U.S. Environmental Protection Agency

Los Angeles Area Lakes TMDLs
March 2012

Appendices

Appendix A. Methodology for Nutrient TMDL Development

(This page left intentionally blank.)

A.1 Introduction

USEPA Region IX is establishing TMDLs for impairments in nine lakes in the Los Angeles Region (Figure A-1). USEPA was assisted in this effort by the Los Angeles Water Quality Control Board (Regional Board). Impairments of these waterbodies include low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, algae, pH, mercury, lead, copper, chlordane, DDT, dieldrin, PCBs, and trash.

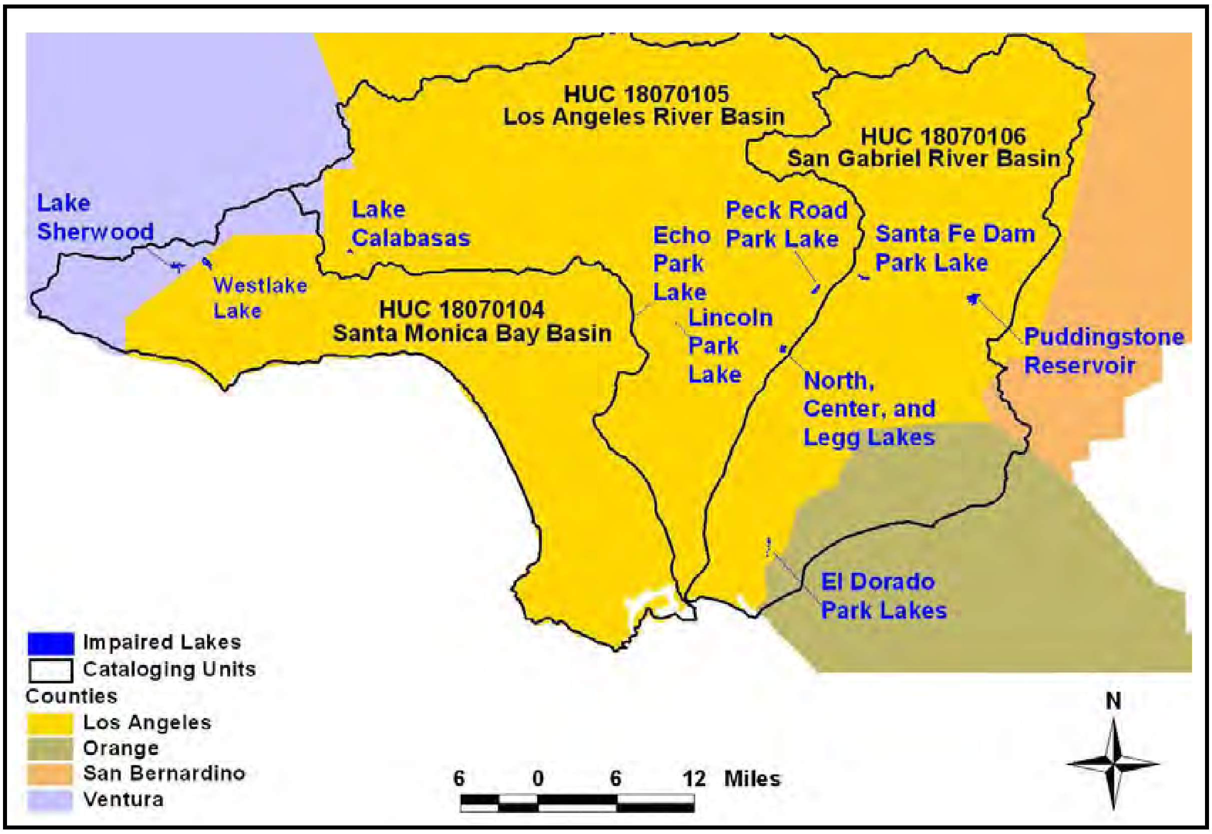


Figure A-1. Location of Impaired Lakes

Eight of these waterbodies have impairments that may be due to elevated nutrient levels: Peck Road Park Lake, Echo Park Lake, Lincoln Park Lake, Lake Calabasas, the El Dorado Park lakes, Legg Lake, Puddingstone Reservoir, and Santa Fe Dam Park Lake. These impairments include algae, ammonia, eutrophication, low dissolved oxygen/organic enrichment, odor, and pH. A steady-state lake response model has been set up for each impaired lake to determine whether or not eutrophication is the primary cause of these impairments. This appendix discusses the problems associated with eutrophication, sources of nutrient loading, and the approach used for determining loading capacities for nitrogen and phosphorus based on observed and simulated levels of chlorophyll *a*.

(This page left intentionally blank.)

A.2 Conceptual Model: Nutrients, Algae, and Eutrophication

Excessive algal growth in the urban lakes of the Los Angeles region has resulted in several waterbodies not supporting their designated beneficial uses associated with aquatic life and recreation (LARWQCB, 1996). Unaesthetic amounts of algal biomass can directly impair swimming and wading recreational uses. Algal growth in some instances has produced algal mats in the lakes (UC Riverside, 1994). Excess growth of algae can also result in loss of invertebrate taxa through habitat alteration (Biggs, 2000). In addition, ammonia, a nitrogen compound, has been measured at concentrations exceeding objectives designed to protect aquatic life (LARWQCB, 1996).

Rates of algal growth depend on the availability of nutrients, light, and other factors. Stimulation of excess algal growth by nutrient loading is referred to as eutrophication. There are many biological responses to nutrients (nitrogen and phosphorus) in lakes. The biologically available nutrients and light will stimulate phytoplankton and or macrophyte growth. As these plants grow, they provide food and habitat for other organisms such as zooplankton and fish. When the aquatic plants die, they will release nutrients (ammonia and phosphorus) back into the water through decomposition. The decomposition of plant material consumes oxygen from the water column; in addition the recycled nutrients are available to stimulate additional plant growth. Physical properties such as light, temperature, residence time, and wind mixing also play integral roles throughout the pathways described.

These typical biological processes can become over-stimulated by the addition of excess nutrients to a waterbody and create a situation in which water quality becomes degraded and beneficial uses are impaired. The following flow chart (Figure A-2) outlines the responses within a lake to excessive nutrient loading and how the beneficial uses will be impacted.

Excessive nutrient loading, from either external or internal processes, can cause excessive phytoplankton and macrophyte growth. The resulting plant biomass may cause increased turbidity, altered planktonic food chains, unaesthetic conditions, reduced dissolved oxygen concentrations, and increased nutrient recycling (Figure A-2). These changes can lead to a cascade of biological responses culminating in impaired beneficial uses.

Typically, excessive plant growth can quickly lead to an altered planktonic community; in many cases the dominant phytoplankton species may become blue-green algae (cyanophytes) and algal blooms may occur, especially in the summer months. These blooms cause fluctuations in dissolved oxygen concentration and pH that can negatively affect aquatic life in the waterbody. Senescence and decay of the biomass present in algal blooms may also cause problems with scum and odors that affect recreational uses of the affected waterbody. Likewise, macrophyte growth may increase and become expansive throughout the lake (Figure A-2). Particularly in shallow lakes, the combination of available nutrients and greater light intensity throughout the water column provides the light that is needed for rapid plant growth. In addition, light can penetrate to the bottom of shallow lakes, promoting macrophyte growth. In comparison, in deep lakes a greater portion of the water column is not able to support photosynthesis as a majority of the water column is below the light penetration depth. Thus, the impacts of nutrient loading and the biological response of planktonic algae and macrophytes are often very apparent in shallow lakes.

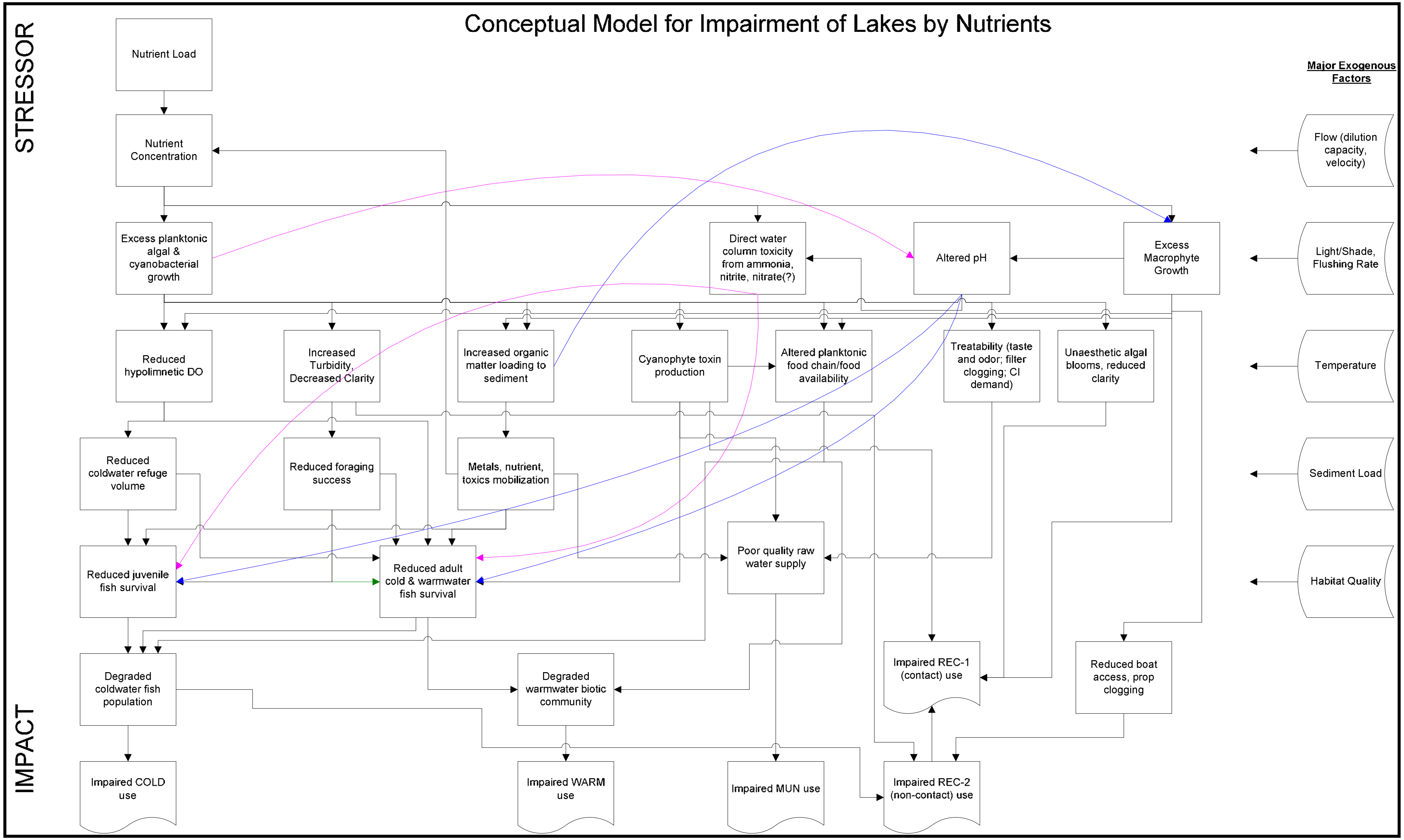


Figure A-2. Conceptual Model for Lakes

As noted above, eutrophication can also lead to increased daytime pH in lakes due to rapid uptake of carbon dioxide by photosynthesizing algae. The elevated pH creates a harmful environment for organisms and can increase the concentration of un-ionized ammonia, potentially leading to direct toxicity to fish and other organisms. Dense algal populations also cause diurnal swings in dissolved oxygen concentrations, as oxygen is released during daytime photosynthesis and consumed during nighttime respiration. Decomposition of algal biomass can consume oxygen and dramatically reduce the oxygen levels found in the lake. Low dissolved oxygen levels can become very stressful for fish and other organisms and may in fact lead to fish kills (Figure A-2). Moreover, as the plant material is decomposed, the nutrients are released and will recycle through the system. Shallow lakes tend to have increased biological productivity because it is likely that the photosynthetic zone and decomposition zone of the water column overlap, creating the situation where as materials are decomposed and the nutrients released, they are also immediately available for photosynthesis and plant growth continuing to drive ongoing impairments.

Control of the deleterious effects of eutrophication in lakes typically requires reduction in nutrient loads. Both external and internal (recycled) nutrient loads may need to be addressed.

(This page left intentionally blank.)

A.3 Source Assessment

Sources of nutrient loading to a lake may include both point and nonpoint sources. For the purposes of allocating loads among nutrient sources, federal regulations distinguish between allocations for point sources regulated under NPDES permits (for which wasteload allocations are established) and nonpoint sources that are not regulated through NPDES permits (for which load allocations are established) (see 40 CFR 130.2). This section describes how the loading from point and nonpoint sources were estimated.

A.3.1 POINT SOURCES

Point sources are discharges that occur at a defined point, or points, such as a pipe or storm drain outlet. Most point sources are regulated through the NPDES permitting process.

A.3.1.1 MS4 Permittees

In 1990 USEPA developed rules establishing Phase 1 of the NPDES stormwater program, designed to prevent pollutants from being washed by stormwater runoff into the Municipal Separate Storm Sewer Systems (MS4), or from being directly discharged into the MS4 and then discharged into local waterbodies. Phase I of the program required operators of medium and large MS4s (those generally serving populations of 100,000 or more) to implement a stormwater management program as a means to control polluted discharges. Phase II of the program extends the requirements to operators of small MS4 systems, which must reduce pollutants in stormwater to the maximum extent practicable (MEP) to protect water quality.

Nitrogen and phosphorus loads from urban stormwater runoff are estimated from event mean concentration (EMC) data and flows predicted from calibrated watershed models (Appendix D, Wet Weather Loading). Two flow-calibrated LSPC models were previously developed for the San Gabriel and Los Angeles river basins (Tetra Tech, 2004; Tetra Tech, 2005). To estimate runoff volumes, average monthly areal flow rates have been extracted for each land use and applied to the land use composition that drains to an MS4 for each lake. The county of Los Angeles and the Southern California Coastal Water Research Project (SCCWRP) have been collecting pollutant concentration data for storm events in the county of Los Angeles for representative land use classes. These concentrations can be applied to the flow volumes predicted by the LSPC models for each land use to estimate average wet weather nutrient loading to each lake. Appendix D (Wet Weather Loading) describes the datasets, assumptions, and loading results for this analysis.

These systems may also discharge during dry weather as a result of irrigation, car washing, etc. Estimation of nutrient loading from MS4 systems in dry weather is based on SCCWRP regional studies and is described in Appendix F (Dry Weather Loading).

A.3.1.2 Non-MS4 NPDES Discharges

In addition to MS4 stormwater dischargers, the NPDES program regulates stormwater discharges associated with industrial and construction activities and non-stormwater discharges (individual and general permits). To quantify nutrient loading from non-MS4 NPDES discharges, the permit databases maintained by the Los Angeles Regional Board were downloaded for the Los Angeles River, San Gabriel River, and Santa Monica Bay Basins. Geographic information listed for each permit was used to determine which facilities are located in the watersheds of the eight nutrient-impaired lakes. Nutrient loading from each facility was estimated based on the reported disturbed area. The facilities and estimated loads are described in more detail in the lake specific sections of this report.

A.3.1.3 Additional Inputs

Several of the lakes addressed by this TMDL have additional point source inputs that do not currently have NPDES permits. Most are supplemental flows from groundwater wells or potable water that maintain lake levels. Information pertaining to flow volumes from these sources was provided by park staff at each lake (generally based on water usage information from the water suppliers). Where accessible, the Regional Board and USEPA sampled water quality from these inputs during the 2009 sampling events. In some cases, the suppliers were able to provide nutrient concentrations. Nutrient loading was calculated from average nutrient species concentration data and an estimate of annual flow volumes to each lake.

A.3.2 NONPOINT SOURCES

Nutrient loading from nonpoint sources originates from sources that do not discharge at a defined point. This section describes the methods used to estimate loading from nonpoint sources.

A.3.2.1 Internal Loading from Lake Sediments

Lake sediments typically store phosphorus that has sorbed to soil particles or settled to the bottom of the lake following the decomposition of organic matter. When these sediments become hypoxic (i.e., when dissolved oxygen concentrations become low) they may release stored phosphorus into the water column which then becomes available for uptake by plants and algae. In some lakes, internal phosphorus loading may comprise a significant portion of the total load.

Hypoxic conditions also promote release of dissolved ammonia from the sediments. Lake sediments do not typically store and release significant quantities of nitrogen relative to other lake inputs. However, the net nitrogen sedimentation rate calibrated for each lake accounts for internal loading of nitrogen as well.

Intensive monitoring studies are typically required to accurately quantify internal nutrient loading. This level of information was not available for the lakes addressed by this TMDL. Though the internal load may not be quantified for these lakes, it is reflected in the net (settling minus resuspension) nutrient sedimentation rates calibrated for each lake (Section A.4.2).

Internal loading is discussed in more detail in Appendix B (Internal Loading).

A.3.2.2 Wind Resuspension

As wind moves across a lake surface, the resulting wave action may disturb lake sediments in shallow areas and release additional stored phosphorus. Appendix B (Internal Loading) describes the impacts of wind resuspension and defines the critical lake levels where additional internal loading may occur. As wind resuspension impacts internal loading rates, the effects were accounted for in the net sedimentation rates for phosphorus and nitrogen.

A.3.2.3 Bioturbation

Bottom feeding fish and benthic macroinvertebrates can also disturb lake sediments and promote release of stored nutrients. As bioturbation further impacts internal loading rates, the effects were accounted for in the net sedimentation rates for phosphorus and nitrogen.

A.3.2.4 Atmospheric Deposition

The National Atmospheric Deposition Program (NADP) monitors wet nitrogen deposition (as nitrate) at two active and two inactive stations in southern California. Isopleth maps were downloaded from the NADP website and brought into a GIS environment to extract site specific precipitation-weighted annual average nitrate concentrations for grid cells overlaying each lake. NADP has produced these isopleth maps for years 1994 through 2006. The time series was extended to previous years by developing a regression equation for each location based on year and cumulative precipitation (Appendix E, Atmospheric Deposition).

The precipitation-weighted annual average nitrate concentrations were then multiplied by the annual rainfall observed at the nearest weather station (Appendix D, Wet Weather Loading)) and the lake surface area to estimate nitrogen loading from atmospheric deposition to each lake surface. Deposition to land surfaces is accounted for in the loading estimates from the watersheds (Appendices D and F; Wet and Dry Weather Loading, respectively).

Unlike nitrogen, phosphorus does not have a significant gaseous phase, and atmospheric deposition is primarily due to fugitive dust. Phosphorus deposition rates are typically much lower than nitrogen deposition rates and are not included in the NADP monitoring program. At this time, measurements of phosphorus deposition rates are not available for this area. SCCWRP has recently begun a deposition monitoring study that will measure phosphorus, but the results are not expected to be published until 2011.

The datasets, assumptions, and resulting loading from atmospheric deposition are described in detail in Appendix E (Atmospheric Deposition).

A.3.2.5 Wet Weather Loading

Nitrogen and phosphorus loads from areas that do not drain to an MS4 system are estimated from event mean concentration (EMC) data and flows predicted from calibrated watershed models (Appendix D, Wet Weather Loading). Two flow-calibrated LSPC models were previously developed for the San Gabriel and Los Angeles river basins (Tetra Tech, 2004; Tetra Tech, 2005). To estimate nonpoint source runoff volumes, average monthly areal flow rates have been extracted for each land use and applied to the land use composition that does not drain to an MS4. The county of Los Angeles and SCCWRP have been collecting pollutant concentration data for storm events in the county of Los Angeles for representative land use classes. These concentrations can be applied to the flow volumes predicted by the LSPC models for each land use to estimate average nutrient loading to each lake. Appendix D (Wet Weather Loading) describes the datasets, assumptions, and loading results for this analysis.

A.3.2.6 Dry Weather Loading

In addition to pollutant loads delivered during storm events (discussed in Appendix D, Wet Weather Loading), it is important to account for loads that are delivered to a waterbody during dry weather. Nonpoint sources during dry weather include irrigation, fertilization of adjacent parkland, and other miscellaneous urban sources. Estimation of dry weather pollutant loading is discussed in Appendix F (Dry Weather Loading).

(This page left intentionally blank.)

A.4 Linkage Analysis

To simulate the impacts of nutrient loading on each impaired lake, the nutrient numeric endpoints (NNE) BATHTUB Tool was set up and calibrated to lake-specific conditions. The NNE BATHTUB Tool is a version of the US Army Corps of Engineers (USACE) BATHTUB model and was developed to support risk-based nutrient numeric endpoints in California (Tetra Tech, 2006). For these TMDLs, target nutrient loads and resulting allocations were determined specifically for each lake based on the secondary target – summer season mean chlorophyll *a* concentration.

Other parameters may be chosen as secondary targets for determining nutrient allocations. Chlorophyll *a*, however, is the best choice for assessing nutrient impacts alone. For example, choosing dissolved oxygen as a secondary target will not only account for fluctuations in concentration caused by algal photosynthesis and respiration but will also include response to loading of organic matter not associated with algal decay (e.g., loading from wastewater treatment plants, organic fertilizers, etc.). The existing dissolved oxygen criteria serves as an additional target in these TMDLs. Light penetration, often measured as Secchi depth, is another indicator of nutrient impairment as greater densities of algae block sunlight penetration and reduce Secchi depth. Light penetration is also impacted by suspended sediment concentrations and low Secchi depth does not always correlate with excessive nutrient loading. This is often the case for waterbodies located in watersheds comprised of silt and clay soils or in areas undergoing land clearing and construction. Thus chlorophyll *a* is the most appropriate parameter for assessing the direct impacts of eutrophication on a waterbody. This section describes how the NNE Tool simulates chlorophyll *a* and its use in developing the nitrogen and phosphorus TMDLs.

A.4.1 MODEL DESCRIPTION

The USACE developed the BATHTUB model (Walker, 1987) to predict eutrophication in reservoirs across the country. BATHTUB is a steady-state model that calculates nutrient concentrations, chlorophyll *a* concentration (or algal density), turbidity, and hypolimnetic oxygen depletion based on nutrient loadings, hydrology, lake morphometry, and internal nutrient cycling processes. BATHTUB uses a typical mass balance modeling approach that tracks the fate of external and internal nutrient loads between the water column, outflows, and sediments. External loads can be specified from various sources including stream inflows, nonpoint source runoff, atmospheric deposition, groundwater inflows, and point sources. Internal nutrient loads from cycling processes may include sediment release and macrophyte decomposition. These processes are accounted for implicitly in the model through the calibration of the net sedimentation rates. If an estimate of internal loading of phosphorus is required, the following methodology described by Nürnberg (1984) provides a reasonable estimate:

TPinlake = TPinflow * (1-Rpred) + Lint / Qs , where

Rpred = 15 / (18 + Qs)

TPinlake = mean summer in-lake phosphorus concentration

TPinflow = mean summer tributary phosphorus concentration

Qs = mean depth over hydraulic residence time

Rpred = annual retention due to sedimentation

Lint = internal phosphorus load (mg/m^2/yr)

Since BATHTUB is a steady-state model, it focuses on long-term average conditions rather than day-to-day variations in water quality. Algal concentrations are predicted for the summer season when water quality problems are most severe. Annual differences in water quality, or differences resulting from different loading or hydrologic conditions (e.g., wet vs. dry years), can be evaluated by running the model separately for each scenario.

BATHTUB first calculates steady-state phosphorus and nitrogen balances based on nutrient loads, nutrient sedimentation, and transport processes (lake flushing, transport between segments, etc.). Several options are provided to allow first-order, second-order, and other loss rate formulations for nutrient sedimentation that have been proposed from various nutrient loading models in the literature. The resulting nutrient levels are then used in a series of empirical relationships to calculate chlorophyll *a*, oxygen depletion, and turbidity. Phytoplankton concentrations are estimated from mechanistically-based steady-state relationships that include processes such as photosynthesis, settling, respiration, grazing mortality, and flushing. Both nitrogen and phosphorus can be considered as limiting nutrients, at the option of the user. Several options are also provided to account for variations in nutrient availability for phytoplankton growth based on the nutrient speciation in the inflows. The empirical relationships used in BATHTUB were derived from field data from many different lakes, including those in USEPA's National Eutrophication Survey and lakes operated by the Army Corps of Engineers. Default values are provided for most of the model parameters based on extensive statistical analyses of these data.

In 2006, Tetra Tech developed the NNE BATHTUB Tool as a simplified method for predicting summer season chlorophyll *a* lake response to a number of inputs. The NNE BATHTUB Tool is a risk-based approach for estimating site-specific nutrient numeric endpoints (NNE) for California waters (Tetra Tech, 2006). The Tool has been tested for several waterbodies in California as a series of case studies (e.g., Tetra Tech, 2007).

The NNE spreadsheet tool allows the user to specify a chlorophyll *a* target and predicts the probability that current conditions will exceed the target, as well as showing a matrix of allowable nitrogen and phosphorus loading combinations necessary to meet the target. The user-defined chlorophyll *a* target can be input directly by the user, or can be calculated based on an allowable change in water transparency measured as Secchi depth.

For both the nitrogen and phosphorus simulations, the NNE BATHTUB Tool has been set up to incorporate the USACE BATHTUB Model default equations for simulating nutrient sedimentation rates. In accordance with the USACE BATHTUB Model Users Manual (Walker, 1987), the NNE Tool incorporates a calibration factor on each sedimentation rate to improve model fit to observed data.

The NNE BATHTUB Tool simulates phosphorus (P) using the 2nd-order P-sedimentation model (presented as P Model 2 in Walker, 1987):

$$\text{P Sedimentation Rate (mg/m}^3\text{-yr)} = K_P \cdot A1 \cdot P^2,$$

where P is the total phosphorus concentration in µg/L.

This yields a solution for P:

$$P = \frac{\sqrt{1 + 4K_P\, A1\, Pi\, T} - 1}{2K_P\, A1 T}, \text{ where}$$

A1= 0.056 Qs/[Fot · (Qs + 13.3)]

Pi = inflow total P concentration (µg/L)

Qs = overflow rate (m/yr), with a minimum of 4

Fot = ratio of inflow ortho P to inflow total P

K_P = P calibration factor, typically ranging from 0.5 to 2.0

T = hydraulic residence time (yr) = Volume/Inflow-per-yr

The nitrogen (N) simulation is implemented using the 2nd order N-sedimentation (presented as N Model 2 in Walker, 1987):

N Sedimentation Rate (mg/m^3-yr) = $K_N \cdot B1 \cdot N^2$,

where N is the total nitrogen concentration in µg/L. This yields a solution for N:

$$N = \frac{\sqrt{1 + 4 K_N \, B1 \, Ni \, T} - 1}{2 K_N \, B1 T}, \text{ where}$$

$B1 = 0.0035\,Qs/[Fin^{0.59} \cdot (Qs + 17.3)]$

K_N = N calibration factor, typically ranging from 0.3 to 3.0

Ni = inflow total N concentration

Fin = ratio of inflow inorganic N to inflow total N

The USACE BATHTUB Model allows the user to choose from five empirical equations for chlorophyll *a* simulation. The NNE Tool incorporates the equation that considers light, flushing rate, and nutrient concentrations to account for the co-risk factors whose cumulative effect determines algal density (presented as Chl Model 1 in Walker, 1987). A calibration factor on simulated chlorophyll *a* concentration allows the user to improve the model fit based on observed data:

$$Chl - a = \frac{K_C \, Bx}{(1 + 0.025 \, Bx \, G)(1 + G \, a)}, \text{ where}$$

$Bx = Xpn^{1.33}/4.31$

$Xpn = [P^{-2} + ((N-150)/12)^{-2}]^{-0.5}$

K_C = Chl-a calibration factor

G = Zmix · (0.14 + 0.0039 Fs)

Zmix = mixed depth (m)

Fs = (summer) flushing rate = (inflow – evap)/vol

A = non-algal turbidity (m^{-1}).

The NNE BATHTUB Tool uses Visual Basic's GoalSeek function to find combinations of N and P loading that result in predicted chlorophyll *a* being equal to the selected target. Because algal growth can be limited by either N or P there is not a unique solution, and the Tool output supplies the user with a curve representing the loading combinations that will result in attainment of the selected chlorophyll *a* target.

Spatial variability in water quality can be simulated with BATHTUB by dividing a lake horizontally into segments, and calculating transport processes such as advection and dispersion between the segments. This is appropriate for large lakes, particularly lakes with multiple sidearms and tributary inflows, that have substantially different water quality in different portions of the lake. However, this was not necessary for the lakes addressed in this TMDL report due to their generally small to moderate sizes, and the lack of detailed data demonstrating significant spatial variations in lake characteristics and water quality. Therefore, the NNE BATHTUB Tool was applied as a whole lake model to each waterbody. In some cases, a chain of multiple lakes was combined into a single lake system for modeling because the multiple lakes had similar characteristics or they functioned essentially as a single lake. The lake-specific chapters describe details associated with each lake model.

A.4.2 MODEL SETUP AND CALIBRATION TO EXISTING CONDITIONS

The NNE BATHTUB Tool was set up individually for each impaired lake or lake system. Bathymetry data for each lake were acquired from various sources to represent the general characteristics of the waterbody, such as surface area, volume, and average depth. The lake specific bathymetry data are discussed in each lake chapter of the TMDL report.

Cumulative nitrogen and phosphorus loads were calculated as a sum of all known, quantifiable sources. Sources of loading resulting from wet weather are discussed in Appendix D; Appendix F summarizes the loading originating during dry weather conditions. Atmospheric deposition to each lake surface is quantified in Appendix E. Internal nutrient loading is discussed in Appendix B, but is not quantified directly due to lack of data (the BATHTUB model accounts for internal loading indirectly by using a net sedimentation rate [sedimentation minus resuspension]). Prior to calibration of the BATHTUB model, the user must determine the appropriate averaging period by calculating the nutrient turnover ratio (Walker, 1987). Average external loading rates are calculated for the summer season (May through September) and for the year. These loads are compared to the mass of nutrients stored in the waterbody (average nutrient concentration times volume) to calculate the mass residence time. Dividing the length of the averaging period (1.0 yr for the annual averaging period or 0.42 yr for the summer season period) by the mass residence time yields the nutrient turnover ratio. The averaging period for the model should be selected such that the nutrient turnover ratio for the limiting nutrient is greater than or equal to 2. The following equations apply:

$$\text{Mass Residence Time (yr)} = \text{Nutrient mass in waterbody (lb)} / \text{External nutrient loading (lb/yr)}$$

$$\text{Nutrient Turnover Ratio} = \text{Length of the averaging period (yr)} / \text{Mass Residence Time (yr)}$$

Once the bathymetry and loading inputs corresponding to the correct averaging period were input, each model was calibrated to observed conditions. Simulated phosphorus concentrations were compared to the average summer season concentrations based on data collected since the early 1990s (Appendix G, Monitoring Data). The calibration factor, K_P, was adjusted until the simulated concentration approximated those observed. The calibration process was repeated using K_N for nitrogen and K_C for chlorophyll *a*.

For some of the lakes, there are other sources of loading associated with the parkland area for which loading estimates were not available (Appendix F, Dry Weather Loading). Examples include inputs from excessive fertilization relative to product recommendations and runoff of nearby residential areas (through the storm drain system or nonpoint source) where fertilizer application rates were unknown, leaking wastewater infrastructure serving visitors at adjacent parks, natural wildlife populations, and abnormally high wildlife populations caused by feeding and inappropriate trash disposal along the shorelines of park lakes. Loads in this additional parkland loading category were quantified using the NNE BATHTUB model by increasing the inputs until simulated concentrations of total phosphorus and total nitrogen matched those observed. The chlorophyll *a* concentrations were then calibrated using K_C.

A.5 TMDL Development

The TMDL is defined by the loading capacity. A waterbody's loading capacity represents the maximum amount of pollutant loading that can be assimilated without violating water quality standards (40 CFR 130.2(f)). For nutrients, this is the maximum amount of nitrogen and phosphorus loading consistent with meeting the numeric target of 20 µg/L of chlorophyll *a* as an average summer concentration in each impaired lake. Selection of the chlorophyll *a* target is discussed in Section 2.2.3.

A.5.1 LOADING CAPACITY AND ALLOCATIONS

The NNE BATHTUB Tool outputs a matrix of nitrogen and phosphorus loads consistent with achieving the chlorophyll *a* target. For lakes where the calibrated chlorophyll *a* concentration is less than the target, it was assumed that the loading capacity is not exceeded under existing conditions and no reductions in nitrogen or phosphorus are required. For those lakes where the chlorophyll *a* concentration is greater than the target and loading reductions are required, the loading combination that is predicted to result in an in-lake ratio of total nitrogen concentration to total phosphorus concentration close to 10 was selected. This ratio was chosen to match that typically observed in natural systems and to balance biomass growth and prevent limitation by one nutrient (Thomann and Mueller, 1987). A ratio of 10 typically limits the growth nuisance species, such as cyanobacteria (blue green algae) (Welch and Jacoby, 2004).

The loading capacity for each nutrient is expressed as pounds per year (lb/yr). The values are further broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margin of Safety (MOS) using the general TMDL equation:

$$TMDL = Loading\ Capacity = \sum WLAs + LAs + MOS$$

Existing loads, loading capacity, WLAs, LAs, and MOS are presented for each individual waterbody or lake system in the respective lake chapters of this TMDL report. As previously mentioned, in-lake concentrations of nitrogen and phosphorus have been determined based on simulation of allowable loads with the NNE BATHTUB model and using a ratio close to 10. These in-lake concentrations are calculated from a complex set of equations that consider internal cycling processes and, therefore, differ from concentrations associated with various inflows. Each lake chapter also presents nutrient concentrations associated with the WLA and LA inputs. These values are provided as examples as they are calculated based on existing flow volumes (and will need to be recalculated if flow volumes change). Because the input concentrations do not consider internal cycling processes and are based on existing flow volumes, they do not match the allowable in-lake nitrogen and phosphorous concentrations.

A.5.2 MARGIN OF SAFETY

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. The nutrient TMDLs for these lakes are based on simulated nitrogen and phosphorous concentrations and include a 10 percent explicit margin of safety when reductions are required. For lakes not currently exceeding the numeric targets, the loading capacity has been set to existing conditions as an antidegradation measure; hence, the MOS is implicitly applied in the TMDL development.

A.5.3 DAILY LOAD EXPRESSION

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River TMDL. The TMDLs developed here each include a daily maximum load estimate consistent with the guidelines provided by USEPA (2007). Because the majority of loads occur during wet weather events, the maximum allowable daily load is calculated from the 99th percentile flow multiplied by the average allowable concentration consistent with achieving the long-term loading targets. In lakes where the majority of loads are associated with supplemental water additions, appropriate flow rates are determined and multiplied by the average allowable concentration to determine the maximum allowable daily load.

A.6 References

Biggs, B.J.F. 2000. Eutrophication of streams and rivers: dissolved nutrient-chlorophyll relationships for benthic algae. J. N. Am. Benthol. Soc. 19(1):17-31.

LARWQCB. 1996. LA Regional Water Quality Control Board 1996 Water Quality Assessment & Documentation – 305(b) Report Supporting Documentation for Los Angeles Region. Developed by the Los Angeles Regional Water Quality Control Board.

Nürnberg, G.K. 1984. The prediction of internal phosphorus load in lakes with anoxic hypolimnia. Limno. Oceanogr. 29, 111-24.

Tetra Tech. 2004. Model Development for Simulation of Wet-Weather Metals Loading from the Los Angeles River Watershed. Prepared for USEPA Region 9 and the Los Angeles Regional Water Quality Control Board, May 2004.

Tetra Tech. 2005. Model Development for Simulation of Wet-Weather Metals Loading from the San Gabriel River Watershed. Prepared for USEPA Region 9 and the Los Angeles Regional Water Quality Control Board, October 2005.

Tetra Tech. 2006. Technical Approach to Develop Nutrient Numeric Endpoints for California. Prepared for USEPA Region 9 and the California State Water Research Control Board, Planning and Standards Implementation Unit.

Tetra Tech. 2007. Nutrient Numeric Endpoints for TMDL Development: Malibu Creek Case Study. Prepared for USEPA Region IX.

Thomann, R.V., and J.A. Mueller, 1987. Principles of Surface Water Quality Modeling and Control, Harper and Row, New York, 1987.

UC Riverside. 1994. Evaluation of water quality for selected lakes in the Los Angeles hydrologic basin. Submitted to LARWQCB, December 1994.

Walker, W.W. 1987. Empirical Methods for Predicting Eutrophication in Impoundments. Report 4–Phase III: Applications Manual. Technical Report E-81-9. U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, MS.

Welch, E.B. and J.M. Jacoby. 2004. Pollutant Effects in Freshwater Applied Limnology, Third Edition. Spon Press, London.

(This page left intentionally blank.)

Appendix B. Internal Loading from Lake Sediments

(This page left intentionally blank.)

B.1 Introduction

USEPA Region IX is establishing TMDLs for impairments in nine lakes in the Los Angeles Region (Figure B-1). USEPA was assisted in this effort by the Los Angeles Water Quality Control Board (Regional Board). The waterbodies are impaired by combinations of low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, algae, pH, mercury, lead, copper, chlordane, DDT, dieldrin, PCBs, and trash.

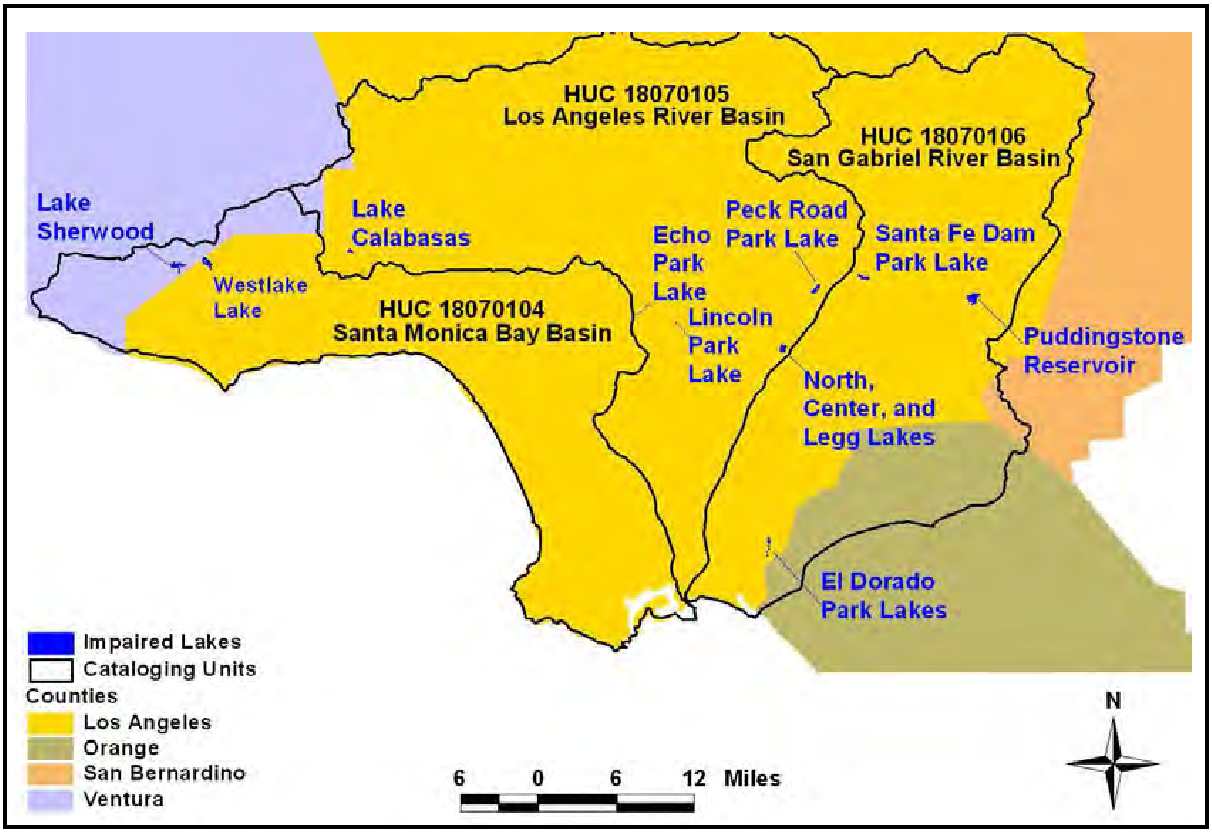


Figure B-1. Location of Impaired Lakes

Internal loading from the lake sediments of impaired waterbodies can be a significant source of pollutant loading, particularly for phosphorus, mercury, and Organochlorine (OC) Pesticides and PCBs. This appendix provides a general overview of the mechanisms that affect rates of internal loading. Although processes affecting internal loads of all pollutants are discussed, internal loads of phosphorus and mercury will not be quantified in the TMDLs because the linkage analyses implicitly account for these mechanisms. For phosphorus, the NNE BATHTUB Tool (Appendix A, Nutrient TMDL Development) accounts for resuspension from internal sediments by applying a net sedimentation rate for phosphorus. For mercury, fish tissue bioaccumulation data reflect both the external and internal loading of methylmercury to the waterbody (Appendix C, Mercury TMDL Development). In addition, loads of phosphorus and mercury continue to enter the impaired waterbodies, although mercury is likely delivered at lower levels than seen previously.

For OC Pesticides and PCBs, the fish tissue bioaccumulation data reflect all sources of loading; however, historic accumulation and internal releases are likely the predominant source of loading to Puddingstone Reservoir, Peck Road Park Lake, and Echo Park Lake as the use of chlordane, DDTs, dieldrin, and PCBs is no longer allowed in the U.S. Thus quantifying internal loading of these pollutants was an important component of TMDL development. Estimation of internal recycling rates of OC Pesticides and PCBs is discussed in Appendix H (Organochlorine Compounds TMDL Development).

This appendix discusses the general process of internal loading from lake sediments and the conditions that tend to increase rates of release of the contaminants addressed by this TMDL report.

B.2 Historic Sediment Stores

External loads of pollutants can enter lakes in surface flow, in groundwater discharge, and by direct atmospheric deposition. Once entering the lake, pollutants may be discharged downstream, degrade, volatilize back to the atmosphere, or settle to the sediment.

Over time nutrients, metals, and OC Pesticides and PCBs that are particle reactive tend to settle and accumulate in sediment on a lake's bottom. The net rate of settling is dependent upon the particular lake's dynamics as well as the compound's chemical characteristics. Sedimentation can also create a concentration gradient in the water column, where water near the water-sediment interface tends to harbor higher concentrations of nutrients, metals, or OC Pesticides and PCBs than the shallower lake levels.

Once material is translocated to the sediment several conceptual pathways may be followed:

- The material may remain in the shallow sediment layers with the potential for continued exchange with the water column and biota.
- The material may degrade or be sequestered in permanently insoluble forms within the sediment, resulting in a net loss of active pollutant mass.
- The material may be buried with clean sediment (either from upland erosion processes or a capping project), sequestering at a depth that minimizes interaction with the water column.
- The material may be released back to the water column.

Release processes such as diffusive exchange, bioturbation, and sediment disturbance by wind mixing or dredging activity can release historical sediment stores, returning pollutants to the water column. For these reasons, sedimentation can act as both a sink and a source of these contaminants in lakes. Refer to Section 2 for a more detailed discussion on the determination of sediment targets.

Most OC Pesticides and PCBs are generally banned from use and no longer manufactured in the US. Despite these efforts, historical loading and sedimentation has often caused a situation in which elevated concentrations continue to be found in lake sediment stores. External loading rates of phosphorus and metals have also often declined over time with better management practices, but historical elevated loading may result in significant stores present in lake sediments. Releases of sediment stores of these compounds may comprise a significant portion of the total load to a lake's water column. For example, internal loading can account for a substantial amount of the total phosphorus within the water column (Moore et al., 1998), creating a situation in which, despite reduction in external loading, phosphorus concentrations in lake water remain high and cause continued impairment (Bachmann, 2005). Authors such as Brumbaugh et al. (2001) have shown a log-log linear relationship between methylmercury in the water and fish tissue, when normalized to fish length. Further, elevated concentrations of OC Pesticides and PCBs in fish tissue can occur as a direct result of food chain pathways that lead back to worms and other invertebrates that feed in contaminated sediments, even when water column concentrations meet all applicable criteria (Thomann et al., 1992).

Estimation of the total mass of pollutants stored in sediment is difficult. Concentrations in sediment often vary by orders of magnitude over short distances in both the lateral and vertical dimension, so large amounts of samples are often needed to obtain an accurate characterization of the sediment storage. Historical bathymetric data can assist in determining the net rates of sediment accumulation. This could be used to obtain rough estimates of sediment storage if combined with assumptions about the changes in concentrations on influent sediment over time.

In theory, removal of contaminated sediment could reduce the amount of accumulated pollutants available for exchange into the water column and biota. Unfortunately, the removal process may disturb and release the metals, nutrients, or OC Pesticides and PCBs, returning these constituents to the water column, and thereby increasing the bioavailability of the compounds. Additionally, removal of the top layers of sediment may uncover more contaminated layers deposited in past decades when the use and management of the pollutants was less adequately controlled in the US.

As an alternative to dredging removal, highly contaminated sediments are sometimes sequestered with engineered caps to prevent releases to the water column. Both approaches are very costly, and are thus most often used at highly contaminated Superfund sites. Less expensive techniques attempt to reduce rates of release relative to the processes discussed in the following sections. For example, oxygenating the bottom water can minimize releases that are facilitated by anoxia, while manipulation of lake levels can sometimes reduce resuspension due to wind mixing. For some pollutants, chemical treatments can be useful. For instance, alum is often used to reduce phosphorus recycling in lakes by converting phosphorus to insoluble precipitates.

B.3 Thermal Stratification and Wind Mixing

Lakes located in the Los Angeles region are exposed to extreme heat during the summer months. Cycles of warming and cooling due to seasonal variations impact the release of suspended sediments and associated constituents into the water column. Several of the lakes addressed by this TMDL report are also relatively shallow and are subject to wind mixing which may disturb lake sediments and associated pollutants and further impair water quality.

Thermal stratification refers to the process in which a warm layer of water develops in the epilimnion (the upper level of a stratified lake) due to the transfer of solar energy, while deeper waters remain cooler and, sometimes, anoxic, particularly in the summer (see Section B.4 for more details on anoxic conditions). The difference in temperature causes a density gradient and increased resistance to mixing between the upper and lower lake depths (cooler water being more dense), which limits the exchange of water and compounds between the layers and typically results in epilimnotic concentrations of sediment-associated pollutants being less than those found in deeper waters. The greater the temperature differential, the more resistant the water column is to vertical mixing. Stratified conditions remain until the thermal density gradient disappears due to cooling of the surface water or wind energy is able to overcome the remaining density gradient, allowing the water to mix; this process is referred to as lake turnover. As deep waters rise to the surface, they may transport significant amounts of sediment-associated pollutants (e.g., metals, nutrients, and OC Pesticides and PCBs) that were released during periods of stratification into surface waters where they may exacerbate algal growth or contaminate fish tissue.

Wind mixing also has the potential to increase resuspension of bed sediments and associated pollutants in shallow waters. The wind-mixed depth, referred to as the “critical depth,” is directly related to the fetch (the distance wind travels across the surface of the lake), the lake depth, and the wind speed. Longer lake fetches tend to allow for a greater critical depth, and lakes unprotected from the wind are more susceptible to increased wind mixing. In most shallow lakes, the critical depth is approximately equal to the average depth of the lake; this allows for areas prone to resuspension. The degree to which wind mixing impacts pollutant resuspension is also related to the lake’s water-level, as there is considerably less potential for sediment resuspension under deep waters; sediments underlying shallow waters have an increased potential for resuspension due to wind action.

The degree to which wind mixing and lake turnover impact water column pollutant concentrations also depends on the physical characteristics of sediment present at the bottom of the lake, the presence or absence of a lake liner, and the presence or absence of benthic algae and macrophytic (rooted plant) communities. Locations with loose organic sediment and sparse plant coverage are more prone to increased rates of resuspension due to wind mixing. Lakes with coarse sediments (sands and gravels), low amounts of settled organic material, or those with artificial liners have less potential for resuspension. Refer to the lake-specific TMDL sections (Sections 4 through 13) for information regarding soil types, lake liners, and bathymetric data.

As described, sediment resuspension has been predicted in studies drawing relationships between resuspension and wind speed, wind direction, fetch and depth to sediment (Carper and Bachmann, 1984). As wind blows over the surface, a deep water wave will be generated when the depth of the water is greater than one half of the wave-length (Wetzel, 2001). The transition of the wave from the deep water to shallow water creates a situation prone to resuspension. The wavelength (L) of a deepwater wave is related to its period (T), in the following relationship, where g is the gravitational constant (Martin and McCutcheon, 1999):

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

The period of a wave can be estimated by using the equation derived by the US Army Coastal Engineering Research Center (Carper and Bachmann, 1984). Where U is the wind speed and F is the fetch:

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

Although the pollutant loads, due to lake turnover and wind mixing, are not explicitly quantified for these TMDLs, they are included inherently in the eutrophication, mercury bioaccumulation, and OC Pesticides and PCBs models developed for each lake (see Appendices A, C, and H, respectively).

B.4 Internal Loading and Anoxic Conditions

The dissolved oxygen concentration at the sediment-water interface plays an important role in the internal loading of various ionic compounds. The condition in which oxygen is fully depleted is called anoxia; partial depletion (below 2 mg/L) is referred to as hypoxia. Deeper lakes will often become thermally stratified in summer months, resulting in anoxic or hypoxic conditions within the lower metalimnion (the middle layer of a stratified lake) and hypolimnion (the bottom layer of a stratified lake). To a certain degree, this is a natural process within deeper lakes; however, it is more common for lakes with small surface areas to become anoxic due to stagnation or limited water exchange. Additionally, the decomposition of the phytoplankton associated with eutrophication requires oxygen, thus decreasing the available dissolved oxygen within the water column, particularly near the sediment-water interface where decaying organic matter tends to settle and accumulate.

The oxidation-reduction (redox) potential of an aquatic system is used to describe the process or degree to which ions are exchanged within a system. Compounds gaining electrons are said to be reduced, while those losing electrons are oxidized. Important biological processes used to create energy (i.e., respiration and photosynthesis) involve the exchange of electrons. The most energetically favorable reaction occurs with the oxidation of organic material (oxic respiration). However, in the absence of oxygen, bacterial processes shift to denitrification, manganese reduction, iron reduction, sulfate reduction, and methanogenesis (releasing compounds such as NH_3/NH_4^+, Mn^{2+}, Fe^{2+}, S^{2-}).

In oxygenated environments, free electrons are readily bound by oxygen and associated compounds are partitioned to sediments. In anoxic environments, particularly at the sediment-water interface or the oxic-anoxic boundary within the water column, electrons and compounds are released into the water column via redox reactions. This release can dramatically increase the concentration of reduced species (NH_3/NH_4^+, Mn^{2+}, Fe^{2+}, S^{2-}) within the waterbody. Artificial aeration of bottom waters impedes these reactions and the release of pollutants.

For example, with limited oxygen, bacterial decomposition of organic material in lake sediments results in the release of inorganic phosphorus into the water column. The iron cycle has a dramatic effect on the rates of recycling of phosphorus: under oxidizing conditions, iron and phosphorus form insoluble ferric hydroxy complexes; under reducing conditions these complexes dissolve, releasing both iron and phosphorus to the water column. In fact, one study found that sediment phosphorus flux was fourfold greater under anoxic conditions (Haggard, 2005) than aerobic. Increased levels of phosphorus resulting from sediment release add to the available nutrient pool and continue the cycle of eutrophication.

Denitrification also occurs under anoxic conditions where nitrates are first reduced to ammonia (NH_4/NH_3) and then to nitrogen gas (N_2). Conversion to ammonia may occur in environments with low oxygen levels; reduction to nitrogen gas requires anoxic conditions.

Under anoxic conditions, sulfates (SO_4^+) are reduced to bisulfide or sulfide (HS^- or S^{2-}). The presence of sulfidic compounds produces a strong sulfur odor, which can lead to an odor impairment in a waterbody. Methylation of mercury is an additional microbial process that occurs under low-oxygen, or reducing conditions. Research shows that sulfur-reducing bacteria may play an important role in the methylation process (Compeau, 1985). The transformation of mercury into methylmercury is of concern as the methylated form, methylmercury, bioaccumulates within the food chain and may accumulate to levels that are unsafe for human or wildlife consumption. A more detailed description of mercury methylation is presented in Appendix C (Mercury TMDL Development).

(This page left intentionally blank.)

B.5 Bioturbation

Bioturbation is the mixing and resuspension of sediment and benthic material by fish and macroinvertebrates. This disturbance of the sediments can have an impact on nutrient cycling and the availability of sediment-associated pollutants. In particular, bioturbation by bottom feeding fish can stir up the sediment and increase the movement of nitrogen and phosphorus into the water. Organic contaminants are typically hydrophobic and prefer to sorb to organic matter that may have settled to the lake bottom where bioturbation may cause resuspension and loading to the water column. Of great concern is the release of historical stores of OC Pesticides and PCBs.

For example, a positive relationship was observed between carp biomass and total suspended sediments within the water column and, more specifically, bream (a benthivorous fish) was shown to cause a 0.03 mg/L increase in total phosphorus per 100 kg of bream per hectare (Breukelaar, 1994). An additional study by Persson and Svensson (2006) showed increased concentrations of nitrogen and phosphorus in the water column of enclosures with benthivorus fish relative to controls with no fish.

Fish are stocked at the El Dorado Park lakes, Santa Fe Dam Park Lake, Echo Park Lake, North, Center and Legg lakes, Puddingstone Reservoir, Lincoln Lake, Peck Road Park Lake, and Westlake (California Department of Fish and Game, 2009). Fish have also been observed in Lake Calabasas and Sherwood Lake during recent monitoring events. Despite the confirmed presence of fish, available data do not include a comprehensive fish population assessment.

Another type of bioturbation is caused by macroinvertebrates that feed in the sediment. This first causes vertical mixing in the sediment. Some macroinvertebrates – particularly tubificid oligochaete worms – maintain burrows that enable them to feed at depth but defecate on the surface of the sediment. Such worms, which often occur at very high densities in organic sediments, can effectively pump significant amounts of both sediment-sorbed and porewater dissolved pollutants from depths of up to 10 inches or more into the water column (e.g., Reible et al., 1996).

Without comprehensive population assessments (species and population size), it is difficult to quantify the amount of pollutants in the water column that are directly related to bioturbation. Although bioturbation may not be precisely calculated without complete population assessments, it is assumed that samples collected at locations containing fish include water column concentrations impacted by bioturbation. In addition, impacts of bioturbation are included inherently in the eutrophication, mercury bioaccumulation, and OC Pesticides and PCBs models developed for each lake (see Appendices A, C, and H, respectively).

(This page left intentionally blank.)

B.6 Impacts of Sedimentation

Under certain conditions, lake sediments behave as significant sinks, removing pollutants from contact with the water column by allowing for deep burial and sequestration. In general, deep burial depends on the net sedimentation rate, which is the external sediment supply less resuspension. Rates of burial loss of specific compounds depend on the extent to which the compound is adsorbed to sediment, and lake dynamics (stratification, internal concentration, wind mixing, and depth) that determine rates of recycling of deposited material. Burial rates are often high for lakes in arid climates due to the sparse vegetative ground cover compared to areas receiving higher amounts of rainfall.

B.6.1 PHOSPHORUS

Inorganic phosphorus is particle-reactive. The burial and sequestration of phosphorus is an important mechanism that can reduce the mass of bioavailable phosphorus within the water column. Sedimentation rates depend on the specific lake dynamics as well as the size and settling velocity of the particulate matter to which the phosphorus is bound (Welch and Jacoby, 2004).

As explained in Sections B.4 through B.5, sediment stores of phosphorus can be released into the water column through multiple mechanisms. Thus, sedimentation may act as a sink under certain conditions and as a source under other conditions. First, anoxic environments, often present at the sediment-water interface, increase the reduction and release of phosphorus (Section B.4). Second, resuspension of sediment by wind mixing (Section B.3), and bioturbation (Section B.5) can result in additional recycling from the sediment to the water column.

B.6.2 MERCURY

In midwestern and eastern lakes, methylation in lake sediments is often the predominant source of methylmercury (MeHg) in the water column. However, in western lakes with high sedimentation rates, rapid burial tends to depress the relative importance of regeneration of MeHg from lake sediments. For instance, in McPhee Reservoir in Colorado (Tetra Tech, 2001), 71 percent of the MeHg present in the water column was estimated to derive from watershed inflows, while much of the MeHg created in lake sediment was apparently buried. Lakes with high sedimentation rates are therefore likely to respond approximately linearly to reductions in the watershed MeHg and total Hg load – although there may well be a delay in the response to load reductions, as found for McPhee Reservoir (Tetra Tech, 2001).

B.6.3 ORGANOCHLORINE PESTICIDES AND PCBS

Many OC Pesticides and PCBs have a high propensity to partition to sediment. For example, chlordane, DDT, dieldrin, and PCBs are hydrophobic and have low water solubilities. These characteristics increase the partitioning and, therefore, these OC Pesticides and PCBs are more likely to bind to sediment. The majority of the pollutant loads for such compounds will be stored in the lake sediments and further concentrated in aquatic organisms through bioaccumulation in the food chain. It is important to note that despite sediment contamination, water column concentrations of OC Pesticides and PCBs are frequently below detectable concentrations.

(This page left intentionally blank.)

B.7 References

Bachmann R.W., M.V. Hoyer, C. Fernandez, and D.E. Canfield. 2003. An alternative to proposed phosphorus TMDLs for the management of Lake Okeechobee. *Lake and Reservoir Management.* 19(3):251-264.

Brumbaugh, W.G., D.P. Krabbenhoft, D.R. Helsel, J.G. Wisner, and K.R. Echols. 2001. A national pilot study of mercury contamination of aquatic ecosystems along multiple gradients: Bioaccumulation in fish. *Biological Science Report USGS/BRD/DSR-2001-0009*. U.S. Geological Survey, Reston, VA.

Breukelaar, A.W., E.H. Lammens, J.G.P. Klein Bretelers Istavan Tartrai. 2006. Effects of benthivorous bream (*Abramis brama*) and carp (*Cyprinus carpio*) on sediment resuspension and concentrations of nutrients and chlorophyll *a*. *Freshwater Bio.* 32(1):113-121.

Carper, G.L., and R.W. Bachmann. 1984. Wind resuspension of sediments in a prairie lake. *Canadian Journal of Fisheries and Aquatic Sciences*. Vol. 41: 1763-1767.

Compeau, G.C., and R. Bartha. 1985. Sulfate-reducing bacteria: principle methylators of mercury in anoxic estuarine sediment. *Appl. Environ. Microbiol.* 50(2):498-502.

California Department of Fish and Game. 2009. Catchable Trout Planting. Available at: http://www.dfg.ca.gov/fish/hatcheries/fishplanting/SouthCoast.asp. Accessed on January 08, 2010.

Haggard, B.E., and P.A. DeLaune. 2005. Phosphorus flux from bottom sediments in Lake Eucha, Oklahoma. *J. Environ. Qual*. 34:724-728.

Martin, J.L., and S.C. McCutcheon. 1999. Hydrodynamics and transport for water quality modeling. Lewis Publ., Boca Raton, FL.

Moore, P.A., K.R. Reddy, and M.M. Fisher. 1998. Phosphorus flux between sediment and overlying water in Lake Okeechobee, Florida: spatial and temporal variations. *J. Environ. Qual.* 27:1428-1439.

Persson, A., and J.M. Svensson. 2006. Effects of benthivorous fish on biogeochemical processes in lake sediments. *Freshwater Bio.* 51(7):1298-1309.

Reible, D.D., V. Popov, K.T. Valsaraj, L.J. Thibodeaux, F. Lin, M. Dikshit, M.A. Todaro and J.W. Fleeger. 1996. Contaminant fluxes from sediment due to tubificid oligochaete bioturbation. *Water Research*, 30(3): 704-714.

Tetra Tech. 2001. Technical Support Document for Developing a Total Maximum Daily Load for Mercury in McPhee and Narraguinnep Reservoirs, Colorado. Report to U.S. Environmental Protection Agency, Region 8. Tetra Tech, Inc., Research Triangle Park, NC.

Thomann, R.V., J.P. Connolly, and T.F. Parkerton. 1992. An equilibrium model of organic chemical accumulation in aquatic food webs with sediment interaction. *Environmental Toxicology and Chemistry*, 11: 615-629.

Welch, E.B. and J.M. Jacoby. 2004. *Pollutant Effects in Freshwater, Applied Limnology (3rd edition).* Spon Press, London.

Wetzel, R. 2001. *Limnology*. Academic Press, 2001.

(This page left intentionally blank.)

Appendix C. Methodology for Mercury TMDL Development

(This page left intentionally blank.)

C.1 Introduction

USEPA Region IX is establishing TMDLs for impairments in nine lakes in the Los Angeles Region (Figure C-1). USEPA was assisted in this effort by the Los Angeles Water Quality Control Board (Regional Board). Impairments of these waterbodies include low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, algae, pH, mercury, lead, copper, chlordane, dieldrin, DDT, PCBs, and trash.

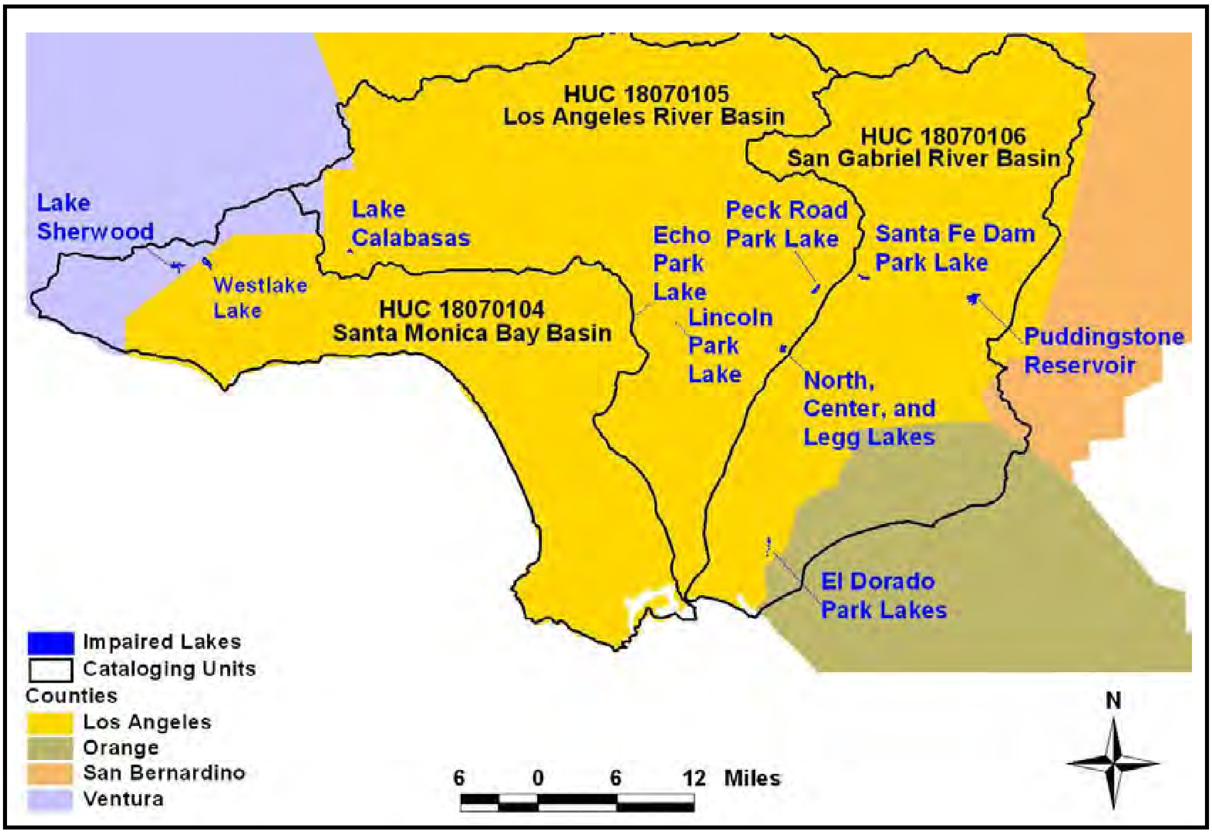


Figure C-1. Location of Impaired Lakes

Three of these waterbodies are listed as impaired by mercury due to elevated fish tissue concentrations: the El Dorado Park lakes, Puddingstone Reservoir, and Lake Sherwood. This appendix discusses the lake specific load allocations based on the measured tissue concentrations observed in each waterbody.

(This page left intentionally blank.)

C.2 Description of the Mercury Cycle

Selected aspects of the lake and watershed mercury cycle are summarized schematically in Figure C-2, based on the representations discussed in Hudson et al. (1994) and Tetra Tech (1999). The boxes represent stores of mercury, and the arrows represent fluxes. The top of the diagram summarizes the various forms of mercury that may be loaded to a lake.

It is important to recognize that mercury exists in a variety of forms, including elemental mercury (Hg(0)), ionic mercury (Hg(I) and Hg(II)), and compounds in which mercury is joined to an organic molecule.

In the figure, Hg(I) is ignored because Hg(II) species generally predominate in aquatic systems. Mercuric sulfide (HgS or cinnabar) is a compound formed from Hg(II) but is shown separately because it is the predominant natural ore. Organic forms of mercury include methylmercury (CH_3Hg or “MeHg”), and other natural forms such as dimethylmercury and manmade compounds such as organic mercury pesticides. (Where sorption and desorption are indicated in Figure C-2, “Hg(II)” and “MeHg” refer to the same common pools of water column Hg(II) and MeHg shown in the compartments at the top of the diagram.)

Dimethylmercury (CH_3-Hg-CH_3) is also ignored in the conceptual model shown in Figure C-2, because this mercury species seems to occur in measurable quantities only in marine waters. Organic mercury pesticides also have been ignored in this TMDL study, because such pesticides are not currently used in this country. Loads delivered to the lake historically have likely been buried under years of accumulated sediment. If contaminated upland sediments continue to contribute loading to the impaired waterbodies, recent tributary monitoring data will include this component of loading.

Ionic mercury and methylmercury form strong complexes with organic substances (including humic acids) and strongly sorb onto soils and sediments. Once sorbed to organic matter, mercury can be ingested by invertebrates, thus entering the food chain. Some of the sorbed mercury will settle to the lake bottom; if buried deeply enough, mercury in bottom sediments will become unavailable to the lake mercury cycle. Burial in bottom sediments can be an important route of removal of mercury from the aquatic environment.

Methylation and demethylation play an important role in determining how mercury will accumulate through the food web. Hg(II) is methylated by a biological process that appears to involve sulfate-reducing bacteria. Rates of biological methylation of mercury can be affected by a number of factors. Methylation can occur in water, sediment, and soil solutions under anaerobic conditions, and to a lesser extent under aerobic conditions. In lakes, methylation occurs mainly at the sediment-water interface and at the oxic-anoxic boundary within the water column. The rate of methylation is affected by the concentration of available Hg(II) (which can be affected by the concentration of certain ions and ligands), the microbial concentration, pH, temperature, redox potential, and the presence of other chemical processes. Methylation rates appear to increase at lower pH. Demethylation of mercury is also mediated by bacteria.

Both Hg(II) and methylmercury (MeHg) sorb to algae and detritus, but only the methylmercury is readily passed up to the next trophic level (inorganic mercury is relatively easily egested). Invertebrates eat both algae and detritus, thereby accumulating any MeHg that has sorbed to these. Fish eat the invertebrates and either grow into larger fish (which continue to accumulate body burdens of mercury), are eaten by larger fish or other piscivores, or die and decay.

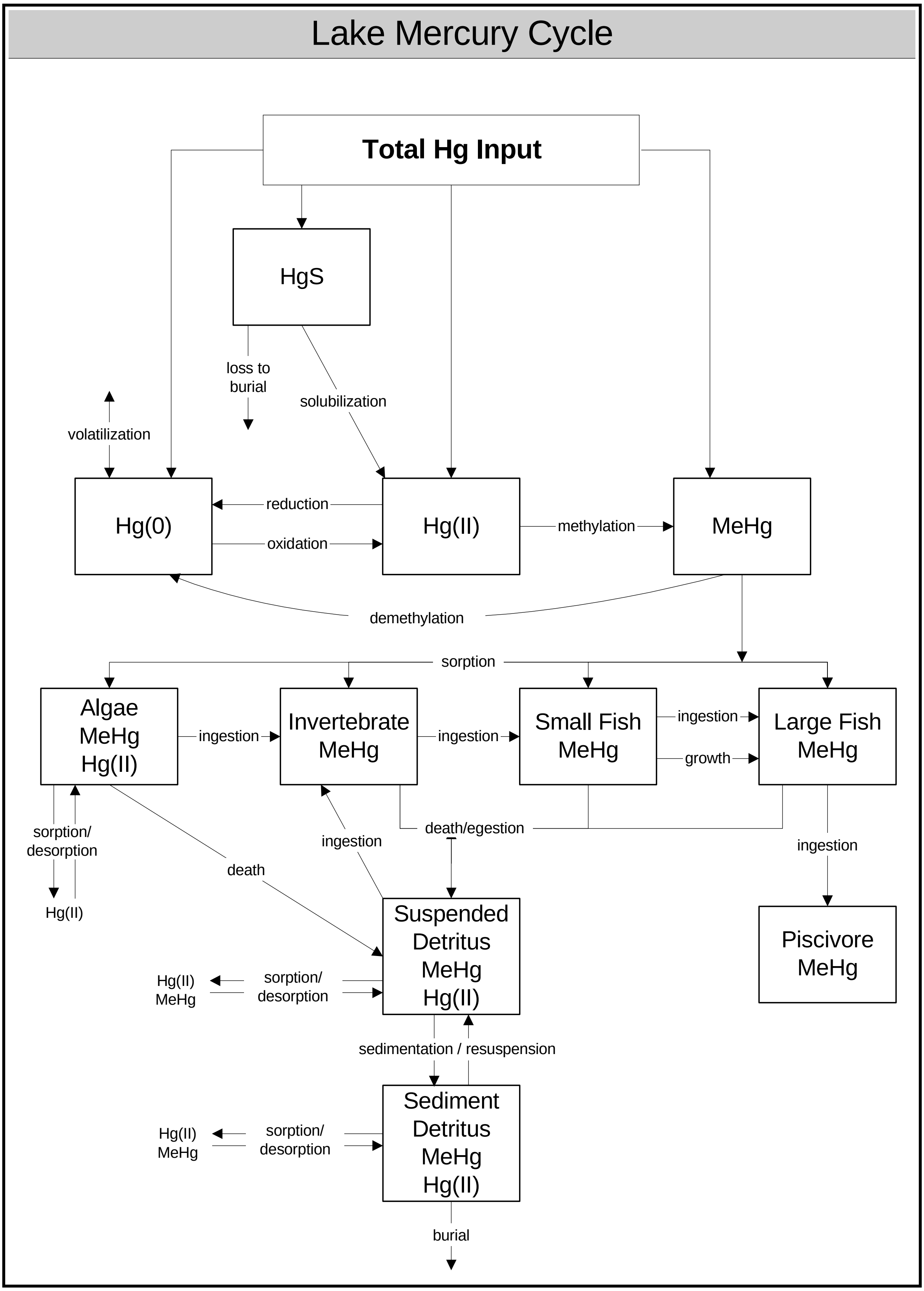


Figure C-2. Conceptual Diagram of Lake Mercury Cycle

Typically, almost all of the mercury found in fish (greater than 95 percent) is in methylmercury form. Studies have shown that fish body burdens of mercury tend to increase concurrently with increasing size or age of the fish, under conditions of constant exposure.

Although it is important to identify external sources of mercury to the reservoir, there may be fluxes of mercury within the reservoir that would continue for some time even if all external sources of mercury load were eliminated. The most important store of mercury within the reservoir is the bed sediment. Mercury in the bed sediment may cause exposure to biota by being:

- Resuspended into the water column, where it is ingested or it adsorbs to organisms that are later ingested.
- Methylated by bacteria. The methylmercury tends to attach to organic matter, which may be ingested by invertebrates and thereby introduced to the lake food web.

(This page left intentionally blank.)

C.3 Source Assessment

Sources of mercury loading to a lake may include both point and nonpoint sources. For purposes of allocations among mercury sources, federal regulations distinguish between allocations for point sources regulated under NPDES permits (for which waste load allocations are established) and nonpoint sources that are not regulated through NPDES permits (for which load allocations are established) (see 40 CFR 130.2). The most significant source of mercury in point source discharges is wastewater associated with the installation or removal of mercury amalgam dental fillings. Sources in the watershed include junkyards housing automobiles where mercury-containing switches have not been removed prior to crushing, and landfills where fluorescent light bulbs have not been properly disposed. Significant releases to the atmosphere may occur from coal-power plants, cement manufacturing facilities, oil refineries, and chlor-alkali plants. This section describes how loading from point and nonpoint sources were estimated for the mercury-impaired watersheds.

C.3.1 POINT SOURCES

Point sources are discharges that occur at a defined point, or points, such as a pipe or storm drain outlet. Most point sources are regulated through the NPDES permitting process.

C.3.1.1 MS4 Permittees

In 1990 USEPA developed rules establishing Phase 1 of the NPDES stormwater program, designed to prevent pollutants from being washed by stormwater runoff into the Municipal Separate Storm Sewer Systems (MS4), or from being directly discharged into the MS4 and then discharged into local waterbodies. Phase I of the program required operators of medium and large MS4s (those generally serving populations of 100,000 or more) to implement a stormwater management program as a means to control polluted discharges. Phase II of the program extends the requirements to operators of small MS4 systems, which must reduce pollutants in stormwater to the maximum extent practicable (MEP) to protect water quality.

Mercury loads from urban stormwater runoff and associated sediment are estimated from monitoring data collected at the mouth of each major tributary that discharges to a mercury impaired lake (Appendix G, Monitoring Data) and simulated flows and sediment loads (Appendix D, Wet Weather Loading). Two flow-calibrated watershed models (using the Loading Simulation Program in C++ [LSPC]) models were previously developed for the San Gabriel and Los Angeles river basins (Tetra Tech, 2004; Tetra Tech, 2005). To estimate stormwater runoff volumes and sediment loads, average monthly areal flow rates have been extracted for each land use and applied to the land use composition that drains to a MS4 for each lake. Sediment event mean concentrations for each land use are used to estimate sediment loads. Appendix D (Wet Weather Loading) describes the LSPC model output, summarizes the mercury monitoring data, and presents the resulting mercury loading from MS4 systems.

These systems may also discharge during dry weather as a result of irrigation, car washing, etc. Estimation of mercury loading from MS4 systems in dry weather is based on SCCWRP regional flow estimates and local monitoring data as described in Appendix F (Dry Weather Loading).

C.3.1.2 Non-MS4 NPDES Discharges

In addition to MS4 stormwater dischargers, the NPDES program regulates stormwater discharges associated with industrial and construction activities and non-stormwater discharges (individual and general permits). . To quantify mercury loading from non-MS4 NPDES discharges, the permit databases maintained by the Los Angeles Regional Board were downloaded for the San Gabriel River and Santa Monica Bay basins. Geographic information listed for each permit was used to determine which facilities are located in the watersheds of the three mercury impaired lakes. Mercury loading from each facility was estimated based on the reported disturbed area. The facilities and estimated loads are described in more detail in the lake specific sections of this report.

C.3.1.3 Additional Inputs

Several of the lakes addressed by this TMDL have additional point source inputs that do not currently have NPDES permits. Most are supplemental flows from groundwater wells, or potable water that maintain lake levels. Information pertaining to flow volumes from these sources was provided by park staff at each lake (generally based on water usage information from the water suppliers). Where accessible, the Regional Board and USEPA sampled water quality from these inputs during the 2009 sampling events. Mercury loading was calculated from observed concentration data and an estimate of annual flow volumes to each lake.

C.3.2 NONPOINT SOURCES

Mercury loading from nonpoint sources originates from sources that do not discharge at a defined point. This section describes the methods used to estimate loading from nonpoint sources.

C.3.2.1 Atmospheric Deposition

Mercury deposition from the atmosphere to the earth’s surface may occur in several forms: gaseous elemental mercury (Hg(0)), divalent ionic mercury (Hg(II)), reactive gaseous mercury (RGM), and aerosol particulate mercury (Hg-P). Atmospheric deposition can be divided into short-range or near-field deposition, which includes deposition from sources located near the watershed, and long-range or far-field deposition, which includes mercury deposition from regional and global sources. Mercury emitted from manmade sources usually contains both gaseous elemental mercury (Hg(0)) and divalent mercury (Hg(II)). Hg(II) species, because of their solubility and their tendency to attach to particles, are redeposited relatively close to their source (probably within a few hundred miles), whereas Hg(0) remains in the atmosphere much longer, contributing to long-range transport.

Deposition may either occur in wet form (associated with precipitation) or dry form (associated with particulate settling). Wet deposition is monitored at select locations across the country by the Mercury Deposition Network (MDN). There is one MDN site in Southern California, but it has only been active since May of 2006. The rates of wet mercury deposition to each lake water surface were estimated with a regression approach that utilized nitrate and sulfate wet deposition data collected by the National Atmospheric Deposition Program (NADP), along with mercury wet deposition data collected by the MDN (see Appendix E, Atmospheric Deposition).

Dry deposition is more difficult to monitor and less localized data are available to estimate this component. To estimate loading from this component, grid-cell output from regional deposition models developed by USEPA were obtained for each lake impaired by mercury (see Appendix E, Atmospheric Deposition).

To evaluate potential nearfield sources at each impaired lake, the USEPA Toxics Release Inventory (TRI) was used to determine the proximity of point sources that may contribute to airborne mercury loads including coal-fired power plants, steel recycling facilities, waste incinerators, cement and lime kilns, smelters and gold mine roasters, pulp and paper mills, and chlor-alkali factories.

Precipitation events following recent forest fires also result in increased loads of total and methylmercury from the watershed and release of elemental mercury to the atmosphere which is then available for deposition.

C.3.2.2 Watershed Loading

Mercury loads from areas that do not drain to an MS4 system are estimated from monitoring data collected at the mouth of each major tributary that discharges to a mercury impaired lake (Appendix G, Monitoring Data) and simulated flows and sediment loads (Appendix D, Wet Weather Loading). Two flow-calibrated LSPC models were previously developed for the San Gabriel and Los Angeles river basins (Tetra Tech, 2004; Tetra Tech, 2005). To estimate runoff volumes and sediment loads, average monthly areal flow rates have been extracted for each land use and applied to the land use composition that does not drain to an MS4 for each lake. Sediment event mean concentrations for each land use are used to estimate sediment loads. Appendix D (Wet Weather Loading) describes the LSPC model output, summarizes the mercury monitoring data, and presents the resulting wet weather mercury loading areas that do not discharge to an MS4.

In addition to pollutant loads delivered during storm events (discussed in Appendix D, Wet Weather Loading), it is important to account for loads that are delivered to a waterbody during dry weather. Nonpoint sources during dry weather include groundwater discharges, irrigation (reclaimed water is used for irrigation of parklands adjacent to two of the waterbodies), fertilization of adjacent parkland, and other miscellaneous urban sources. Estimation of dry weather pollutant loading is discussed in Appendix F (Dry Weather Loading).

C.3.2.3 Methylation

Accumulation of mercury in biota is determined by methylmercury concentrations, not total mercury. These concentrations reflect both methylation within the lake and external loading of methylmercury. Methylation of mercury occurs under oxygen-poor, reducing conditions. Wetland areas are particularly likely sites for methylation in the watershed. Other likely sites include shallow riparian groundwater, the bottom waters and sediment of impoundments that stratify and go anoxic, and beaver ponds and their associated wetlands. Sampling for methylmercury concentrations in the water column and sediment was performed at each tributary or input to the impaired lakes. One of the tributaries at Lake Sherwood exhibited characteristics associated with high methylation (see Section 12). The implementation section for each lake will address how to best manage these loads.

Dredging activities to remove accumulated sediment from lakes and sedimentation basins may have significant impacts on total and methylmercury loading to lake waters. In theory, removal of accumulated sediment should reduce the amount of total and methylated mercury stored in the sediments. Unfortunately, the removal process may disturb and release methylated mercury into the water column and increase the bioavailability of the metal. Additionally, removal of the top layers of sediment may uncover layers deposited during the 1960s through 1980s when air emissions of mercury were less adequately controlled. Proper testing and planning is required to ensure that removal activities do not add to the overall mercury burden.

C.3.2.4 Direct Geologic Sources

Geological formations containing significant mercury concentrations have a higher probability of occurrence in mineralized areas along fault lines, intrusive dikes in igneous formations, or resulting from natural springs. Volcanic activity has the potential to release mercury into the air, so areas with large ash deposits may contain higher concentrations of mercury. Mercury is also more likely to occur in shale and slate deposits as they are derived from clays, which have high affinities for adsorbing metals such as mercury (this affinity explains why coal burning power plants emit mercury). Sediment mercury concentrations measured at the mouth of each major tributary include the geologic component as well as anthropogenic sources of mercury.

The California Geological Survey has posted a map online of the earthquake hazard across the state (http://www.consrv.ca.gov/cgs/rghm/psha/Pages/index.aspx). This map indicates that fault line activity in these three watersheds is moderate.

The U.S. Geological Survey conducted a geochemical survey of stream sediments and generated estimates of mercury concentrations in soil by county in their Open-File Report 2004-1001 accessible via their website (http://tin.er.usgs.gov/geochem/doc/home.htm). The mean concentration estimated for Ventura County (where Lake Sherwood is located) is 0.064 ppm with a standard deviation of 0.034 ppm (minimum of 0.022 and maximum of 0.232 ppm). The mean concentration estimated for the County of Los Angeles (where El Dorado Park lakes and Puddingstone Reservoir are located) is 0.149 ppm with a standard deviation of 0.217 ppm (minimum of 0.010 and maximum of 1.849 ppm). The nearest sample to Lake Sherwood was NURE record ID RA000197, which had a mercury concentration of 0.02 ppm whereas the nearest sample to El Dorado Park lakes was NURE record ID RA000163 with a mercury concentration of 0.07 ppm. At Puddingstone Reservoir the nearest sampling location, which was analyzed for mercury was NURE record ID RA000425, had a mercury concentration of 0.08 ppm.

C.3.2.5 Indirect Geologic Sources

Geological formations containing deposits of precious metals (e.g., gold, silver, and copper) have been targets of historic and current mining activities. In cases where the desired metals are contained in ore as opposed to veins, extraction of the desired metal commonly occurs through the process of amalgamation, in which mercury is used as the amalgam. Amalgamation is an easy and inexpensive process of removing fine metal particles from ore, but when poorly implemented, it can lead to spillage of mercury, contaminated mine tailings, and localized atmospheric deposition.

Oil production may also release mercury into the environment, particularly in California. Mercury often causes corrosion and fouling problems in pipelines and equipment and is easily transferred from oil to water during refinement processes. Researchers at the University of North Dakota found that typical mercury concentrations in crude oil across the globe are less than 20 ppb, but that some measurements taken in California have been as high as 24,000 ppb (http://www.undeerc.org/catm/pdf/area3/MJH_Crude_2002.pdf). Wilhelm et al. (2004) also report that average concentrations measured from crude oil samples in California were higher than those measured from other states in the US (11.3 ppb compared to 4.3 ppb).

Thus, in relation to mining potential and oil production, the geological formations in a watershed can indirectly influence mercury loadings. No precious metal mines or oil refineries are known to have operated within the watersheds of the three mercury impaired lakes. However, the presence of oil refineries in the general region indicates that high mercury sediment concentrations may exist.

C.4 Linkage Analysis

The linkage analysis provides the quantitative basis for determining the loading capacity of each impaired lake. This in turn allows estimation of the Total Maximum Daily Load (TMDL), and allocation of that load to urban sources (wasteload allocations) and rural sources (load allocations). The TMDL also contains a Margin of Safety, which is described in detail below.

Neither data nor resources are available to create and calibrate detailed lake response models for mercury cycling in the El Dorado Park lakes, Puddingstone Reservoir, and Lake Sherwood. The key to the TMDL target is achieving acceptable concentrations in fish. The mercury TMDLs for these three lakes are being developed in a similar fashion to the Big Bear Lake TMDL, which applies watershed specific mercury concentrations to simulated sediment loads and water volumes (Tetra Tech, 2008; note: as of the writing of these TMDLs, the Big Bear Lake TMDL for mercury has not been finalized and approved). For these three lakes, previously developed LSPC models provide a mechanism for incorporating wet, normal, and dry simulation years into the TMDL.

In midwestern and eastern lakes, methylation in lake sediments is often the predominant source of MeHg in the water column. However, in western lakes with high sedimentation rates, rapid burial tends to depress the relative importance of regeneration of MeHg from lake sediments. For instance, in McPhee Reservoir in Colorado (Tetra Tech, 2001), 71 percent of the MeHg present in the water column was estimated to derive from watershed inflows, while much of the MeHg created in lake sediment was apparently buried. Lakes with high sedimentation rates are therefore likely to respond approximately linearly to reductions in the watershed MeHg and total Hg load – although there may well be a delay in the response to load reductions, as found for McPhee Reservoir (Tetra Tech, 2001).

Lakes in arid climates are predisposed to high rates of sedimentation given the lower density of vegetative ground cover compared to areas receiving higher amounts of rainfall. Each of the three mercury impaired systems addressed by this TMDL likely experience average to high rates of sedimentation. In fact, Lake Sherwood is listed as impaired by sedimentation, as was Big Bear Lake (Tetra Tech, 2008). Two studies have summarized sedimentation rates for Puddingstone Reservoir. According to the Reservoir Sedimentation Database (accessed 6/5/2009), the average annual historical sedimentation rate measured from 1915 to 1941 for Puddingstone Reservoir was 16 ac-ft per year (approximately 0.76 inches per year). The Department of Boating and Waterways and State Coastal Conservancy (2002) reports that the average annual sedimentation rate measured in Puddingstone Reservoir from 1925 to 1980 was 31 ac-ft per year (approximately 1.5 inches per year). For Lake Sherwood, the reported average annual sedimentation rate measured from 1905 to 1938 ranged from 2.5 to 10 acre-feet per year (0.22 to 0.88 inches per year); this rate has likely increased with development around the perimeter of the lake. Site specific data for the El Dorado Park lakes are not available. However, watershed loading at El Dorado Park is less significant than loads associated with the groundwater source used for lake filling (see Appendix F, Dry Weather Loading).

The available evidence suggests that sedimentation rates are likely to diminish the relative importance of MeHg recycling from lake sediments if coupled with reductions in mercury. This, in turn, suggests that MeHg exposure concentrations in each lake should respond approximately linearly to reductions in mercury load, particularly if conditions favoring methylation are discouraged (i.e., anoxic conditions near the sediment-water interface). While this is the best assumption that can be made with the current data, two caveats should be mentioned. First, the burial and sequestration of MeHg due to sedimentation may be counteracted by dredging activities that may occur periodically as part of an overall lake management plan. Second, the potential role of peripheral wetlands or forebays as a locus of mercury methylation and subsequent loading to each lake is currently unknown. It is clear that reductions in external mercury loads to each waterbody will be beneficial, although a program of adaptive implementation may need to be pursued if elevated fish tissue concentrations persist.

Nationally, authors such as Brumbaugh et al. (2001) have shown a log-log linear relationship between MeHg in water and MeHg in fish tissue normalized to length. However, this relationship is well-approximated by a linear relationship for the ranges of fish tissue concentration of concern for these impaired lakes. Until such time as lake response models for mercury are constructed for these waterbodies, and sufficient calibration data collected to develop them, an assumption of an approximately linear response of fish tissue concentrations to changes in external loads is sufficient for the development of these TMDLs.

Each of the three lakes shows exceedances of the fish tissue mercury concentration in largemouth bass. Exceedances of the total mercury water quality standard were not observed in any of the impaired waterbodies; however, two lakes had exceedances of the dissolved methylmercury water quality standard (Lake Sherwood and Puddingstone Reservoir; Note: the observed data were based on the total fraction, while the water column target is for the dissolved fraction, resulting in more conservative assessments). Because limited samples were available to compare to the dissolved methylmercury target and the long-term average fish tissue concentrations are more predictive of exposure pathways for humans and wildlife, the TMDLs were based on the reduction required to meet the fish tissue guideline. In addition, the mercury reductions required by the fish tissue data were consistently higher than the reductions required to meet the methylmercury water column target; therefore, meeting the reductions for fish tissue should also result in attainment of the water column target for methylmercury.

C.5 TMDL Development

The TMDL is defined by the loading capacity. A waterbody's loading capacity represents the maximum amount pollutant loading that can be assimilated without violating water quality standards (40 CFR 130.2(f)). For mercury, this is the maximum amount of mercury loading and methylation uptake consistent with meeting the numeric target of 0.22 ppm for mercury in 350mm largemouth bass.

C.5.1 LOADING CAPACITY AND ALLOCATIONS

A model of lake response and fish bioaccumulation has not been created at this time for these impaired lakes. Rather, it is assumed that, in the long term, fish tissue concentrations will respond approximately linearly to reductions in mercury loads. This assumption has been found to be a reasonable first-order approximation in other systems with high burial rates, such as McPhee and Narraguinnep reservoirs in Colorado (Tetra Tech, 2001). For McPhee in particular, a detailed model of lake mercury cycling and bioaccumulation was created using the D-MCM model (Tetra Tech, 1999). The calibrated model yielded predictions that were well-approximated by the assumption of a linear response of fish tissue concentration to reductions in external mercury loads.

Calculating the loading capacity first requires an estimate of the existing mercury concentration in largemouth bass, the predominant trophic level 4 fish in each waterbody. To do this, a linear regression analysis was performed on tissue concentrations versus length for each lake, which was then used to predict the existing concentration associated with the target size fish (see Appendix G [Monitoring Data] for details regarding fish tissue monitoring data). The resulting linear regression equations are presented as

$$Hg(fish) = Y\text{-}intercept + Slope \cdot Len$$

where *Hg(fish)* is the total mercury concentration in largemouth bass (ppm), *Len* is length in mm, and *Y-intercept* and *Slope* are constants representing the point at which the line crosses the y-axis and the slope or gradient of the line, respectively. In addition, the one-sided 95 percent upper confidence limits on mean predictions about the regression line (95 percent UCL) and the 95 percent upper prediction intervals on individual predicted concentrations (95 percent UPI) were calculated. The UPI gives the confidence limit on the individual predictions for a given length while the UCL gives the confidence limit on the average of the predictions for a given length. These regressions have non-zero intercepts and should not be considered valid for lengths less than the representative dataset (150 to 200 mm depending on the lake).

For mercury, long-term cumulative exposure is the primary concern. Therefore, it is appropriate to use the 95 percent UCL rather than the UPI to provide a Margin of Safety on the appropriate age class. Use of the UCL provides a Margin of Safety because it represents an upper confidence bound on the long-term exposure concentration.

The one-sided 95 percent UCL is given by

$$UCL_{0.95} = \mu_{y|x_0} + t_{0.05,n-2} \cdot s_{\mu_y|x_0}$$

where $\mu_{y|x}$ is the predicted value of y given $x=x_0$, t is the Student's t-statistic with n-1 degrees of freedom, and n is the number of observations used in the regression. The variance on the prediction at $x=x_0$, $s^2{}_{\mu_y|x_0}$, is given by

$$s^2_{\mu_y|x_0} = s^2_{y|x} \cdot \left[1/n + (x_0 - \bar{x})^2 \Big/ \sum (x_i - \bar{x})^2 \right],$$

where x_0 is the value of the independent variable (*Len*) at which the prediction is made, $\bar{x}$ is the mean of the observed independent variables, x_i, and $s^2_{y|x}$ is the standard error of the model estimates. For example, for the El Dorado Park lakes data (see Section 8), this yields:

$$UCL_{0.95}[Hg(fish)] = -0.15316 + 0.001461 \cdot Len + 2.0796 \cdot 0.14285 \cdot \sqrt{1/23 + (Len - 365.9)^2 / 106245}$$

This equation expresses the upper 95 percent confidence limit on predicted fish tissue mercury concentrations for any length (*Len*). The first two terms alone would generate the prediction line; the addition of the last term results in the UCL line.

Both the observed data and the predicted concentrations show that mercury concentrations in largemouth bass typically exceed the target of 0.22 ppm in each lake. The target length for assessing compliance with this tissue concentration is 325-375 mm for largemouth bass. A range is provided for compliance; however, an average of 350 mm largemouth bass is used for TMDL calculations. The predicted mercury concentration based on a one-sided 95 percent upper confidence limit on mean predictions about the regression line (95 percent UCL) for this length is compared to the target fish concentration to determine the required reduction in mercury loading, which includes a Margin of Safety as described above.

For each lake, the fraction of existing load consistent with attaining the target (the loading capacity) is the ratio of the target (0.22 ppm) to the best estimate of current average concentrations in the target fish population. The difference between the direct regression estimate and the 95 percent UCL provides the Margin of Safety. Therefore, the allocatable fraction of the existing load (the loading capacity less the Margin of Safety) is the ratio of the target to the 95 percent UCL. The resulting loading capacities and allocatable loads are expressed as fractions of the existing load in the lake-specific chapters. For example, at Lake Sherwood the predicted total mercury concentration for a 350 mm largemouth bass is 0.607 ppm, and the 95th percent UCL is 0.744 ppm. The following calculations apply:

Loading capacity as fraction of existing load = 0.22 ppm / 0.607 ppm = 0.362

Allocatable load as fraction of existing load = 0.22 ppm / 0.744 ppm = 0.296

Margin of safety as fraction of the existing load = 0.362 – 0.296 = 0.067

The loading capacity can also be expressed as grams per year (g/yr) using the existing load from the source assessments and the calculated fractions of the existing load. Estimates of the existing mercury load to each lake are discussed in Appendices D, E, and F. Specifically, the loading capacity is presented as a percentage of the existing load (in grams per year). This value can be further broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margin of Safety (MOS) using the general TMDL equation:

$$TMDL = Loading\ Capacity = \sum WLAs + LAs + MOS$$

For division of WLAs and LAs, the percent reduction in mercury loading was applied equally to all sources of mercury in each watershed based on the results of the lake-specific source assessments.

C.5.2 MARGIN OF SAFETY

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. An implicit MOS is included based on comparison of total mercury concentrations in fish tissue to the methylmercury guideline (most, but not all, of the total mercury in fish is in the methyl form) (Note: additional lake-specific conditions or assumptions may also have been included in the implicit MOS). An explicit MOS is provided by the use of the 95 percent UCL to determine the allocatable load.

C.5.3 DAILY LOAD EXPRESSION

USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River. Although it is long-term cumulative load rather than daily loads of mercury that are driving the bioaccumulation of mercury in fish in, these TMDLs do present a maximum daily load according to the guidelines provided by USEPA (2007). Because the majority of loads occur during wet weather events, the daily maximum allowable load of mercury is calculated from the maximum daily storm flow rate (estimated from the 99th percentile flow) multiplied by the allowable concentration for mercury consistent with achieving the long-term loading target. For lakes with significant loading from other sources, such as supplemental water additions, appropriate daily flow rates were identified and multiplied by the allowable concentration for mercury to determine the daily maximum allowable load.

(This page left intentionally blank.)

C.6 References

Brumbaugh, W.G., D.P. Krabbenhoft, D.R. Helsel, J.G. Wisner, and K.R. Echols. 2001. A National Pilot Study of Mercury Contamination of Aquatic Ecosystems Along Multiple Gradients: Bioaccumulation in Fish. Biological Science Report USGS/BRD/DSR-2001-0009. U.S. Geological Survey, Reston, VA.

Department of Boating and Waterways and State Coastal Conservancy. 2002. California Beach Restoration Study. http://www.dbw.ca.gov/Environmental/BeachReport.aspx.

Hudson, R.J.M., S.A. Gherini, C.J. Watras, and D.B. Porcella. 1994. Modeling the biogeochemical cycle of mercury in lakes: The Mercury Cycling Model (MCM) and its application to the MCL study lakes. In *Mercury as a Global Pollutant* ed. C.J. Watras and J.W. Huckabee, pp. 475-523. Lewis Publishers, Chelsea, MI.

Tetra Tech. 1999. Dynamic Mercury Cycling Model for Windows 95/NTJ - A Model for Mercury Cycling in Lakes, D-MCM Version 1.0, Users Guide and Technical Reference. Electric Power Research Institute, Palo Alto, CA.

Tetra Tech. 2001. Technical Support Document for Developing a Total Maximum Daily Load for Mercury in McPhee and Narraguinnep Reservoirs, Colorado. Report to U.S. Environmental Protection Agency, Region 8. Tetra Tech, Inc., Research Triangle Park, NC.

Tetra Tech. 2004. Model Development for Simulation of Wet-Weather Metals Loading from the Los Angeles River Watershed. Prepared for USEPA Region 9 and the Los Angeles Regional Water Quality Control Board, May 2004.

Tetra Tech. 2005. Model Development for Simulation of Wet-Weather Metals Loading from the San Gabriel River Watershed. Prepared for USEPA Region 9 and the Los Angeles Regional Water Quality Control Board, October 2005.

Tetra Tech. 2008. Big Bear Lake Technical Support Document for Mercury TMDL. Prepared for USEPA Region 9 and the Santa Ana Regional Water Quality Control Board, October 2008.

Wilhelm, S. M., L. Liang, D. Cussen, and D.A. Kirchgessner. 2004. Mercury in Crude Oil Processed in the United States. Environ. Sci. Technol., 2007, 41 (13), pp 4509–4514.

(This page left intentionally blank.)

Appendix D. Estimation of Wet Weather Loading from Runoff and Sediment Transport

(This page left intentionally blank.)

D.1 Introduction

USEPA Region IX is establishing TMDLs for impairments in nine lakes in the Los Angeles Region (Figure D-1). USEPA was assisted in this effort by the Los Angeles Water Quality Control Board (Regional Board). The waterbodies are impaired by low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, algae, pH, mercury, lead, copper, chlordane, DDT, dieldrin, PCBs, and trash.

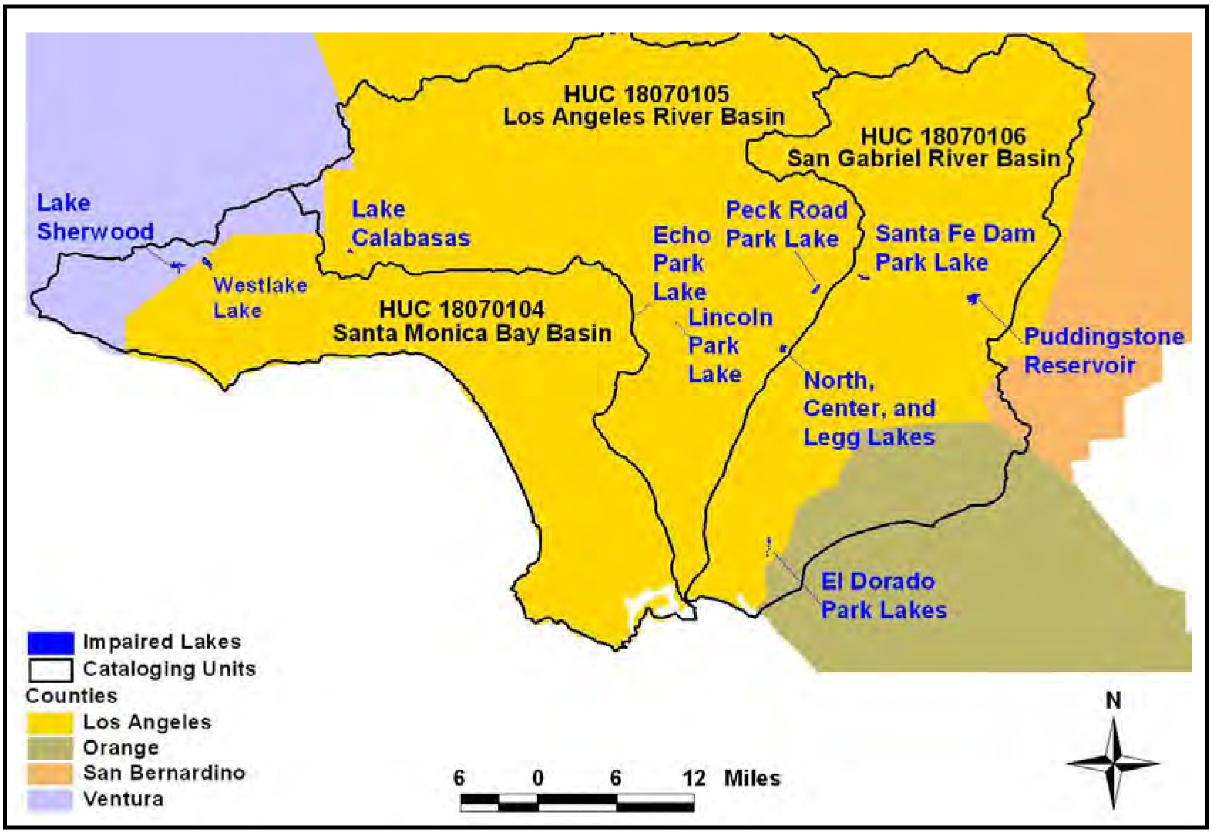


Figure D-1. Location of Impaired Lakes

Estimation of watershed loading as a result of wet weather events is based on calibrated watershed models developed for the Los Angeles and San Gabriel river basins. Each model was previously calibrated for flow and metals loading. For the purposes of developing nutrient and mercury TMDLs, the simulated flows predicted for land uses in spatially relevant modeling subbasins were used along with regional event mean concentrations (EMCs) of total suspended sediment, nitrogen, and phosphorus to estimate loading to each impaired waterbody. Mercury concentrations were based on monitoring data collected at the mouth of each major storm drain or tributary to the mercury-impaired lakes.

Each of the impaired lakes, with the exception of Lake Sherwood, is in either the Los Angeles or San Gabriel River Basin. Lake Sherwood, however, is in close proximity to the Los Angeles River Basin, and the land use coverage compiled for this model covers the Lake Sherwood drainage area.

Each of the river basin watershed models (using the Loading Simulation Program in C++ [LSPC] model) was calibrated for flow. Model output available for the years 1983 to 2006 was used to estimate average monthly runoff depths by land use for each LSPC modeling subbasin that contains one of the impaired lakes addressed by this TMDL document (Note: all references to runoff in this appendix are associated with both the storm drain system and nonpoint sources). These years represent dry, normal, and wet conditions for the Los Angeles area and provide a reasonable estimate of average runoff conditions for these waterbodies.

The TMDLs are allocated based on subwatershed and MS4 stormwater permittee. A GIS environment was used to overlay the subwatersheds, jurisdictions, and the LSPC land use coverages to estimate the area of each modeled land use within a subwatershed/jurisdiction area. Monthly runoff volumes were then calculated for each combination of land use/subwatershed/jurisdiction based on land use area and simulated runoff depth.

To estimate loading of nutrients, metals, and Organochlorine (OC) Pesticides and PCBs to each waterbody from upland areas, event mean concentrations (EMCs) based on the Southern California Coastal Water Research Project (SCCWRP) and the county of Los Angeles monitoring studies were applied to the average monthly runoff volumes calculated for each land use/subwatershed/jurisdiction area (i.e., water quality EMCs and runoff volume were used to calculate loadings for nutrients, metals, and OC Pesticides and PCBs). Mercury loads are estimated from simulated runoff volumes, predicted sediment loads (based on EMCs), and watershed monitoring data. Specifically, mercury loading is associated with both sediment and runoff from upland areas. To determine sediment loading of mercury, the sediment EMCs and runoff volumes were used to calculate sediment loads, and the sediment mercury concentrations from monitoring data were then applied to these sediment loads. Similar to the nutrients, metals, and OC Pesticides and PCBs, mercury loading associated with runoff from upland areas was calculated using the water quality monitoring data and simulated runoff volumes. Section D.3 provides more details on these calculations.

These calculated loads represent a portion of the existing pollutant load to each impaired waterbody. Estimates of loading from other sources are described in other sections or appendices of the TMDL report. The summation of loads from all sources will then be used to estimate existing loading to each lake.

D.2 Simulation of Urban Runoff

D.2.1 MODEL OVERVIEW

The U.S. Environmental Protection Agency's (USEPA) Loading Simulation Program C++ (LSPC) has been used to represent the hydrological and water quality conditions in the Los Angeles River and San Gabriel River watersheds (Tetra Tech, 2004; Tetra Tech, 2005). LSPC is a component of the USEPA's TMDL Modeling Toolbox, which has been developed through a joint effort between USEPA and Tetra Tech. It integrates a geographical information system (GIS), comprehensive data storage and management capabilities, a dynamic watershed model (a re-coded version of USEPA's Hydrological Simulation Program – FORTRAN [HSPF] [Bicknell et al., 2001]), and a data analysis/post-processing system into a convenient PC-based Windows interface that dictates no software requirements. LSPC is capable of representing loading, both flow and water quality, from nonpoint and point sources, and simulating in-stream processes. LSPC can simulate flow, sediment, metals, nutrients, pesticides, and other conventional pollutants, for pervious and impervious lands and waterbodies. Each river basin LSPC model was configured to simulate the respective watershed as a series of hydrologically connected subwatersheds.

Each watershed model represented the variability of nonpoint source contributions through dynamic representation of hydrology and land practices. Each model also included all point and nonpoint source contributions. Key components of the watershed modeling included:

- Watershed segmentation
- Meteorological data
- Land use representation
- Soils
- Reach characteristics
- Point source discharges
- Hydrology representation
- Pollutant representation
- Flow data

D.2.2 WATERSHED SEGMENTATION

In order to evaluate sources contributing to an impaired waterbody and to represent the spatial variability of these sources, the contributing drainage area was represented by a series of subwatersheds. This subdivision was primarily based on the stream networks and topographic variability, and secondarily on the locations of flow and water quality monitoring stations, consistency of hydrologic factors, land use consistency, and existing watershed boundaries.

The subwatersheds for the Los Angeles River basin were delineated after dividing the watershed into two general components: headwaters and lower-elevation urban areas. The headwaters were generally more mountainous and had steeper slopes than the downstream portion of the watershed. In this mountainous region, Digital Elevation Models (DEMs) were utilized for delineating subwatersheds. Specifically, subwatershed boundaries were based upon slopes, ridges, and projected drainage patterns. Alternatively, in the downstream flatter areas of the watershed, maps illustrating the catchment network and drainage pipes were used to isolate sewer-sheds. The Los Angeles River watershed was ultimately delineated into 35 subwatersheds for appropriate hydrologic connectivity and representation (Figure D-2).

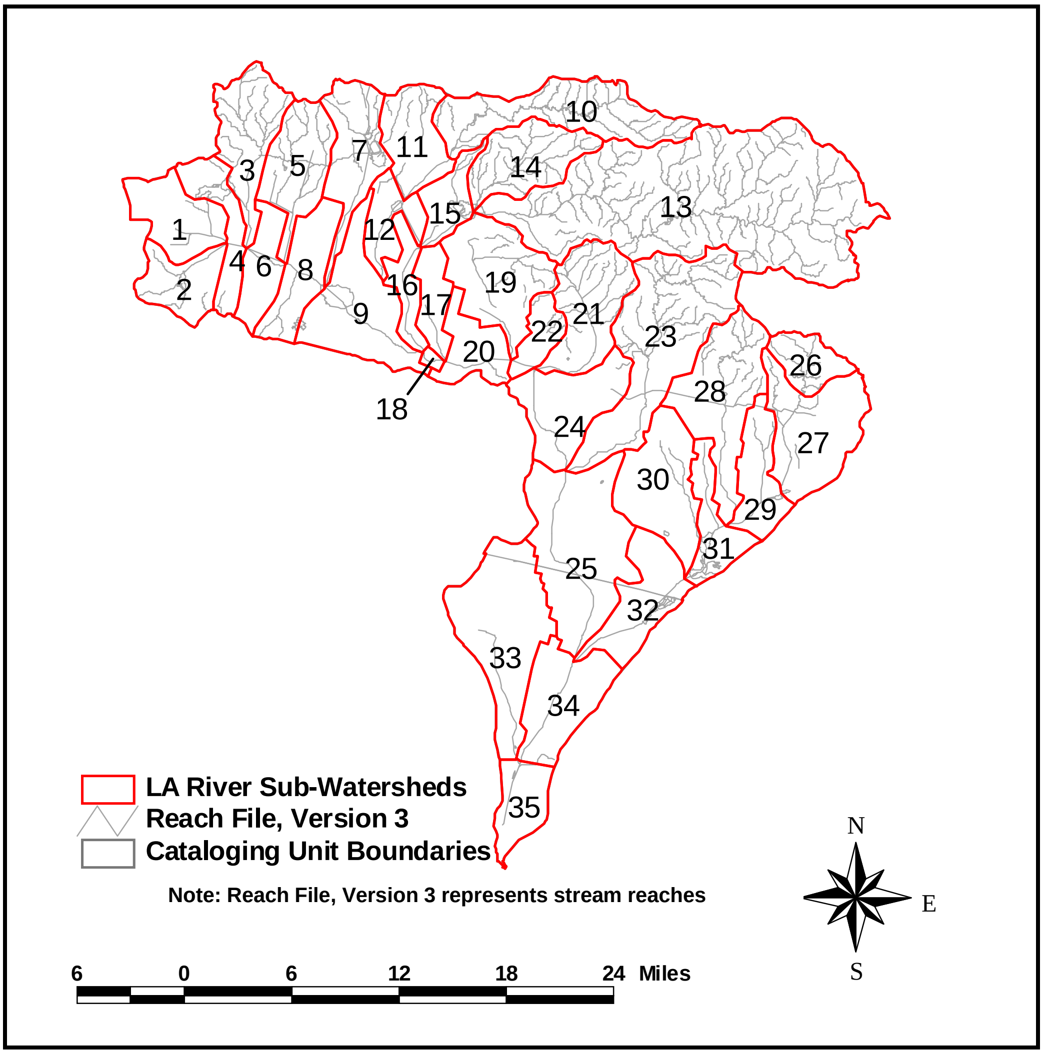


Figure D-2. Subwatershed Delineation for the Los Angeles River Watershed

For the San Gabriel River LSPC model, watershed segmentation was primarily based on the stream networks and topographic variability, and secondarily on the locations of flow and water quality monitoring stations, consistency of hydrologic factors, land use consistency, and existing watershed boundaries (based on CALWTR 2.2 watershed boundaries and municipal storm sewer-sheds). The San Gabriel River watershed was divided into 139 subwatersheds for appropriate hydrologic connectivity and representation (Figure D-3).

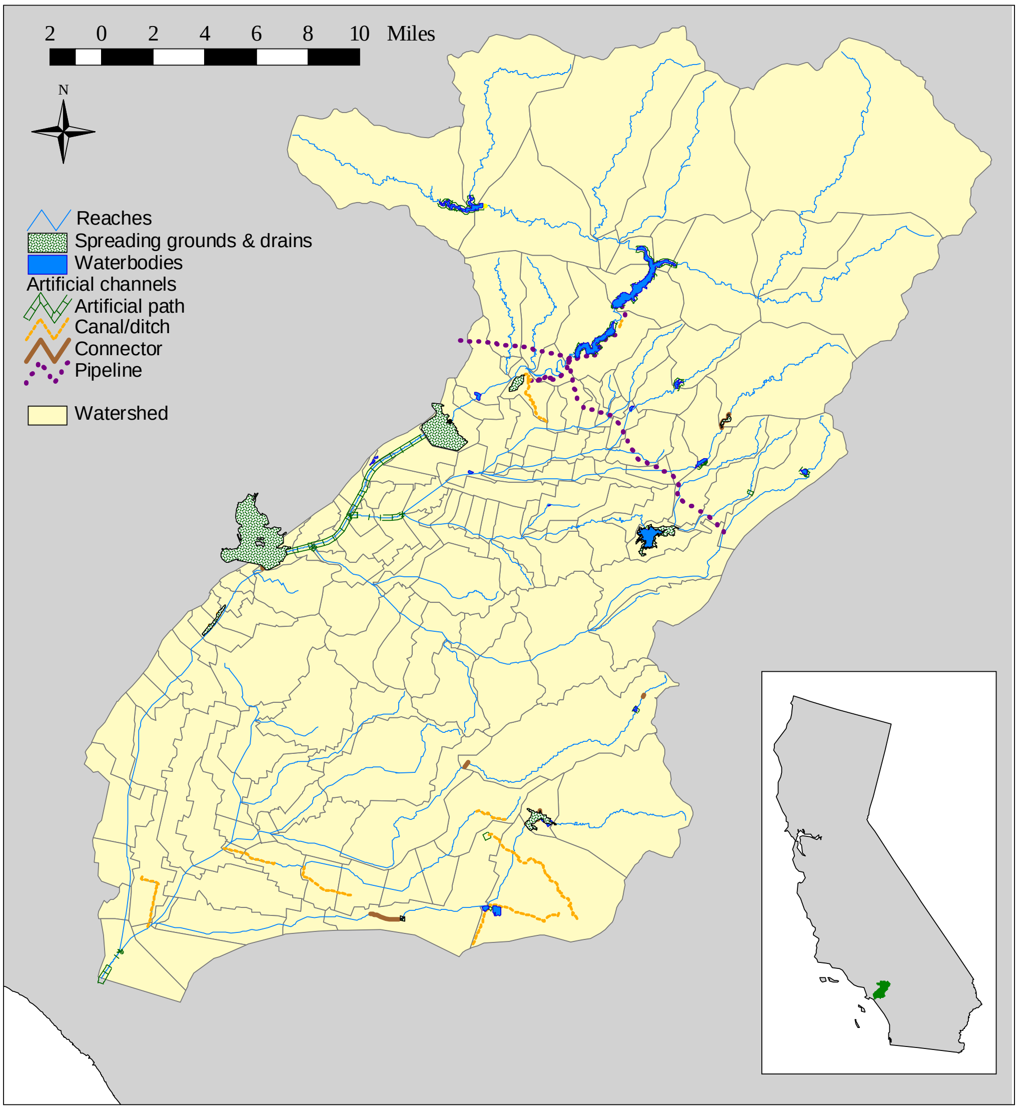


Figure D-3. Subwatershed Delineation for the San Gabriel River Watershed

D.2.3 METEOROLOGICAL DATA

Meteorological data are a critical component of the watershed model. LSPC requires appropriate representation of precipitation and potential evapotranspiration. In general, hourly precipitation (or finer resolution) data are recommended for nonpoint source modeling. Therefore, only weather stations with hourly-recorded data were considered in the precipitation data selection process (note: stations with daily evapotranspiration data were also used and the data were disaggregated to hourly, as describe below). Rainfall-runoff processes for each subwatershed were driven by precipitation data from the most representative station. These data provide necessary input to LSPC algorithms for hydrologic and water quality representation.

Precipitation data available from the National Climatic Data Center (NCDC) were reviewed based on geographic location, period of record, and missing data to determine the most appropriate meteorological stations. Ultimately, hourly rainfall data were obtained from 11 weather stations located in and around the Los Angeles River watershed (see Table D-1 and Figure D-4).

Long-term hourly wind speed, cloud cover, temperature, and dew point data were available for the Los Angeles International Airport (WBAN #23174). These data were obtained from NCDC for the characterization of meteorology of the modeled watersheds. Using these data, hourly potential evapotranspiration was calculated for the Los Angeles River LSPC model.

Table D-1. Precipitation and Meteorological Stations Used in the Los Angeles River LSPC Watershed Model

Station #	Description	Elevation (ft)	Latitude	Longitude
CA1194	BURBANK VALLEY PUMP PLA	655	34.183	-118.333
CA1682	CHATSWORTH RESERVOIR	910	34.225	-118.618
CA3751	HANSEN DAM	1087	34.261	-118.385
CA5085	LONG BEACH AP	31	33.812	-118.146
CA5114	LOS ANGELES WSO ARPT	100	33.938	-118.406
CA5115	LOS ANGELES DOWNTOWN	185	34.028	-118.296
CA5637	MILL CREEK SUMMIT R S	4990	34.387	-118.075
CA7762	SAN FERNANDO PH 3	1250	34.317	-118.500
CA7926	SANTA FE DAM	425	34.113	-117.969
CA8092	SEPULVEDA DAM	680	34.166	-118.473
CA9666	WHITTIER NARROWS DAM	200	34.020	-118.086

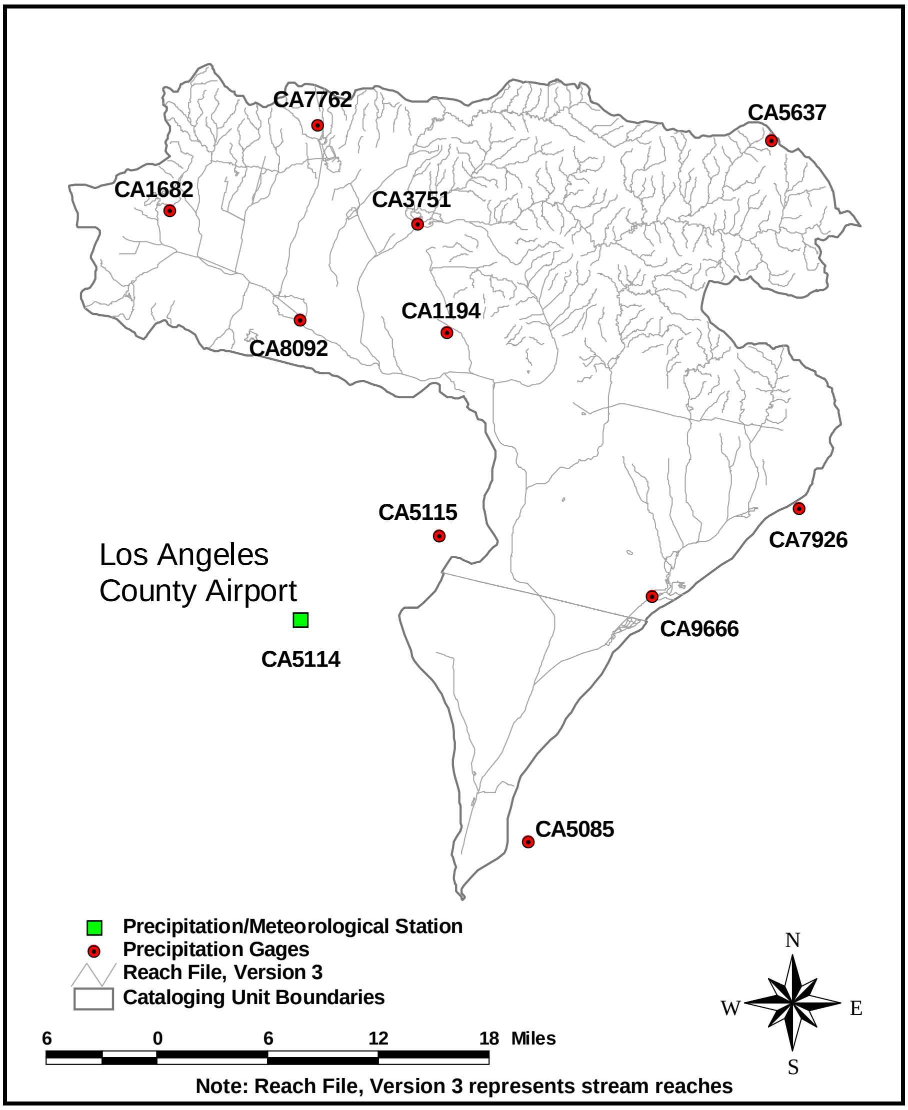


Figure D-4. Location of Precipitation and Meteorological Stations

For the San Gabriel model, hourly rainfall data were obtained from nine weather stations located in and around the watershed (Table D-2 and Figure D-5).

Table D-2. Precipitation Datasets Used for the San Gabriel River Model

Station #	Description	Elevation (ft)
CA1057	Brea Dam	275
CA1272	Cajon West Summit	4,780
CA1520	Carbon Canyon - Workman	1,180
CA5085	Long Beach	31

Station #	Description	Elevation (ft)
CA6473	Orange County Reservoir	660
CA7779	San Gabriel Dam	1,481
CA7926	Santa Fe Dam	425
CA8436	Spadra Lanterman Hospital	676
CA9666	Whittier Narrows Dam	200

Figure D-5. Location of Precipitation Stations for the San Gabriel LSPC Model

Because rainfall gages are not always in operation and accurately recording data, the resulting dataset may contain various intervals of accumulated, missing, or deleted data. Missing or deleted intervals are periods over which either the rainfall gage malfunctioned or the data records were somehow lost.

Accumulated intervals represent cumulative precipitation over several hours, but the exact hourly distribution of the data is unknown. To address the incomplete portions of each dataset, it was necessary to patch the rainfall data with information from nearby gages (see Tetra Tech, 2005 for more information).

Evapotranspiration (ET) data are also required by the LSPC model and were obtained for 10 weather stations from the Los Angeles County Department of Public Works (LADPW) and the California Irrigation Management Information System (CIMIS) (Table D-3 and Figure D-6). The six LADPW stations provided daily ET data while the four CIMIS stations recorded hourly ET. For model input, the daily values were averaged and then disaggregated to hourly increments using hourly data. Specifically, the average hourly percent of total ET from the CIMIS stations was applied to the daily LADPW data, resulting in hourly LADPW ET values. The hourly averages for all 10 stations were then averaged and incorporated into the model weather files.

Table D-3. Evapotranspiration Datasets Used for the San Gabriel River Model

Station #	Description	Elevation (ft)	Source
78	Brea Dam	730	CIMIS
82	Cajon West Summit	1,620	CIMIS
159	Carbon Canyon - Workman	595	CIMIS
174	Long Beach Airport	17	CIMIS
89B	San Dimas Dam	1,350	LADPW
96C	Puddingstone Dam	1,030	LADPW
223B	Big Dalton Dam	1,587	LADPW
334B	Cogswell Dam	2,300	LADPW
390B	Morris Dam	1,210	LADPW
425B	San Gabriel Dam	1,481	LADPW

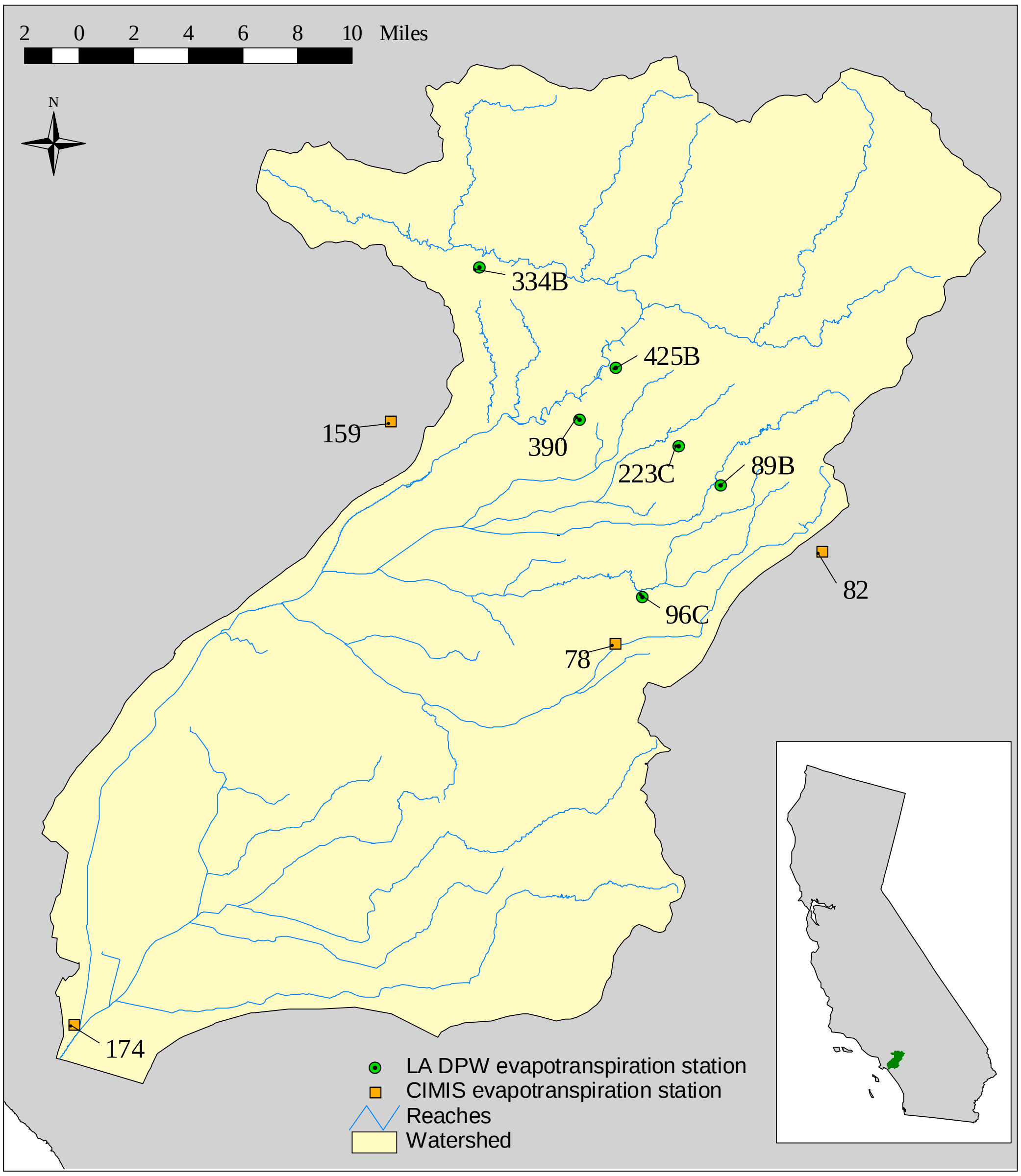


Figure D-6. Location of Evapotranspiration Stations

D.2.4 LAND USE REPRESENTATION

A watershed model requires a basis for distributing hydrologic and pollutant loading parameters. This is necessary to appropriately represent hydrologic variability throughout the basin, which is influenced by land surface and subsurface characteristics. It is also necessary to represent variability in pollutant loading, which is highly correlated to land practices. The basis for this distribution was provided by land use coverages developed for each watershed.

Two sources of land use data were used for the original Los Angeles River LSPC Model. The primary source of data was the County of Los Angeles Department of Public Works (LADPW) 1994 land use dataset that covers the county of Los Angeles. This dataset was supplemented with land use data from the

1993 USGS Multi-Resolution Land Characteristic (MRLC) dataset. For the original San Gabriel Model, the primary source of data was the Southern California Association of Governments (SCAG) 2000 land use dataset that covers the county of Los Angeles. This dataset was supplemented with land use data from the 1993 USGS Multi-Resolution Land Characteristic (MRLC) dataset. More recent land use data (SCAG, 2005) are currently available that did not exist during configuration of the original Los Angeles River and San Gabriel River LSPC models.

For development of these TMDLs, Tetra Tech verified the accuracy, to the extent practicable, of the land use coverages provided with the LSPC models relative to SCAG 2005 land use data. When discrepancies were observed, the land use categorization was updated. Current satellite imagery was used when necessary. Special attention was given to areas classified in the LSPC models as agriculture or strip mines due to the prevalence with which these areas are developed and their relatively high pollutant loading rates (details associated with these modifications are discussed in the lake-specific sections below).

Although the multiple categories in the land use coverages provide much detail regarding spatial representation of land practices in the watershed, such resolution is unnecessary for watershed modeling if many of the categories share hydrologic or pollutant loading characteristics. Therefore, many land use categories were grouped into similar classifications, resulting in a subset of 7 categories for the Los Angeles River model and 12 categories for the San Gabriel River model. Selection of the land use categories was based on the availability of monitoring data and literature values that could be used to characterize individual land use contributions and critical metals-contributing practices associated with different land uses. Land use areas by modeling subbasin are presented in the modeling reports (Tetra Tech, 2004 and 2005).

LSPC algorithms require that land use categories be divided into separate pervious and impervious land units for modeling. This division was made for the appropriate land uses to represent impervious and pervious areas separately. The division was based on typical impervious percentages associated with different land use types defined by LADPW (DePoto et al., 1991).

D.2.5 SOILS

Soil data for each watershed were obtained from the State Soil Geographic Data Base (STATSGO). There are four main Hydrologic Soil Groups (Groups A, B, C, and D). These groups, which are described below, range from soils with low runoff potential to soils with high runoff potential (USDA, 1986).

- Group A Soils have low runoff potential and high infiltration rates even when wet. They consist chiefly of sand and gravel and are well drained to excessively-drained.
- Group B Soils have moderate infiltration rates when wet and consist chiefly of soils that are moderately-deep to deep, moderately- to well-drained, and moderately coarse textured.
- Group C Soils have low infiltration rates when wet and consist chiefly of soils having a layer that impedes downward movement of water with moderately-fine to fine texture.
- Group D Soils have high runoff potential, very low infiltration rates and consist chiefly of clay soils. These soils also include urban areas.

The total area associated with each specific soil type was determined for each model subbasin. The representative soil group for each model subbasin was based on the dominant soil type found in that subwatershed.

D.2.6 REACH CHARACTERISTICS

Each delineated subbasin was represented with a single stream assumed to be a completely mixed, one-dimensional segment with a trapezoidal cross-section. The National Hydrography Dataset (NHD) stream reach network for USGS hydrologic unit 18070105 was used to determine the representative stream reach for each Los Angeles River subbasin. The NHD stream reach network for USGS hydrologic unit 18070106 was used to determine the representative stream reach for each San Gabriel River subbasin. Once the representative reach was identified, slopes were calculated based on DEM data and stream lengths measured from the original NHD stream coverage. In addition to stream slope and length, mean depths and channel widths are required to route flow and pollutants through the hydrologically connected subwatersheds. Mean stream depth and channel width were estimated for the Los Angeles River LSPC model from as-built channel construction drawings provided by the LADPW and were supplemented or verified through field reconnaissance. For the San Gabriel model, mean stream depth and channel width were estimated using regression curves that relate upstream drainage area to stream dimensions.

D.2.7 POINT SOURCE DISCHARGES

Both LSPC models incorporate flows and pollutant loads from major NPDES dischargers in the basin. However, none of these facilities impact the impaired lakes addressed by this TMDL.

D.2.8 HYDROLOGY REPRESENTATION

Watershed hydrology plays an important role in the determination of nonpoint source flow and ultimately nonpoint source loadings to a waterbody. The watershed model must appropriately represent the spatial and temporal variability of hydrological characteristics within a watershed. Key hydrological characteristics include interception storage capacities, infiltration properties, evaporation and transpiration rates, and watershed slope and roughness. LSPC’s algorithms are identical to those in the Hydrologic Simulation Program – FORTRAN (HSPF). The LSPC/HSPF modules used to represent watershed hydrology for TMDL development included PWATER (water budget simulation for pervious land units) and IWATER (water budget simulation for impervious land units). A detailed description of relevant hydrological algorithms is presented in the HSPF User’s Manual (Bicknell et al., 2001).

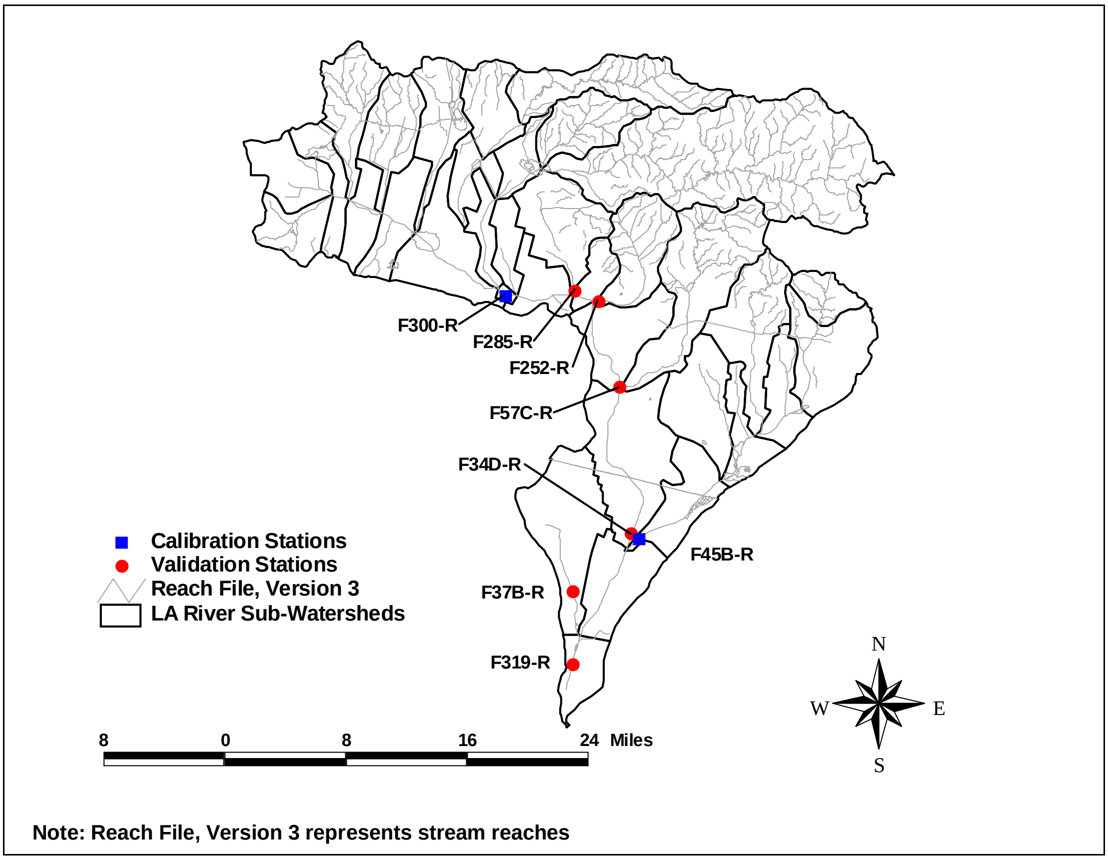
Key hydrologic parameters in the PWATER and IWATER modules are infiltration, groundwater flow, and overland flow. USDA’s STATSGO Soils Database served as a starting point for designation of infiltration and groundwater flow parameters. For parameter values not easily derived from these sources, documentation on past HSPF applications was accessed. Starting values were refined through the hydrologic calibration process (Tetra Tech, 2004; Tetra Tech, 2005).

D.2.9 FLOW DATA

Flow gaging stations representing relatively diverse hydrologic regions were used for calibration and validation of each LSPC model. Eight stations were selected for calibration of the Los Angeles River LSPC Model because they either had a robust historical record or they were in a strategic location (i.e., along a 303(d)-listed waterbody). The selected flow stations are maintained by the LADPW. Information about each flow station, including location and use in model calibration or validation, is presented in Table D-4 and illustrated in Figure D-7.

Table D-4. Calibration and Validation Stations used in the Los Angeles River LSPC Model

Number	Station Description	Latitude	Longitude	Comment
F45B-R	Rio Hondo above Stuart and Gray Road	33.946	-118.164	Calibration
F300-R	Los Angeles River at Tujunga Ave.	34.141	-118.379	Calibration
F285-R	Burbank Western Storm Drain at Riverside Dr.	34.161	-118.304	Validation
F37B-R	Compton Creek near Greenleaf Drive	33.882	-118.224	Validation
F252-R	Verdugo Wash at Estelle Avenue	34.156	-118.273	Validation
F57C-R	Los Angeles River above Arroyo Seco	34.082	-118.226	Validation
F34D-R	Los Angeles River below Firestone Blvd.	33.949	-118.174	Validation
F319-R	Los Angeles River below Wardlow River Rd.	33.815	-118.205	Validation



Note: Reach File, Version 3 represents stream reaches

Figure D-7. Location of Hydrology Calibration and Validation Stations for the Los Angeles River LSPC Model

For the San Gabriel Model, 12 flow gaging stations containing full or partial records of flow for the simulation period were identified. These flow stations are maintained by LADPW or the United Stated Geological Survey (USGS). Information about each flow station, including outflow subwatershed, the station identification number (which also indicates the responsible agency) and period(s) used for model calibration and validation, is presented in Table D-5, and their locations are illustrated in Figure D-8.

Table D-5. Flow Data Used for San Gabriel River LSPC Model Calibration and Validation

Gaging Station	Station Description	Outflow Subwatershed	Calibration Dates	Validation Dates
USGS 11089500	Fullerton Creek	56	7/01/94 – 9/30/97	10/01/97 – 9/30/02
USGS 11088500	Brea Creek	59	7/01/94 – 9/30/97	10/01/97 – 9/30/02
LADPW F304-R[a]	Walnut Creek	83	1/01/98 – 12/30/02	none
LADPW F274B-R	Dalton Wash	99	10/01/92 – 9/30/95	none
LADPW F312B-R[a]	San Jose Channel	67	10/01/92 – 9/30/94	1/01/98 – 9/30/02
USGS 11087020	San Gabriel River	18	7/01/94 – 9/30/97	10/01/97 – 9/30/02
LADPW F262C-R[a]	San Gabriel River	8	1/01/98 – 12/30/02	none
LADPW F42B-R[a]	San Gabriel River	2	1/01/98 – 12/30/02	none
USGS 11085000	San Gabriel River	24	7/01/94 – 9/30/97	10/01/97 – 9/30/02
LADPW F190-R	San Gabriel River	26	7/01/94 – 9/30/95	none
LADPW U8-R	San Gabriel River	29	7/01/94 – 9/30/95	none
LADPW F354-R[a]	Coyote Creek	37	12/01/01 – 12/30/02	none

[a] There are various periods of missing data from this gage station.

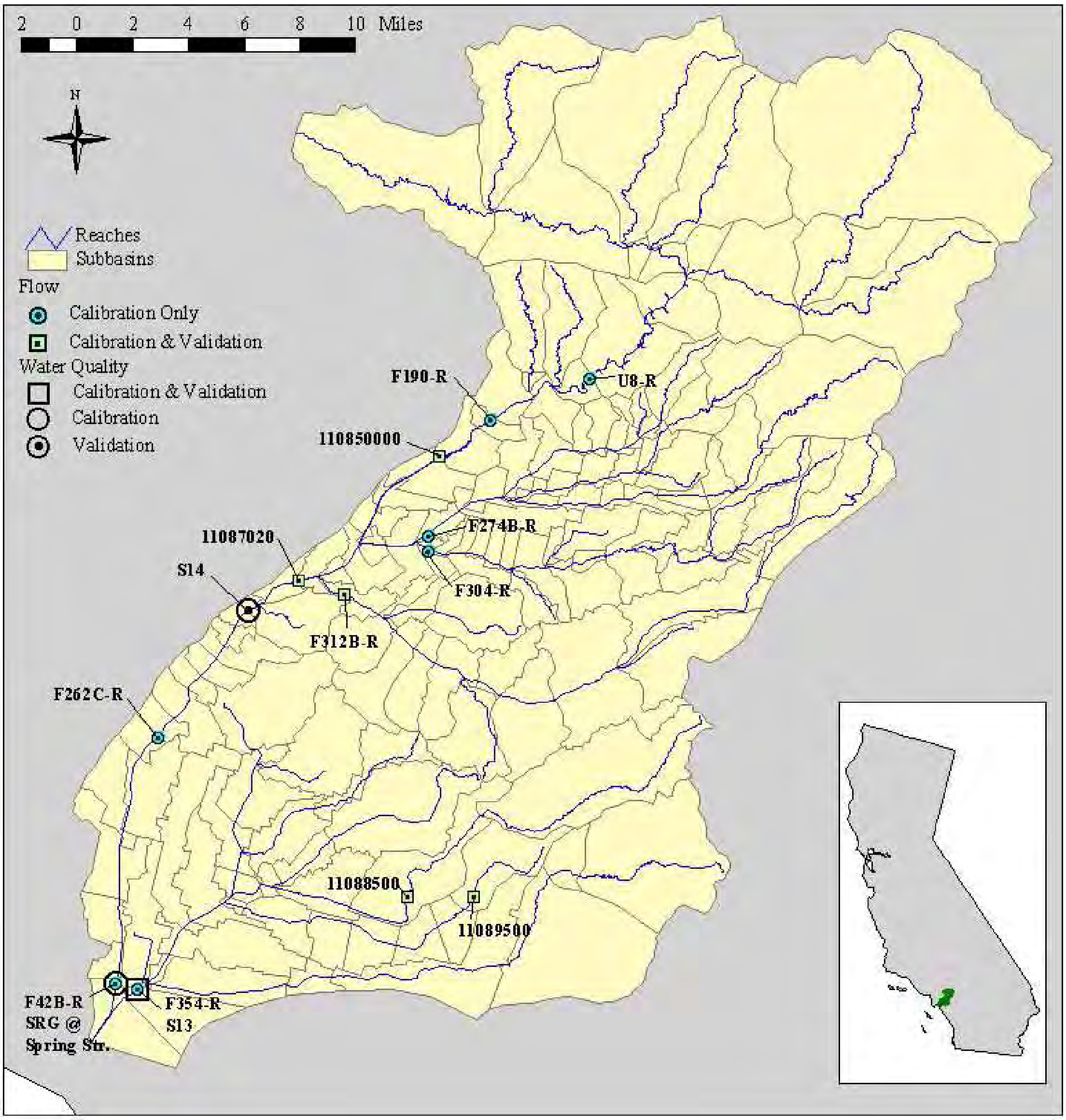


Figure D-8. Locations of Monitoring Stations Used for Model Calibration and Validation of the San Gabriel LSPC Model

The process and results of the flow calibration for each LSPC model is discussed in detail in the respective modeling reports (Tetra Tech, 2004 and 2005).

D.2.10 MODEL OUTPUT

Both of the LSPC models were used to estimate the average monthly runoff depths from land uses present in the watersheds of the impaired waterbodies addressed by this TMDL. Runoff depths by land use were extracted from the LSPC modeling subbasin that contains the watershed of each impaired lake. Table D-6 lists the impaired lakes and associated LSPC model, modeling subbasin, weather station, and dominant hydrologic soil group used to drive the runoff simulation.

Table D-6. Impaired Waterbodies and Associated LSPC Modeling Subbasins and Meteorological Stations

Impaired Lake/Reservoir	LSPC Model	Subbasin #	Meteorological Station	Dominant Soil Group
Peck Road Park	Los Angeles River	27	CA7926	C
Lincoln Park	Los Angeles River	25	CA5115	D
Echo Park	Los Angeles River	25	CA5115	D
Calabasas	Los Angeles River	2	CA1682	D
El Dorado Park	San Gabriel River	46	CA5085	NL[1]
Legg	Los Angeles River	31	CA9666	D
Puddingstone	San Gabriel River	93, 94, 95	CA8436	NL[1]
Santa Fe Dam Park	San Gabriel River	24	CA7926	NL[1]
Sherwood	Los Angeles River	2	CA1682	D

[1] NL: dominant soil hydrologic group is not listed by subbasin in the San Gabriel River Basin LSPC Modeling Report.

Average monthly runoff depths by land use for each impaired waterbody are listed in the respective lake sections of this appendix.

D.3 Event Mean Concentrations

Event mean concentrations (EMCs) represent flow-weighted average concentrations delivered during storm events. Because the LSPC models have not been calibrated to generate loading estimates of key parameters of concern, pollutant EMCs applied to calibrated flow volumes for representative land uses are the best approximation of wet weather loading to the impaired lakes at this time. For these TMDLs, EMCs for nutrients are used to estimate loading associated with runoff volumes. For the mercury and OC Pesticides and PCBs TMDLs, sediment EMCs are used along with simulated runoff volumes and watershed specific water column and sediment concentrations of pollutants to estimate wet weather loading.

For sediment and sediment-associated parameters, the observed instream concentration can be significantly affected by channel scour and deposition processes. The LSPC models are not fully calibrated for such channel processes, which tend to be location-specific. The magnitude of this component for the TMDL watersheds is significantly reduced by the fact that many of the channels are either piped or hardened with concrete and thus not subject to channel degradation. There are, however, portions of the channel network that are not hardened, and even within concrete-lined channels it is expected that there were cycles of deposition and scour of sediment.

Because sufficient data are not available to calibrate detailed models of sediment scour and deposition in reaches, the TMDL analysis is based on an assumption of long-term dynamic equilibrium in the stream network. This approach makes the approximate assumption that the amount of sediment moving through the streams is equivalent (as a long-term average) to the rate of sediment loading to those streams, as estimated from the reported EMCs. Such an assumption was clearly not valid during the earlier period of land use change, construction, and rapid development in the study watersheds, but is believed to provide a reasonable approximation for current conditions.

EMC data for several monitoring years were provided by SCCWRP (Ackerman and Schiff, 2003) and Los Angeles County Department of Public Works (LACDPW) (LACDPW, 2000) for various land uses. Though an EMC may be the same for two seemingly different land uses, loading rates will vary due to differences in runoff volumes. Table D-7 summarizes the EMCs for modeled land uses in the Los Angeles River and San Gabriel River LSPC models.

Table D-7. EMCs for Modeled Land Uses in the Los Angeles and San Gabriel LSPC Models

Los Angeles Model	San Gabriel Model	Sediment (mg/L)	Nitrogen (mg/L)	Phosphorus (mg/L)
Agriculture	Cropland and Pasture	1,520	8.6	0.56
Commercial Other Urban	Commercial and Services Other Urban or Built Up	56.5	4.41	0.67
Industrial	Industrial Transportation, Communication, Utilities	84.7	4.55	0.58
Open	Evergreen Forest Land Herbaceous Rangeland Mixed Rangeland Shrub and Brush Rangeland	28.83	3.2	0.11
Residential	Residential Transitional Areas	55.2	4.51	0.73
NA	Strip Mines	1,520	4.55	0.58

The 12 land uses simulated by the San Gabriel River Basin LSPC Model were aggregated to modeled land use categories as presented in Table D-8. The table also lists the impervious fractions of each urban land use simulated by the model (rural land uses are assumed 100 percent pervious). The impervious area is the major determinant of total runoff volume for urban land uses. At the basin-wide scale, areas classified as "other urban or built-up" were simulated as commercial areas with 65 percent imperviousness. Comparison of the LSPC land use coverage to SCAG 2005 data and recent satellite imagery indicate that areas adjacent to the impaired waterbodies classified as "other urban or built-up" are actually parkland. To predict runoff volumes and sediment loading from these areas, model output for the pervious fraction of commercial areas was assumed representative of parkland.

Table D-8. Aggregation of Land Use Classes in the LSPC Model

Original Land Use	Modeled Land Use
Commercial and services	Commercial (80 percent impervious)
Cropland and pasture	Cropland
Evergreen forest land	Forest
Herbaceous rangeland	Pasture
Industrial	Industrial (80 percent impervious)
Mixed rangeland	Pasture
Other urban or built-up[1]	Commercial (65 percent impervious)
Residential	Residential (19 percent impervious)
Shrub & brush rangeland	Pasture
Trans, comm, util	Transportation (80 percent impervious)
Transitional areas	Residential (10 percent impervious)

[1] Other urban or built-up areas surrounding impaired waterbodies are parkland and are simulated as commercial with zero percent imperviousness.

Runoff depths for the simulated land uses vary by LSPC modeling subbasin. The subsequent sections of this report summarize the monthly average runoff depths for land uses draining to each of the impaired lakes. EMCs are applied to runoff depths for a corresponding area to estimate pollutant loading associated with a particular area. For example, the average runoff depth in January for agricultural lands in the Peck Road Park Lake watershed is 0.5361 inches (Table D-12). There are 4.19 acres of agriculture present in the Western Subwatershed (Table D-9).

The following calculation estimates the total nitrogen load delivered during January from this area:

$$\frac{8.6mg-N}{L}\cdot\frac{0.536in}{mo}\cdot\frac{1ft}{12in}\cdot 4.19ac\cdot\frac{28.32L}{ft^3}\cdot\frac{43{,}560\,ft^2}{ac}\cdot\frac{1g}{1{,}000mg}\cdot\frac{1lb}{453.6g}=4.38lb-N$$

For the waterbodies impaired by mercury and OC Pesticides and PCBs, watershed specific monitoring data were available to estimate loading to each lake. The dissolved portion of the load can be represented as flow multiplied by the observed water column concentration, and the sediment-associated portion of the load can be represented as average sediment movement times the observed concentration in stream sediment.

For example, the near lake undeveloped subwatershed draining to Lake Sherwood is comprised of 197 acres of open space (Table D-64). The monthly average runoff depth for open space in January is 0.3808 inches (Table D-66), and the sediment EMC is 28.83 mg/L (Table D-7). The winter season water column and sediment-associated concentrations of total mercury are 2.96 ng/L (Table D-68) and 129 µg/kg (Table D-69), respectively. The total mercury delivered from runoff generated in January from this area is

$$197ac \cdot \frac{0.3808in}{mo} \cdot \frac{1ft}{12in} \cdot \frac{43{,}560\,ft^2}{ac} \cdot \frac{28.32L}{ft^3} \cdot \frac{2.96ng-Hg}{L} \cdot \frac{1g}{1 \cdot 10^9\,ng} = 0.0228g-Hg$$

The total mercury (Hg) associated with the delivered sediment (sed) in January is

$$197ac \cdot \frac{0.3808in}{mo} \cdot \frac{1ft}{12in} \cdot \frac{43{,}560\,ft^2}{ac} \cdot \frac{28.32L}{ft^3} \cdot \frac{28.83mg-sed}{L} \cdot \frac{1g-sed}{1{,}000mg-sed} \cdot \frac{129ng-Hg}{g-sed} \cdot \frac{1g-Hg}{1 \cdot 10^9\,ng-Hg} = 0.0287g-Hg$$

(This page left intentionally blank.)

D.4 Peck Road Park Lake

Peck Road Park Lake is located in the Los Angeles River Basin. However, the LACDPW diverts flows from the San Gabriel River to Peck Road Park Lake via the Santa Fe Diversion Channel.

Impairments of this lake include low dissolved oxygen/organic enrichment, eutrophication (originally on the consent decree, but currently delisted), odor, lead, chlordane, DDT, dieldrin, PCBs, and trash. Output from the Los Angeles River LSPC model coupled with regional pollutant event mean concentrations have been used to estimate loads from upland areas of OC Pesticides and PCBs and nutrients, which may be contributing to the low dissolved oxygen/organic enrichment, eutrophication, and odor impairments. Loads from the diversion are estimated from measured flow volumes and area-weighted event mean concentrations for the land use classes upstream of the diversion channel.

Three subwatersheds comprise the drainage area to Peck Road Park Lake. The subwatershed draining the western part of the watershed via Santa Anita Wash is 12,686 acres, and the eastern subwatershed draining to Saw Pit Wash is 10,557 acres. There is an inwardly draining mining operation in the southern part of the eastern watershed that has been removed from the loading analysis. The area surrounding the lake is 321 acres. Each subwatershed drains to a storm sewer system so all allocations for the TMDLs are wasteload allocations (except for the trash TMDL which also has a load allocation).

Figure D-9 shows the MS4 stormwater permittees in the Peck Road Park Lake watershed. The western subwatershed is comprised of the county of Los Angeles, Sierra Madre, Arcadia, Monrovia, Angeles National Forest, and Caltrans areas. The eastern subwatershed is comprised of the county of Los Angeles, Monrovia, Duarte, Bradbury, Arcadia, Irwindale, Angeles National Forest, and Caltrans areas. The county of Los Angeles, Monrovia, Irwindale, Arcadia, and El Monte comprise the drainage around the lake. The park area is comprised of 152 acres adjacent to the lake.

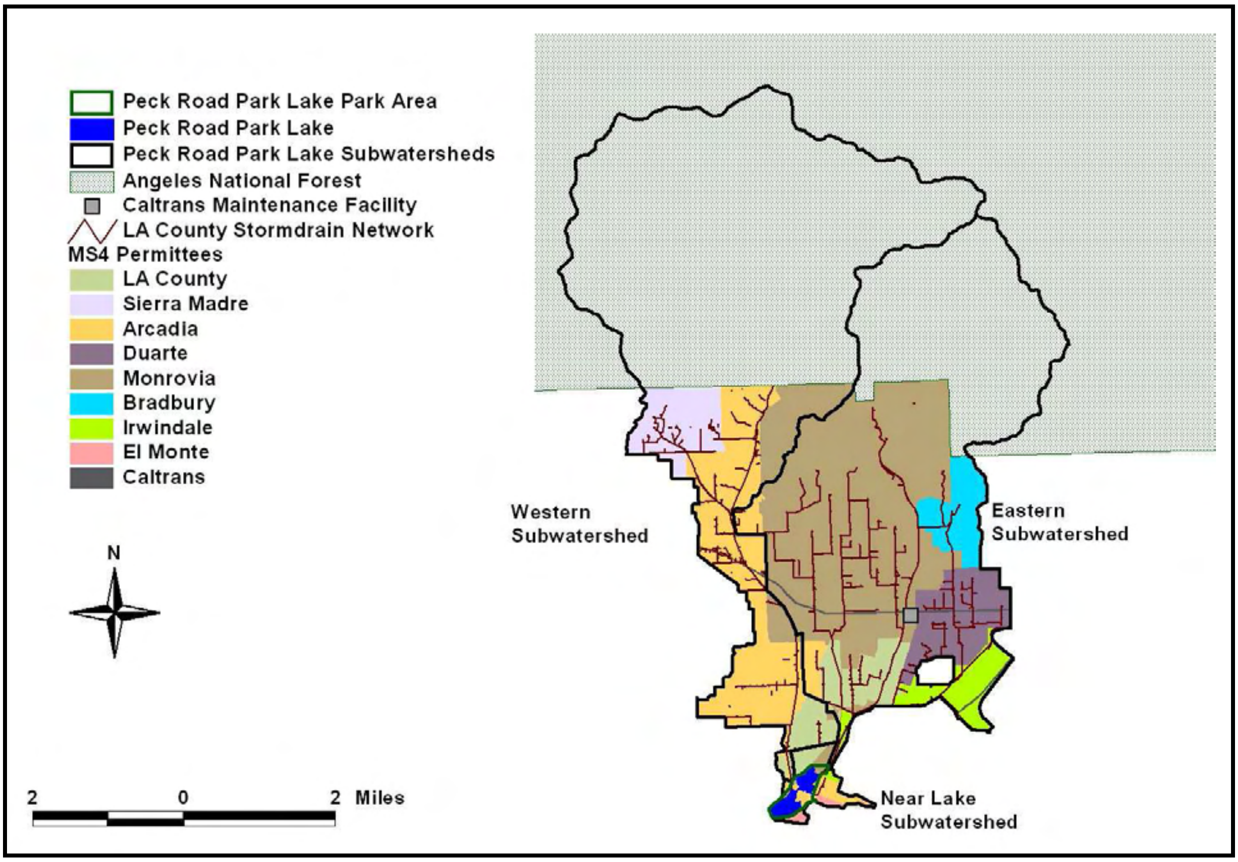


Figure D-9. MS4 Permittees and the County of Los Angeles Storm Drain Network in the Peck Road Park Lake Subwatersheds

Land uses identified in the Los Angeles River LSPC model are shown in Figure D-10. Upon review of the SCAG 2005 database, as well as current satellite imagery, it was evident that a portion of the areas classified by the LSPC model as agriculture were inaccurate. Land use classifications were changed to accurately reflect the conditions identified in the more recent data. Approximately 82 acres classified by LSPC as agriculture corresponded to orchards, vineyards, and horse farms and were not altered. However, approximately 27 acres of agriculture was reclassified as open space and 28 acres were reclassified as residential. Areas classified as industrial or commercial in the Angeles National Forest were also inaccurate and were reclassified as open. Inaccuracies in land use assignment were corrected for each subwatershed and jurisdiction to reflect the more recent SCAG 2005 dataset and current satellite imagery. All areas within the Caltrans jurisdiction were simulated as industrial since the Los Angeles River Basin LSPC model lumped transportation uses into the industrial category. Table D-9, Table D-10, and Table D-11 summarize the post-processed land use areas used to estimate pollutant loading from upland areas draining to Peck Road Park Lake.

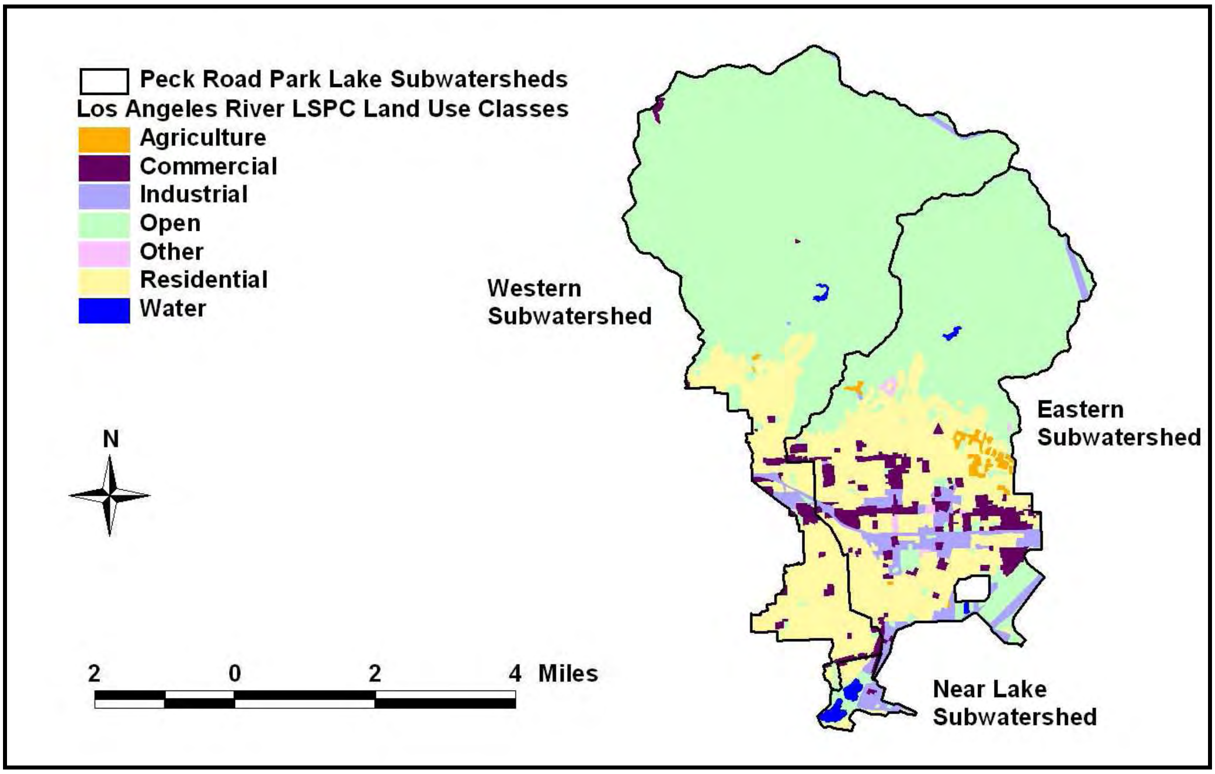


Figure D-10. LSPC Land Use Classes for the Peck Road Park Lake Subwatersheds

Table D-9. Land Use Areas (ac) Draining from the Western Subwatershed of Peck Road Park Lake

Land Use	County of Los Angeles	Sierra Madre	Arcadia	Monrovia	Caltrans	Angeles National Forest	Total
Agriculture	0	4.19	0	0	0	0	**4.19**
Commercial	34.8	2.62	124	13.0	0	0	**175**
Industrial	0	0	70.4	0.319	16.9	0	**87.6**
Open	3.50	377	319	483	0	9,104	**10,286**
Other Urban	0	0	0.053	0	0	0	**0.053**
Residential	207	296	1,516	114	0	0	**2,133**
Total	**245**	**679**	**2,030**	**611**	**16.9**	**9,104**	**12,686**

Table D-10. Land Use Areas (ac) Draining from the Eastern Subwatershed of Peck Road Park Lake

Land Use	County of Los Angeles	Monrovia	Duarte	Bradbury	Arcadia	Irwindale	Caltrans	Angeles National Forest	Total
Agriculture	0	0	0	78.1	0	0	0	0	**78.1**
Commercial	24.8	430	232	0	33.9	12.7	0	0	**733**
Industrial	1.27	407	107	0	0	180	78.4	0	**774**
Open	5.29	1,419	53.5	229	16.0	274	0	3,511	**5,508**
Other Urban	0	51.0	1.74	2.90	1.71	0	0	0	**57.3**
Residential	467	2,149	424	193	158	15.5	0	0	**3,406**
Total	**499**	**4,456**	**818**	**503**	**209**	**483**	**78.4**	**3,511**	**10,557**

Table D-11. Land Use Areas (ac) Draining from the Near Lake Subwatershed of Peck Road Park Lake

Land Use	County of Los Angeles	Monrovia	Irwindale	Arcadia	El Monte	Total
Agriculture	0	0	0	0	0	**0**
Commercial	7.10	7.90	0	3.86	0	**18.9**
Industrial	0.0003	14.4	13.9	69.7	10.2	**108**
Open	0.233	24.6	0.187	61.6	0.984	**87.5**
Other Urban	0	0	0	0	0	**0**
Residential	60.4	1.30	0	4.18	40.9	**107**
Total	**67.7**	**48.1**	**14.1**	**139**	**52.1**	**321**

The land use composition upstream of the San Gabriel River at the diversion to Peck Road Park Lake is primarily rangeland (56 percent) and forest (40 percent). The remaining 4 percent is comprised of other types of open areas and urban development. To estimate the pollutant concentrations associated with the diverted flows, EMCs (Section D.3) were area-weighted based on the land use composition upstream of the diversion.

D.4.1 RUNOFF AND DIVERTED FLOWS

LSPC-predicted runoff from the Peck Road Park Lake subwatersheds is primarily driven by the land use and soil characteristics of the drainage area and the nearest meteorological station represented in the model. Figure D-11 shows the simulated annual rainfall for the Peck Road Park Lake subwatersheds. The annual average rainfall is 19.1 inches.

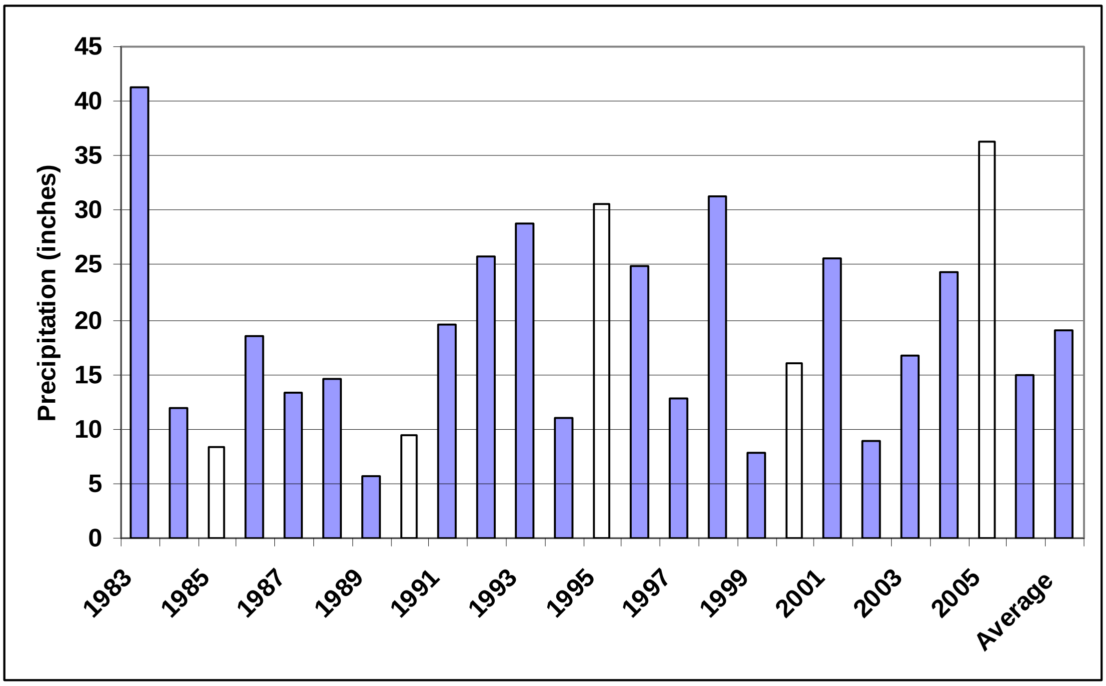


Figure D-11. Annual Rainfall for the Peck Road Park Lake Subwatersheds

The simulated monthly average runoff depths for land uses in the Peck Road Park Lake subwatersheds are shown in Table D-12.

Table D-12. Monthly Average Runoff Depths (inches/month) for Land Uses in the Peck Road Park Lake Subwatersheds, 1983 - 2006

Month	Agriculture	Commercial	Industrial	Open	Other Urban	Residential
January	0.5361	3.0291	2.7414	0.1966	2.0223	1.9645
February	0.8942	3.9665	3.6105	0.4150	2.7206	2.6491
March	0.5614	2.4735	2.2559	0.2416	1.7120	1.6683
April	0.1153	0.8499	0.7608	0.0548	0.5381	0.5202
May	0.0531	0.2477	0.2250	0.0216	0.1682	0.1636
June	0.0097	0.1020	0.0904	0.0053	0.0614	0.0591
July	0.0010	0.0090	0.0080	0.0006	0.0054	0.0052
August	0.0047	0.0632	0.0558	0.0024	0.0373	0.0358
September	0.0163	0.2219	0.1959	0.0080	0.1312	0.1260
October	0.0407	0.5706	0.5037	0.0202	0.3364	0.3230
November	0.0684	0.9569	0.8447	0.0339	0.5641	0.5416
December	0.1226	1.5882	1.4051	0.0575	0.9475	0.9108

The LACDPW provided Tetra Tech with mean daily flows measured over the past 15 years (October 1994 through May 2009) in the diversion channel that directs flow from the San Gabriel River to Peck Road Park Lake. The average monthly flows from this diversion are summarized in Table D-13.

Table D-13. Average Monthly Flow Volumes Diverted to Peck Road Park Lake

Month	Diverted Flow (ac-ft)
January	223
February	229
March	981
April	717
May	1,028
June	2,039
July	1,134
August	343
September	854
October	718
November	76.8
December	395
Total	**8,737**

Figure D-12 summarizes the monthly average runoff and diversion volumes delivered to Peck Road Park Lake. The total annual volume delivered to the lake is 16,529 ac-ft, and approximately half the flow is from the San Gabriel diversion. Flows during the months May through October are primarily from the diversion channel.

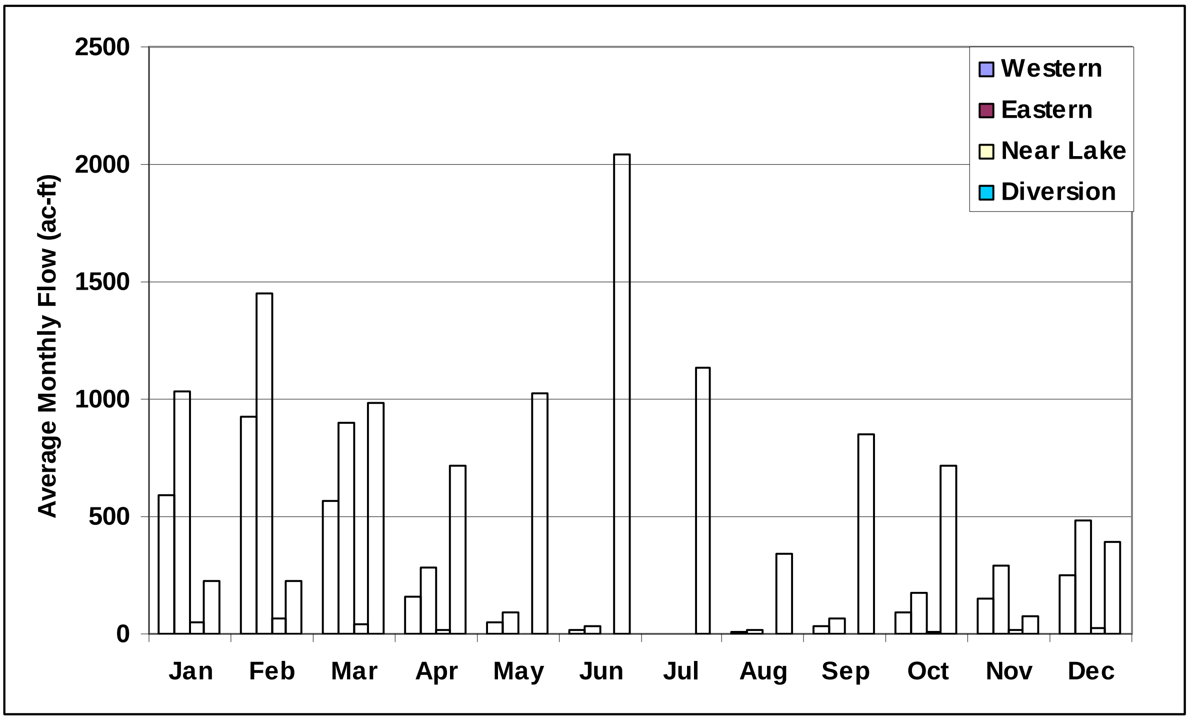


Figure D-12. Monthly Average Runoff Volumes to Peck Road Park Lake

D.4.2 SEDIMENT LOADS

Sediment loads are calculated from simulated volumes and suspended sediment event mean concentrations for each modeled land use (Section D.3). Table D-14 summarizes the average annual sediment loads for each jurisdiction by subwatershed. Sediment loads estimated for the diversion are included as well. See example calculations in Section D.3.

Table D-14. Average Annual Sediment Loads to Peck Road Park Lake

Subwatershed	Jurisdiction	Sediment (tons/yr)
Eastern	Arcadia	12.1
Eastern	Bradbury	44.4
Eastern	Caltrans	9.55
Eastern	Duarte	58.0
Eastern	Irwindale	24.9
Eastern	County of Los Angeles	28.6
Eastern	Monrovia	217
Eastern	Angeles National Forest	12.1
Near Lake	Arcadia	9.29
Near Lake	El Monte	3.55
Near Lake	Irwindale	1.70
Near Lake	County of Los Angeles	4.03

Subwatershed	Jurisdiction	Sediment (tons/yr)
Near Lake	Monrovia	2.62
Western	Arcadia	106
Western	Caltrans	2.06
Western	County of Los Angeles	14.7
Western	Monrovia	9.27
Western	Sierra Madre	19.9
Western	Angeles National Forest	31.4
Diversion		379
Total		**990**

D.4.3 NUTRIENT LOADS

Nutrient loads are estimated from simulated volumes and event mean concentration data collected by SCCWRP and the county of Los Angeles (Section D.3). Table D-15 summarizes the total nitrogen and total phosphorus loads delivered to Peck Road Park Lake from each jurisdiction and subwatershed or from the diversion. See example calculations in Section D.3.

The loads presented in the table are existing loads, not allocated loads.

Table D-15. Average Annual Nutrient Loads to Peck Road Park Lake

Subwatershed	Jurisdiction	Nitrogen (lb/yr)	Phosphorus (lb/yr)
Eastern	Arcadia	1,951	309
Eastern	Bradbury	2,337	320
Eastern	Caltrans	1,027	131
Eastern	Duarte	8,606	1,307
Eastern	Irwindale	2,891	358
Eastern	County of Los Angeles	4,653	749
Eastern	Monrovia	32,627	4,894
Eastern	Angeles National Forest	2,692	92.5
Near Lake	Arcadia	1,053	132
Near Lake	El Monte	510	77.8
Near Lake	Irwindale	183	23.3
Near Lake	County of Los Angeles	653	105
Near Lake	Monrovia	330	43.4
Western	Arcadia	16,812	2,641
Western	Caltrans	221	28.2
Western	County of Los Angeles	2,386	381
Western	Monrovia	1,601	210
Western	Sierra Madre	3,056	456
Western	Angeles National Forest	6,981	240

Subwatershed	Jurisdiction	Nitrogen (lb/yr)	Phosphorus (lb/yr)
Diversion		76,970	2,960
Total		**167,539**	**15,458**

D.4.4 ORGANOCHLORINE PESTICIDES AND PCBS LOADS

The existing loading rates from upland areas for OC Pesticides and PCBs are estimated for each pollutant of concern using monitoring data collected by USEPA, the Regional Board, and UCLA, between 2008 and 2009. Only data from sites representing inflows are used; these sites include locations in an inflow, or in the lake near an inflow. Inflows considered for wet weather loading were tributaries, drainage paths, and channels. For Peck Road Park Lake, this included PRPL-6, PRPL-7, PRPL-12 and PRPL-13 (Figure D-13).

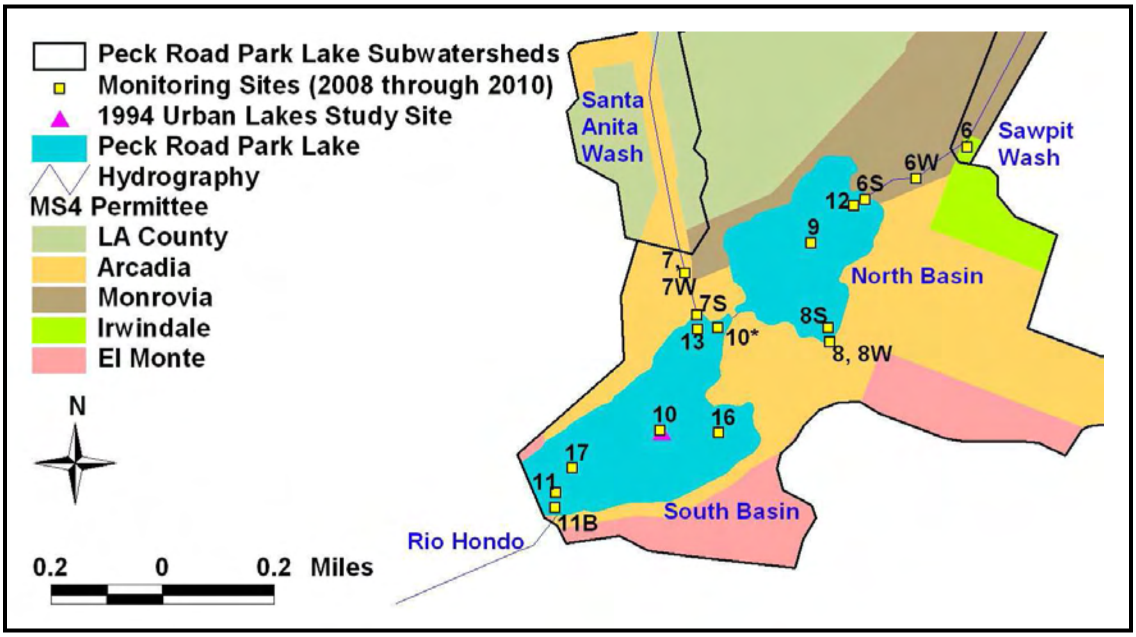


Figure D-13. Monitoring Stations at Peck Road Park Lake

The OC Pesticides and PCBs of concern are not currently in use and are more likely to have been historically loaded to the lake sediments; therefore, current tributary loading is likely to be small. The OC Pesticides and PCBs are hydrophobic and the majority of the pollutant mass in wet weather loads will be associated with the sediment. The measured levels of OC Pesticides and PCBs in inflow sediments were the only data that could be used to quantify current inflow loads because nearly all of the water column, porewater, suspended sediment, and suspended sediment in porewater samples did not yield reportable results. For OC Pesticides and PCBs where some of the samples had detectable quantities of a pollutant, the average inflow concentration was calculated assuming samples analyzed below detection limits were equal to one-half the detection limit. For all of the sediment samples, dieldrin was below detection and reporting levels . Instead, an upper-bound analysis was performed using the detection limit as the incoming concentration associated with the sediment. The average concentration of total chlordane in sediments associated with inputs was 3.15 µg/kg dry weight and the average level of PCBs was 15.38 µg/kg dry weight. The average concentration of DDT was 5.57 µg/kg dry weight. The inflow sediment data are summarized in Table D-16 and all data collected in the watershed are discussed in detail in Appendix G (Monitoring Data).

Table D-16. Summary of Sediment Data near Inflow Locations at Peck Road Park Lake

Parameter	Number of Samples	Number of Samples Above Detection Limits[1]	Average Concentration (µg/kg dry weight)	Detection Limit (µg/kg dry weight)
Chlordane	6	2	3.15	0.34-1
DDT	6	3	5.57	0.69-1.18
Dieldrin	6	0	(0.91)[2]	0.69-1.18
Total PCBs	6	5	15.38	0.34-1

[1] Non-detect samples were included in reported averages as one-half of the detection limit.

[2] All sample results were below detection limits. An upper-bound analysis was performed using the highest reported detection limit for dieldrin.

These input sediment concentrations were applied to the calculated sediment loads (Section D.4.2) to estimate sediment-associated OC Pesticides and PCBs loads entering the lake. Specifically, to determine sediment loading of OC Pesticides and PCBs, the sediment EMCs and LSPC predicted runoff volumes were used to calculate sediment loads (Table D-17), and the sediment OC Pesticides and PCBs concentrations from monitoring data (Table D-16) were then applied to these sediment loads.

Sediment loads and subsequently calculated OC Pesticides and PCBs loads were determined for each jurisdiction based on the land use types and areas within each subwatershed. The jurisdictional areas are presented for the three Peck Road Park Lake subwatersheds in Table D-17 along with the predicted sediment loads for each land use. Dissolved concentrations in inflows are assumed insignificant.

Table D-17. Annual Sediment Load to Peck Road Park Lake

Subwatershed	Jurisdiction	Area (ac)	Annual Sediment Load (tons/yr)	Percent of Total Load
Eastern	Arcadia	209	12.1	1.22%
Eastern	Bradbury	503	44.4	4.48%
Eastern	Caltrans	78.4	9.6	0.96%
Eastern	Duarte	785	57.2	5.78%
Eastern	General Industrial Stormwater Permittees* (in the city of Duarte)	33	0.8	0.08%
Eastern	Irwindale	463	23.3	2.36%
Eastern	General Industrial Stormwater Permittees (in the city of Irwindale)	19.9	1.6	0.16%
Eastern	County of Los Angeles	499	28.6	2.89%
Eastern	Monrovia	4,323	200	20.2%
Eastern	General Industrial Stormwater Permittees (in the city of Monrovia)	134	16.3	1.65%

Subwatershed	Jurisdiction	Area (ac)	Annual Sediment Load (tons/yr)	Percent of Total Load
Eastern	Angeles National Forest	3,511	12.1	1.22%
Diversion	Los Angeles County Department of Public Works	-	379	38.3%
Near Lake	Arcadia	125	7.59	0.77%
Near Lake	General Industrial Stormwater Permittees (in the city of Arcadia)	14	1.70	0.17%
Near Lake	El Monte	52.1	3.55	0.36%
Near Lake	Irwindale	14.1	1.70	0.17%
Near Lake	County of Los Angeles	67.7	4.03	0.41%
Near Lake	Monrovia	48.1	2.62	0.26%
Western	Arcadia	1,720	68.1	6.88%
Western	General Industrial Stormwater Permittees (in the city of Arcadia)	310	37.8	3.82%
Western	Caltrans	16.9	2.06	0.21%
Western	County of Los Angeles	245	14.7	1.49%
Western	Monrovia	611	9.27	0.94%
Western	Sierra Madre	679	19.9	2.01%
Western	Angeles National Forest	9,104	31.4	3.18%
Total		**23,564**	**990.3**	**100%**

*The disturbed area associated with general industrial stormwater permittees was subtracted out of the appropriate city area and allocated to these permits.

The chlordane, PCB, DDT, and dieldrin loads were calculated by applying the input sediment concentrations (Table D-16) to the calculated sediment load of 900.3 tons per year (Table D-17). See example calculations in Section D.3. Loads for each jurisdiction are shown by subwatershed in Table D-18.

Table D-18. Total Organic Loads Estimated for Each Jurisdiction and Subwatershed in the Peck Road Park Lake Watershed (g/yr)

Subwatershed	Jurisdiction	Annual PCB Load	Annual Chlordane Load	Annual DDT Load[1]	Annual Dieldrin Load[1]	Percent of Total Load
Eastern	Arcadia	0.17	0.034	0.061	<0.010	1.22%
Eastern	Bradbury	0.62	0.127	0.224	<0.037	4.48%
Eastern	Caltrans	0.13	0.027	0.048	<0.008	0.96%
Eastern	Duarte	0.80	0.163	0.289	<0.047	5.78%

Subwatershed	Jurisdiction	Annual PCB Load	Annual Chlordane Load	Annual DDT Load[1]	Annual Dieldrin Load[1]	Percent of Total Load
Eastern	General Industrial Stormwater Permittees[2] (in the city of Duarte)	0.01	0.002	0.004	<0.001	0.08%
Eastern	Irwindale	0.33	0.067	0.118	<0.019	2.36%
Eastern	General Industrial Stormwater Permittees (in the city of Irwindale)	0.02	0.005	0.008	<0.001	0.16%
Eastern	County of Los Angeles	0.40	0.082	0.145	<0.024	2.89%
Eastern	Monrovia	2.80	0.573	1.013	<0.165	20.24%
Eastern	General Industrial Stormwater Permittees (in the city of Monrovia)	0.23	0.047	0.0821.65%0.061	<0.013	1.65%
Eastern	Angeles National Forest	0.17	0.035	1.917	<0.010	1.22%
Diversion	Los Angeles County Department of Public Works	5.29	1.084	0.038	<0.313	38.3%
Near Lake	Arcadia	0.11	0.022	0.009	<0.006	0.77%
Near Lake	General Industrial Stormwater Permittees (in the city of Arcadia)	0.02	0.005	0.018	<0.001	0.17%
Near Lake	El Monte	0.05	0.010	0.009	<0.003	0.36%
Near Lake	Irwindale	0.02	0.005	0.020	<0.001	0.17%
Near Lake	County of Los Angeles	0.06	0.012	0.013	<0.003	0.41%
Near Lake	Monrovia	0.04	0.007	0.344	<0.002	0.26%
Western	Arcadia	0.95	0.195	0.191	<0.056	6.88%
Western	General Industrial Stormwater Permittees (in the city of Arcadia)	0.53	0.108	0.010	<0.031	3.82%
Western	Caltrans	0.03	0.006	0.074	<0.002	0.21%
Western	County of Los Angeles	0.21	0.042	0.047	<0.012	1.49%
Western	Monrovia	0.13	0.026	0.100	<0.008	0.94%
Western	Sierra Madre	0.28	0.057	0.159	<0.016	2.01%
Western	Angeles National Forest	0.44	0.090	0.061	<0.026	3.18%
Total		**13.7**	**2.83**	**5.00**	**0.818**	**100%**

[1] Results from upper-bound analysis representing the maximum possible dieldrin load.

[2] The disturbed area associated with general industrial stormwater permittees was subtracted out of the appropriate city area and allocated to these permits.

D.5 Lincoln Park Lake

Lincoln Park Lake is located in the Los Angeles River Basin. Impairments of this lake include low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, lead, and trash. Output from the Los Angeles River LSPC model coupled with regional pollutant event mean concentrations has been used to estimate loads from upland areas of nutrients, which may be contributing to the low dissolved oxygen/organic enrichment, odor, eutrophication, and ammonia impairments.

Figure D-14 shows the MS4 stormwater permittee in the Lincoln Park Lake watershed (the city of Los Angeles). Though the lake appears to be connected to the county of Los Angeles storm drain network, this system actually passes under Lincoln Park Lake and does not discharge stormwater to the lake. The subwatershed for Lincoln Park Lake (37.1 acres) is comprised only of the surrounding parklands. All loads generated from this area are assigned load allocations for TMDL development.

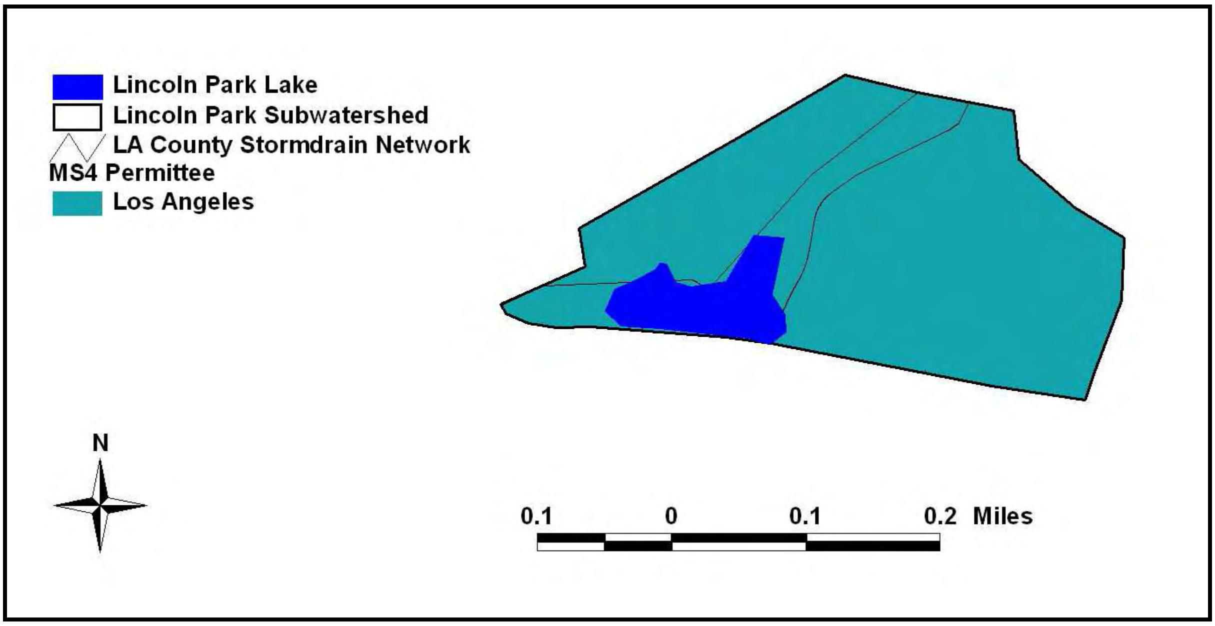


Figure D-14. MS4 Permittee and the County of Los Angeles Storm Drain Network in the Lincoln Park Lake Watershed

Land uses identified in the Los Angeles River LSPC model are shown in Figure D-15. The watershed is comprised of open space and industrial areas. Table D-19 summarizes the land use areas used to estimate pollutant loading from upland areas draining to Lincoln Park Lake.

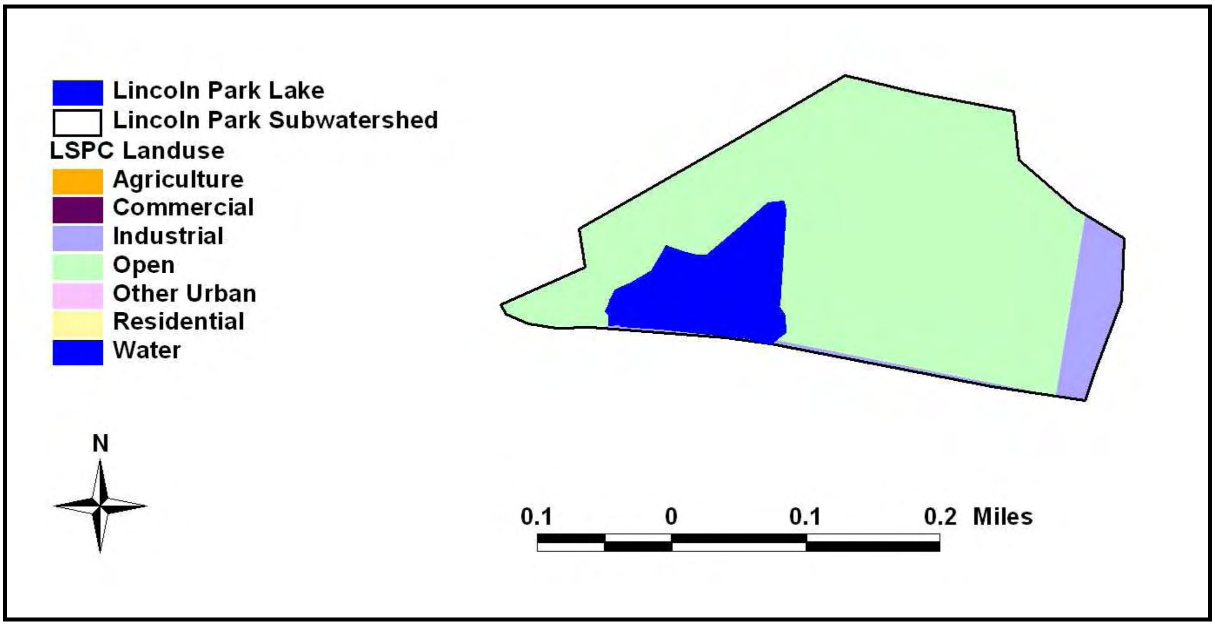


Figure D-15. LSPC Land Use Classes for the Lincoln Park Lake Watershed

Table D-19. Land Use Areas (ac) Draining to Lincoln Park Lake

Land Use	Los Angeles
Agriculture	0
Commercial	0
Industrial	3.40
Open	33.7
Other Urban	0
Residential	0
Total	**37.1**

D.5.1 RUNOFF

LSPC-predicted runoff from the Lincoln Park Lake watershed is primarily driven by the land use and soil characteristics of the drainage area and the nearest meteorological station represented in the model. Figure D-16 shows the simulated annual rainfall for the Lincoln Park Lake watershed. The annual average rainfall is 15.2 inches.

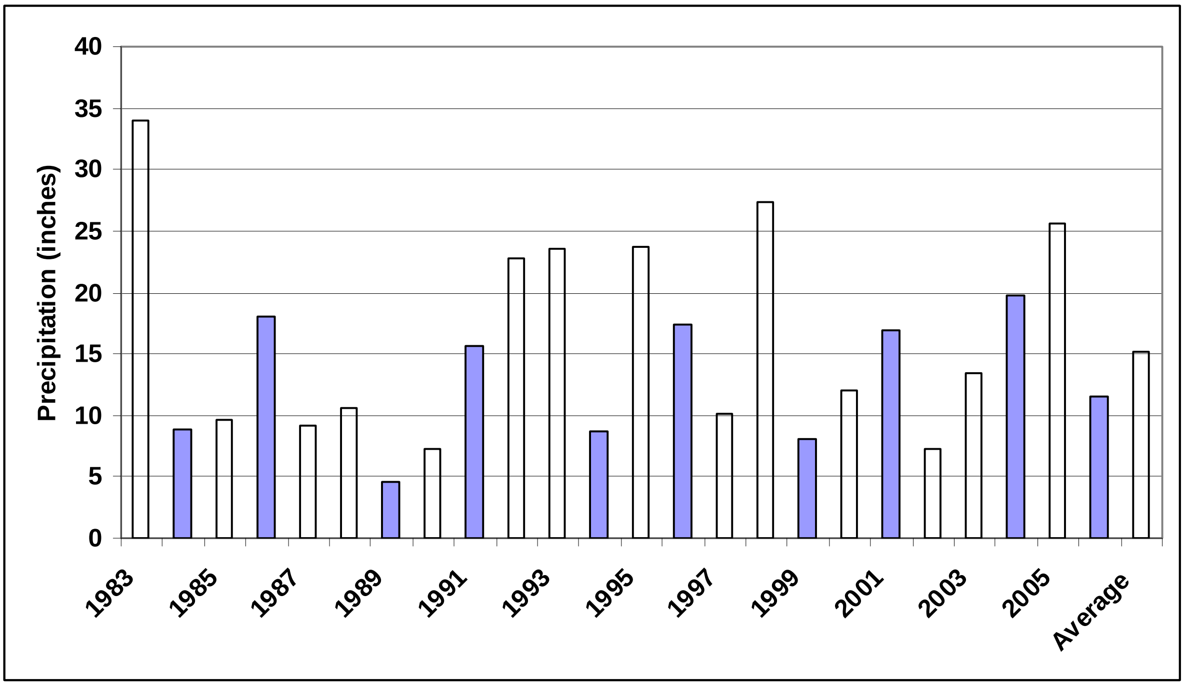


Figure D-16. Annual Rainfall for the Lincoln Park Lake Watershed

The simulated monthly average runoff depths for land uses in the Lincoln Park Lake watershed are shown in Table D-20.

Table D-20. Monthly Average Runoff Depths (inches/month) for Land Uses in the Lincoln Park Lake Watershed, 1983 - 2006

Month	Industrial	Open
January	2.0170	0.0963
February	2.7225	0.1613
March	1.7918	0.1136
April	0.5372	0.0334
May	0.1602	0.0094
June	0.0475	0.0024
July	0.0024	0.0002
August	0.0232	0.0010
September	0.1352	0.0055
October	0.3393	0.0136
November	0.6098	0.0244
December	1.2099	0.0487

Figure D-17 summarizes the monthly average runoff volumes delivered to Lincoln Park Lake. The total annual volume delivered to the lake is 4.15 ac-ft. The months May through October each contribute less than 5 percent of the annual runoff volume.

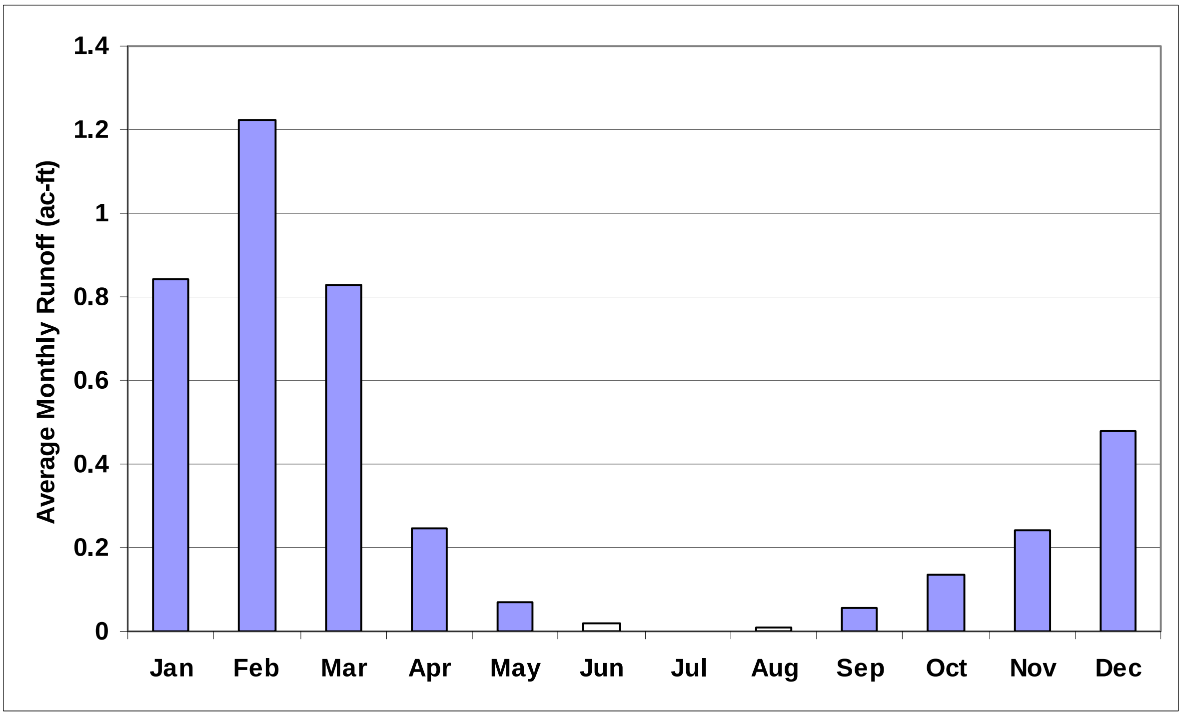


Figure D-17. Monthly Average Runoff Volumes to Lincoln Park Lake

D.5.2 NUTRIENT LOADS

Nutrient loads are estimated from simulated volumes and event mean concentration data collected by SCCWRP and the county of Los Angeles (Section D.3). Table D-21 summarizes the total nitrogen and total phosphorus loads estimated for Lincoln Park Lake. See example calculations in Section D.3.

Table D-21. Average Annual Nutrient Loads to Lincoln Park Lake

Jurisdiction	Nitrogen (lb/yr)	Phosphorus (lb/yr)
Los Angeles	46.1	4.72

D.6 Echo Park Lake

Echo Park Lake is located in the Los Angeles River Basin. Impairments of this lake include odor, ammonia, eutrophication, algae, pH, copper, lead, PCBs, dieldrin, chlordane, and trash. Output from the Los Angeles River LSPC model coupled with regional pollutant event mean concentrations have been used to estimate loads from upland areas of OC Pesticides and PCBs and nutrients, which may be contributing to the odor, ammonia, eutrophication, algae, and pH impairments.

Two subwatersheds comprise the drainage area to Echo Park Lake. The subwatershed draining the northern part of the watershed is 614 acres and the southern subwatershed drains 170 acres. Both subwatersheds drain to a storm drain system, so all allocations for the TMDLs are wasteload allocations (except for the trash TMDL which also has a load allocation). Dry weather flows from the storm drain system are diverted downstream of Echo Park Lake. Figure D-18 shows the MS4 stormwater permittees in the Echo Park Lake watershed. Both subwatersheds are located entirely within the city of Los Angeles with a small portion of Caltrans area. The park is comprised of 15.5 acres of land adjacent to the lake.

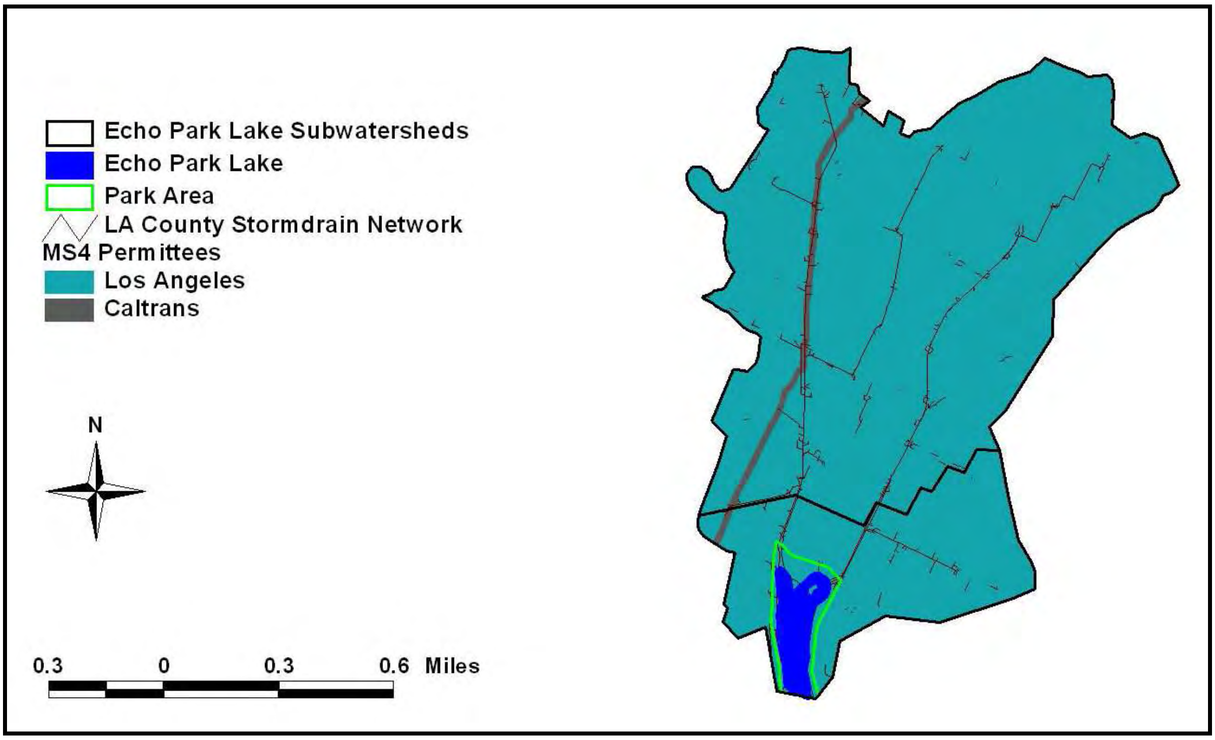


Figure D-18. MS4 Permittees and the County of Los Angeles Storm Drain Network in the Echo Park Lake Subwatersheds

Land uses identified in the Los Angeles River LSPC model are shown in Figure D-19. The watershed is comprised primarily of residential development as well as commercial, other urban, industrial, and open space areas. Table D-22 and Table D-23 summarize the land use areas used to estimate pollutant loading from the Northern and Southern subwatersheds, respectively.

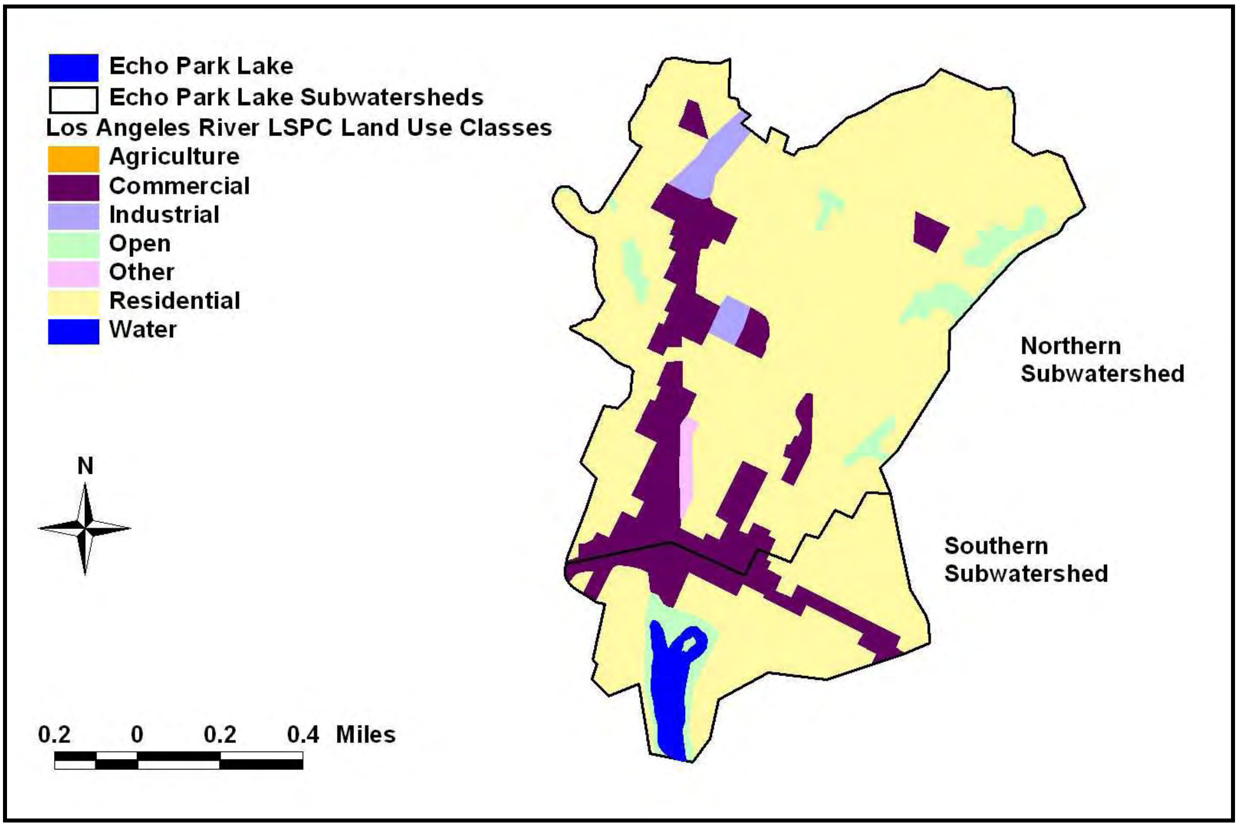


Figure D-19. LSPC Land Use Classes for the Echo Park Lake Subwatersheds

Table D-22. Land Use Areas (ac) Draining to Echo Park Lake from the Northern Subwatershed

Land Use	Los Angeles	Caltrans	Total
Agriculture	0	0	**0**
Commercial	78.4	0	**78.4**
Industrial	12.2	13.0	**25.2**
Open	27.5	0	**27.5**
Other Urban	4.67	0	**4.67**
Residential	479	0	**479**
Total	**601**	**13.0**	**614**

Table D-23. Land Use Areas (ac) Draining to Echo Park Lake from the Southern Subwatershed

Land Use	Los Angeles	Caltrans	Total
Agriculture	0	0	**0**
Commercial	31.6	0	**31.6**
Industrial	0	1.10	**1.10**
Open	15.5	0	**15.5**
Other Urban	0	0	**0**
Residential	122	0	**122**
Total	**169**	**1.10**	**170**

D.6.1 RUNOFF

LSPC-predicted runoff from the Echo Park Lake subwatersheds is primarily driven by the land use and soil characteristics of the drainage area and the nearest meteorological station represented in the model. Figure D-20 shows the simulated annual rainfall for the Echo Park Lake subwatersheds. The annual average rainfall is 15.2 inches.

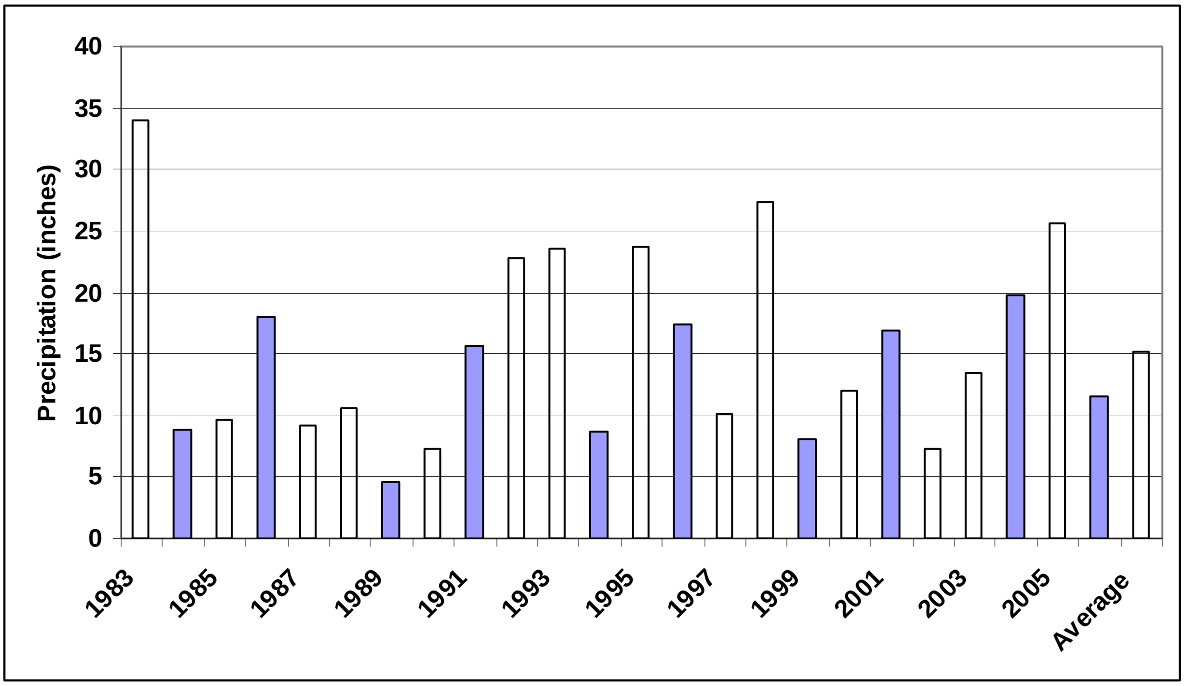


Figure D-20. Annual Rainfall for the Echo Park Lake Subwatersheds

The simulated monthly average runoff depths for land uses in the Echo Park Lake subwatersheds are shown in Table D-24.

Table D-24. Monthly Average Runoff Depths (inches/month) for Land Uses in the Echo Park Lake Subwatersheds, 1983 - 2006

Month	Agriculture	Commercial	Industrial	Open	Other Urban	Residential
January	0.2843	2.2493	2.0170	0.0963	1.4365	1.3899
February	0.4635	3.0258	2.7225	0.1613	1.9644	1.9036
March	0.3191	1.9875	1.7918	0.1136	1.3028	1.2636
April	0.0826	0.6010	0.5372	0.0334	0.3779	0.3651
May	0.0264	0.1783	0.1602	0.0094	0.1147	0.1111
June	0.0047	0.0537	0.0475	0.0024	0.0322	0.0309
July	0.0004	0.0027	0.0024	0.0002	0.0017	0.0016
August	0.0020	0.0263	0.0232	0.0010	0.0155	0.0149
September	0.0110	0.1532	0.1352	0.0055	0.0903	0.0867
October	0.0272	0.3845	0.3393	0.0136	0.2263	0.2172
November	0.0489	0.6911	0.6098	0.0244	0.4067	0.3904
December	0.0995	1.3697	1.2099	0.0487	0.8103	0.7783

The majority of the runoff from the Echo Park Lake watershed is diverted downstream of the lake and on average, only 16.7 ac-ft/yr are delivered through the storm drain network (personal communication, Charlie Yu, City of Los Angeles, 3/4/2010). The simulated runoff volumes and associated pollutant loading were scaled down by the ratio of delivered flow (16.7 ac-ft/yr) to simulated flow (452 ac-ft/yr) to estimate the amount of loading reaching Echo Park Lake. It was assumed that all runoff (0.6 ac-ft/yr) and associated pollutant loading from the 15.5 acres of park adjacent to the lake were not diverted downstream. Figure D-12 summarizes the monthly average runoff volumes delivered to Echo Park Lake. The total annual volume delivered to the lake is 17.3 ac-ft. The months May through October each contribute less than 5 percent of the annual runoff volume.

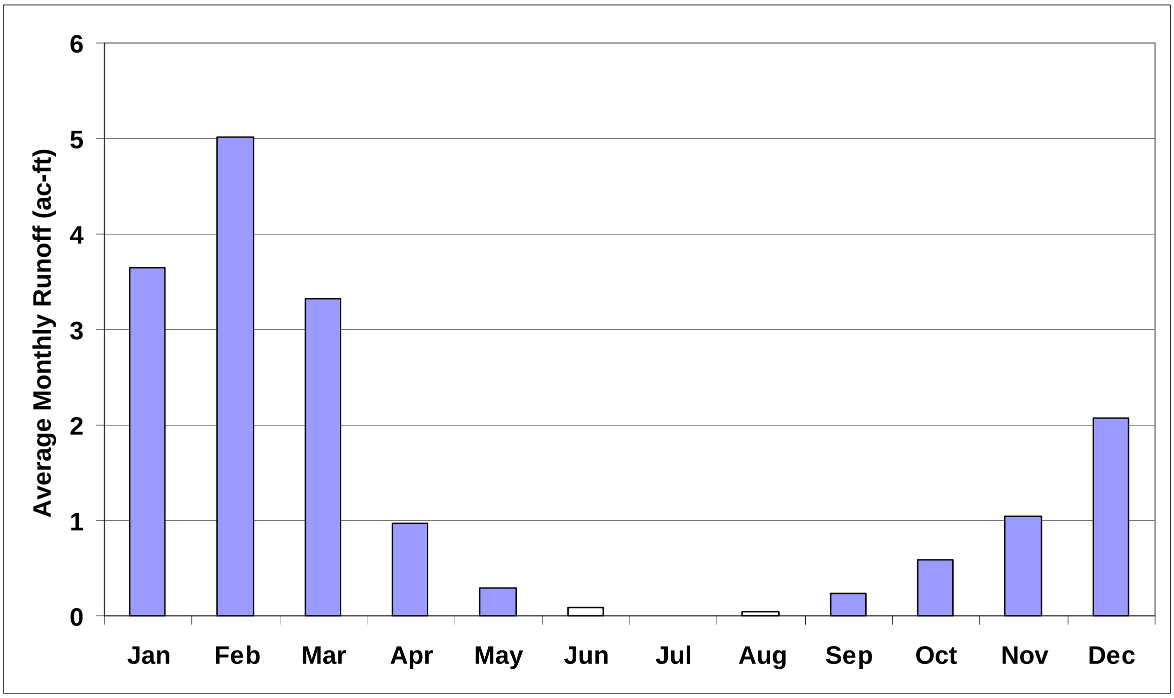


Figure D-21. Monthly Average Runoff Volumes to Echo Park Lake

D.6.2 SEDIMENT LOADS

Sediment loads are calculated from delivered runoff volumes and suspended sediment event mean concentrations for each modeled land use (Section D.3). Table D-25 summarizes the average annual sediment loads for each jurisdiction by subwatershed. See example calculations in Section D.3.

Table D-25. Average Annual Sediment Loads to Echo Park Lake

Subwatershed	Jurisdiction	Sediment (tons/yr)
Northern	City of Los Angeles	0.976
Northern	Caltrans	0.044
Southern	City of Los Angeles	0.291
Southern	Caltrans	0.0037
Total		**1.32**

D.6.3 NUTRIENT LOADS

Nutrient loads are estimated from delivered volumes and event mean concentration data collected by SCCWRP and the county of Los Angeles (Section D.3). Table D-26 summarizes the total nitrogen and total phosphorus loads estimated for Echo Park Lake from each jurisdiction and subwatershed. See example calculations in Section D.3.

Table D-26. Average Annual Nutrient Loads to Echo Park Lake

Subwatershed	Jurisdiction	Nitrogen (lb/yr)	Phosphorus (lb/yr)
Northern	City of Los Angeles	155	24.7
Northern	Caltrans	4.80	0.608
Southern	City of Los Angeles	169	6.99
Southern	Caltrans	1.10	0.051
Total		**209**	**32.3**

D.6.4 ORGANOCHLORINE PESTICIDES AND PCBS LOADS

The existing loading rates from upland areas for OC Pesticides and PCBs are estimated for each pollutant of concern using monitoring data collected by USEPA, the Regional Board, and UCLA, between 2008 and 2009. Only data from sites representing inflows are used; these sites include locations in an inflow, or in the lake near an inflow. Inflows considered for wet weather loading were tributaries, drainage paths, and channels. For Echo Park Lake, data from the following stations was included: EPL-1, EPL-2, and EPL-12 (Figure D-22).

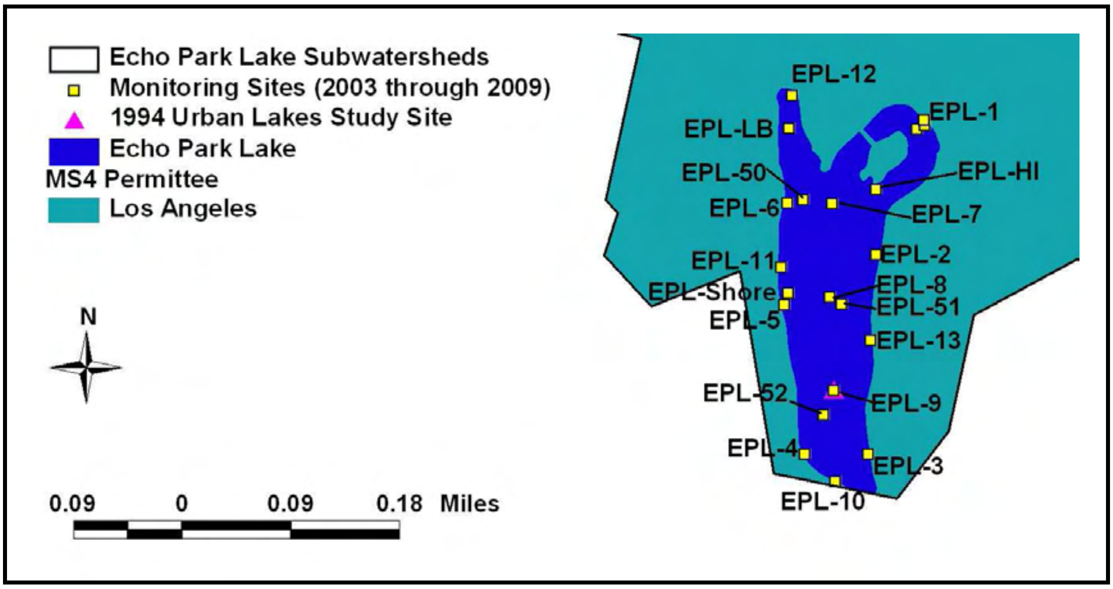


Figure D-22. Echo Park Monitoring Stations

The OC Pesticides and PCBs of concern are not currently in use and are more likely to have been historically loaded to the lake sediments; therefore, current tributary loading is likely to be small. The OC Pesticides and PCBs are hydrophobic and the majority of the pollutant mass in wet weather loads were associated with the sediment. The measured levels of OC Pesticides and PCBs in inflow sediments were the only data that could be used to quantify current inflow loads because nearly all of the water column, porewater, suspended sediment, and suspended sediment in porewater samples did not yield reportable results. For chlordane and PCBs, samples below detection limits were assumed to be one-half of the detection limits. For all of the sediment samples, dieldrin was below detection levels; therefore an inflow concentration could not be determined. Instead, an upper-bound analysis was performed using the detection limit as the incoming concentration associated with the sediment. The inflow sediment data are summarized in Table D-27 and all data collected in the watershed are discussed in detail in Appendix G (Monitoring Data).

Table D-27. Summary of Sediment Data near Inflow Locations at Echo Park Lake

Parameter	Number of Samples	Number of Samples Above Detection Limits[1]	Average Concentration (µg/kg dry weight)	Detection Limit Range (µg/kg dry weight)
Chlordane	6	2	8.31	0.44-1.23
Dieldrin	6	0	(1.32)[2]	0.83- 3.00
Total PCBs	6	5	24.16	0.44-1.23

[1] Non-detect samples were included in reported averages as one-half of the detection limit.

[2] All sample results were below detection limits. An upper-bound analysis was performed using the highest reported detection limit for dieldrin.

These input sediment concentrations were applied to the calculated sediment loads (Section D.6.2) to estimate the sediment-associated OC Pesticides and PCBs loads entering the lake. Sediment loads and subsequently calculated OC Pesticides and PCBs loads were determined for each jurisdiction based on the land use types and areas within each subwatershed. The jurisdictional areas are presented for the two Echo Park Lake subwatersheds in Table D-28. Dissolved concentrations in inflows are assumed insignificant.

Table D-28. Annual Sediment Load to Echo Park Lake

Subwatershed	Jurisdiction	Area (ac)	Annual Sediment Load (tons/yr)	Percent of Total Load
Northern	Caltrans	13.0	0.044	3.44%
Northern	City of Los Angeles	601	0.98	75.66%
Southern	Caltrans	1.10	0.0037	0.29%
Southern	City of Los Angeles	169	0.29	20.61%
Total		**784**	**1.32**	**100%**

The chlordane, PCB, and dieldrin loads were calculated by applying the input sediment concentrations (Table D-27) to the calculated sediment load of 1.32 tons per year (Table D-28). See example calculations in Section D.3. Loads for each jurisdiction are shown by subwatershed in Table D-29.

Table D-29. Total Organic Loads Estimated for Each Jurisdiction and Subwatershed in the Echo Park Watershed (g/yr)

Subwatershed	**Jurisdiction**	**Annual PCB Load**	**Annual Chlordane Load**	**Annual Dieldrin Load[1]**	**Percent of Total Load**
Northern	Caltrans	0.0010	0.0003	<0.00005	3.44%
Northern	Los Angeles	0.021	0.0074	<0.00117	75.66%
Southern	Caltrans	0.0001	0.00003	<0.00000	0.29%
Southern	Los Angeles	0.0064	0.0022	<0.00035	20.61%
Total		**0.029**	**0.0099**	**<0.0016**	**100%**

[1] Results from upper-bound analysis representing the maximum possible dieldrin load.

D.7 Lake Calabasas

Lake Calabasas is located in the Los Angeles River Basin. Impairments of this lake include low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, and pH. A DDT impairment was previously reported for this lake, but was delisted by the Regional Board in 2009. Output from the Los Angeles River LSPC model coupled with regional pollutant event mean concentrations have been used to estimate nutrient loads from upland areas, which may be contributing to the low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, and pH impairments.

One subwatershed draining 86.5 acres comprises the drainage area to Lake Calabasas. Figure D-23 shows the MS4 stormwater permittee in the Lake Calabasas watershed. The entire subwatershed is comprised of the city of Calabasas. This subwatershed drains to a storm drain system, so all allocations for the TMDLs are wasteload allocations.

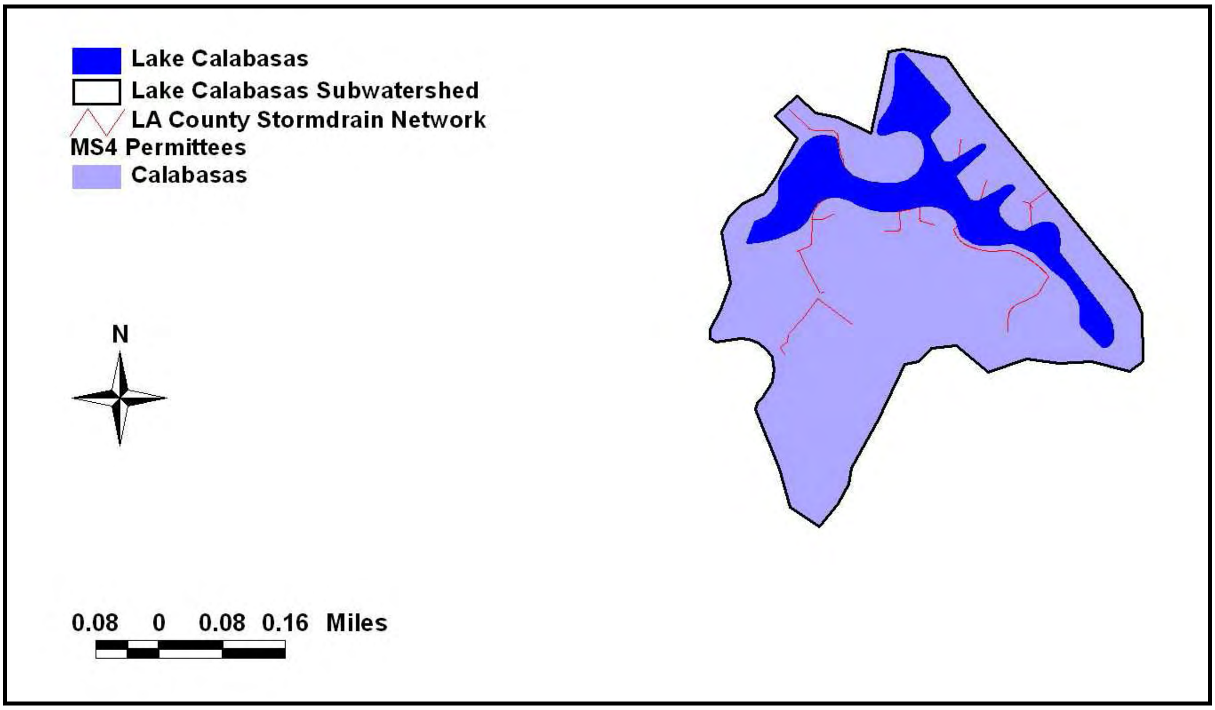


Figure D-23. MS4 Permittee and the County of Los Angeles Storm Drain Network in the Lake Calabasas Subwatersheds

Land uses identified in the Los Angeles River LSPC model are shown in Figure D-24. The watershed is comprised of residential development and open space. Table D-30 summarizes the land use areas used to estimate pollutant loading from upland areas draining to Lake Calabasas.

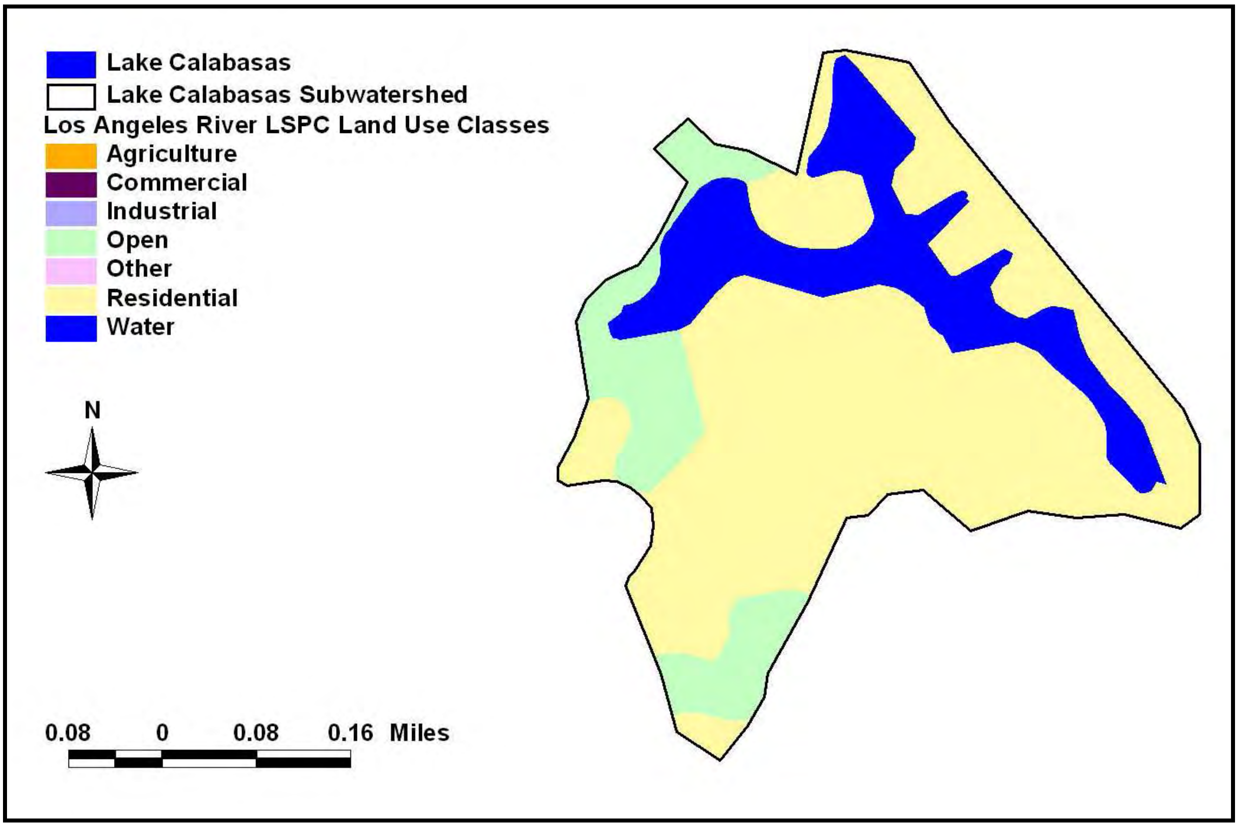


Figure D-24. LSPC Land Use Classes for the Lake Calabasas Subwatershed

Table D-30. Land Use Areas (ac) Draining to Lake Calabasas

Land Use	City of Calabasas
Agriculture	0
Commercial	0
Industrial	0
Open	14.2
Other Urban	0.0
Residential	72.3
Total	**86.5**

D.7.1 RUNOFF

LSPC-predicted runoff from the Lake Calabasas subwatershed is primarily driven by the land use and soil characteristics of the drainage area and the nearest meteorological station represented in the model. Figure D-25 shows the simulated annual rainfall for the Lake Calabasas subwatershed. The annual average rainfall is 17.5 inches.

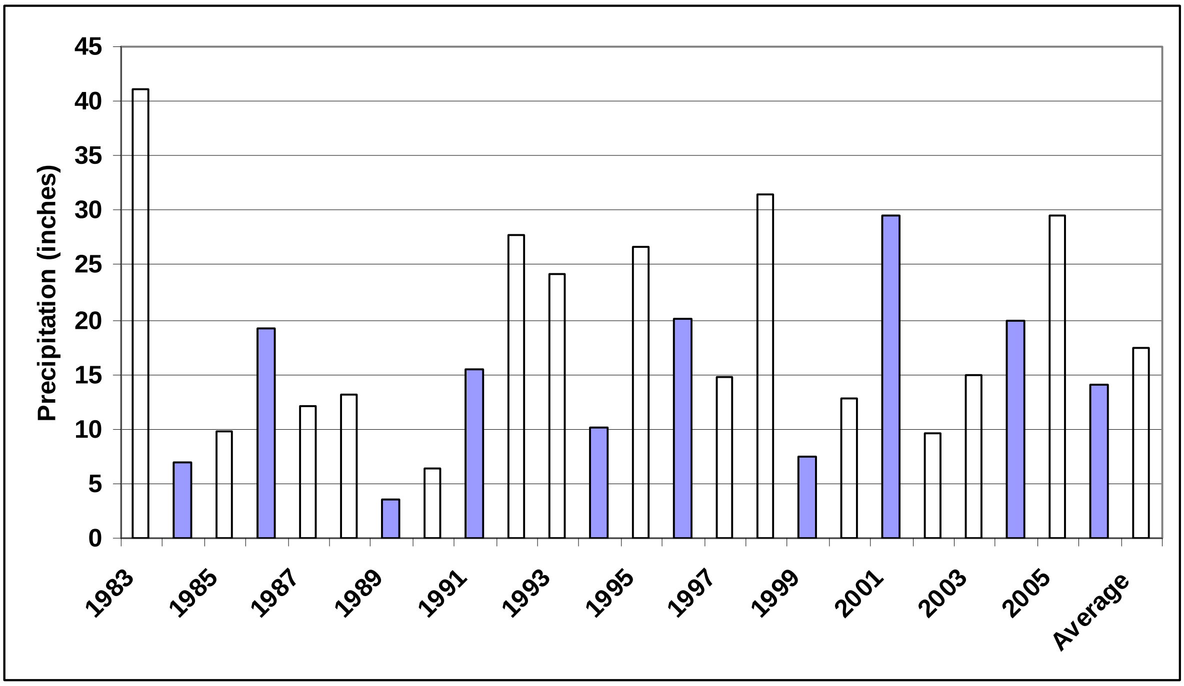


Figure D-25. Annual Rainfall for the Lake Calabasas Subwatershed

The simulated monthly average runoff depths for land uses in the Lake Calabasas subwatershed are shown in Table D-31.

Table D-31. Monthly Average Runoff Depths (inches/month) for Land Uses in the Lake Calabasas Subwatershed, 1983 - 2006

Month	Open	Residential
January	0.1271	1.6687
February	0.3202	2.5495
March	0.2219	1.5042
April	0.0473	0.4536
May	0.0174	0.1452
June	0.0020	0.0134
July	0.0005	0.0023
August	0.0009	0.0116
September	0.0056	0.0878
October	0.0192	0.3065
November	0.0342	0.5464
December	0.0585	0.9309

Figure D-26 summarizes the monthly average runoff volumes delivered to Lake Calabasas. The total annual volume delivered to the lake is 50.6 ac-ft. The months May through October each contribute less than 5 percent of the annual runoff volume.

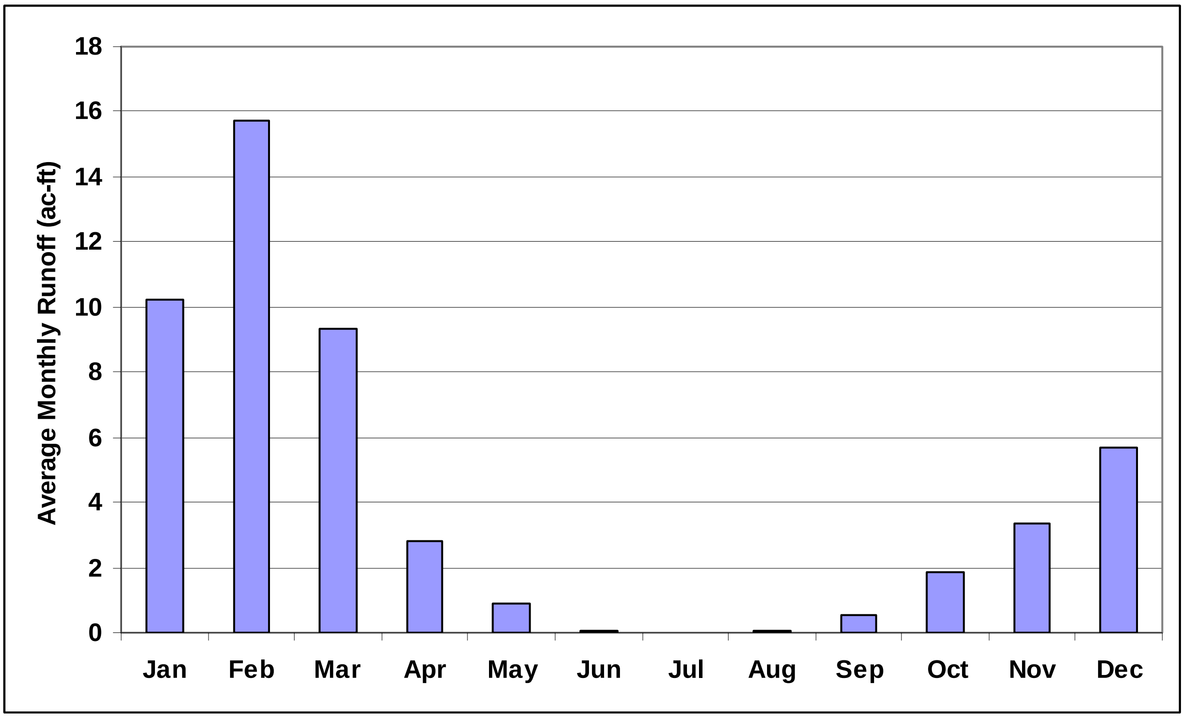


Figure D-26. Monthly Average Runoff Volumes to Lake Calabasas

D.7.2 NUTRIENT LOADS

Nutrient loads are estimated from runoff volumes and event mean concentration data collected by SCCWRP and the county of Los Angeles (Section D.3). Table D-32 summarizes the total nitrogen and total phosphorus loads delivered to Lake Calabasas. See example calculations in Section D.3.

Table D-32. Average Annual Nutrient Loads to Lake Calabasas

Subwatershed	Jurisdiction	Nitrogen (lb/yr)	Phosphorus (lb/yr)
Calabasas	Calabasas	616	98.7

D.8 El Dorado Park Lakes

The El Dorado Park lakes are located in the San Gabriel River Basin. Six lakes are located in the park. The northern four lakes are hydraulically connected and separate from the system comprised by the two southern lakes, also hydraulically connected. These lakes are listed as impaired by algae, ammonia, eutrophication, pH, copper, lead, and mercury. Output from the San Gabriel River LSPC model, coupled with regional pollutant event mean concentrations, has been used to estimate loads of nutrients, which may be contributing to the algae, ammonia, eutrophication, and pH impairments. LSPC model output and monitoring data collected in 2009 are used to estimate mercury loading.

Two separate watersheds have been delineated for these separate lake systems. The subwatershed draining to the northern four lakes is comprised of 185 acres, and the subwatershed draining to the southern two lakes is comprised of 33.8 acres.

Figure D-27 shows the MS4 stormwater permittee that comprises both the northern and southern drainages of the El Dorado Park lake systems as well as the Los Angeles County storm drain network. Though both watersheds are in the city of Long Beach incorporated area, there are no major drains that divert runoff directly to the lake: a few small culverts pass water beneath walking paths and park roads. Because both watersheds are comprised solely of parklands that do not drain to a major storm drain system, the watershed loads to the El Dorado Park lakes are assigned load allocations in the TMDLs.

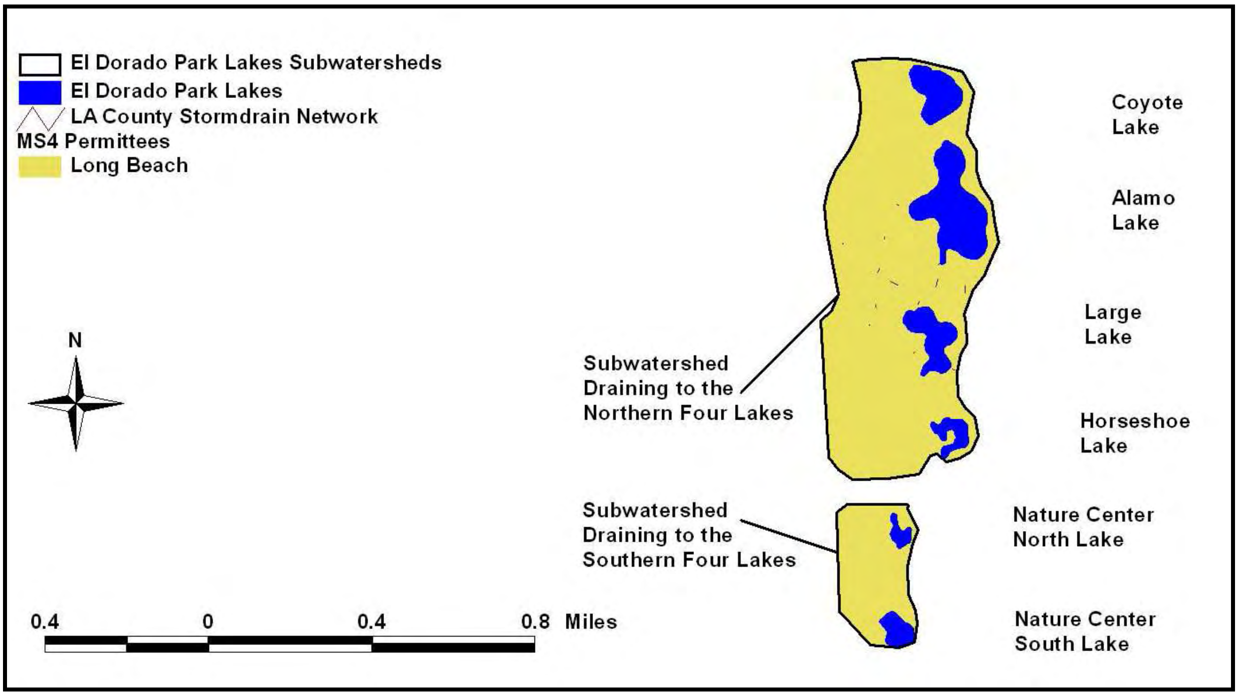


Figure D-27. MS4 Permittee and the County of Los Angeles Storm Drain Network in the El Dorado Park Lake Subwatersheds

Both subwatersheds are comprised of land classified by the San Gabriel LSPC model as “other urban or built-up” except for the two polygons classified as water (Figure D-28). To improve accuracy in land use areas, the SCAG 2005 database was used to estimate the area of the lakes in each subwatershed. Runoff loads from the lakes are assumed zero. All remaining areas in each subwatershed were assumed other urban or built-up (185 acres of the northern subwatershed and 33.8 acres in the southern subwatershed).

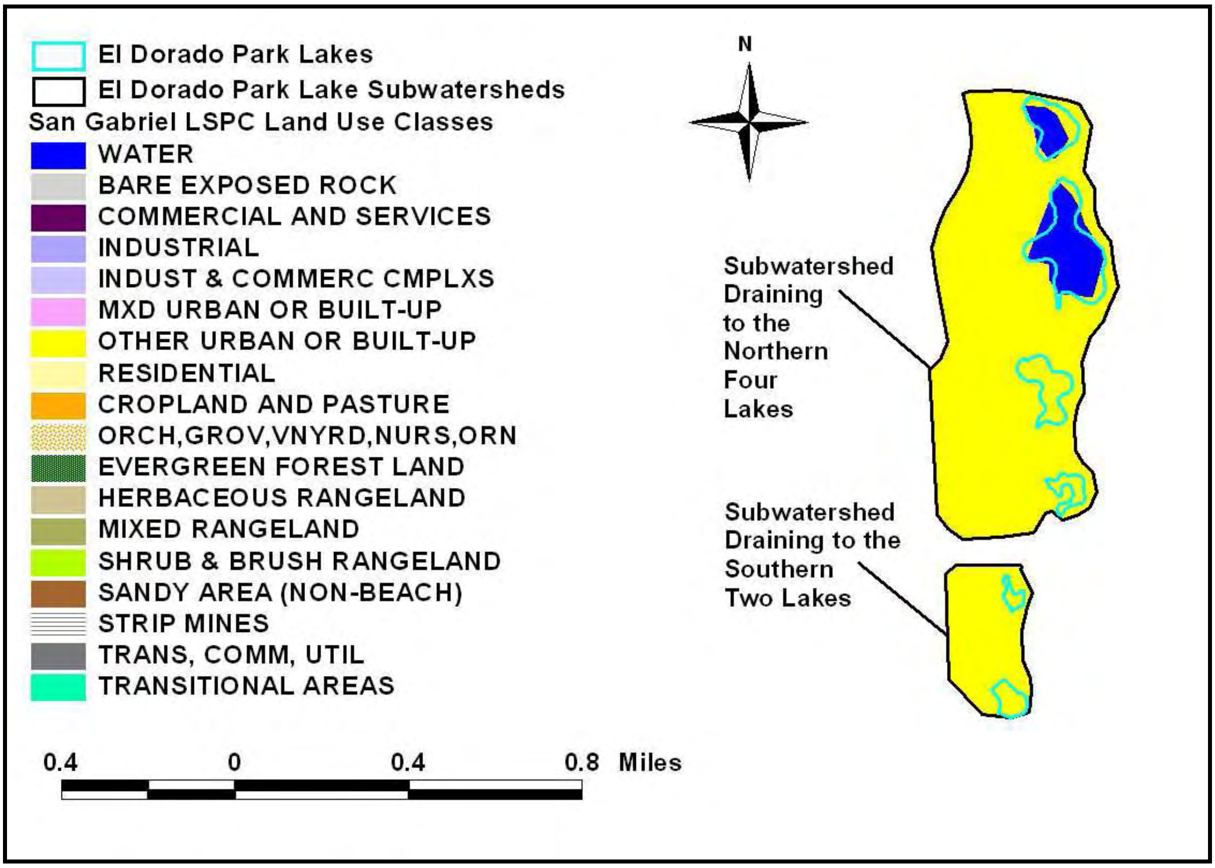


Figure D-28. LSPC Land Use Classes for the El Dorado Park Lakes Subwatersheds

The San Gabriel LSPC Model aggregated the identified land uses into modeled land uses. In the original model, lands classified as "other urban or built-up" were modeled as commercial areas, which is reasonable at the larger basin scale for which the model was developed. Comparison to the SCAG 2005 dataset and current satellite imagery indicate that these areas around the El Dorado Park lakes are actually parkland. To simulate pollutant loading from these areas, LSPC output for pervious commercial areas was assumed representative of park areas. Table D-33 summarizes the areas draining to the El Dorado Park lakes.

Table D-33. Areas Draining to the El Dorado Park Lakes

Land Use	Long Beach – Northern Lake System	Long Beach – Southern Lake System
Other urban or built-up (parkland)	185	33.8

D.8.1 RUNOFF

LSPC-predicted runoff from the El Dorado Park lakes subwatersheds is primarily driven by the land use and soil characteristics of the drainage area and the nearest meteorological station represented in the model. Figure D-29 shows the simulated annual rainfall for the El Dorado Park lakes subwatersheds. The annual average rainfall is 11.7 inches.

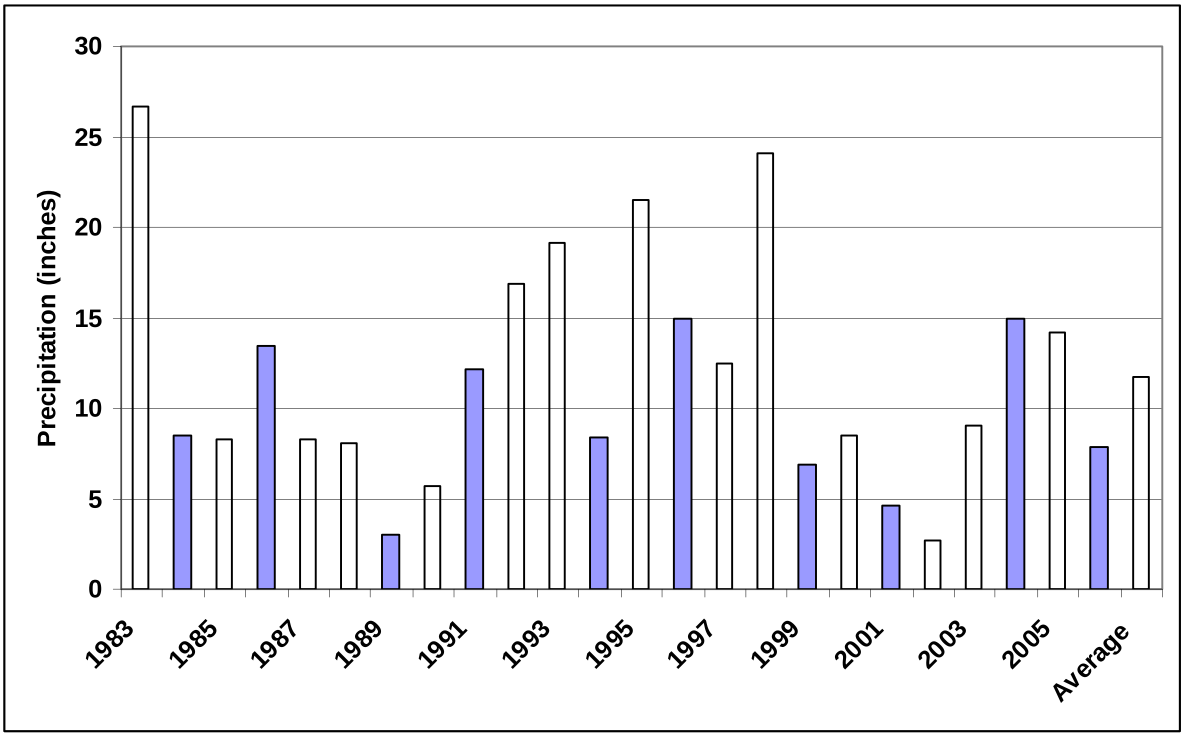


Figure D-29. Annual Rainfall for the El Dorado Park Lakes Subwatersheds

The simulated monthly average runoff for parkland (commercial pervious) areas in the El Dorado Park lakes subwatersheds is shown in Table D-34.

Table D-34. Monthly Average Runoff Depths (inches/month) for Land Uses in the El Dorado Park Lakes Subwatersheds, 1983 - 2006

Month	Runoff from Parkland
January	0.0154
February	0.0268
March	0.0345
April	0.0206
May	0.0069
June	0.0018
July	0.0005
August	0.0002
September	0.0001
October	0.0003
November	0.0007
December	0.0020

Table D-35 summarizes the monthly average runoff volumes from each subwatershed draining to the El Dorado Park lakes.

Table D-35. Monthly Average Runoff Volumes (ac-ft/month) from the El Dorado Park Lakes Subwatersheds

Subwatershed:	Northern	Southern
Month Land Use:	**Parkland**	**Parkland**
January	0.2377	0.0435
February	0.4130	0.0755
March	0.5308	0.0971
April	0.3167	0.0579
May	0.1060	0.0194
June	0.0277	0.0051
July	0.0071	0.0013
August	0.0025	0.0005
September	0.0013	0.0002
October	0.0051	0.0009
November	0.0106	0.0019
December	0.0311	0.0057
Annual Volume (ac-ft/yr)	**1.69**	**0.309**

D.8.2 SEDIMENT LOADS

Sediment loads from each subwatershed are based on simulated runoff volumes and suspended sediment event mean concentrations. The assumed suspended sediment event mean concentration for the LSPC model for other urban areas is 56.5 mg/L (Table D-7). Table D-36 summarizes the monthly sediment loads from each subwatershed in El Dorado Park. See example calculations in Section D.3.

Table D-36. Monthly Average Sediment Loads (lbs) from the El Dorado Park Lakes Subwatersheds

Month	Northern Lake System	Southern Lake System
January	36.5	6.68
February	63.5	11.6
March	81.6	14.9
April	48.7	8.90
May	16.3	2.98
June	4.26	0.778
July	1.09	0.199

Month	**Northern Lake System**	**Southern Lake System**
August	0.382	0.070
September	0.195	0.036
October	0.787	0.144
November	1.63	0.299
December	4.79	0.875
Annual Load (lb/yr)	**259.6**	**47.5**

D.8.3 NUTRIENT LOADS

Nutrient loads are estimated from event mean concentration data collected by SCCWRP and the county of Los Angeles. For "other urban" land uses, the total nitrogen event mean concentration is 4.41 mg-N/L and the total phosphorus event mean concentration is 0.67 mg-P/L (Table D-7). Table D-37 and Table D-38 summarize the total nitrogen and total phosphorus loads, respectively, from each subwatershed draining to the El Dorado Park lakes. See example calculations in Section D.3.

Table D-37. Monthly Average Nitrogen Loads (pounds) from the El Dorado Park Lakes Subwatersheds

Month	**Northern Lake System**	**Southern Lake System**
January	2.85	0.521
February	4.95	0.906
March	6.37	1.16
April	3.80	0.695
May	1.27	0.232
June	0.332	0.061
July	0.085	0.015
August	0.030	0.005
September	0.015	0.003
October	0.061	0.011
November	0.127	0.023
December	0.374	0.068
Annual Load (lb/yr)	**20.26**	**3.71**

Table D-38. Monthly Average Phosphorus Loads (lbs) from the El Dorado Park Lakes Subwatersheds

Month	Northern Lake System	Southern Lake System
January	0.433	0.0792
February	0.752	0.138
March	0.967	0.177
April	0.577	0.106
May	0.193	0.0353
June	0.0505	0.0092
July	0.0129	0.0024
August	0.0045	0.0008
September	0.0023	0.0004
October	0.0093	0.0017
November	0.0194	0.0035
December	0.0567	0.0104
Annual Load (lb/yr)	**3.08**	**0.563**

D.8.4 MERCURY LOADS

Mercury loads from each subwatershed are based on monitoring data collected by the Regional Board and USEPA during the winter and summer of 2009. Mercury loading is associated with both sediment and runoff from upland areas. To determine sediment loading of mercury, the sediment EMCs and runoff volumes were used to calculate sediment loads, and the sediment mercury concentrations from monitoring data were then applied to the sediment loads. Mercury loading associated with runoff from upland areas was calculated by applying the mercury water column concentrations to simulated runoff volumes (Section D.3 provides examples of these calculations). However, during both the February and July 2009 sampling events, the only visible inputs to the El Dorado Park lakes were the groundwater input to Coyote Lake and the potable water input to Nature Center North Lake. Loads associated with these inputs are discussed in Appendix F (Dry Weather Loading).

To estimate loading associated with wet weather events, concentrations measured from culverts and tributaries around Puddingstone Reservoir in the southern subwatershed (Section D.10.4) were assumed representative of concentrations for the El Dorado Park lakes. Puddingstone Reservoir is located in Bonelli Regional Park and the land uses surrounding the reservoir are similar to those in El Dorado Park. Table D-39 and Table D-40 present the assumed concentrations and resulting loads for total mercury and methylmercury, respectively. Example calculations are presented in Section D.3.

Table D-39. Total Mercury Loads Estimated for Each Subwatershed in El Dorado Park

Sub-watershed	Juris-diction	Area (ac)	Summer Water Column Hg (ng/L)[1]	Winter Water Column Hg (ng/L)[1]	Summer Sediment Hg (µg/kg)[2]	Winter Sediment Hg (µg/kg)[2]	Annual Water Column Hg Load (g/yr)	Annual Sediment Hg Load (g/yr)	Total Annual Hg Load (g/yr)
Northern Lake System	Long Beach	185	7.55	2.65	50.3	36.4	0.00643	0.00443	0.0109
Southern Lake System	Long Beach	33.8	7.55	2.65	50.3	36.4	0.00118	0.000810	0.00199

[1] Concentrations are based on observations around Puddingstone Reservoir (Table D-50).

[2] Concentrations are based on observations around Puddingstone Reservoir (Table D-51).

Table D-40. Methylmercury Loads Estimated for Each Subwatershed in El Dorado Park

Sub-watershed	Juris-diction	Area (ac)	Summer Water Column MeHg (ng/L)[1]	Winter Water Column MeHg (ng/L)[1]	Summer Sediment MeHg (µg/kg)[2]	Winter Sediment MeHg (µg/kg)[2]	Annual Water Column MeHg Load (g/yr)	Annual Sediment MeHg Load (g/yr)	Total Annual MeHg Load (g/yr)
Northern Lake System	Long Beach	185	0.046	0.010	0.716	0.002	2.75E-05	7.68E-06	3.52E-05
Southern Lake System	Long Beach	33.8	0.046	0.010	0.716	0.002	5.03E-06	1.40E-06	6.43E-06

[1] Concentrations are based on observations around Puddingstone Reservoir (Table D-50).

[2] Concentrations are based on observations around Puddingstone Reservoir (Table D-51).

(This page left intentionally blank.)

D.9 North, Center, and Legg Lakes

North, Center, and Legg lakes are hydraulically connected waterbodies in Whittier Narrows Regional Park located in the Los Angeles River Basin. Legg Lake is listed as impaired by odor, ammonia, pH, copper, and lead (note: trash impairment has been addressed by a previous TMDL). Output from the Los Angeles River LSPC model coupled with regional pollutant event mean concentrations have been used to estimate existing loading rates from upland areas of nutrients, which may be contributing to the odor, ammonia, and pH impairments.

Five subwatersheds comprise the drainage area to these lakes. The northwestern and northeastern subwatersheds each drain to a storm drain that enters North Lake on the north side. Three separate drainage areas have been delineated around the lakes to designate respective overland flow.

The northwestern, northeastern, and direct to North Lake subwatersheds flow into North Lake which is basically separate from Center and Legg lakes during dry periods; North Lake discharges to Morris Creek. Legg Lake receives inputs from the direct to Legg Lake subwatershed, from a Superfund site that discharges treated groundwater to the lake, and from pumped groundwater that is split between North and Center lakes to maintain water levels. Legg Lake drains into Center Lake via a connecting channel which then discharges to Morris Creek. There are two culverts connecting Center and North lakes that allow water to flow between them when levels are sufficiently high.

Figure D-34 shows the MS4 stormwater permittees in the North, Center, and Legg lakes watershed. Loads generated from El Monte, South El Monte, the county of Los Angeles, and Caltrans from either the northwestern or northeastern subwatersheds are assigned wasteload allocations in the TMDLs because they drain to the storm drain network. Loads generated by South El Monte or the county of Los Angeles areas in the direct drainage subwatersheds are assigned load allocations; Caltrans areas in these subwatersheds are assigned wasteload allocations.

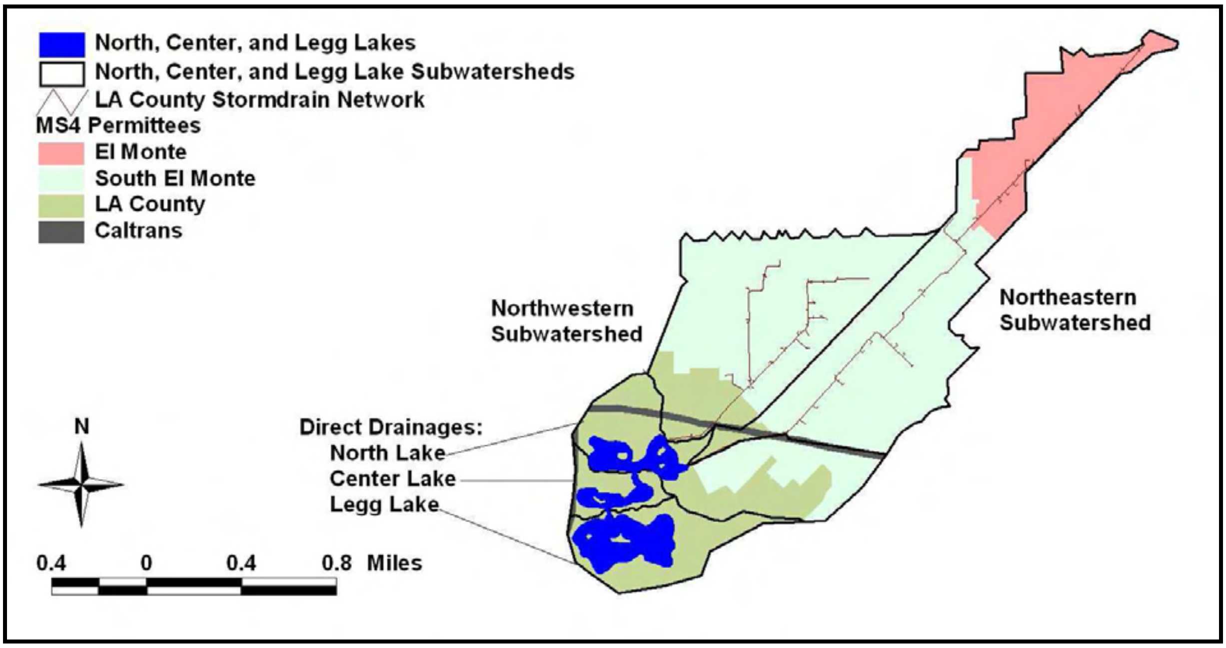


Figure D-30. MS4 Permittees and the County of Los Angeles Storm Drain Network in the North, Center, and Legg Subwatersheds

Land uses identified in the Los Angeles River LSPC model for these subwatersheds are shown in Figure D-35. Tetra Tech reviewed the SCAG 2005 database and current satellite imagery to confirm the acreage of agricultural areas present in the LSPC model. Land use classifications were changed to accurately reflect the conditions identified in the more recent data. Specifically, the following changes were made to maintain consistency with the SCAG 2005 land use database: in the direct drainage subwatershed to Legg Lake, approximately half of the agricultural area was modified as it is actually parkland, and the agricultural areas assigned in the direct to North Lake and north-eastern subwatersheds were changed to vacant land. In addition, the agricultural area present in the northwestern subwatershed is classified by SCAG 2005 as nurseries; however, this area was reclassified to parkland as current satellite imagery shows this area to be Shiveley Park. For the purposes of estimating flows and pollutant loads to this lake system, all agricultural areas were re-assigned as open space, with the exception of 1.04 acres located in the direct to Legg Lake subwatershed. The area classified as "other" is a high school according to SCAG 2005. Table D-41 and Table D-42 summarize the land use types present in the northern two subwatersheds and direct drainage subwatersheds, respectively.

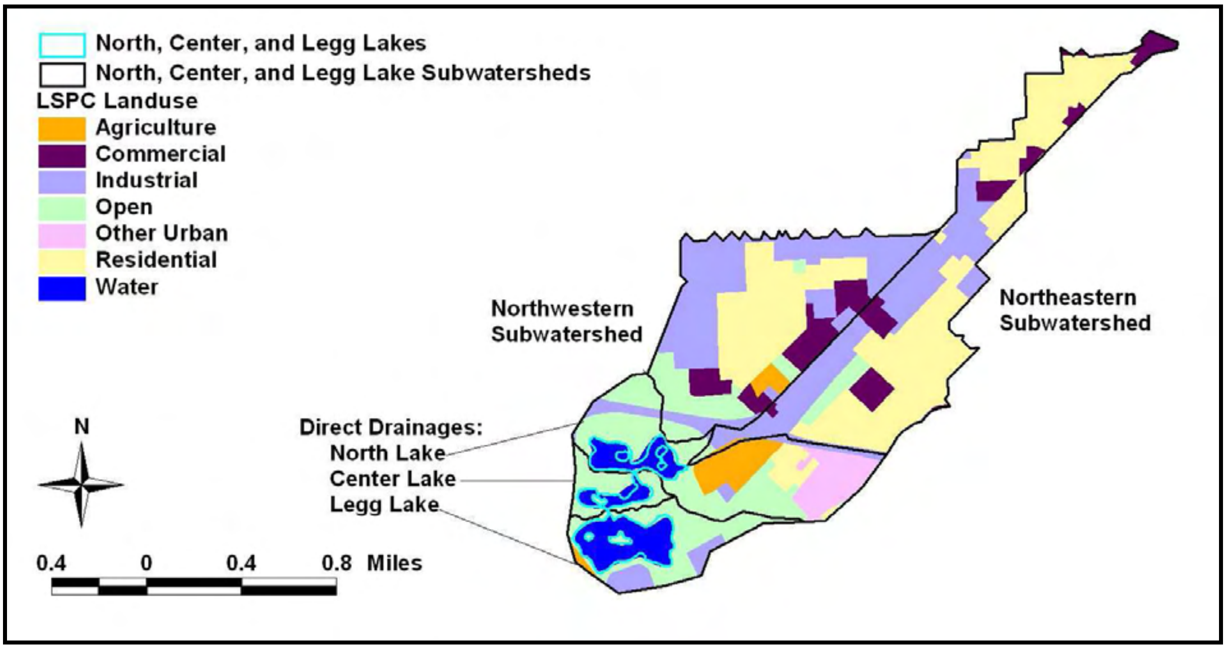


Figure D-31. LSPC Land Use Classes for the North, Center, and Legg Lake Subwatersheds

Table D-41. Land Use Areas (ac) Draining from the Northern Subwatersheds to North, Center, and Legg Lakes

Land Use	El Monte	South El Monte	County of Los Angeles	Caltrans	Total
Agriculture	0	0	0	0	**0**
Commercial	23.5	58.0	11.9	0	**93.5**
Industrial	6.49	269	13.4	11.5	**300**
Open	0	29.3	44.6	0	**73.9**
Other Urban	0	0.0	0	0	**0**
Residential	104	267	0.271	0	**371**
Total	**134**	**623**	**70.2**	**11.5**	**838**

Table D-42. Land Use Areas (ac) Draining from the Direct Drainage Subwatersheds to North, Center, and Legg Lakes

Land Use	South El Monte	County of Los Angeles	Caltrans	Total
Agriculture	0	1.04	0	**1.04**
Commercial	0	0	0	**0**
Industrial	1.78	24.1	17.6	**43.4**
Open	29.8	202	0	**232**
Other Urban	28.2	12.1	0	**40.3**
Residential	15.8	1.19	0	**17.0**
Total	**75.7**	**240**	**17.6**	**334**

D.9.1 RUNOFF

LSPC-predicted runoff is primarily driven by the land use and soil characteristics of the drainage area and the nearest meteorological station represented in the model. Figure D-32 shows the simulated annual rainfall for the North, Center, and Legg lakes subwatersheds. The annual average rainfall is 16.5 inches.

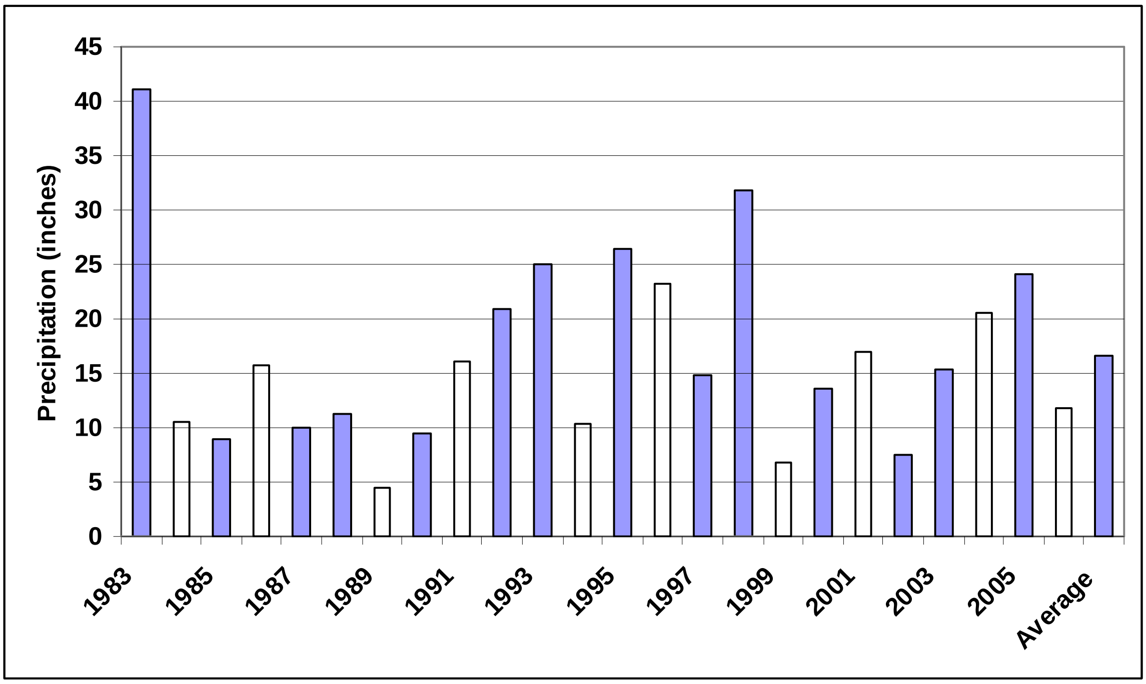


Figure D-32. Annual Rainfall for the North, Center, and Legg Lake Subwatersheds

The simulated monthly average runoff depths for land uses in the North, Center, and Legg lakes subwatersheds are shown in Table D-43.

Table D-43. Monthly Average Runoff Depths (inches/month) for Land Uses in the North, Center, and Legg Lake Subwatersheds, 1983 - 2006

Month	Agriculture	Commercial	Industrial	Open	Other Urban	Residential
January	0.3332	2.5316	2.2740	0.1144	1.6302	1.5787
February	0.5856	3.2297	2.9171	0.2374	2.1356	2.0731
March	0.3748	2.2020	1.9888	0.1469	1.4557	1.4130
April	0.0825	0.6283	0.5613	0.0372	0.3938	0.3804
May	0.0467	0.2536	0.2289	0.0179	0.1672	0.1622
June	0.0073	0.0825	0.0731	0.0038	0.0494	0.0475
July	0.0002	0.0004	0.0004	0.0001	0.0003	0.0003
August	0.0067	0.0922	0.0814	0.0033	0.0544	0.0522
September	0.0132	0.1843	0.1627	0.0066	0.1086	0.1042
October	0.0378	0.5315	0.4691	0.0188	0.3133	0.3008
November	0.0533	0.7505	0.6623	0.0265	0.4418	0.4242
December	0.1103	1.4977	1.3232	0.0534	0.8870	0.8521

Figure D-33 summarizes the monthly average runoff volumes from each subwatershed from 1983 through 2006. The total annual volume estimated for the lakes is 682 ac-ft. The months May through October each contribute less than 5 percent of the annual runoff volume.

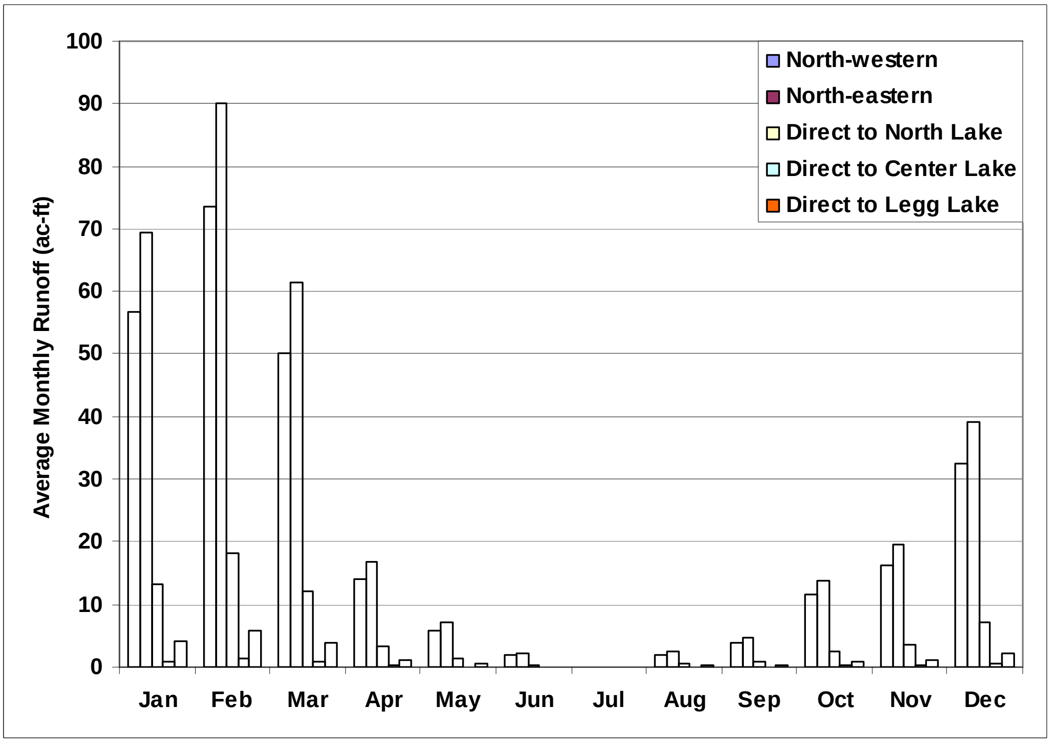


Figure D-33. Monthly Average Runoff Volumes to North, Center, and Legg Lakes (1983-2006)

D.9.2 NUTRIENT LOADS

Nutrient loads are estimated from event mean concentration data collected by SCCWRP and the county of Los Angeles (Section D.3). Table D-44 summarizes the total nitrogen and total phosphorus loads delivered from each subwatershed and jurisdiction contributing to the Legg Lake system. See example calculations in Section D.3.

Table D-44. Average Annual Nutrient Loads to North, Center, or Legg Lakes

Subwatershed	Jurisdiction	Area (ac)	Nitrogen (lb/yr)	Phosphorus (lb/yr)
Direct to Center Lake	Caltrans	3.26	36.1	4.60
Direct to Center Lake	County of Los Angeles	30.4	14.7	0.505
Direct to Legg Lake	Caltrans	0.837	9.28	1.18
Direct to Legg Lake	County of Los Angeles	83.1	228	26.0
Direct to North Lake	Caltrans	13.5	149	19.1
Direct to North Lake	County of Los Angeles	127	226	26.6
Direct to North Lake	South El Monte	75.7	369	55.1
Northwestern	Caltrans	5.32	58.9	7.51
Northwestern	County of Los Angeles	60.1	241	32.4
Northwestern	South El Monte	317	2,982	420
Northeastern	Caltrans	6.18	68.5	8.73
Northeastern	El Monte	134	1,140	179
Northeastern	County of Los Angeles	10.0	73.7	9.24
Northeastern	South El Monte	305	2,716	391
Total		**1,172**	**8,313**	**1,182**

(This page left intentionally blank.)

D.10 Puddingstone Reservoir

Puddingstone Reservoir is located in the San Gabriel River Basin. Impairments include low dissolved oxygen/organic enrichment, mercury, chlordane, DDT, dieldrin, and PCBs. Output from the San Gabriel River LSPC model coupled with regional pollutant event mean concentrations has been used to estimate loads of nutrients from upland areas, which may be contributing to the low dissolved oxygen/organic enrichment impairment. LSPC model output and monitoring data collected in 2009 are used to estimate mercury and OC Pesticides and PCBs loading.

Two subwatersheds comprise the drainage area to Puddingstone Reservoir. The subwatershed draining the northern part of the watershed is 6,959 acres, and the southern subwatershed is 1,169 acres. The subwatershed boundaries were chosen to separate those areas that drain to a storm drain (the northern subwatershed) and those that enter the reservoir via natural tributaries or overland flow (the southern subwatershed).

Figure D-34 shows the MS4 stormwater permittees in the Puddingstone Reservoir watershed. The northern subwatershed is primarily comprised of the county of Los Angeles, Claremont, and La Verne areas with a small amount of San Dimas, Caltrans, and Angeles National Forest areas. Loads generated from these jurisdictions in the northern subwatershed were assigned wasteload allocations because they drain to the Los Angeles County storm drain network. The southern subwatershed is comprised of San Dimas, La Verne, and Pomona areas. Loads from these jurisdictions originating in the southern subwatershed were assigned load allocations. The small amount of Caltrans area in the southern subwatershed were assigned a wasteload allocation.

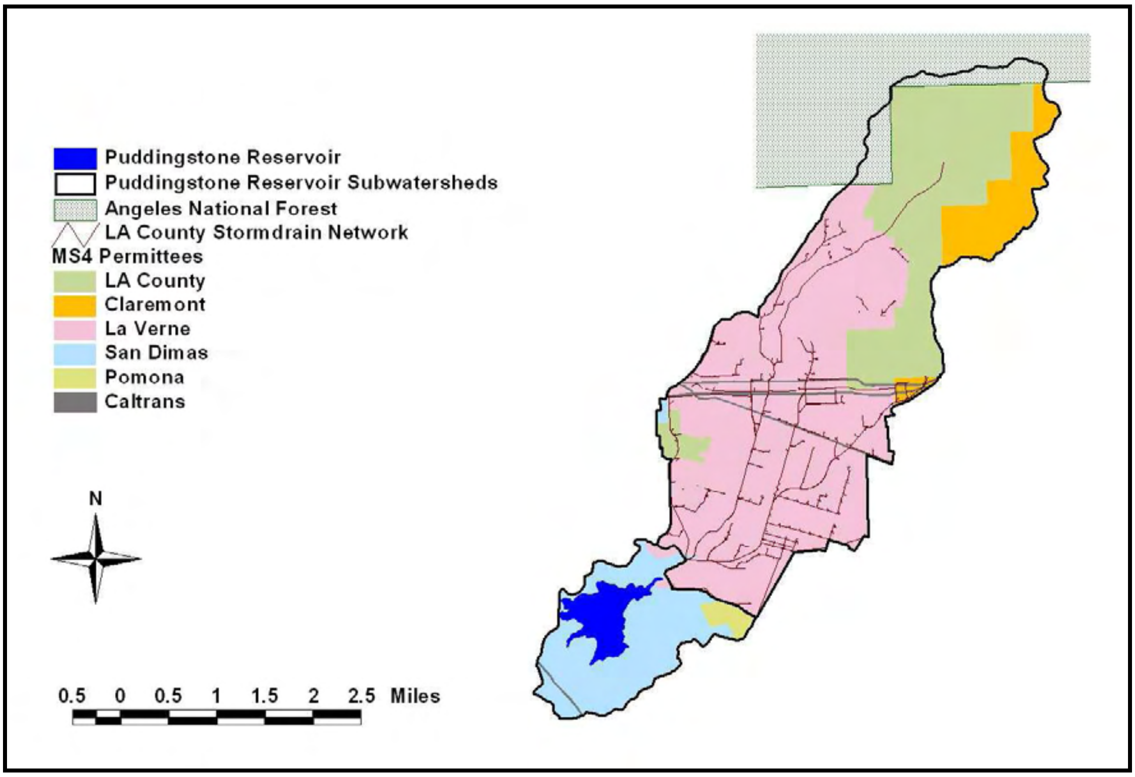


Figure D-34. MS4 Permittees and the Los Angeles County Storm Drain Network in the Puddingstone Reservoir Subwatersheds

Land uses identified in the San Gabriel River LSPC model are shown in Figure D-35. Upon review of the SCAG 2005 database as well as current satellite imagery, it was evident that some of the areas classified by the LSPC model as agriculture or strip mines were inaccurate. Land use classifications were changed to accurately reflect the conditions identified in the more recent data. Specifically, the strip mine area in the northern basin (271 ac) was modified as it is currently in residential development; a portion of the agricultural lands in the watershed were changed to either residential or mixed rangeland; and the reservoir identified in the northern basin is a flood control structure that is essentially vacant land based on the aerial, so this area was assigned to mixed rangeland. The “other urban or built-up” areas in the southern subwatershed were reclassified because review of aerial imagery indicates that these areas are currently parkland surrounding the reservoir; therefore, they were simulated as commercial areas with zero percent imperviousness (see discussion in Section D.3). Inaccuracies in land use assignment were corrected for each subwatershed and jurisdiction to reflect the more recent SCAG 2005 dataset and current satellite imagery. All areas within the Caltrans jurisdiction were simulated as transportation. Table D-45 and Table D-46 summarize the land use areas used to estimate pollutant loading from upland areas draining to Puddingstone Reservoir.

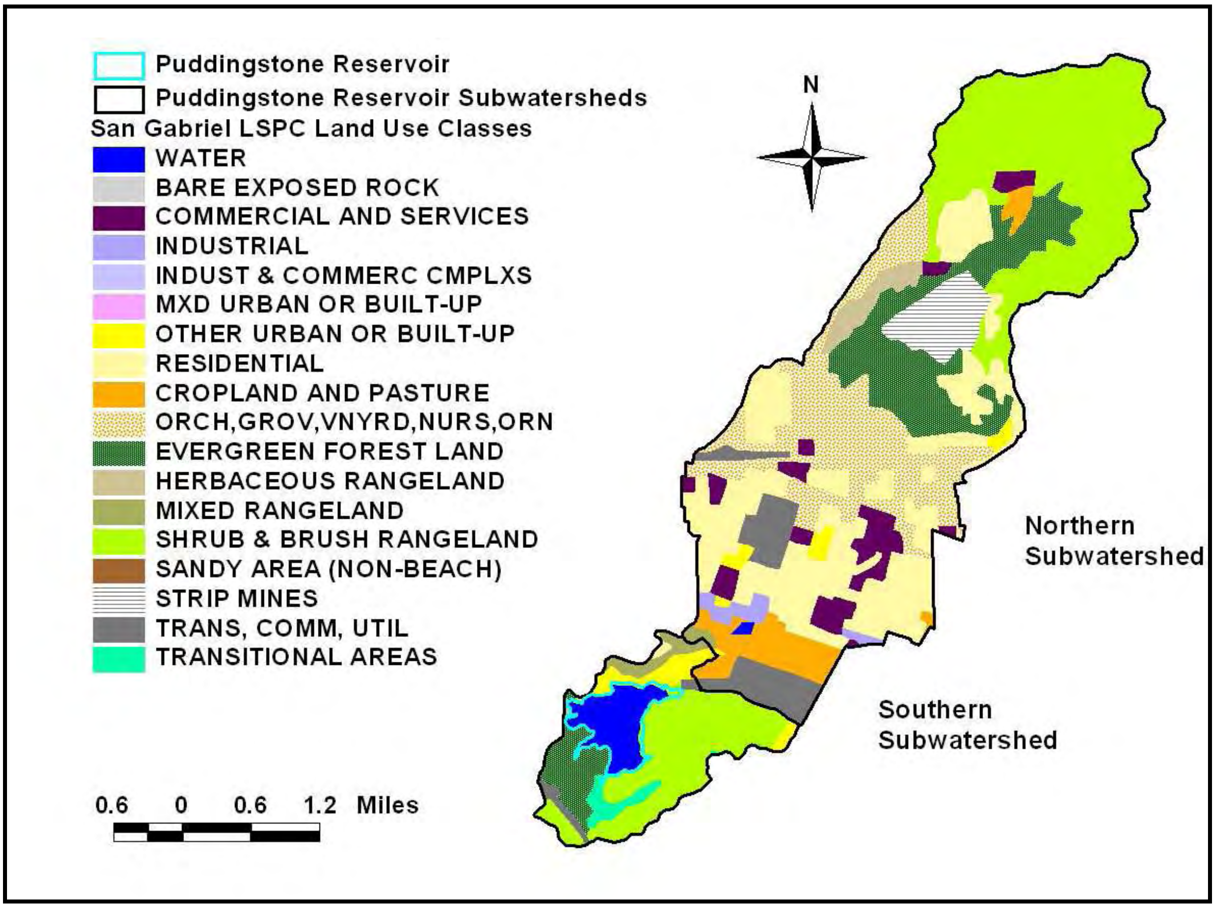


Figure D-35. LSPC Land Use Classes for the Puddingstone Reservoir Subwatersheds

Table D-45. Land Use Areas (ac) Draining from the Northern Subwatershed of Puddingstone Reservoir

Land Use	Claremont	County of Los Angeles	La Verne	Pomona	San Dimas	Caltrans	Angeles National Forest	Total
Commercial and services	0	38.8	295	0.291	11.0	0	**0**	**345**
Cropland and pasture	2.91	22.5	199	0	0	0	**0**	**225**
Evergreen forest land	42.9	378	376	0	0	0	**0**	**797**
Herbaceous rangeland	0	0	123	0	0	0	**0**	**123**
Industrial	0	0	82.3	0	0	0	**0**	**82.3**
Mixed rangeland	0	21.5	111	1.08	1.95	0	**0**	**135**
Other urban or built-up	8.07	9.24	58.2	0.005	2.90	0	**0**	**78.4**
Residential	28.4	467	2,469	0.260	10.0	0	**0**	**2,975**
Shrub & brush rangeland	496	926	19.7	0.097	0.53	0	**293**	**1,736**
Transportation, communications, utilities	0	0.97	346	3.55	2.12	110	**0**	**463**
Transitional areas	0	0	0	0	0	0	**0**	**0**
Total	**578**	**1,865**	**4,079**	**5.28**	**28.5**	**110**	**293**	**6,959**

Table D-46. Land Use Areas (ac) Draining from the Southern Subwatershed of Puddingstone Reservoir

Land Use	La Verne	Pomona	San Dimas	Caltrans	Total
Commercial and services	0	0	0	0	**0**
Cropland and pasture	0	0	0	0	**0**
Evergreen forest land	0	0	184	0	**184**
Herbaceous rangeland	0	0	4.33	0	**4.33**
Industrial	0	0	0	0	**0**
Mixed rangeland	23.7	0	48.5	0	**72.2**
Other urban or built-up	1.35	19.1	101	0	**122**
Residential	0	0	10.7	0	**10.7**
Shrub & brush rangeland	0.006	62.1	602	0	**664**
Transportation, communications, utilities	8.44	0.616	23.0	11.6	**43.6**
Transitional areas	0	0	68.2	0	**68.2**
Total	**33.5**	**81.8**	**1,042**	**11.6**	**1,169**

D.10.1 RUNOFF

LSPC-predicted runoff from the Puddingstone Reservoir subwatersheds is primarily driven by the land use and soil characteristics of the drainage area and the nearest meteorological station represented in the model. Figure D-36 shows the simulated annual rainfall for the Puddingstone Reservoir subwatersheds. The annual average rainfall is 17.4 inches.

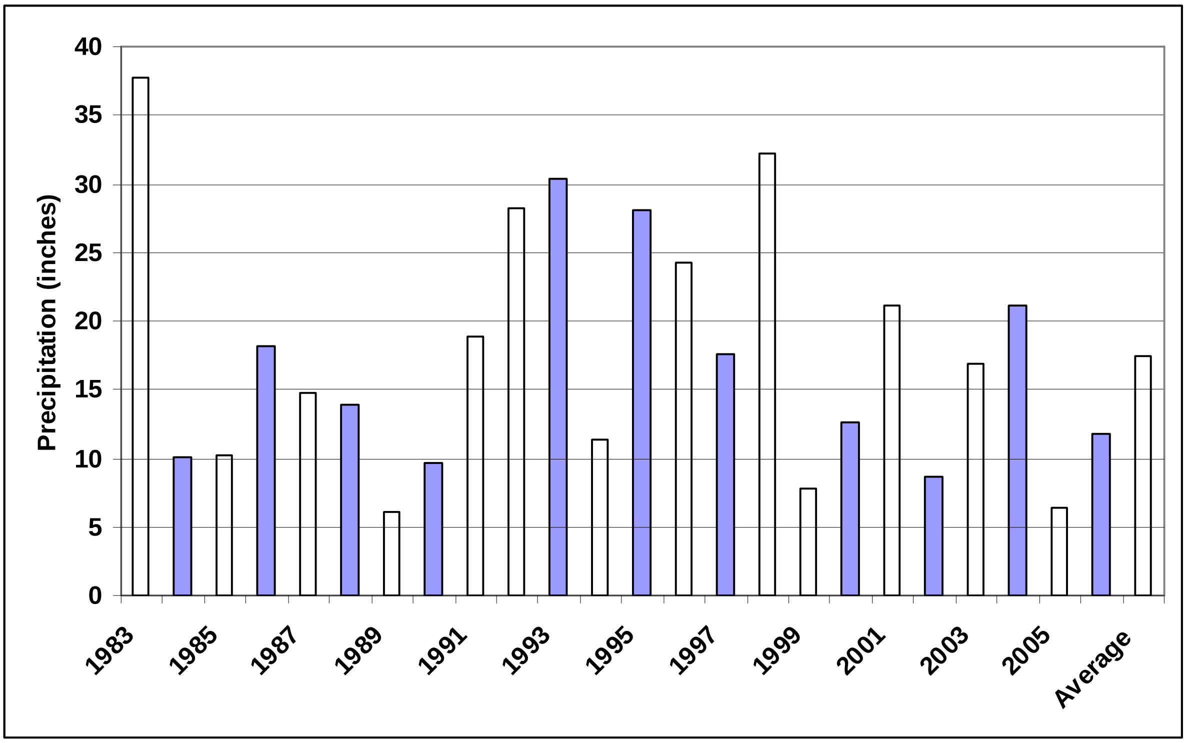


Figure D-36. Annual Rainfall for the Puddingstone Reservoir Subwatersheds

The simulated monthly average runoff depths for land uses in the Puddingstone Reservoir subwatersheds are shown in Table D-47.

Table D-47. Monthly Average Runoff Depths (inches/month) for Land Uses in the Puddingstone Reservoir Subwatersheds, 1983 - 2006

Land Use	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Commercial and services	2.545	3.469	2.162	0.777	0.227	0.133	0.040	0.040	0.124	0.624	1.029	1.636
Cropland and pasture	0.161	0.217	0.228	0.147	0.110	0.085	0.078	0.073	0.071	0.074	0.078	0.093
Evergreen forest land	0.150	0.204	0.214	0.138	0.104	0.081	0.074	0.069	0.068	0.070	0.074	0.086
Herbaceous rangeland	0.161	0.217	0.228	0.147	0.110	0.085	0.078	0.073	0.071	0.074	0.078	0.093
Industrial	2.576	3.501	2.198	0.807	0.251	0.154	0.059	0.058	0.141	0.643	1.050	1.660

Land Use	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Mixed rangeland	0.161	0.217	0.228	0.147	0.110	0.085	0.078	0.073	0.071	0.074	0.078	0.093
Other urban or built-up	2.104	2.866	1.807	0.664	0.208	0.127	0.049	0.048	0.115	0.523	0.853	1.350
Residential	0.737	1.002	0.699	0.305	0.144	0.100	0.072	0.068	0.086	0.208	0.309	0.465
Shrub & brush rangeland	0.161	0.217	0.228	0.147	0.110	0.085	0.078	0.073	0.071	0.074	0.078	0.093
Transportation, communications, utilities	2.545	3.469	2.162	0.777	0.227	0.133	0.040	0.040	0.124	0.624	1.029	1.636
Transitional areas	0.471	0.638	0.484	0.236	0.132	0.096	0.077	0.072	0.081	0.147	0.202	0.293
Parkland*	0.192	0.252	0.267	0.175	0.129	0.097	0.088	0.082	0.080	0.085	0.092	0.112

*Previously “other urban or built-up” areas in the southern subwatershed (see discussion in Section D.3).

Figure D-37 summarizes the monthly average runoff volumes delivered to Puddingstone Reservoir from 1983 through 2006. The total annual runoff to the reservoir is 2,692 ac-ft. The months May through October each contribute less than 5 percent of the annual runoff volume.

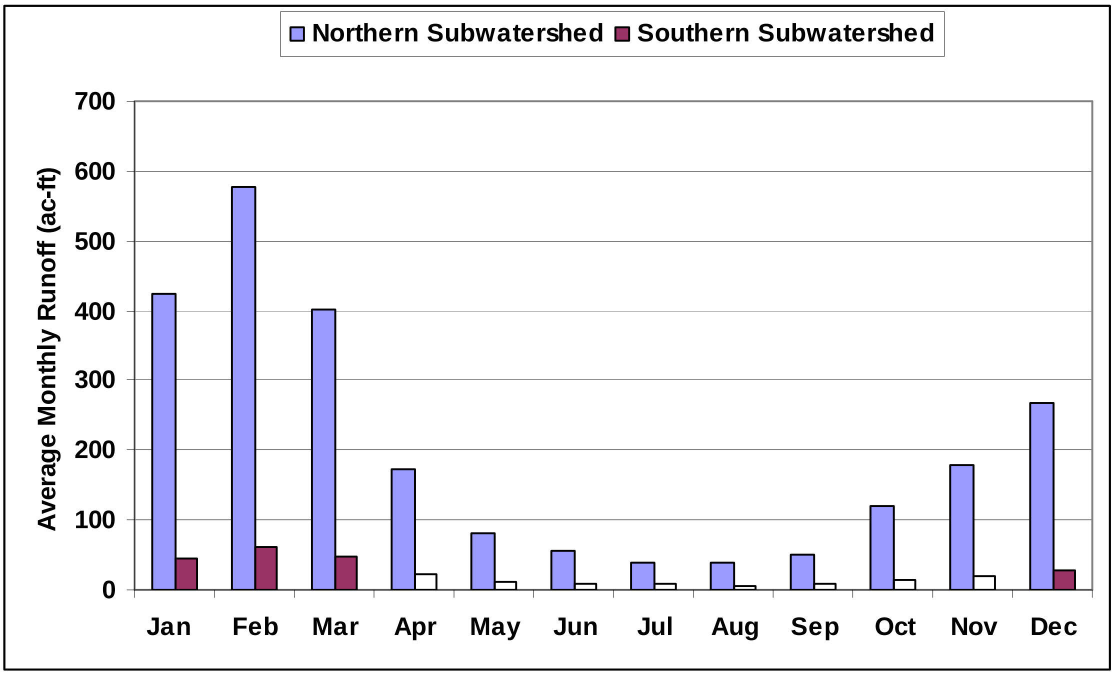


Figure D-37. Monthly Average Runoff Volumes to Puddingstone Reservoir (1983-2006)

Though the Metropolitan Water District can divert water to Puddingstone Reservoir from outside the watershed, this practice is seldom used (personal communication, Adam Walden, Los Angeles County Department of Public Works, 9/16/09) and does not impact the average conditions for this reservoir.

D.10.2 SEDIMENT LOADS

Sediment loads associated with upland areas are calculated from simulated runoff volumes and suspended sediment event mean concentrations for each modeled land use (Section D.3). Table D-48 summarizes the average annual sediment loads for each jurisdiction by subwatershed. See example calculations in Section D.3.

Table D-48. Average Annual Sediment Loads to Puddingstone Reservoir

Subwatershed	Jurisdiction	Sediment (tons/yr)
Northern	Caltrans	13.5
Northern	Claremont	4.49
Northern	County of Los Angeles	27.7
Northern	La Verne	197
Northern	Pomona	0.473
Northern	San Dimas	1.63
Northern	Angeles National Forest	1.36
Southern	Caltrans	1.42
Southern	La Verne	1.24
Southern	Pomona	1.68
Southern	San Dimas	14.8
Total		**266**

Sedimentation data collected by the USACE from 1925 to 1980 indicate that approximately 31 acre-feet per year (approximately 1.5 inches per year) have been delivered to Puddingstone Reservoir in the past (Department of Boating and Waterways and State Coastal Conservancy, 2002). Measurements occurred in 10- to 20-year increments that likely captured anomalous events such as flooding and fires followed by precipitation that typically result in mass wasting of sediment. In addition, rates were measured during periods of rapid development when the use of erosion control practices on construction sites was uncommon. During this development period, natural channels were replaced with hardened structures, decreasing sediment loading associated with channel erosion. Though these sediment loads have impacted Puddingstone Reservoir in the past, they are not considered to represent average current conditions (the average annual sediment load of 266 tons/year is equivalent to 0.00465 inches per year; Table D-48). Also, large pulses of sediment are likely delivered during a few events with much of the associated pollutant loading quickly buried and sequestered and therefore unavailable for release to the water column or entrance to the food chain via benthic organisms. Thus, no additional pollutant loads were assumed for mass wasting events.

D.10.3 NUTRIENT LOADS

Nutrient loads from upland areas are estimated from event mean concentration data collected by SCCWRP and the county of Los Angeles (Section D.3). Table D-49 summarizes the total nitrogen and total phosphorus loads delivered to Puddingstone Reservoir from each jurisdiction and subwatershed. See example calculations in Section D.3.

Table D-49. Average Annual Nutrient Loads to Puddingstone Reservoir

Subwatershed	Jurisdiction	Nitrogen (lb/yr)	Phosphorus (lb/yr)
Northern	Caltrans	1,409	214
Northern	Claremont	766	52.3
Northern	County of Los Angeles	4,011	467
Northern	La Verne	21,698	3,254
Northern	Pomona	51.6	7.71
Northern	San Dimas	244	37.2
Northern	Angeles National Forest	301	10.3
Southern	Caltrans	148	22.5
Southern	La Verne	147	19.4
Southern	Pomona	276	34.5
Southern	San Dimas	2,433	272
Total		**31,484**	**4,391**

D.10.4 MERCURY LOADS

Mercury loads resulting from upland areas are based on monitoring data collected by the Regional Board and USEPA during the winter and summer of 2009. Water column mercury concentrations measured from major inputs to the lakes are applied to simulated runoff volumes and input sediment mercury concentrations are applied to the calculated sediment loads (Section D.10.2) to estimate water column and sediment associated mercury loads, respectively. Figure D-38 shows the locations of the monitoring stations in the Puddingstone Reservoir watershed. Stations PR19 and PR19SD are in close proximity and display as one yellow square in Figure D-38.

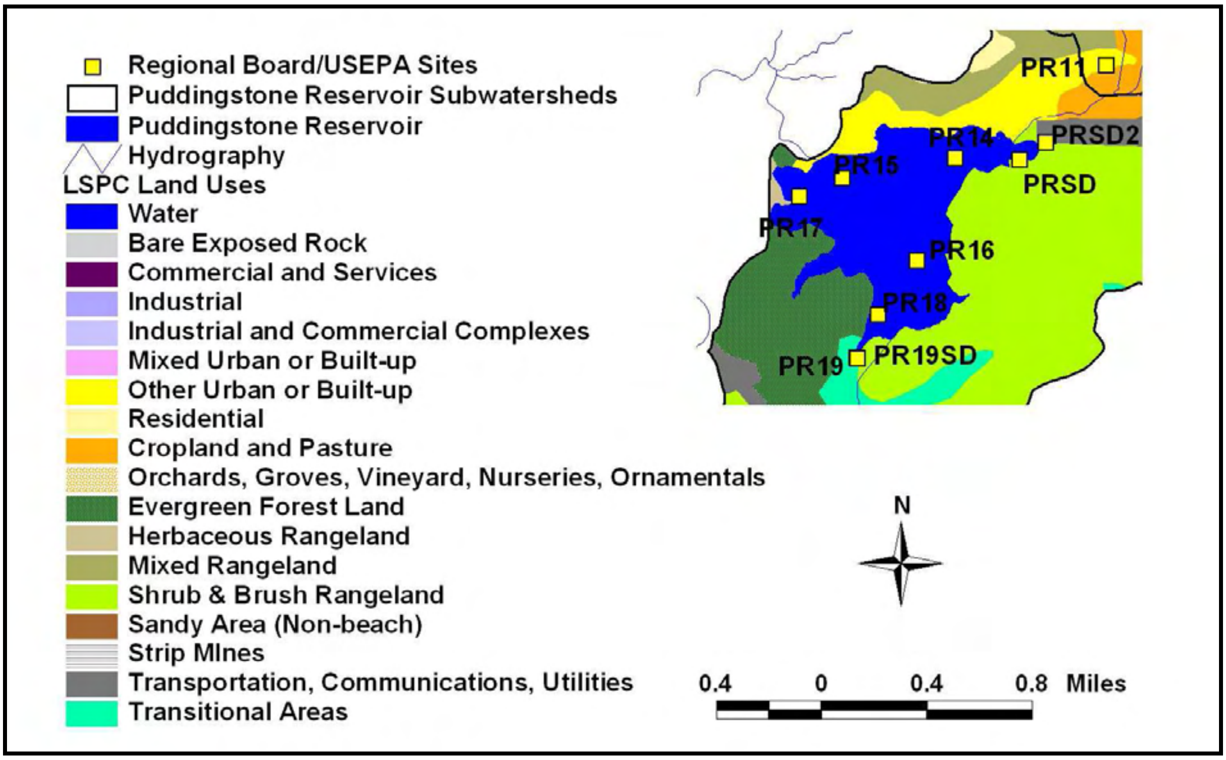


Figure D-38. Monitoring Stations in the Puddingstone Reservoir Watershed

Table D-50 and Table D-51 present the methyl and total mercury concentrations observed at the mouth of each major input in the water column and sediments, respectively. More details regarding these data are presented in Appendix G (Monitoring Data).

Table D-50. Tributary/Inflow Mercury Water Column Measurements for Puddingstone Reservoir

Location	Date	Time	MeHg (ng/L)	Total Hg (ng/L)
PR11	2/24/2009	14:30	0.043	3.52
PRSD		13:10	<0.020	2.65
PR11	7/16/2009	11:45	0.553	4.24
PRSD2		13:10	0.046	7.55

Table D-51. Inflow Mercury Sediment Concentrations for Puddingstone Reservoir

Location	Date	Time	MeHg (µg/kg)	Total Hg (µg/kg)
PR11	2/24/2009	14:30	<0.011	52.9
PR11	7/16/2009	11:45	1.71	73.1
PR19		14:05	0.068	34.3
PR19SD		14:10	0.940	66.2
PRSD2		13:10	1.14	50.4

Concentrations of total and methylmercury vary seasonally at each input. Water column and sediment concentration data are available during both the summer and winter season at station PR11, which represents loading from the northern subwatershed. Mercury loading is associated with both sediment and runoff from upland areas and given the availability of seasonal data, concentrations of total and methylmercury in water (Table D-50) and sediment (Table D-51) varied for the summer and winter. To determine sediment loading of mercury, the sediment EMCs and LSPC predicted runoff volumes were used to calculate sediment loads, and the sediment mercury concentrations from monitoring data (Table D-51) were then applied to these sediment loads. Mercury loading associated with runoff from upland areas was calculated by applying the mercury water column concentrations (Table D-50) to LSPC simulated runoff volumes (Section D.3 provides examples of these calculations). The July 2009 monitoring data were used to estimate loads for the summer season (May through October), and the February 2009 data were used to estimate loads for the winter season (November through April). The sediment methlymercury concentration was below the detection limit for the winter sampling so it was assumed equal to one-half the detection limit, or 0.006 µg/kg (Table D-51).

In the southern subwatershed, similar calculations were performed to estimate mercury loading associated with runoff and sediment; however, additional monitoring stations were available to represent the loading throughout this subwatershed. Specifically, water column concentrations of methyl and total mercury were measured at two storm drain outlets located near the campground in the northeastern section of the watershed. Each drain was measured during only one season. Measurements at PRSD were used to estimate winter concentrations, and measurements at PRSD2 were used to estimate summer concentrations. The winter methylmercury concentration in water was below the detection limit so it was assumed to be equal to one-half the detection limit, or 0.01 ng/L (Table D-50). Mercury sediment concentrations for inlets located in the southern subwatershed were measured only during the summer season. The summer season total mercury sediment concentration was assumed equal to the average of the concentrations measured at PR19, PR19SD, and PRSD2 (50.3 µg/kg; Table D-51). The winter season total mercury sediment concentration (36.4 µg/kg) was assumed equal to the summer concentration divided by the ratio of summer to winter total mercury sediment concentrations observed at PR11 (73.1 µg/kg ÷ 52.9 µg/kg = ratio of 1.38; Table D-51). Similar assumptions were used to estimate the summer and winter methylmercury sediment concentrations applicable to the southern subwatershed.

The assumed concentrations were applied to the runoff and sediment loads estimated from each jurisdiction within the watershed. Assumed total mercury concentrations and resulting loads are summarized in Table D-52. See example calculations in Section D.3. Results for methylmercury are presented in Table D-53. Approximately 92 percent of the wet weather total mercury load and 98 percent of the wet weather methylmercury load originate in the northern subwatershed, which accounts for 86 percent of the watershed area.

Table D-52. Total Mercury Loads Estimated for Each Jurisdiction and Subwatershed in the Puddingstone Reservoir Watershed

Sub-watershed	Jurisdiction	Area (ac)	Summer Water Column Hg (ng/L)	Winter Water Column Hg (ng/L)	Summer Sediment Hg (µg/kg)	Winter Sediment Hg (µg/kg)	Annual Water Column Hg Load (g/yr)	Annual Sediment Hg Load (g/yr)	Total Annual Hg Load (g/yr)
Northern	Caltrans	110	4.24	3.52	73.1	52.9	0.520	0.672	1.19
Northern	Claremont	578	4.24	3.52	73.1	52.9	0.372	0.239	0.611
Northern	County of Los Angeles	1,865	4.24	3.52	73.1	52.9	1.68	1.45	3.13
Northern	La Verne	4,079	4.24	3.52	73.1	52.9	7.97	10.1	18.1
Northern	Pomona	5.28	4.24	3.52	73.1	52.9	0.019	0.024	0.043
Northern	San Dimas	28.5	4.24	3.52	73.1	52.9	0.090	0.082	0.172
Northern	Angeles National Forest	293	4.24	3.52	73.1	52.9	0.161	0.074	0.234
Southern	Caltrans	11.6	7.55	2.65	50.3	36.4	0.047	0.049	0.096
Southern	La Verne	33.5	7.55	2.65	50.3	36.4	0.054	0.043	0.097
Southern	Pomona	81.8	7.55	2.65	50.3	36.4	0.108	0.058	0.166
Southern	San Dimas	1,043	7.55	2.65	50.3	36.4	1.05	0.522	1.57
Total		**8,128**	**N/A**	**N/A**	**N/A**	**N/A**	**12.1**	**13.3**	**25.4**

N/A = Not applicable

Table D-53. Methylmercury Loads Estimated for Each Jurisdiction and Subwatershed in the Puddingstone Reservoir Watershed

Sub-watershed	Jurisdiction	Area (ac)	Summer Water Column MeHg (ng/L)	Winter Water Column MeHg (ng/L)	Summer Sediment MeHg (µg/kg)	Winter Sediment MeHg (µg/kg)	Annual Water Column MeHg Load (g/yr)	Annual Sediment MeHg Load (g/yr)	Total Annual MeHg Load (g/yr)
Northern	Caltrans	110	0.553	0.043	1.710	0.006	1.31E-02	2.01E-03	1.51E-02
Northern	Claremont	578	0.553	0.043	1.710	0.006	1.96E-02	2.00E-03	2.16E-02
Northern	County of Los Angeles	1,865	0.553	0.043	1.710	0.006	7.34E-02	9.95E-03	8.34E-02
Northern	La Verne	4,079	0.553	0.043	1.710	0.006	2.52E-01	5.65E-02	3.08E-01
Northern	Pomona	5.28	0.553	0.043	1.710	0.006	5.09E-04	7.31E-05	5.82E-04
Northern	San Dimas	28.5	0.553	0.043	1.710	0.006	2.48E-03	2.77E-04	2.75E-03
Northern	Angeles National Forest	293	0.553	0.043	1.710	0.006	9.38E-03	7.34E-04	1.01E-02
Southern	Caltrans	11.6	0.046	0.010	0.716	0.002	2.03E-04	8.84E-05	2.92E-04
Southern	La Verne	33.5	0.046	0.010	0.716	0.002	2.46E-04	9.55E-05	3.41E-04
Southern	Pomona	81.8	0.046	0.010	0.716	0.002	5.00E-04	1.57E-04	6.58E-04
Southern	San Dimas	1,043	0.046	0.010	0.716	0.002	5.01E-03	1.70E-03	6.70E-03
Total		**8,128**	**N/A**	**N/A**	**N/A**	**N/A**	**0.376**	**0.074**	**0.450**

N/A = Not applicable

D.10.5 ORGANOCHLORINE PESTICIDES AND PCBS LOADS

The existing loading rates from upland areas for OC Pesticides and PCBs are estimated for each pollutant using monitoring data collected by USEPA, the Regional Board, and UCLA between 2008 and 2009. Only data from sites representing inflows are used; these include locations in an inflow or in the lake near an inflow. Inflows considered for wet weather loading were tributaries, drainage paths, and channels. For Puddingstone Reservoir, this included PR-11, PR-19, PR-19SD, and PR-SD2 (Figure D-39).

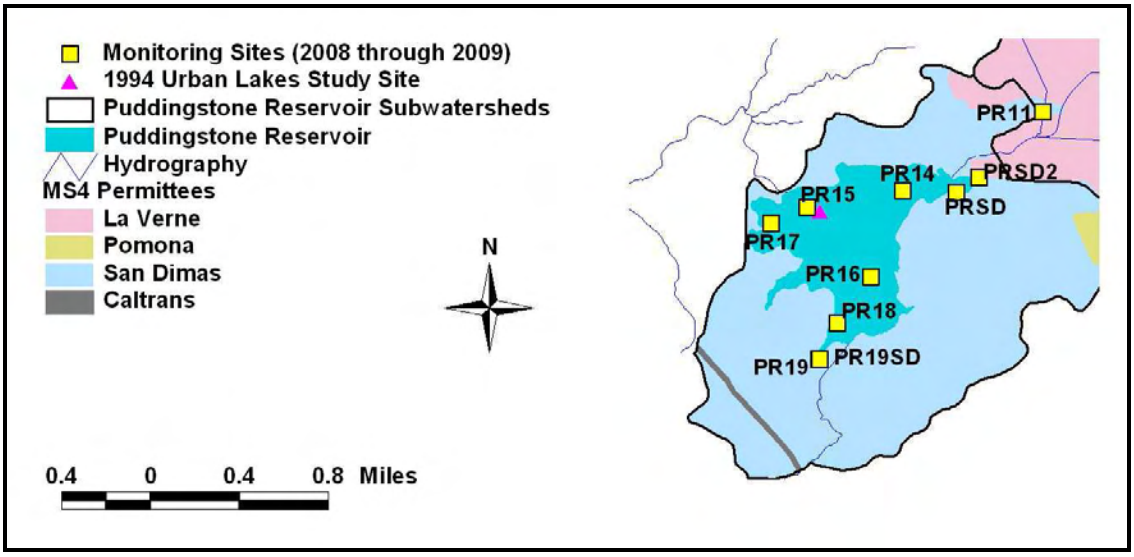


Figure D-39. Puddingstone Reservoir Monitoring Stations

The OC Pesticides and PCBs of concern are not currently in use and are more likely to have been historically loaded to the lake sediments; therefore, current tributary loading is likely to be small. The OC Pesticides and PCBs are hydrophobic and the majority of the pollutant mass in wet weather loads were associated with the sediment. The measured levels of OC Pesticides and PCBs in inflow sediments were the only data that could be used to quantify current inflow loads because nearly all of the water column, porewater, suspended sediment, and suspended sediment in porewater samples did not yield reportable results. For OC Pesticides and PCBs where some of the samples had detectable quantities of a pollutant, the average inflow concentration was calculated assuming samples analyzed below detection limits were equal to one-half the detection limit. All dieldrin samples were below detection limits; for dieldrin the concentration was calculated directly from the detection limits. The inflow sediment data are summarized in Table D-54 and all data collected in the watershed are discussed in detail in Appendix G (Monitoring Data).

Table D-54. Summary of Sediment Data near Inflow Locations for Puddingstone Reservoir

Parameter	No. of Samples	Number of Samples Above Detection Limits[1]	Average Concentration (µg/kg dry weight)	Detection Limit (µg/kg dry weight)
Chlordane	3	3	5.11	0.39-1.58
DDT	3	2	5.50	0.77-3.17
Total PCBs	3	3	50.3	0.39-1.58
Dieldrin	3	0	< 1.0	1.0

[1] Non-detect samples were included in reported averages as one-half of the detection limit.

These input sediment concentrations were applied to the calculated sediment loads (Section D.10.2) to estimate the sediment-associated OC Pesticides and PCBs loads entering the lake. Sediment loads and subsequently calculated OC Pesticides and PCBs loads were determined for each jurisdiction based on the land use types and areas within each subwatershed. The jurisdictional areas are presented for the two Puddingstone Reservoir subwatersheds in Table D-55 along with the predicted sediment loads for each land use. Dissolved concentrations in inflows are assumed insignificant.

Table D-55. Annual Sediment Load for Puddingstone Reservoir

Subwatershed	Jurisdiction	Area (ac)	Annual Sediment Load (tons/yr)	Percent of Total Load
Northern	Caltrans	110	13.5	5.10%
Northern	Claremont	578	4.5	1.69%
Northern	County of Los Angeles	2,056	27.7	10.43%
Northern	La Verne	4,181	168	63.23%
Northern	General Industrial Stormwater Permittees* (in the city of La Verne)	1,865	24.8	9.33%
Northern	General Construction Stormwater Permittees (in the city of La Verne)	4,079	4.5	1.70%
Northern	Pomona	5.28	0.5	0.18%
Northern	San Dimas	28.5	1.6	0.62%
Northern	Angeles National Forest	293	1.4	0.51%
Southern	Caltrans	11.6	1.4	0.54%
Southern	La Verne	33.5	1.2	0.47%
Southern	Pomona	81.8	1.7	0.63%
Southern	San Dimas	1,042	14.8	5.59%
Southern	County of Los Angeles (Irrigation)		0.0	0.00%
Total		**8,128**	**265.5**	**100%**

*The disturbed area associated with general construction and general industrial stormwater permittees was subtracted out of the appropriate city area and allocated to these permits.

The chlordane, PCB, DDT, and dieldrin loads were calculated by applying the input sediment concentrations (Table D-54) to the calculated sediment loads (Table D-55; 265.5 tons per year). See example calculations in Section D.3. The dieldrin calculation is based on the detection limit of 1 μg/kg dry weight. Loads for each jurisdiction are shown by subwatershed in Table D-56.

Table D-56. Total Organic Loads Estimated for Each Jurisdiction and Subwatershed in the Puddingstone Watershed (g/yr)

Subwatershed	Jurisdiction	Annual PCB Load	Annual Chlordane Load	Annual Dieldrin Loads	Annual DDT Load	Percent of Total Load
Northern	Caltrans	0.62	0.063	0.012	0.068	5.10%
Northern	Claremont	0.20	0.021	0.004	0.022	1.69%
Northern	County of Los Angeles	1.30	0.128	0.025	0.138	10.43%
Northern	La Verne	7.68	0.778	0.152	0.838	63.23%
Northern	General Industrial Stormwater Permittees* (in the city of La Verne)	1.13	0.115	0.022	0.124	9.33%
Northern	General Construction Stormwater Permittees (in the city of La Verne)	0.21	0.021	0.004	0.022	1.69%
Northern	Pomona	0.02	0.002	0.000	0.002	0.18%
Northern	San Dimas	0.07	0.008	0.001	0.008	0.62%
Northern	Angeles National Forest	0.06	0.006	0.001	0.007	0.51%
Southern	Caltrans	0.06	0.007	0.001	0.007	0.54%
Southern	La Verne	0.06	0.006	0.001	0.006	0.47%
Southern	Pomona	0.08	0.008	0.001	0.008	0.63%
Southern	San Dimas	0.68	0.069	0.002	0.074	5.59%
Southern	County of Los Angeles (Irrigation)	0.00	0.000	0.013	0.00	0.00%
Total		**12.12**	**1.23**	**0.24**	**1.32**	**100%**

*The disturbed area associated with general construction and general industrial stormwater permittees was subtracted out of the appropriate city area and allocated to these permits.

D.11 Santa Fe Dam Park Lake

Santa Fe Dam Park Lake is located in the San Gabriel River Basin. Though the park lake is located near the Santa Fe Dam Diversion Channel, it does not receive any diverted flow from the San Gabriel River (personal communication, Arthur Gotingco, Los Angeles County Public Works Department, 7/13/2009).

Impairments of this lake include pH, copper, and lead. Output from the San Gabriel River LSPC model coupled with regional pollutant event mean concentrations have been used to estimate loads of nutrients from upland areas, which may be contributing to the pH impairment.

One subwatershed comprises the drainage area to Santa Fe Dam Park Lake. This subwatershed is comprised of 362 acres. No storm water sewer system is present in the watershed, so all allocations for the TMDLs were load allocations.

Figure D-40 shows the MS4 stormwater permittees in the Santa Fe Dam Park Lake watershed. Most of the area is located in Irwindale, with a small portion in Azusa.

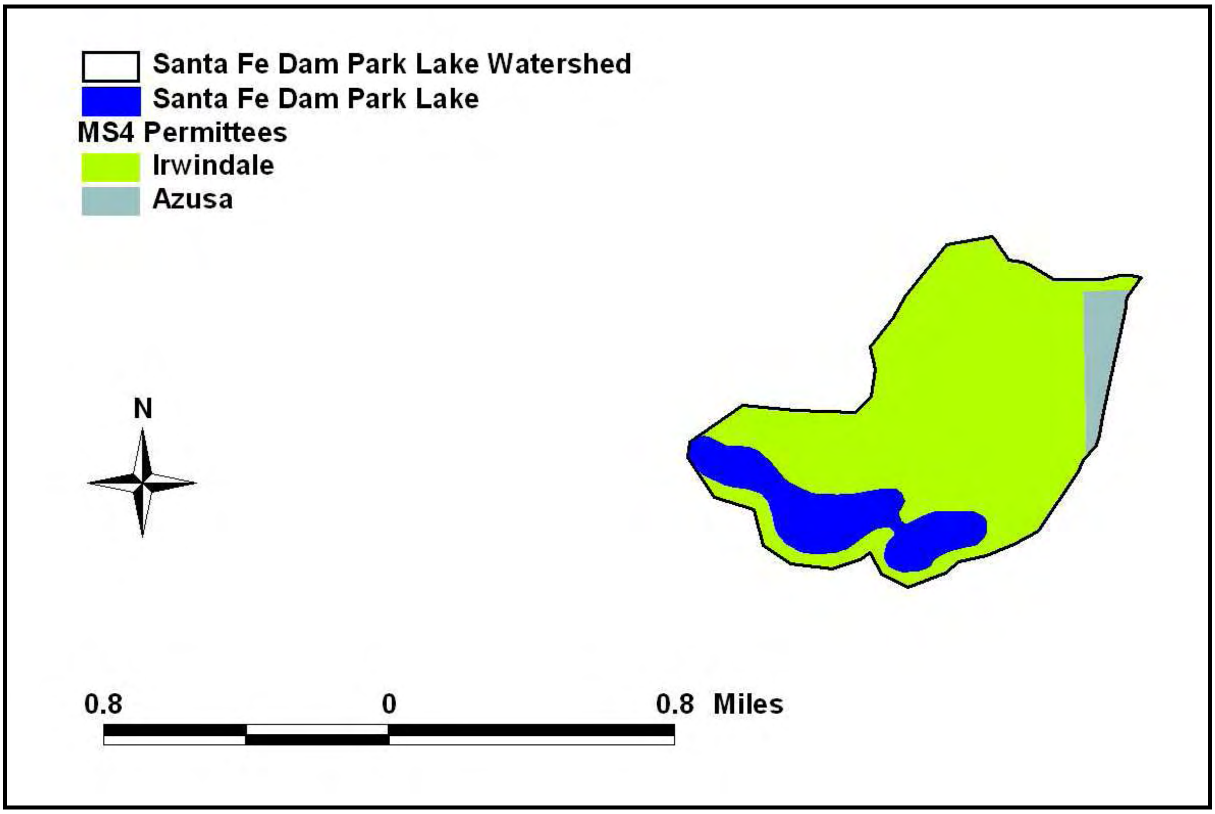


Figure D-40. MS4 Permittees in the Santa Fe Dam Park Lake Subwatershed

Land uses identified in the San Gabriel River LSPC model are shown in Figure D-41. Upon review of the SCAG 2005 database as well as current satellite imagery, it was evident that the portion of area classified by the LSPC model as strip mines had not been mined for some time. The SCAG 2005 database classified this area as vacant; the current satellite imagery shows this area to be re-established shrub/brush rangeland. Land use classifications were changed to accurately reflect the conditions identified in the more recent data. Specifically, the 6.25 acres classified by the LSPC model as strip mines were therefore converted to shrub and brush rangeland for this loading analysis. Table D-57 summarizes the post-processed land use areas used to estimate pollutant loading from upland areas draining to Santa Fe Dam Park Lake.

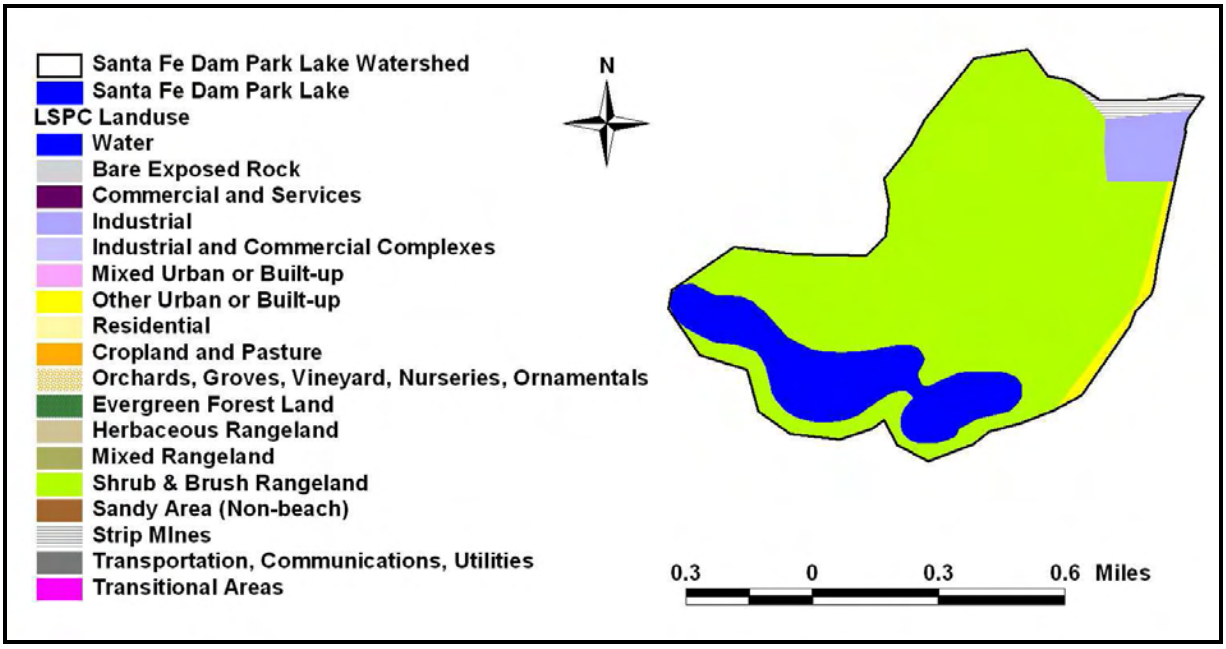


Figure D-41. LSPC Land Use Classes for the Santa Fe Dam Park Lake Subwatershed

Table D-57. Land Use Areas (ac) Draining to Santa Fe Dam Park Lake

Land Use	Azusa	Irwindale	Total
Industrial	11.5	7.16	**18.7**
Other urban or built-up	3.94	4.54	**8.48**
Shrub & brush rangeland	6.94	328	**335**
Total	**22.4**	**340**	**362**

D.11.1 RUNOFF

LSPC-predicted runoff from the Santa Fe Dam Park Lake subwatershed is primarily driven by the land use and soil characteristics of the drainage area and the nearest meteorological station represented in the model. Figure D-42 shows the simulated annual rainfall for the Santa Fe Dam Park Lake subwatershed. The annual average rainfall is 18.5 inches.

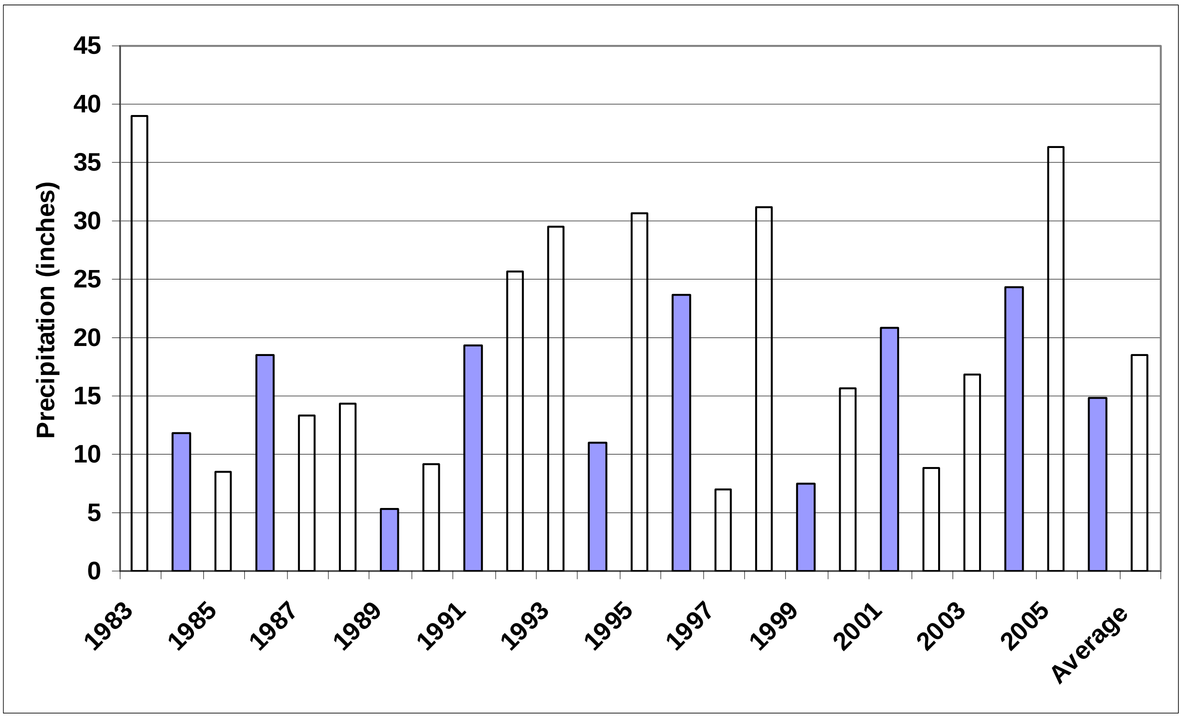


Figure D-42. Annual Rainfall for the Santa Fe Dam Park Lake Subwatershed

The simulated monthly average runoff depths for land uses in the Santa Fe Dam Park Lake subwatershed are shown in Table D-58.

Table D-58. Monthly Average Runoff Depths (inches/month) for Land Uses in the Santa Fe Dam Park Lake Subwatershed, 1983 - 2006

Month	Other Urban or Built Up	Heavy Industrial	Shrub/Brush Rangeland
January	2.2518	2.7522	0.0771
February	3.1443	3.8356	0.1393
March	1.8647	2.2633	0.1246
April	0.7178	0.8713	0.0462
May	0.2278	0.2735	0.0261
June	0.0950	0.1149	0.0076
July	0.0105	0.0121	0.0031
August	0.0504	0.0616	0.0018
September	0.1539	0.1890	0.0016
October	0.4457	0.5482	0.0019
November	0.7481	0.9202	0.0022
December	1.3029	1.6019	0.0061

Figure D-43 summarizes the monthly average runoff volumes delivered to Santa Fe Dam Park Lake. The total annual runoff to the reservoir is 40.9 ac-ft. The months May through October each contribute less than 5 percent of the annual runoff volume.

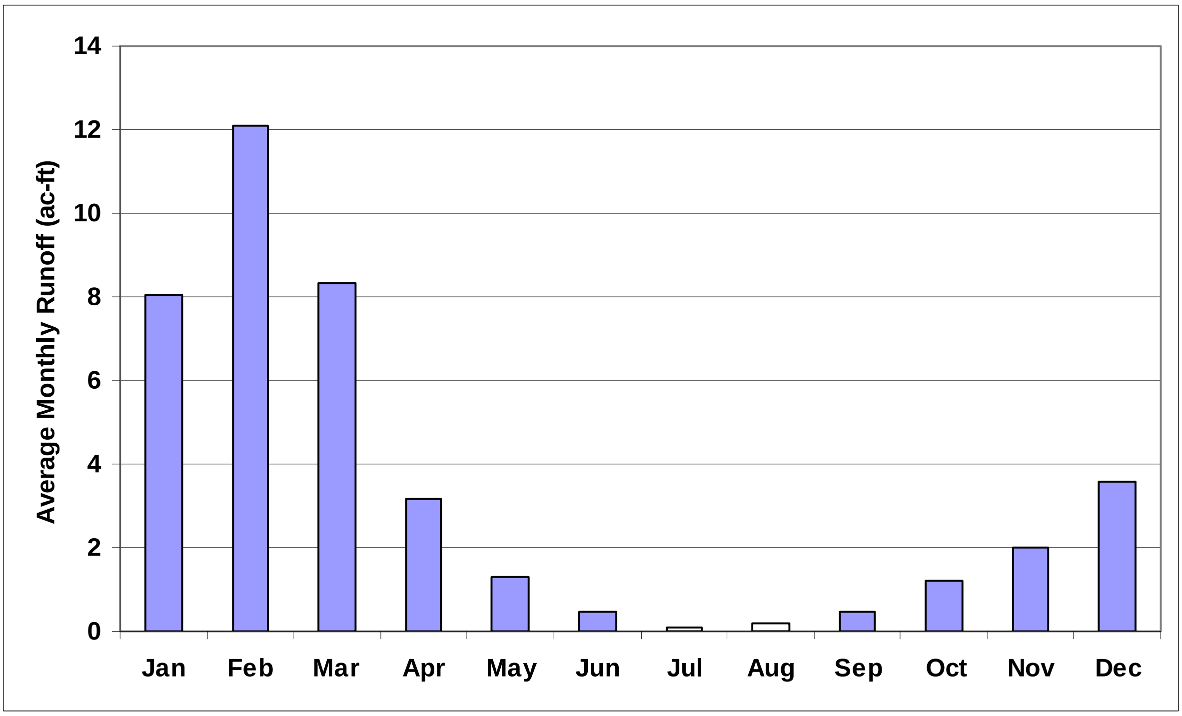


Figure D-43. Monthly Average Runoff Volumes to Santa Fe Dam Park Lake

D.11.2 NUTRIENT LOADS

Nutrient loads are estimated from simulated runoff volumes and event mean concentration data collected by SCCWRP and the county of Los Angeles (Section D.3). Table D-59 summarizes the total nitrogen and total phosphorus loads delivered to Santa Fe Dam Park Lake from each jurisdiction. See example calculations in Section D.3.

Table D-59. Average Annual Nutrient Loads to Santa Fe Dam Park Lake

Jurisdiction	Nitrogen (lb/yr)	Phosphorus (lb/yr)
Azusa	205	27.0
Irwindale	253	23.8
Total	**458**	**50.8**

D.12 Lake Sherwood

Lake Sherwood is located in the Santa Monica Bay Basin and is impaired by mercury (note: algae, ammonia, eutrophication, and low dissolved oxygen impairments have been addressed by a previous TMDL). For consistency with the other two mercury impaired lakes addressed by this TMDL (Puddingstone Reservoir and the El Dorado Park lakes), the upland mercury loads were calculated from tributary monitoring data collected in 2009 and estimates of runoff volumes and sediment loading predicted by an LSPC model. Though an LSPC model has not been developed for the Santa Monica Bay Basin, the land use coverage for the Los Angeles River Basin LSPC model covers the drainage area to Lake Sherwood and was used to predict runoff volumes and sediment loads by land use to Lake Sherwood.

Six subwatersheds comprise the drainage area (10,656 acres) to Lake Sherwood. Figure D-44 shows the MS4 stormwater permittees comprising each subwatershed. Ventura County is the only stormwater permittee in the Western Subwatershed. The Hidden Valley Wash subwatershed is mostly in Ventura County with small portion in Thousand Oaks. The Northern, Near Lake Undeveloped, and Near Lake Developed subwatersheds are comprised of both Ventura County and Thousand Oaks MS4 areas. The Carlisle Canyon subwatershed contains Ventura and Los Angeles County areas as well as Thousand Oaks, California Department of Transportation (Caltrans), and California State Park areas. Neither Ventura or Los Angeles counties (the MS4 stormwater permittees in the watershed) maintain storm drain systems in the Lake Sherwood watershed. However, there are residential developments in the vicinity of the lake which drain to culverts and storm drains. These areas are generally associated with the Sherwood Valley Homeowner's Association (SVHOA) and Sherwood Development Company. All subwatersheds will receive wasteload allocations except for the Carlisle Canyon and Near Lake Undeveloped subwatersheds. The small Caltrans area in the Carlisle Canyon subwatershed will also receive a wasteload allocation.

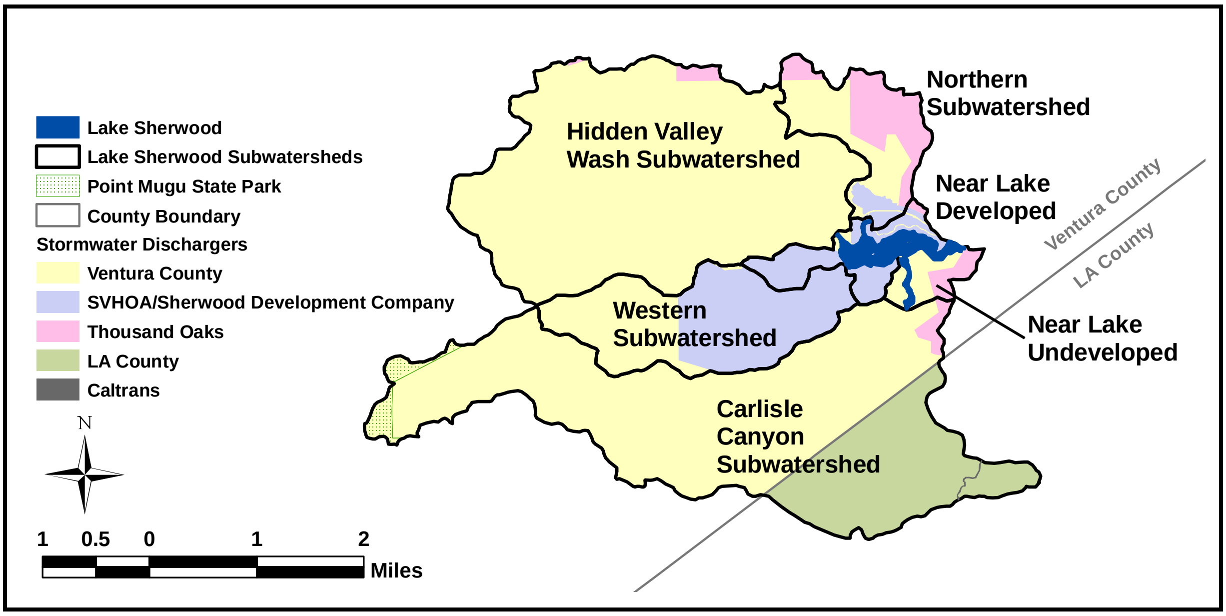


Figure D-44. MS4 Permittees in the Lake Sherwood Subwatersheds

Land uses identified in the Los Angeles River LSPC model are shown in Figure D-45. The watershed is comprised mostly of open space, agriculture, residential, and other urban areas. A single parcel of commercial development was identified in the Near Lake Developed Subwatershed. Review of SCAG 2005 land use data confirmed that much of the watershed is currently used for agriculture. The area in the Carlisle Canyon subwatershed under the Caltrans jurisdiction (Figure D-44) was simulated as industrial to estimate sediment loading and runoff volumes from the area associated with this State highway (i.e., changed from open to industrial land use). This was the only modification to the land use classifications for the Lake Sherwood subwatersheds. Table D-60 through Table D-65 summarize the land use areas used to estimate pollutant loading from upland areas draining to Lake Sherwood.

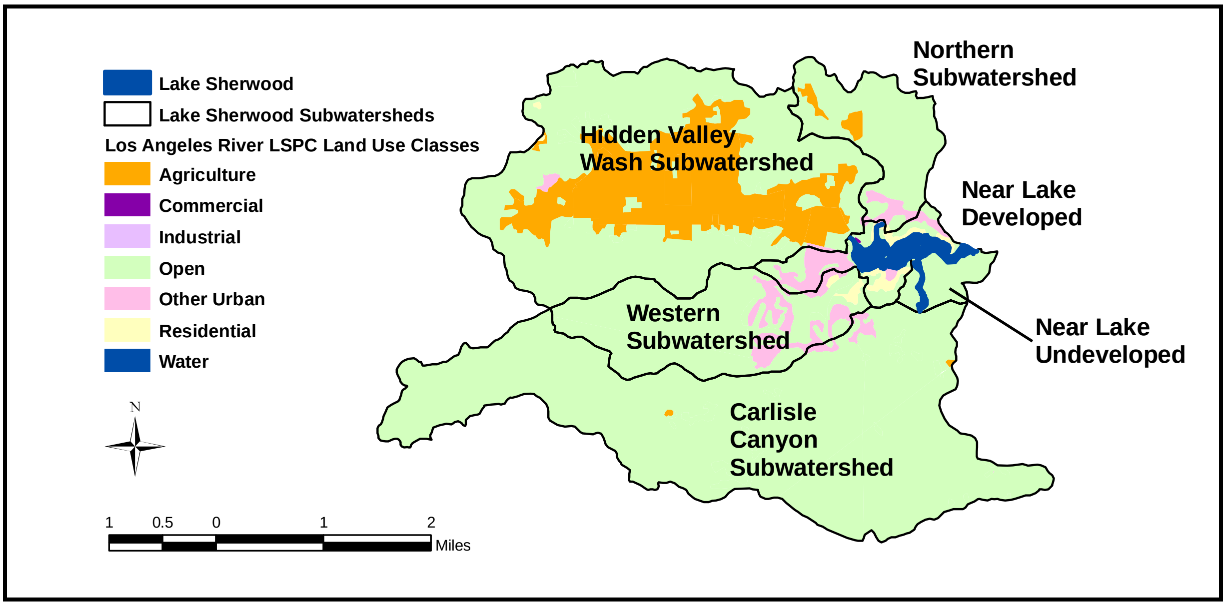


Figure D-45. LSPC Land Use Classes for the Lake Sherwood Subwatersheds

Table D-60. Land Use Areas (ac) Draining from the Northern Subwatershed to Lake Sherwood

Land Use	Ventura County	Thousand Oaks	SVHOA	Total
Agriculture	42	0	0	**42**
Commercial	0	0	0	**0**
Industrial	0	0	0	**0**
Open	301	338	29	**669**
Other Urban	7.2	0	34	**41**
Residential	0.20	0	2	**2**
Total	**351**	**338**	**65**	**754**

Table D-61. Land Use Areas (ac) Draining from the Hidden Valley Wash Subwatershed to Lake Sherwood

Land Use	Ventura County	Thousand Oaks	Total
Agriculture	1,328	0	**1,328**
Commercial	0	0	**0**
Industrial	0	0	**0**
Open	2,441	40.4	**2,482**
Other Urban	19.7	0	**20**
Residential	3.97	0	**4**
Total	**3,793**	**40.4**	**3,833**

Table D-62. Land Use Areas (ac) Draining from the Western Subwatershed to Lake Sherwood

Land Use	Ventura County	SVHOA	Total
Agriculture	0	0	**0**
Commercial	0	0	**0**
Industrial	0	0	**0**
Open	548	587	**1,136**
Other Urban	0	165	**165**
Residential	0	20	**20**
Total	**548**	**772**	**1,321**

Table D-63. Land Use Areas (ac) Draining from the Carlisle Canyon Subwatershed to Lake Sherwood

Land Use	Ventura County	Thousand Oaks	County of Los Angeles	Caltrans	Point Mugu State Park	Total
Agriculture	5.24	0	0.118	0	0	**5.36**
Commercial	0	0	0	0	0	**0**
Industrial	0	0	0	2.75	0	**2.75**
Open	2,866	50.4	1,149	0	101	**4,166**
Other Urban	34.2	0	0.06	0	0	**34**
Residential	0	0	0	0	0	**0**
Total	**2,905**	**50**	**1,149**	**2.75**	**101**	**4,209**

Table D-64. Land Use Areas (ac) Draining from the Near Lake Undeveloped Subwatershed to Lake Sherwood

Land Use	Ventura County	Thousand Oaks	Total
Agriculture	0	0	**0**
Commercial	0	0	**0**
Industrial	0	0	**0**
Open	126	70.9	**197**
Other Urban	0	0	**0**
Residential	0.004	0	**0.004**
Total	**126**	**70.9**	**197**

Table D-65. Land Use Areas (ac) Draining from the Near Lake Developed Subwatershed to Lake Sherwood

Land Use	Ventura County	Thousand Oaks	SVHOA	Total
Agriculture	0	0	0	**0.0**
Commercial	1.13	0	0	**1.1**
Industrial	0	0	0	**0**
Open	15	8.8	143	**167**
Other Urban	3.3	0	110	**113**
Residential	4.4	0	57	**61**
Total	**24**	**8.8**	**310**	**343**

D.12.1 RUNOFF

LSPC-based runoff from the Lake Sherwood subwatersheds is primarily driven by the land use and soil characteristics of a nearby drainage area and the nearest meteorological station represented in the model. Figure D-46 shows the simulated annual rainfall for the Lake Sherwood subwatersheds. The annual average rainfall is 17.5 inches.

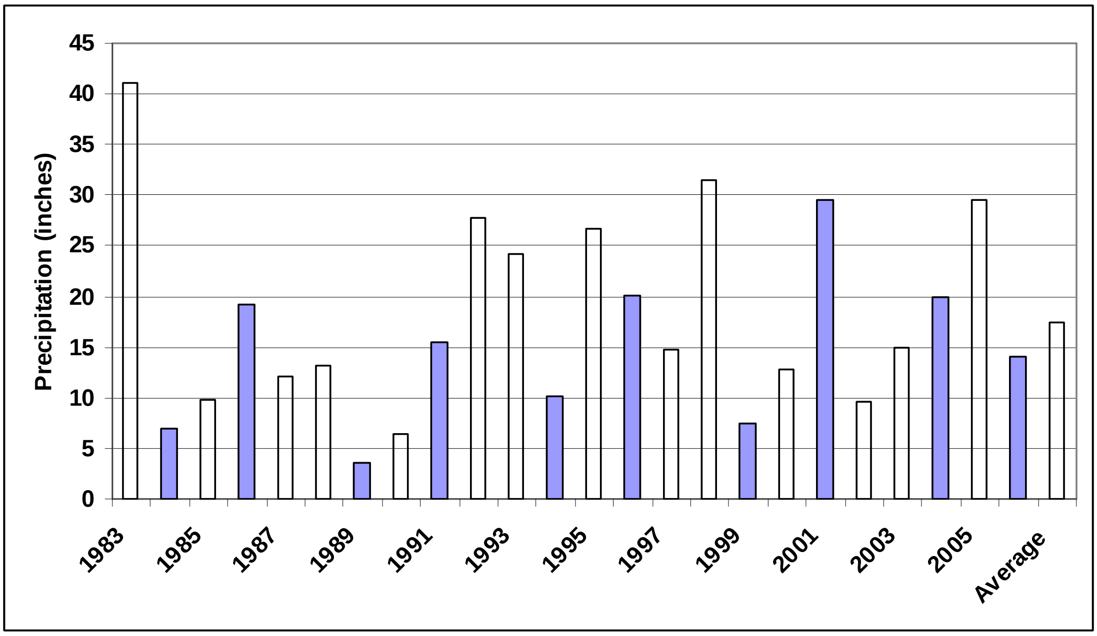


Figure D-46. Annual Rainfall for the Lake Sherwood Subwatersheds

The simulated monthly average runoff depths for land uses in the Lake Sherwood subwatersheds are shown in Table D-66.

Table D-66. Monthly Average Runoff Depths (in/mo) for Land Uses in the Lake Sherwood Subwatersheds, 1983 – 2006

Month	Agriculture	Commercial	Industrial	Open	Other Urban	Residential
January	0.3808	2.6527	2.3868	0.1271	1.7220	1.6687
February	0.8634	3.7749	3.4439	0.3202	2.6159	2.5495
March	0.5419	2.2277	2.0322	0.2219	1.5434	1.5042
April	0.1031	0.7350	0.6590	0.0473	0.4689	0.4536
May	0.0452	0.2213	0.2008	0.0174	0.1493	0.1452
June	0.0032	0.0224	0.0199	0.0020	0.0138	0.0134
July	0.0007	0.0037	0.0033	0.0005	0.0024	0.0023
August	0.0016	0.0205	0.0181	0.0009	0.0121	0.0116
September	0.0112	0.1552	0.1370	0.0056	0.0915	0.0878
October	0.0385	0.5420	0.4784	0.0192	0.3193	0.3065
November	0.0689	0.9656	0.8524	0.0342	0.5691	0.5464
December	0.1247	1.6257	1.4379	0.0585	0.9685	0.9309

Figure D-47 summarizes the monthly average runoff volumes delivered to Lake Sherwood. The total annual volume delivered to the lake is 1,205 ac-ft. The months May through October each contribute less than 3 percent of the annual runoff volume.

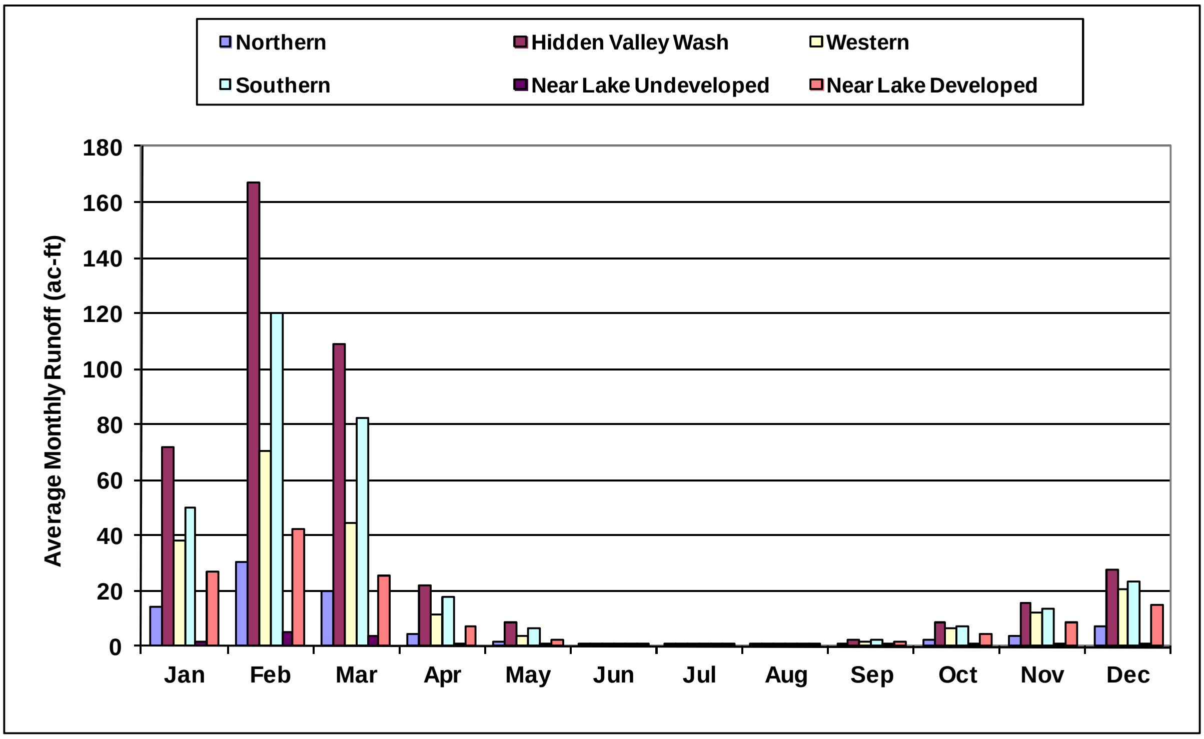


Figure D-47. Monthly Average Runoff Volumes to Lake Sherwood

D.12.2 SEDIMENT LOADS

Sediment loads associated with upland areas are calculated from simulated runoff volumes and suspended sediment event mean concentrations for each modeled land use (Section D.3). Table D-67 summarizes the average annual sediment loads for each jurisdiction by subwatershed. See example calculations in Section D.3.

Table D-67. Average Annual Sediment Loads to Lake Sherwood

Subwatershed	Jurisdiction	Sediment (tons/yr)
Western	Ventura County	1.53
Western	SVHOA	11.60
Hidden Valley Wash	Thousand Oaks	0.11
Hidden Valley Wash	Ventura County	507.42
Near Lake Undeveloped	Thousand Oaks	0.198
Near Lake Undeveloped	Ventura County	0.353
Near Lake Developed	Thousand Oaks	0.02

Subwatershed	Jurisdiction	Sediment (tons/yr)
Near Lake Developed	Ventura County	0.54
Near Lake Developed	SVHOA	9.29
Northern	Thousand Oaks	0.94
Northern	Ventura County	17.02
Northern	SVHOA	2.01
Carlisle Canyon	Caltrans	0.31
Carlisle Canyon	County of Los Angeles	3.26
Carlisle Canyon	Thousand Oaks	0.14
Carlisle Canyon	Ventura County	11.83
Carlisle Canyon	Point Mugu State Park	0.28
Total		**567**

For Lake Sherwood, the reported average annual sedimentation rate measured from 1905 to 1938 ranged from 2.5 to 10 acre-feet per year (0.22 to 0.88 inches per year) (Department of Boating and Waterways and State Coastal Conservancy, 2002). These measurements likely capture anomalous events such as flooding and fires that result in mass wasting of sediment and are not considered average conditions for the lake (the predicted average annual sediment load of 567 tons/yr is equal to 0.018 in/yr; Table D-67). Because large pulses of sediment are likely delivered during a few events, much of the associated pollutant loading is quickly buried and sequestered and therefore unavailable for release to the water column or entrance to the food chain via benthic organisms. Thus no additional pollutant loads were assumed for mass wasting events.

D.12.3 MERCURY LOADS

Mercury loads from each subwatershed are based on monitoring data collected by the Regional Board and USEPA during the winter and summer of 2009. Water column mercury concentrations measured from major inputs to the lakes are applied to simulated runoff volumes, and input mercury sediment concentrations are applied to the calculated sediment loads (see Section D.12.2) to estimate water column and sediment associated mercury loads, respectively. Figure D-48 shows the locations of the monitoring stations in the Lake Sherwood Watershed.

Table D-68 and Table D-69 present the methyl and total mercury concentrations observed at the mouth of each major input in the water column and sediments, respectively. More details regarding this data are presented in Appendix G (Monitoring Data).

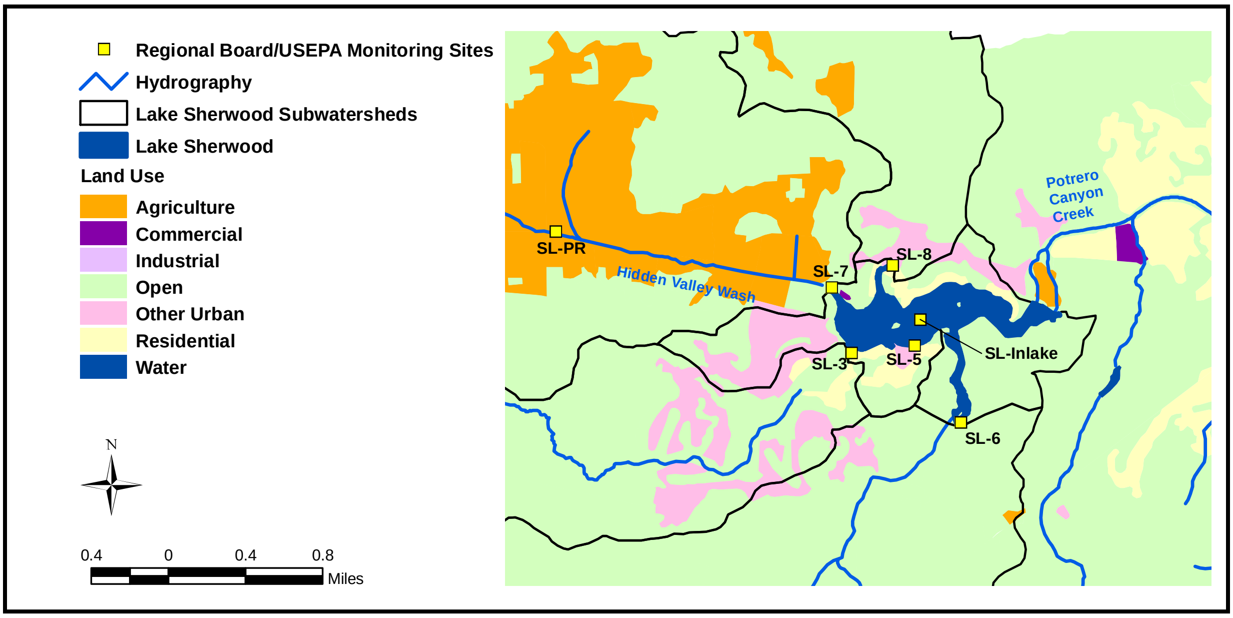


Figure D-48. Monitoring Stations in the Lake Sherwood Watershed

Table D-68. Tributary/Inflow Mercury Water Column Measurements for Lake Sherwood

Location	Date	Time	MeHg (ng/L)	Total Hg (ng/L)
SL-3	2/25/2009	13:00	0.157	6.00
SL-6		11:00	0.216	2.96
SL-8		11:45	0.025[1]	23.9
SL-3	7/13/2009	8:55	0.536	4.58
SL-3D		8:55	NA	4.63
SL-7		10:15	3.41	11.3
SL-8		10:00	0.096	54.0

[1] Temperature requirements for methylmercury analysis not met.

Table D-69. Inflow Mercury Sediment Concentrations for Lake Sherwood

Location	Date	Time	MeHg (µg/kg)	Total Hg (µg/kg)
SL-3	2/25/2009	13:00	0.269	92.7
SL-6		11:00	0.136	129
SL-5		13:15	0.145	51.0
SL-7[1]		08:30	2.53	243
SL-3	7/13/2009	8:55	0.397	392
SL-3D		8:55	NA	265

Location	Date	Time	MeHg (µg/kg)	Total Hg (µg/kg)
SL-5		9:45	0.657	62.9
SL-7		10:15	0.453	275
SL-PR		10:50	0.009	60.3
SL-8		10:00	0.696	63.3

[1] Sediment concentration data indicated that this site, which is located in a stagnant backwater area at the mouth of Hidden Valley Wash, may be a "hot spot" for methylation and not representative of the sediment methylmercury concentrations delivered from the subwatershed as a whole (sediment methylmercury concentrations were an order of magnitude greater than those observed at other sites in the watershed).

Concentrations of total and methylmercury vary seasonally at each input. Water column and sediment concentration data are available during both the summer and winter season at SL-3, which represents loading from the Western Subwatershed. Mercury loading is associated with both sediment and runoff from upland areas and given the availability of seasonal data, concentrations of total and methylmercury in water (Table D-68) and sediment (Table D-69) varied for the summer and winter. To determine sediment loading of mercury, the sediment EMCs and LSPC predicted runoff volumes were used to calculate sediment loads and the sediment mercury concentrations from monitoring data (Table D-69) were then applied to these sediment loads. Mercury loading associated with runoff from upland areas was calculated by applying the mercury water column concentrations (Table D-68) to LSPC simulated runoff volumes (Section D.3 provides examples of these calculations). The July 2009 monitoring data were used to estimate loadings for the summer season (May through October) and the February 2009 data were used to estimate loadings for the winter season (November through April). Similar calculations were performed to estimate mercury loading associated with runoff and sediment in the other subwatersheds.

The northern subwatershed is represented by Site SL-8, which has water column total mercury concentration data during both seasons but sediment data only in the summer season. Winter season total mercury concentration in the sediments (32.1 µg/kg) was assumed equal to the summer season total mercury concentration divided by the average observed ratio of summer to winter sediment total mercury concentrations observed at SL-3 (based on an average of SL-3 and its duplicate, SL-3D), SL-5, and SL-7 (average of [328.5 µg/kg ÷ 92.7 µg/kg; 62.9 µg/kg ÷ 51.0 µg/kg; 275 µg/kg ÷ 243 µg/kg] = ratio of 1.97; Table D-69). In addition, the water column methylmercury sample collected during the winter event was not maintained within the temperature constraints required for accurate analysis. Therefore, the winter season water column methylmercury concentration for the northern subwatershed (0.028 ng/L) was assumed equal to the summer season concentration divided by the ratio of summer to winter methylmercury observed at site SL-3 (0.536 ng/L ÷ 0.157 ng/L = ratio of 3.41; Table D-68) [water column data were not available for either season at SL-5, and SL-7 is likely not reflective of methylmercury concentrations in the watershed as a whole]. Winter sediment concentrations of methylmercury (0.232 µg/kg) were assumed equal to the summer concentration divided by the average observed ratio at SL-3 and SL-5 (average of [0.397 µg/kg ÷ 0.269 µg/kg; 0.657 µg/kg ÷ 0.145 µg/kg] = ratio of 3.0; Table D-69).

Sediment concentration data are available for both seasons at SL-5, which represents loading from the Near Lake Developed Subwatershed. Water was not flowing at SL-5 during either monitoring event so SL-8 data and related assumptions are used to represent water column concentrations for the Near Lake Developed Subwatershed.

Observations at SL-6 are used to estimate loading from the Carlisle Canyon and Near Lake Undeveloped Subwatersheds; the land use in both of these subwatersheds is primarily undeveloped. However, monitoring data are only available at SL-6 during the winter sampling event. To estimate summer season total mercury concentrations (3.4 ng/L in the water column; 254.1 µg/kg in the sediment), the average observed ratio of summer to winter water column concentrations at SL-3 (based on an average of SL-3 and its duplicate, SL-3D) and SL-8 (average of [4.60 ng/L ÷ 6.0 ng/L; 54.0 ng/L ÷ 23.9 ng/L] = ratio of 1.51 ; Table D-68) and sediment concentrations observed at SL-3 (based on an average of SL-3 and its duplicate, SL-3D), SL-5, and SL-7 (average of [328.5 µg/kg ÷ 92.7 µg/kg; 62.9 µg/kg ÷ 51.0 µg/kg; 275 µg/kg ÷ 243 µg/kg] = ratio of 1.97; Table D-69) are applied to the winter concentrations. To estimate summer season methylmercury concentrations at SL-6 (0.737 ng/L in the water column; 0.408 µg/kg in the sediment), the average observed ratio of summer to winter water column concentrations at SL-3 (0.536 ng/L ÷ 0.157 ng/L = ratio of 3.41; Table D-68) and sediment concentrations observed at SL-3 and SL-5 (average of [0.397 µg/kg ÷ 0.269 µg/kg; 0.657 µg/kg ÷ 0.145 µg/kg] = ratio of 3.0; Table D-69) are applied to the winter concentrations.

Site SL-7 was originally chosen to represent water column and sediment concentrations from the Hidden Valley Wash Subwatershed. Sediment concentration data collected during the winter monitoring event indicated that this site, which is located in a stagnant backwater area at the mouth of Hidden Valley Wash, may be a “hot spot” for methylation and not representative of the subwatershed as a whole (sediment methylmercury concentrations were an order of magnitude greater than those observed at other sites in the watershed). Water column concentrations observed during the summer event confirm this assumption as methylmercury concentrations were again an order of magnitude higher than those observed at other sites in the watershed. For the summer sampling event, Site SL-7 was re-sampled and site SL-PR was added as an upstream site on Hidden Valley Wash. Both water and sediment were sampled during this event at SL-7, but water was not flowing at SL-PR, so only sediment was sampled.

Summer sediment concentrations of total and methylmercury for the Hidden Valley Wash subwatershed were assumed equal to those observed during the summer at SL-PR. Winter sediment concentrations (30.6 µg/kg total mercury; 0.003 µg/kg methylmercury) were scaled down based on the average ratio of summer to winter sediment concentrations observed at SL-3 (based on an average of SL-3 and its duplicate, SL-3D), SL-5, and SL-7 (average of [328.5 µg/kg ÷ 92.7 µg/kg; 62.9 µg/kg ÷ 51.0 µg/kg; 275 µg/kg ÷ 243 µg/kg] = ratio of 1.97; Table D-69) for total mercury and at sites SL-3 and SL-5 (average of [0.397 µg/kg ÷ 0.269 µg/kg; 0.657 µg/kg ÷ 0.145 µg/kg] = ratio of 3.0; Table D-69) for methylmercury. Water column summer total mercury concentrations for Hidden Valley Wash were assumed equal to those observed during the summer event at SL-7 because 1) no data were available at SL-PR, and 2) total mercury water column concentrations were within the range of those observed at other sites in the watershed during the summer event. To estimate winter water column concentrations of total mercury at this site (7.48 ng/L), the average observed ratio of summer to winter water column concentrations at SL-3 (based on an average of SL-3 and its duplicate, SL-3D) and SL-8 (average of [4.61 ng/L ÷ 6.0 ng/L; 54.0 ng/L ÷ 23.9 ng/L] = ratio of 1.51; Table D-68) was applied. Summer (0.667 ng/L) and winter (0.374 ng/L) methylmercury water column concentrations were estimated from the total mercury concentrations assumed for each season multiplied by the fraction of mercury observed in the methyl form at other sites. For the summer methyl fraction, the average ratio observed at SL-3 and SL-8 was used (average of [0.536 ng/L ÷ 4.58 ng/L; 0.096 ng/L ÷ 54.0 ng/L] = ratio of 0.059; Table D-68); for the winter methyl fraction, the average ratio observed at SL-3 and SL-6 was used (average of [0.157 ng/L ÷ 6.00 ng/L; 0.216 ng/L ÷ 2.96 ng/L] = ratio of 0.050; Table D-68).

The assumed concentrations were applied to the runoff and sediment loads estimated from each jurisdiction within the watershed. Assumed total mercury concentrations and resulting loads are summarized in Table D-70. See example calculations in Section D.3. Results for methylmercury are presented in Table D-71. The Hidden Valley Wash subwatershed generates approximately 60 percent of the total and methylmercury loads to Lake Sherwood due to its acreage and predominance of agricultural land use relative to the other subwatersheds. Based on monitoring data collected in 2009, these loads are discharged to a stagnant, backwater area that exhibits high rates of methylation. These loads are thus greater and more bioavailable relative to other sources in the watershed.

Table D-70. Total Mercury Loads Estimated for Each Jurisdiction and Subwatershed in the Lake Sherwood Watershed

Subwatershed	Jurisdiction	Area (ac)	Summer Water Column Hg (ng/L)	Winter Water Column Hg (ng/L)	Summer Sediment Hg (µg/kg)	Winter Sediment Hg (µg/kg)	Annual Water Column Hg Load (g/yr)	Annual Sediment Hg Load (g/yr)	Total Annual Hg Load (g/yr)
Western	Ventura County	548	4.6	6.0	328.5	92.7	0.286	0.146	0.432
Western	SVHOA	772	4.6	6.0	328.5	92.7	1.253	1.142	2.395
Hidden Valley Wash	Thousand Oaks	40	11.3	7.5	60.3	30.6	0.027	0.003	0.031
Hidden Valley Wash	Ventura County	3,793	11.3	7.5	60.3	30.6	4.083	14.725	18.808
Near Lake Undeveloped	Thousand Oaks	70.9	4.48	2.96	254.1	129	0.019	0.024	0.043
Near Lake Undeveloped	Ventura County	126	4.48	2.96	254.1	129	0.034	0.043	0.077
Near Lake Developed	Thousand Oaks	9	54	23.9	62.9	51.0	0.020	0.001	0.021
Near Lake Developed	Ventura County	24	54	23.9	62.9	51.0	0.243	0.025	0.268
Near Lake Developed	SVHOA	310	54	23.9	62.9	51.0	4.060	0.437	4.497
Northern	Thousand Oaks	338	54	23.9	63.3	32.1	0.757	0.029	0.786
Northern	Ventura County	351	54	23.9	63.3	32.1	1.080	0.519	1.599
Northern	SVHOA	65	54	23.9	63.3	32.1	0.871	0.062	0.934
Carlisle Canyon	Caltrans	2.75	4.48	2.96	254.1	129	0.010	0.039	0.049
Carlisle Canyon	County of Los Angeles	1,149	4.48	2.96	254.1	129	0.307	0.401	0.708
Carlisle Canyon	Thousand Oaks	50.4	4.48	2.96	254.1	129	0.013	0.017	0.031
Carlisle Canyon	Ventura County	2,905	4.48	2.96	254.1	129	0.861	1.457	2.318
Carlisle Canyon	Point Mugu State Park	101	4.48	2.96	254.1	129	0.027	0.035	0.062
Total		**10,655**	**N/A**	**N/A**	**N/A**	**N/A**	**13.95**	**19.11**	**33.06**

N/A = Not applicable

Table D-71. Methylmercury Loads Estimated for Each Jurisdiction and Subwatershed in the Lake Sherwood Watershed

Subwatershed	Jurisdiction	Area (ac)	Summer Water Column MeHg (ng/L)	Winter Water Column MeHg (ng/L)	Summer Sediment MeHg (µg/kg)	Winter Sediment MeHg (µg/kg)	Annual Water Column MeHg Load (g/yr)	Annual Sediment MeHg Load (g/yr)	Total Annual MeHg Load (g/yr)
Western	Ventura County	548	0.536	0.157	0.397	0.269	8.54E-03	3.83E-04	8.92E-03
Western	SVHOA	772	0.536	0.157	0.397	0.269	3.85E-02	2.92E-03	4.15E-02
Hidden Valley Wash	Thousand Oaks	40	0.672	0.370	0.009	0.003	1.37E-03	3.40E-07	1.37E-03
Hidden Valley Wash	Ventura County	3,793	0.672	0.370	0.009	0.003	2.05E-01	1.51E-03	2.07E-01
Near Lake Undeveloped	Thousand Oaks	70.9	0.737	0.216	0.408	0.136	1.52E-03	2.70E-05	1.55E-03
Near Lake Undeveloped	Ventura County	126	0.737	0.216	0.408	0.136	2.71E-03	4.82E-05	2.76E-03
Near Lake Developed	Thousand Oaks	9	0.096	0.028	0.657	0.145	2.47E-05	3.86E-06	2.85E-05
Near Lake Developed	Ventura County	24	0.096	0.028	0.657	0.145	3.06E-04	8.79E-05	3.94E-04
Near Lake Developed	SVHOA	310	0.096	0.028	0.657	0.145	5.12E-03	1.52E-03	6.64E-03
Northern	Thousand Oaks	338	0.096	0.028	0.696	0.232	9.42E-04	2.19E-04	1.16E-03
Northern	Ventura County	351	0.096	0.028	0.696	0.232	1.35E-03	3.91E-03	5.26E-03
Northern	SVHOA	65.08	0.096	0.028	0.696	0.232	1.10E-03	4.81E-04	1.58E-03
Carlisle Canyon	Caltrans	2.75	0.737	0.216	0.408	0.136	8.40E-04	4.36E-05	8.84E-04
Carlisle Canyon	County of Los Angeles	1,149	0.737	0.216	0.408	0.136	2.46E-02	4.45E-04	2.51E-02
Carlisle Canyon	Thousand Oaks	50.4	0.737	0.216	0.408	0.136	1.08E-03	1.92E-05	1.10E-03
Carlisle Canyon	Ventura County	2,905	0.737	0.216	0.408	0.136	6.92E-02	1.62E-03	7.08E-02
Carlisle Canyon	Point Mugu State Park	101	0.737	0.216	0.408	0.136	2.16E-03	3.84E-05	2.20E-03
Total		**10,655**	**N/A**	**N/A**	**N/A**	**N/A**	**0.36**	**0.01**	**0.38**

N/A = Not applicable

D.13 References

Ackerman and Schiff. 2003. Modeling Storm Water Mass Emissions to the Southern California Bight. *Journal of Environmental Engineering*, ASCE, April 2003.

Bicknell, B.R., J.C. Imhoff, J. John, L. Kittle, T.H. Jobes and J.A.S. Donigian. 2001. Hydrological simulation program - FORTRAN, Version 12. AQUA TERRA Consultants. Mountain View, California. 873 pp.

Department of Boating and Waterways and State Coastal Conservancy. 2002. California Beach Restoration Study. http://www.dbw.ca.gov/Environmental/BeachReport.aspx.

DePoto, W., I. Gindi and M. Schleikorn. 1991. Hydrology Manual. Los Angeles County Department of Public Works, Hydraulic/Water Conservation Division. Alhambra, California. 79 pp.

LACDPW. 2000. Stormwater Quality Summary Data. Los Angeles County Department of Public Works. http://dpw.lacounty.gov/wmd/npdes/wq_data.cfm.

Tetra Tech. 2004. Model Development for Simulation of Wet-Weather Metals Loading from the Los Angeles River Watershed. Prepared for USEPA Region 9 and the Los Angeles Regional Water Quality Control Board, May 2004.

Tetra Tech. 2005. Model Development for Simulation of Wet-Weather Metals Loading from the San Gabriel River Watershed. Prepared for USEPA Region 9 and the Los Angeles Regional Water Quality Control Board, October 2005.

USDA (U.S. Department of Agriculture). 1986. U.S. Soil Conservation Service. Technical Release 55: Urban Hydrology for Small Watersheds. June 1986.

(This page left intentionally blank.)

Appendix E. Pollutant Loading from Atmospheric Deposition

(This page left intentionally blank.)

E.1 Introduction

USEPA Region IX is establishing TMDLs for impairments in nine lakes in the Los Angeles Region (Figure E-1). USEPA was assisted in this effort by the Los Angeles Water Quality Control Board (Regional Board) . Impairments of these waterbodies include low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, algae, pH, mercury, lead, copper, chlordane, dieldrin, DDT, PCBs, and trash. These impairments are typically associated with pollutant loading from various sources, one of which may be atmospheric deposition.

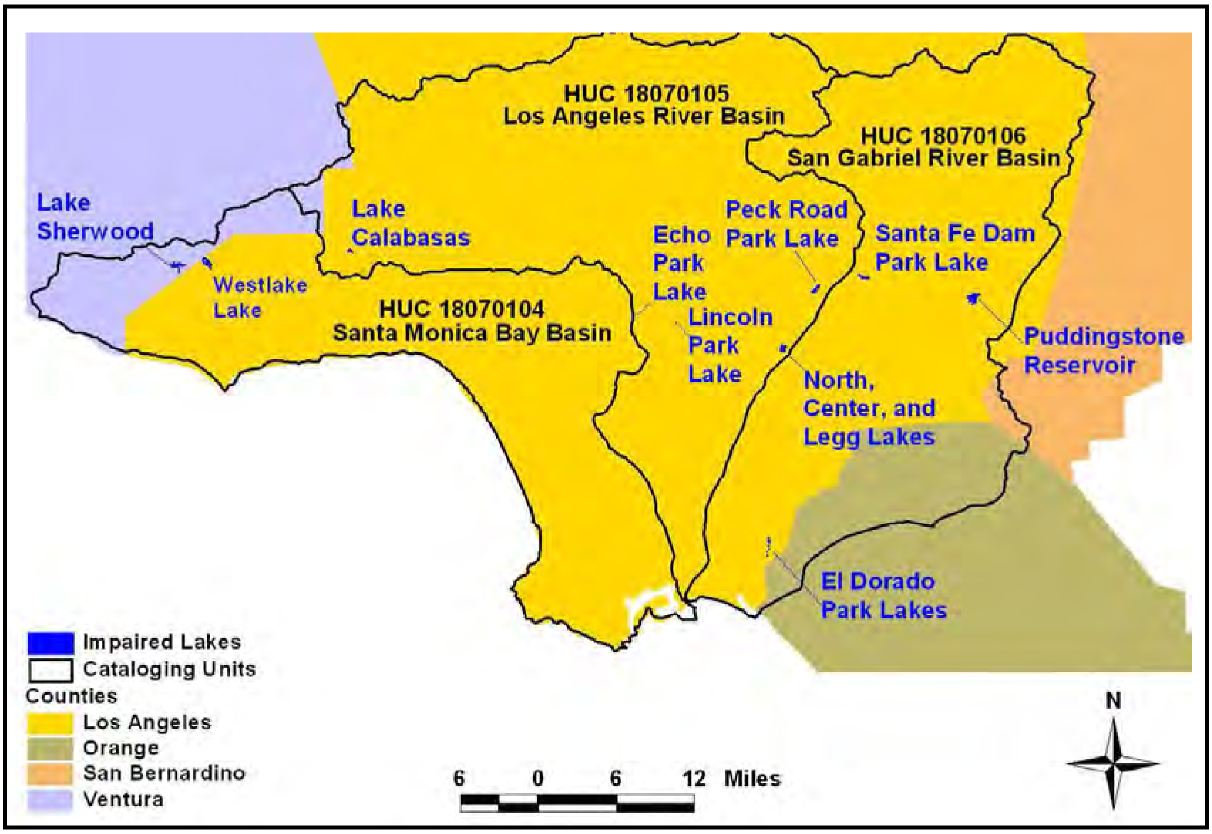


Figure E-1. Location of Impaired Lakes

Atmospheric deposition of pollutants may occur as either wet deposition (associated with precipitation) or dry deposition (associated with particulates). Wet deposition of nitrate, sulfate, and mercury are monitored nationally by the National Atmospheric Deposition Program (NADP) and the Mercury Deposition Network (MDN). Dry deposition of these parameters is less frequently monitored. Pollutants such as Organochlorine (OC) Pesticides and PCBs s have been studied regionally.

This Appendix summarizes the monitoring data, modeling efforts, and regional studies available to estimate pollutant loading from atmospheric deposition to the water surfaces of the lakes addressed by this TMDL.

(This page left intentionally blank.)

E.2 Phosphorus Deposition

Eight lakes shown in Figure E-1 (all except Lake Sherwood and Westlake Lake) have impairments addressed by this TMDL report that may be due to excessive nutrient loading. A potential source of phosphorus loading to a lake surface is atmospheric deposition. However, phosphorus does not have a significant gaseous phase, and atmospheric deposition is primarily due to fugitive dust. Phosphorus deposition rates are typically much lower than other pollutant deposition rates and are not included in the NADP monitoring program.

Currently, direct measurements of phosphorus deposition rates in Southern California are not available. Given the likelihood that direct deposition of phosphorus to a waterbody is insignificant relative to other sources of loading, the nutrient TMDLs for these eight lakes will assume zero phosphorus loading from atmospheric deposition. The Southern California Coastal Water Research Project (SCCWRP) has recently begun a deposition monitoring study that will measure phosphorus, but the results are not expected to be published until 2011. If this study indicates that atmospheric deposition of phosphorus is a significant source of phosphorus to waterbodies in the region, the nutrient TMDLs may be amended to reflect these data.

(This page left intentionally blank.)

E.3 Nitrate Deposition

The National Atmospheric Deposition Program (NADP) monitors wet nitrate and sulfate deposition at two active and two inactive stations in southern California (Figure E-2). [Though site CA94 is also a NADP site, the period of record is not sufficient to assess nitrate trends with time.] Originally, data from these stations were to be combined to develop a regression equation that could be used to predict annual precipitation-weighted nitrate concentrations and sulfate at each impaired lake. Table E-1 lists the NADP monitoring stations, elevations, and periods of record used for the analysis.

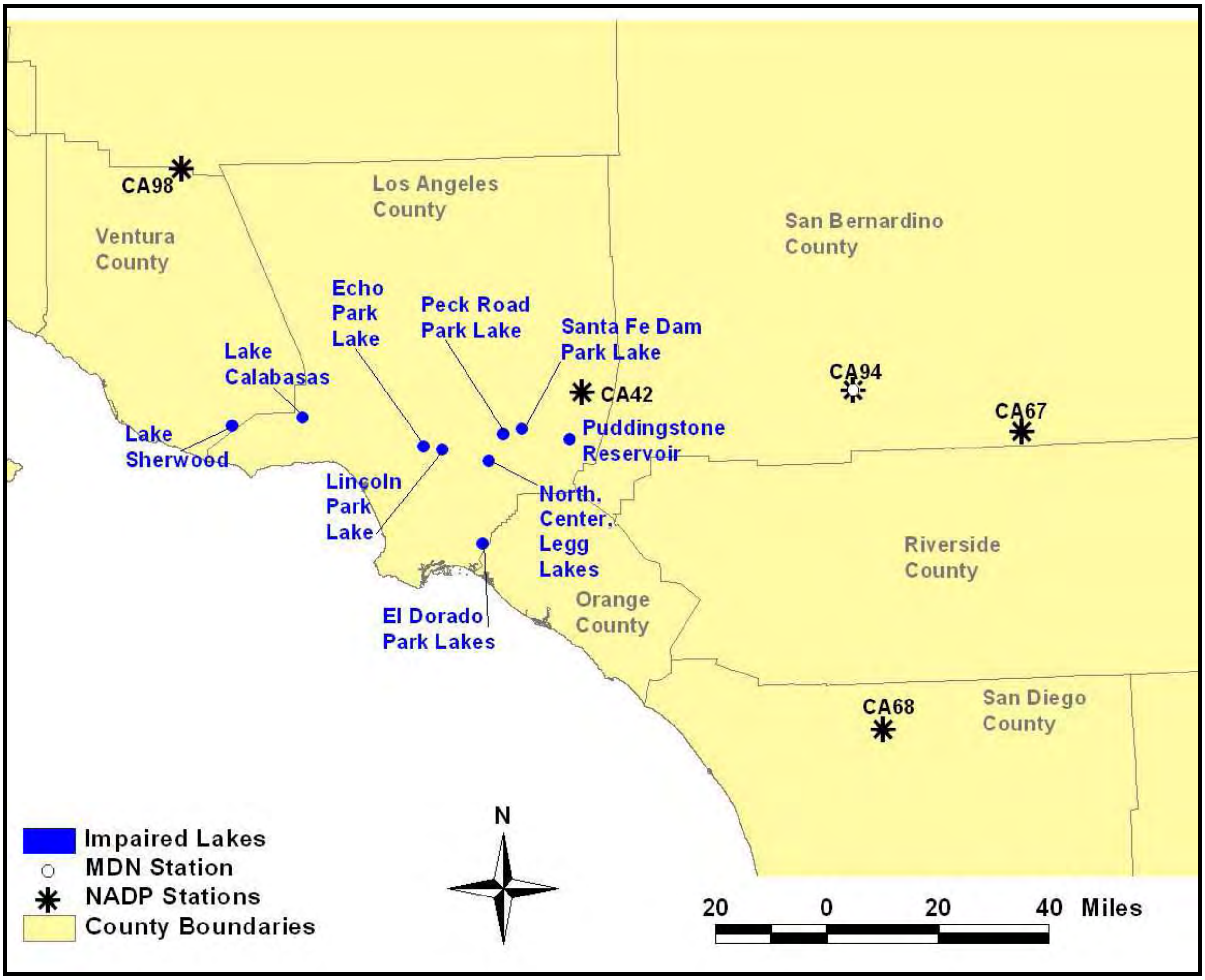


Figure E-2. Location of NADP Monitoring Stations

Table E-1. NADP Stations Used to Develop Nitrate Regression Based on Elevation and Year

ID	Name	Period of Record	Elevation (m)
CA42	Tanbark Flat	January 1982 to February 2008	853
CA67	Joshua Tree	September 2000 to February 2008	1,239
CA68	Palomar Mountain	March 1983 to January 1988	1,695
CA98	Chuchupate Ranger Station	March 1983 to January 1996	1,614

Figure E-3 shows the annual precipitation-weighted nitrate concentrations at the four sites used to develop the regression analysis. At each of the four stations, concentrations of nitrate show a decreasing trend with time.

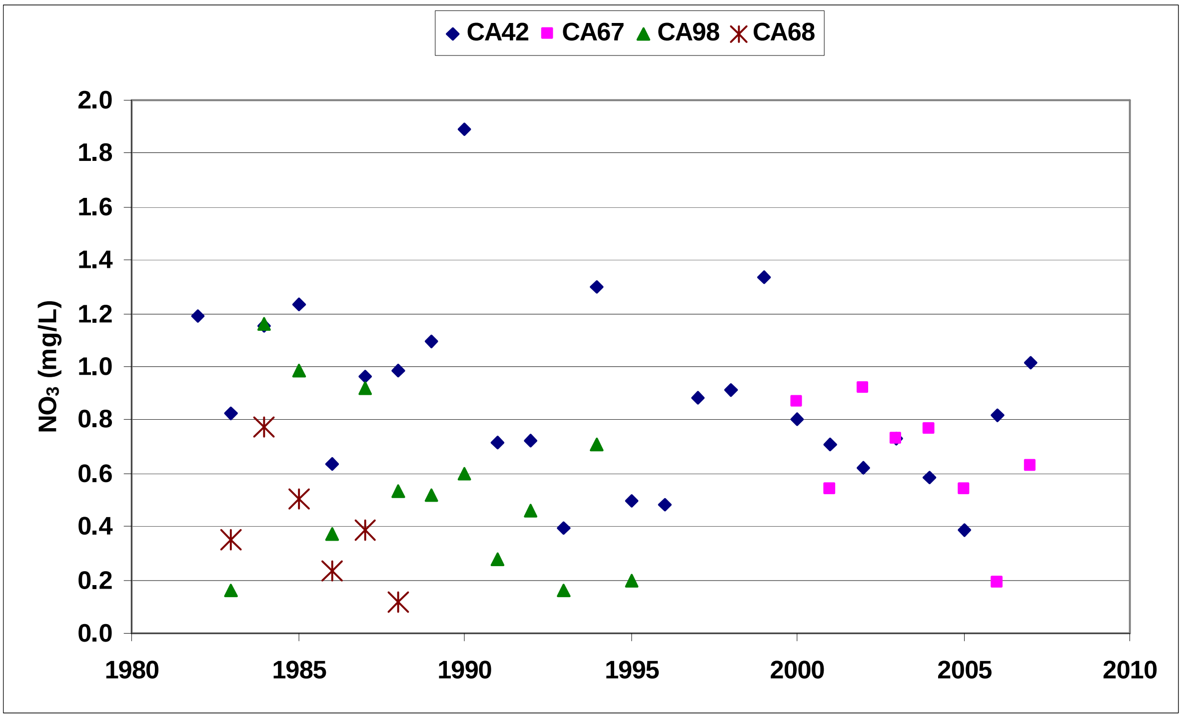


Figure E-3. Annual Precipitation-Weighted Nitrate Concentrations at Four Locations in Southern California

The regression analysis combining the elevation of each station along with year resulted in the following equation for predicting annual precipitation-weighted nitrate concentrations:

$$LOG_{10}\ (NO_3,\ mg/L) = 88.56 - 25.82\ LOG_{10}\ (Year) - 1.167\ LOG_{10}\ (Elevation,\ m),\ R^2 = 31.2\%$$

In the past, Tetra Tech has used this regression approach to estimate nitrate concentrations at varying elevations for TMDLs developed in Colorado, Arizona, and elsewhere in California. Unfortunately, the elevations of the impaired lakes addressed by this TMDL, ranging from 7 meters to 293 meters, are significantly less than the elevations of the four NADP stations available for developing the regression (853 meters to 1,695 meters). The predicted nitrate concentrations over the range of elevations of the impaired lakes are therefore significantly overestimated. Figure E-4 shows the predicted nitrate concentrations, respectively, for an example year (2000) over a range of elevations.

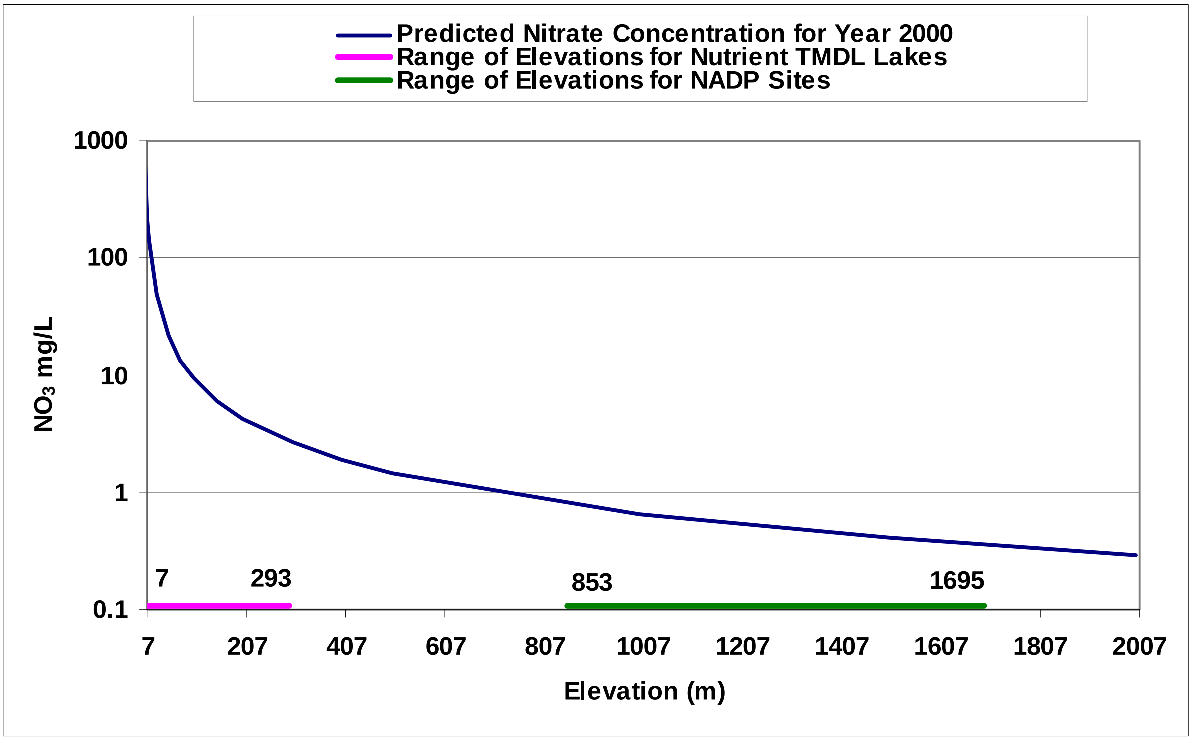


Figure E-4. Predicted Precipitation-Weighted Nitrate Concentrations for the Year 2000

As an alternative approach for predicting annual precipitation-weighted nitrate concentrations for the impaired lakes, Tetra Tech downloaded geospatial annual isopleth maps published by NADP and extracted the nitrate concentrations for grid cells overlaying each lake. NADP has produced the isopleth maps for 1994 to 2006. Tetra Tech extended the time series to previous years to correspond with available LSPC model output (Appendix D, Wet Weather Loading) for other source load estimates by developing a regression equation for each location based on year and cumulative precipitation. Table E-2 presents the annual precipitation-weighted concentrations measured (1994 to 2006) or estimated (1983 to 1993) at each lake. Although this TMDL does not address nutrient impairments at Lake Sherwood, the nitrate analysis is relevant for the mercury wet deposition estimates (Section E.4).

Table E-2. Annual Precipitation-Weighted Nitrate Concentrations (mg-NO_3/L)

Year	Peck Road Park Lake	Lincoln Park Lake	Echo Park Lake	Lake Calabasas	El Dorado Park Lakes	Puddingstone Reservoir	Legg, Center, and North Lakes	Santa Fe Dam Park Lake	Lake Sherwood
1983	0.73	0.74	0.73	0.64	1.07	1.10	0.79	0.74	0.58
1984	1.29	1.30	1.29	1.29	1.41	1.54	1.44	1.03	1.24
1985	1.33	1.25	1.25	1.21	1.38	1.49	1.44	1.06	1.17
1986	1.11	1.04	1.03	1.01	1.23	1.32	1.26	0.94	0.97
1987	1.19	1.22	1.22	1.13	1.30	1.34	1.35	0.99	1.09
1988	1.14	1.16	1.16	1.09	1.27	1.31	1.29	0.97	1.05
1989	1.30	1.28	1.28	1.26	1.34	1.40	1.41	1.06	1.22
1990	1.20	1.20	1.19	1.18	1.25	1.30	1.27	1.01	1.15
1991	0.98	0.98	0.97	0.99	1.08	1.10	1.09	0.89	0.96
1992	0.84	0.79	0.79	0.73	0.95	0.90	0.95	0.81	0.70
1993	0.75	0.75	0.75	0.78	0.87	0.83	0.83	0.76	0.75
1994	1.30	1.29	1.29	1.27	1.29	1.30	1.30	1.30	1.26
1995	0.49	0.48	0.47	0.35	0.48	0.49	0.49	0.49	0.29
1996	0.48	0.48	0.48	0.47	0.48	0.48	0.48	0.48	0.47
1997	0.89	0.88	0.88	0.87	0.88	0.89	0.89	0.89	0.86
1998	0.92	0.92	0.92	0.91	0.92	0.92	0.95	0.92	0.91
1999	1.33	1.33	1.33	1.33	1.33	1.33	1.33	1.33	1.32
2000	0.80	0.80	0.80	0.79	0.80	0.80	0.80	0.80	0.78
2001	0.71	0.71	0.71	0.72	0.71	0.71	0.71	0.71	0.73
2002	0.62	0.62	0.62	0.63	0.63	0.62	0.62	0.63	0.63
2003	0.73	0.73	0.73	0.73	0.73	0.73	0.73	0.73	0.73
2004	0.58	0.59	0.59	0.59	0.59	0.58	0.58	0.58	0.58
2005	0.39	0.39	0.39	0.39	0.39	0.39	0.39	0.39	0.39
2006	0.81	0.81	0.81	0.81	0.81	0.81	0.81	0.81	0.80

Table E-3 lists the surface areas and annual precipitation assumed for each impaired lake. A discussion of the weather stations used to estimate annual precipitation at each lake is discussed in Appendix D (Wet Weather Loading).

The annual direct deposition load to a water surface depends on the amount of precipitation, the lake surface area, and the precipitation-weighted nitrate concentration measured or estimated for that year. For example, the nitrogen load deposited to the surface of Peck Road Park Lake in 1983 may be estimated as follows:

1) Convert the units of the precipitation-weighted nitrate concentration for 1983 from NO_3 to N.

$$\frac{0.73mg-NO_3}{L}\cdot\frac{1mmolNO_3}{62mgNO_3}\cdot\frac{1mmolN}{1mmolNO_3}\cdot\frac{14mgN}{mmolN}=\frac{0.165mgN}{L}$$

2) Estimate the volume of precipitation to the lake surface in 1983.

$$41.2in\cdot 87.4ac\cdot\frac{1ft}{12in}=300.07ac-ft$$

3) Multiply concentration by volume to calculate load.

$$\frac{0.165mgN}{L}\cdot 300.7ac-ft\cdot\frac{43{,}560ft^2}{1ac}\cdot\frac{28.32L}{1ft^3}\cdot\frac{g}{1000mg}\cdot\frac{1lb}{453.6g}=134.5lbN$$

Table E-4 presents the average nitrogen load to each lake due to atmospheric deposition.

Table E-3. Annual Precipitation (inches) and Surface Area of Impaired Lakes

Year	Peck Road Park Lake (87.4 ac)	Lincoln Park Lake (4.9 ac)	Echo Park Lake (14.1 ac)	Lake Calabasas (17.8 ac)	El Dorado Park Lakes (35.3 ac)	Puddingstone Reservoir (252.4 ac)	Legg, Center, and North Lakes (76.6 ac)	Santa Fe Dam Park Lake (70.6 ac)	Lake Sherwood (136.8 ac)
1983	41.2	34.0	34.0	41.0	26.7	37.7	41.1	39.1	41.0
1984	11.9	8.9	8.9	6.9	8.5	10.1	10.6	11.9	6.9
1985	8.4	9.7	9.7	9.8	8.2	10.2	8.9	8.4	9.8
1986	18.6	18.1	18.1	19.1	13.5	18.1	15.7	18.6	19.1
1987	13.4	9.1	9.1	12.0	8.2	14.7	10.0	13.3	12.0
1988	14.7	10.7	10.7	13.1	8.1	13.9	11.2	14.4	13.1
1989	5.7	4.5	4.5	3.6	3.0	6.1	4.4	5.4	3.6
1990	9.3	7.2	7.2	6.4	5.7	9.6	9.5	9.2	6.4
1991	19.7	15.7	15.7	15.6	12.2	18.9	16.1	19.4	15.6
1992	25.7	22.8	22.8	27.7	16.9	28.2	20.8	25.6	27.7
1993	28.9	23.5	23.5	24.2	19.2	30.3	25.0	29.6	24.2
1994	11.0	8.7	8.7	10.1	8.4	11.3	10.4	11.0	10.1
1995	30.6	23.7	23.7	26.6	21.5	28.2	26.5	30.6	26.6
1996	25.0	17.4	17.4	20.1	14.9	24.3	23.3	23.8	20.1
1997	12.8	10.2	10.2	14.8	12.4	17.5	14.9	6.9	14.8
1998	31.2	27.3	27.3	31.5	24.1	32.2	31.8	31.2	31.5
1999	7.8	8.0	8.0	7.5	6.8	7.8	6.8	7.5	7.5
2000	16.1	12.0	12.0	12.8	8.5	12.6	13.6	15.7	12.8
2001	25.6	17.0	17.0	29.5	4.7	21.1	17.0	20.8	29.5
2002	8.9	7.3	7.3	9.5	2.7	8.7	7.6	8.8	9.5
2003	16.8	13.4	13.4	14.9	9.1	16.8	15.3	16.8	14.9
2004	24.4	19.8	19.8	19.9	14.9	21.1	20.5	24.4	19.9
2005	36.3	25.6	25.6	29.6	14.2	6.4	24.0	36.3	29.6
2006	14.9	11.6	11.6	14.0	7.8	11.8	11.9	14.9	14.0
Average	**19.1**	**15.3**	**15.3**	**17.5**	**11.7**	**17.4**	**16.5**	**18.5**	**17.5**

Table E-4. Annual Nitrogen Load (lb) from Atmospheric Deposition to Impaired Lakes

Year	Peck Road Park Lake	Lincoln Park Lake	Echo Park Lake	Lake Calabasas	El Dorado Park Lakes	Puddingstone Reservoir	Legg, Center, and North Lakes	Santa Fe Dam Park Lake	Lake Sherwood
1983	134.5	6.3	17.9	23.9	51.6	535.6	127.3	104.5	166.5
1984	68.7	2.9	8.3	8.1	21.6	200.9	59.8	44.3	59.9
1985	50.0	3.0	8.7	10.8	20.4	196.3	50.2	32.2	80.3
1986	92.3	4.7	13.5	17.6	30.0	308.6	77.5	63.2	129.7
1987	71.3	2.8	8.0	12.4	19.3	254.4	52.9	47.6	91.6
1988	74.9	3.1	9.0	13.0	18.6	235.2	56.6	50.5	96.3
1989	33.1	1.4	4.2	4.1	7.3	110.3	24.3	20.7	30.7
1990	49.9	2.2	6.2	6.9	12.9	161.2	47.3	33.6	51.5
1991	86.3	3.9	11.0	14.1	23.8	268.5	68.8	62.4	104.8
1992	96.5	4.5	13.0	18.4	29.0	327.8	77.5	74.9	135.7
1993	96.9	4.4	12.7	17.2	30.2	324.8	81.3	81.3	127.1
1994	64.0	2.8	8.1	11.7	19.6	189.7	53.0	51.7	89.1
1995	67.1	2.9	8.0	8.5	18.6	178.5	50.9	54.2	54.0
1996	53.7	2.1	6.0	8.6	12.9	150.6	43.8	41.3	66.1
1997	50.9	2.3	6.5	11.7	19.7	201.2	52.0	22.2	89.1
1998	128.4	6.3	18.1	26.1	40.0	382.6	118.4	103.7	200.7
1999	46.4	2.7	7.7	9.1	16.3	134.0	35.4	36.0	69.3
2000	57.6	2.4	6.9	9.2	12.3	130.2	42.6	45.4	69.9
2001	81.3	3.0	8.7	19.3	6.0	193.5	47.3	53.4	150.7
2002	24.7	1.1	3.3	5.5	3.1	69.7	18.5	20.0	41.9
2003	54.8	2.5	7.1	9.9	12.0	158.4	43.8	44.3	76.1
2004	63.3	2.9	8.4	10.7	15.9	158.1	46.6	51.1	80.8
2005	63.3	2.5	7.2	10.5	10.0	32.2	36.7	51.1	80.8
2006	54.0	2.4	6.8	10.3	11.4	123.4	37.8	43.6	78.4
Average	**69.3**	**3.1**	**9.0**	**12.4**	**19.3**	**209.4**	**56.3**	**51.4**	**92.5**

(This page left intentionally blank.)

E.4 Mercury Deposition

Mercury deposition from the atmosphere to the earth's surface may occur in several forms: gaseous elemental mercury (Hg(0)), divalent ionic mercury (Hg(II)), reactive gaseous mercury (RGM), and aerosol particulate mercury (Hg-P). Atmospheric deposition can be divided into short-range or near-field deposition, which includes deposition from sources located near the watershed, and long-range or far-field deposition, which includes mercury deposition from regional and global sources. Mercury emitted from manmade sources usually contains both gaseous elemental mercury (Hg(0)) and divalent mercury (Hg(II)). Hg(II) species, because of their solubility and their tendency to attach to particles, are redeposited relatively close to their source (probably within a few hundred miles), whereas Hg(0) remains in the atmosphere much longer, contributing to long-range transport.

Deposition may either occur in wet form (associated with precipitation) or dry form (associated with particulate settling). Wet deposition is monitored at select locations across the country by the Mercury Deposition Network (MDN). There is one MDN site in Southern California, but it has only been active since May of 2006. The rates of wet mercury deposition to each lake water surface were estimated with a regression approach that utilizes nitrate and sulfate wet deposition data collected by the National Atmospheric Deposition Program (NADP) along with mercury wet deposition data collected by the MDN.

Dry deposition is more difficult to monitor and less localized data are available to estimate this component. To estimate loading from dry deposition, grid-cell output from regional deposition models developed by USEPA was obtained for each lake impaired by mercury.

To evaluate potential near-field sources at each impaired lake, the USEPA Toxics Release Inventory (TRI) was used to determine the proximity of point sources that may contribute to airborne mercury loads including coal-fired power plants, steel recycling facilities, waste incinerators, cement and lime kilns, smelters and gold mine roasters, pulp and paper mills, and chlor-alkali factories.

Precipitation events following recent forest fires also result in increased loads of total and methylmercury from the watershed and release of elemental mercury to the atmosphere which is then available for deposition.

E.4.1. NEAR FIELD SOURCES OF ATMOSPHERIC MERCURY

Major atmospheric point sources of mercury can cause locally elevated areas of near-field atmospheric deposition downwind. Mercury emitted from manmade sources usually contains both gaseous elemental mercury (Hg(0)) and divalent mercury (Hg(II)). Hg(II) species, because of their solubility and their tendency to attach to particles, are redeposited relatively close to their source (probably within a few hundred miles), whereas Hg(0) remains in the atmosphere much longer, contributing to long-range transport. Reactive gaseous mercury and particulate mercury are also associated with manmade sources and typically deposit within approximately 100 miles of the source.

Significant potential near-field emission sources of airborne mercury include coal-fired power plants, steel recycling facilities, waste incinerators, cement and lime kilns, smelters and gold mine roasters, pulp and paper mills, and chlor-alkali factories. Emissions from such sources are summarized in USEPA's Toxic Release Inventory (TRI). Facilities that reported emissions of mercury in southern California in 2007 to the USEPA (2009) within 100 miles of the El Dorado Park lakes, Puddingstone Reservoir, or Lake Sherwood are shown in Figure E-5, Figure E-6, and Figure E-7, respectively. Emissions data for 2008 have not yet been released.

Table E-5 summarizes the loads reported from each facility in the 2007 TRI within 100 miles of either of these three waterbodies. Thirty-five out of 64 facilities listed in the database reported zero pounds of mercury released in 2006 (these are not included in the table); 19 reported emissions less than 10 pounds per year. Four of the top five sources of mercury emissions were due to cement manufacturing facilities; one of the top five is an oil refinery. Total reported mercury air emissions in 2007 within 100 miles of these three mercury impaired lakes were 1,043 pounds.

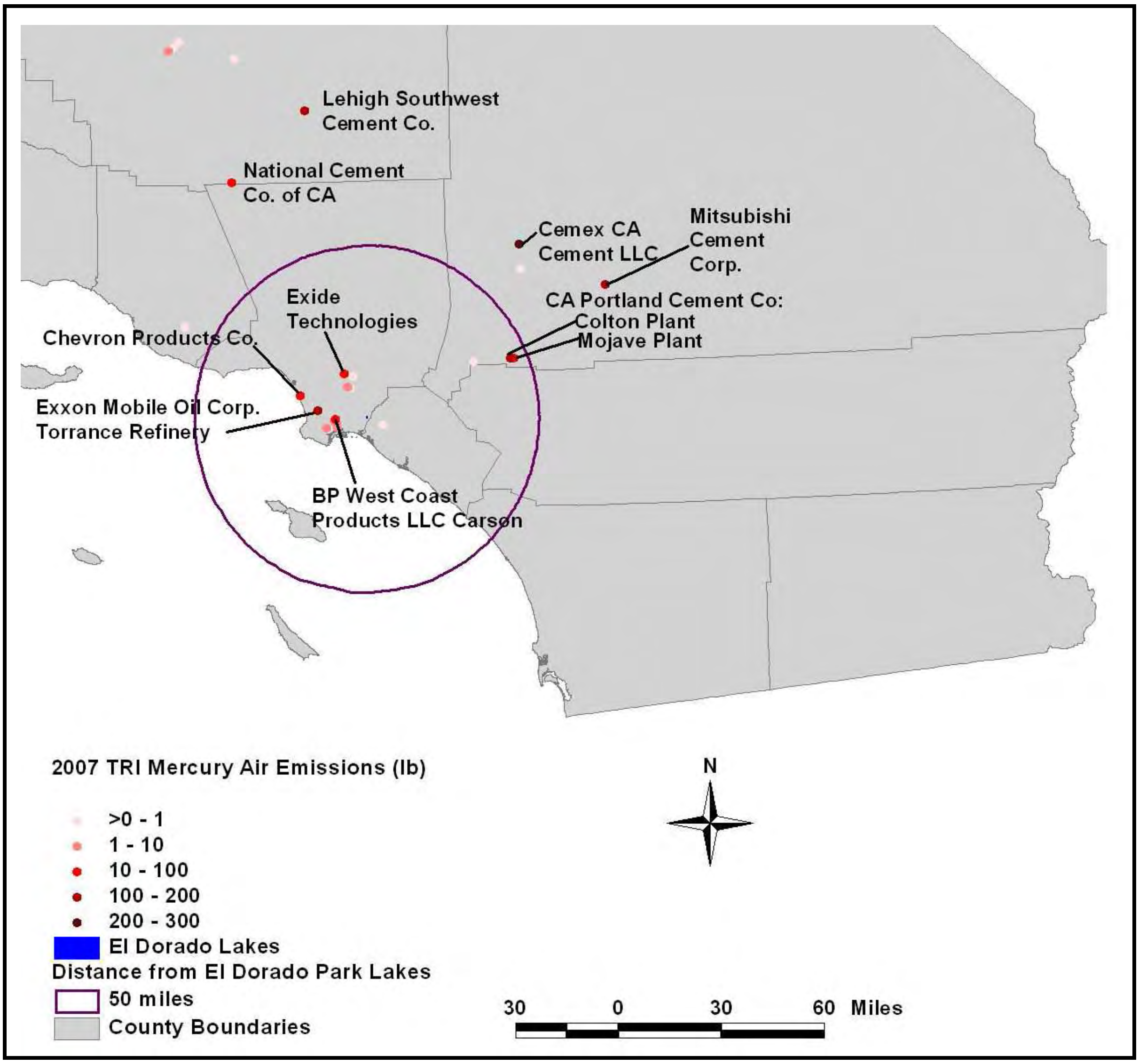


Figure E-5. Location of Facilities Reporting Mercury Emissions within 50 miles of the El Dorado Park Lakes

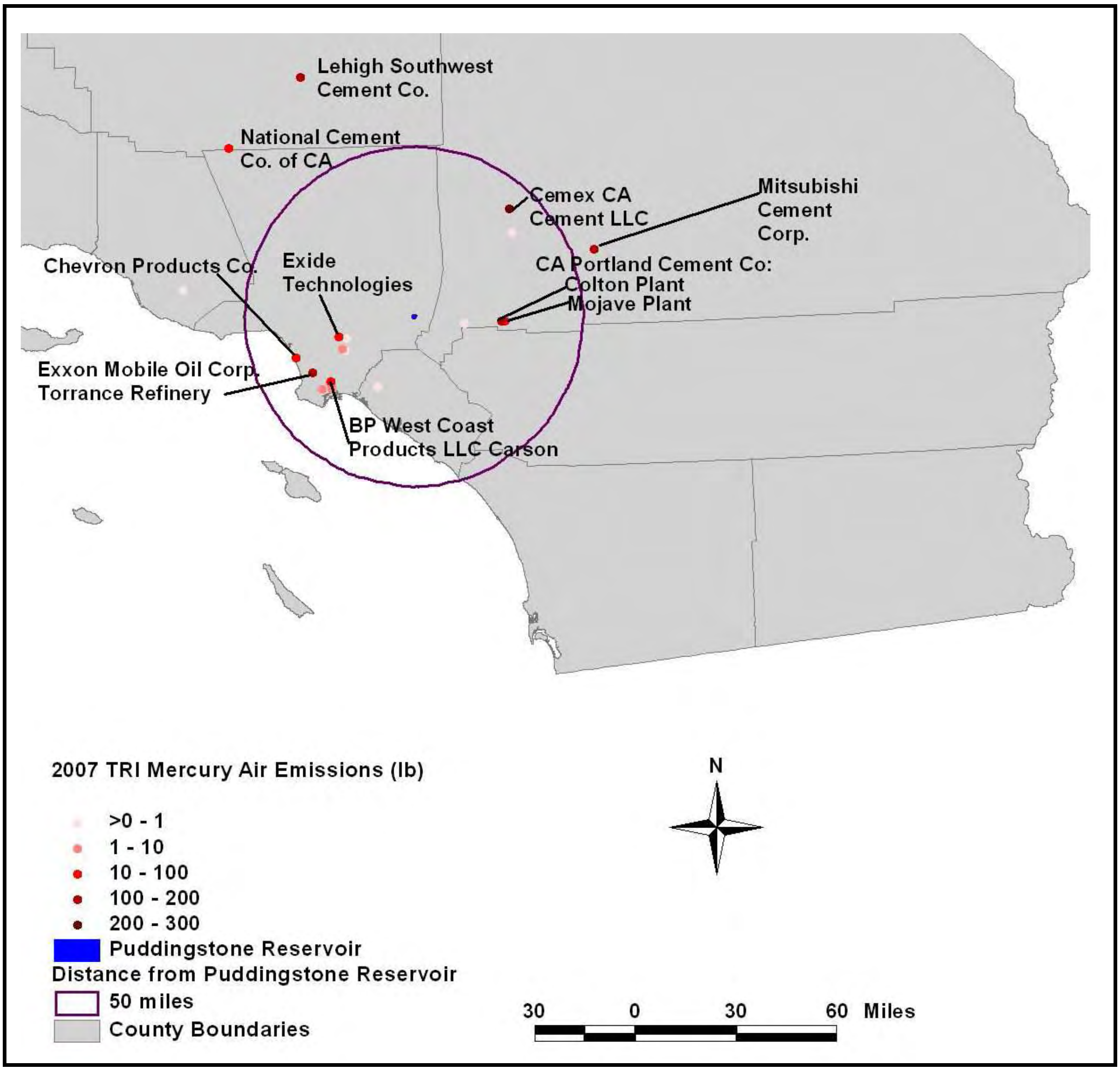


Figure E-6. **Location of Facilities Reporting Mercury Emissions within 50 miles of Puddingstone Reservoir**

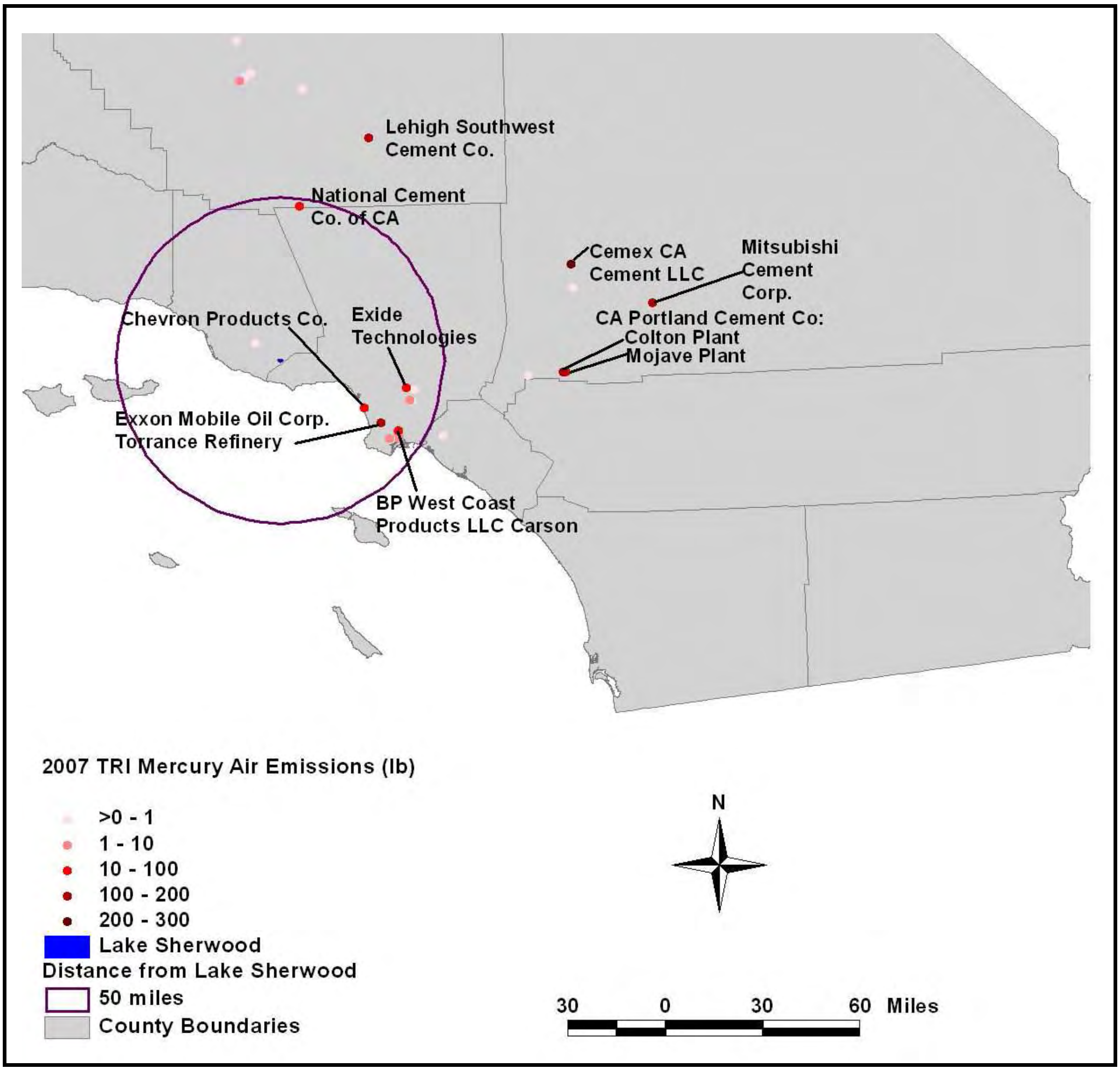


Figure E-7. Location of Facilities Reporting Mercury Emissions within 50 miles of Lake Sherwood

Table E-5. Mercury Emissions Reported in the 2007 USEPA Toxic Release Inventory

Facility Name	Total Air Emissions (lbs)
Cemex California Cement LLC	273.30
Exxon Mobil Oil Corp - Torrance Refinery	162.70
Mitsubishi Cement Corp.	160.00
Lehigh Southwest Cement Co.	144.12
California Portland Cement Co. Colton Plant	124.61
National Cement Co Of California Inc	55.00
Exide Technologies	51.74
Bp West Coast Products LLC Carson	17.10
Chevron Products Co. Div Of Chevron USA Inc.	14.90
California Portland Cement Co. Mojave Plant	13.40
Ultramar Inc. Wilmington Refinery	4.51
Conoco Phillips Co La Refinery Wilmington Plant	3.80
Tesoro Los Angeles Refinery	3.40
Arnco	3.00
Conoco Phillips Co Los Angeles Refinery Carson Plant	2.30
Big West Of California Refinery	1.90
Mt. Poso Cogeneration	1.41
Commerce Refuse-To-Energy Facility	1.29
Rio Bravo Poso	0.96
Rio Bravo Jasmin	0.86
GHN Neon Inc	0.74
Tin, Inc DBA Temple Inland	0.60
Lunday-Thagard Co.	0.53
Alltech Associates Inc.	0.50
San Joaquin Refining Co Inc.	0.33
Big West Of California Refinery	0.30
Teledyne Imaging Sensors	0.20
Tricor Refining LLC	0.01
GS Roofing Products Co Inc. (DBA Certainteed)	0.01
Total	**1,043.52**

E.4.2. SIMULATED MERCURY DEPOSITION RATES

USEPA has undertaken several national-scale modeling efforts to characterize mercury deposition. For the 1997 Report to Congress, USEPA developed the Regional Lagrangian Model of Air Pollution (RELMAP) modeling (USEPA, 1997, Section 5.1.3) to produce gridded estimates of deposition rates. The report included comparisons between wet deposition of mercury from local anthropogenic sources and a global-scale background concentration. While the RELMAP modeling is now believed to be outdated and does not fully reflect the current state of understanding of atmospheric chemistry leading to deposition of mercury (personal communication, O. Russell Bullock, USEPA, to J. B. Butcher, Tetra Tech, July 25, 2001), these results suggested that the deposition of mercury in the southwest has a strong global or long-range component.

The RELMAP modeling had considerable uncertainty, particularly for the Southwest, where monitoring data were scarce and dry deposition of mercury may play a larger role. The broad-scale RELMAP modeling also could not take into account the effects of local topography on deposition, nor did it account for the interaction of chloride ions in power plant emissions with elemental mercury to form species such as mercuric chloride that are subject to more rapid deposition. USEPA subsequently developed a more sophisticated regional mercury transport model (Community Multiscale Air Quality (CMAQ-Hg)) based on the Models-3/CMAQ system (Byun and Ching, 1999), which incorporated a more sophisticated representation of mercury chemistry. In support of the Clean Air Mercury Rule, the CMAQ-Hg model was used to predict mercury deposition for the 2001 base case on a 36x36 km model grid (USEPA, 2005). The baseline scenario was used to estimate wet and dry mercury deposition rates.

The CMAQ 2001 analysis was also conducted with US power plant emissions set to zero. Wet and dry rates of deposition were not distinguished in the output supplied to Tetra Tech. In most of the southwest region of the US, turning off US power plants in the model did not significantly impact the rate of total mercury deposition (see the bottom row of Table E-6). Simulated mercury deposition rates for the CMAQ grid cells that contain each impaired lake are summarized in Table E-6.

Table E-6. CMAQ 2001 Output for Grid Cells Underlying the Watersheds of the Mercury Impaired Lakes

Component	Mercury Deposition Rate g/km^2/yr		
	El Dorado Park Lakes	Puddingstone Reservoir	Lake Sherwood
Wet – Baseline	9.6988	4.1082	2.9007
Dry – Baseline	77.5962	29.8365	12.1748
Total – Baseline	87.2950	33.9447	15.0755
Total – Zero Power Plant Emissions	87.2822	33.9293	15.0682

An additional run of the CMAQ model was undertaken for 2002 meteorological conditions, with alterations to the functional description of processes leading to the dry deposition of mercury. The 2002 CMAQ results are summarized in Table E-7. At the El Dorado Park lakes, the CMAQ 2001 simulation predicts higher rates of both wet and dry deposition, and the total deposition rate is approximately 44 percent higher than the 2002 simulation results. For Puddingstone Reservoir, the 2001 simulation predicts a higher wet deposition rate, but the 2002 simulation predicts a higher dry deposition rate. The total deposition rate predicted by the 2002 simulation is approximately 11 percent higher than the 2001 simulation. At Lake Sherwood, the 2001 simulation predicts higher rates of both wet and dry deposition, and the total deposition rate is approximately 240 percent higher than the 2002 prediction. Both the 2001 and 2002 CMAQ simulations estimate that the rate of dry mercury deposition is higher than the rate of wet mercury deposition. The CMAQ 2002 results are assumed to represent a more accurate estimate of dry deposition because this model included alterations to the functional deposition of processes associated with the dry deposition of mercury. Therefore, the dry deposition is primarily based on the CMAQ 2002 results, with the exception of Lake Sherwood. For Lake Sherwood, the CMAQ 2001 dry deposition rates will be used as described in Section E.4.5).

Table E-7. CMAQ 2002 Output for Grid Cell Underlying the Watersheds of the Mercury Impaired Lakes

	Mercury Deposition Rate g/km^2/yr		
Component	**El Dorado Park Lakes**	**Puddingstone Reservoir**	**Lake Sherwood**
Wet	3.5642	2.5400	0.5863
Dry	57.0656	35.2323	5.6784
Total	60.6298	37.7723	6.2647

E.4.3. WET DEPOSITION MONITORING

Deposition may either occur in wet form (associated with precipitation) or dry form (associated with particulate or gaseous settling). Wet deposition is monitored at select locations across the country by the Mercury Deposition Network (MDN). In May 2006, a MDN station was installed at Converse Flats, California in San Bernardino County. Quality-assured data are available from the MDN website through December 2007; provisional data were provided to Tetra Tech through December 2008.

Figure E-8 through Figure E-10 show the measurements of precipitation, mercury concentration, and mercury deposition at Converse Flats. Points connected by lines indicate successive weeks with measured precipitation and mercury wet deposition measurements. Single points indicate that no precipitation fell the week prior or the week after. Weekly precipitation measurements range from 0 to 130 mm (0 to 5.1 inches). The average observed mercury concentration during precipitation events is 18.5 ng/L, and the volume-weighted average concentration is 11.2 ng/L. Weekly deposition rates measured at Converse Flats range from 0 to 1,442 ng/m^2, and the average annual deposition rate, including weeks with zero precipitation, is 3.48 g/km^2/yr.

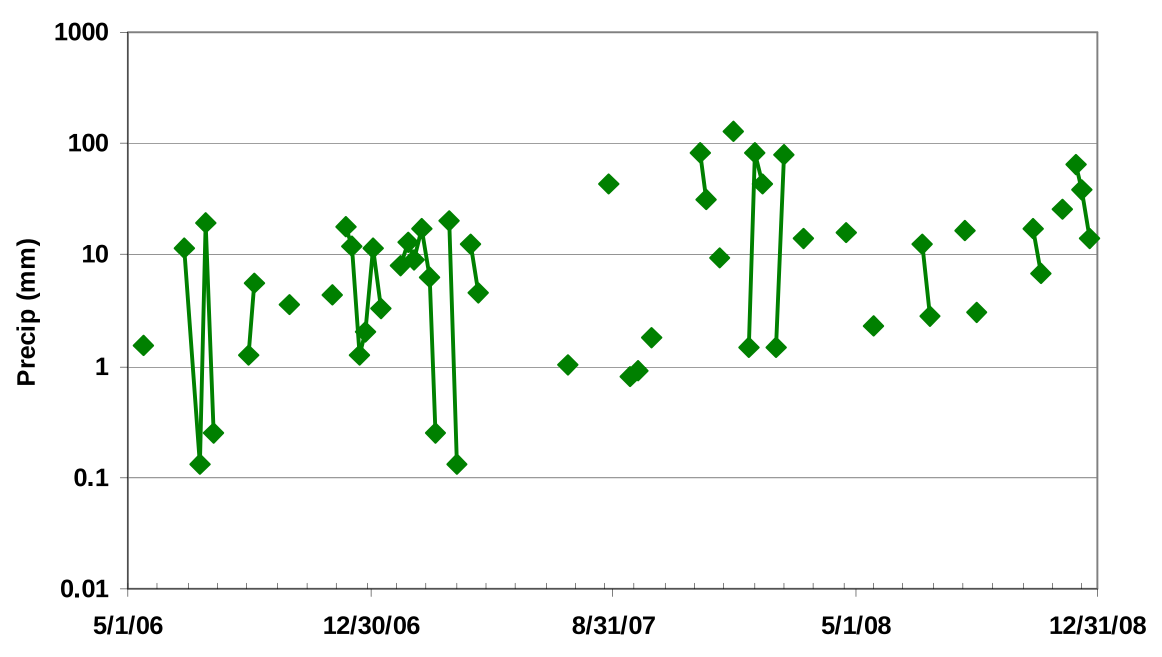


Figure E-8. Weekly Precipitation Measurements at CA94

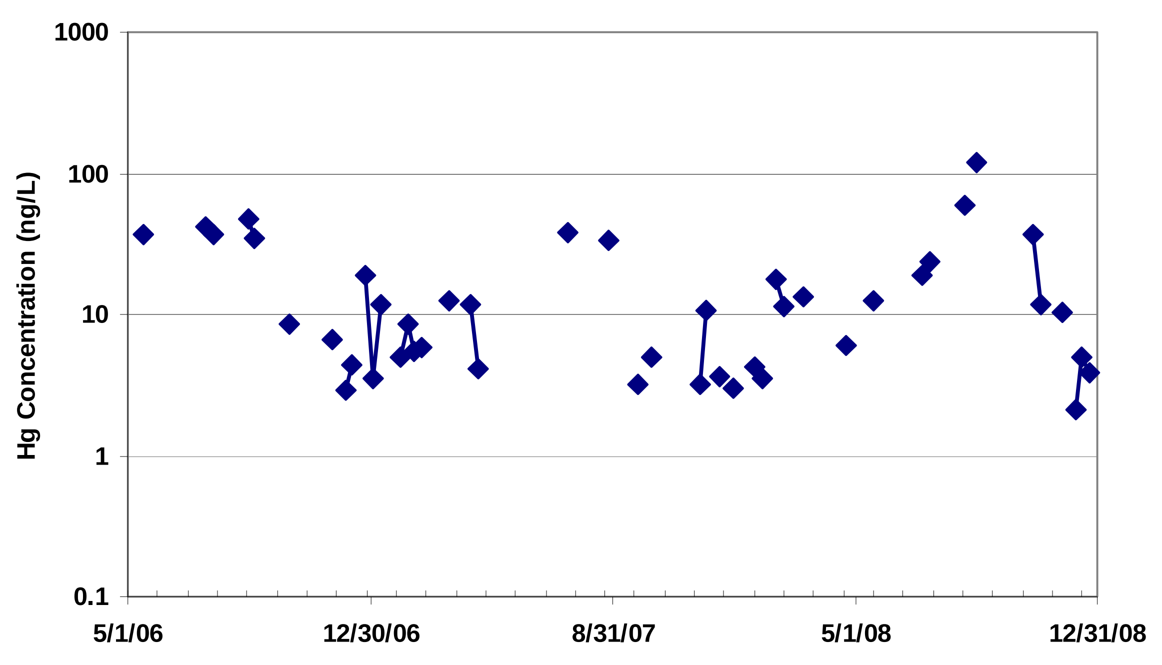


Figure E-9. Weekly Mercury Concentrations at CA94

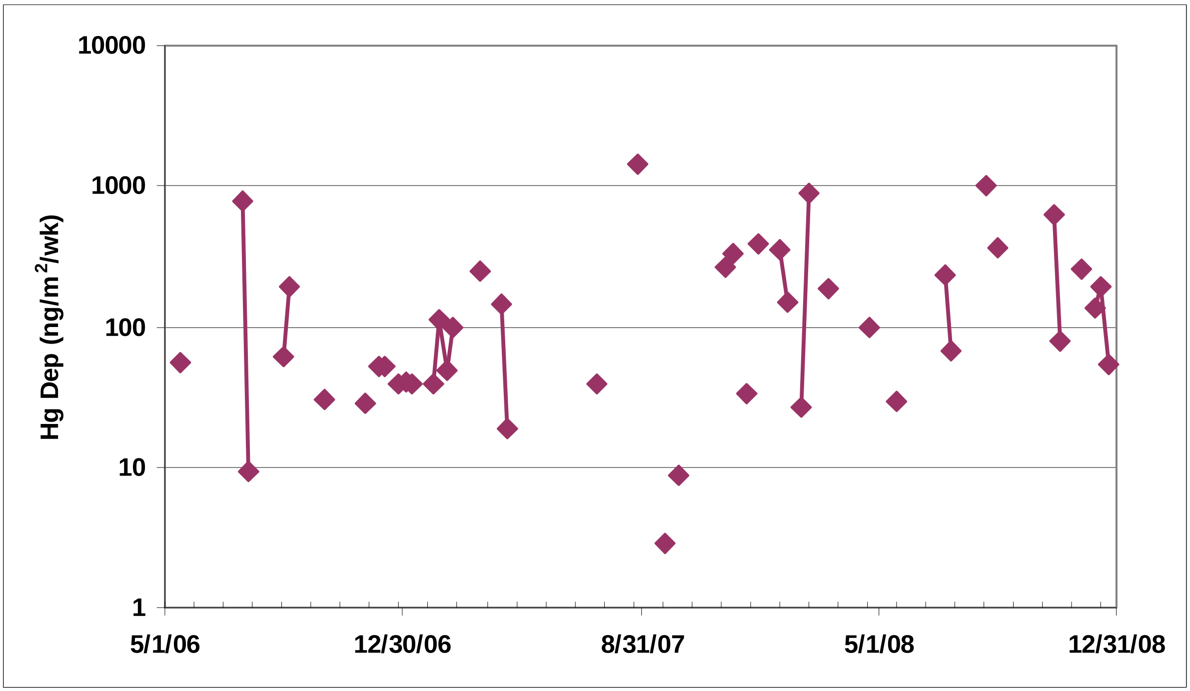


Figure E-10. Weekly Mercury Wet Deposition Rates at CA94

E.4.4. WET DEPOSITION ESTIMATION

MDN station CA94 (Converse Flats) was installed in May 2006 to support development of mercury TMDLs in Southern California. During the period of record, the average annual wet deposition rate is 3.48 g/km^2/yr. In addition to mercury concentrations, this site also monitored nitrate and sulfate wet deposition concentrations through the National Atmospheric Deposition Program (NADP). Deposition of particulate and reactive gaseous mercury derived from combustion sources is often correlated with nitrate and sulfate deposition. A multiple regression on nitrate and sulfate deposition concentrations measured at CA94 yields an estimate of mercury concentration with an R^2 of 0.54. Figure E-11 shows a comparison of the measured and estimated mercury concentrations resulting from the following equation:

$$\mathrm{LOG}_{10}\,(\mathrm{Hg,\ ng/L}) = 1.2102 + 0.1285\ \mathrm{LOG}_{10}\,(\mathrm{NO_3,\ mg/L}) + 0.4579\ \mathrm{LOG}_{10}\,(\mathrm{SO_4,\ mg/L}),\ R^2 = 53.6\%$$

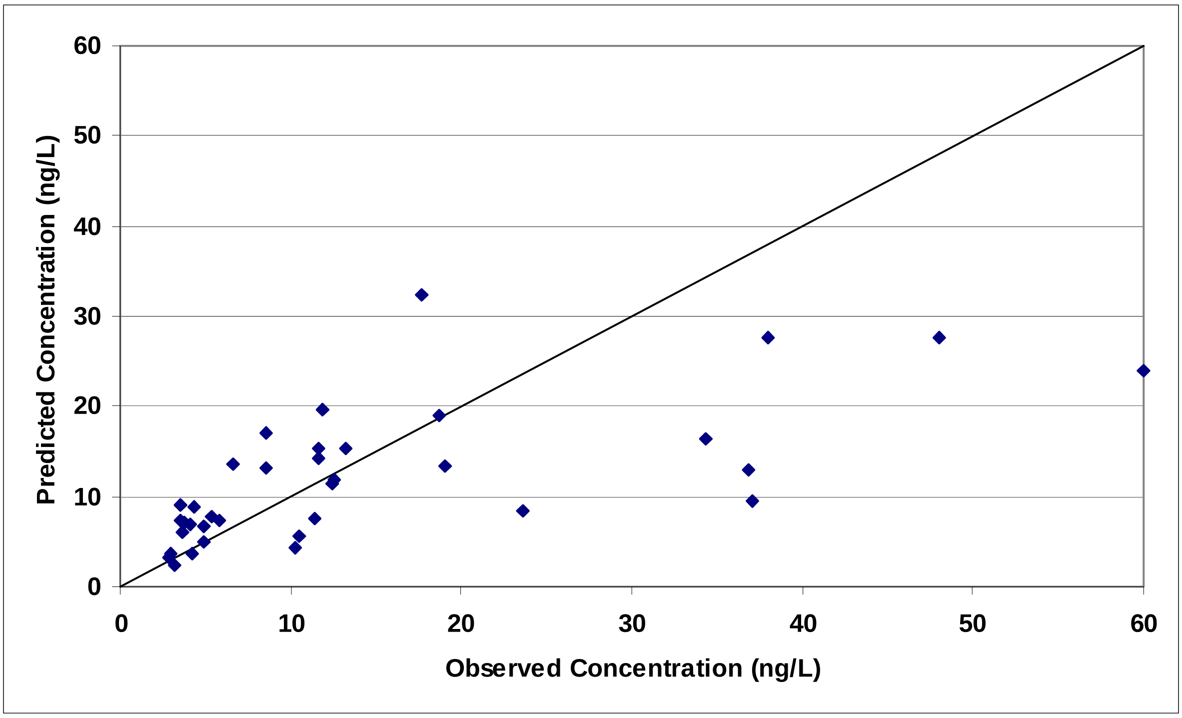


Figure E-11. Comparison of Measured and Predicted Mercury Wet Deposition Concentrations at Converse Flats

In order to use the mercury regression equation to estimate concentrations of mercury in precipitation at other locations, estimates of nitrate and sulfate concentrations are needed at each mercury impaired lake. Section E.3 explained how annual precipitation-weighted nitrate concentrations were obtained for the impaired lakes addressed by this TMDL. A similar method was used to obtain annual precipitation-weighted sulfate concentrations for the three mercury impaired lakes. Table E-8, Table E-9, and Table E-10 list the annual precipitation-weighted nitrate, sulfate, and predicted mercury concentrations for each lake and year.

Table E-8. Annual Precipitation-Weighted Concentrations at El Dorado Park Lakes

Year	Nitrate (mg-NO_3/L)	Sulfate (mg-SO_4/L)	Mercury (ng/L)
1983	1.07	0.45	11.37
1984	1.41	0.56	13.03
1985	1.38	0.56	12.91
1986	1.23	0.51	12.24
1987	1.30	0.54	12.61
1988	1.27	0.53	12.47
1989	1.34	0.55	12.82
1990	1.25	0.52	12.40
1991	1.08	0.47	11.59

Year	Nitrate (mg-NO_3/L)	Sulfate (mg-SO_4/L)	Mercury (ng/L)
1992	0.95	0.43	10.92
1993	0.87	0.40	10.49
1994	1.29	0.60	13.25
1995	0.48	0.26	7.93
1996	0.48	0.27	8.11
1997	0.88	0.42	10.72
1998	0.92	0.43	10.93
1999	1.33	0.57	12.99
2000	0.80	0.36	9.90
2001	0.71	0.39	10.08
2002	0.63	0.30	8.76
2003	0.73	0.30	9.02
2004	0.59	0.33	9.11
2005	0.39	0.30	8.26
2006	0.81	0.46	11.09

Table E-9. Annual Precipitation-Weighted Concentrations at Puddingstone Reservoir

Year	Nitrate (mg-NO_3/L)	Sulfate (mg-SO_4/L)	Mercury (ng/L)
1983	1.10	0.46	11.52
1984	1.54	0.62	13.82
1985	1.49	0.61	13.64
1986	1.32	0.55	12.77
1987	1.34	0.56	12.89
1988	1.31	0.55	12.78
1989	1.40	0.59	13.28
1990	1.30	0.55	12.80
1991	1.10	0.48	11.75
1992	0.90	0.41	10.65
1993	0.83	0.38	10.21
1994	1.30	0.60	13.28
1995	0.49	0.26	8.02
1996	0.48	0.27	8.13

Year	Nitrate (mg-NO_3/L)	Sulfate (mg-SO_4/L)	Mercury (ng/L)
1997	0.89	0.42	10.73
1998	0.92	0.43	10.93
1999	1.33	0.57	13.00
2000	0.80	0.36	9.90
2001	0.71	0.39	10.07
2002	0.62	0.29	8.63
2003	0.73	0.30	8.95
2004	0.58	0.33	9.10
2005	0.39	0.29	8.19
2006	0.81	0.46	11.10

Table E-10. Annual Precipitation-Weighted Concentrations at Lake Sherwood

Year	Nitrate (mg-NO_3/L)	Sulfate (mg-SO_4/L)	Mercury (ng/L)
1983	0.58	0.29	8.57
1984	1.24	0.49	12.03
1985	1.17	0.47	11.71
1986	0.97	0.41	10.75
1987	1.09	0.45	11.38
1988	1.05	0.44	11.23
1989	1.22	0.49	12.06
1990	1.15	0.47	11.75
1991	0.96	0.42	10.81
1992	0.70	0.34	9.49
1993	0.75	0.36	9.80
1994	1.26	0.58	13.00
1995	0.29	0.19	6.42
1996	0.47	0.26	7.95
1997	0.86	0.41	10.55
1998	0.91	0.43	10.84
1999	1.32	0.56	12.93
2000	0.78	0.36	9.79
2001	0.73	0.39	10.13

Year	Nitrate (mg-NO_3/L)	Sulfate (mg-SO_4/L)	Mercury (ng/L)
2002	0.63	0.30	8.83
2003	0.73	0.31	9.10
2004	0.58	0.33	9.05
2005	0.39	0.30	8.32
2006	0.80	0.46	11.00

Lake surface area and annual precipitation (see Table E-3) combined with precipitation-weighted mercury concentrations provide an estimate of annual wet deposition of mercury to a lake surface. Table E-11 presents the mercury load from wet deposition calculated for each lake.

Table E-11. Mercury Load from Wet Deposition to Mercury Impaired Lakes

	Mercury Load from Wet Deposition (g/yr)		
Year	El Dorado Park Lakes	Puddingstone Reservoir	Lake Sherwood
1980	1.10	11.26	4.94
1981	0.40	3.61	1.17
1982	0.39	3.61	1.61
1983	0.60	5.99	2.89
1984	0.38	4.91	1.93
1985	0.37	4.61	2.07
1986	0.14	2.11	0.61
1987	0.26	3.19	1.07
1988	0.51	5.76	2.36
1989	0.67	7.77	3.69
1990	0.73	8.02	3.33
1991	0.40	3.89	1.84
1992	0.62	5.86	2.40
1993	0.44	5.12	2.25
1994	0.48	4.88	2.20
1995	0.96	9.12	4.79
1996	0.32	2.62	1.36
1997	0.31	3.23	1.75
1998	0.17	5.51	4.20
1999	0.08	1.94	1.18

Year	Mercury Load from Wet Deposition (g/yr)		
	El Dorado Park Lakes	Puddingstone Reservoir	Lake Sherwood
2000	0.30	3.91	1.91
2001	0.49	4.99	2.53
2002	0.42	1.37	3.46
2003	0.32	3.38	2.16
2004	0.453	11.26	4.94
2005	1.10	3.61	1.17
2006	0.40	3.61	1.61
Average	**0.39**	**4.86**	**2.40**

Table E-12 compares the average wet deposition rate based on monitoring data and regression analyses to the CMAQ 2001 and 2002 runs. The calculated rates are generally in agreement with the CMAQ runs with the exception of Lake Sherwood where calculated rates are 50 percent higher than the greater of the two CMAQ estimates. As discussed in Section E.4.6, the calculated wet deposition rates will be used for TMDL development; the CMAQ model runs are only presented for comparison (Note: There are only two published CMAQ model runs for consideration in the analyses and only grid-scale model output was available; therefore, additional model runs could not be performed for TMDL development).

Table E-12. Summary of Wet Deposition Estimates to Each Impaired Lake

Deposition Load	El Dorado Park Lakes	Puddingstone Reservoir	Lake Sherwood
Lake Surface Area (km^2)	0.143	1.021	0.554
Calculated Wet (g/yr)	0.453	4.86	2.40
CMAQ 2001 Wet (g/yr)	1.39	4.19	1.61
CMAQ 2002 Wet (g/yr)	0.510	2.59	0.325

Note: Shaded cells represent the selected wet deposition loads for each waterbody.

E.4.5. DRY DEPOSITION

Although there are few direct measurements to support well-characterized estimates, dry deposition of mercury often is assumed to be approximately equal to wet deposition (e.g., Lindberg et al., 1991; Lindqvist et al., 1991). This assumption is not always valid in the southwest. Dry and wet deposition were measured in the Pecos River basin of eastern New Mexico in 1993–1994 (Popp et al., 1996). Average weekly deposition rates were calculated to be 140 ng/m^2-wk of mercury from dry deposition and 160 ng/m^2-wk of mercury from wet deposition. These data demonstrate the importance of both dry and wet deposition as sources of mercury. Early throughfall studies in a coniferous forest indicate that dry deposition beneath a forest canopy could be on the order of 50 percent of the wet deposition signal (Lindqvist et al., 1991). However, the local university cooperator at the Caballo, New Mexico MDN station (NM10) estimated dry deposition as up to six times wet deposition at this arid site (Caldwell et al., 2003). A recent study sponsored by the Arizona Department of Environmental Quality indicates that dry deposition may be two to nine times higher than wet deposition (Tetra Tech, 2008).

Atmospheric dry deposition involves three groups of mercury species: reactive gaseous mercury (RGM), aerosol particulate mercury (Hg-P), and gaseous elemental mercury (Hg(0)). All three forms may deposit to land and water surfaces, but there are significant differences in chemistry and rates. Hg(0) is the dominant species in terms of ambient concentration; however, net deposition rates are much higher for the other forms (Lindberg et al., 1992).

Dry mercury deposition to water surfaces is typically comprised of the reactive gaseous and particulate forms of mercury only. Elemental mercury contributes to the loading to land surfaces as it is accumulated in vegetation through stomatal vapor uptake (Eriksen et al., 2003). Contributions to soil systems occur as vegetative material falls and decays on the soil surface. No direct measurements of dry deposition are available for this region. As a conservative estimate, the greater of the two CMAQ simulation results (Section E.4.2) may be used to estimate the rate of dry mercury deposition to the land (direct deposition plus foliar accumulation).

The TMDL process for mercury loading generally divides loading into two components: watershed loading and direct atmospheric deposition to the water surface. Though the watershed load typically originates from atmospheric sources, whether historic, recent, near, or distant, delivery to the waterbody depends on runoff, erosion, and sedimentation processes that occur on the land surface and in the tributary network. In some cases, direct sources of mercury loading may be present in a watershed, such as mine tailings or geological formations with naturally high mercury concentrations. Watershed loading models that predict runoff and sediment delivery to a receiving waterbody are typically coupled with direct measurements of mercury concentrations in the sediments and water column of major tributaries to estimate mercury loading from the watershed.

The direct loading from the atmosphere to water surfaces may be estimated as wet deposition plus total dry deposition minus the foliar accumulation component. Because the CMAQ model runs estimate dry deposition to the land surface, the output includes the amount of mercury that has accumulated in leafy material (via stomatal uptake) and is eventually deposited to the land surface following leaf fall and decomposition. Direct dry deposition to a waterbody should not include this component. Foliar accumulation typically accounts for approximately 7 g/km^2/yr in the southwest region (Tetra Tech, 2008).

The CMAQ 2002 results are assumed to represent a more accurate estimate of dry deposition because this model included alterations to the functional deposition of processes associated with the dry deposition of mercury. Therefore, the dry deposition is primarily based on the CMAQ 2002 results, with the exception of Lake Sherwood. For Lake Sherwood, the 2001 results for dry deposition are assumed because 1) subtracting the foliar accumulation rate from the 2002 results would yield a negative deposition rate and 2) the net 2001 dry deposition rate (minus foliar accumulation) is similar to the 2002 gross dry deposition rate (see Table E-13 for a comparison of the 2001 and 2002 CMAQ results). It is important to note that there are only two published CMAQ model runs and only grid-scale model output was available; therefore, additional model runs could not be performed for TMDL development. The total dry deposition rates to each lake surface are summarized in Table E-14.

Table E-13. CMAQ Output for Grid Cells Underlying the Watersheds of the Mercury Impaired Lakes

	Mercury Deposition Rate (g/km^2/yr)		
Component	**El Dorado Park Lakes**	**Puddingstone Reservoir**	**Lake Sherwood**
CMAQ 2001 Dry Deposition	77.5962	29.8365	12.1748
CMAQ 2002 Dry Deposition	57.0656	35.2323	5.6784

Note: Shaded cells represent the selected dry deposition rates for each waterbody.

Table E-14. Summary of Dry Deposition Estimates to Each Impaired Lake

Calculation Term	El Dorado Park Lakes	Puddingstone Reservoir	Lake Sherwood
Dry Deposition Rate (g/km^2/yr)*	57.0656	35.2323	12.1748
Dry Deposition Rate Minus Foliar Accumulation Rate (g/km^2/yr)	50.0656	28.2323	5.1748
Lake Surface Area (km^2)	0.143	1.021	0.554
Direct Dry Deposition Load (g/yr)	7.16	28.82	2.87

*Values are from shaded cells in Table E-13.

E.4.6. TOTAL MERCURY DEPOSITION TO LAKE SURFACES

As discussed previously, mercury deposition to a lake surface may occur in either wet or dry form. Table E-15 summarizes the average wet, dry, and total deposition estimates for each mercury impaired lake.

Table E-15. Summary of Direct Mercury Deposition to Impaired Lakes

Deposition Load	El Dorado Park Lakes	Puddingstone Reservoir	Lake Sherwood
Lake Surface Area (km^2)	0.143	1.021	0.554
Calculated Wet (g/yr)[1]	0.45	4.86	2.40
Direct Dry Deposition Load (g/yr)[2]	7.16	28.82	2.87
Total (g/yr)	7.61	33.68	5.27

[1] See Table E-12.

[2] See Table E-14.

E.5 Organochlorine Pesticides and PCBs Deposition

An additional source of Organochlorine (OC) Pesticides and PCBs is atmospheric deposition, which occurs as a result of both local and global atmospheric transport. Unfortunately, atmospheric deposition is difficult to measure, and detailed information on atmospheric deposition rates of most OC Pesticides and PCBs is not available for southern California. (SCCWRP recently undertook a study of OC Pesticides and PCBs deposition, but has withdrawn the results based on methodological concerns.) It is well established, however, that atmospheric deposition of OC Pesticides and PCBs plays a significant role in contamination of lakes, even in remote areas, including national parks in the western US (Landers et al., 2010; Hageman et al., 2006).

The current atmospheric flux of OC Pesticides and PCBs to the lakes is thus unknown. Two factors help simplify the TMDL analysis. First, OC Pesticides and PCBs derived from atmospheric deposition on the watershed are implicitly included in estimates of watershed loading. Second, hydrophopic OC Pesticides and PCBs both deposit to and degas from waterbodies, and it is the net balance of these processes that is of most concern for the TMDL. The OC Pesticides and PCBs of concern are no longer in use, with atmospheric deposition rates declining, and elevated fish tissue concentrations appear to be largely due to legacy sediment contamination. In such situations, the net flux is typically outward from contaminated waterbodies to the atmosphere, thus rendering the net atmospheric flux to the lake less than or near zero. In the early 1990s, PCBs and dieldrin in the Great Lakes showed a net loss to the atmosphere, although DDT was still accruing (Hoff et al., 1996). In 1998-99, Park et al. (2002) reported that Corpus Christi Bay in Texas was a net source of PCBs to the atmosphere, and that the annual water-surface-exchange fluxes of most pesticides appeared to be close to a net of zero.

Given these considerations, direct *net* loading to the lake surface is assumed to be near zero. The associated load allocation for atmospheric deposition is also set to zero.

(This page left intentionally blank.)

E.6 References

Byun, D.W. and J.K.S. Ching, eds. 1999. Science Algorithms of the EPA Models-3 Community Multiscale Air Quality (CMAQ) Modeling System. EPA/600/R-99/030. U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC.

Caldwell, C.A., R. Arimoto, P. Swartzendruber, and E.M. Prestbo. 2003. Air Deposition of Mercury and Other Airborne Pollutants in the Arid Southwest. Project Number A-00-1. Southwest Consortium for Environmental Research and Policy, San Diego, CA. http://www.scerp.org/projs/00rpts/A-00-l.pdf.

Eriksen, J., M.S. Gustin, D. Schorran, D. Johnson, S. Lindberg, and J. Coleman. 2003. Accumulation of atmospheric mercury in forest foliage. *Atmospheric Environment*, 37: 1613-1622.

Hageman, K.J., S.L. Simonich, D.H. Campbell, G.R. Wilson, and D. H. Landers. 2006. Atmospheric deposition of current-use and historic-use pesticides in snow at national parks in the western United States. *Environmental Science and Technology*, 44(3): 855-859.

Hoff, R.M., W.M.J. Strachan, C.W. Sweet, C.H. Chan, M. Schackleton, T.F. Bidelman, K.A. Brice, D.A. Burniston, S. Cussion, D.F. Gatz, K. Harlin, and W.H. Schroeder. 1996. Atmospheric deposition of toxic chemicals to the Great Lakes: A review of data through 1994. *Atmospheric Environment*, 30(20): 3505-3527.

Landers, D.H., S.M. Simonich, D. Jaffe, L. Geiser, D.H. Compbell, A. Schwindt, C. Schreck, M. Kent, W. Hafner, H.E. Taylor, K. Hageman, S. Usenko, L. Ackerman, J. Schrlau, N. Rose, T. Blett, and M.M. Erway. 2010. The Western Airborne Contaminant Assessment Project (WACAP): An interdisciplinary evaluation of the impacts of airborne contaminants in western U.S. national parks. *Environmental Science and Technology*, 40(10): 3174-3180.

Lindberg, S.E., R.R. Turner, T.P. Meyers, G.E. Taylor Jr., and W.H. Schroeder. 1991. Atmospheric concentrations and deposition of Hg to a deciduous forest at Walker Branch watershed, Tennessee, USA. Water, Air, and Soil Pollution 56: 577-594.

Lindberg, S.E., T.P. Meyers, G.E. Taylor Jr., R.R. Turner, and W.H. Schroeder. 1992. Atmosphere-surface exchange of mercury in a forest: results of modeling and gradient approaches. *Journal of Geophysical Research*, 97(D2): 2519-2528.

Lindqvist, O., K. Johansson, M. Aastrup, A. Andersson, L. Bringmark, G. Hovsenius, L. Hakanson, A. Iverfeldt, M. Meili, and B. Timm. 1991. *Mercury in the Swedish Environment: Recent Research on Causes, Consequences, and Corrective Methods*. Kluwer Academic Publishers, Dordrecht, Netherlands.

Park, J.-S., T.L. Wade, and S.T. Sweet. 2002. Atmospheric deposition of PAHs, PCBs, and organochlorine pesticides to Corpus Christi Bay, Texas. *Atmospheric Environment*, 36(10): 1707-1720.

Popp, C.J., D.K. Brandvold, K. Kirk, L.A. Brandvold, V. McLemore, S. Hansen, R. Radtke, and P. Kyle. 1996. Reconnaissance and Investigation of Trace Metal Sources, Sinks, and Transport in the Upper Pecos River Basin, New Mexico. Cooperative Agreement No. 3-FC-40-13830. New Mexico Institute of Mining and Technology, U.S. Department of the Interior and U.S. Bureau of Reclamation.

Tetra Tech. 2008. Arizona Mercury Air Deposition Data Analysis Memorandum to Karen Irwin, USEPA Region IX and Jason Sutter, Arizona Department of Environmental Quality, September 2008. Tetra Tech, Inc., Research Triangle Park, NC.

USEPA. 1997. Mercury Study Report to Congress, Vol. 3, Fate and Transport of Mercury in the Environment. EPA-452-R/97-005. U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards and Office of Research and Development, Washington, DC.

USEPA. 2005. Technical Support Document for the Final Clean Air Mercury Rule – Air Quality Modeling. US Environmental Protection Agency, Office of Air Quality Planning and Standards, Research Triangle Park, NC. March 2005. http://www.epa.gov/ttn/atw/utility/agm_oar-2002-0056-6130.pdf.

USEPA. 2009. Facility Report. U.S. Environmental Protection Agency, 2007 TRI Public Data Release. http://www.epa.gov/tri/tridata/tri07/index.htm.

Appendix F. Estimation of Loading During Dry Weather

(This page left intentionally blank.)

F.1 Introduction

USEPA Region IX is establishing TMDLs for impairments in nine lakes in the Los Angeles Region (Figure F-1). USEPA was assisted in this effort by the Los Angeles Water Quality Control Board (Regional Board) (Figure F-1). Impairments of these waterbodies include low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, algae, pH, mercury, lead, copper, chlordane, DDT, dieldrin, PCBs, and trash.

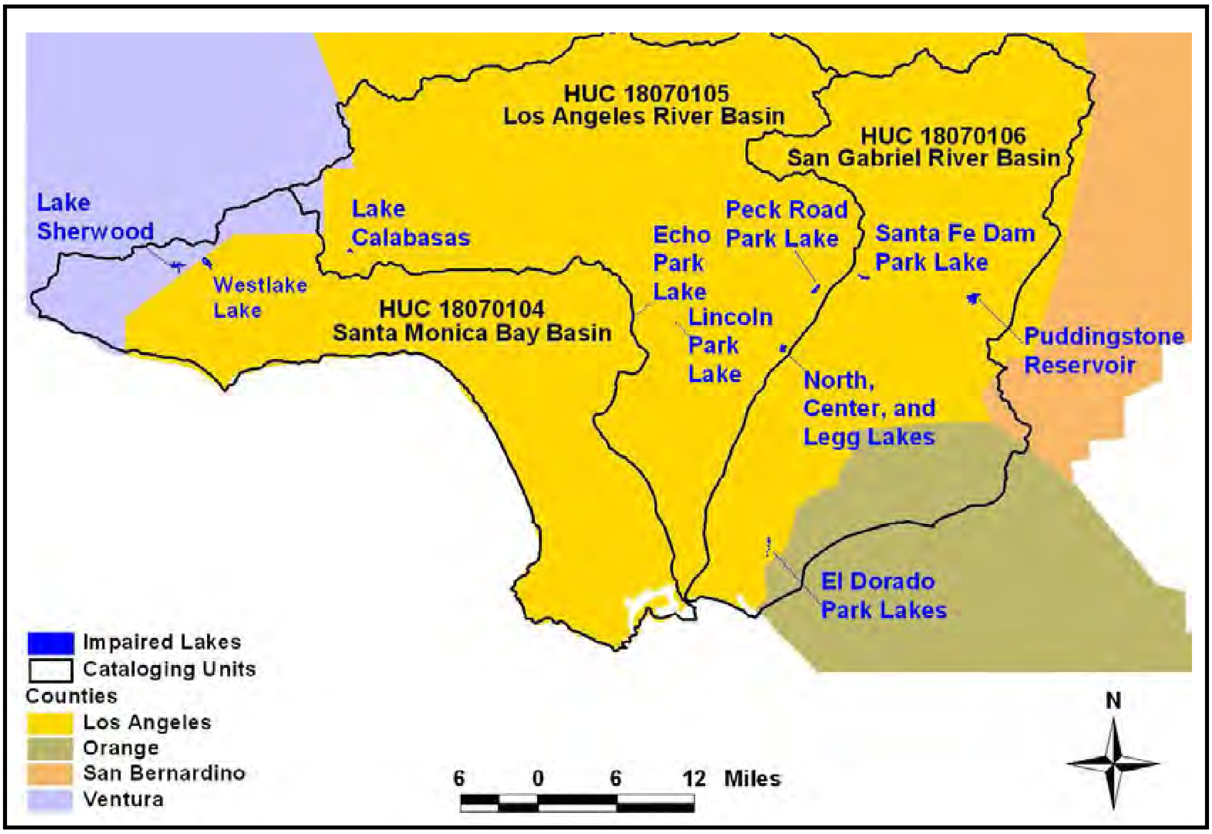


Figure F-1. Location of Impaired Lakes

In addition to pollutant loads delivered during storm events (discussed in Appendix D, Wet Weather Loading), it is important to account for loads that are delivered to a waterbody during dry weather. These may include point source discharges, imported water, direct groundwater or potable water inputs, and flows resulting from irrigation. This appendix discusses these sources of pollutant loading and the methods used to estimate average annual dry weather loading to each impaired lake.

Dry weather loading was estimated for constituents with significant dry weather loads. Since organochlorine pesticides (chlordane, DDT, dieldrin) and PCBs are strongly sorbed to sediment, loading and transport during dry weather flow is assumed to be insignificant for these constituents and no separate load calculation is performed for dry weather flows.

The calculated dry weather loads represent a portion of the existing pollutant load to each impaired waterbody. Estimates of loading from other sources are described in other sections or appendices of the TMDL report. The summation of loads from all sources will then be used to estimate existing loading to each lake.

(This page left intentionally blank.)

F.2 Dry Weather Loads from Storm Drains

Two Loading Simulation Program in C++ (LSPC) watershed models were previously developed by Tetra Tech to estimate wet weather loading of metals to the Los Angeles and San Gabriel rivers (Tetra Tech, 2004; Tetra Tech, 2005). The models are large- scale models that estimate loading from water reclamation facilities and major storm drains discharging to either one of the main river bodies or to a major tributary. These models were developed to address wet weather metals impairments and were not used to estimate nutrient loading during dry weather.

The lakes addressed by this TMDL are each in small drainages relative to the dry weather models discussed above. In addition, all of the impaired lakes except Lake Sherwood include nutrient TMDLs. For these reasons, Tetra Tech estimated dry-weather loading from upland areas delivered via storm drains based on dry weather monitoring studies conducted by the Southern California Coastal Water Research Project (SCCWRP).

In 2002 and 2003, SCCWRP measured dry weather flows and concentrations of nutrients, metals, and bacteria in six watersheds in the Los Angeles River Basin (Stein and Ackerman, 2007). Concentration data collected during this study are applicable to a majority of impairments addressed by this TMDL. Two of the watersheds (Walnut Creek and Ballona Creek) do not receive inputs from wastewater treatment plant (WWTP) dischargers, and so the flows and loads measured reflect inputs from storm drains and their upland catchments only. Because the watersheds for these TMDLs do not contain WWTPs, data for the Walnut Creek and Ballona Creek watersheds can be used to represent storm drain inputs to the TMDL watersheds. The SCCWRP study only monitored those parameters for which the corresponding waterbody was listed as impaired. Thus, nutrient monitoring data and corresponding dry weather loading estimates are only available for the Walnut Creek watershed (Ballona Creek is not impaired for nutrients); therefore, only the Walnut Creek data are applicable to calculate nutrient loads to the TMDL watersheds. Table F-1 summarizes the mean concentrations of nutrients measured in the Walnut Creek watershed, which were used to estimate storm drain nutrient loads.

Table F-1. Mean Pollutant Concentrations Measured During Dry Weather Periods

Parameter	Walnut Creek Watershed
Total ammonia (mg-N/L)	0.1
Nitrate plus nitrite (mg-N/L)	1.0
TKN (mg-N/L)	2.0
Total phosphate (mg-P/L)	0.3

Total nitrogen concentration in dry weather runoff may be estimated from the species monitored and is approximately 3 mg-N/L. Total phosphorus concentration was estimated based on a total phosphate concentration of 0.3 mg-P/L and organic fractions observed under median flow conditions on the San Gabriel River (Tetra Tech, 2007). Assuming the median of observed flows is representative of dry weather conditions, the organic fraction observed (50 percent) is a reasonable approximation. Thus, total phosphorus in dry weather flows is approximately 0.6 mg-P/L.

Dry weather flows in urban areas tend to exhibit diurnal variability due to the nature of the primary sources of flow (irrigation, car washing, etc.). In 2005, SCCWRP presented results of a more intensive flow monitoring study where data were collected at five minute increments over a three month period. During periods identified as dry weather, the areal flow rate (flow rate divided by contributing area) was approximately 180 $m^3/d/km^2$, or 2.6 in/yr, in three watersheds (Ackerman and Stein, 2005).

The TMDLs are allocated based on subwatershed and jurisdiction. A GIS environment was used to overlay the subwatersheds, jurisdictions, and storm drain coverage to estimate the upland area that may contribute dry weather loading via storm drains. These areas were then multiplied by the annual average dry weather flow rates (2.6 inches/yr) and loading rates for total nitrogen and phosphorus (1.77 lb-N/ac/yr) and total phosphorus (0.354 lb-P/ac/yr):

$$\frac{3.0mg-N}{L}\cdot\frac{2.6in}{yr}\cdot\frac{1ft}{12in}\cdot\frac{28.32L}{ft^3}\cdot\frac{43{,}560\,ft^2}{ac}\cdot\frac{1g}{1{,}000mg}\cdot\frac{1lb}{453.6g}=1.77lb-N/ac/yr$$

$$\frac{0.6mg-P}{L}\cdot\frac{2.6in}{yr}\cdot\frac{1ft}{12in}\cdot\frac{28.32L}{ft^3}\cdot\frac{43{,}560\,ft^2}{ac}\cdot\frac{1g}{1{,}000mg}\cdot\frac{1lb}{453.6g}=0.354lb-P/ac/yr$$

For example, 100 acres of area draining to a storm drain network would contribute the following flows and nutrient loads:

$$100ac\cdot\frac{2.6in}{yr}\cdot\frac{1ft}{12in}=21ac-ft/yr$$

$$100ac\cdot 1.77lb-N/ac/yr=177lb-N/yr$$

$$100ac\cdot 0.354lb-P/ac/yr=35.4lb-P/yr$$

F.3 Contributions from Other Dry Weather Inputs

The lakes addressed in this TMDL report may receive inputs from several sources during dry periods. The majority of the impaired lakes receive potable water or groundwater as a supplemental source to offset evaporation and keep lake levels within a normal range.

Water used for irrigation around each lake also has the potential to deliver pollutants via runoff into the lake. Unit areas of urban land were set up for the LSPC modeling subbasins surrounding each impaired lake to estimate the percentage of irrigation water applied that would enter the lake via runoff or interflow.

During the 2009 and 2010 water quality monitoring events, known and accessible dry weather inputs were sampled for mercury, nutrients, and Organochlorine (OC) Pesticides and PCBs. (The groundwater, potable water, and reclaimed water sources at El Dorado Park lakes were also sampled for total mercury and methylmercury.) Measured concentrations were applied to known or estimated volumes to calculate loading to each lake. If water quality or flow estimates were not available for a potential source, assumptions were made to estimate loading.

The following sample calculation estimates average annual nitrogen load given a flowrate of 250 ac-ft/yr and a total nitrogen concentration of 1.2 mg/L:

$$\frac{250ac-ft}{yr}\cdot\frac{1.2mg}{L}\cdot\frac{43{,}560\,ft^2}{ac}\cdot\frac{28.32L}{ft^3}\cdot\frac{1g}{1{,}000mg}\cdot\frac{1lb}{453.6g}=816lb-N/yr$$

Mercury loading is calculated in a similar manner, although the units on the concentration and load are different (the gram is used to summarize mercury loads because the pound is too large for the quantities delivered to the impaired waterbodies). To estimate mercury loading from an input that has an average flowrate of 250 ac-ft/yr with an average total mercury concentration of 10 ng/L, the following equation would be used,

$$\frac{250ac-ft}{yr}\cdot\frac{10ng-Hg}{L}\cdot\frac{43{,}560\,ft^2}{ac}\cdot\frac{28.32L}{ft^3}\cdot\frac{1g}{1{,}000{,}000{,}000ng}=3.08g-Hg/yr$$

Table F-2 summaries the dry weather sources that may contribute pollutant loading to each impaired lake. The sections that follow describe the sources and loading estimates specifically for each waterbody.

Table F-2. Dry Weather Loading Sources to the Impaired Lakes

Lake/ Reservoir	Storm Drains	Potable Water	Groundwater	Irrigation	NPDES
Peck Road Park Lake	Yes	No	No	Yes	No
Lincoln Park Lake	No[1]	Yes	No	Yes	No
Echo Park Lake	No[2]	Yes	No	Yes	No
Lake Calabasas	Yes	Yes	No	Yes	No
El Dorado Park Lakes	No	Yes	Yes	Yes	No

Lake/ Reservoir	**Storm Drains**	**Potable Water**	**Groundwater**	**Irrigation**	**NPDES**
North, Center, Legg Lakes	Yes	No	Yes	Yes	No
Puddingstone Reservoir	Yes	No	No	Yes	No
Santa Fe Dam Park Lake	No	Yes	Yes	Yes	No
Lake Sherwood	Yes	No	No	NA[3]	No

[1] The storm drain network passes under Lincoln Park Lake with no outfalls to the lake.

[2] Dry weather flows from the storm drain network are diverted downstream of Echo Park Lake.

[3] Information regarding irrigation was not collected because this TMDL does not address nutrient impairments for Lake Sherwood.

F.4 Peck Road Park Lake

Peck Road Park Lake is located in the Los Angeles River Basin. However, the Los Angeles County Department of Public Works (LACDPW) diverts flows from the San Gabriel River to Peck Road Park Lake via the Santa Fe Diversion Channel.

Impairments of this lake include low dissolved oxygen/organic enrichment, eutrophication (originally on the consent decree, but currently delisted), odor, lead, chlordane, dieldrin, DDT, PCBs, and trash. Dry weather contributions include storm drain inputs delivering dry weather flows from upland areas.

F.4.1 POLLUTANT LOADS FROM STORM DRAINS

Three subwatersheds comprise the drainage area to Peck Road Park Lake. The subwatershed draining the western part of the watershed via Santa Anita Wash is 12,686 acres, and the eastern subwatershed draining to Saw Pit Wash is 10,557 acres. There is an inwardly draining mining operation in the southern part of the eastern watershed that has been removed from the loading analysis. The subwatershed surrounding the lake is 321 acres. Each subwatershed drains to a storm sewer system so all allocations for the TMDLs are wasteload allocations, except for the trash TMDL which also has a load allocation.

Figure F-2 shows the MS4 stormwater permittees in the Peck Road Park Lake watershed. The western subwatershed is comprised of the county of Los Angeles, Sierra Madre, Arcadia, Monrovia, Angeles National Forest, and Caltrans areas. The eastern subwatershed is comprised of the county of Los Angeles, Monrovia, Duarte, Bradbury, Arcadia, Irwindale, Angeles National Forest, and Caltrans areas. The county of Los Angeles, Monrovia, Irwindale, Arcadia, and El Monte comprise the drainage around the lake. The park area is comprised of 152 acres adjacent to the lake (see the Peck Road Park Lake chapter for a more detailed map of the park area).

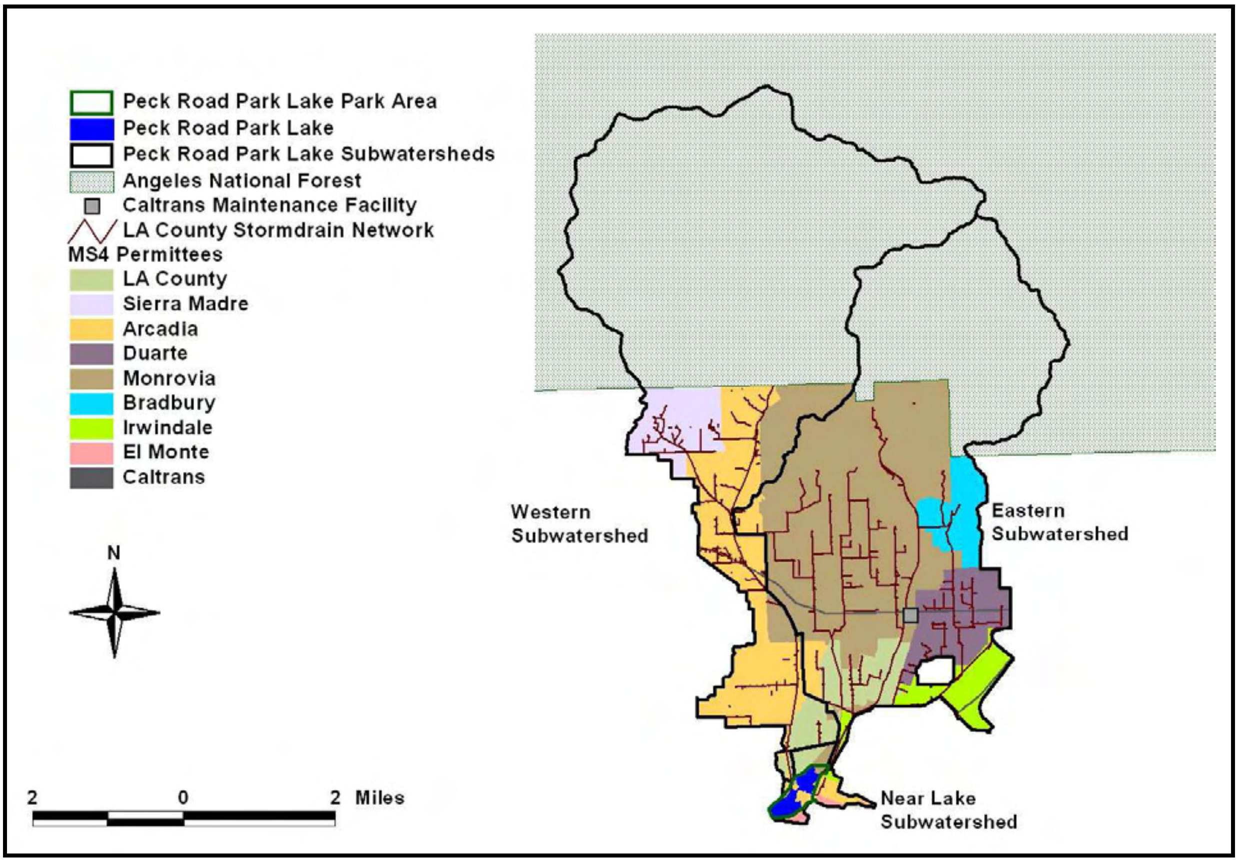


Figure F-2. MS4 Permittees and the County of Los Angeles Storm Drain Network in the Peck Road Park Lake Subwatersheds

Table F-3 summarizes the upland areas draining to Peck Road Park Lake by subwatershed and jurisdiction. Dry weather loading from the Angeles National Forest is assumed to be zero (wet weather loading is described in Appendix D, Wet Weather Loading). Table F-4 through Table F-6 list the estimated dry-weather flows and nutrient loads corresponding to these areas. Sample calculations are provided in Section F.2.

Table F-3. Land Use Areas (ac) Draining to Peck Road Park Lake

Subwater-shed	County of Los Angeles	Mon-rovia	Duarte	Brad-bury	Arcadia	Irwin-dale	Sierra Madre	El Monte	Caltrans	Angeles National Forest	Total
Western	245	611	0	0	2,030	0	679	0	16.9	**9,104**	**12,686**
Eastern	499	4,456	818	503	209	483	0	0	78.4	**3,511**	**10,557**
Near Lake	67.7	48.1	0	0	139	14.1	0	52.1	0	**0**	**321**
Total	**812**	**5,115**	**818**	**503**	**2,378**	**497**	**679**	**52.1**	**95.3**	**12,615**	**23,564**

Table F-4. Estimated Dry Weather Flows to Peck Road Park Lake (ac-ft/yr)

Subwatershed	County of Los Angeles	Mon-rovia	Duarte	Brad-bury	Arcadia	Irwin-dale	Sierra Madre	El Monte	Caltrans	Total
Western	52.9	132	0	0	439	0	147	0	3.65	**774**
Eastern	108	963	177	109	45.2	104	0	0	16.9	**1523**
Near Lake	14.6	10.4	0	0	30.0	3.05	0	11.3	0	**69.4**
Total	**175**	**1,105**	**177**	**109**	**515**	**107**	**147**	**11.3**	**20.6**	**2,366**

Table F-5. Estimated Dry Weather Nitrogen Loads to Peck Road Park Lake (lb/yr)

Subwatershed	County of Los Angeles	Mon-rovia	Duarte	Brad-bury	Arcadia	Irwin-dale	Sierra Madre	El Monte	Caltrans	Total
Western	432	1,077	0	0	3,579	0	1,197	0	29.8	**6,316**
Eastern	880	7,856	1,442	887	369	852	0	0	138	**12,424**
Near Lake	119	84.8	0	0	245	24.9	0	91.9	0	**566**
Total	**1,431**	**9,019**	**1,442**	**887**	**4,193**	**876**	**1,197**	**91.9**	**168**	**19,305**

Table F-6. Estimated Dry Weather Phosphorus Loads to Peck Road Park Lake (lb/yr)

Subwatershed	County of Los Angeles	Mon-rovia	Duarte	Brad-bury	Arcadia	Irwin-dale	Sierra Madre	El Monte	Caltrans	Total
Western	86.4	215	0	0	717	0	240	0	5.96	**1,263**
Eastern	176	1,571	288	177	73.7	170	0	0	27.6	**2,485**
Near Lake	23.9	17.0	0	0	49.0	4.97	0	18.4	0	**113**
Total	**286**	**1,803**	**288**	**177**	**840**	**175**	**240**	**18.4**	**33.6**	**3,861**

F.4.2 POLLUTANT LOADS FROM OTHER DRY WEATHER INPUTS

Water levels at Peck Road Park Lake are supplemented with flows from the San Gabriel River through a diversion channel. Estimates of flows and loads from this source are discussed in Appendix D (Wet Weather Loading) because this diversion is only used during wet weather.

A potable water source at Peck Road Park Lake is used to irrigate approximately 2 acres in a picnic area that is approximately 200 yards away from the lake. This area is fertilized when funding permits. Given the distance of this area from the lake, it is unlikely that irrigation or fertilization contributes significant nutrient loads to Peck Road Park Lake.

Other sources of nutrient loading may exist at Peck Road Park Lake such as wildlife and pets depositing feces that may wash off into the reservoir during rain events. While no bird feeding has been observed during recent fieldwork, birds do feed from trash cans and food litter at the park. It is difficult to estimate nutrient loading from animal wastes without information on populations and pet owner waste-disposal practices. Loads from animal wastes, as well as other sources that are difficult to quantify with the available information (e.g., park-area wastewater infrastructure systems) were not accounted for in the Peck Road Park Lake nutrient TMDLs because no additional loading was required to simulate observed nutrient concentrations at this lake (see Appendix A, Nutrient TMDL Development).

F.5 Lincoln Park Lake

Lincoln Park Lake is located in the Los Angeles River Basin. Impairments of this lake include low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, lead, and trash. Dry weather contributions to this lake include lake filling and irrigation/fertilization of adjacent parkland.

Figure F-3 shows the MS4 stormwater permittee comprising the Lincoln Park Lake watershed (the city of Los Angeles). Though the lake appears to be connected to the county of Los Angeles storm drain network, this system actually passes under Lincoln Park Lake and does not discharge stormwater or dry weather flows to the lake.

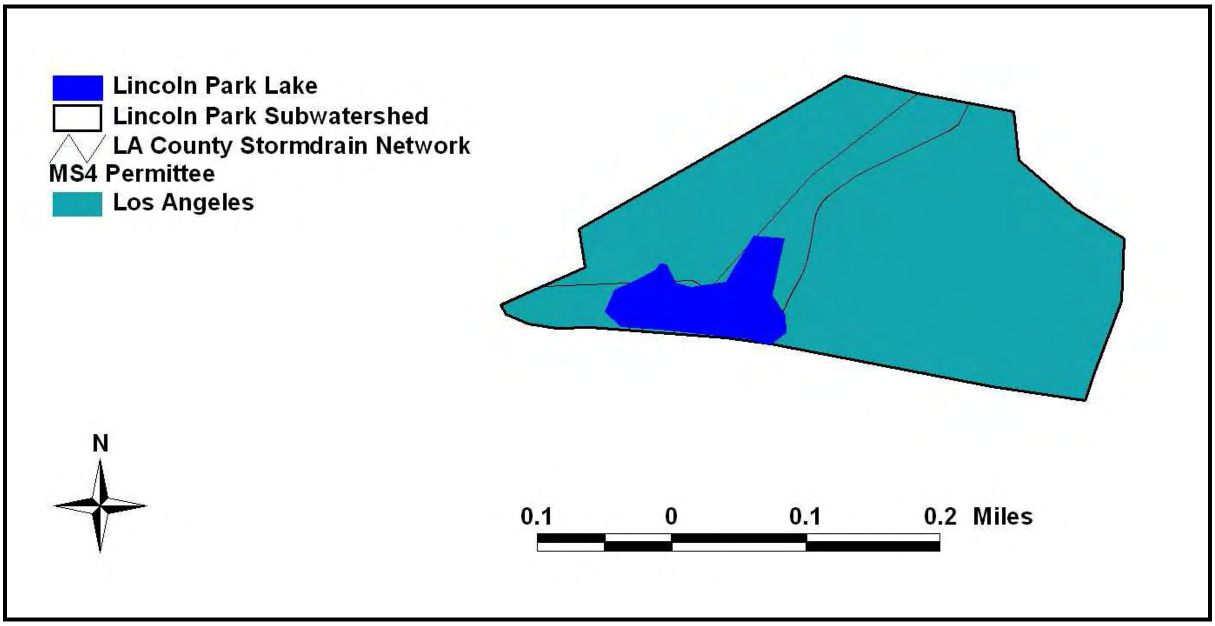


Figure F-3. MS4 Permittee and the County of Los Angeles Storm Drain Network in the Lincoln Park Lake Subwatersheds

F.5.1 POLLUTANT LOADS FROM STORM DRAINS

Lincoln Park Lake is not hydraulically connected to the county of Los Angeles storm drain system although part of the system passes under the lake. Thus, dry weather loads to this lake delivered from storm drains are zero (wet weather loads from the watershed are discussed in Appendix D).

F.5.2 POLLUTANT LOADS FROM OTHER DRY WEATHER INPUTS

A potable water source at Lincoln Park Lake is used for lake filling as well as irrigation of parkland. Based on monthly usage summaries for May 2007 through April 2009, the average annual usage is 30.8 ac-ft/yr. All usage reported is applied directly to the lake to supplement lake levels. Park staff indicate that 32 acres surrounding the lake are irrigated with an additional 1 foot of potable water annually. Water is observed to percolate into the ground. The annual net evapotranspiration minus precipitation depth is 34.5 inches based on CIMIS data for this zone and precipitation data for a nearby weather station (Appendix D, Wet Weather Loading). Thus, the majority of the applied water likely percolates into the ground or is lost to evapotranspiration. A unit area model setup in LSPC for this subbasin indicates that approximately 5.6 percent of applied irrigation water reaches Lincoln Park Lake.

The potable water input at Lincoln Park Lake was sampled for water quality by USEPA and the Regional Board in August 2009. Table F-7 summarizes the observed water quality and estimated loads from lake filling and irrigation with the potable water source. See Section F.3 for sample calculations.

Table F-7. Summary of Potable Water Quality and Resulting Direct Loads to Lincoln Park Lake

Parameter	Concentration (mg-N/L or mg-P/L)	Load from Irrigation (lb-N/yr or lb-P/yr)	Supplemental Water Addition (lb-N/yr or lb-P/yr)
Ammonia-N	0.335	1.64	28.1
Nitrate- N	0.33	1.61	27.6
Nitrite-N	0.03	0.147	2.51
TKN (mg-N/L)	0.531	2.60	44.5
Orthophosphate (mg-P/L)	0.017	0.083	1.42
Total Phosphorus (mg-P/L)	0.118	0.58	9.88
Total Nitrogen (calculated) (mg-N/L)	0.891	4.36	74.6

Note: Potable water concentrations are from data collected at Lincoln Park Lake.

The area surrounding Lincoln Park Lake (32 acres) is fertilized twice per year with 16-6-8 fertilizer at a rate of 7.5 lb/1,000 ft^2. The technical sheet for the product recommends applying this fertilizer at a rate of 6.25 lb/1,000 ft^2. It is difficult to estimate nutrient loading from fertilization as application methods, turf grass harvesting, and proximity of application to subsequent precipitation events impact transport via runoff.

During sampling events at Lincoln Park Lake, people were observed feeding the birds and a local person(s) was/were leaving piles of food along the shoreline of the lake. In addition, birds may feed from trash cans and food litter at the park. These practices increases nutrient loading to the lake by attracting birds and other animals that may deposit feces in and around the lake. Loads associated with feeding wildlife, as well as other sources that are difficult to quantify with the available information (normal wildlife populations, pets, park-area wastewater infrastructure, fertilization, etc.) were accounted for in a category called “Additional Parkland Loading.” During calibration of the BATHTUB model (see Appendix A, Nutrient TMDL Development), loads in this category were quantified by increasing inputs until simulated nutrient concentrations match those observed.

Precise bird counts for Lincoln Park Lake are not available; however, field notes indicate excess bird populations which are likely a significant portion of the nutrient loading associated with additional parkland areas. At Echo Park Lake, total phosphorus and total nitrogen loads of 78 lb-P/yr and 780 lb-N/yr were estimated for the approximately 1,000 birds observed to reside at that lake (Black and Veatch, 2010). The bird population at Lincoln Park like is likely one-half to one-quarter of that. Thus total phosphorus loads due to the bird population at Lincoln Park Lake likely range from 19.5 lb-P/yr to 39 lb-P/yr; total nitrogen loads range from 195 lb-N/yr to 390 lb-N/yr. The estimated loading from the resident bird population at Lincoln Park Lake is greater than the additional parkland loading estimated from the BATHTUB model. This overestimation may be due to 1) an inaccurate estimate of the bird population at Lincoln Park Lake, and 2) the conservative assumption that 100 percent of bird waste and associated nutrient loading reach the lake. Regardless of the accuracy of the estimated loading associated with bird waste, this analysis indicates that nutrient loading associated with the excess bird population comprises a significant portion of the additional parkland loading.

F.6 Echo Park Lake

Echo Park Lake is located in the Los Angeles River Basin. Impairments of this lake include odor, ammonia, eutrophication, algae, pH, copper, lead, chlordane, dieldrin, PCBs, and trash. Dry weather contributions to this lake include lake filling and irrigation/fertilization of adjacent parkland.

Two subwatersheds comprise the drainage area to Echo Park Lake. The subwatershed draining the northern part of the watershed is 614 acres, and the southern subwatershed drains 170 acres. Both subwatersheds drain to a storm drain system, so all allocations for the TMDLs are wasteload allocations, except the trash TMDL which also has a load allocation. Dry weather flows from the storm drain system are diverted downstream of Echo Park Lake (Black and Veatch, 2008). Figure F-4 shows the MS4 stormwater permittee in the Echo Park Lake watershed. Both subwatersheds are located entirely within the city of Los Angeles with a small portion of Caltrans area. The park is comprised of 15.5 acres of land adjacent to the lake.

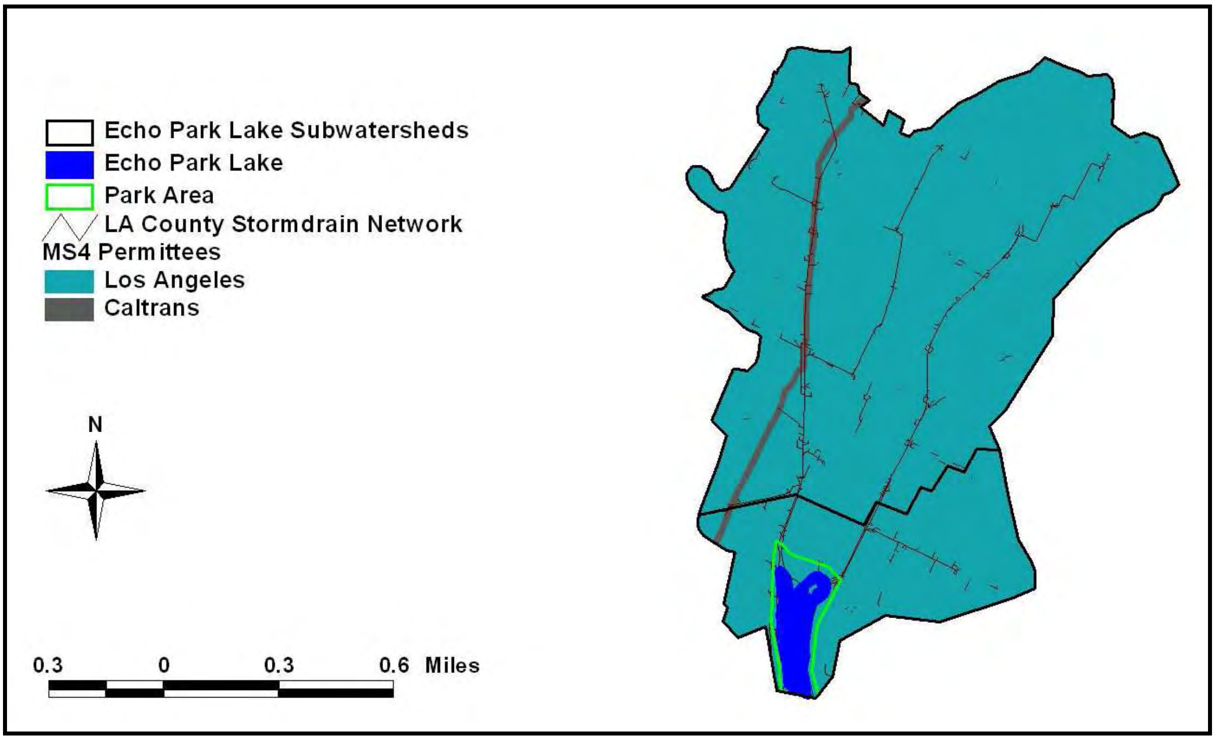


Figure F-4. MS4 Permittee and the County of Los Angeles Storm Drain Network in the Echo Park Lake Subwatersheds

F.6.1 POLLUTANT LOADS FROM STORM DRAINS

A recent study performed for the city of Los Angeles Bureau of Sanitation found that dry season flows through storm drains generally bypassed Echo Park Lake (Black and Veatch, 2008). Thus, dry weather loads to this lake delivered from storm drains are zero (wet weather loads from the MS4 stormwater system are discussed in Appendix D).

F.6.2 POLLUTANT LOADS FROM OTHER DRY WEATHER INPUTS

A potable water source at Echo Park Lake is used for both lake filling and irrigation of surrounding parklands. According to a hydrologic study of the park lake conducted by Black & Veatch (2008), 162 ac-ft/yr of potable water are pumped annually. Staff at Echo Park indicate that approximately 9 acres in the vicinity of the lake are irrigated at a rate of approximately 1 foot per year and that the water mainly percolates into the ground with occasional runoff into the lake. The annual net evapotranspiration minus precipitation depth is 34.5 inches based on CIMIS data for this zone and precipitation data for a nearby weather station (Appendix D, Wet Weather Loading). Thus, the majority of the applied water likely percolates into the ground or is lost to evapotranspiration. A unit area model set up in LSPC for this subbasin indicates that approximately 4.6 percent of applied irrigation water reaches Echo Park Lake. The remainder of the pumped water (162 ac-ft minus 9 ac-ft) is assumed applied directly to the lake to maintain water levels.

The potable water source was sampled and analyzed for nutrients and metals on August 4, 2009. Table F-8 summarizes the nutrient water quality data as well as the resulting loads from irrigation and lake filling. Calculated nitrogen loads assume that parameters analyzed at less than detection have concentrations equivalent to ½ the detection limit. See sample calculations in Section F.3.

Table F-8. Summary of Potable Water Quality and Resulting Direct Loads to Echo Park Lake

Parameter	Concentration (mg-N/L or mg-P/L)	Load from Irrigation (lb-N/yr or lb-P/yr)	Supplemental Water Addition (lb-N/yr or lb-P/yr)
Ammonia-N	<0.03	0.017	6.24
Nitrate- N	0.9	1.024	374.45
Nitrite-N	<0.01	0.006	2.08
TKN	<0.456	0.259	94.86
Orthophosphate	0.020	0.023	8.32
Total Phosphorus	0.122	0.139	50.76
Total Nitrogen (calculated)	1.133	1.289	471

Note: Potable water concentrations are from data collected at Echo Park Lake.

Nine acres surrounding Echo Park Lake are fertilized twice per year with 16-6-8 fertilizer at a rate of 7.5 lb/1,000 ft^2. The technical sheet for the product recommends applying this fertilizer at a rate of 6.25 lb/1,000 ft^2. It is difficult to estimate nutrient loading from fertilization as application methods, turf grass harvesting, and proximity of application to subsequent precipitation events impact transport via runoff.

During sampling events at Echo Park Lake, people were observed feeding the birds and a local person(s) was/were leaving piles of food along the shoreline of the lake. This practice increases nutrient loading to the lake by attracting birds and other animals that may deposit feces in and around the lake. In addition, birds may feed from trash cans and food litter at the park. Loads associated with feeding wildlife, as well as other sources that are difficult to quantify with the available information (normal wildlife populations, pets, park-area wastewater infrastructure, fertilization, etc.) were accounted for in a category called "Additional Parkland Loading." During calibration of the BATHTUB model (see Appendix A, Nutrient TMDL Development), loads in this category were quantified by increasing inputs until simulated nutrient concentrations matched those observed.

A significant portion of loading from the additional local sources is likely due to excessive bird populations. According to a recent water quality modeling study conducted by Black and Veatch (2010), there is a year-round, resident bird population of approximately 1,000 Rock Doves and American Coots. Estimates of nutrient loading from these birds were based on literature values and an assumption that all waste generated by the birds would reach the lake (i.e., no uptake or trapping in adjacent areas). The estimated total phosphorus loading from these birds is 78 lb-P/yr, and the estimated total nitrogen loading is 780 lb-N/yr. Both loading estimates are greater than the additional parkland loading estimated from the BATHTUB model. This overestimation may be due to 1) an inaccurate estimate of the year-round bird population at Echo Park Lake, and 2) the conservative assumption that 100 percent of bird waste and associated nutrient loading reach the lake. Regardless of the accuracy of the estimated loading associated with bird waste, this analysis indicates that nutrient loading associated with the excess bird population comprises a significant portion of the additional parkland loading.

(This page left intentionally blank.)

F.7 Lake Calabasas

Lake Calabasas is located in the Los Angeles River Basin. Impairments of this lake include low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, and pH. A DDT impairment was previously reported for this lake but was delisted by the Regional Board in 2009. Dry weather contributions to this lake include lake filling with a potable water source, storm drain inputs delivering dry weather flows from surrounding development, and irrigation and fertilization of areas around the lake.

F.7.1 POLLUTANT LOADS FROM STORM DRAINS

One subwatershed draining 86.5 acres comprises the drainage area to Lake Calabasas. Figure F-5 shows the MS4 stormwater permittee in the Lake Calabasas watershed. The entire subwatershed is located in the city of Calabasas. This subwatershed drains to a storm drain system, so all allocations for the TMDLs are wasteload allocations.

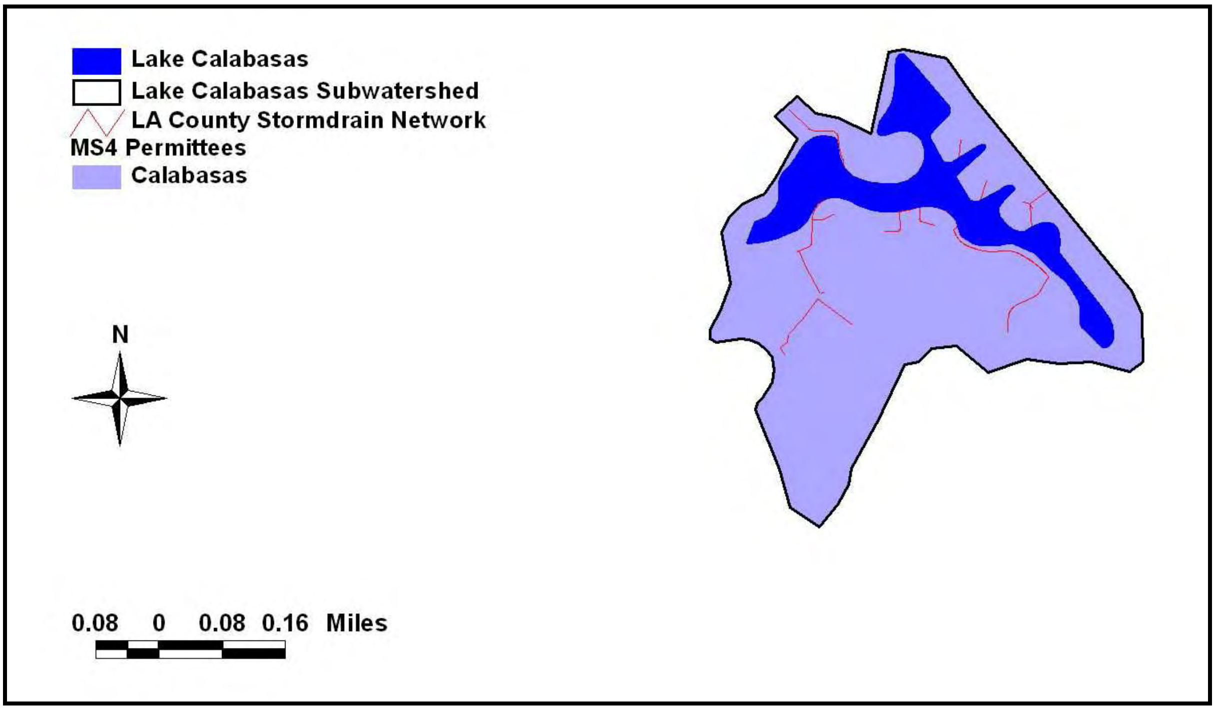


Figure F-5. MS4 Permittee and the County of Los Angeles Storm Drain Network in the Lake Calabasas Subwatersheds

Table F-9 summarizes the upland areas draining to Lake Calabasas as well as the associated dry weather flows and nutrient loads. Sample calculations are provided in Section F.2.

Table F-9. Land Use Areas and Associated Dry Weather Inputs to Lake Calabasas

Area (ac)	Flow (ac-ft/yr)	Total Nitrogen (lb/yr)	Total Phosphorus (lb/yr)
86.5	18.7	152	30.5

F.7.2 POLLUTANT LOADS FROM OTHER DRY WEATHER INPUTS

A potable water source at Lake Calabasas is used for both lake filling and irrigation of approximately 2 acres around the lake. Based on monthly data provided for 1995 to 2009, the average annual water usage is 60.8 ac-ft.

Only one day of irrigation usage has been monitored. On October 6, 2009, 0.011 ac-ft of potable water was applied over a 16-hour period. Assuming irrigation occurs five days a week throughout the year, the total applied irrigation volume is 2.86 ac-ft. The runoff depth based on this assumption is 17 inches. The annual net evapotranspiration minus precipitation depth is 37.6 inches based on CIMIS data for this zone and precipitation data for a nearby weather station (Appendix D, Wet Weather Loading). Thus, the majority of the applied water likely percolates into the ground or is lost to evapotranspiration. Staff at Lake Calabasas indicate that some of the applied water runs off into the lake. A unit area model setup in LSPC for this subbasin indicates that approximately 5.3 percent of applied irrigation water reaches Lake Calabasas.

The potable water source at Lake Calabasas was sampled for water quality on August 6, 2009. Table F-10 summarizes the nutrient parameters sampled. The total phosphorus concentration was analyzed as less than the detection limit of 0.016 mg-P/L. Phosphate measured greater than the detection limit and is used to estimate total phosphorus loading from this source. The total nitrogen concentration is calculated assuming the nitrite concentration is equal to half the detection limit. Nutrient loading associated with irrigation and lake filling are also presented. Estimated total nitrogen and total phosphorus loads from irrigation are 0.655 lb-N/yr and 0.00852 lb-P/yr, respectively, assuming irrigation occurs five days a week throughout the year. Assuming the remainder of the usage is discharged directly to the lake, the additional nutrient loading to Lake Calabasas is 252 lb-N/yr and 3.28 lb-P/yr. See Section F.3 for sample calculations.

Table F-10. Water Quality Data for the Potable Water Source at Lake Calabasas

Parameter	Concentration (mg-N/L or mg-P/L)	Load from Irrigation (lb-N/yr or lb-P/yr)	Supplemental Water Additions (lb-N/yr or lb-P/yr)
Ammonia-N	0.35	0.143	55.1
Nitrate- N	1.13	0.463	178
Nitrite-N	<0.01	0.0020	0.788
TKN	0.464	0.190	73.1
Orthophosphate	0.0208	0.00852	3.28
Total Phosphorus	<0.016	0.00852	3.28
Total Nitrogen (calculated)	1.60	0.655	252

Note: Potable water concentrations are from data collected at Lake Calabasas.

A portion of the common area surrounding Lake Calabasas is fertilized three times per year. The type of fertilizer applied varies depending on turf requirements. The average rate applied is approximately 1 lb per 250-275 sq ft of turf grass. The shrub and ground cover fertilizer is applied at approximately 5 lbs per 1,000 square feet. Staff at Lake Calabasas indicate that recommended rates are applied.

Residential properties surround Lake Calabasas, and maintenance staff indicate that some homeowners irrigate and fertilize their lawns. It is difficult to estimate nutrient loading from fertilization, from either the residential or common areas, because application rates and methods, turf grass harvesting, and proximity of application to subsequent precipitation events impact transport via runoff. Loads from fertilization, as well as other sources that are difficult to quantify with the available information (wildlife, pets, etc.) were not accounted for in the Lake Calabasas nutrient TMDLs because no additional loading was required to simulate observed nutrient concentrations at this lake (see Appendix A, Nutrient TMDL Development).

(This page left intentionally blank.)

F.8 El Dorado Park Lakes

The El Dorado Park lakes are located in the San Gabriel River Basin. Six lakes are located in the park. The northern four lakes are hydraulically connected and separate from the system comprised by the two southern lakes, also hydraulically connected. These lakes are listed as impaired by algae, ammonia, eutrophication, pH, copper, lead, and mercury. Dry weather contributions to these lakes include groundwater, potable water, and reclaimed water used for irrigation. No storm drains exist in the watershed that would deliver dry weather loads from areas outside of the park.

Two separate watersheds have been delineated for these separate lake systems. The subwatershed draining to the northern four lakes is comprised of 185 acres, and the subwatershed draining to the southern two lakes is comprised of 33.8 acres.

Figure F-6 shows the MS4 stormwater permittee that comprises both the northern and southern subwatersheds of the El Dorado Park lakes systems as well as the county of Los Angeles storm drain network. Although both watersheds are in the city of Long Beach incorporated area, there are no major drains that divert runoff directly to the lakes; a few small culverts pass water beneath walking paths and park roads. Because both watersheds are comprised solely of parklands that do not drain to a major storm drain system, the watershed loads to the El Dorado Park lakes are assigned load allocations in the TMDLs.

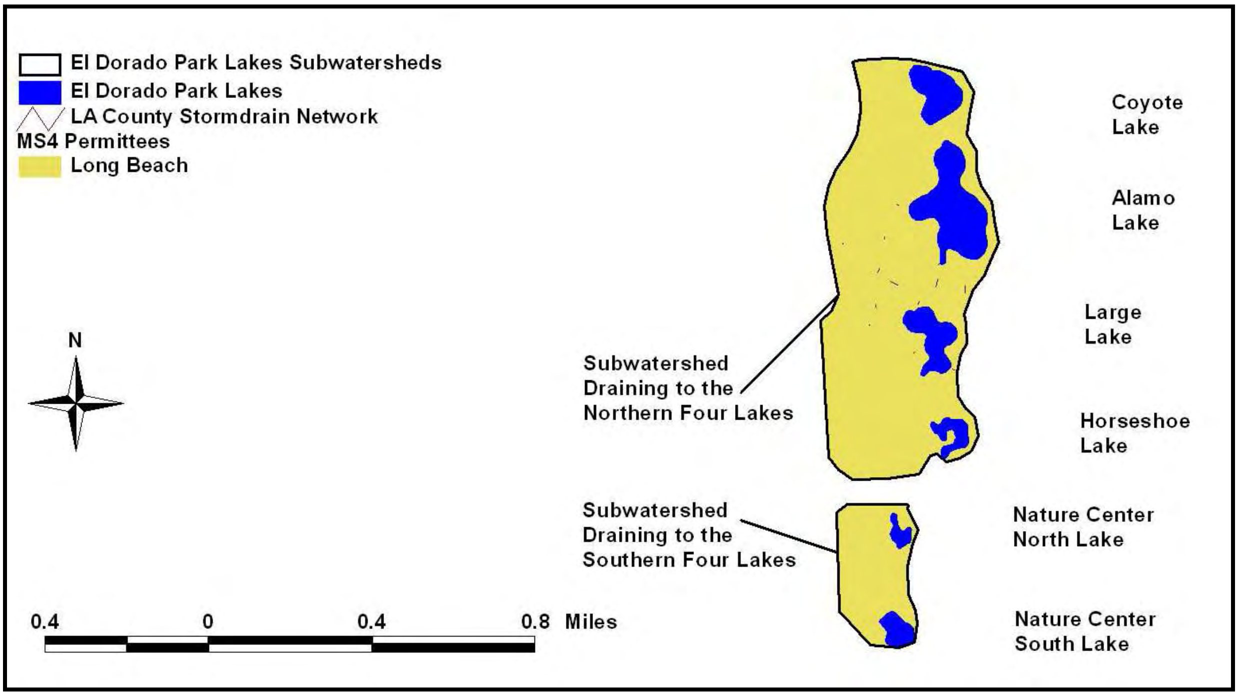


Figure F-6. MS4 Permittee and the County of Los Angeles Storm Drain Network in the El Dorado Park Lake Subwatersheds

F.8.1 POLLUTANT LOADS FROM STORM DRAINS

The El Dorado Park lakes watersheds are isolated from upland areas; no storm drains deliver dry weather runoff from outside the park. Thus dry weather loads from storm drains are zero (wet weather loads from the watershed are discussed in Appendix D).

F.8.2 POLLUTANT LOADS FROM OTHER DRY WEATHER INPUTS

The El Dorado Park lakes are comprised of two hydraulically separate systems. The northern four lakes receive groundwater that is pumped into Coyote Lake at a rate of approximately 110 ac-ft/yr. During a typical year, 80 percent of flows are discharged during the summer season (May through September) (personal communication, Keith McDonald, Long Beach Water, August 25, 2009).

The groundwater input was sampled twice during 2009 for water quality. Table F-11 summarizes the observed water quality for this input and presents the average of the observed values used for load estimation for the northern four lakes (this source is not expected to exhibit seasonal variations of water quality).

Table F-11. Groundwater Quality Data for the El Dorado Park Lakes

Parameter	2/26/2009	7/15/2009	Average
Ammonia (mg-N/L)	0.325	0.28	0.302
Nitrate (mg-N/L)	<0.01	<0.01	<0.01
Nitrite (mg-N/L)	<0.01	<0.01	<0.01
TKN (mg-N/L)	0.805	1.1	0.952
Orthophosphate (mg-P/L)	0.072	0.071	0.0715
Total Phosphorus (mg-P/L)	0.189	0.291	0.240
Total Nitrogen (calculated) (mg-N/L)	0.815	1.11	0.962
Total Mercury (ng/L)	142	131	136.5
Methylmercury (ng/L)	0.215	0.109	0.162

The calculated total nitrogen values assume nitrate and nitrite concentrations are each equal to one-half the detection limits. Based on the average of concentrations observed and an annual average flow rate of 110 ac-ft/yr, the groundwater input delivers 287 lb-N and 71.5 lb-P per year. Total and methyl mercury loads are 18.4 g and 0.022 g, respectively. Example calculations are presented in Section F.3.

The southern lakes at El Dorado Park lakes receive supplemental flows from a potable water source. On average, 105 ac-ft are pumped annually into Nature Center North Lake. This source was sampled for water quality during the August 2009, August 2010, and September 2010 sampling events (Table F-12). Resulting average nutrient loads are 269 pounds of nitrogen and 13.7 pounds of phosphorus annually. Total and methyl mercury loads are 0.368 g and 0.00259 g, respectively. Example calculations are presented in Section F.3.

Table F-12. Potable Water Quality Data for El Dorado Park Lakes

Parameter	August 2009	August 2010	September 2010	Average
Ammonia (mg-N/L)	0.365	0.0359	0.292	0.231
Nitrate (mg-N/L)	0.37	0.173	0.173	0.239
Nitrite (mg-N/L)	<0.01	0.054	0.060	0.040
TKN (mg-N/L)	0.84	0.480	0.672	0.664
Orthophosphate (mg-P/L)	<0.0075	0.026	0.009	0.013
Total Phosphorus (mg-P/L)	0.1085	<0.0165	<0.0165	0.0478
Total Nitrogen (calculated) (mg-N/L)	1.21	0.707	0.905	0.942
Total Mercury (ng/L)	2.84	Not sampled	Not sampled	2.84
Methylmercury (ng/L)	0.020	Not sampled	Not sampled	0.020

The park area surrounding the El Dorado Park lakes is irrigated with reclaimed water. This source was sampled for water quality in December 2009. Table F-13 summarizes the water quality data relevant to the nutrient and mercury TMDLs.

Table F-13. Reclaimed Water Quality Data for El Dorado Park Lakes

Parameter	Concentration
Ammonia (mg-N/L)	0.62
Nitrate (mg-N/L)	4.45
Nitrite (mg-N/L)	0.05
TKN (mg-N/L)	1.22
Orthophosphate (mg-P/L)	0.084
Total Phosphorus (mg-P/L)	0.166
Total Nitrogen (calculated) (mg-N/L)	5.72
Total Mercury (ng/L)	1.46
Methylmercury (ng/L)	0.021

Irrigation water is applied to 221 acres surrounding Coyote and Alamo lakes (known as Area III) and 179 acres surrounding Large and Horseshoe lakes (known as Area II). At the Nature Center where the two southern lakes are located, 91.1 acres are irrigated. The applied average annual volumes to these respective areas (based on utility bills) are 244 ac-ft, 280 ac-ft, and 64.7 ac-ft; applied depths range from 8.5 inches to 18.8 inches. The annual net evapotranspiration minus precipitation depth is 34.9 inches based on CIMIS data for this zone and precipitation data for a nearby weather station (Appendix D, Wet

Weather Loading). Thus, the majority of the applied water likely percolates into the ground or is lost to evapotranspiration. Officials at the park state that most of the reclaimed irrigation water percolates into the ground, but some runs off into the lakes and some sprinkler heads spray across the stream. A unit area model setup in LSPC for this subbasin indicates that approximately 3.9 percent of applied irrigation water may reach the El Dorado Park lakes. No additional fertilization has occurred on the parkland. This condition is assumed to represent existing conditions. Table F-14 summarizes the pollutant loads delivered to the two separate lake systems based on this information. To estimate loading from irrigation, the results from applying the example calculation used in Section F.3 were multiplied by 0.039 (the fraction of applied flow assumed to reach the lakes).

Table F-14. Estimated Loads Resulting from Irrigation around the El Dorado Park Lakes

Pollutant	Loading to Northern Lake System	Loading to Southern Lake System
Ammonia (lb/yr)	34.7	4.29
Nitrate (lb/yr)	249	30.8
Nitrite (lb/yr)	2.80	0.35
Organic Nitrogen (lb/yr)	33.6	4.15
Total Nitrogen (lb/yr)	320	39.6
Phosphate (lb/yr)	4.70	0.58
Phosphorus (lb/yr)	9.29	1.15
Total Mercury (g/yr)	0.0371	0.00458
Methylmercury (g/yr)	0.000533	0.0000659

Note: Reclaimed water concentrations used in the loading calculations are from data collected at El Dorado Park lakes (Table F-13).

There are some additional sources of nutrient loading that may exist at El Dorado Park lakes, such as feces deposited in near lake areas by wildlife and pets. These loads are difficult to estimate with out information on wildlife populations, number of pets visiting annually, and percentage of pet owners properly disposing of pet wastes. Additionally, during sampling events at El Dorado Park lakes people were observed feeding the birds and the birds may also feed from trash cans and food litter at the park. This practice increases nutrient loading to the lake by attracting birds and other animals that may deposit feces in or around the lake. Loads associated with feeding wildlife, as well as other sources that are difficult to quantify with the available information (normal wildlife populations, pets, park-area wastewater infrastructure, etc.) were accounted for in a category called “Additional Parkland Loading.” During calibration of the BATHTUB model (see Appendix A, Nutrient TMDL Development), loads in this category were quantified by increasing inputs until simulated nutrient concentrations match those observed.

F.9 North, Center, and Legg Lakes

North, Center, and Legg lakes are hydraulically connected waterbodies in Whittier Narrows Regional Park located in the Los Angeles River Basin. Legg Lake is listed as impaired by odor, ammonia, pH, copper, and lead (note: trash impairment has been addressed by a previous TMDL). Dry weather contributions to these lakes include storm drains, groundwater, irrigation, fertilization, and treated groundwater from a Superfund site.

F.9.1 POLLUTANT LOADS FROM STORM DRAINS

Five subwatersheds comprise the drainage area to these lakes. The northwestern and northeastern subwatersheds each drain to a storm drain that enters North Lake on the north side. Three separate subwatersheds areas have been delineated around the lakes to designate respective overland flow directly to each lake.

The northwestern, northeastern, and direct to north subwatersheds flow into North Lake which is basically separate from Center and Legg lakes during dry periods; North Lake discharges to Morris Creek. Legg Lake receives inputs from the direct to Legg subwatershed, from a Superfund site that discharges remediated water to the lake, and from pumped groundwater that is split between North and Legg lakes to maintain water levels. Legg Lake drains into Center Lake via a connecting channel which then discharges to Morris Creek. There are two culverts connecting Center and North lakes that allow water to flow between them when levels are sufficiently high.

Figure F-8 shows the MS4 stormwater permittees in the North, Center, and Legg lakes watershed. Loads generated from El Monte, South El Monte, Los Angeles County, and Caltrans from either the northwestern or northeastern subwatersheds are assigned wasteload allocations in the TMDLs because they drain to the storm drain network. Loads generated by South El Monte or the county of Los Angeles areas in the subwatersheds contributing directly to the lake are assigned load allocations; Caltrans areas in these subwatersheds are assigned wasteload allocations.

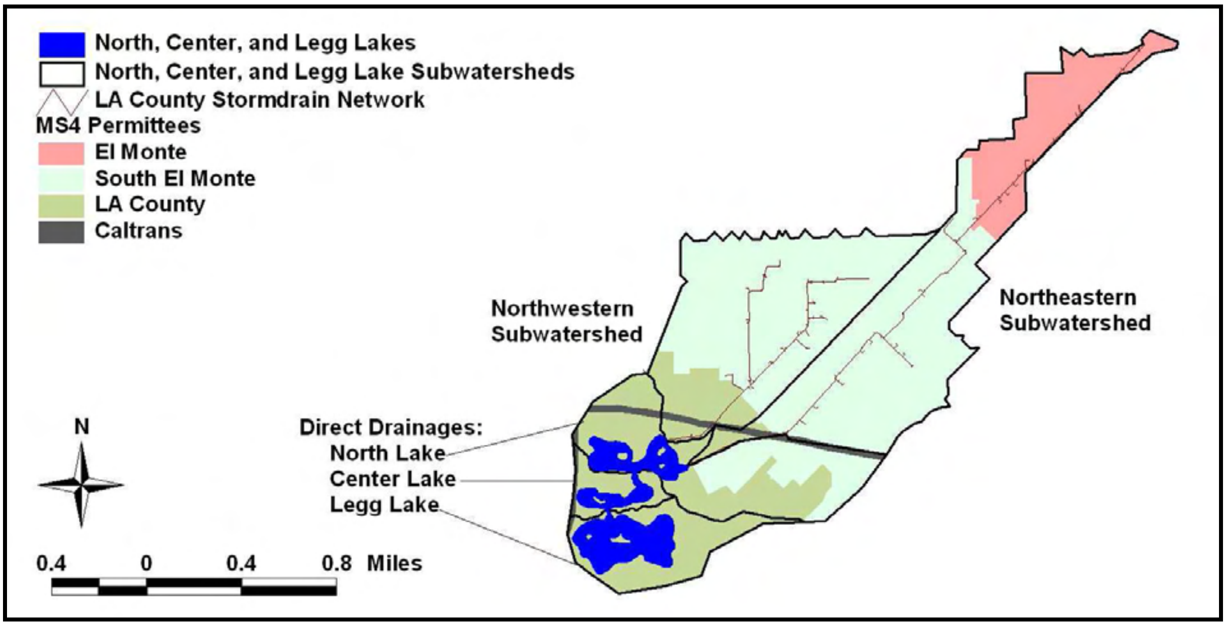


Figure F-7. MS4 Permittees and the County of Los Angeles Storm Drain Network in the North, Center, and Legg Lake Subwatersheds

Two subwatersheds contain a portion of the county of Los Angeles storm drain network that may deliver pollutant loads during dry weather. Table F-15 summarizes the upland areas draining to the Legg Lake system by subwatershed and jurisdiction. Table F-16 through Table F-18 list the estimated dry-weather flows and nutrient loads corresponding to these areas. Total nitrogen loading during dry weather is 1,478 lb/yr, and total phosphorus loading from dry weather flows is 296 lb/yr. Sample calculations are provided in Section F.2.

Table F-15. Land Use Areas (ac) Draining to North, Center, and Legg Lakes

Subwatershed	El Monte	South El Monte	County of Los Angeles	Caltrans	Total
Northwestern	0	317	60.1	5.32	**383**
Northeastern	134	305	10.0	6.18	**456**

Table F-16. Estimated Dry Weather Flows (ac-ft/yr) to North, Center, and Legg Lakes

Subwatershed	El Monte	South El Monte	County of Los Angeles	Caltrans	Total
Northwestern	0	68.6	13.0	1.15	**82.8**
Northeastern	29.0	65.9	2.17	1.34	**98.4**

Table F-17. Estimated Dry Weather Nitrogen Loads (lb/yr) to North, Center, and Legg Lakes

Subwatershed	El Monte	South El Monte	County of Los Angeles	Caltrans	Total
Northwestern	0	560	106	9.38	**675**
Northeastern	237	538	17.7	10.9	**803**

Table F-18. Estimated Dry Weather Phosphorus Loads (lb/yr) to North, Center, and Legg Lakes

Subwatershed	El Monte	South El Monte	County of Los Angeles	Caltrans	Total
Northwestern	0	112	21.2	1.88	**135**
Northeastern	47.3	108	3.53	2.18	**161**

F.9.2 POLLUTANT LOADS FROM OTHER DRY WEATHER INPUTS

To supplement water levels, treated groundwater from a Superfund site is pumped continuously into North and Legg lakes at an estimated rate of 2,534 ac-ft per year. This annual flow rate was estimated by extrapolating the flow measured by EPA for May through September 2010. Flows are split equally between these two lakes. Prior to May 2010 additional groundwater had been used to supplement water levels, but this input was discontinued.

In the summer of 2002, the city of Whittier began operating a Liquid Phase Granular Activated Carbon treatment facility associated with the San Gabriel Valley Area 1 Whittier Narrows Operable Unit Superfund site (EPA #CAD980677355) that treats groundwater contaminated with volatile organic chemicals (Stetson Engineers, 2009). In addition to monitoring levels of organic chemicals, the City is required to monitor concentrations of nitrate in the raw and treated water. Monitoring data collected over the period indicate that concentrations of nitrate did not change significantly in the treatment plant. The average nitrate concentration in the treated effluent, based on monthly samples collected in 2008, was 12 mg/L as NO_3 or 2.73 mg-N/L (Stetson Engineers, 2009).

EPA sampled the treated groundwater input during the June 8, August 11, and September 2010 sampling events. Table F-19 provides the mean observed concentrations for water quality parameters related to nutrient loading for these sampling events. Based on these concentrations, the treated Superfund discharge contributes 12,355 lb-N and 172 lb-P to the lake system, split equally between North and Legg lakes. Sample calculations are presented in Section F.3.

Table F-19. Mean Observed Concentrations for the Superfund Site at North, Center, and Legg Lakes for June, August, and September 2010 sampling events

Parameter	Mean Observed Concentration
Ammonia (mg-N/L)	0.05
Nitrate (mg-N/L)	1.60
Nitrite (mg-N/L)	0.13
TKN (mg-N/L)	0.07
Orthophosphate (mg-P/L)	0.03
Total Phosphorus (mg-P/L)	0.03
Total Nitrogen (calculated) (mg-N/L)	1.79

Runoff resulting from irrigation of 568 acres of parkland adjacent to the Legg Lake system is another potential source of nutrient loading. Water usage data for the Whittier Narrows Regional Recreation Area was provided for water years 2005 through 2009. Based on the average of the two most recent, complete water years, the total water usage at Whittier Narrows was 1,239 ac-ft. Staff at the park indicate that approximately 10 percent of this water is potable and 90 percent is reclaimed. Irrigation with the reclaimed water source began in 2006.

The usage also includes irrigation at Norman’s Nursery, which is outside the watershed of the Legg Lake system. In 2006, Norman’s Nursery used approximately 6.7 percent of the reclaimed water applied at Whittier Narrows. Subtracting out the usage at Norman’s Nursery leaves approximately 1,040 ac-ft of reclaimed water applied around the Legg Lake system. An additional 124 ac-ft of potable water is also applied. On average, 24.6 inches of irrigation water are applied. The annual net evapotranspiration minus precipitation depth is 38.6 inches based on CIMIS data for this zone and precipitation data for a nearby weather station (Appendix D, Wet Weather Loading). Thus, the majority of the applied water likely percolates into the ground or is lost to evapotranspiration. A unit area model setup in LSPC for this subbasin indicates that approximately 6.3 percent of applied irrigation water reaches the lake system.

Water quality data for the reclaimed water source were provided for fiscal year 2006/2007. The total phosphorus concentration was not reported. To estimate the total phosphorus concentration, the ratio of total phosphorus to orthophosphate observed in reclaimed water at El Dorado Park lakes (0.166 mg-P/L ÷ 0.84 mg-P/L = ratio of 1.98; Table F-13) was applied to the reported orthophosphate concentration. The resulting total phosphorus concentration is 1.45 mg-P/L.

The potable water source at Whittier Narrows was not sampled. Assumed concentrations for the potable water source are based on average values observed at the El Dorado Park lakes, Echo Park Lake, Lincoln Park Lake, and Lake Calabasas potable water inputs. Table F-20 summarizes the reported concentrations for the reclaimed water source and the assumed concentrations for the potable water source. Only parameters relevant to the nutrient TMDLs are included.

Table F-20. Average Water Quality Data for the Irrigation Water Sources at Whittier Narrows

Parameter	Reclaimed Water	Potable Water
Ammonia (mg-N/L)	0.86	0.266
Nitrate (mg-N/L)	7.07	0.682
Nitrite (mg-N/L)	<0.03	0.011
Organic Nitrogen (mg-N/L)	1.43	0.249
Phosphate (mg-P/L)	0.733	0.015
Total Phosphorus (mg-P/L)	Not Reported	0.089
Total Nitrogen (calculated) (mg-N/L)	8.52	0.942

Note: Reclaimed water concentrations are from the reclaimed water source for fiscal year 2006/2007 (total phosphorous was not reported); Potable water concentrations are an average of the potable water values observed at El Dorado Park lakes, Echo Park Lake, Lincoln Park Lake, and Lake Calabasas presented previously.

Table F-21 summarizes the nutrient loads delivered to the Legg Lake system due to irrigation based on the volumes and water quality data described above. See Section F.3 for example calculations.

Table F-21. Estimated Annual Pollutant Loading Resulting from Irrigation at Whittier Narrows

Pollutant	Load
Ammonia (lb/yr)	158
Nitrate (lb/yr)	1,266
Nitrite (lb/yr)	2.89
Organic Nitrogen (lb/yr)	284
Total Nitrogen (lb/yr)	1,711
Phosphate (lb/yr)	130
Phosphorus (lb/yr)	258

Information regarding use of fertilizer has not been received from park staff. Even if the types and application rates were known, it would still be difficult to estimate nutrient loading from fertilization because application methods, turf grass harvesting, and proximity of application to subsequent precipitation events impact transport via runoff.

During sampling events at the Legg Lake system, people were observed feeding the birds and birds may feed from trash cans and food litter at the park. These practices increase nutrient loading to the lake by attracting animals that may deposit feces around the lake. Loads associated with food waste dumping, as well as other sources that are difficult to quantify with the available information (normal wildlife populations, pets, park-area septic and other wastewater infrastructure systems, fertilization, etc.) were not accounted for in the North, Center, and Legg Lake nutrient TMDLs because no additional loading was required to simulate observed nutrient concentrations in this system (see Appendix A, Nutrient TMDL Development).

(This page left intentionally blank.)

F.10 Puddingstone Reservoir

Puddingstone Reservoir is located in the San Gabriel River Basin. Impairments include low dissolved oxygen/organic enrichment, mercury, chlordane, DDT, and PCBs. Dry weather contributions to this lake include dry weather runoff to storm drains in the Northern Subwatershed and irrigation of parkland in the Southern Subwatershed.

F.10.1 POLLUTANT LOADS FROM STORM DRAINS

Two subwatersheds comprise the drainage area to Puddingstone Reservoir. The subwatershed draining the northern part of the watershed is 6,959 acres, and the southern subwatershed is 1,169 acres. The subwatershed boundaries were chosen to separate those areas that drain to a storm drain (the northern subwatershed) and those that enter the reservoir via natural tributaries or overland flow (the southern subwatershed).

Figure F-8 shows the MS4 stormwater permittees in the Puddingstone Reservoir watershed. The northern subwatershed is primarily comprised of the county of Los Angeles, Claremont, and La Verne areas with a small amount of San Dimas, Caltrans, and Angeles National Forest areas. Loads generated from these jurisdictions in the northern subwatershed are assigned wasteload allocations because they drain to the county of Los Angeles storm drain network. The southern subwatershed is comprised of San Dimas, La Verne, and Pomona areas. Loads from these jurisdictions originating in the southern subwatershed are assigned load allocations. The small amount of Caltrans area in the Southern Subwatershed are assigned a wasteload allocation.

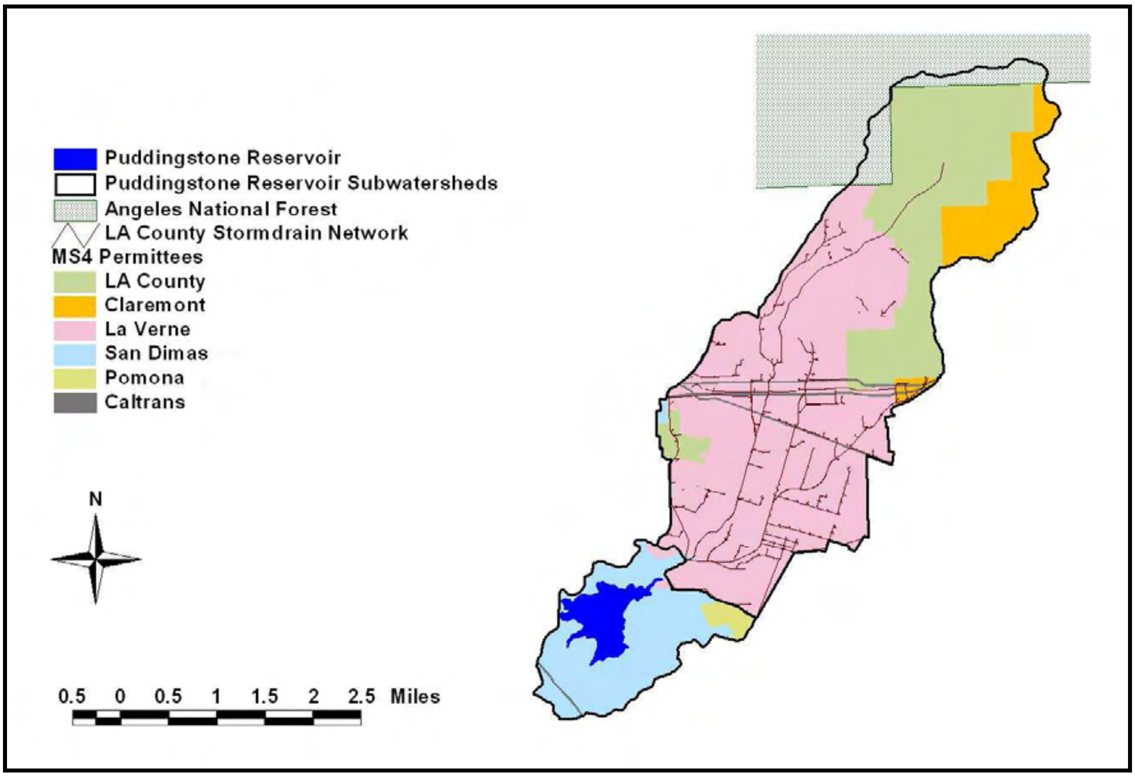


Figure F-8. MS4 Permittees and the County of Los Angeles Storm Drain Network in the Puddingstone Reservoir Subwatersheds

The county of Los Angeles storm drain system is only present in the Northern subwatershed. The contributing areas by jurisdiction and the estimated dry-weather flows and pollutant loads are presented in Table F-22. Nutrient loads were calculated from the SCCWRP data presented in Section F.2. The relevant monitoring study did not include mercury data, so the mercury concentration observed in the summer of 2009 near the outlet of the Northern Subwatershed was used to represent dry weather concentrations of total mercury (4.24 ng/L) and methylmercury (0.553 ng/L). Dry weather loads from National Forest lands were assumed zero; wet weather loading from these areas is described in Appendix D (Wet Weather Loading). The Southern Subwatershed does not contain areas serviced by the storm drain network. Dry weather loads associated with the small pipes that drain surrounding parkland are likely due to irrigation of adjacent areas, which is discussed in the following section. Sample calculations for dry weather loads from the storm drain network are provided in Section F.2.

Table F-22. Estimated Dry Weather Storm Drain Inputs to Puddingstone Reservoir from the Northern Subwatershed

Watershed	Claremont	County of Los Angeles	La Verne	Pomona	San Dimas	Caltrans	Angeles National Forest	Total
Area (ac)	578	1,865	4,079	5.28	28.5	110	**293**	**6,959**
Flow (ac-ft/yr)	125	403	881	1.14	6.15	23.8	**0**	**1,440**
Nitrogen (lb/yr)	1,019	3,288	7,191	9.32	50.2	194	**0**	**11,752**
Phosphorus (lb/yr)	204	658	1,438	1.86	10.0	38.8	**0**	**2,350**
Total Mercury (g/yr)	0.654	2.11	4.61	0.00597	0.0322	0.124	**0**	**7.53**
Methylmercury (g/yr)	0.085	0.275	0.601	0.000779	0.00419	0.0162	**0**	**0.983**

F.10.2 POLLUTANT LOADS FROM OTHER DRY WEATHER INPUTS

Puddingstone Reservoir does not receive supplemental flows from a potable or groundwater source. Though the Metropolitan Water District can divert water to Puddingstone Reservoir from outside the watershed, this practice is seldom used (personal communication, Adam Walden, Los Angeles County Department of Public Works, 9/16/09) and does not impact the average conditions for this reservoir.

The park area around Puddingstone Reservoir is irrigated with a combination of reclaimed and potable water. Water quality data for the reclaimed water source were provided by the city of Pomona. The potable water source at Puddingstone Reservoir was not sampled during the 2009 monitoring event and data were not available from the source. To estimate water quality for this source, concentrations were assumed equal to the average concentrations observed at the El Dorado Park lakes, Echo Park Lake, Lincoln Park Lake, and Lake Calabasas potable water inputs. Table F-23 summarizes the monitoring data for the reclaimed water provided by the city of Pomona and the average concentrations assumed for the potable water source. Only those parameters relevant to the nutrient and mercury TMDLs are included in the table.

Table F-23. Average Water Quality Data for the Irrigation Water Sources at Puddingstone Reservoir

Parameter	Reclaimed Water	Potable Water
Ammonia (mg-N/L)	1.33	0.266
Nitrate (mg-N/L)	5.49	0.682
Nitrite (mg-N/L)	0.094	0.011
Organic Nitrogen (mg-N/L)	1.27	0.249
Phosphate (mg-P/L)	Not reported	0.015
Total Phosphorus (mg-P/L)	Not reported	0.089
Total Nitrogen (calculated) (mg-N/L)	8.18	1.21
Total Mercury (ng/L)	24	2.84

Note: Reclaimed water concentrations were provided by the city of Pomona (phosphate and total phosphorous were not reported); Potable water concentrations are an average of the potable water values observed at El Dorado Park lakes, Echo Park Lake, Lincoln Park Lake, and Lake Calabasas presented previously.

The monitoring data for the reclaimed water at Puddingstone Reservoir does not include phosphorus parameters. A phosphate concentration of 0.408 mg-P/L was assumed, based on averaging the phosphate concentrations reported for Legg Lake and El Dorado Park lakes. The total phosphorus concentration was estimated by applying the ratio of total phosphorus to orthophosphate observed at El Dorado Park lakes (0.166 mg-P/L ÷ 0.84 mg-P/L = ratio of 1.98; Table F-13). The resulting total phosphorus concentration is 0.807 mg-P/L.

Park staff report that approximately 1,180 acres in the park are irrigated. Utility bills indicate that on average, 1,510 ac-ft of reclaimed water and 104 ac-ft of potable water are used for irrigation each year. This volume equates to a depth of 16.4 inches, which is significantly less than the net evaporation minus precipitation depth (37.7 inches) estimated from data posted on the CIMIS website for this zone and precipitation data for a nearby weather station (Appendix D, Wet Weather Loading).

Officials at the park state that the majority of the irrigation water percolates into the ground, although areas along the shoreline do produce runoff to the reservoir during irrigation. A unit area model set up in LSPC for this subbasin indicates that approximately 10.1 percent of applied irrigation water reaches Puddingstone Reservoir. During the past three years, no additional fertilization has occurred due to budget considerations. This condition is assumed to represent existing conditions. The resulting loads due to irrigation are summarized in Table F-24. Example calculations are presented in Section F.3.

Table F-24. Estimated Annual Pollutant Loading Resulting from Irrigation at Puddingstone Reservoir

Pollutant	Load
Ammonia (lb/yr)	559
Nitrate (lb/yr)	2,294
Nitrite (lb/yr)	39.3
Organic Nitrogen (lb/yr)	533
Total Nitrogen (lb/yr)	3,425

Pollutant	**Load**
Phosphate (lb/yr)[1]	170
Phosphorus (lb/yr)[2]	337
Total Mercury (g/yr)	4.55

[1] The monitoring data for the reclaimed water at Puddingstone Reservoir does not include phosphorus parameters. A phosphate concentration of 0.408 mg-P/L was assumed, based on averaging the phosphate concentrations reported for Legg Lake and El Dorado Park lakes.

[2] The total phosphorus concentration was estimated by applying the ratio of total phosphorus to orthophosphate observed at El Dorado Park lakes (0.166 mg-P/L ÷ 0.84 mg-P/L = ratio of 1.98; Table F-13). The resulting total phosphorus concentration is 0.807 mg-P/L.

Other sources of nutrient loading may exist at Puddingstone Reservoir such as wildlife and pets depositing feces that may wash off into the reservoir during rain events. While no bird feeding has been observed during recent fieldwork, birds may feed from trash cans and food litter at the park. It is difficult to estimate nutrient loading from animal wastes without information on populations and pet owner waste-disposal practices. Loads from animal wastes, as well as other sources that are difficult to quantify with the available information (e.g., park-area wastewater infrastructure systems), were not accounted for in the Puddingstone Reservoir nutrient TMDLs because no additional loading was required to simulate observed nutrient concentrations at this lake (see Appendix A, Nutrient TMDL Development).

F.11 Santa Fe Dam Park Lake

Santa Fe Dam Park Lake is located in the San Gabriel River Basin. Impairments of this lake include pH, copper, and lead. The waterbody is a recreational lake that was constructed within the Santa Fe Flood Control Basin, but no water from the basin is diverted into the lake (personal communication, Chris Graham, Los Angeles County Department of Parks and Recreation). Dry weather inputs include groundwater and potable water used for maintaining lake levels and runoff from irrigation.

One 362-acre subwatershed comprises the drainage area to Santa Fe Dam Park Lake. No storm drain system is present in the watershed. Figure F-9 shows the jurisdictions present in the Santa Fe Dam Park Lake watershed. Most of the drainage area is located in Irwindale, with a small portion in Azusa.

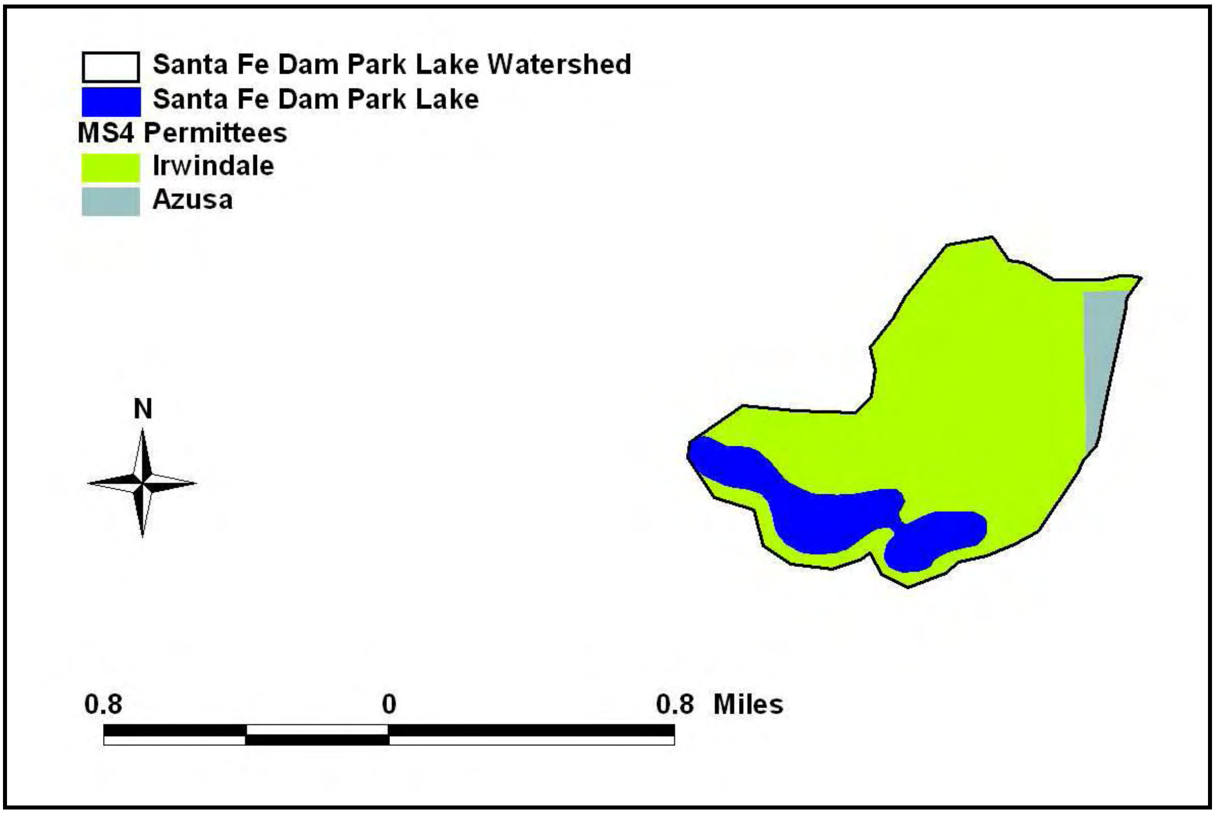


Figure F-9. Jurisdictions in the Santa Fe Dam Park Lake Subwatershed

F.11.1 POLLUTANT LOADS FROM STORM DRAINS

There are no storm drains in the Santa Fe Dam Park Lake watershed. Thus dry weather loads from storm drains are zero for this waterbody (wet weather loads from the watershed are discussed in Appendix D).

F.11.2 POLLUTANT LOADS FROM OTHER DRY WEATHER INPUTS

Santa Fe Dam Park Lake receives supplemental flows from groundwater and potable water sources to maintain lake levels. Ten years of monthly usage data were used to estimate the average annual volume pumped from each source. Groundwater and potable water are pumped at average rates of 1,319 ac-ft/yr and 544 ac-ft/yr, respectively.

The groundwater input at Santa Fe Dam Park Lake was sampled on August 3, 2009 and August 12, 2010. The calculated total nitrogen value for the groundwater concentrations reported as less than the detection limit are equal to one-half the detection limits. During both sampling events, total phosphorus was analyzed as less than the detection limit of 0.016 mg/L; therefore, the phosphate concentration was used to represent the total phosphorus content of the groundwater. The potable water input at the discharge point to Santa Fe Dam Park Lake has not been sampled. The average of measurements obtained from potable water sources at other impaired lakes sampled for this TMDL study were used to estimate the nutrient concentrations for this source (El Dorado Park lakes, Echo Park Lake, Lake Calabasas, and Lincoln Park Lake). Table F-25 summarizes the average observed and estimated concentrations for the groundwater and potable water inputs at Santa Fe Dam Park Lake.

Table F-25. Water Quality Data for the Groundwater and Potable Water Inputs at Santa Fe Dam Park Lake

Parameter	Groundwater	Potable Water
Ammonia (mg-N/L)	0.03	0.266
Nitrate (mg-N/L)	2.3	0.682
Nitrite (mg-N/L)	0.02	0.011
TKN (mg-N/L)	0.67	0.516
Orthophosphate (mg-P/L)	0.026	0.015
Total Phosphorus (mg-P/L)	0.026	0.0923
Total Nitrogen (calculated) (mg-N/L)	2.99	1.21

Note: Groundwater concentrations are from data collected at Santa Fe Dam Park lake (total phosphorous was less than the detection limit, so the phosphate concentration was used to represent total phosphorous); Potable water concentrations are an average of the potable water values observed at El Dorado Park lakes, Echo Park Lake, Lincoln Park Lake, and Lake Calabasas presented previously.

Nutrient loads discharged directly to Santa Fe Dam Park Lake from these sources can be calculated from the average annual volume discharged and the water quality concentrations. Total nitrogen loads from groundwater and potable water are estimated to be 10,734 lb-N/yr and 1,790 lb-N/yr, respectively. Total phosphorus loads from these sources are 93.3 lb-P/yr and 137 lb-P/yr, respectively. Example calculations are presented in Section F.3.

In addition to inputs of potable water and groundwater, the swim beach area of Santa Fe Dam Park Lake is chlorinated during the summer months. Chlorination typically occurs seven days per week via five pumps. However, due to reduced funding available in 2009, the swim beach was closed Monday through Wednesday and only one chlorine pump was being utilized (personal communication, Chris Graham, Los Angeles County Department of Parks and Recreation, September 19, 2009). Chlorination alters the pH of the water and may be contributing to the pH impairment.

The groundwater source at Santa Fe Dam Park Lake is also used to irrigate 175 acres of parkland. Irrigation water is observed to percolate into the ground. Application volumes were not available. To estimate loading from this source, it is assumed that a depth of 1 foot of water is applied annually. A unit area model setup in LSPC for this subbasin indicates that approximately 9.6 percent of applied irrigation water reaches the lake. These assumptions yield nutrient loads to the lake of 137 lb-N/yr and

1.19 lb-P/yr. There is no fertilization schedule for this area, so loads from fertilizer are assumed zero. Example calculations are presented in Section F.3.

Other sources of nutrient loading may exist at Santa Fe Dam Park Lake such as wildlife and pets depositing feces that may wash off into the reservoir during rain events. While no bird feeding has been observed during recent fieldwork, it is likely a recreational activity at the lake and birds may feed from trash cans and food litter at the park. It is difficult to estimate nutrient loading from animal wastes without information on populations and pet owner waste-disposal practices. Loads from animal wastes, as well as other sources that are difficult to quantify with the available information (e.g., park-area wastewater infrastructure systems) were not accounted for in the Santa Fe Dam Park Lake nutrient TMDLs because no additional loading was required to simulate observed nutrient concentrations at this lake (see Appendix A, Nutrient TMDL Development).

(This page left intentionally blank.)

F.12 Lake Sherwood

Lake Sherwood is located in the Santa Monica Bay Basin and is impaired by mercury (note: algae, ammonia, eutrophication, and low dissolved oxygen impairments have been addressed by a previous TMDL). Dry weather contributions to this lake include dry weather runoff to storm drains in the developed subwatersheds.

F.12.1 POLLUTANT LOADS FROM STORM DRAINS

Six subwatersheds comprise the drainage area (10,656 acres) to Lake Sherwood. Figure F-10 shows the MS4 stormwater permittees comprising each subwatershed.

Ventura County is the only stormwater permittee in the Western Subwatershed. The Hidden Valley Wash subwatershed is mostly in Ventura County with small portion in Thousand Oaks. The Northern, Near Lake Undeveloped, and Near Lake Developed subwatersheds are comprised of both Ventura County and Thousand Oaks MS4 areas. The Carlisle Canyon subwatershed contains Ventura and Los Angeles County areas as well as Thousand Oaks, California Department of Transportation (Caltrans), and California State Park areas. Neither Ventura or Los Angeles counties (the MS4 stormwater permittees in the watershed) maintain storm drain systems in the Lake Sherwood watershed. However, there are residential developments in the vicinity of the lake which drain to culverts and storm drains. These areas are generally associated with the Sherwood Valley Homeowner's Association (SVHOA) and Sherwood Development Company. All subwatersheds will receive wasteload allocations except for the Carlisle Canyon and Near Lake Undeveloped subwatersheds. The small Caltrans area in the Carlisle Canyon subwatershed will also receive a wasteload allocation.

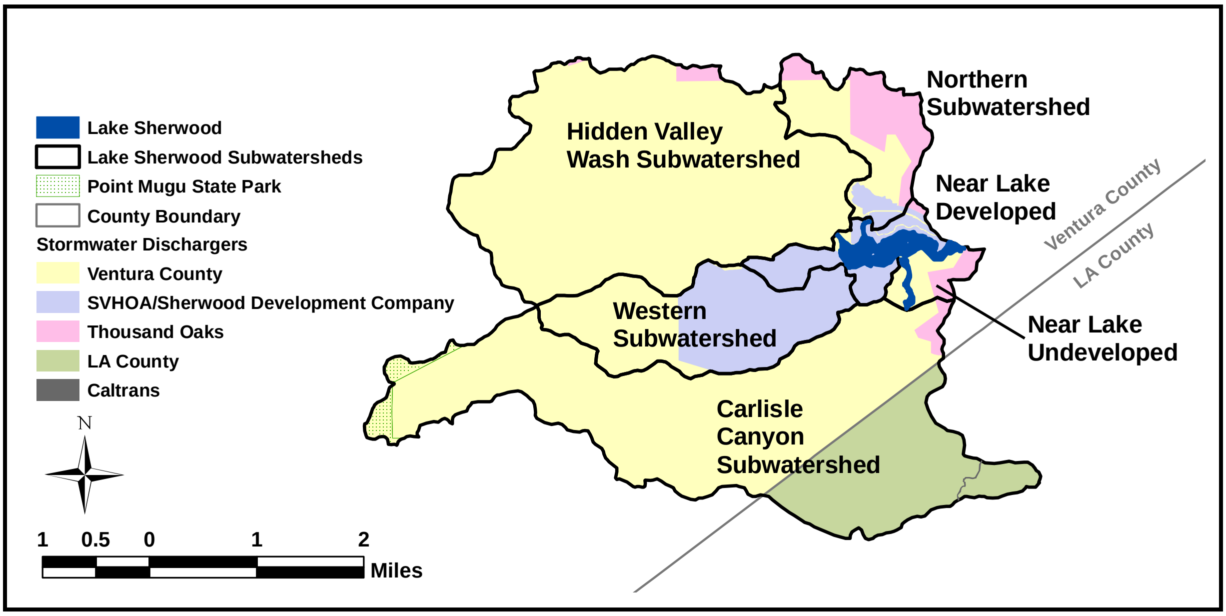


Figure F-10. MS4 Permittees in the Lake Sherwood Subwatersheds

The developed subwatersheds (Northern, Western, and Near Lake Developed) likely contribute dry weather flows and loading to Lake Sherwood via the storm drain system. [No flows were observed along Hidden Valley Wash during dry weather sampling, so dry weather loads from this subwatershed are assumed zero.] Dry weather flow volumes (described in Section F.2) were estimated for the "other urban" and residential land uses in each developed subwatershed.. All developed lands in these subwatersheds are in Ventura County. Mercury concentrations observed during the summer monitoring event (Appendix G, Monitoring Data) were assumed to represent the dry weather concentrations at the mouth of each tributary or storm drain. Table F-26 summarizes the resulting total mercury loads, and Table F-27 summarizes the methylmercury loads. Example calculations are presented in Section F.3.

Table F-26. Dry Weather Total Mercury Loading to Lake Sherwood

Subwatershed	Developed Area (ac)	Dry Weather Flows (ac-ft/yr)	Dry Weather Total Mercury Concentration (ng/L)	Dry Weather Total Mercury Load (g/yr)
Northern	43.0	9.28	54.0	0.618
Western	185	39.9	4.58	0.226
Near Lake Developed	175	37.7	54.0[1]	2.51

[1] Concentrations for this subwatershed are assumed similar to those observed in the Northern Subwatershed based on land use similarity.

Table F-27. Dry Weather Methylmercury Loading to Lake Sherwood

Subwatershed	Developed Area (ac)	Dry Weather Flows (ac-ft/yr)	Dry Weather Methylmercury Concentration (ng/L)	Dry Weather Methylmercury Load (g/yr)
Northern	43.0	9.28	0.096	0.0011
Western	185	39.9	0.536	0.0264
Near Lake Developed	175	37.7	0.096[1]	0.00447

[1] Concentrations for this subwatershed are assumed similar to those observed in the Northern Subwatershed based on land use similarity.

F.12.2 POLLUTANT LOADS FROM OTHER DRY WEATHER INPUTS

There are no additional dry weather sources of mercury loading to Lake Sherwood.

F.13 References

Ackerman, D. and E.D. Stein. 2005. Dry Weather Flow in Arid, Urban Watersheds. Presented at the 2005 Headwaters to Oceans (H2O) Conference. Available online at http://coastalconference.org/h20_2005/pdf/2005/2005_10-27-Thursday/Session3C-Watershed-Water_Quality_Modeling/Ackerman-Dry_Weather_Flow_in_Arid_Urban_Watersheds.pdf

Black and Veatch. 2008. Echo Park Lake Exfiltration and Flow Monitoring. Prepared for the City of Los Angeles Bureau of Sanitation, November 2008.

Black and Veatch. 2010. Echo Park Lake Water Quality Modeling Study. Prepared for the City of Los Angeles Department of Public Works Bureau of Engineering. March 2010.

Stein, E.D. and D. Ackerman. 2007. Dry weather water quality loadings in arid, urban watersheds of Los Angeles Basin, California, USA. Journal of American Water Resources Association 43(2):398-413.

Stetson Engineers Inc. 2009. Whittier Narrows Operable Unit Groundwater Treatment Plant Technical Performance Report.

Tetra Tech. 2004. Modeling Analysis for Development of TMDLs for Metals in the Los Angeles River and Tributaries. Prepared for USEPA Region 9 and the Los Angeles Regional Water Quality Control Board, July 2004.

Tetra Tech. 2005. Model Development for Simulation of Dry-Weather Metals Loading from the San Gabriel River Watershed. Prepared for USEPA Region 9 and the Los Angeles Regional Water Quality Control Board, November 2005.

Tetra Tech. 2007. Evaluation of the Numeric Nutrient Endpoints for the San Gabriel River Watershed. Prepared for US Environmental Protection Agency Region 9. Tetra Tech, Inc., Owings Mills, MD.

(This page left intentionally blank.)

Appendix G. Monitoring Data for the Los Angeles Area Lakes TMDLs

(This page left intentionally blank.)

G.1 Introduction

USEPA Region IX is establishing TMDLs for impairments in nine lakes in the Los Angeles Region (Figure G-1). Los Angeles Water Quality Control Board (Regional Board) assisted USEPA in this effort by compiling historic data associated with 1998 list and with collecting recent (2008-2010) monitoring results. The waterbodies are impaired by low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, algae, pH, mercury, lead, copper, chlordane, DDT, dieldrin, PCBs, and trash. This appendix describes the monitoring data relevant to TMDL development, determinations of nonimpairment, and determinations of new impairments for these waterbodies.

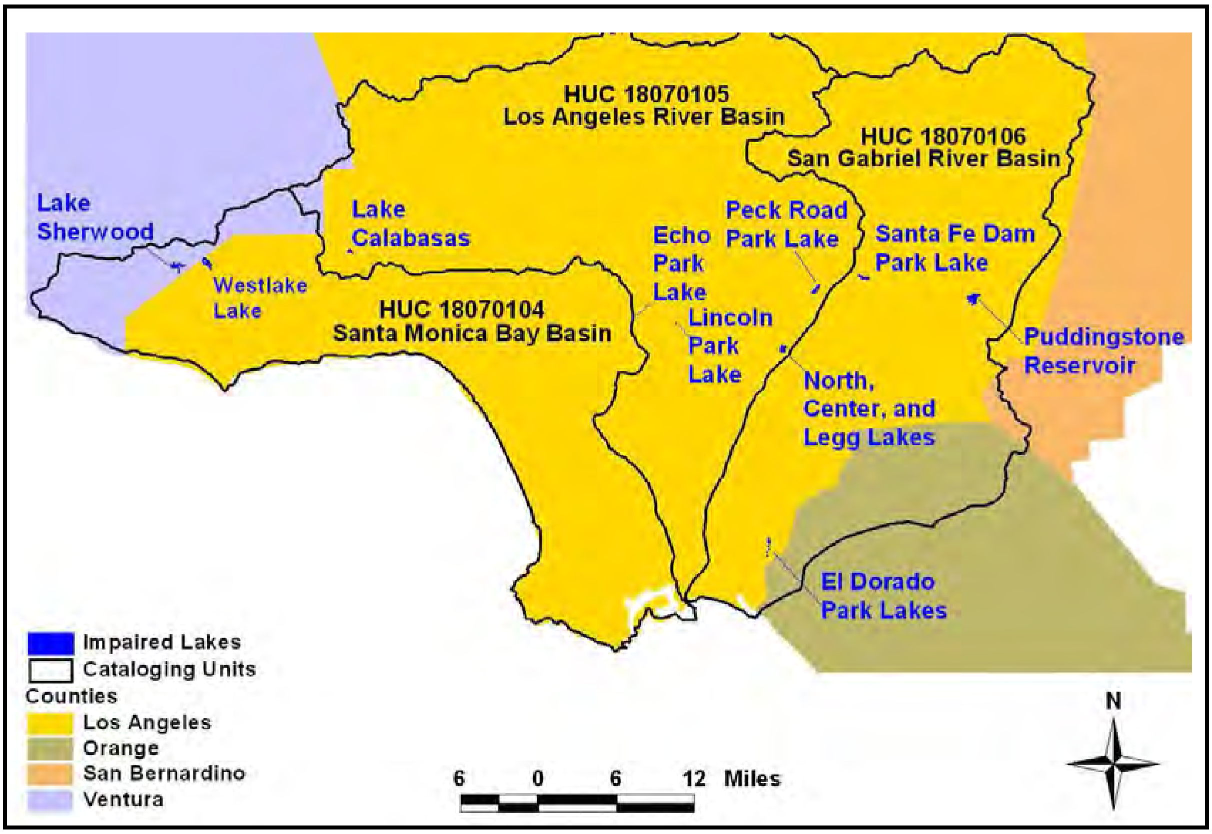


Figure G-1. Location of Impaired Lakes

(This page left intentionally blank.)

G.2 Overview of Monitoring Parameters

The impairments in the Los Angeles area lakes presented in Table 2-30 can be grouped into five categories: nutrients, mercury, metals, Organochlorine (OC) Pesticides and PCBs, and trash. Various monitoring parameters are helpful to evaluate impairments and characterize watershed and in-lake conditions for TMDL analyses. The parameters and their associated category are presented in Table G-1. This table also identifies the media and whether available data were reported by an analytical laboratory and/or measured in the field. Specifics for nutrients, mercury, metals, and OC Pesticides and PCBs data are described for each lake in Section G.4 through Section G.13. Trash monitoring measurements are described in the lake-specific chapters.

Table G-1. Monitoring Parameters by Category

Analysis Category	Monitoring Parameter	Media	Analytical Laboratory Result	Field Measurement
Nutrients	Ammonia/Ammonium	Water column	●	
	Biochemical Oxygen Demand	Water column	●	
	Chloride	Water column	●	
	Chlorophyll *a*	Water column	●	
	Depth	Water column		●
	Dissolved Organic Carbon	Water column	●	
	Dissolved Oxygen	Water column	●	●
	Electrical Conductivity	Water column		●
	Nitrate	Water column	●	
	Nitrite	Water column	●	
	Organic Nitrogen	Water column	●	
	Orthophosphate	Water column	●	
	pH	Water column	●	●
	Secchi Depth	Water column		●
	Sulfate	Water column	●	
	Suspended Solids	Water column	●	
	Temperature	Water column		●
	Total Alkalinity	Water column	●	
	Total Dissolved Solids	Water column	●	
	Total Kjeldahl Nitrogen	Water column	●	
	Total Organic Carbon	Water column	●	
	Total Phosphate	Water column	●	
	Total Phosphorous	Water column	●	
Mercury	Methylmercury (total)	Water column	●	

Analysis Category	Monitoring Parameter	Media	Analytical Laboratory Result	Field Measurement
	Methylmercury (total)	Sediment	●	
	Sulfate	Sediment	●	
	Sulfate	Water column	●	
	Total Mercury	Water column	●	
	Total Mercury	Sediment	●	
	Total Mercury	Fish tissue	●	
	Total Suspended Solids	Water column	●	
Metals	Cadmium (dissolved)	Water column	●	
	Copper (dissolved)	Water column	●	
	Copper (total)	Sediment	●	
	Lead (dissolved)	Water column	●	
	Lead (total)	Water column	●	
	Lead (total)	Sediment	●	
	Total hardness	Water column	●	
	Zinc (dissolved)	Water column	●	
OC Pesticides and PCBs	Chlordane*	Water column	●	
	Chlordane*	Porewater	●	
	Chlordane*	Fish tissue	●	
	Chlordane*	Sediment (bed and suspended sediment)	●	
	DDTs*	Water column	●	
	DDTs*	Porewater	●	
	DDTs*	Fish tissue	●	
	DDTs*	Sediment (bed and suspended sediment)	●	
	Dieldrin	Water column	●	
	Dieldrin	Porewater	●	
	Dieldrin	Fish tissue	●	
	Dieldrin	Sediment (bed and suspended sediment)	●	
	PCBs*	Water column	●	
	PCBs*	Porewater	●	
	PCBs*	Fish tissue	●	
	PCBs*	Sediment (bed and suspended sediment)	●	

* May include various chemicals that make up the total compound.

G.3 Overview of Monitoring Studies

Several studies have been conducted over the past few decades to monitor water quality in the county of Los Angeles. The University of California at Riverside conducted a study of urban lakes in the county of Los Angeles (UC Riverside, 1994). Most of the monitoring data were collected in 1992 and 1993, and the findings were summarized in 1994 as the "Evaluation of Water Quality for Selected Lakes in the Los Angeles Hydrologic Basin." Each lake was sampled at one location. Samples reported with the same date were likely replicates. Although raw data were available for the nutrient parameters, pH, total organic carbon (TOC), and total dissolved solids (TDS), only ranges and average chlorophyll *a* concentrations were provided.

The Regional Board completed a Clean Water Act Section 305(b) Water Quality Assessment and Documentation Report for the Los Angeles Region in 1996 (LARWQCB, 1996). This report identifies the impaired waters, summarizes the impairments for each lake, and provides data summaries for dissolved oxygen (DO), pH, and ammonia. A database of water quality monitoring was provided to Tetra Tech along with the Water Quality Assessment and Documentation Report. Although the database does contain limited water quality data for a few of the nutrient impaired lakes, it does not contain the raw data associated with the data summaries of DO, pH, and ammonia as listed in the report. While the data summaries are useful to explain the initial listings, they do not provide the level of detail required to directly apply the data (sampling location, depth, time, relationship to other monitored parameters, etc.).

More recently the Regional Board has collected water quality data in several of these lakes. Much of these data were collected in 2008 and 2009, with some additional metals, organics, and nutrient sampling in 2010. In addition, a few of the lakes have been studied independently over the past several years by other municipal agencies.

This appendix summarizes the water quality data collected in each lake and associated watershed through fall 2010. Where applicable, these data were used to support model development and/or TMDL calculations.

In addition to displaying the locations of the various monitoring stations for each study, the figures in this appendix also show the subwatershed boundaries and the incorporated areas comprising each watershed. In general, the areas draining to storm drain networks will receive waste load allocations for the TMDL, while the other drainage areas will receive load allocations. The areas associated with wasteload and load allocations are described in each lake chapter. Tetra Tech made slight modifications to the subwatershed boundaries that were downloaded from the county of Los Angeles GIS data depot. These minor modifications were based on aerial photographs and digital elevation models. Most changes were made to coordinate subwatershed boundaries with a sampling location, to move the boundary outside of the arms of each lake, or to aggregate subwatersheds to larger TMDL subwatersheds. Modifications are explained in the general information section for each lake.

(This page left intentionally blank.)

G.4 Monitoring Data for Peck Road Park Lake

Monitoring data relevant to the impairments of Peck Road Park Lake are available from 1992, 1993, 2008, 2009, and 2010. In addition, tributary data are available sporadically from 1977 through 1997 and fish tissue data are available from 1986 through 2007. Figure G-2 shows the historical and recent monitoring locations for Peck Road Park Lake.

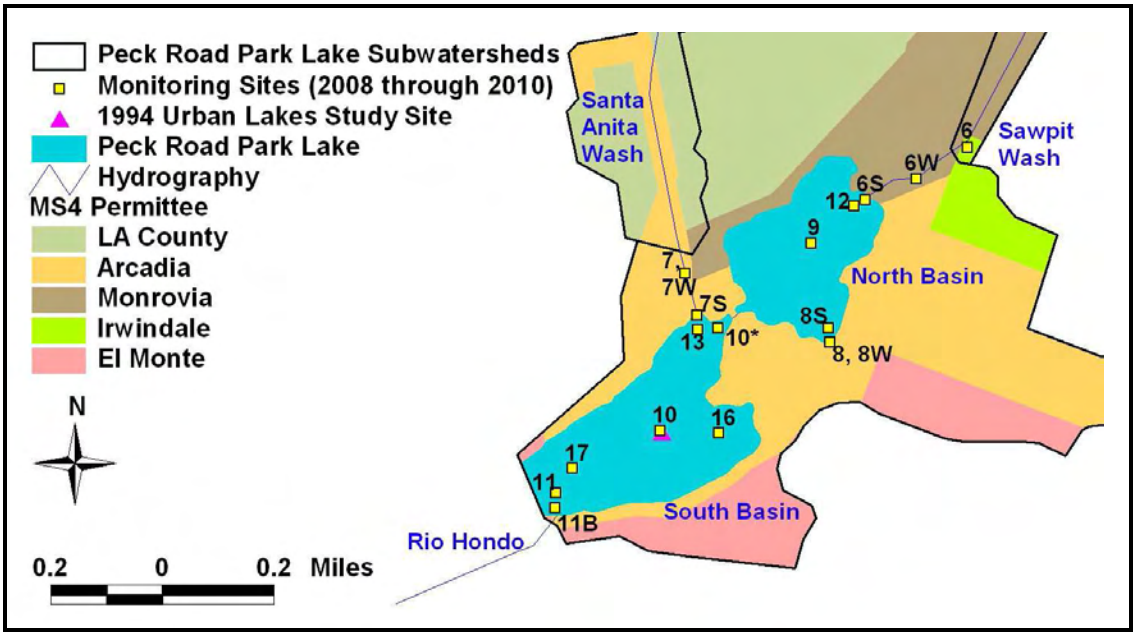


Figure G-2. Peck Road Park Lake Monitoring Sites

G.4.1 MONITORING RELATED TO NUTRIENT IMPAIRMENTS

The nutrient and pH data collected during the 1992-93 monitoring period in support of the Urban Lakes Study are shown in Table G-2. Samples were collected from the middle of the south basin (pink triangle, Figure G-2). Unfortunately, nutrient levels were analyzed at relatively high detection limits.

Of the 90 orthophosphate samples collected, only one exceeds the reporting limit of 0.1 mg/L. This measurement was collected at a depth of 8 meters and had a value of 0.4 mg/L. Only 1 of 90 total phosphorus samples exceeded the reporting limit of 0.1 mg/L: at a depth of 5 meters the TP measurement was 0.9 mg/L.

Three nitrite samples exceeded the reporting limit for this dataset of 0.1 mg/L. All three had values of 0.2 mg/L and were located at depths ranging from 7 to 14 meters. For nitrate, 23 samples were less than the reporting limit and the maximum nitrate concentration measured was 1.1 mg/L. Twelve measurements of Total Kjeldahl Nitrogen (TKN), which includes the organic and ammonia species of nitrogen, were less than the reporting limit and the maximum TKN concentration observed was 2.0 mg/L. For ammonia, 55 out of 90 measurements were less than the reporting limit and 35 samples ranged from 0.1 mg/L to 1.2 mg/L. pH ranged from 7.3 to 8.8. Total organic carbon (TOC) concentrations ranged from 0.4 mg/L to 4.7 mg/L.

The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from <1 µg/L to 19 µg/L with an average of 8 µg/L. The graphs displaying the depth profile data for Peck Road Park Lake show that dissolved oxygen typically declines to 0 mg/L during the summer months at depths greater than 5 meters. At depths less than 5 meters, dissolved oxygen concentrations were typically around 7 mg/L during the summer months.

Table G-2. Peck Road Park Lake 1992/1993 Monitoring Data for Nutrients

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
7/7/1992	0	2	<0.1	<0.1	0.1	<0.1	<0.1	7.3	3	173
	3	0.2	<0.1	<0.1	0.1	<0.1	<0.1	7.8	2.2	169
	5.5	0.2	<0.1	<0.1	0.2	<0.1	<0.1	8	4	172
	7	0.3	<0.1	<0.1	0.2	<0.1	<0.1	7.9	2.3	156
	9	0.5	0.2	<0.1	0.1	<0.1	<0.1	7.8	1.8	162
7/7/1992	0	0.2	<0.1	<0.1	<0.1	<0.1	<0.1	8.3	2.2	151
	3	0.2	<0.1	<0.1	0.1	<0.1	<0.1	8.3	3.1	169
	5.5	0.3	<0.1	<0.1	0.2	<0.1	<0.1	8.2	2.8	171
	6	0.1	<0.1	<0.1	0.2	<0.1	<0.1	8.2	2	171
	7.5	0.4	0.3	<0.1	0.1	<0.1	<0.1	8	3.5	170
7/23/1992	0	0.5	0.9	<0.1	0.1	<0.1	<0.1	8.1	1.2	260
	3.5	0.1	<0.1	<0.1	0.1	<0.1	<0.1	8.1	3	245
	6.5	0.2	<0.1	<0.1	0.2	<0.1	<0.1	8	1.2	242
	8.5	0.4	0.2	<0.1	<0.1	<0.1	<0.1	7.9	1.3	240
	10.5	0.5	0.2	<0.1	<0.1	<0.1	<0.1	7.8	1.1	255
	13	1	0.9	<0.1	<0.1	0.1	<0.1	7.7	2.1	223
7/23/1992	0	0.3	0.5	<0.1	<0.1	<0.1	<0.1	8.4	1.3	174
	2	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	8.4	3.9	185
	4	0.1	<0.1	<0.1	0.1	<0.1	<0.1	8.2	1.2	198
7/23/1992	0	0.1	<0.1	<0.1	<0.1	<0.1	<0.1	8.5	1.5	167
	2	0.2	<0.1	<0.1	<0.1	<0.1	<0.1	8.5	4.2	185
	4	0.1	<0.1	<0.1	0.1	<0.1	<0.1	8.3	1	189
	6	<0.1	0.3	<0.1	0.2	<0.1	<0.1	8.1	1.1	216
	8	0.4	0.3	<0.1	<0.1	<0.1	0.1	7.9	1.3	174
9/9/1992	0	0.2	0.3	<0.1	<0.1	<0.1	<0.1	8.4	1.9	182
	2.5	0.2	<0.1	<0.1	<0.1	<0.1	<0.1	8.4	1.9	177
	4.5	0.2	<0.1	<0.1	<0.1	<0.1	<0.1	8.4	1.9	175
	6.5	0.4	<0.1	<0.1	<0.1	<0.1	<0.1	8.3	3.3	174
	8.5	1.2	0.7	<0.1	<0.1	<0.1	<0.1	8.2	4.4	168

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
	10	1.3	1.2	<0.1	<0.1	0.1	<0.1	8.4	4.5	167
10/8/1992	0	0.1	0.1	<0.1	<0.1	<0.1	<0.1	8.6	2.3	185
	2	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	8.7	2.3	180
	4	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	8.6	2.2	180
	6	<0.1	0.1	<0.1	<0.1	<0.1	<0.1	8.3	2.8	176
	8	1.4	1.2	<0.1	<0.1	0.4	<0.1	7.8	4.4	182
	9	1.7	1.2	<0.1	<0.1	<0.1	<0.1	7.7	4.7	169
11/3/1992	0	0.7	<0.1	<0.1	0.7	<0.1	<0.1	8.4	2.9	222
	2.5	0.6	<0.1	<0.1	0.7	<0.1	<0.1	8.3	3.2	229
	5	1.1	0.1	<0.1	0.8	<0.1	<0.1	8.2	2.9	221
	7.5	0.5	0.2	<0.1	0.8	<0.1	<0.1	8.1	2.7	209
	9.5	0.6	0.1	<0.1	0.7	<0.1	<0.1	8	2.7	271
	11.6	1	0.4	<0.1	0.2	<0.1	<0.1	7.9	3.5	153
12/17/1992	0	0.6	0.3	<0.1	0.5	<0.1	<0.1	7.9	3.1	188
	2	0.9	0.2	<0.1	0.5	<0.1	<0.1	7.9	3.2	191
	4.5	1	0.2	<0.1	0.4	<0.1	<0.1	8	3	180
	7.5	1.1	0.2	<0.1	0.5	<0.1	<0.1	8	3	184
	10.5	0.7	0.3	<0.1	0.4	<0.1	<0.1	8	3.6	179
	12.5	0.8	0.3	<0.1	0.5	<0.1	<0.1	8	3	184
1/27/1993	0	0.2	<0.1	<0.1	1.1	<0.1	<0.1	8.1	2	133
	4	0.3	<0.1	<0.1	1	<0.1	<0.1	8	2.5	116
	8	0.3	0.1	<0.1	1	<0.1	0.1	8	2.3	116
	12	0.4	0.1	<0.1	1.1	<0.1	<0.1	8	2.1	133
	16	0.4	0.1	<0.1	1.1	<0.1	<0.1	8.1	2	137
	20	0.3	<0.1	<0.1	1.1	<0.1	<0.1	8.1	1.9	129
2/16/1993	0	0.8	<0.1	<0.1	1	<0.1	0.1	8.6	2.4	148
	2	0.6	<0.1	<0.1	1	<0.1	0.1	8.5	3	123
	5	0.7	<0.1	<0.1	1.1	<0.1	0.9	8.3	2	145
	8	0.3	0.1	<0.1	1.1	<0.1	<0.1	8.2	1.9	142
	11	<0.1	0.2	<0.1	1.1	<0.1	<0.1	8.2	2.3	167
	14.5	0.5	0.2	<0.1	1.1	<0.1	<0.1	8.2	1.9	151
2/25/1993	0	0.3	<0.1	<0.1	1	<0.1	<0.1	8.1	2.2	126

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
	3	0.3	<0.1	<0.1	1	<0.1	<0.1	8.1	2.1	134
	6	0.2	<0.1	<0.1	0.9	0.1	<0.1	8.1	2.3	135
	9	0.1	<0.1	<0.1	0.9	0.1	<0.1	8.1	2.2	128
	12	0.3	<0.1	<0.1	0.9	<0.1	<0.1	8.2	2	131
	15.5	0.3	<0.1	<0.1	0.9	<0.1	<0.1	8.2	2	122
3/17/1993	0	0.2	<0.1	<0.1	0.8	<0.1	<0.1	8.7	1.8	163
	3	0.3	<0.1	<0.1	0.8	<0.1	<0.1	8.6	1.7	167
	6	0.2	<0.1	<0.1	0.9	<0.1	<0.1	8.3	2.3	148
	9	0.3	<0.1	<0.1	1.1	<0.1	<0.1	8.3	1.9	141
	12.5	0.1	<0.1	<0.1	1.1	<0.1	<0.1	8.1	1.9	154
	16	0.2	<0.1	<0.1	1.1	<0.1	<0.1	8.3	1.8	146
4/22/1993	0	<0.1	<0.1	<0.1	0.5	<0.1	<0.1	8.8	0.8	178
	2	<0.1	<0.1	<0.1	0.4	<0.1	<0.1	8.8	1.1	173
	5	<0.1	<0.1	<0.1	0.5	<0.1	<0.1	8.6	0.4	191
	8	<0.1	<0.1	<0.1	0.9	<0.1	<0.1	8.1	0.9	157
	11	<0.1	<0.1	<0.1	1	<0.1	<0.1	7.9	0.8	159
	14.5	<0.1	<0.1	<0.1	1.1	<0.1	<0.1	7.9	0.8	155
5/25/1993	0	0.4	<0.1	<0.1	0.3	<0.1	<0.1	8.7	2	201
	3.5	0.4	<0.1	<0.1	0.3	<0.1	<0.1	8.7	2.4	185
	6.5	0.4	<0.1	<0.1	0.6	<0.1	<0.1	8.2	2	183
	9.5	0.5	0.1	0.2	0.7	<0.1	<0.1	7.8	2	197
	12.5	0.4	<0.1	<0.1	0.6	<0.1	<0.1	8.1	1.7	190
	14	0.4	<0.1	0.2	0.7	<0.1	<0.1	7.8	1.7	162
6/23/1993	0	0.3	<0.1	<0.1	0.1	<0.1	<0.1	8.5	1.1	192
	2	0.2	<0.1	<0.1	0.1	<0.1	<0.1	7.9	1.2	167
	4	0.4	<0.1	<0.1	0.2	<0.1	<0.1	8.3	1.4	187
	7	0.4	<0.1	0.2	0.4	<0.1	<0.1	8.1	1.2	223
	9.5	0.6	0.3	<0.1	<0.1	<0.1	<0.1	8	1.3	173
	12	0.7	0.4	<0.1	<0.1	<0.1	<0.1	8.5	1.4	184

The Regional Board completed its Water Quality Assessment and Documentation Report for waterbodies in the Los Angeles Region in 1996 (LARWQCB, 1996). The summary table for Peck Road Park Lake states that dissolved oxygen (DO) was not supporting the aquatic life use: 195 measurements of DO were collected in the lake with concentrations ranging from 0.2 mg/L to 15.2 mg/L. The accompanying database does not contain the raw data associated with these measurements, so depth, temperature, date, and time cannot be established. The summary table also lists the odor impairment as not supporting both contact and non-contact recreation uses.

For Peck Road Park Lake, the 1996 water quality database contained eight station locations in the watershed; no stations were located within the lake for direct comparison to water quality standards. Table G-3 describes the stations contained in the database. Tetra Tech assigned labels to each site for mapping purposes. Data for Waterbody/Station ID combinations that had identical spatial coordinates were combined under one label. Table G-4 lists the nutrient data contained in the Water Quality Assessment Database for these locations.

Table G-3. Site Locations in the 1996 Water Quality Database for the Peck Road Park Lake Watershed

Waterbody	Station ID	Label
PECK ROAD SPREADING BASIN	Inlet	PRSB
Sawpit Wash	gage abv Peck Rd	SPWPR
Sawpit Wash	Peck Rd	SPWPR
Sawpit Wash	HUNTINGTON DRIVE	SPWHD
SAWPIT WASH DNS	MONROVIA CREEK	MCASPW
Santa Anita	[Blank]	SAWBEF
Santa Anita Cyn	[Blank]	SAWBEF
Santa Anita Wash	blw Live Oak Ave	SAWLO
Santa Anita Wsh	Live Oak Ave	SAWLO
Santa Anita Wsh	Colorado Blvd	SAWCB
STAFTH	SANTAANITAWASH@FOOTHILLBLVD	SAWFB

Table G-4. Water Quality Assessment Data for Tributaries in the Peck Road Park Lake Watershed

Date	Time	Station ID	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	pH
12/27/1977	10:34	PRSB	N/A	N/A	N/A	N/A	N/A	7.1
12/27/1977	07:00	MCASPW	N/A	N/A	N/A	N/A	N/A	7.6
12/28/1977	12:54	SPWHD	N/A	N/A	N/A	N/A	N/A	7.5
1/4/1978	16:15	MCASPW	N/A	N/A	N/A	N/A	N/A	7.4
1/6/1978	11:00	SPWHD	N/A	N/A	N/A	N/A	N/A	7.9
1/6/1978		SAWFB	N/A	N/A	N/A	N/A	N/A	7.2
1/10/1978	11:15	PRSB	N/A	N/A	N/A	N/A	N/A	7.5
1/16/1978	21:15	PRSB	N/A	N/A	N/A	N/A	N/A	7.8

Date	Time	Station ID	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	pH
2/7/1978	20:45	PRSB	N/A	N/A	N/A	N/A	N/A	7.0
2/9/1978	10:20	SPWHD	N/A	N/A	N/A	N/A	N/A	7.5
2/9/1978	10:09	SAWFB	N/A	N/A	N/A	N/A	N/A	7.2
2/28/1978	20:05	PRSB	N/A	N/A	N/A	N/A	N/A	7.5
3/2/1978	10:26	SPWHD	N/A	N/A	N/A	N/A	N/A	7.6
3/2/1978	10:20	SAWFB	N/A	N/A	N/A	N/A	N/A	7.4
3/4/1978	08:10	PRSB	N/A	N/A	N/A	N/A	N/A	7.3
3/18/1980		SAWBEF	0.60	N/A	N/A	N/A	<0.01	N/A
8/13/1980		SAWBEF	<0.1	N/A	N/A	N/A	0.08	N/A
12/3/1980		SAWBEF	0.10	N/A	N/A	N/A	0.05	N/A
4/13/1981		SAWBEF	N/A	N/A	N/A	1.30	<0.01	8.5
8/5/1981		SAWBEF	N/A	N/A	N/A	0.70	N/A	8.3
11/9/1981		SAWBEF	N/A	N/A	N/A	0.50	0.03	8.1
3/11/1982		SAWBEF	N/A	N/A	N/A	0.60	0.03	8.1
5/27/1982		SAWBEF	N/A	N/A	N/A	0.56	0.02	7.8
8/13/1982	14:40	SAWBEF	N/A	N/A	N/A	1.37	0.06	8.3
10/26/1982	14:00	SAWBEF	N/A	N/A	N/A	1.00	<0.03	8.1
5/6/1983		SAWBEF	N/A	N/A	N/A	0.47	0.09	8.2
9/22/1983	14:45	SAWBEF	N/A	N/A	N/A	0.70	0.05	8.0
8/21/1987		SAWBEF	N/A	N/A	N/A	0.23	0.07	7.9
9/30/1988		SAWBEF	N/A	N/A	N/A	0.20	N/A	7.8
10/2/1989		SAWBEF	N/A	N/A	N/A	0.16	<0.16	7.9
10/2/1990		SAWBEF	N/A	N/A	N/A	0.19	<0.1	7.8
4/18/1991		SPWPR	N/A	0.80	<0.03	0.80	0.06	9.0
5/8/1991		SAWBEF	N/A	N/A	N/A	0.40	0.14	8.2
5/14/1992		SAWCB	N/A	N/A	N/A	N/A	N/A	8.4
5/14/1992		SAWLO	N/A	N/A	N/A	N/A	N/A	8.4
12/22/1992		SAWLO	N/A	N/A	<0.03	4.40	N/A	N/A
12/22/1992		SPWPR	N/A	N/A	<0.03	<0.2	N/A	N/A
5/13/1997	09:55	SAWLO	3.60	0.80	<0.03	<0.2	0.10	8.7
5/13/1997	10:15	SPWPR	0.60	0.20	<0.03	<0.2	0.02	8.6

N/A = No data available.

In 2008, Regional Board collected water quality samples from several locations in Peck Road Park Lake and its inflows (Figure G-2). Site location information is listed in Table G-5.

Table G-5. Site Locations for the 2008 Peck Road Park Lake Monitoring Event

Site number	Project Site	Comment
Inflows		
6	Sawpit Wash (SPW)	Inflow
7	Santa Anita Wash (SAW)	Inflow
20	Santa Anita Wash (SAW)	Site 7-Field Duplicate
8	North Basin Outfall (NBO)	Stormwater outfall to North Basin
Mid-Lake Sites		
9	North Basin (NB)	
10	South Basin (SB)	This site was moved to the narrow section connecting the north and south basins for the December sampling event
16	South Basin (SB)	
17	South Basin (SB)	

Analytical data for the June 17, 2008 sampling event are listed in Table G-6 (sites 16 and 17 were not monitored during this event). Four of the six sites had NH_3-N concentrations less than the reporting limit of 0.1 mg/L; the maximum ammonia concentration was 0.437 mg/L. TKN ranged from 1.2 mg/L to 10 mg/L, with the higher concentrations observed at the two major inflow sites (6, 7/20). Nitrate concentrations ranged from 0.22 mg/L to 0.58 mg/L with two measurements less than the reporting limit of 0.1 mg/L. Each site had measurements of nitrite and orthophosphate less than the reporting limits of 0.1 mg/L and 0.4 mg/L, respectively. All but one site (Sawpit Wash) had total phosphate concentrations less than the reporting limit of 0.5 mg/L.

Table G-6. Analytical Data for the June 17, 2008 Peck Road Park Lake Sampling Event

Station Number	NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Ortho-phosphate (mg/L)	Total phosphate (mg/L)	Total Dissolved Solids (mg/L)
Inflows							
6	0.179	6.5	<0.1	<0.1	<0.4	0.715	303
7	<0.1	9.26	0.58	<0.1	<0.4	<0.5	1,517
20	<0.1	10	0.44	<0.1	<0.4	<0.5	1,401
8	<0.1	1.32	0.22	<0.1	<0.4	<0.5	145
Mid-lake Sites							
9	<0.1	1.2	0.24	<0.1	<0.4	<0.5	142
10	0.437	2.08	<RL	<0.1	<0.4	<0.5	134

Field data collected at these sites are summarized in Table G-7. Field data were collected in Peck Road Park Lake (sites 10 and 9) at depths ranging from the water surface to 2.5 meters. Temperature varied by approximately 1 ºC in the south basin and approximately 4 ºC in the north basin over the sampling depth. Dissolved oxygen in the lake was elevated at all depths except station 9 at a depth of 2.5 meters. Electrical conductivity was fairly constant at both sites and ranged from 0.17 mS/cm to 0.185 mS/cm. pH measurements in the lake ranged from 8.0 to 9.4, although the meter was not calibrated due to equipment malfunction and this data should not be used quantitatively. Chlorophyll *a* measurements in the lake ranged from 4.0 µg/L to 11.4 µg/L. The total depth at site 10 was approximately 8 meters and the Secchi depth was 3.2 meters; the total depth at site 9 was approximately 5.2 meters and the Secchi depth was 2.3 meters.

Site 6 (Sawpit Wash) had the highest observed temperature (33.5 ºC) and chlorophyll *a* concentration (16 µg/L). Dissolved oxygen was 9 mg/L; electrical conductivity was about two times higher than that observed in the lake. Site 7 (Santa Anita Wash) had observed temperatures slightly less than that measured in the lake; dissolved oxygen was approximately 11 mg/L. Electrical conductivity at this site was the highest (1.726 mS/cm); chlorophyll *a* was slightly higher than the majority of measurements taken in the lake (10.4 µg/L). Site 8 is a stormwater outfall at the downstream end of the north basin. Readings at this site were generally similar to those measured in the lake.

Table G-7. Field Data for the June 17, 2008 Peck Road Park Lake Sampling Event

Site	Time	Depth (m)	Temp (C)	DO (mg/L)	EC (mS/cm)	pH[1]	Chl a (µg/L)	Secchi Depth (m)
10	Samples at this site collected between 10:17 and 11:05	surface	26.4	17.8	0.17	9.3	4	3.2
		0.5	26	18.7	0.17	9.4	9.4	
		1	25.7	19	0.17	9.4	4.9	
		1.5	25.6	19.2	0.17	9.4	6	
		2	25.5	19.4	0.17	9.4	6.7	
		2.5	25.4	19.5	0.17	9.4	7.5	
9	Samples at this site collected between 12:38 and 13:30	surface	28.32	18.67	0.185	9.32	4.1	2.3
		0.5	27.62	18.77	0.184	9.3	3.8	
		1	26.59	19.46	0.183	9.23	4.8	
		1.5	26.18	19.78	0.182	9.14	6.6	
		2	25.8	17	0.18	8.9	7.8	
		2.5	23.9	3	0.18	8	11.4	
8	14:57	surface	29.5	20.1	0.18	9.4	6.2	NA
6	17:18	surface	33.5	9	0.37	9.9	16	NA
7	18:40	surface	24.31	11.17	1.726	9.62	10.4	NA

[1] pH calibration was outside of accepted range. Data should not be used quantitatively.

Four sites were sampled by the Regional Board on December 11, 2008. Samples were collected from the surface at each site. Table G-8 summarizes the nutrient data collected. Measurements of TKN, nitrite, orthophosphate, and total phosphate were less than the reporting limit at each site. Ammonia concentrations ranged from 0.209 mg/L to 0.273 mg/L; nitrate ranged from 0.162 mg/L to 0.287 mg/L. Total dissolved solids ranged from 154 mg/L to 178 mg/L. Suspended solids were less than the reporting limit at each site except for site 16. Chlorophyll *a* ranged from 1.8 µg/L to 4.0 µg/L.

Table G-8. Analytical Data for the December 11, 2008 Peck Road Park Lake Sampling Event

Station Number	NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Ortho phosphate (mg/L)	Total phosphate (mg/L)	Total Dissolved Solids (mg/L	Suspended Solids (mg/L)	Chl *a* (µg/L)
Mid-Lake Sites									
9	0.273	<1.0	0.287	<0.1	<0.4	<0.5	154	<10	4.0
10*	0.209	<1.0	0.173	<0.1	<0.4	<0.5	178	<10	3.6
16	0.262	<1.0	0.164	<0.1	<0.4	<0.5	175	24	3.1
17	0.269	<1.0	0.162	<0.1	<0.4	<0.5	165	<10	1.8

Field data for the December 11, 2008 event are summarized in Table G-9.

Table G-9. Field Data for the December 11, 2008 Peck Road Park Lake Sampling Event

Station Number	Time	Depth (m)	Temp (C)	DO[1] (mg/L)	EC (mS/cm)	pH	Secchi Depth (m)	Total Depth (m)
9	8:15	Surface	15.61	NA	0.206	7.51	1.6	6.4
		0.5	15.6	2.44	0.204	7.48		
		1.0	15.6	2.29	0.204	7.48		
		1.5	15.59	2.23	0.204	7.47		
		2.0	15.59	2.21	0.204	7.47		
10*	10:07	Surface	16.30	6.15	0.234	7.79	1.1	2.3
		0.5	15.99	6.20	0.231	7.80		
		1.0	15.72	5.41	0.227	7.70		
		1.5	15.52	3.91	0.215	7.53		
		2.0	15.47	3.27	0.213	7.51		
16	11:45	Surface	17.29	5.77	0.237	7.74	1.6	2.1
		0.5	16.44	6.15	0.236	7.81		
		1.0	16.26	6.07	0.236	7.80		
		1.5	16.19	5.71	0.236	7.78		
		2.0	16.15	5.57	0.236	7.76		
17	12:24	Surface	16.62	4.94	0.236	7.70	1.8	11.1
		0.5	16.20	4.78	0.236	7.69		
		1.0	16.14	4.61	0.236	7.68		
		1.5	16.11	4.54	0.236	7.68		
		2.0	16.09	4.56	0.236	7.69		

[1] Field team questioned measurement of DO for this event. Meter was not calibrated prior to sampling.

Water quality monitoring was also conducted by the Regional Board on August 5, 2009 at Stations 9 and 10. The data from this event are shown in Table G-10.

Table G-10. Analytical Data for the August 5, 2009 Peck Road Park Lake Sampling Event

Station Number	NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Ortho phosphate (mg/L)	Total phosphorus (mg/L)	Total Dissolved Solids (mg/L)	Suspended Solids (mg/L)	Chl *a* (µg/L)
PRPL-9	<0.03	<0.456	<0.01	<0.01	0.0135	0.022	205	3.9	8.0
PRPL-10	<0.03	<0.456	<0.01	<0.01	0.0112	0.116	194	3.2	5.3

Profile data were also collected at stations PRPL-9 and PRPL-10 on August 5, 2009. Profiles were performed at 9:00 a.m. at station PRPL-9, and at 8:00 a.m. and 3:00 p.m. at station PRPL-10. These data are displayed in Figure G-3 through Figure G-5. At station PRPL-9, the specific conductivity was between 0.340 and 0.373 mS/cm. The pH ranged from 7.69 and 8.56. The maximum DO in the lake was 11.79 mg/L, at a depth of 1.48 meters. Below 1.48 meters, the DO steadily declines to 3.34 mg/L at 7 meters of depth. The temperature near the surface of the water was 24.5 °C and starts to decline at 1 meter of depth. The minimum temperature was 16.05 °C.

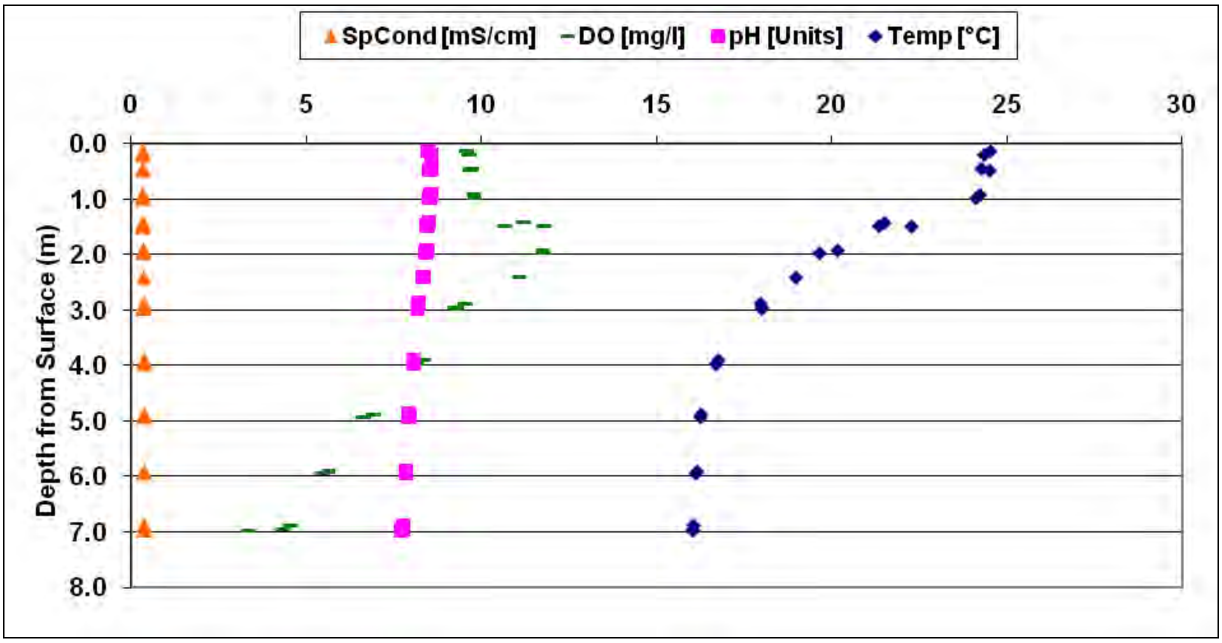


Figure G-3. Profile Data Collected at PRPL-9 in Peck Road Park Lake on August 5, 2009

Profile data were collected at the PRPL-10 at 8:00 a.m. and 3:00 p.m. The morning profile is shown in Figure G-4. The temperature in the lake ranges from 16.44 to 23.38 °C. The maximum DO is 11.78 mg/L and occurs at 2.22 meters of depth. The minimum DO was 2 mg/L at 10 meters of depth. The pH ranged from 8.45 to 7.51. The specific conductivity was constant with depth.

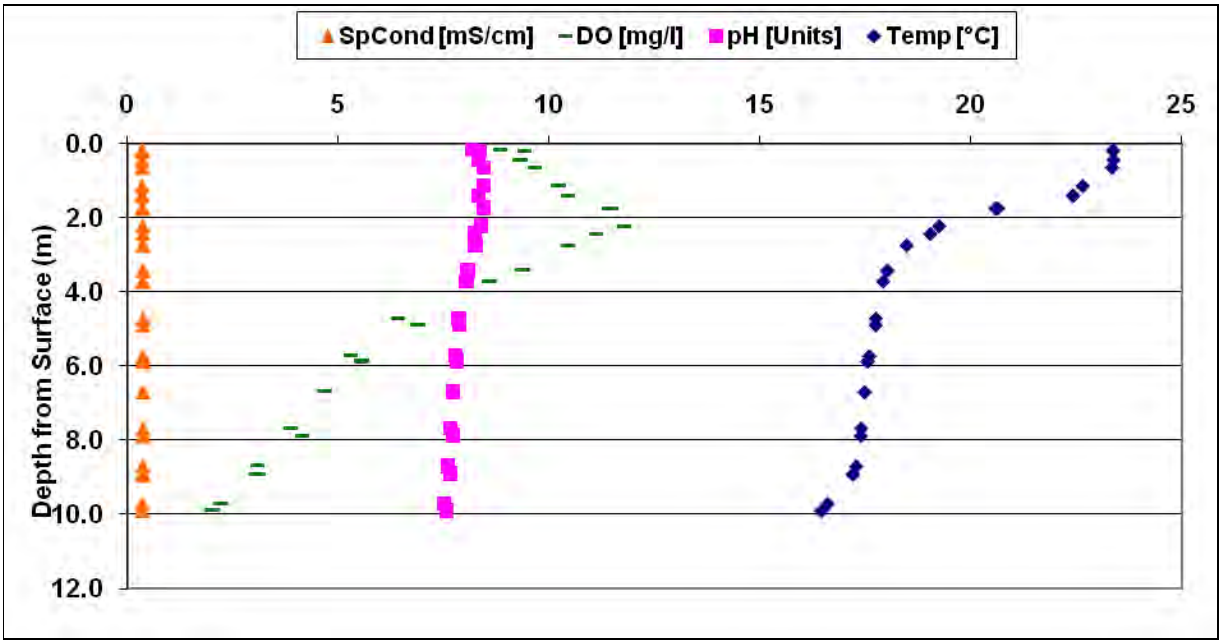


Figure G-4. **Profile Data Collected at PRPL-10 in Peck Road Park Lake on August 5, 2009 at 8:00 a.m.**

The profile data collected in the afternoon at Station PRPL-10 is shown in Figure G-5. The specific conductivity was constant with depth and the pH ranged from 7.53 to 8.71. The maximum DO was 12.02 mg/L at 2.03 meters. The DO decreased with depth after 2 meters, to a minimum of 1.22 mg/L. The temperature of the lake was between 24.04 and 17.07°C. Similar to the DO, temperature decreased with depth after 2.03 meters.

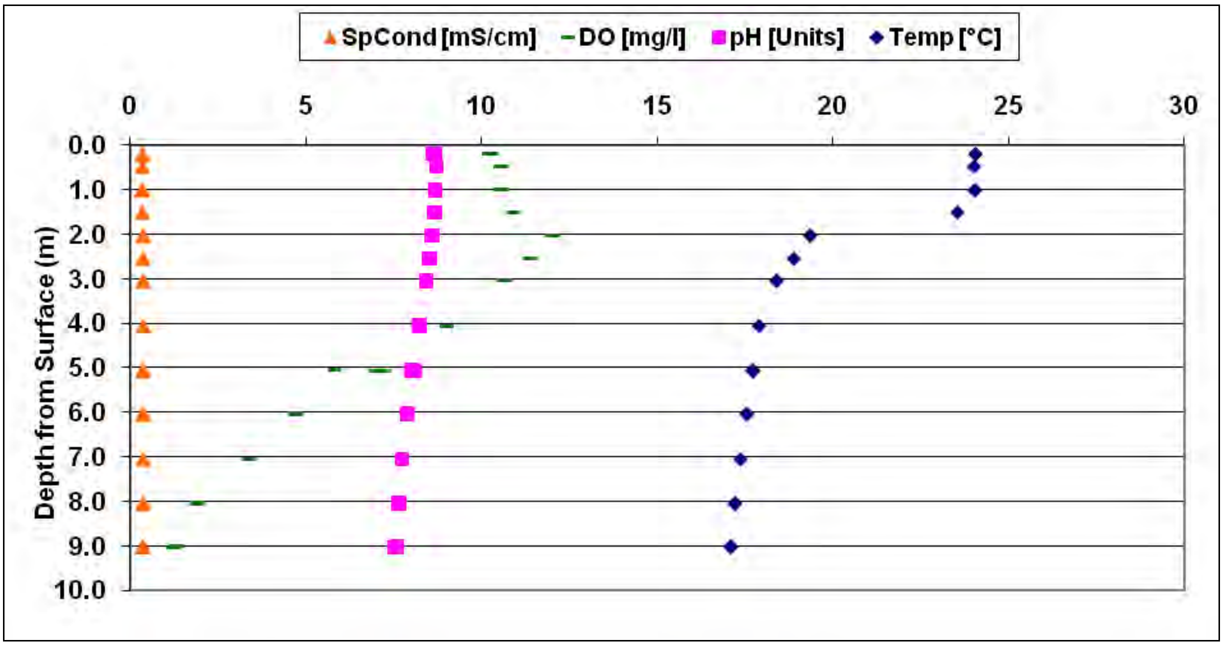


Figure G-5. **Profile Data Collected at PRPL-10 in Peck Road Park Lake on August 5, 2009 at 3:00 p.m.**

The DO saturation for the morning and afternoon readings at PRPL-10 are shown below in Figure G-6. The DO saturation ranges from 13 to 131 percent. DO saturation above 100 percent indicates additional oxygen input from algal productivity. The maximum DO saturation occurs at 2 meters of depth in the euphotic zone.

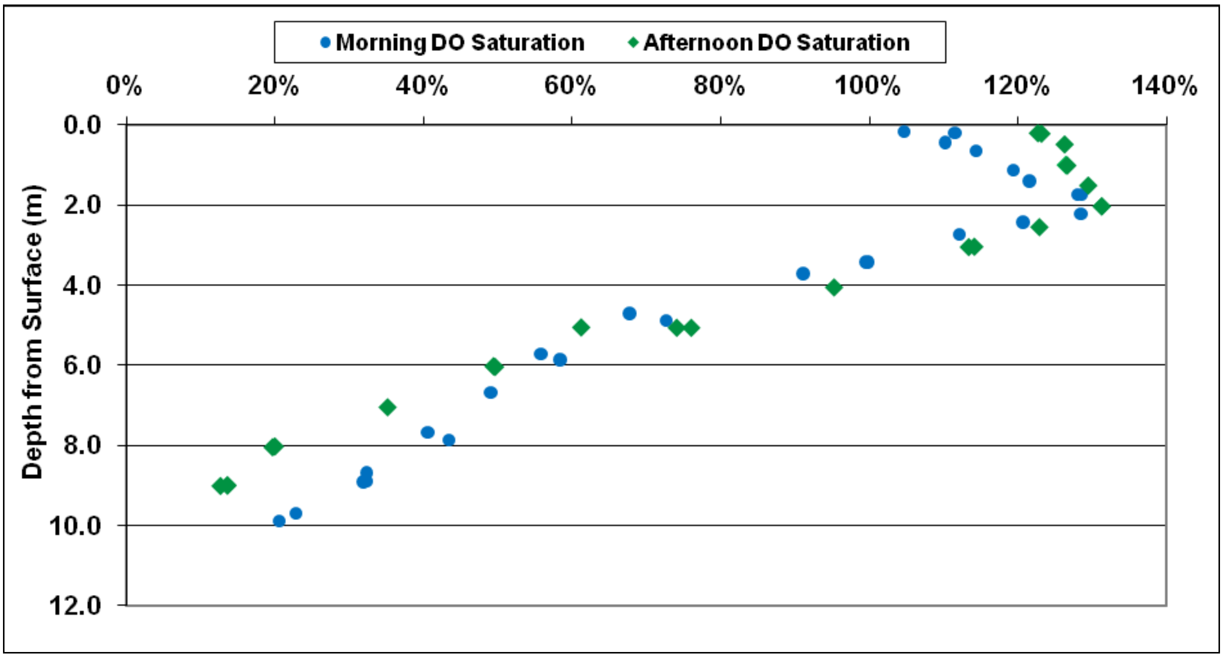


Figure G-6. DO Saturation from Profile Data Collected at PRPL-10 in Peck Road Park Lake on August 5, 2009

On September 30, 2010, additional sampling was conducted at the mid-lake sites (Table G-11). Ammonia concentrations were below the detection limit of 0.03 mg-N/L. Nitrite ranged from 0.041 to 0.043 mg-N/L, and nitrate was below the detection limit of 0.01 mg-N/L (note: nitrite values were higher than nitrate. These samples passed the laboratory QA/QC protocols, so they are considered valid). TKN ranged from 0.562 to 0.634 mg-N/L. Orthophosphate and total phosphorus ranged from 0.02 mg-P/L to 0.04 mg-P/L. Chlorophyll *a* ranged from 6.7 µg/L to 13.4 µg/L.

Table G-11. Analytical Data for the September 30, 2010 Peck Road Park Lake Sampling Event

Station Number	NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Ortho Phosphate (mg/L)	Total Phosphorus (mg/L)	Total Dissolved Solids (mg/L)	Suspended Solids (mg/L)	Chl *a* (µg/L)
PRPL-9	<0.03	0.634	<0.01	0.041	0.040	0.040	220	3.25	13.4
PRPL-10	<0.03	0.574	<0.01	0.043	0.024	0.022	200	0.75	6.68
PRPL-10 (Duplicate)	<0.03	0.562	<0.01	0.042	0.025	<0.0165	160	1.50	7.12

During the September 2010 sampling event, two continuous monitoring probes were deployed over a 24-hour period (Figure G-7 and Figure G-8). At an average depth of 0.6 meters, DO concentrations ranged from 8.6 mg/L to 10.1 mg/L. pH ranged from about 8.5 to 8.8. On September 30, 2010, depth profile measurements were also taken and are shown in Table G-12, Figure G-9, and Figure G-10. DO measurements collected from the surface of the lake ranged from 8.5 mg/L to 10.9 mg/L. At 2 meters above the bottom, DO ranged from 0.2 to 4.0 mg/L. Specific conductivity was not recorded during the profile measurements.

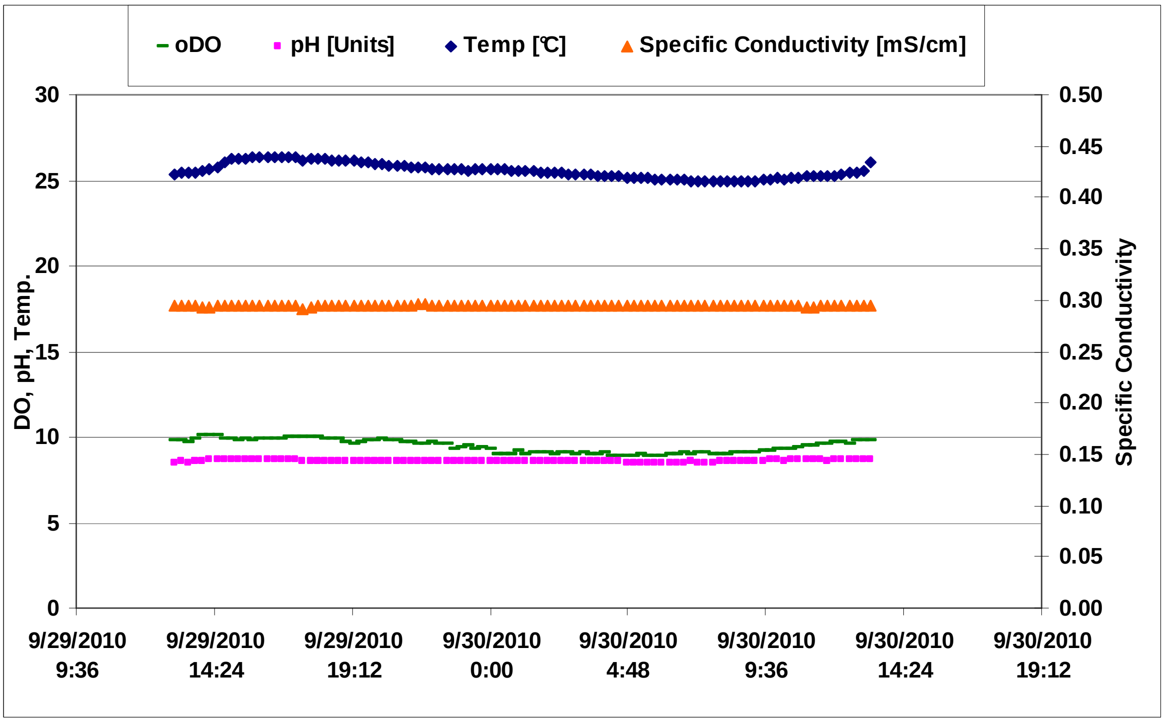


Figure G-7. 24-Hour Probe Data Collected at PRPL-9 on September 29, 2010

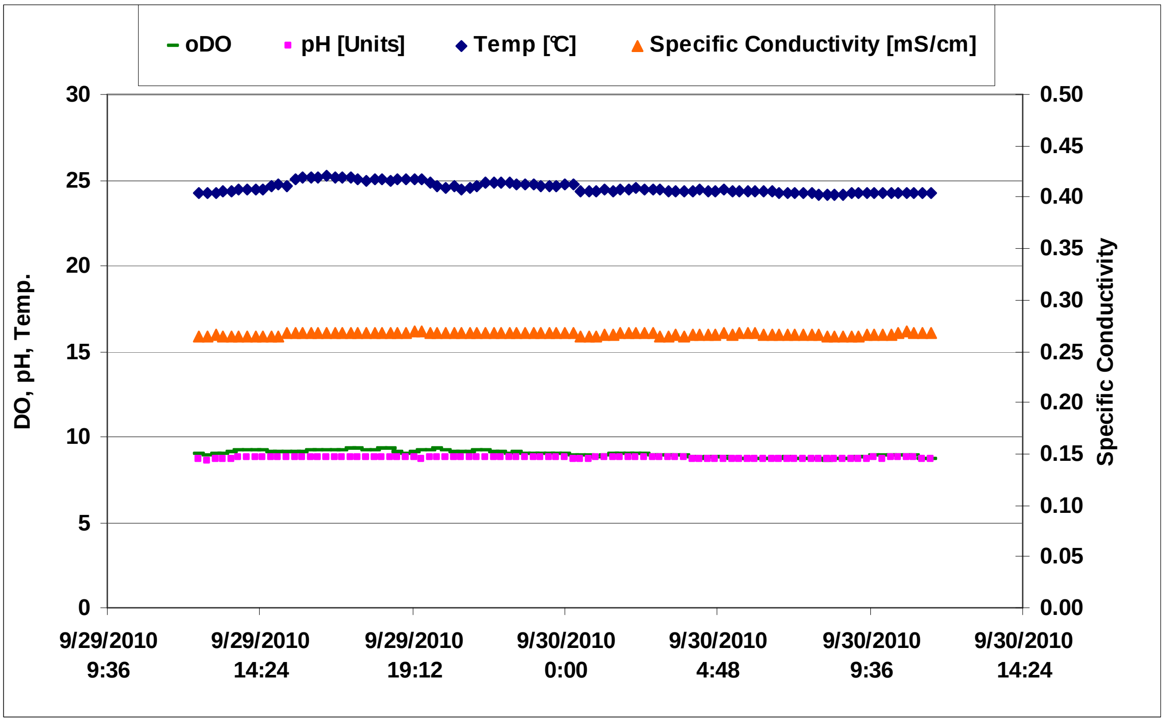


Figure G-8. 24-Hour Probe Data Collected at PRPL-10 on September 29, 2010

Table G-12. Profile Data Collected at Peck Road Park Lake on September 30, 2010

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Orp (mV)
PRPL-9	11:15	0.5	25.61	8.49	10.87	25.2
		1	25.10	8.51	11.04	24.7
		1.5	24.76	8.52	11.20	24.9
		2	24.32	8.36	10.40	32.8
		2.5	23.37	8.18	9.10	37.8
		3	22.70	7.83	--	48.7
		3.5	22.25	7.41	2.83	50.3
		4	22.02	7.25	0.20	19.5
PRPL-10	11:33	0.5	24.85	8.34	8.54	18.0
		1	24.71	8.37	8.46	17.9
		1.5	24.57	8.41	8.73	15.6
		2	24.51	8.41	8.73	14.4
		2.5	24.40	8.39	8.50	13.1
		3	24.06	8.21	7.12	13.4
		3.5	23.88	8.18	6.92	11.2
		4	23.07	7.71	4.02	20.0

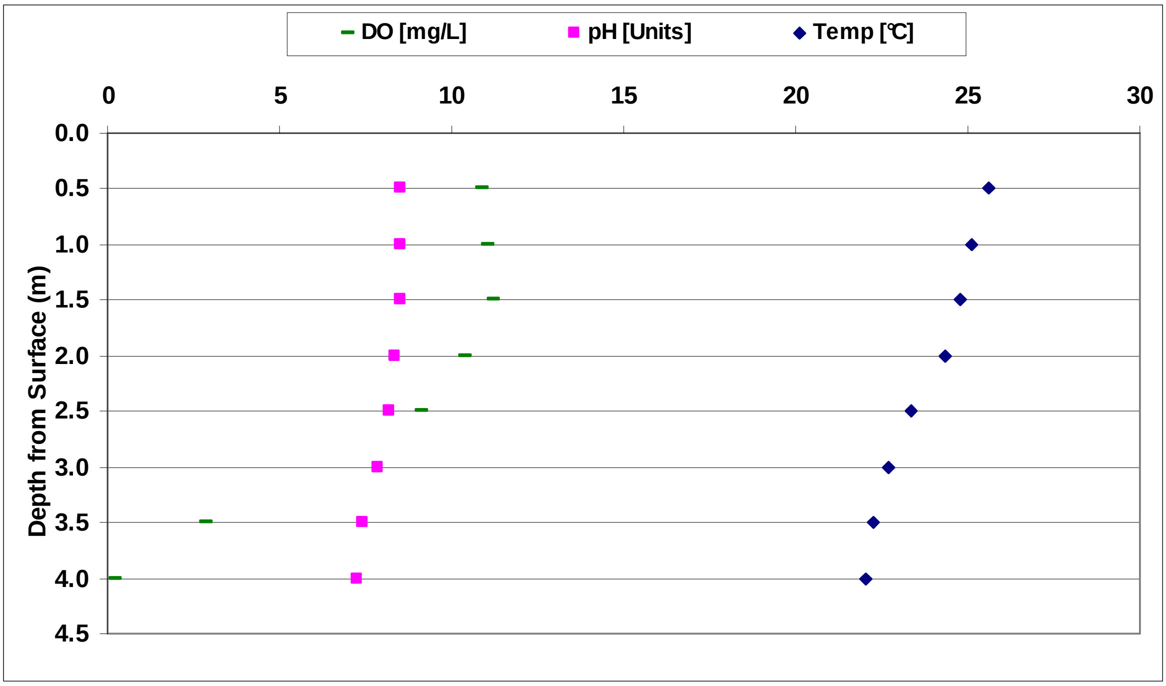


Figure G-9. **Profile Data Collected at PRPL-9 on September 30, 2010**

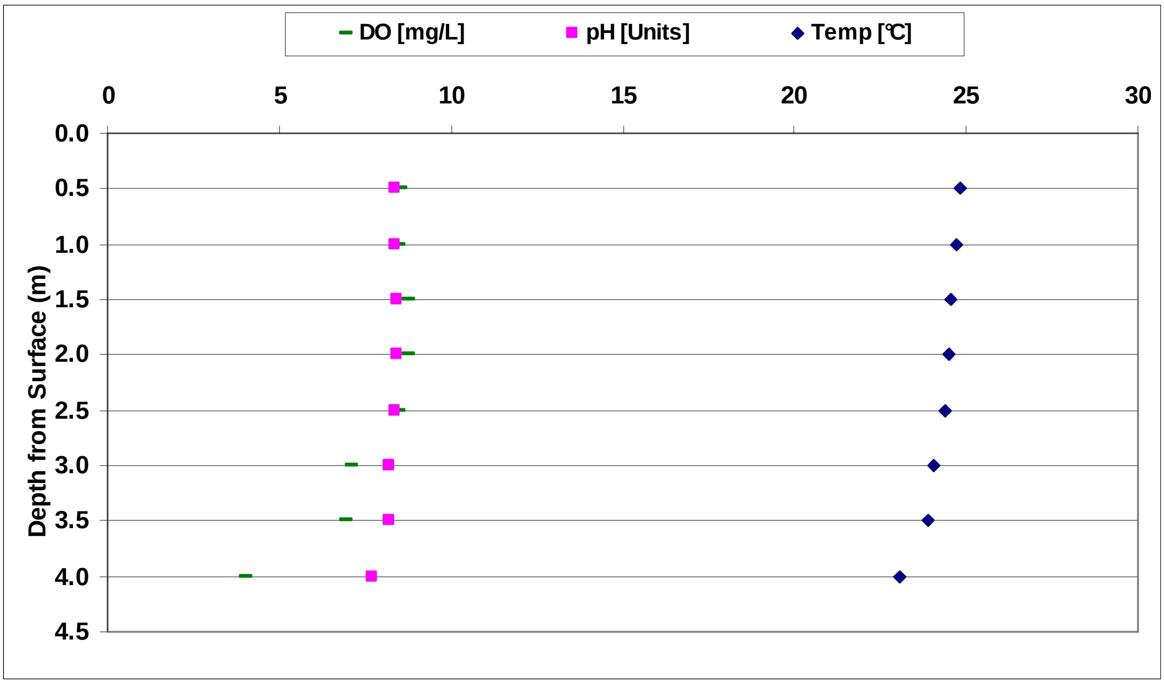


Figure G-10. **Profile Data Collected at PRPL-10 on September 30, 2010**

The DO saturation at PRPL-10 is shown below in Figure G-11. The DO saturation ranges from 47 to 106 percent. DO saturation above 100 percent indicates additional oxygen input from algal productivity. The maximum DO saturation occurs at 1.5 and 2 meters of depth in the euphotic zone.

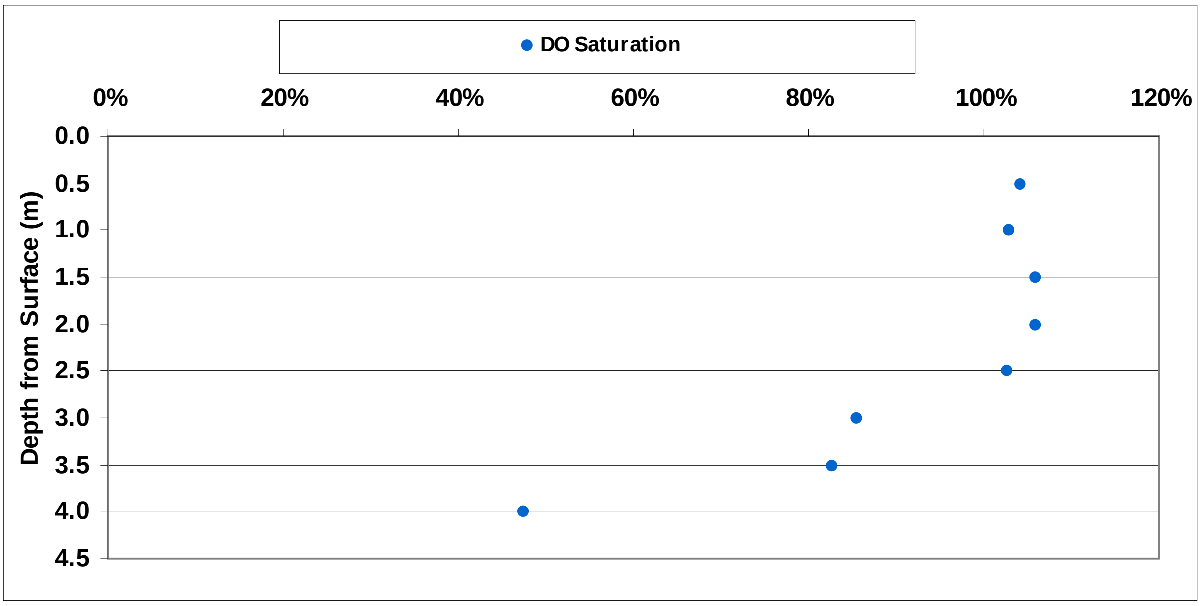


Figure G-11. DO Saturation from Profile Data Collected at PRPL-10 in Peck Road Park Lake on September 30, 2010

Sediment samples were also collected during the September 2010 monitoring event. Table G-13 summarizes these data.

Table G-13. September 30, 2010 Sediment Monitoring Data for Peck Road Park Lake

Location	Time	TKN (mg/kg)	NH_3-N (mg/kg)	NO_2-N (mg/kg)	NO_3-N (mg/kg)	PO_4-P (mg/kg)	Total P (mg/kg)	Total Organic Carbon (% by wt.)	Acid Volatile Sulfides (mg/kg)	Percent Solids	Total Hard-ness (mg/kg)
PRPL-9	11:30	5500	36.2	1.15	1.51	0.231	35.3	4.81	30.7	36.5	38400
PRPL-10	10:45	3180	26.6	1.34	1.64	0.0446	92.3	3.98	135	33.6	27400
PRPL-10 (Duplicate)	10:45	4170	28.5	1.31	1.67	0.0337	16.2	4.07	145	33.6	28400

G.4.2 MONITORING RELATED TO LEAD IMPAIRMENT

In 1996 Peck Road Park Lake was deemed impaired by lead. Monitoring data for cadmium, copper, lead, and zinc are presented in this section. Peck Road Park Lake is not listed for cadmium, copper, or zinc, but those data are presented here for completeness because other waterbodies in the region are affected by some of these contaminants.

Metals data collected at Peck Road Park Lake, as part of the 1992-1993 Urban Lakes Study (UC Riverside, 1994), are presented in Table G-14. Samples were collected from the middle of the south basin (pink triangle, Figure G-2) and included dissolved copper and dissolved lead. Dissolved copper samples were collected throughout the water column at depths from the surface to 20 meters. The range of the 90 dissolved copper samples was between less than 10 µg/L and 69 µg/L. Similarly, dissolved lead samples were also collected throughout the water column, again at depths from the surface to 20 meters. The 90 samples collected ranged in concentration from less than 1 µg/L to 82 µg/L.

The Regional Board completed its Water Quality Assessment and Documentation Report for waterbodies in the Los Angeles Region in 1996 (LARWQCB, 1996). The 1996 summary table for Peck Road Park Lake states that lead was not supporting the assessed uses: 90 measurements had a maximum lead concentration of 73 µg/L, a maximum copper concentration of 69 µg/L, and a maximum zinc concentration of 47 µg/L (raw data were not provided, but it is assumed that most of these samples are associated with the Urban Lake Study [UC Riverside, 1994]).

Unfortunately, metals levels were analyzed at relatively high detection limits compared to current detection limits; dissolved copper minimum detection 10 µg/L while dissolved lead was 1 µg/L. No hardness data were collected as part of the Urban Lakes Study, thus it cannot be compared to the hardness-based water quality objectives.

Table G-14. Peck Road Park Lake 1992/1993 Monitoring Data for Metals

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
7/7/1992	0	<10	<1
	3	<10	<1
	5.5	<10	<1
	7	18	9
	9	47	21
7/7/1992	0	21	<1
	3	22	1
	5.5	15	<1
	6	26	15
	7.5	41	18
7/23/1992	0	<10	11
	3.5	12	<1
	6.5	<10	<1
	8.5	<10	<1
	10.5	<10	<1
	13	N/A	<1
7/23/1992	0	<10	<1
	2	<10	<1

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
	4	<10	<1
7/23/1992	0	<10	<1
	2	<10	<1
	4	<10	<1
	6	<10	<1
	8	<10	1
9/9/1992	0	12	<1
	2.5	<10	<1
	4.5	<10	1
	6.5	<10	<1
	8.5	10	2
	10	<10	2
10/8/1992	0	<10	<1
	2	<10	<1
	4	<10	<1
	6	<10	<1
	8	<10	<1
	9	<10	<1
11/3/1992	0	55	1
	2.5	18	1
	5	19	1
	7.5	36	1
	9.5	53	2
	11.6	19	3
12/17/1992	0	<10	<1
	2	<10	<1
	4.5	<10	<1
	7.5	<10	<1
	10.5	<10	1
	12.5	<10	<1
1/27/1993	0	<10	<1
	4	<10	<1

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
	8	<10	27
	12	<10	16
	16	<10	18
	20	<10	2
2/16/1993	0	<10	<1
	2	15	4
	5	16	6
	8	<10	<1
	11	<10	<1
	14.5	<10	<1
2/25/1993	0	<10	<1
	3	<10	3
	6	<10	<1
	9	<10	<1
	12	<10	<1
	15.5	<10	<1
3/17/1993	0	69	<1
	3	<10	39
	6	<10	43
	9	<10	66
	12.5	<10	53
	16	<10	73
4/22/1993	0	<10	17
	2	<10	43
	5	<10	64
	8	<10	31
	11	<10	33
	14.5	<10	12
5/25/1993	0	<10	6
	3.5	<10	3
	6.5	<10	<1
	9.5	<10	1

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
	12.5	<10	6
	14	<10	11
6/23/1993	0	<10	82
	2	<10	2
	4	<10	<1
	7	<10	<1
	9.5	<10	<1
	12	<10	<1

Table G-15 presents 30 additional metals samples that were collected by the USEPA, Regional Board, and/or the County of Los Angeles between December 2008 and September 2010. Samples were collected at locations PRPL-8, PRPL-9, PRPL-10, and PRPL-11B in 2009 and 2010, while PRL 10*/16/17 and PRL 09 were sampled in 2008. Sites were analyzed for dissolved cadmium, copper, lead, and zinc.

Detection limits were lower than the 1992-1993 study with a cadmium detection limit of 0.2 µg/L, dissolved copper detection limit of 0.4 µg/L, dissolved lead detection limit of 0.05 µg/L, and dissolved zinc detection limit of 0.2 µg/L. All dissolved cadmium concentrations were less than 0.2 µg/L; copper concentrations ranged from <0.4 µg/L to 10.2 µg/L; lead concentrations were between <0.05 µg/L and 1 µg/L; and zinc concentrations ranged from <0.1 µg/L to 14.8 µg/L. Metals toxicity is affected by hardness; therefore, each sample was also analyzed for hardness. The 2008-2010 sampling resulted in a hardness range of 40 mg/L to 102 mg/L. In addition, two total lead samples were collected by the Regional Board in June 2008 at PRL 09 (North Basin) and PRL 10 (South Basin). The total lead concentrations were 5.8 µg/L and 11.8 µg/L, respectively (with 96 mg/L and 88 mg/L hardness values, respectively). Since dissolved results pertain to the applicable standard and recent data more closely represents current conditions, data in Table G-15 were weighted more heavily in the assessment.

Table G-15. Metals Data for the 2008-2010 Peck Road Park Lake Sampling Events

Organi-zation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
RB	12/11/2008	PRL 10*/16/17	102	<0.2	1.1	<0.1	2.8	average of stations 10*, 16, and 17
RB	12/11/2008	PRL 09	84	<0.2	1.7	0.1	4.2	average of replicates
RB/EPA	8/5/2009	PRPL 8	121	<0.2	4.7	0.2	4.7	
RB/EPA	8/5/2009	PRPL 9	121	<0.2	10.2	0.3	11.2	average of replicates
RB/EPA	8/5/2009	PRPL 10	122	<0.2	5.1	0.2	7.1	
RB/EPA	8/5/2009	PRPL 11	122	<0.2	4.4	0.1	3.7	
EPA/ County	11/16/2009	PRPL-10	116	<0.2	0.4	<0.1	1.6	average of filtered samples

Organi-zation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
EPA/ County	11/16/2009	PRPL-11B	117	<0.2	0.4	<0.1	<0.1	average of filtered samples
EPA/ County	11/16/2009	PRPL-8	109	<0.2	1.1	0.6	1.7	average of replicates and filtered samples
EPA/ County	11/16/2009	PRPL-9	108	<0.2	0.9	0.3	8	average of duplicate and filtered samples
County	12/8/2009	PRPL-10	114	<0.2	<0.4	<0.1	<0.1	
County	12/8/2009	PRPL-11B	113	<0.2	<0.4	<0.1	<0.1	
County	12/8/2009	PRPL-8	88	<0.2	3	1	5.9	average of replicates
County	12/8/2009	PRPL-9	87	<0.2	3.3	1	9.4	average of duplicates
EPA	12/14/2009	PRPL-11B	89	<0.2	1.1	<0.1	4.1	
EPA	12/14/2009	PRPL-9	40	<0.2	2.8	0.3	14.6	
EPA	12/14/2009	PRPL-10	83	<0.2	1.2	<0.1	3.7	average of duplicates
EPA	12/14/2009	PRPL-8	40	<0.2	2.9	0.5	14.8	average of replicates
County	1/28/2010	PRPL-11B	63	<0.2	1.8	0.2	3.7	
County	1/28/2010	PRPL-9	59	<0.2	2.3	0.2	7.2	
County	1/28/2010	PRPL-10	63	<0.2	1.8	0.2	6.6	average of duplicates
County	1/28/2010	PRPL-8	59	<0.2	2.3	0.2	4.1	average of replicates
County	2/17/2010	PRPL-11B	59	<0.2	1.8	0.1	3.7	
County	2/17/2010	PRPL-9	73	<0.2	2.1	0.2	6.4	
County	2/17/2010	PRPL-10	64	<0.2	2.1	0.1	3.1	average of duplicates
County	2/17/2010	PRPL-8	59	<0.2	2.0	0.2	5.1	average of replicates
EPA / RB	9/30/2010	PRPL-8	76	<0.2	<0.4	<0.05	<0.1	
EPA / RB	9/30/2010	PRPL-9	75	<0.2	<0.4	<0.05	<0.1	
EPA / RB	9/30/2010	PRPL-10	66	<0.2	<0.4	<0.05	<0.1	
EPA / RB	9/30/2010	PRPL-11	66	<0.2	<0.4	<0.05	<0.1	

RB = Regional Board

EPA = USEPA

County = County of Los Angeles

USEPA also collected two sediment samples during September 2010 to further evaluate lake conditions. Table G-16 summarizes the lead concentrations measured in the samples. There were zero sediment lead exceedances of the 128 ppm freshwater (Probable Effect Concentrations) sediment target.

Table G-16. Sediment Metals Data for the September 2010 Peck Road Park Lake Sampling Event

Organization	Date	Station ID	Lead (mg/kg))	Notes
EPA	09/30/2010	PRPL9	86.8	
EPA	09/30/2010	PRPL10	82.5	Average of duplicates

G.4.3 MONITORING RELATED TO ORGANOCHLORINE PESTICIDES AND PCBS IMPAIRMENTS

The extent of Organochlorine (OC) Pesticides and PCBs in Peck Road Park Lake was assessed through Regional Board sampling and their contracted study with UCLA. Peck Road Park Lake is specifically impaired by chlordane, DDT, dieldrin, and PCBs. The collected data for these contaminants are shown below for the water column, bottom lake sediments, suspended sediment in the water column, porewater, and suspended sediments in the porewater. Fish tissue level data from research by the Toxic Substances Monitoring Program (TSMP) (TSMP, 2009) and Surface Water Ambient Monitoring Program (SWAMP) (SWAMP, 2009; Davis et al., 2008) are also presented here and used in the OC Pesticides and PCBs TMDL.

G.4.3.1 Water Column Data Observed in Peck Road Park Lake

Water column samples were collected in the summer and fall of 2008 for the UCLA study and also on December 11, 2008 by the Regional Board. All pollutants were below detection limits (ND) or quantifiable/reportable levels (DNQ). PCB-31 was detected but not quantifiable at PRPL-10 in summer 2008 and also at PRPL-6W and PRPL-7W in fall 2008. Other PBCs that were DNQ were PCB-18 at PRPL-6W in summer 2008 and PCB-44, PCB-110, and PCB-153 at PRPL-7W in fall 2008. The results from the summer 2008 and fall 2008 samples are shown in Table G-17 and Table G-18 and results from the December 11, 2008 sampling are shown in Table G-19.

Table G-17. Results from Water Column Samples Collected at Peck Road Park Lake in Summer 2008

	PRPL-6W			PRPL-7W			PRPL-8W			PRPL-8W (dup)			PRPL-9			PRPL-10		
	DL	RL	Result	DL	RL	Result	DL	RL	Result	DL	RL	Result	DL	RL	Result	DL	RL	Result
Contaminant	(ng/L)																	
Chlordane-gamma	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
Chlordane-alpha	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
4,4'-DDE	3.16	31.58	ND	3.00	30.00	ND	3.09	30.93	ND	3.00	30.00	ND	3.00	30.00	ND	3.00	30.00	ND
4,4'-DDD	3.16	31.58	ND	3.00	30.00	ND	3.09	30.93	ND	3.00	30.00	ND	3.00	30.00	ND	3.00	30.00	ND
4,4'-DDT	3.16	31.58	ND	3.00	30.00	ND	3.09	30.93	ND	3.00	30.00	ND	3.00	30.00	ND	3.00	30.00	ND
Dieldrin	3.16	31.58	ND	3.00	30.00	ND	3.09	30.93	ND	3.00	30.00	ND	3.00	30.00	ND	3.00	30.00	ND
PCB 5	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 18	1.58	15.79	8.64*	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 31	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	4.07*
PCB 52	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 44	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 66	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 101	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 87	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 151	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 110	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 153	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 141	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 138	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 187	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 183	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 180	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 170	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 206	1.58	15.79	ND	1.50	15.00	ND	1.55	15.46	ND	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND

*Result was above the detection limit, but below the reporting limit.

Table G-18. Results from Water Column Samples Collected at Peck Road Park Lake in Fall 2008

Contaminant	PRPL-6W			PRPL-7W (duplicate)			PRPL-7W		
	DL	RL	Result	DL	RL	Result	DL	RL	Result
	(ng/L)								
Chlordane-gamma	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
Chlordane-alpha	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
4,4'-DDE	3.33	33.33	ND	3.33	33.33	ND	3.00	30.00	ND
4,4'-DDD	3.33	33.33	ND	3.33	33.33	ND	3.00	30.00	ND
4,4'-DDT	3.33	33.33	ND	3.33	33.33	ND	3.00	30.00	ND
Dieldrin	3.33	33.33	ND	3.33	33.33	ND	3.00	30.00	ND
PCB 5	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 18	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 31	1.67	16.67	4.31*	1.67	16.67	ND	1.50	15.00	7.76*
PCB 52	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 44	1.67	16.67	ND	1.67	16.67	1.93*	1.50	15.00	ND
PCB 66	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 101	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 87	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 151	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 110	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	3.02*
PCB 153	1.67	16.67	ND	1.67	16.67	2.88*	1.50	15.00	ND
PCB 141	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 138	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 187	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 183	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 180	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 170	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND
PCB 206	1.67	16.67	ND	1.67	16.67	ND	1.50	15.00	ND

*Result was above the detection limit, but below the reporting limit.

Table G-19. Results from Water Column Samples Collected at Peck Road Park Lake on December 11, 2008

Contaminant (ng/L)	PRPL-9	PRPL-10*	PRPL-16	PRPL-17	MDL
Chlordane-alpha	ND	ND	ND	ND	1.0
Chlordane-gamma	ND	ND	ND	ND	1.0
cis-Nonachlor	ND	ND	ND	ND	1.0
trans-Nonachlor	ND	ND	ND	ND	1.0
Oxychlordane	ND	ND	ND	ND	1.0
2-4’DDD	ND	ND	ND	ND	1.0
2-4’DDE	ND	ND	ND	ND	1.0
2-4’DDT	ND	ND	ND	ND	1.0
4-4’DDD	ND	ND	ND	ND	1.0
4-4’DDE	ND	ND	ND	ND	1.0
4-4’DDT	ND	ND	ND	ND	1.0
Dieldrin	ND	ND	ND	ND	1.0
PCB003	ND	ND	ND	ND	1.0
PCB008	ND	ND	ND	ND	1.0
PCB018	ND	ND	ND	ND	1.0
PCB028	ND	ND	ND	ND	1.0
PCB031	ND	ND	ND	ND	1.0
PCB033	ND	ND	ND	ND	1.0
PCB037	ND	ND	ND	ND	1.0
PCB044	ND	ND	ND	ND	1.0
PCB049	ND	ND	ND	ND	1.0
PCB052	ND	ND	ND	ND	1.0
PCB056/060	ND	ND	ND	ND	1.0
PCB066	ND	ND	ND	ND	1.0
PCB070	ND	ND	ND	ND	1.0
PCB074	ND	ND	ND	ND	1.0
PCB077	ND	ND	ND	ND	1.0
PCB081	ND	ND	ND	ND	1.0
PCB087	ND	ND	ND	ND	1.0
PCB095	ND	ND	ND	ND	1.0
PCB097	ND	ND	ND	ND	1.0
PCB099	ND	ND	ND	ND	1.0
PCB101	ND	ND	ND	ND	1.0
PCB105	ND	ND	ND	ND	1.0
PCB110	ND	ND	ND	ND	1.0
PCB114	ND	ND	ND	ND	1.0
PCB118	ND	ND	ND	ND	1.0

Contaminant (ng/L)	PRPL-9	PRPL-10*	PRPL-16	PRPL-17	MDL
PCB119	ND	ND	ND	ND	1.0
PCB123	ND	ND	ND	ND	1.0
PCB126	ND	ND	ND	ND	1.0
PCB128	ND	ND	ND	ND	1.0
PCB138	ND	ND	ND	ND	1.0
PCB141	ND	ND	ND	ND	1.0
PCB149	ND	ND	ND	ND	1.0
PCB151	ND	ND	ND	ND	1.0
PCB153	ND	ND	ND	ND	1.0
PCB156	ND	ND	ND	ND	1.0
PCB157	ND	ND	ND	ND	1.0
PCB158	ND	ND	ND	ND	1.0
PCB167	ND	ND	ND	ND	1.0
PCB168+132	ND	ND	ND	ND	1.0
PCB169	ND	ND	ND	ND	1.0
PCB170	ND	ND	ND	ND	1.0
PCB174	ND	ND	ND	ND	1.0
PCB177	ND	ND	ND	ND	1.0
PCB180	ND	ND	ND	ND	1.0
PCB183	ND	ND	ND	ND	1.0
PCB187	ND	ND	ND	ND	1.0
PCB189	ND	ND	ND	ND	1.0
PCB194	ND	ND	ND	ND	1.0
PCB195	ND	ND	ND	ND	1.0
PCB200	ND	ND	ND	ND	1.0
PCB201	ND	ND	ND	ND	1.0
PCB203	ND	ND	ND	ND	1.0
PCB206	ND	ND	ND	ND	1.0
PCB209	ND	ND	ND	ND	1.0

G.4.3.2 Porewater Data Observed in Peck Road Park Lake

Analysis of porewater and porewater suspended solids were performed for PRPL-6S, PRPL-7S, PRPL-9, and PRPL-10 in summer 2008. None of the contaminants were found in the porewater or associated solids at PRPL-6S. PCBs were detected below reporting limits (DNQ) in the water and suspended solids in porewater samples from PRPL-7S and PRPL-9. Three different PCB congeners were detected in the porewater suspended sediment from PRPL-10. No pollutants were detected in the porewater at PRPL-10. The analysis of porewater and suspended solids in porewater are shown in Table G-20, and Table G-21, respectively (see Stenstrom et al., 2009 for raw data).

Table G-20. Results from Porewater Samples Collected at Peck Road Park Lake in Summer 2008

Contaminant (ng/L)	PRPL-6S	PRPL-7S	PRPL-9	PRPL-10	MDL
Chlordane	ND	ND	ND	ND	15
DDT	ND	ND	ND	ND	30
Dieldrin	ND	ND	ND	ND	30
Total PCBs	ND	DNQ[1]	DNQ[2]	ND	15

[1] PCB-31 was detected below reporting limit (150 ng/L)

[2] PCB-5 was detected below reporting limit (150 ng/L).

Table G-21. Results of Porewater Suspended Sediments Samples Collected at Peck Road Park Lake in Summer 2008

Contaminant (µg/kg dry weight)	PRPL- 6S	PRPL-7S	PRPL-9	PRPL-10	MDL
Chlordane	ND	ND	ND	ND	2.26 – 9.25
DDT	ND	ND	ND	ND	4.51 – 18.50
Dieldrin	ND	ND	ND	ND	4.51 – 18.50
Total PCBs	ND	DNQ[1]	DNQ[2]	DNQ[3]	2.26 – 9.25

[1] PCB-52 was detected below reporting limit (22.55 µg/kg dry weight).

[2] PCB-87, PCB-153, PCB-180 were detected below reporting limit (66.03 µg/kg dry weight for each congener).

[3] PCB-160, PCB-145, PCB-187 were detected below reporting limit (59.72 µg/kg dry weight for each congener).

In fall 2008 samples from PRPL-7S, PRPL-9 and PRPL-10 were analyzed for contaminants in porewater. None of the organic chemicals of interest were detected in the samples. The porewater had insufficient TSS for analysis. The results from the fall 2008 analysis are shown in Table G-22.

Table G-22. Results of porewater sampling collected at Peck Road Park Lake in Fall 2008

Contaminant (ng/L)	PRPL-7S	PRPL-9	PRPL-10	MDL
Chlordane	ND	ND	ND	15
DDT	ND	ND	ND	30
Dieldrin	ND	ND	ND	30
Total PCBs	ND	ND	ND	15

G.4.3.3 Fish Tissue Data Observed in Peck Road Park Lake

Concentrations of Aroclor PCBs, chlordane, DDTs, dieldrin, and PCBs in fish tissue are shown for Peck Road Park Lake in Table G-23. Largemouth bass were the only fish species collected from Peck Road Park Lake. Aroclor PCBs were not detected in the fish samples. The average chlordane and DDT concentrations (17.2 ppb chlordane and 21.8 ppb DDTs) are both above OEHHA 2008 Fish Contaminant Goals (FCGs) for these contaminants (5.6 ppb for chlordane and 21 ppb for DDTs). The average PCBs concentration was 34.4 ppb, higher than the 3.6 ppb FCG for PCBs. The average dieldrin concentrations (1.06 ppb) are higher than the 0.45 ppb FCG for dieldrin.

Table G-23. Compiled Fish Tissue Analytical Data for Peck Road Park Lake

Program	Pollutant	Sample Date	Common Name	Concentration (ppb, wet wt)	Mean Length (mm)	Mean Weight (g)
TSMP	Aroclor PCBs	7/21/1986	Largemouth Bass	ND	332	788
TSMP	Aroclor PCBs	7/21/1986	Largemouth Bass	ND	175	90
TSMP	Aroclor PCBs	4/17/1991	Largemouth Bass	ND	126	29.6
TSMP	Aroclor PCBs	4/27/1992	Largemouth Bass	ND	160	68.5
SWAMP	Total PCBs	Summer 2007	Largemouth Bass	55.307	361.4	526.2
SWAMP	Total PCBs	Summer 2007	Largemouth Bass	22.651	360.4	499.2
SWAMP	Total PCBs	4/19/2010	Largemouth Bass	25.345	359.6	846
TSMP	Chlordane	7/21/1986	Largemouth Bass	42	332	788
TSMP	Chlordane	7/21/1986	Largemouth Bass	7	175	90
TSMP	Chlordane	4/17/1991	Largemouth Bass	14.1	126	29.6
TSMP	Chlordane	4/27/1992	Largemouth Bass	ND	160	68.5
SWAMP	Chlordane	Summer 2007	Largemouth Bass	19.212	361.4	526.2
SWAMP	Chlordane	Summer 2007	Largemouth Bass	8.637	360.4	499.2
SWAMP	Chlordane	4/19/2010	Largemouth Bass	12.465	359.6	846
TSMP	DDTs	7/21/1986	Largemouth Bass	35	332	788
TSMP	DDTs	7/21/1986	Largemouth Bass	18	175	90
TSMP	DDTs	4/17/1991	Largemouth Bass	39	126	29.6
TSMP	DDTs	4/27/1992	Largemouth Bass	14	160	68.5
SWAMP	DDTs	Summer 2007	Largemouth Bass	24.416	361.4	526.2
SWAMP	DDTs	Summer 2007	Largemouth Bass	8.982	360.4	499.2
SWAMP	DDTs	4/19/2010	Largemouth Bass	13.109	359.6	846
TSMP	Dieldrin	4/17/1991	Largemouth Bass	N/A	126	29.6
TSMP	Dieldrin	4/27/1992	Largemouth Bass	N/A	160	68.5
SWAMP	Dieldrin	Summer 2007	Largemouth Bass	0.965	361.4	526.2
SWAMP	Dieldrin	Summer 2007	Largemouth Bass	0.542	360.4	499.2
SWAMP	Dieldrin	4/19/2010	Largemouth Bass	1.66	359.6	846

ND = Non-detect

N/A = Not applicable

G.4.3.4 Sediment Data Observed in Peck Road Park Lake

Sediment samples for Peck Road Park Lake were collected by USEPA and the county of Los Angeles on November 16, 2009, and in the summer and fall of 2008 by UCLA. UCLA collected sediment samples at PRPL-6S, PRPL-7S, PRPL-9 and PRPL-10 in the summer 2008. Each sample also had laboratory duplicates and PRPL-7S had a field duplicate. At PRPL-6S (laboratory duplicate), DDE was detected at 20 µg/kg dry weight and PCB-180 was detected at 11 µg/kg dry weight. PCB-18 was detected at PRPL-7S (laboratory duplicate of the field duplicate sample) with a sediment concentration of 17 µg/kg dry weight. Chlordane-gamma was detected in PRPL-9S (laboratory duplicate) sediment samples at 7 µg/kg dry weight. The chlordane-gamma level at PRPL-9 was the only detected contaminant above the CBSQG for TEC and PEC levels. No contaminants were above reporting levels at PRPL-10S. The results of the sampling are shown in Table G-24.

The results of the UCLA fall 2008 sediment analysis are shown in Table G-25. Sediments from PRPL-7S, PRPL-9 and PRPL-10 were collected. Each sample also had laboratory duplicates and PRPL-9 had a field duplicate. PCB-31 at PRPL-7S and PCB-66 at PRPL-9 were the only pollutants detected above reportable levels. At PRPL-9 PCB-66 was detected at 8.60 µg/kg dry weight. At PRPL-7S, PCB-31 was quantified at 276.41 µg/kg dry weight. No contaminants were above reporting levels at PRPL-10.

Chlordane-gamma was detected at all four stations on November 16, 2009, in concentrations ranging from 1.0 to 6.6 µg/kg. Chlordane-alpha was detected at PRPL-9, PRPL-10 and PRPL-13 with concentrations ranging from 3.4 to 6.5 µg/kg. The DDT compound was not detected at any of the sites, but DDT-associated degradation products (DDD and DDE) were detected at three of the four stations (PRPL-9, PRPL-10 and PRPL-13). Several PCB congeners were also detected; however, dieldrin was not detected in any of the sediment samples. The raw data for these samples are reported in Table G-26. The detection limit for all samples was 1 µg/kg dry sediment.

Table G-24. Results from Sediment Samples Collected at Peck Road Park Lake in Summer 2008

Contaminant	PRPL-6S			PRPL-6S (lab dup)			PRPL-7S			PRPL-7S (lab dup)			PRPL-7SB (field dup)			PRPL-7SB (lab dup of field dup)			PRPL-9			PRPL-9 (lab dup)			PRPL-10			PRPL-10 (lab dup)		
	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.
	µg/kg dry weight																													
Chlordane-gamma	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	7.14	0.72	7.20	ND	0.65	6.48	ND
Chlordane-alpha	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
4,4'-DDE	0.76	7.62	ND	0.72	7.20	20.07	0.72	7.17	ND	0.81	8.07	ND	0.69	6.87	ND	0.83	8.30	ND	0.96	9.57	ND	0.99	9.89	ND	1.44	14.40	ND	1.30	12.95	ND
4,4'-DDD	0.76	7.62	ND	0.72	7.20	ND	0.72	7.17	ND	0.81	8.07	ND	0.69	6.87	ND	0.83	8.30	ND	0.96	9.57	ND	0.99	9.89	ND	1.44	14.40	ND	1.30	12.95	ND
4,4'-DDT	0.76	7.62	ND	0.72	7.20	ND	0.72	7.17	ND	0.81	8.07	ND	0.69	6.87	0.90*	0.83	8.30	ND	0.96	9.57	ND	0.99	9.89	ND	1.44	14.40	ND	1.30	12.95	ND
Dieldrin	0.76	7.62	ND	0.72	7.20	ND	0.72	7.17	ND	0.81	8.07	ND	0.69	6.87	ND	0.83	8.30	ND	0.96	9.57	ND	0.99	9.89	ND	1.44	14.40	ND	1.30	12.95	ND
PCB 5	0.38	3.81	0.40*	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 18	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	17.09	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 31	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 52	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 44	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 66	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	4.19*	0.65	6.48	ND
PCB 101	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 87	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 151	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 110	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	0.24*	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 153	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 141	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	0.85*	0.72	7.20	ND	0.65	6.48	ND
PCB 138	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	0.87	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 187	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 183	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 180	0.38	3.81	ND	0.36	3.60	11.38	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 170	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND
PCB 206	0.38	3.81	ND	0.36	3.60	ND	0.36	3.58	ND	0.40	4.03	ND	0.34	3.44	ND	0.41	4.15	ND	0.48	4.78	ND	0.49	4.95	ND	0.72	7.20	ND	0.65	6.48	ND

*Results were above the detection level, but below the reporting level.

Table G-25. Results from Sediment Samples Collected at Peck Road Park Lake in Fall 2008

Contaminant	PRPL-7S			PRPL-7S (lab dup)			PRPL-9			PRPL-9 (lab dup)			PRPL-9 (field dup)			PRPL-9 (lab dup of field dup)			PRPL-10			PRPL-10 (lab dup)		
	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.
	µg/kg dry weight																							
Chlordane-gamma	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
Chlordane-alpha	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
4,4'-DDE	1.18	11.82	ND	1.18	11.82	ND	1.30	13.01	ND	1.40	14.02	ND	1.09	10.90	ND	1.31	13.06	ND	0.97	9.69	ND	0.98	9.82	ND
4,4'-DDD	1.18	11.82	ND	1.18	11.82	ND	1.30	13.01	ND	1.40	14.02	ND	1.09	10.90	ND	1.31	13.06	4.26*	0.97	9.69	ND	0.98	9.82	ND
4,4'-DDT	1.18	11.82	ND	1.18	11.82	ND	1.30	13.01	ND	1.40	14.02	ND	1.09	10.90	ND	1.31	13.06	3.81*	0.97	9.69	ND	0.98	9.82	ND
Dieldrin	1.18	11.82	ND	1.18	11.82	ND	1.30	13.01	ND	1.40	14.02	ND	1.09	10.90	ND	1.31	13.06	ND	0.97	9.69	ND	0.98	9.82	ND
PCB 5	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 18	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 31	0.59	5.91	ND	0.59	5.91	276.41	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 52	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 44	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 66	0.59	5.91	2.38*	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	8.60	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 101	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 87	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 151	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 110	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 153	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 141	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	3.55*	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 138	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 187	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 183	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 180	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	1.83*	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 170	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND
PCB 206	0.59	5.91	ND	0.59	5.91	ND	0.65	6.50	ND	0.70	7.01	ND	0.54	5.45	ND	0.65	6.53	ND	0.48	4.84	ND	0.49	4.91	ND

*Results were above the detection level but below the reporting level.

Table G-26. Results from Sediment Samples Collected at Peck Road Park Lake on November 16, 2009

Contaminant (µg/kg dry weight)	PRPL-9			PRPL-10	PRPL-12	PRPL-13	MDL
	Results	Field Dup	Field and Lab Dup				
Chlordane-gamma	5.3	6.2	6.6	5.6	1	3.1	1
Chlordane-alpha	5.4	6.4	6.5	5.6	ND	3.4	1
cis-Nonachlor	2.5	2.7	3.4	3	ND	1.5	1
trans-Nonachlor	6.3	6.3	6.4	4.1	ND	3.2	1
Oxychlordane	ND	ND	ND	ND	ND	ND	1
2,4` - DDD	ND	ND	ND	ND	ND	ND	1
2,4` - DDE	ND	ND	ND	ND	ND	ND	1
2,4` - DDT	ND	ND	ND	ND	ND	ND	1
4,4` - DDD	3	ND	ND	4.1	ND	2.8	1
4,4` - DDE	7.3	9.7	8.4	7.7	ND	8.2	1
4,4` - DDT	ND	ND	ND	ND	ND	ND	1
Dieldrin	ND	ND	ND	ND	ND	ND	1
PCB037	ND	2.1	ND	ND	ND	ND	1
PCB074	2.9	2.3	2	ND	ND	ND	1
PCB095	1.2	1.7	1.4	1.6	ND	ND	1
PCB099	ND	ND	ND	1.0	ND	ND	1
PCB101	1.4	2.2	1.1	1.4	ND	1.0	1
PCB110	1.8	ND	1.1	ND	ND	1.2	1
PCB118	ND	1.6	1.4	ND	ND	ND	1
PCB138	5.1	3.1	ND	ND	ND	ND	1
PCB149	1.3	2	2.2	1.6	ND	1.3	1
PCB151	ND	1	1	ND	ND	ND	1
PCB153	2.1	ND	ND	1.8	ND	1.6	1
PCB174	1.8	2	2.5	1.1	ND	ND	1
PCB177	ND	1.4	1	ND	ND	ND	1
PCB180	1.1	1.6	2.5	1.8	ND	ND	1
PCB187	1.8	2.4	1.6	1.1	ND	ND	1
PCB194	ND	ND	ND	ND	1.0	ND	1
PCB206	ND	2.3	ND	1.3	ND	ND	1

G.4.3.5 Suspended Sediment Data Observed in Peck Road Park Lake

Suspended solids (TSS) from Peck Road Park Lake were collected in the summer and fall of 2008. Summer samples were taken at PRPL-6S, PRPL-6W, PRPL-7S, PRPL-9 and PRPL-10. PRPL-6W was the only sample that had enough suspended matter to perform the analysis. None of the pesticides were detected in the sample. PCB-110 was detected, but not quantifiable. The results of the summer sampling are shown in Table G-27.

Table G-27. Results from Suspended Sediment Samples Collected at Peck Road Park Lake in Summer 2008

Contaminant	PRPL-6W		
	DL	RL	Result
	µg/kg dry suspended solids		
Chlordane-gamma	5.14	51.35	ND
Chlordane-alpha	5.14	51.35	ND
4,4'-DDE	10.27	102.71	ND
Dieldrin	10.27	102.71	ND
4,4'-DDD	10.27	102.71	ND
4,4'-DDT	10.27	102.71	ND
PCB 5	5.14	51.35	ND
PCB 18	5.14	51.35	ND
PCB 31	5.14	51.35	ND
PCB 52	5.14	51.35	ND
PCB 44	5.14	51.35	ND
PCB 66	5.14	51.35	ND
PCB 101	5.14	51.35	ND
PCB 87	5.14	51.35	ND
PCB 151	5.14	51.35	ND
PCB 110	5.14	51.35	27.15*
PCB 153	5.14	51.35	ND
PCB 141	5.14	51.35	ND
PCB 138	5.14	51.35	ND
PCB 187	5.14	51.35	ND
PCB 183	5.14	51.35	ND
PCB 180	5.14	51.35	ND
PCB 170	5.14	51.35	ND
PCB 206	5.14	51.35	ND

*Result was above detection limit, but below reporting limits.
Note: Samples were collected at PRPL-7S, PRPL-9 and PRPL-10, but had insufficient sample for analysis.

In fall 2008, TSS from PRPL-6W and PRPL-7W were analyzed for the contaminants. The only chemicals detected were PCB-138 at PRPL-6W and PCB-180 at PRPL-7S, both below reportable limits. These results are shown in Table G-28.

Table G-28. Results from Suspended Sediment Samples Collected at Peck Road Park Lake in Fall 2008

Contaminant	PRPL-6W			PRPL-7W			PRPL-7W (field duplicate)		
	DL	RL	Result	DL	RL	Result	DL	RL	Result
	µg/kg dry suspended solids								
Chlordane-gamma	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
Chlordane-alpha	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
4,4'-DDE	4.73	47.26	ND	40.82	408.16	ND	28.85	288.46	ND
Dieldrin	4.73	47.26	ND	40.82	408.16	ND	28.85	288.46	ND
4,4'-DDD	4.73	47.26	ND	40.82	408.16	ND	28.85	288.46	ND
4,4'-DDT	4.73	47.26	ND	40.82	408.16	ND	28.85	288.46	ND
PCB 5	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 18	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 31	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 52	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 44	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 66	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 101	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 87	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 151	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 110	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 153	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 141	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 138	2.36	23.63	3.56*	20.41	204.08	ND	14.42	144.23	ND
PCB 187	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 183	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 180	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	48.23*
PCB 170	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND
PCB 206	2.36	23.63	ND	20.41	204.08	ND	14.42	144.23	ND

*Results are above the detection limits but below the reporting limits.

G.5 Monitoring Data for Lincoln Park Lake

Monitoring data relevant to the impairments of Lincoln Park Lake are available from 1992, 1993, 2008, and 2009. Figure G-12 shows the historical and recent monitoring locations for Lincoln Park Lake.

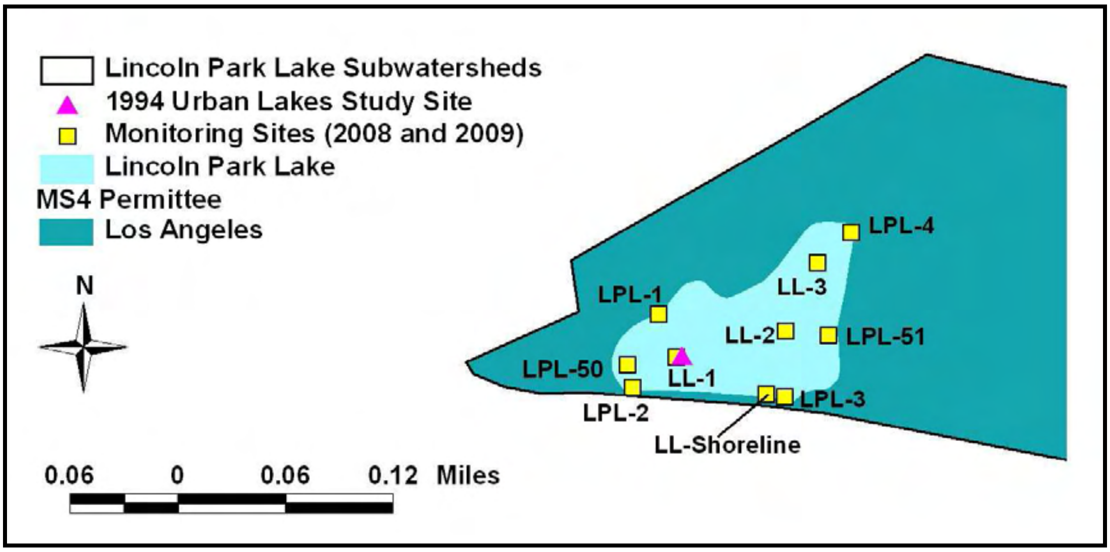


Figure G-12. Lincoln Park Lake Monitoring Sites

G.5.1 MONITORING RELATED TO NUTRIENT IMPAIRMENTS

Water quality sampling was conducted in Lincoln Park Lake in 1992 and 1993 for the Urban Lakes Study (Table G-29) from a station located in the western half of the lake (UC Riverside, 1994) (pink triangle, Figure G-12). Sampling occurred over 2 meters of depth on 12 sampling days. TKN ranged from 0.3 mg/L to 2.8 mg/L; eight of 28 samples for ammonia were less than detection and the maximum observed ammonia concentration was 1.1 mg/L. All nitrite samples were less than the reporting limit, and 17 of 28 nitrate samples were less than the reporting limit. The maximum nitrate concentration was 0.3 mg/L. Orthophosphate concentrations in 1992 were less than or equivalent to the reporting limit, while concentrations in 1993 ranged from 0.2 mg/L to 0.3 mg/L. Total phosphorus was also higher in 1993 with concentrations ranging from 0.2 mg/L to 0.5 mg/L compared to concentrations in 1992 of which nine samples were less than the reporting limit and the maximum observed concentration was 0.2 mg/L. pH measurements ranged from 7.7 to 9.1. TOC ranged from 6.0 mg/L to 14.5 mg/L, with one outlier of 132 mg/L. The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from <1 µg/L to 97 µg/L with an average of 33 µg/L.

Table G-29. Lincoln Park Lake 1992/1993 Monitoring Data for Nutrients

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
7/13/1992	0	1.4	0.4	<0.01	<0.01	0.1	<0.01	7.8	9	672
	2	1.5	0.3	<0.01	<0.01	<0.01	<0.01	7.8	9.1	653
7/13/1992	0	1.5	0.3	<0.01	<0.01	<0.01	<0.01	7.9	9.4	671
	1.5	1.4	0.3	<0.01	<0.01	<0.01	<0.01	7.9	9.1	668
7/13/1992	0	1.5	0.3	<0.01	<0.01	<0.01	<0.01	7.9	8.9	667
	1.5	1.4	0.3	<0.01	<0.01	<0.01	<0.01	8.1	8.6	649
8/19/1992	0	2.8	0.3	<0.01	0.1	<0.01	0.1	8.4	8.5	701
	2	2.2	0.3	<0.01	<0.01	<0.01	0.1	8.5	8.6	697
9/17/1992	0	1.9	0.2	<0.01	0.1	<0.01	0.2	8.2	8.4	631
	2	1.6	0.1	<0.01	<0.01	<0.01	0.1	8.4	7.9	629
10/15/1992	0	1.4	0.3	<0.01	<0.01	<0.01	<0.01	8.2	6.6	645
	2	1.1	0.4	<0.01	0.1	<0.01	<0.01	8.2	6.6	638
11/5/1992	0	1.9	0.7	<0.01	0.2	<0.01	0.1	8.1	6.3	602
	1.7	1.7	0.8	<0.01	0.2	<0.01	0.1	8.2	7.1	581
12/8/1992	0	1.7	0.5	<0.01	0.3	<0.01	<0.01	7.7	6	575
	1.5	1.9	0.5	<0.01	0.3	<0.01	0.1	7.7	6.1	568
1/14/1993	0	2.7	0.6	<0.01	0.3	0.2	0.4	7.8	7	419
	2	2.3	0.7	<0.01	0.3	0.2	0.3	8	6.5	446
2/2/1993	0	2.4	1.1	<0.01	0.3	0.2	0.2	8.1	6.2	539
	2	2.1	1.1	<0.01	0.3	0.2	0.2	8.1	6.1	598
3/24/1993	0	1.9	<0.01	<0.01	<0.01	0.3	0.5	8.8	9.5	634
	2	1.8	<0.01	<0.01	<0.01	0.3	0.4	8.8	9.1	617
4/6/1993	0	1.6	<0.01	<0.01	<0.01	0.3	0.4	8.9	7.9	594
	1.5	1.6	<0.01	<0.01	<0.01	0.3	0.5	8.9	8.6	604
5/3/1993	0	0.3	<0.01	<0.01	<0.01	0.3	0.4	9.1	11.1	640
	2	0.7	<0.01	<0.01	<0.01	0.3	0.4	9.0	11.2	650
6/7/1993	0	1.6	<0.01	<0.01	<0.01	0.3	0.2	8.8	132	674
	2	1.9	<0.01	<0.01	<0.01	0.3	0.3	8.7	14.5	674

There are no stations in Lincoln Park Lake or its drainage area listed in the Regional Board Water Quality Assessment Database. The Water Quality Assessment Report, however, states that DO was partially supporting the aquatic life use with 78 measurements of dissolved oxygen ranging from 0.1 mg/L to

13.7 mg/L. Ammonia was listed as not supporting the aquatic life or contact recreation uses. Twenty-eight ammonium samples were collected ranging from non-detect to 1.14 mg/L, the upper end of this range is below the acute target, but above the chronic target (for assessment purposes, we are assuming that the analysis methodology converted all ammonia to ammonium). Raw data are not available to assess location, date, time, depth, temperature, or pH with regard to these samples.

The Regional Board sampled water quality at four stations around the shoreline of Lincoln Park Lake in 2008. All samples were collected from the edge of the lake using a 6-ft extension pole. Samples were collected approximately 4 inches below the water surface.

During the October 29, 2008 sampling event, concentrations of total phosphate and ammonia at each station were less than the reporting limits of 0.5 mg/L and 0.1 mg/L, respectively. TKN at each site ranged from 1.49 mg/L to 2.32 mg/L. Total dissolved solids ranged from 847 mg/L to 868 mg/L. Suspended solids ranged from less than the reporting limit of 10 mg/L to 12 mg/L. Chlorophyll *a* ranged from 44 µg/L to 123 µg/L.

During the November 6, 2008 sampling event, concentrations of orthophosphate, nitrate, and nitrite at each site were less than the reporting limits of 0.4 mg/L, 0.1 mg/L, and 0.1 mg/L, respectively. No other parameters were measured during this event.

Field data for these two sampling events are summarized in Table G-30.

Table G-30. Field Data for 2008 Monitoring Events at Lincoln Park Lake

Site	Date	Time	Temperature	pH	Total Depth (m)
LPL-1	10/29/2008	15:15	22	8.9	0.6
	11/6/2008	9:34	17	8.5	0.6
LPL-2	10/29/2008	14:05	20	8.9	0.9
	11/6/2008	10:05	17	8.5	0.5
LPL-3	10/29/2008	15:45	20	9.0	0.4
	11/6/2008	10:30	17	8.7	0.3
LPL-4	10/29/2008	16:50	22	9.0	0.5
	11/6/2008	10:45	17	8.5	0.4

In 2009, the City of Los Angeles Bureau of Sanitation, Watershed Protection Division began collecting water quality samples at three locations in Lincoln Park Lake. Table G-31 summarizes the analyses for samples collected on February 18 through July 28, 2009. The nitrate in the lake at all locations and sampling times was below the detection level. After February, all nitrite levels were also below detection level; and after March, all ammonia samples were also below detection. The fraction of organic nitrogen was between 0.8 and 1.8 throughout the sampling period. The chlorophyll *a* was lowest in February; at LL-3 it was 13 µg/L. The maximum amount of chlorophyll (47 µg/L) was sampled in July at LL-2. Suspended solids were also higher in the summer months. In July, the average TSS was 18.2 mg/L and only 11.2 mg/L in February.

Table G-31. 2009 City of Los Angeles Bureau of Sanitation 2009 Lincoln Park Lake Monitoring Data

Date	Station	NH_3-N (mg/L)	Org N (mg/L)	NO_3-N (mg/L)	NO_2-N (mg/L)	TP (mg/L)	Chlorophyll *a* (µg/L)	BOD (mg/L)	TSS (mg/L)
2/18/2009	LL-1	0.24	1.3	<0.02	0.11	0.14	35	6	10.5
	LL-2	0.24	1.0	<0.02	0.11	0.09	44	5	12.0
	LL-3	0.27	1.7	<0.02	0.13	0.09	13	4	11.0
3/26/2009	LL-1	0.09	1.3	<0.02	<0.02	0.13	26	4	14.0
	LL-2	0.08	1.3	<0.02	<0.02	0.12	32	5	13.3
	LL-3	0.10	1.8	<0.02	<0.02	0.11	22	5	10.0
4/27/2009	LL-1	<0.05	1.5	<0.02	<0.02	0.18	24	6	12.0
	LL-2	<0.05	1.0	<0.02	<0.02	0.17	28	3	12.0
	LL-3	<0.05	1.3	<0.02	<0.02	0.18	23	4	13.2
5/28/2009	LL-1	<0.05	1.0	<0.02	<0.02	0.13	26	<3	13.0
	LL-2	<0.05	1.2	<0.02	<0.02	0.16	21	<3	16.0
	LL-3	<0.05	1.2	<0.02	<0.02	0.16	25	<3	8.0
7/28/2009	LL-1	<0.05	0.9	<0.02	<0.02	0.15	40	4	16.0
	LL-2	<0.05	1.4	<0.02	<0.02	0.16	47	4	19.5
	LL-3	<0.05	0.8	<0.02	<0.02	0.14	44	4	19.0

Sonde data were also collected by the City of Los Angeles Bureau of Sanitation. Table G-32 presents the mean daily values measured at stations LL-1, LL-2, and LL-3 for temperature, specific conductivity, dissolved oxygen, and pH at three depths. For a given collection day, there was little variability between the stations or depths for temperature, specific conductivity, dissolved oxygen, or pH, indicating absence of significant stratification.

Table G-32. Mean Values of Sonde Data Collected in Lincoln Park Lake at Stations 1, 2, and 3

	Temperature (℃)			Specific Conductivity (mS/cm)			Dissolved Oxygen (mg/L)			pH		
Date	Surface (< 0.5 m)	0.5 - 1.0 m	1.0 - 1.5 m	Surface (< 0.5 m)	0.5 - 1.0 m	1.0 - 1.5 m	Surface (< 0.5 m)	0.5 - 1.0 m	1.0 - 1.5 m	Surface (< 0.5 m)	0.5 - 1.0 m	1.0 - 1.5 m
7/28/2008	27.30	27.47	27.48	1.20	1.19	1.19	8.35	7.69	8.05	8.65	8.67	8.71
2/18/2009	12.77	12.35	11.96	1.04	1.04	1.04	8.79	8.74	8.42	8.24	8.22	8.17
3/26/2009	17.90	17.74	N/A	1.09	1.09	N/A	8.61	8.48	N/A	8.34	8.31	N/A
4/27/2009	20.54	20.55	20.76	1.14	1.14	1.14	7.42	7.05	6.49	8.36	8.34	8.31
5/28/2009	23.67	23.77	23.76	1.21	1.21	1.21	7.94	7.74	7.75	8.43	8.45	8.46

N/A = no data available

On March 10, 2009, the Regional Board and USEPA sampled water quality in Lincoln Park Lake. Two sites were accessed by wading in from boat access ramps located on either side of the lake. Samples were collected from 1 foot at each site and the total depth at each site was approximately 2.2 feet. Table G-33 summarizes the nutrient and chlorophyll *a* measurements for these two stations. Ammonia concentrations were relatively high and ranged from 1.2 mg/L to 1.26 mg/L; TKN was 2.2 mg/L at both stations. Nitrate and nitrite were both relatively low with concentrations averaging 0.07 mg/L and 0.04 mg/L, respectively. Orthophosphate concentrations were approximately 0.08 mg/L and total phosphorus concentrations were approximately 0.126 mg/L. Chlorophyll *a* concentrations at both sites were less than the detection limit of 1 µg/L.

Table G-33. In-lake and Shoreline Water Column Measurements for Lincoln Park Lake

Station Label	NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Ortho-phos-phate (mg/L)	TP (mg/L)	TDS (mg/L)	TSS (mg/L)	Chloro-phyll *a* (µg/L)
LPL-50	1.20	2.2	0.07	0.04	0.0762	0.125	703	4	<1 µg/L
LPL-50 (duplicate)	1.24	NA	0.07	0.04	0.0835	0.125	NA	NA	<1 µg/L
LPL-51	1.26	2.2	0.06	0.04	0.0802	0.127	664	5.2	<1 µg/L

Profile data collected in Lincoln Park Lake on March 10, 2009 are summarized in Table G-34. DO concentrations in the lake generally ranged from 5.9 mg/L to 6.2 mg/L with one reading of 7.0 mg/L from a surface sample. pH ranged from 6.7 to 7.0. Profile depths listed in the field notes (ranging from surface to 1.3 meters) were multiplied by the ratio of total depth reported in the field notes to the depth measured on the probe cable at each monitoring station because the probe was drifting and indicating depths greater than actual (Anna Sofranko, USEPA Region IX, personal communication, May 12, 2009).

Table G-34. Field Data for the March 10, 2009 Lincoln Park Lake Sampling Event

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Secchi Depth (m)	Total Depth (m)
LPL-50	13:45	Surface	18.9	7.0	7.6	Greater than total depth	0.69
		0.26	18.8	6.8	6.0		
		0.53	18.6	6.8	6.2		
		~0.69 (bottom)	17.2	6.7	6.0		
LPL-51	15:10	Surface	19.1	6.7	6.2	Greater than total depth	0.66
		0.26	19.2	6.7	6.1		
		0.53	19.2	6.7	6.2		
		~0.66 (bottom)	19.2	6.7	5.9		

On August 4, 2009, USEPA and the Regional Board collected additional nutrient samples from Lincoln Park Lake. Ammonia, TKN, nitrite, and nitrate were all less than the detection limits of 0.03 mg-N/L, 0.456 mg-N/L, 0.01 mg-N/L, and 0.01 mg-N/L, respectively. Orthophosphate was less than the detection limit (0.0075 mg-P/L), and total phosphorus was 0.182 mg-P/L. The chlorophyll *a* concentration was 27.3 µg/L. The potable water input was also sampled during this event. Ammonia and nitrate were both 0.33 mg-N/L; nitrite was 0.03 mg-N/L. TKN measured 0.531 mg-N/L. Orthophosphate and total phosphorus were 0.017 mg-P/L and 0.118 mg-P/L, respectively.

Profile data associated with this event were collected at LL-1, shown in Table G-35. The DO concentration ranged from 8.32 to 10.19 mg/L. The total depth at this station was 1.7 meters, and the Secchi depth was 0.66 meters. The pH was approximately 9.1 at all depths.

Table G-35. Field Data for the August 4, 2009 Lincoln Park Lake Sampling Event

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Secchi Depth (m)	Total Depth (m)
LL-1	14:40	0.1	28.7	9.1	9.14	0.66	1.70
		0.52	28.4	9.1	9.51		
		1.01	26.7	9.2	10.19		
		1.5	26.1	9.0	8.32		

Field data were collected for the potable water source during the August sampling event. After purging the line for approximately ten minutes, the pH was 7.82, the DO was 6.62 mg/L, and the temperature was 26.8 ºC.

Additional supplemental water quality samples were collected from Lincoln Park Lake. Table G-36 presents the chloride, sulfate, total alkalinity, total dissolved solids, and total organic carbon data measured in the lake. Temperature and pH measurements reported in the field notes are also shown in this table. Both temperature and pH significantly increased between March and August. The average temperature in March was 19.0 °C and the temperature in August was 28.7 °C. The pH ranged from 6.7 in the winter to 9.1 in the summer. Chloride, sulfate, TDS, DOC, and TOC all significantly increased in August. The alkalinity in the summer was 61 mg/L lower than the level measured in March.

Table G-36. Supplemental Water Quality Monitoring for Lincoln Park Lake

Date	Location	Time	Temper-ature (℃)	pH	Chloride (mg/L)	Sulfate (mg/L)	Total Alka-linity (mg/L)	Hardness (mg/L as $CaCO_3$)	TDS (mg/L)	DOC (mg/L)	TOC (mg/L)
3/10/2009	LPL-50	13:40	18.9	7.0	97	247	142	262	703	5.6	6.1
	LPL-51	14:30	19.1	6.7	99	250	142	257	664	5.4	5.3
8/4/2009	LL-1[1]	14:05	28.7	9.1	134	305	81	281	826	9.8	10.5

[1] These data were averages of laboratory replicates, except for temperature and pH data (which were surface samples collected at 14:40).

The city of Los Angeles provided water quality monitoring data for the Glendale Water Reclamation Plant, which may be used to supplement lake levels and irrigate parkland at Lincoln Park in the future.

Table G-37 summarizes the average water quality for this source based on monthly averages reported for 2008 and 2009.

Table G-37. Average Water Quality for the Glendale Water Reclamation Plant

NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Orthophosphate (mg/L)	TP (mg/L)
1.3	3.17	5.64	0.009	1.76	1.93

G.5.2 MONITORING RELATED TO LEAD IMPAIRMENT

In 1996, Lincoln Park Lake was deemed impaired by lead. Monitoring data for cadmium, copper, lead, and zinc are presented in this section. Lincoln Park Lake is not listed for cadmium, copper, or zinc, but those data are presented here for completeness because other waterbodies in the region are affected by some of these contaminants.

Metals data collected at Lincoln Park Lake, as part of the 1992-1993 Urban Lakes Study (UC Riverside, 1994), are shown in Table G-38. Specifically, samples were collected from a station located in the western half of the lake (UC Riverside, 1994) (pink triangle, Figure G-12) and included dissolved copper and dissolved lead. Dissolved copper samples were collected throughout the water column at depths from the surface to two meters. The range of the 28 dissolved copper samples was between less than 10 µg/L and 81 µg/L. Similarly, dissolved lead samples were also collected throughout the water column at depths from the surface to two meters. The 28 samples collected ranged in concentration from less than 1 µg/L to 94 µg/L.

The Regional Board completed its Water Quality Assessment and Documentation Report for waterbodies in the Los Angeles Region in 1996 (LARWQCB, 1996). The summary table for Lincoln Park Lake states that lead was not supporting the assessed uses: 28 measurements reported a maximum lead concentration of 94 µg/L, a maximum copper concentration of 61 µg/L, a maximum cadmium concentration of 1.6 µg/L, and a maximum zinc concentration of 13 µg/L (raw data were not provided, but it is assumed that most of these samples are associated with the Urban Lake Study [UC Riverside, 1994]).

Unfortunately, metals levels were analyzed at relatively high detection limits compared to current detection limits; dissolved copper minimum detection 10 µg/L while dissolved lead was 1 µg/L. No hardness data were collected as part of the Urban Lakes Study, thus it cannot be compared to the hardness-based water quality objectives.

Table G-38. Lincoln Park Lake 1992/1993 Monitoring Data for Metals

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
7/13/1992	0	16	<1
	2	<10	<1
7/13/1992	0	13	<1
	1.5	<10	N/A
7/13/1992	0	20	21
	1.5	11	<1
8/19/1992	0	22	<1

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
	2	23	<1
9/17/1992	0	17	N/A
	2	16	1
10/15/1992	0	<10	<1
	2	<10	<1
11/5/1992	0	<10	2
	1.7	<10	2
12/8/1992	0	<10	7
	1.5	<10	7
1/14/1993	0	11	9
	2	<10	1
2/2/1993	0	<10	<1
	2	<10	<1
3/24/1993	0	<10	<1
	2	<10	<1
4/6/1993	0	81	2
	1.5	47	<1
5/3/1993	0	<10	<1
	2	<10	3
6/7/1993	0	17	94
	2	16	33

Table G-39 presents 40 additional metal samples that were collected by the USEPA, Regional Board, and/or the city of Los Angeles between October 2008 and December 2010 at Lincoln Park Lake. Samples were collected at locations LPL 1/2/3/4, LPL 50/51, LL-1, LL-2, LL-3, LPL-2/4, and LL-Shoreline. Sites were analyzed for dissolved cadmium, copper, lead, and/or zinc (only lead data are reported for the city of Los Angeles samples).

Detection limits were lower than the 1992-1993 study with a cadmium detection limit of 0.2 µg/L, dissolved copper detection limit of 0.4 µg/L, dissolved lead detection limit of 0.05 µg/L, and dissolved zinc detection limit of 0.2 µg/L. All dissolved cadmium concentrations were < 0.2 µg/L to 0.4 µg/L; copper concentrations were between 2.1 µg/L and 8.12 µg/L; lead concentrations ranged from <0.05 µg/L to 2.0 µg/L; and zinc concentrations were 0.3 µg/L to 1.3 µg/L. Metal toxicity is affected by hardness; therefore, each sample was also analyzed for hardness. The 2008-2010 sampling resulted in a hardness range of 166 mg/L to 356 mg/L. Since dissolved results pertain to the applicable standard and recent data more closely represents current conditions, data in Table G-39 were weighted more heavily in the assessment.

Table G-39. Metals Data for the 2008-2010 Lincoln Park Lake Sampling Events

Organi-zation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
RB	10/29/2008	LPL 1/2/3/4	247.8	0.4	4.4	<0.1	0.5	average of replicates; average of sites 1-4
RB	3/10/2009	LPL 50/51	258.9	<0.2	2.1	0.2	1.3	average of replicates & duplicates; average of sites 50 and 51
City LA	2/18/2009	LL-1/2/3	292.0	N/A	N/A	<2	N/A	average of sites 1, 2, and 3
City LA	3/26/2009	LL-1/2/3	257.0	N/A	N/A	0.1	N/A	average of sites 1, 2, and 3
City LA	4/27/2009	LL-1/2/3	311.3	N/A	N/A	0.2	N/A	average of sites 1, 2, and 3
City LA	5/28/2009	LL-1/2/3	316.7	N/A	N/A	0.1	N/A	average of sites 1, 2, and 3
City LA	7/28/2009	LL-1/2/3	279.3	N/A	N/A	0.4	N/A	average of sites 1, 2, and 3
RB/EPA	8/4/2009	LL 1	281.0	<0.2	4	0.3	0.8	average of replicates
RB/EPA	8/4/2009	LPL 2 / 4	282.3	<0.2	2.1	0.1	0.3	average of shore sites 2 and 6
City LA	8/28/2009	LL-Shoreline	324	N/A	N/A	0.1	N/A	
City LA	9/4/2009	LL-Shoreline	312	N/A	N/A	<0.1	N/A	
City LA	9/11/2009	LL-Shoreline	328	N/A	N/A	<0.1	N/A	
City LA	9/18/2009	LL-Shoreline	320	N/A	N/A	0.2	N/A	
City LA	9/25/2009	LL-Shoreline	331	N/A	N/A	0.2	N/A	
City LA	10/2/2009	LL-Shoreline	315	N/A	N/A	<0.1	N/A	

Organi-zation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
City LA	10/9/2009	LL-Shoreline	316	N/A	N/A	0.2	N/A	
City LA	10/16/2009	LL-Shoreline	356	N/A	N/A	0.2	N/A	
City LA	10/23/2009	LL-Shoreline	331	N/A	N/A	0.3	N/A	
City LA	10/30/2009	LL-Shoreline	332	N/A	N/A	0.3	N/A	
City LA	11/6/2009	LL-Shoreline	330	N/A	N/A	0.3	N/A	
City LA	11/13/2009	LL-Shoreline	349	N/A	N/A	0.2	N/A	
City LA	11/20/2009	LL-Shoreline	307	N/A	N/A	0.4	N/A	
City LA	12/4/2009	LL-Shoreline	323	N/A	N/A	0.3	N/A	
City LA	12/11/2009	LL-Shoreline	321	N/A	N/A	0.4	N/A	
City LA	12/18/2009	LL-Shoreline	318	N/A	N/A	0.2	N/A	
City LA	1/8/2010	LL-Shoreline	333	N/A	N/A	0.5	N/A	
City LA	1/15/2010	LL-Shoreline	315	N/A	N/A	1.6	N/A	
City LA	1/22/2010	LL-Shoreline	271	N/A	N/A	0.4	N/A	
City LA	2/5/2010	LL-Shoreline	286	N/A	N/A	0.3	N/A	
City LA	2/12/2010	LL-Shoreline	265	N/A	N/A	0.2	N/A	
City LA	2/19/2010	LL-Shoreline	236	N/A	N/A	<0.1	N/A	
City LA	2/26/2010	LL-Shoreline	260	N/A	N/A	<0.1	N/A	
EPA / RB	9/28/2010	LL-1	166	<0.4	8.12	<0.1	<0.5	
EPA / RB	9/28/2010	LPL-4	167	<0.2	3.73	<0.05	<0.1	
City LA	10/8/2010	LL-Shoreline	256	N/A	N/A	0.17	N/A	
City LA	10/22/2010	LL-Shoreline	269	N/A	N/A	0.17	N/A	

Organi-zation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
City LA	11/5/2010	LL-Shoreline	253	N/A	N/A	0.19	N/A	
City LA	11/19/2010	LL-Shoreline	266	N/A	N/A	0.19	N/A	
City LA	12/3/2010	LL-Shoreline	253	N/A	N/A	0.34	N/A	
City LA	12/17/2010	LL-Shoreline	238	N/A	N/A	0.16	N/A	

N/A = No data available
RB = Regional Board
EPA = USEPA
City LA = City of Los Angeles

USEPA also collected one sediment sample in September 2010 to further evaluate lake conditions. Table G-40 summarizes the lead concentrations measured in these samples. There were zero sediment lead exceedances of the 128 ppm freshwater (Probable Effect Concentrations) sediment target.

Table G-40. Sediment Metals Data for the September 2010 Lincoln Park Lake Sampling Event

Organi zation	Date	Station ID	Lead (mg/kg)	Notes
EPA	09/28/2010	LL1	105	Average of duplicates

(This page left intentionally blank.)

G.6 Monitoring Data for Echo Park Lake

Echo Park Lake has been monitored more frequently than many other lakes addressed in this memo. Sampling has occurred in 1992, 1993, and 2003 through 2009. In addition, fish tissue data are available for 1987 to 2007. Figure G-13 shows the location of historic and recent monitoring locations in Echo Park Lake.

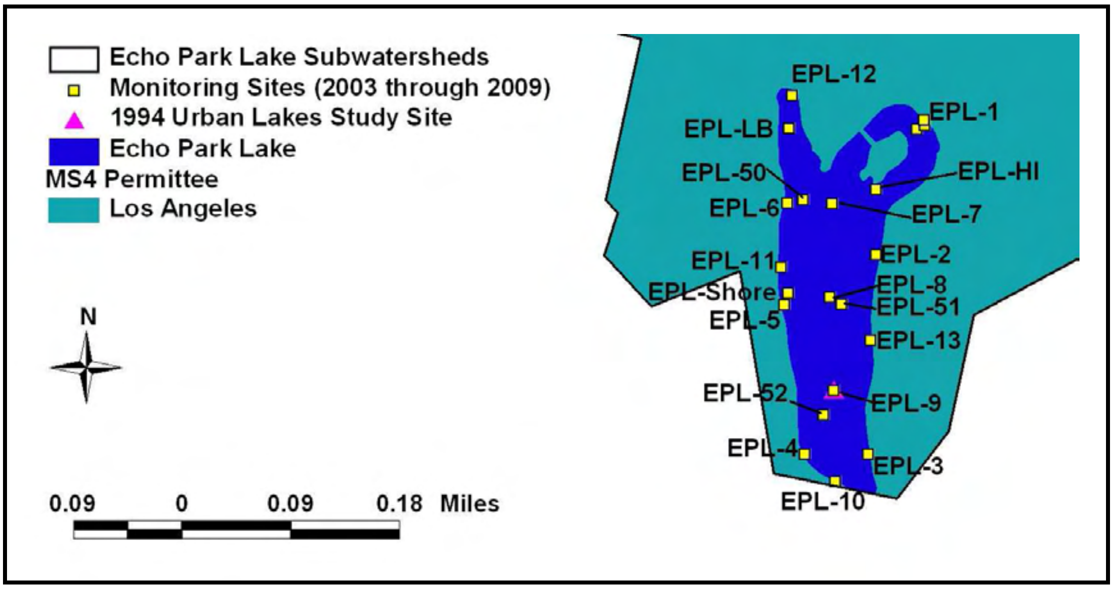


Figure G-13. Historic and Recent Sampling Sites at Echo Park Lake

G.6.1 MONITORING RELATED TO NUTRIENT IMPAIRMENTS

Results of the 1992/1993 Urban Lakes Study sampling are summarized in Table G-41. Sampling occurred near the center of the lower half of the lake (UC Riverside, 1994) (pink triangle, Figure G-13). TKN concentrations during this sampling period ranged from 0.9 mg/L to 1.9 mg/L. Ammonium concentrations were less than the reporting limit for 22 of 31 samples, and the maximum observed ammonium concentration was 0.7 mg/L. Nitrite concentrations were less than the reporting limit in all samples; 24 of 31 nitrate samples were less than the reporting limit. The maximum observed nitrate concentration was 0.2 mg/L. Orthophosphate concentrations were generally less than or equivalent to the reporting limit with some observations of 0.2 mg/L. Total phosphorus concentrations ranged from less than the reporting limit to 0.3 mg/L. pH measurements ranged from 7.7 to 9.4, and TOC ranged from 4.8 mg/L to 7.6 mg/L. The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from 6 µg/L to 66 µg/L with an average of 24 µg/L.

Table G-41. Echo Park Lake 1992/1993 Monitoring Data

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
7/14/1992	0	1	0.3	<0.1	<0.1	0.1	<0.1	8.8	6.9	333
	1.5	1	<0.1	<0.1	<0.1	0.1	<0.1	8.9	7.1	321
7/14/1992	0	1.1	<0.1	<0.1	<0.1	0.1	<0.1	9.0	7.4	319
	1.5	1.2	<0.1	<0.1	<0.1	0.1	<0.1	9.0	7.4	317
7/14/1992	0	1.4	0.1	<0.1	<0.1	0.1	0.1	9.0	7.6	316
	1.3	1.4	<0.1	<0.1	<0.1	0.1	0.1	8.9	6.8	322
8/13/1992	0	0.9	<0.1	<0.1	<0.1	<0.1	<0.1	9.3	6.5	322
	1.5	1.1	<0.1	<0.1	<0.1	<0.1	0.1	9.3	6.7	323
8/13/1992	0	0.9	<0.1	<0.1	<0.1	<0.1	0.1	9.3	6.4	322
	2	1.2	<0.1	<0.1	<0.1	<0.1	0.1	9.3	6.6	319
	0	1.1	<0.1	<0.1	<0.1	<0.1	0.1	9.1	6.5	334
9/17/1992	0	1.1	<0.1	<0.1	<0.1	<0.1	0.2	8.3	6.9	364
	1.7	1.1	<0.1	<0.1	<0.1	<0.1	0.1	8.2	7.2	359
10/15/1992	0	0.9	<0.1	<0.1	<0.1	<0.1	<0.1	8.1	6.7	447
	1.7	1.3	0.1	<0.1	<0.1	<0.1	<0.1	8.1	6.8	450
11/5/1992	0	1.9	<0.1	<0.1	<0.1	<0.1	0.2	8.7	6.8	411
	1.5	1.9	<0.1	<0.1	<0.1	<0.1	0.2	8.7	7.2	428
12/8/1992	0	1.3	0.2	<0.1	0.2	<0.1	<0.1	7.7	6.1	443
	1.5	1.3	0.2	<0.1	0.1	<0.1	<0.1	7.8	5.9	453
1/12/1993	0	1.8	0.7	<0.1	0.2	0.2	<0.1	7.8	5.5	350
	1.5	1.7	0.7	<0.1	0.1	0.2	0.1	7.8	5.4	357
2/2/1993	0	1.7	0.5	<0.1	0.2		<0.1	8.5	4.8	323
	1.5	1.6	0.6	<0.1	0.2	<0.1	<0.1	8.5	4.8	299
3/17/1993	0	0.9	<0.1	<0.1	0.1	<0.1	<0.1	8.8	5.4	252
	1.5	1	<0.1	<0.1	<0.1	<0.1	<0.1	8.8	5.2	251
4/7/1993	0	1.1	<0.1	<0.1	<0.1	<0.1	0.1	9.4	5.4	249
	1.5	1.1	<0.1	<0.1	<0.1	<0.1	0.1	9.4	4.8	251
5/3/1993	0	1.2	<0.1	<0.1	<0.1	0.2	0.3	8.9	5.3	352
	2	1.2	<0.1	<0.1	<0.1	0.2	0.3	8.9	5	321
6/8/1993	0	1.1	<0.1	<0.1	<0.1	0.2	0.1	8.9	7.1	386
	1.5	1.1	<0.1	<0.1	<0.1	0.2	0.1	8.6	7.1	411

There were no stations in Echo Park Lake or its drainage area in the Regional Board Water Quality Assessment Database. The Water Quality Assessment Report, however, states pH was not supporting the contact recreation use and partially supporting the aquatic life use: 69 measurements of pH were collected which ranged from 7.0 to 9.4. Thirty-one ammonium samples were collected with values ranging from non-detect to 0.71 mg/L, the upper end of this range is below the acute target, but above the chronic target (for assessment purposes, we are assuming that the analysis methodology converted all ammonia to ammonium); ammonia was listed as not supporting the aquatic life and contact recreation uses. Raw data are not available to assess location, date, time, depth, temperature, or pH with regards to these samples. Odor and algae were both listed as not supporting the contact and non-contact recreation uses. Eutrophication was listed as not supporting the aquatic life use.

In 2003, the City of Los Angeles Bureau of Sanitation, Watershed Protection Division began collecting water quality samples from Echo Park Lake. Stations EPL-1 through EPL-6 are perimeter stations that were only sampled for bacterial parameters and EPL-7, EPL-8, and EPL-9 are mid-lake stations. Table G-42 lists the nutrient data collected through February 2010 for the three in-lake stations. Of the 84 samples collected during this period, 38 were non-detect for ammonia; the maximum ammonia concentration was 0.93 mg/L. Organic nitrogen concentrations ranged from 0.28 mg/L to 3.14 mg/L. Thirty-five nitrate samples were non detect, and the maximum observed concentration was 1.0 mg/L. Fifty-five of the nitrite samples were non detect; the other two samples had concentrations of 0.02 and 0.09 mg/L. Total nitrogen concentrations, calculated from the sum of ammonia, organic nitrogen, nitrate, and nitrite, ranged from 0.28 mg/L to 3.48 mg/L. Total phosphate measurements generally ranged from 0.06 mg/L to 0.51 mg/L with three measurements less than detection. Biochemical oxygen demand (BOD) ranged from 4 mg/L to 18 mg/L with 25 measurements less than the detection limit; the length and type of the BOD test was not specified in the data set received. TSS measurements ranged from 3 mg/L to 31 mg/L. No chlorophyll *a* data were reported.

Table G-42. City of Los Angeles Bureau of Sanitation Echo Park Lake Monitoring Data

Date	Station	NH_3-N (mg/L)	Org N (mg/L)	NO_3-N (mg/L)	NO_2-N (mg/L)	TN calc. (mg/L)	Total Phosphate (mg/L)	BOD (mg/L)	TSS (mg/L)
5/14/2003	EPL-7	<0.10	0.3	<0.02	<0.02	0.28	0.12	<4	8
	EPL-8	<0.10	0.3	<0.02	<0.02	0.28	0.14	<4	11
	EPL-9	<0.10	0.3	<0.02	<0.02	0.28	0.15	4	16
8/12/2003	EPL-7	0.34	3.1	<0.02	<0.10	3.48	0.28	17	29
	EPL-8	0.56	2.0	<0.02	<0.10	2.58	0.08	18	25
	EPL-9	0.34	1.9	<0.02	<0.10	2.24	0.23	16	25
11/20/2003	EPL-7	0.30	1.0	0.16	<0.02	1.46	0.09	<4	5
	EPL-8	0.30	1.3	0.15	<0.02	1.75	0.08	<4	5
	EPL-9	0.60	1.0	0.16	<0.02	1.76	0.07	<4	4
2/18/2004	EPL-7	<0.10	< 0.1	1.00	0.09	1.09	0.08	10	6
	EPL-8	<0.10	0.6	0.08	<0.02	0.68	0.08	9	3
	EPL-9	<0.10	1.2	0.10	<0.02	1.30	0.16	9	5
5/18/2004	EPL-7	0.10	1.1	<0.02	<0.02	1.20	<0.05	<4	8
	EPL-8	0.10	1.0	<0.02	<0.02	1.10	<0.05	<4	11
	EPL-9	0.20	1.0	<0.02	<0.02	1.20	<0.05	<4	8

Date	Station	NH_3-N (mg/L)	Org N (mg/L)	NO_3-N (mg/L)	NO_2-N (mg/L)	TN calc. (mg/L)	Total Phosphate (mg/L)	BOD (mg/L)	TSS (mg/L)
8/25/2004	EPL-7	0.10	1.2	<0.02	<0.02	1.30	0.09	5	19
	EPL-8	<0.10	1.1	<0.02	<0.02	1.10	0.07	6	14
	EPL-9	<0.10	1.0	<0.02	<0.02	1.00	0.08	6	13
11/17/2004	EPL-7	0.50	1.3	0.17	<0.02	1.97	0.48	16	16
	EPL-8	0.57	1.0	0.18	<0.02	1.75	0.40	7	7
	EPL-9	0.49	1.3	0.18	<0.02	1.97	0.51	8	8
2/17/2005	EPL-7	<0.05	1.5	0.18	<0.02	1.65	0.13	4	16
	EPL-8	<0.05	0.8	0.06	<0.02	0.85	0.12	<4	9
	EPL-9	<0.05	0.7	0.06	0.02	0.80	0.09	<4	8
5/19/2005	EPL-7	<0.05	1.0	0.07	<0.02	1.07	0.15	<4	31
	EPL-8	<0.05	0.8	0.02	<0.02	0.82	0.11	<4	18
	EPL-9	<0.05	1.0	<0.02	<0.02	1.00	0.13	<4	21
8/18/2005	EPL-7	<0.05	1.4	<0.02	<0.02	1.40	0.06	7	23
	EPL-8	<0.05	1.2	<0.02	<0.02	1.20	0.06	7	20
	EPL-9	<0.05	1.2	<0.02	<0.02	1.20	0.07	6	20
11/17/2005	EPL-7	0.21	0.7	<0.02	<0.02	0.91	0.25	4	16
	EPL-8	0.24	2.2	<0.02	<0.02	2.44	0.26	4	20
	EPL-9	0.69	1.4	<0.02	<0.02	2.09	0.26	4	16
2/9/2006	EPL-7	<0.05	1.0	0.04	<0.02	1.04	0.16	AE[1]	AE[1]
	EPL-8	<0.05	1.0	<0.02	<0.02	1.00	0.21	AE[1]	AE[1]
	EPL-9	<0.05	1.1	<0.02	<0.02	1.10	0.25	AE[1]	AE[1]
5/11/2006	EPL-7	0.45	0.3	<0.02	<0.02	0.75	0.07	<3	18
	EPL-8	0.30	0.4	<0.02	<0.02	0.70	0.18	<3	21
	EPL-9	0.35	0.3	<0.02	<0.02	0.65	0.15	<3	16
8/17/2006	EPL-7	0.09	1.2	<0.02	<0.02	1.29	0.15	8	28
	EPL-8	0.10	0.9	<0.02	<0.02	1.00	0.16	4	10
	EPL-9	0.17	1.1	<0.02	<0.02	1.27	0.19	6	16
11/16/2006	EPL-7	0.10	0.9	0.07	<0.02	1.07	0.07	<3	18
	EPL-8	0.14	1.8	0.06	<0.02	2.00	0.08	<3	13
	EPL-9	0.19	1.5	0.04	<0.02	1.73	0.08	<3	14
2/8/2007	EPL-7	0.59	0.7	0.06	<0.02	1.35	0.21	<3	8
	EPL-8	0.67	0.4	0.06	<0.02	1.13	0.21	<3	9
	EPL-9	0.93	0.4	0.06	<0.02	1.39	0.22	<3	9

Date	Station	NH_3-N (mg/L)	Org N (mg/L)	NO_3-N (mg/L)	NO_2-N (mg/L)	TN calc. (mg/L)	Total Phosphate (mg/L)	BOD (mg/L)	TSS (mg/L)
5/17/2007	EPL-7	0.10	1.2	<0.20	<0.20	1.30	0.14	<3	24
	EPL-8	0.09	1.4	<0.20	<0.20	1.49	0.31	<3	27
	EPL-9	0.08	1.6	<0.20	<0.20	1.68	0.24	<3	22
8/16/2007	EPL-7	<0.05	0.9	<0.02	<0.02	0.90	0.11	4	19
	EPL-8	<0.05	1.0	<0.02	<0.02	1.00	0.12	5	11
	EPL-9	<0.05	0.8	0.02	<0.02	0.82	0.12	5	12
11/7/2007	EPL-7	<0.05	0.9	<0.02	<0.02	1.00	0.13	5	12
	EPL-8	0.10	0.9	<0.02	<0.02	1.00	0.14	5	12
	EPL-9	0.10	0.7	<0.02	<0.02	0.80	0.12	4	12
2/14/2008	EPL-7	0.18	0.2	0.13	< 0.02	0.51	0.11	< 3	8
	EPL-8	0.28	0.5	0.12	< 0.02	0.90	0.12	< 3	11
	EPL-9	0.27	0.3	0.12	< 0.02	0.69	0.12	< 3	10
5/8/2008	EPL-7	< 0.05	1.1	0.08	< 0.02	1.18	0.13	5	18
	EPL-8	0.09	1.1	0.08	< 0.02	1.27	0.14	6	16
	EPL-9	0.17	0.6	0.09	< 0.02	0.86	0.12	< 3	14
8/7/2008	EPL-7	< 0.05	0.8	< 0.02	< 0.02	0.80	0.11	< 3	19
	EPL-8	< 0.05	0.9	< 0.02	< 0.02	0.90	0.13	< 3	19
	EPL-9	< 0.05	1.0	< 0.02	< 0.02	1.00	0.12	3	16
11/20/2008	EPL-7	0.31	1.0	0.38	0.03	1.72	0.08	3	10
	EPL-8	0.32	1.1	0.33	< 0.02	1.75	0.10	3	14
	EPL-9	0.28	1.0	0.31	< 0.02	1.59	< 0.05	3	12
2/19/2009	EPL-7	< 0.05	0.5	0.28	0.10	0.88	0.13	< 3	4
	EPL-8	< 0.05	0.4	0.29	0.11	0.80	0.10	< 3	4
	EPL-9	0.05	0.5	0.30	0.10	0.95	0.11	< 3	6
5/21/2009	EPL-7	< 0.05	1	< 0.02	< 0.02	1.00	0.15	3	17
	EPL-8	< 0.05	0.8	< 0.02	< 0.02	0.80	0.15	4	19
	EPL-9	< 0.05	1.1	< 0.02	< 0.02	1.10	0.16	5	15
8/18/2009	EPL-7	< 0.05	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	EPL-8	< 0.05	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	EPL-9	< 0.05	n/a	n/a	n/a	n/a	n/a	n/a	n/a
12/22/2009	EPL-7	< 0.05	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	EPL-8	< 0.05	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	EPL-9	< 0.05	n/a	n/a	n/a	n/a	n/a	n/a	n/a

Date	Station	NH_3-N (mg/L)	Org N (mg/L)	NO_3-N (mg/L)	NO_2-N (mg/L)	TN calc. (mg/L)	Total Phosphate (mg/L)	BOD (mg/L)	TSS (mg/L)
2/16/2010	EPL-7	0.10	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	EPL-8	0.11	n/a	n/a	n/a	n/a	n/a	n/a	n/a
	EPL-9	0.10	n/a	n/a	n/a	n/a	n/a	n/a	n/a

[1] AE indicates analyst error: no value reported.

Sonde data were also collected by the City of Los Angeles Bureau of Sanitation. Table G-43 presents the mean daily values measured at stations EPL-7, EPL-8, and EPL-9 for temperature, specific conductivity, dissolved oxygen, and pH at four depths. For a given collection day, there was little variability between the stations or depths for temperature, specific conductivity, dissolved oxygen, or pH, indicating absence of significant stratification.

Table G-43. Mean Values of Sonde Data Collected in Echo Park Lake at Stations 7, 8, and 9

	Temperature (℃)				Specific Conductivity (mS/cm)				Dissolved Oxygen (mg/L)				pH			
Date	**Surface**	**0.5 - 1.0 m**	**1.0 - 1.5m**	**>1.5m**	**Surface**	**0.5 - 1.0 m**	**1.0 - 1.5m**	**>1.5m**	**Surface**	**0.5 - 1.0 m**	**1.0 - 1.5 m**	**>1.5m**	**Surface**	**0.5 - 1.0 m**	**1.0 - 1.5 m**	**>1.5m**
8/25/2004	23.8	23.4	23.1	22.9	0.750	0.749	0.750	0.750	8.9	8.8	8.6	7.9	8.5	8.4	8.4	7.9
11/17/2004	15.9	15.7	15.5	15.3	0.656	0.653	0.654	0.656	9.1	8.9	8.9	8.8	7.9	7.9	7.9	7.9
2/17/2005	16.0	15.9	15.7	15.5	0.417	0.417	0.418	0.418	10.5	10.5	10.4	9.8	8.4	8.4	8.3	8.3
5/19/2005	22.5	22.0	21.9	21.9	0.448	0.448	0.447	0.451	9.8	9.8	9.7	9.6	8.6	8.6	8.6	8.5
8/18/2005	23.6	23.6	23.3	23.5	0.460	0.459	0.459	0.467	9.3	9.4	9.4	8.8	8.9	8.9	8.9	8.1
11/17/2005	16.2	16.0	15.9	15.8	0.537	0.537	0.538	0.538	11.9	12.0	11.9	11.3	8.6	8.6	8.5	8.3
2/9/2006	13.7	13.7	13.7	13.8	0.540	0.541	0.541	0.541	10.9	10.9	10.8	10.7	8.4	8.4	8.4	8.3
5/11/2006	20.5	20.6	20.6	20.3	0.499	0.498	0.499	0.500	10.5	10.6	10.5	9.8	8.6	8.6	8.6	8.5
8/17/2006	24.9	24.8	24.6	24.5	0.485	0.485	0.486	0.492	7.7	7.5	6.7	6.1	8.4	8.3	8.2	8.2
11/16/2006	16.5	16.4	16.2	16.1	0.591	0.590	0.591	0.592	10.3	10.2	10.0	9.7	8.2	8.2	8.2	8.2
2/8/2007	13.6	13.6	13.5	13.5	0.589	0.588	0.588	0.589	8.7	9.0	9.0	8.9	8.0	8.1	8.1	8.1
5/17/2007	20.8	20.7	20.6	20.4	0.638	0.635	0.633	0.633	8.8	8.8	8.7	8.2	8.4	8.3	8.3	8.3
8/16/2007	25.8	25.7	25.5	25.3	0.671	0.669	0.671	0.672	9.3	9.6	9.4	8.7	9.0	9.0	9.0	8.9
11/7/2007	17.6	17.6	17.5	17.5	0.724	0.724	0.724	0.724	8.5	8.5	8.4	8.2	8.39	8.39	8.37	8.34
2/14/2008	14.49	14.48	14.45	14.40	0.64	0.64	0.64	0.64	9.60	9.58	9.60	9.61	8.26	8.28	8.29	8.30
5/8/2008	18.83	18.80	18.70	18.97	0.78	0.78	0.78	0.79	7.65	7.64	7.52	6.41	8.09	8.17	8.20	8.26
8/7/2008	26.66	26.44	26.33	26.06	0.90	0.90	0.89	0.89	6.58	6.24	6.00	5.71	8.21	8.18	8.14	8.11
11/20/2008	16.69	16.46	16.36	16.34	1.04	1.04	1.04	1.04	9.90	9.82	9.61	9.46	8.09	8.07	8.04	8.02
2/19/2009	12.44	12.30	12.17	12.13	0.80	0.80	0.80	0.80	11.03	11.02	10.98	10.99	8.31	8.30	8.29	8.29
5/21/2009	23.91	23.71	23.62	23.51	0.85	0.85	0.85	0.85	8.60	8.44	8.29	8.14	8.28	8.28	8.27	8.26

In 2008, the Regional Board sampled nine locations in Echo Park Lake. Site location descriptions are listed in Table G-44. As the lake is relatively shallow and well mixed by wind action and aerators, the sampling team collected analytical samples from the lake surface only. To avoid confusion with the City of Los Angeles Bureau of Sanitation numbering scheme, all sites were assigned an alternate label.

Table G-44. Site Locations for the 2008 Echo Park Lake Monitoring Event

Sampling Event	Regional Board Site Number	Project Site	Alternate Label
June	1	Below City of LA storm drain	EPL-1
	2	Below County of LA storm drain	EPL-6
	3	Lake mid-point	EPL-8
	4	Lotus Beds	EPL-LB
	5	Hydroponic Island	EPL-HI
December	1	Shoreline sample at northwest segment of lake	EPL-12
	2	Shoreline sample at western side of lake	EPL-11
	3	Shoreline sample on southern edge of lake	EPL-10
	4	Shoreline sample on eastern side of lake	EPL-13
	5	Shoreline sample near City of LA storm drain	EPL-1 (shoreline)

Ammonia concentrations in Echo Park Lake were fairly similar at all three sampled locations (Sites EPL-8, EPL-LB, and EPL-HI) on June 25, 2008 and ranged from 0.131 mg/L to 0.136 mg/L (Table G-45). TKN at the lake midpoint and near the hydroponic island ranged from 1.38 mg/L to 1.49 mg/L; the concentration was higher in the lotus beds at 4.72 mg/L. Concentrations of nitrate, nitrite, orthophosphate, and total phosphate were all less than the reporting limits of 0.1, 0.1, 0.4, and 0.5 mg/L, respectively. Total dissolved solids ranged from 565 mg/L to 651 mg/L, and suspended solids ranged from 11.2 mg/L to 96 mg/L.

Table G-45. Analytical Data for the June 25, 2008 Echo Park Lake Sampling Event

Alternate Station Label	NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Ortho-phosphate (mg/L)	Total Phosphate (mg/L)	Total Dissolved Solids (mg/L)	Suspended Solids (mg/L)
EPL-8	0.136	1.41	<0.1	<0.1	<0.4	<0.5	603	13.2
EPL-LP	0.131	4.72	<0.1	<0.1	<0.4	<0.5	651	96.0
EPL-HI	0.133	1.38	<0.1	<0.1	<0.4	<0.5	565	11.6
EPL-8 (dup.)	0.129	1.49	<0.1	<0.1	<0.4	<0.5	582	11.2

Field data were collected at five sites in Echo Park Lake by the Regional Board (Table G-46). Temperatures at sites EPL-6, EPL-8, and EPL-HI ranged from 25.46 ºC to 27.76 ºC. Temperatures on the north end of the lake were higher (28.95 and 32.0 ºC) but were also collected later in the day. Dissolved oxygen concentrations ranged from 4.95 mg/L to 9.82 mg/L, and pH ranged from 8.21 to 8.56 (note that the pH meter was not producing calibration results within the acceptable range). Electrical conductivity did not vary significantly at any location and ranged from 0.680 mS/cm to 0.688 mS/cm. The Secchi depth readings at sites EPL-6, EPL-8, and EPL-HI ranged from 0.66 m to 0.68 m. Chlorophyll *a* samples collected at depths less than the Secchi depth at each site ranged from 10.9 µg/L to 15 µg/L, with the exception of Site 4 in the lotus beds where the concentration was 26.7 µg/L. At depths greater than the Secchi depth at each site, chlorophyll *a* concentrations were generally higher with concentrations ranging from 16.1 µg/L to 53.6 µg/L. These higher numbers may reflect chlorophyll *a* contained in decaying algae that has settled to the bottom of the lake. A description of the methodology or equipment used to measure chlorophyll *a* concentrations in the field was not provided.

Table G-46. Field Data for the June 25, 2008 Echo Park Lake Sampling Event

Site	Time	Depth (m)	Temp (C)	DO (mg/L)	EC (mS/cm)	pH[1]	Chl a (ug/L)	Secchi Depth (m)
EPL-8	Begin time: 10:06	Surface	26.63	6.57	0.685	8.27	12.7	0.66
		0.5	25.80	6.51	0.685	8.30	15.0	
		1.0	25.54	6.26	0.686	8.26	15.8	
		1.5	25.46	4.95	0.686	8.21	53.6	
		2	25.52	5.33	0.684	8.22	0.8	
EPL-HI	Begin time: 11:05	Surface	26.69	8.11	0.686	8.46	11.9	0.68
		0.5	26.48	7.44	0.685	8.46	11.8	
		1.0 (bottom)	25.83	6.68	0.687	8.43	16.1	
EPL-6	Begin time: 12:23	Surface	27.76	8.3	0.685	8.53	10.9	0.68
		0.5	26.94	7.6	0.686	8.52	12.0	
		1.0 (bottom)	26.76	7.46	0.685	8.50	20.6	
EPL-LB	14:10	Surface	32.0	6.25	0.688	8.29	26.7	NA
EPL-1	14:49	Surface	28.95	9.82	0.680	8.56	14.0	NA

[1] pH calibration was outside of accepted range. Data should not be used quantitatively.

Samples were also collected on December 18, 2008 from five shoreline locations at a depth of approximately 4 inches (Table G-47). Ammonia ranged from 0.206 mg/L to 0.344 mg/L. TKN ranged from 1.1 mg/L to 1.55 mg/L with one measurement near the lotus pond that was less than the reporting limit of 1 mg/L. Nitrate ranged from 0.215 mg/L to 0.325 mg/L. All samples of nitrite, orthophosphate, and total phosphate were less than the reporting limits of 0.1 mg/L, 0.4 mg/L, and 0.5 mg/L, respectively. Total dissolved solids ranged from 549 mg/L to 576 mg/L. All measurements of suspended solids were less than the reporting limit of 10 mg/L except at EPL-1. Chlorophyll *a* ranged from 8.5 µg/L to 20.2 µg/L.

Table G-47. Analytical Data for the December 18, 2008 Echo Park Lake Sampling Event

Alternate Station Label	NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Ortho-phos-phate (mg/L)	Total phos-phate (mg/L)	Total Dissolved Solids (mg/L)	Suspended Solids (mg/L)	Chloro-phyll *a* (µg/L)
EPL-12	0.344	1.55	0.215	<0.1	<0.1	<0.4	549	46.5	20.2
EPL-11	0.209	1.1	0.325	<0.1	<0.1	<0.4	576	<RL	10.4
EPL-10	0.239	1.1	0.3	<0.1	<0.1	<0.4	576	<RL	15.1
EPL-13	0.215	<RL	0.317	<0.1	<0.1	<0.4	576	<RL	8.9
EPL-1 (shoreline)	0.234	1.24	0.309	<0.1	<0.1	<0.4	567	<RL	10.7
EPL-11 (dup.)	0.206	1.16	0.312	<0.1	<0.1	<0.4	576	<RL	8.5

Field data from the December 2008 sampling event are summarized in Table G-48. Temperature in the lake ranged from 8 ºC to 10.5 ºC; pH ranged from 7.7 to 8.1.

Table G-48. Field Data for the December 18, 2008 Echo Park Lake Sampling Event

Site	Time	Temp (C)	pH	Total Depth (m)
EPL-12	9:00	8	7.7	0.2
EPL-11	10:15	10	8.0	0.7
EPL-10	11:20	10	8.0	0.9
EPL-13	12:10	10.5	8.0	0.6
EPL-1 (shoreline)	13:50	10.5	8.1	Not reported

On March 10, 2009, USEPA and the Regional Board sampled Echo Park Lake at three locations (Table G-49). Samples were collected at Site EPL-50 at 9:50 from a depth of 0.61 m. Site EPL-51 was also sampled from a depth of 0.61 m at 10:30. Site EPL-52 was sampled at 11:00 from a depth of 0.46 m. Ammonia concentrations ranged from 0.04 mg/L to 0.06 mg/L, and TKN ranged from 0.7 mg/L to 1.3 mg/L. Nitrate was approximately 0.15 mg/L at each station and nitrite was less than detection. Orthophosphate was less than detection at each station and total phosphorus generally ranged from 0.033 mg/L to 0.071 mg/L. The total phosphorus measured at EPL-52 was 0.762 mg/L, though the field duplicate had a value of 0.071 mg/L. Chlorophyll *a* measurements in the lake ranged from 14.2 µg/L to 15.2 µg/L.

Sites EPL-51 and 52 and the potable water input (PW) were sampled again on August 4th, 2009 (also in Table G-49). Site-51 was sampled at 8:15 and had a total depth of 1.7 meters and a Secchi depth of 0.3 meters. Site-52 was sampled at 9:00, had a depth of 1.8 meters, and a Secchi depth of 0.6 meters. All nitrogen parameters (ammonia, TKN, nitrate, and nitrite) were below detection limits at both in-lake sites. Total phosphorus was 0.196 mg/L at EPL-51 and 0.195 mg/L at EPL-52. The orthophosphate concentrations were 0.0850 and 0.0917 at sites EPL-51 and EPL-52, respectively. The TSS average at the stations was 15.2 mg/L and the chlorophyll *a* average was 15.3µg/L. The TDS was 505 mg/L at EPL-51 and 494 mg/L at EPL-52. The lab noted that these samples were analyzed after the allowable holding time limit, but the data are included here because the TDS amounts were in the expected range and the extended holding time does not appear to have greatly affected these measurements.

Table G-49. In-lake Water Column Measurements for Echo Park Lake

Date	Station Label	NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Ortho-phos-phate (mg/L)	TP (mg/L)	TDS (mg/L)	TSS (mg/L)	Chl *a* (µg/L)
3/10/2009	EPL-50	0.04	0.97	0.15	<0.01	<0.008	0.033	427	4.6	15.2
	EPL-51	0.05	1.30	0.14	<0.01	<0.008	0.046	438	7.0	14.2
	EPL-52	0.06	0.70	0.15	<0.01	<0.008	0.762	442	7.3	14.6
	EPL-52 (duplicate)	0.06	NA	NA	NA	NA	0.071	NA	NA	NA
8/4/2009	EPL-51	<0.03	<0.456	<0.01	<0.01	0.0850	0.196	505	18.0	15.5
	EPL-52	<0.03	<0.456	<0.01	<0.01	0.0917	0.195	494	12.3	15.0
	EPL-PW	<0.03	<0.456	0.9	<0.01	0.0202	0.122	348	<0.5	NA

Additional data taken during the sampling events on March 10 and August 4, 2009, are shown in Table G-50. The chloride concentrations increased significantly between the winter and summer events. The winter average of chloride was 76.2 mg/L and the summer concentration was 92.9 mg/L. The sulfate concentrations were lower in the summer, at an average of 91 mg/L compared to the winter average of 131 mg/L. There were not significant changes between the March and August measurements of alkalinity, DOC, and TOC.

Table G-50. Supplemental Water Quality Monitoring for In-lake Samples in the Echo Park Lake

Date	Location	Time	Chloride (mg/L)	Sulfate (mg/L)	Total Alkalinity (mg/L)	Hardness (mg/L as $CaCO_3$)	DOC (mg/L)	TOC (mg/L)
3/10/2009	EPL-50	9:50	75.8	130	133	181	3.2	2.6
	EPL-51	10:30	76.7	132	126	178	3.1	4.8
	EPL-52	11:00	76.1	130	126	178	3.0	3.4
8/4/2009	EPL-51	8:15	92.7	91	136	191.4	3.6	5.3
	EPL-52	9:00	93.1	91	140	187.6	3.6	5.5
	EPL-PW	10:40	66.4	65.4	141	128.6	0.95	0.85

Profile data collected in Echo Park Lake are summarized in Table G-51. Based on this data the lake appears well mixed both vertically and spatially. DO concentrations in the lake generally ranged from 7.0 mg/L to 8.6 mg/L with one reading of 10.0 mg/L from a surface sample. pH ranged from 7.5 to 7.9. Profile depths listed in the field notes (ranging from surface to 2.5 meters) were multiplied by the ratio of total depth reported in the field notes to the depth measured on the probe cable at each monitoring station because the probe was drifting and indicating depths greater than actual (Anna Sofranko, USEPA Region IX, personal communication, May 12, 2009).

Table G-51. Field Data for the March 10, 2009 Echo Park Lake Sampling Event

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Secchi Depth (m)	Total Depth (m)
EPL-50	9:40	Surface	15.2	7.1	10.0	1.27	1.40
		0.28	15.3	7.5	8.4		
		0.56	15.3	7.5	8.3		
		0.84	15.2	7.6	8.3		
		1.12	15.1	7.6	8.1		
		1.40	15.1	7.7	7.9		
EPL-51	10:30	Surface	15.5	7.7	8.6	1.14	1.67
		0.34	15.4	7.8	8.2		
		0.67	15.3	7.8	8.2		
		1.00	15.0	7.8	8.3		
		1.34	15.0	7.9	8.2		
		1.68	15.0	7.9	8.0		
EPL-52	11:00	Surface	16.2	7.9	7.5	0.91	1.83
		0.36	16.1	7.9	7.5		
		0.73	16.0	7.9	7.2		
		1.10	15.7	7.9	7.4		
		1.46	15.5	7.9	7.3		
		1.82	15.3	7.8	7.0		

Profile data collected in Echo Park Lake in the summer of 2009 are summarized in Table G-52. DO concentrations in the lake ranged from 6.4 mg/L to 7.6 mg/L. The pH ranged from 8.3 to 8.6. Based on this data the lake appears well mixed vertically in the summer as well as in the spring. The Secchi depths at EPL-51 and EPL-52 were much less during the August sampling. The Secchi depth at EPL-51 is 0.84 meters less than the spring Secchi depth. At station EPL-52, the Secchi depth is 0.35 meters less in August than in March. The temperature of the lake ranged from 26.1 to 26.5°C, approximately 10°C higher than the March temperatures. The increased temperature and decreased lake clarity illustrate an increase in algal productivity during this summer season.

Table G-52. Field Data for the August 4, 2009 Echo Park Lake Sampling Event

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Secchi Depth (m)	Total Depth (m)
EPL-51	9:05	Surface	26.1	8.3	6.7	0.30	1.73
		0.5	26.1	8.3	6.6		
		1.0	26.1	8.3	6.5		
		1.5	26.1	8.4	6.4		

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Secchi Depth (m)	Total Depth (m)
EPL-52	9:45	Surface	26.5	8.6	7.5	0.56	1.78
		0.1	26.5	8.6	7.5		
		0.5	26.4	8.6	7.5		
		1.0	26.3	8.6	7.6		
		1.5	26.3	8.6	7.5		

Field data were collected for the potable water source during the August sampling event. After purging the line for approximately five minutes, the pH was 7.48, the DO was 8.73 mg/L, and the temperature was 21.95 ºC.

The city of Los Angeles provided water quality monitoring data for the Glendale Water Reclamation Plant, which may be used to supplement lake levels and irrigate parkland at Echo Park in the future. Table G-53 summarizes the average water quality for this source based on monthly averages reported for 2008 and 2009.

Table G-53. Average Water Quality for the Glendale Water Reclamation Plant

NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Orthophosphate (mg/L)	TP (mg/L)
1.3	3.17	5.64	0.009	1.76	1.93

G.6.2 MONITORING RELATED TO METALS IMPAIRMENTS

In 1996 Echo Park Lake was deemed impaired by copper and lead. Monitoring data for cadmium, copper, lead, and zinc are presented in this section. Echo Park Lake is not listed for cadmium or zinc, but those data are presented here for completeness because other waterbodies in the region are affected by some of these contaminants.

Metals data collected at Echo Park Lake (pink triangle, Figure G-13), as part of the 1992-1993 Urban Lakes Study (UC Riverside, 1994), are shown in Table G-54. Specifically, sampling included dissolved copper and dissolved lead. Dissolved copper samples were collected throughout the water column at depths from the surface to two meters. The range of the 31 dissolved copper samples was between less than 10 µg/L and 105 µg/L. Similarly, dissolved lead samples were also collected throughout the water column, again at depths from the surface to two meters. The 31 samples collected ranged in concentration from less than 1 µg/L to 105 µg/L.

The Regional Board completed its Water Quality Assessment and Documentation Report for waterbodies in the Los Angeles Region in 1996 (LARWQCB, 1996). The summary table for Echo Park Lake states that copper and lead were not supporting the assessed uses: 31 measurements had a maximum lead concentration of 105 µg/L, a maximum copper concentration of 105 µg/L, and a maximum zinc concentration of 14 µg/L (raw data were not provided, but it is assumed that most of these samples are associated with the Urban Lake Study [UC Riverside, 1994]).

Unfortunately, metals levels were analyzed at relatively high detection limits compared to current detection limits; dissolved copper minimum detection 10 µg/L while dissolved lead was 1 µg/L. No hardness data were collected as part of the Urban Lakes Study, thus it cannot be compared to the hardness-based water quality objectives.

Table G-54. Echo Park Lake 1992/1993 Monitoring Data for Metals

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
7/14/1992	0	14	<1
	1.5	15	<1
7/14/1992	0	20	<1
	1.5	19	<1
7/14/1992	0	N/A	<1
	1.3	20	<1
8/13/1992	0	87	2
	1.5	N/A	<1
8/13/1992	0	42	<1
	2	33	<1
	0	31	2
9/17/1992	0	105	1
	1.7	95	2
10/15/1992	0	40	6
	1.7	16	<1
11/5/1992	0	<10	1
	1.5	<10	1
12/8/1992	0	<10	6
	1.5	<10	6
1/12/1993	0	<10	4
	1.5	15	1
2/2/1993	0	15	7
	1.5	<10	2
3/17/1993	0	<10	37
	1.5	13	97
4/7/1993	0	<10	5
	1.5	<10	18
5/3/1993	0	11	1
	2	10	1
6/8/1993	0	<10	105
	1.5	11	60

Table G-55 presents 61 additional metal samples that were collected by the USEPA, Regional Board, and/or the city of Los Angeles between October 2008 and March 2010. Samples were collected at locations: EPL-1, EPL-2, EPL-3, EPL-4, EPL-5, EPL-7, EPL-8, EPL-9, EPL-12, EPL-50, EPL-51, EPL-52 and EPL-Shore. Sites were analyzed for dissolved cadmium, copper, lead and/or zinc.

Detection limits were lower than the 1992-1993 study with a cadmium detection limit of 0.2 µg/L, dissolved copper detection limit of 0.1 µg/L, dissolved lead detection limit of 0.05 µg/L, and dissolved zinc detection limit of 0.2 µg/L. All dissolved cadmium concentrations were < 0.2 µg/L to 0.2 µg/L; copper concentrations were between 0.7 µg/L and 26.3 µg/L; lead concentrations ranged from 0.1 µg/L to 5.5 µg/L; and zinc concentrations were <0.1 µg/L to 2.7 µg/L. In addition, three total lead samples were collected by the Regional Board in June 2008 at EPL3, EPL4, and EPL5. The total lead concentrations ranged from 4.1µg/L to 14.3 µg/L (with a hardness range of 265 mg/L to 267 mg/L). Metal toxicity is affected by hardness; therefore, each sample was also analyzed for hardness. The 2008-2010 dissolved metals sampling resulted in a hardness range of 106 mg/L to 283 mg/L. Since dissolved results pertain to the applicable standard and recent data more closely represents current conditions, data in Table G-55 were weighted more heavily in the assessment.

Table G-55. Metals Data for the 2008-2010 Echo Park Lake Sampling Events

Organiz-ation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
City LA	11/17/2004	EPL 7/8/9	153	N/A	7.3	4.7	N/A	average of 7, 8, 9
City LA	2/17/2005	EPL 7/8/9	106	N/A	11.0	3.0	N/A	average of 7, 8, 9
City LA	5/19/2005	EPL 7/8/9	133	N/A	7.7	1.0	N/A	average of 7, 8, 9
City LA	8/18/2005	EPL 7/8/9	130	N/A	26.3	4.0	N/A	average of 7, 8, 9
City LA	11/17/2005	EPL 7/8/9	154	N/A	4.7	5.5	N/A	average of 7, 8, 9
City LA	2/9/2006	EPL 7/8/9	167	N/A	4.0	1.0	N/A	average of 7, 8, 9
City LA	5/11/2006	EPL 7/8/9	147	N/A	9.0	1.2	N/A	average of 7, 8, 9
City LA	8/17/2006	EPL 7/8/9	111	N/A	13.0	2.3	N/A	average of 7, 8, 9
City LA	11/16/2006	EPL 7/8/9	166	N/A	10.3	1.6	N/A	average of 7, 8, 9
City LA	2/8/2007	EPL 7/8/9	168	N/A	6.3	1.6	N/A	average of 7, 8, 9
City LA	5/17/2007	EPL 7/8/9	179	N/A	17.7	1.8	N/A	average of 7, 8, 9
City LA	8/16/2007	EPL 7/8/9	167	N/A	5.0	1.1	N/A	average of 7, 8, 9
City LA	11/7/2007	EPL 7/8/9	184	N/A	1.0	1.1	N/A	average of 7, 8, 9
City LA	2/14/2008	EPL 7/8/9	186	N/A	2.7	1.1	N/A	average of 7, 8, 9
City LA	5/8/2008	EPL 7/8/9	231	N/A	5.6	3.1	N/A	average of 7, 8, 9
City LA	8/7/2008	EPL 7/8/9	214	N/A	6.5	2.0	N/A	average of 7, 8, 9
City LA	11/20/2008	EPL 7/8/9	283	N/A	3.1	2.0	N/A	average of 7, 8, 9
RB	12/18/2008	EPL 1 R1	208	<0.2	1.6	0.5	2.3	average of replicates; lotus bed location
RB	12/18/2008	EPL 2/3/4	216	<0.2	2.5	0.2	2.4	average of duplicates; average of site 2, 3, &

Organiz-ation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
								4 (shoreline)
RB	12/18/2008	EPL 5 R1	209	<0.2	2.6	0.2	2.7	oxbow location
City LA	2/19/2009	EPL 7/8/9	226	N/A	3.1	2.0	N/A	average of 7, 8, 9
RB/EPA	3/10/2009	EPL 50/51/52	178	<0.2	1.8	1.0	1.5	average of duplicates & MS; average of 50, 51 & 52
City LA	3/26/2009	EPL-shore	234	N/A	4.9	0.5	N/A	
City LA	5/21/2009	EPL 7/8/9	230	N/A	2.7	2.0	N/A	average of 7, 8, 9
RB/EPA	8/4/2009	EPL 1	186	<0.2	1.7	0.7	<0.1	
RB/EPA	8/4/2009	EPL 12	192	<0.2	0.7	0.5	2.1	
RB/EPA	8/4/2009	EPL 5	190	<0.2	1.5	0.6	0.4	
RB/EPA	8/4/2009	EPL 51/52	190	0.2	1.5	0.6	0.1	average of replicates and 51/52
City LA	8/18/2009	EPL 7/8/9	189	N/A	2.9	0.4	N/A	average of replicates and sites
City LA	8/28/2009	EPL 8	199	N/A	3.8	0.7	N/A	
City LA	8/28/2009	EPL-shore	204	N/A	4.8	0.6	N/A	
City LA	9/4/2009	EPL 8	198	N/A	1.4	0.2	N/A	
City LA	9/4/2009	EPL-shore	209	N/A	1.7	0.2	N/A	
City LA	9/11/2009	EPL-shore	206	N/A	2.8	0.5	N/A	
City LA	9/25/2009	EPL-shore	207	N/A	3.8	0.5	N/A	
City LA	10/2/2009	EPL-shore	203	N/A	3.0	0.2	N/A	
City LA	10/9/2009	EPL-shore	196	N/A	4.0	0.7	N/A	
City LA	10/16/2009	EPL-shore	222	N/A	4.8	0.5	N/A	sampling after rainy weather
City LA	10/23/2009	EPL-shore	213	N/A	4.8	0.8	N/A	
City LA	10/30/2009	EPL-shore	229	N/A	1.6	0.4	N/A	
City LA	11/6/2009	EPL-shore	231	N/A	3.7	0.7	N/A	
City LA	11/13/2009	EPL-shore	216	N/A	2.8	0.3	N/A	
City LA	11/20/2009	EPL-shore	208	N/A	3.4	0.5	N/A	
RB/EPA	12/1/2009	EPL 1	193	0.2	1.9	0.2	0.1	average of replicates
RB/EPA	12/1/2009	EPL 12	193	<0.2	1.5	0.7	0.4	
RB/EPA	12/1/2009	EPL 51/52	191	<0.2	1.4	0.4	0.3	average of replicates and 51/52

Organiz-ation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
City LA	12/4/2009	EPL-shore	216	N/A	4.0	0.4	N/A	
City LA	12/11/2009	EPL-shore	214	N/A	4.2	0.3	N/A	
City LA	12/18/2009	EPL-shore	215	N/A	N/A	0.3	N/A	
City LA	12/18/09 & 12/22/09	Mid Lake EPL 7/8/9	231	N/A	3.3	0.4	N/A	average of dry mid-lake samples
City LA	1/8/2010	EPL-shore	250	N/A	4.5	0.5	N/A	
City LA	1/15/2010	EPL-shore	227	N/A	4.6	0.4	N/A	
City LA	1/22/2010	EPL-shore	197	N/A	3.0	0.4	N/A	
City LA	2/5/2010	EPL-shore	218	N/A	4.0	0.4	N/A	
City LA	2/12/2010	EPL-shore	212	N/A	4.9	0.2	N/A	
City LA	2/16/2010	Mid Lake EPL 7/8/9	215	N/A	2.9	0.2	N/A	average of mid-lake samples
City LA	2/19/2010	EPL-shore	226	N/A	2.7	0.3	N/A	
City LA	2/26/2010	EPL-shore	212	N/A	2.2	0.1	N/A	
City LA	3/5/2010	EPL-shore	236	N/A	2.4	0.2	N/A	
City LA	3/12/2010	EPL-shore	234	N/A	3.6	0.4	N/A	
City LA	3/19/2010	EPL-shore	236	N/A	6.6	2.5	N/A	

N/A = No data available.
RB = Regional Board
EPA = USEPA
City LA = City of Los Angeles

G.6.3 MONITORING RELATED TO ORGANOCHLORINE PESTICIDES AND PCBS IMPAIRMENT

Echo Park Lake is impaired by chlordane, dieldrin, and PCBs. Monitoring data for chlordane, DDT, dieldrin, and PCBs in Echo Park Lake are reviewed in this section. Echo Park Lake is not listed for DDT but those data are presented here for completeness because other lakes in the region (Peck Road Park Lake and Puddingstone Reservoir) are affected by this contaminant.

In 2008, UCLA conducted organics measurements at Echo Park Lake at five locations. Site location descriptions are listed in Table G-56. To avoid confusion with the City of Los Angeles Bureau of Sanitation numbering scheme, all sites were assigned an alternate label.

Table G-56. Site Locations for the 2008 UCLA Echo Park Lake Monitoring Event

UCLA Site Number	Project Site	Alternate Label
1	Below City of LA storm drain	EPL-1
2	Below County of LA storm drain	EPL-6
3	Lower Lake	EPL-9
4	Lotus Beds	EPL-LB
5	Hydroponic Island	EPL-HI

The Regional Board conducted organics monitoring in Echo Park Lake in December 2008. To avoid confusion with the City of Los Angeles Bureau of Sanitation numbering scheme, all sites were assigned an alternate label (Table G-57).

Table G-57. Site Locations for the 2008 Regional Board Echo Park Lake Monitoring Event

Regional Board Site Number	Project Site	Alternate Label
1	Shoreline sample at northwest segment of lake	EPL-12
2	Shoreline sample at western side of lake	EPL-11
3	Shoreline sample on southern edge of lake	EPL-10
4	Shoreline sample on eastern side of lake	EPL-13
5	Shoreline sample near City of LA storm drain	EPL-1 (shoreline)

Additional samples were collected by the USEPA and the Regional Board in December 2009 at EPL-12, EPL-51, and EPL-52. Alternative labels were not needed for these locations.

G.6.3.1 Water Column Data Observed in Echo Park Lake

Lake water samples were collected from EPL-9 and EPL-LB in the summer of 2008 as part of an organics study performed by UCLA and funded by a grant managed by the Regional Board.

The samples were analyzed for chlordane, DDT, dieldrin, and PCBs. All contaminants were below reportable levels. Table G-58 shows the results and detection limits for each constituent. PCB-31 was detected at EPL-9 and PCB-5 was detected at EPL-LB, but not at reportable levels .

Table G-58. Results from Water Column Samples Collected at Echo Park Lake in Summer 2008

	EPL-9			EPL-9 (field duplicate)			EPL-LB		
	DL	RL	Result	DL	RL	Result	DL	RL	Result
Contaminant	(ng/L)								
Chlordane-gamma	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
Chlordane-alpha	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
4,4'-DDE	3.00	30.00	ND	3.00	30.00	ND	3.00	30.00	ND

Contaminant	EPL-9			EPL-9 (field duplicate)			EPL-LB		
	DL	**RL**	**Result**	**DL**	**RL**	**Result**	**DL**	**RL**	**Result**
	(ng/L)								
4,4'-DDD	3.00	30.00	ND	3.00	30.00	ND	3.00	30.00	ND
4,4'-DDT	3.00	30.00	ND	3.00	30.00	ND	3.00	30.00	ND
Dieldrin	3.00	30.00	ND	3.00	30.00	ND	3.00	30.00	ND
PCB 5	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	8.43*
PCB 18	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 31	1.50	15.00	3.50*	1.50	15.00	4.15*	1.50	15.00	ND
PCB 52	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 44	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 66	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 101	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 87	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 151	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 110	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 153	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 141	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 138	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 187	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 183	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 180	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 170	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND
PCB 206	1.50	15.00	ND	1.50	15.00	ND	1.50	15.00	ND

*Result was above detection limit but below the reporting limit

Water samples from Echo Park Lake were also collected by the Regional Board on December 18, 2008 at EPL-12, EPL-11, EPL-10, EPL-13, and EPL-1. The samples were only analyzed for PCB congeners, no organochloride pesticides were analyzed. PCBs at all stations were not detected. Each congener had a detection limit of 1 ng/L (Table G-59).

Table G-59. Results from Water Column Samples Collected at Echo Park Lake on December 18th, 2008

Contaminant (ng/L)	EPL-12	EPL-11	EPL-10	EPL-13	EPL-1	MDL
PCB003	ND	ND	ND	ND	ND	1.0
PCB008	ND	ND	ND	ND	ND	1.0
PCB018	ND	ND	ND	ND	ND	1.0

Contaminant (ng/L)	EPL-12	EPL-11	EPL-10	EPL-13	EPL-1	MDL
PCB028	ND	ND	ND	ND	ND	1.0
PCB031	ND	ND	ND	ND	ND	1.0
PCB033	ND	ND	ND	ND	ND	1.0
PCB037	ND	ND	ND	ND	ND	1.0
PCB044	ND	ND	ND	ND	ND	1.0
PCB049	ND	ND	ND	ND	ND	1.0
PCB052	ND	ND	ND	ND	ND	1.0
PCB056/060	ND	ND	ND	ND	ND	1.0
PCB066	ND	ND	ND	ND	ND	1.0
PCB070	ND	ND	ND	ND	ND	1.0
PCB074	ND	ND	ND	ND	ND	1.0
PCB077	ND	ND	ND	ND	ND	1.0
PCB081	ND	ND	ND	ND	ND	1.0
PCB087	ND	ND	ND	ND	ND	1.0
PCB095	ND	ND	ND	ND	ND	1.0
PCB097	ND	ND	ND	ND	ND	1.0
PCB099	ND	ND	ND	ND	ND	1.0
PCB101	ND	ND	ND	ND	ND	1.0
PCB105	ND	ND	ND	ND	ND	1.0
PCB110	ND	ND	ND	ND	ND	1.0
PCB114	ND	ND	ND	ND	ND	1.0
PCB118	ND	ND	ND	ND	ND	1.0
PCB119	ND	ND	ND	ND	ND	1.0
PCB123	ND	ND	ND	ND	ND	1.0
PCB126	ND	ND	ND	ND	ND	1.0
PCB128	ND	ND	ND	ND	ND	1.0
PCB138	ND	ND	ND	ND	ND	1.0
PCB141	ND	ND	ND	ND	ND	1.0
PCB149	ND	ND	ND	ND	ND	1.0
PCB151	ND	ND	ND	ND	ND	1.0
PCB153	ND	ND	ND	ND	ND	1.0
PCB156	ND	ND	ND	ND	ND	1.0
PCB157	ND	ND	ND	ND	ND	1.0
PCB158	ND	ND	ND	ND	ND	1.0

Contaminant (ng/L)	**EPL-12**	**EPL-11**	**EPL-10**	**EPL-13**	**EPL-1**	**MDL**
PCB167	ND	ND	ND	ND	ND	1.0
PCB168+132	ND	ND	ND	ND	ND	1.0
PCB169	ND	ND	ND	ND	ND	1.0
PCB170	ND	ND	ND	ND	ND	1.0
PCB174	ND	ND	ND	ND	ND	1.0
PCB177	ND	ND	ND	ND	ND	1.0
PCB180	ND	ND	ND	ND	ND	1.0
PCB183	ND	ND	ND	ND	ND	1.0
PCB187	ND	ND	ND	ND	ND	1.0
PCB189	ND	ND	ND	ND	ND	1.0
PCB194	ND	ND	ND	ND	ND	1.0
PCB195	ND	ND	ND	ND	ND	1.0
PCB200	ND	ND	ND	ND	ND	1.0
PCB201	ND	ND	ND	ND	ND	1.0
PCB203	ND	ND	ND	ND	ND	1.0
PCB206	ND	ND	ND	ND	ND	1.0
PCB209	ND	ND	ND	ND	ND	1.0

G.6.3.2 Porewater Data Observed in Echo Park Lake

Samples of porewater from summer 2008 were analyzed for EPL-1, EPL-6, EPL-9, EPL-LB and EPL-HI. PCB-5 was detected in the porewater at EPL-6 and EPL-9, and PCB-31 was detected at EPL-HI. None of the organic pollutants were detected at EPL-LB. EPL-1 was not reported due to a laboratory error during analysis. The porewater from EPL-6 was the only sample with sufficient TSS for analysis. PCB-66 was detected in the TSS, but not at reportable levels. The results of the porewater analysis are shown in Table G-60 (see Stenstrom et al., 2009 for raw data).

Table G-60. Results from Porewater Samples Collected at Echo Park Lake in Summer 2008

Contaminant	**Porewater (ng/L)**					**TSS in Porewater (µg/kg)**	
	EPL-6	**EPL-9**	**EPL-LB**	**EPL-HI**	**MDL**	**EPL-6**	**MDL**
Chlordane	ND	ND	ND	ND	15	ND	3.65
DDT	ND	ND	ND	ND	30	ND	7.31
Dieldrin	ND	ND	ND	ND	30	ND	7.31
Total PCBs	DNQ[1]	DNQ[1]	ND	DNQ[2]	15	DNQ[3]	3.65

[1] PCB-5 was detected in these samples at less than the reporting level (150 ng/L)

[2] PCB-31 was detected in this sample at less than the reporting level (150 ng/L)

[3] PCB- 66 was detected less than the reporting level (36.54 µg/kg)

Porewater from EPL-1 and EPL-LB in fall 2008 was also analyzed as part of the UCLA study. No contaminants were detected in the porewater from either site. The results of this analysis are shown in Table G-61 (see Stenstrom et al., 2009 for raw data).

Table G-61. Results from Porewater Samples Collected at Echo Park Lake in Fall 2008

Contaminant (ng/L)	EPL-1	EPL-LB	MDL
Chlordane	ND	ND	15 – 1,500
DDT	ND	ND	30 – 3,000
Dieldrin	ND	ND	30 – 3,000
Total PCBs	ND	ND	15 – 1,500

G.6.3.3 Fish Tissue Levels Observed in Echo Park Lake

Concentrations of Aroclor PCBs, chlordane, DDT, dieldrin, and PCBs in largemouth bass, common carp, and bullhead species were reported by SWAMP (Davis et al., 2008) and TSMP (2009), shown in Table G-62. Concentrations of chlordane were highest in bullhead fish; 66.0 ppb on average. Bullhead fish also had higher average concentrations of DDTs and dieldrin than the other species (60.0 ppb and 7.0 ppb, respectively). Largemouth bass had the lowest average concentrations of the organochlorine pesticides. PCBs were only tested in common carp and largemouth bass. The average concentrations of PCBs in common carp was 81.8 ppb and 49.0 ppb in largemouth bass. The average dieldrin concentrations (1.13 ppb) are higher than the 0.45 ppb FCG for dieldrin.

Table G-62. Compiled Fish Tissue Analytical Data for Echo Park Lake

Pollutant	Sample Date	Common Name	Concentration (ppb, w wt)	Mean Length (mm)	Mean Weight (g)
Aroclor PCBs	6/17/1987	Bullhead	50	236	205.6
Aroclor PCBs	6/17/1987	Largemouth Bass	84	145	42.6
Aroclor PCBs	4/19/1991	Largemouth Bass	ND	244	271.3
Aroclor PCBs	4/24/1992	Largemouth Bass	60	315	581.8
Total PCBs	Summer 2007	Common Carp	119.01	501	1714.4
Total PCBs	Summer 2007	Common Carp	82.618	380	807.4
Total PCBs	Summer 2007	Largemouth Bass	64.716	498	1823.6
Total PCBs	Summer 2007	Largemouth Bass	31.478	380	916
Total PCBs	4/13/2010	Largemouth Bass	50.863	377.2	901
Total PCBs	4/13/2010	Common Carp	43.861	377.2	928
Chlordane	6/17/1987	Bullhead	66	236	205.6
Chlordane	6/17/1987	Largemouth Bass	17.8	145	42.6
Chlordane	4/19/1991	Largemouth Bass	ND	244	271.3
Chlordane	4/24/1992	Largemouth Bass	ND	315	581.8
Chlordane	Summer 2007	Common Carp	32.19	501	1714.4
Chlordane	Summer 2007	Common Carp	21.96	380	807.4

Pollutant	Sample Date	Common Name	Concentration (ppb, w wt)	Mean Length (mm)	Mean Weight (g)
Chlordane	Summer 2007	Largemouth Bass	15.484	498	1823.6
Chlordane	Summer 2007	Largemouth Bass	0.844	380	916
Chlordane	4/13/2010	Largemouth Bass	2.517	377.2	901
Chlordane	4/13/2010	Common Carp	4.216	377.2	928
DDTs	6/17/1987	Bullhead	60	236	205.6
DDTs	6/17/1987	Largemouth Bass	30	145	42.6
DDTs	4/19/1991	Largemouth Bass	ND	244	271.3
DDTs	4/24/1992	Largemouth Bass	11	315	581.8
DDTs	Summer 2007	Common Carp	23.458	501	1714.4
DDTs	Summer 2007	Common Carp	14.87	380	807.4
DDTs	Summer 2007	Largemouth Bass	13.029	498	1823.6
DDTs	Summer 2007	Largemouth Bass	6.35	380	916
DDTs	4/13/2010	Largemouth Bass	7.448	377.2	901
DDTs	4/13/2010	Common Carp	7.3	377.2	928
Dieldrin	6/17/1987	Bullhead	7	236	205.6
Dieldrin	6/17/1987	Largemouth Bass	ND	145	42.6
Dieldrin	4/19/1991	Largemouth Bass	ND	244	271.3
Dieldrin	4/24/1992	Largemouth Bass	ND	315	581.8
Dieldrin	Summer 2007	Common Carp	1.08	501	1714.4
Dieldrin	Summer 2007	Common Carp	0.79	380	807.4
Dieldrin	Summer 2007	Largemouth Bass	0.848	498	1823.6
Dieldrin	Summer 2007	Largemouth Bass	0.585	380	916
Dieldrin	4/13/2010	Largemouth Bass	[0.45]*	377.2	901
Dieldrin	4/13/2010	Common Carp	0.538	377.2	928

ND = Non-detect

* Values in square brackets are reported concentrations below the practical reporting limit and are included in the averages.

G.6.3.4 Sediment Data Observed in Echo Park Lake

Sediment samples from Echo Park Lake were collected for the UCLA study in the fall and summer of 2008 and then by USEPA and the Regional Board in December 2009. The results from the UCLA study are shown in Table G-63 and Table G-64 for fall and summer, respectively. All samples had laboratory duplicates and a field duplicate was also collected during each event. The only contaminant above reportable limits in the fall was PCB-5 at EPL-1. PCB-66 and PCB-153 were also detected but not quantified at EPL-1. DDT was detected at EPL-LB, but not quantifiable (Table G-63).

Table G-63. Results from Sediment Samples Collected at Echo Park Lake in Fall 2008

	EPL-1			EPL-1 (lab dup)			EPL-1 (field dup)			EPL-1 (lab dup of field dup)			EPL-LB			EPL-LB (lab dup)		
	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.
Contaminant	**µg/kg dry weight**																	
Chlordane-gamma	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
Chlordane-alpha	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
4,4'-DDE	2.07	20.70	ND	1.98	19.78	ND	1.35	13.49	ND	1.63	16.25	ND	1.75	17.54	ND	1.85	18.49	ND
4,4'-DDD	2.07	20.70	ND	1.98	19.78	ND	1.35	13.49	ND	1.63	16.25	ND	1.75	17.54	ND	1.85	18.49	ND
4,4'-DDT	2.07	20.70	ND	1.98	19.78	ND	1.35	13.49	ND	1.63	16.25	ND	1.75	17.54	15.79*	1.85	18.49	ND
Dieldrin	2.07	20.70	ND	1.98	19.78	ND	1.35	13.49	ND	1.63	16.25	ND	1.75	17.54	ND	1.85	18.49	ND
PCB 5	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	9.13	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 18	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 31	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 52	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 44	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 66	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	4.60*	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 101	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 87	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 151	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 110	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 153	1.04	10.35	1.44*	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 141	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 138	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 187	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 183	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 180	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 170	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND
PCB 206	1.04	10.35	ND	0.99	9.89	ND	0.67	6.75	ND	0.81	8.13	ND	0.88	8.77	ND	0.92	9.24	ND

Chlordane and several PCB congeners were detected in the summer 2008 sediment samples and concentrations of each detected congener are shown in Table G-64. Chlordane-gamma was detected at EPL-6 with a concentration of 8 µg/kg dry weight. EPL-6 was also found to have concentration of PCBs over the reporting limits for the following congeners: PCB-5, PCB-44, PCB-52 and PCB-66. EPL-9 had concentration of PCBs over the reporting limit for PCB-5 and PCB-52, while PCB-5 and PCB-138 were reportable at EPL-LB. Dieldrin and DDT were not detected at any of the sampled locations.

Table G-64. Results from Sediment Samples Collected at Echo Park Lake in Summer 2008

	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.
	µg/kg dry weight																	
Contaminant	**EPL-1**			**EPL-1 (lab dup)**			**EPL-6**			**EPL-6 (lab dup)**			**EPL-6 (field dup)**			**EPL-6 (lab dup of field dup)**		
Chlordane-gamma	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	8.02	0.83	5.37	ND	0.60	6.04	ND
Chlordane-alpha	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
4,4'-DDE	1.66	16.63	ND	1.85	18.45	ND	1.06	10.59	ND	0.88	8.77	ND	1.66	10.75	ND	1.21	12.09	ND
4,4'-DDD	1.66	16.63	ND	1.85	18.45	ND	1.06	10.59	ND	0.88	8.77	ND	1.66	10.75	ND	1.21	12.09	ND
4,4'-DDT	1.66	16.63	ND	1.85	18.45	ND	1.06	10.59	ND	0.88	8.77	ND	1.66	10.75	ND	1.21	12.09	ND
Dieldrin	1.66	16.63	ND	1.85	18.45	ND	1.06	10.59	ND	0.88	8.77	ND	1.66	10.75	ND	1.21	12.09	ND
PCB 5	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	12.68
PCB 18	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 31	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 52	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	50.32	0.60	6.04	5.49
PCB 44	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	41.98	0.60	6.04	2.56
PCB 66	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	12.11	0.60	6.04	ND
PCB 101	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 87	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 151	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 110	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 153	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 141	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 138	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 187	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 183	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 180	0.83	8.31	1.32*	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 170	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
PCB 206	0.83	8.31	ND	0.92	9.23	ND	0.53	5.30	ND	0.44	4.38	ND	0.83	5.37	ND	0.60	6.04	ND
Contaminant	**EPL-9**			**EPL-9 (lab dup)**			**EPL-LB**			**EPL-LB (lab dup)**			**EPL-HI**			**EPL-HI (lab dup)**		
Chlordane-gamma	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
Chlordane-alpha	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
4,4'-DDE	1.79	17.85	ND	1.87	18.66	ND	2.46	24.58	ND	2.46	24.60	ND	1.26	12.58	ND	1.13	11.32	ND
4,4'-DDD	1.79	17.85	ND	1.87	18.66	ND	2.46	24.58	ND	2.46	24.60	ND	1.26	12.58	ND	1.13	11.32	ND
4,4'-DDT	1.79	17.85	ND	1.87	18.66	ND	2.46	24.58	ND	2.46	24.60	ND	1.26	12.58	ND	1.13	11.32	ND
Dieldrin	1.79	17.85	ND	1.87	18.66	ND	2.46	24.58	ND	2.46	24.60	ND	1.26	12.58	ND	1.13	11.32	ND
PCB 5	0.89	8.93	ND	0.93	9.33	36.49	1.23	12.29	164.7	1.23	12.30	94.62	0.63	6.29	ND	0.57	5.66	ND
PCB 18	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 31	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 52	0.89	8.93	10.84	0.93	9.33	12.37	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 44	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 66	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 101	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 87	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND

Contaminant	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.
	µg/kg dry weight																	
PCB 151	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 110	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 153	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 141	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 138	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	19.84	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 187	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 183	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 180	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 170	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND
PCB 206	0.89	8.93	ND	0.93	9.33	ND	1.23	12.29	ND	1.23	12.30	ND	0.63	6.29	ND	0.57	5.66	ND

*Results were above detection limit but lower than reporting limits

Sediment sampling was conducted by USEPA and the Regional Board at EPL-12, EPL-51, and EPL-52 on December 1, 2009. Similar to the fall and summer 2008 sampling, DDT and dieldrin were not detected in any of the sediment samples. However, 4,4 - DDE was detected at all three sites. Chlordane-gamma was detected at all three locations between 2 and 5 µg/kg dry weight and chlordane-alpha was detected at all three locations between 2 and 4.8 µg/kg dry weight. PCBs were detected at all three stations. Many of the same congeners were found at different EPL stations; PCB018, PCB095, PCB101, and PCB110 were detected at all locations. Other congeners detected and the results of the sediment sampling are shown in Table G-65.

Table G-65. Results from Sediment Samples Collected at Echo Park Lake on December 1, 2009

Contaminant (µg/kg dry weight)	EPL-12	EPL-51		EPL-52	MDL
		Result	Field Dup		
Chlordane-gamma	2.5	5	ND	2	1
Chlordane-alpha	2	4.8	ND	2.1	1
cis-Nonachlor	ND	ND	ND	ND	1
trans-Nonachlor	ND	ND	ND	ND	1
Oxychlordane	ND	ND	ND	ND	1
2,4 - DDD	ND	ND	ND	ND	1
2,4 - DDE	ND	ND	ND	ND	1
2,4 - DDT	ND	ND	ND	ND	1
4,4 - DDD	ND	ND	ND	ND	1
4,4 - DDE	18.6	21.1	20	5.8	1
4,4 - DDT	ND	ND	ND	ND	1
Dieldrin	ND	ND	ND	ND	1
PCB008	ND	ND	ND	2.1	1
PCB018	7.0	ND	7.4	3.5	1
PCB044	ND	ND	ND	6.7	1
PCB049	ND	ND	ND	3.0	1

Contaminant (µg/kg dry weight)	EPL-12	EPL-51		EPL-52	MDL
		Result	Field Dup		
PCB052	ND	ND	ND	9.0	1
PCB066	ND	ND	ND	8.1	1
PCB087	ND	30.5	ND	ND	1
PCB095	5.0	9.1	8.2	3.5	1
PCB097	ND	ND	ND	5.0	1
PCB099	ND	10.3	8.4	3.0	1
PCB101	7.3	11.4	9.1	4.7	1
PCB105	ND	ND	ND	12.3	1
PCB110	18.8	7.6	19.9	10.3	1
PCB119	ND	9.6	ND	1.3	1
PCB132	ND	ND	ND	4.6	1
PCB149	ND	13.6	ND	ND	1

G.6.3.5 Suspended Sediment Data Observed in Echo Park Lake

Echo Park Lake samples from summer 2008 were analyzed for pollutant concentrations associated with suspended sediments in the lake. Samples were collected at EPL-9, EPL-LB and EPL-HI. At EPL-9, PCB-31 was detected at 117 µg/kg dry weight and PCB-153 was also detected, but not within reportable limits. All other PCB congeners were less than the detection limits. No contaminants were detected at EPL-LB. The sample at EPL-HI did not have sufficient suspended solids for analysis. The results of the sampling are shown in Table G-66.

Table G-66. Results from Suspended Sediment Samples Collected at Echo Park Lake in Summer 2008

	EPL-9			EPL-LB		
	DL	RL	Result	DL	RL	Result
Contaminant	µg/kg dry suspended solids					
Chlordane-gamma	10.05	100.50	ND	3.19	31.95	ND
Chlordane-alpha	10.05	100.50	ND	3.19	31.95	ND
4,4'-DDE	20.10	201.01	ND	6.39	63.90	ND
4,4'-DDD	20.10	201.01	ND	6.39	63.90	ND
4,4'-DDT	20.10	201.01	ND	6.39	63.90	ND
Dieldrin	20.10	201.01	ND	6.39	63.90	ND
PCB 5	10.05	100.50	ND	3.19	31.95	ND
PCB 18	10.05	100.50	ND	3.19	31.95	ND
PCB 31	10.05	100.50	116.74	3.19	31.95	ND
PCB 52	10.05	100.50	ND	3.19	31.95	ND
PCB 44	10.05	100.50	ND	3.19	31.95	ND
PCB 66	10.05	100.50	ND	3.19	31.95	ND
PCB 101	10.05	100.50	ND	3.19	31.95	ND
PCB 87	10.05	100.50	ND	3.19	31.95	ND

PCB 151	10.05	100.50	ND	3.19	31.95	ND
PCB 110	10.05	100.50	ND	3.19	31.95	ND
PCB 153	10.05	100.50	37.43*	3.19	31.95	ND
PCB 141	10.05	100.50	ND	3.19	31.95	ND
PCB 138	10.05	100.50	ND	3.19	31.95	ND
PCB 187	10.05	100.50	ND	3.19	31.95	ND
PCB 183	10.05	100.50	ND	3.19	31.95	ND
PCB 180	10.05	100.50	ND	3.19	31.95	ND
PCB 170	10.05	100.50	ND	3.19	31.95	ND
PCB 206	10.05	100.50	ND	3.19	31.95	ND

*Results were above detection limits, but below reporting limits.
Note: EPL-HI was sampled but could not be analyzed.

G.7 Monitoring Data for Lake Calabasas

Monitoring data relevant to the impairments of Lake Calabasas are available from 1992, 1993, and 2004 through 2009. Figure G-14 shows the historical and recent monitoring locations for Lake Calabasas.

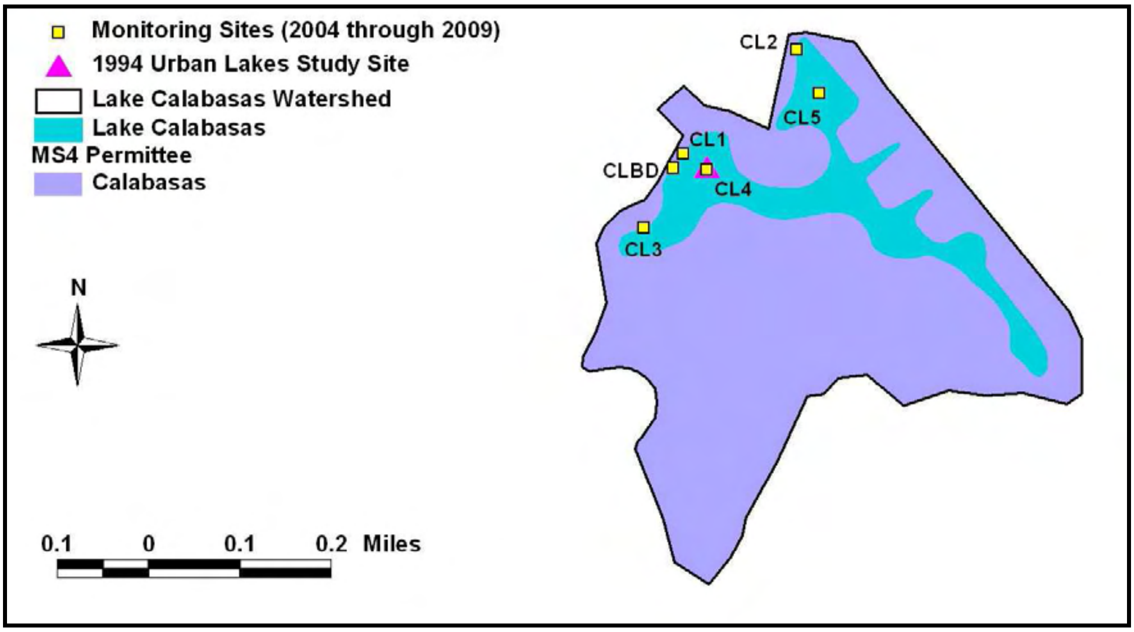


Figure G-14. Lake Calabasas Monitoring Sites

G.7.1 MONITORING RELATED TO NUTRIENT IMPAIRMENTS

Lake Calabasas was monitored from the western side of lake (pink triangle, Figure G-14) in 1992/1993 for water quality (Table G-67) as part of the Urban Lakes Study (UC Riverside, 1994). TKN ranged from 1.0 mg/L to 1.8 mg/L with two samples less than the reporting limit. Ammonium concentrations were usually less than or equivalent to the reporting limit although four samples collected in February and March 1993 ranged from 0.3 mg/L to 0.5 mg/L , the upper end of this range is below the acute target, but above the chronic target (for assessment purposes, we are assuming that the analysis methodology converted all ammonia to ammonium). All of the nitrite and nitrate samples were less than the reporting limit except one nitrate sample of 0.1 mg/L. Five of 28 phosphate samples measured 0.1 mg/L; the others were less than the reporting limit. Total phosphorus concentrations ranged from 0.1 mg/L to 0.2 mg/L with seven samples less than detection. pH in the lake ranged from 8.3 to 9.3. TOC ranged from 5.3 mg/L to 11.5 mg/L. The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from 5 µg/L to 172 µg/L with an average of 39 µg/L.

Table G-67. Lake Calabasas 1992/1993 Monitoring Data

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
7/30/1992	0	1.4	<0.01	<0.01	<0.01	<0.01	0.1	8.8	8.2	479
	1.5	1.5	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	10	445
7/30/1992	0	1.6	<0.01	<0.01	<0.01	0.1	<0.01	9.1	8.8	430
	1.5	1.8	<0.01	<0.01	<0.01	0.1	<0.01	9.0	9.9	451
7/30/1992	0	1.3	<0.01	<0.01	<0.01	<0.01	0.1	9.2	11.2	444
	1.5	1.2	<0.01	<0.01	<0.01	0.1	0.1	9.2	9.9	455
8/17/1992	0	1.5	<0.01	<0.01	<0.01	<0.01	0.1	9.2	9.5	547
	1.5	1.5	0.1	<0.01	<0.01	<0.01	0.1	9.2	9.5	516
9/23/1992	0	1.5	<0.01	<0.01	<0.01	<0.01	0.1	8.8	9.4	495
	1.5	1.6	<0.01	<0.01	<0.01	<0.01	0.1	9.0	10	490
10/21/1992	0	1.5	<0.01	<0.01	<0.01	<0.01	0.1	8.9	11.5	550
	2	1.5	<0.01	<0.01	<0.01	<0.01	0.1	8.9	10.4	561
11/9/1992	0	-	<0.01	<0.01	0.1	<0.01	-	8.3	10.2	481
	1.5	1.7	<0.01	<0.01	<0.01	<0.01	0.1	8.3	10.4	491
12/14/1992	0	1.8	<0.01	<0.01	<0.01	<0.01	0.1	8.8	9.1	539
	1.5	1.7	<0.01	<0.01	<0.01	<0.01	<0.01	8.9	9	541
1/20/1993	0	1.4	<0.01	-	<0.01	<0.01	0.1	8.9	5.3	363
	1.5	1.1	<0.01	<0.01	<0.01	<0.01	<0.01	8.9	6.8	362
2/17/1993	0	1.2	0.5	<0.01	<0.01	0.1	<0.01	8.3	5.7	385
	1.5	1.2	0.4	<0.01	<0.01	0.1	<0.01	8.3	5.4	361
3/18/1993	0	1	0.3	<0.01	<0.01	<0.01	0.1	8.4	6.7	344
	1.5	1.4	0.3	<0.01	<0.01	<0.01	0.2	8.4	6.5	348
4/20/1993	0	<0.01	<0.01	<0.01	<0.01	<0.01	0.2	8.9	6.2	346
	2	<0.01	<0.01	<0.01	<0.01	<0.01	0.1	8.9	5.9	328
5/20/1993	0	1.8	<0.01	<0.01	<0.01	<0.01	0.2	9.2	5.9	322
	1.5	1.8	<0.01	<0.01	<0.01	<0.01	0.1	9.3	5.4	329
6/16/1993	0	1.2	<0.01	<0.01	<0.01	<0.01	0.1	9.0	7.7	356
	1.5	1.3	<0.01	<0.01	<0.01	<0.01	0.1	9.0	8	357

There were no stations in Lake Calabasas or its watershed in the Regional Board Water Quality Assessment Database. The Report (LARWQCB, 1996), however, states that DO was partially supporting the aquatic life use and that 92 measurements of dissolved oxygen were collected which ranged from

0.2 mg/L to 15.7 mg/L. pH was partially supporting the aquatic life use and not supporting the secondary drinking water standards. pH was measured 85 times, and values ranged from 7.4 to 9.3. Ammonia was listed as not supporting the aquatic life or contact recreation uses. Twenty-eight ammonia samples were collected ranging from non-detect to 0.45 mg/L. Raw data are not available to assess location, date, time, depth, temperature, or pH with regard to these samples. Odor was listed as not supporting the contact and non-contact recreation uses. Eutrophication was not supporting the aquatic life use.

The city of Calabasas has been monitoring water quality in Lake Calabasas from a boat dock since 2004 at station CLBD. Table G-68 presents the monthly monitoring data collected through 2008. Nitrate concentrations have ranged from 0.04 mg/L to 1.6 mg/L; phosphate concentrations ranged from 0.03 mg/L to 0.77 mg/L. Secchi depths range from 0.5 m to greater than 2.7 m, and pH ranged from 7.91 to 9.69. Dissolved oxygen has been observed ranging from 4.8 mg/L to 15.82 mg/L with water temperatures ranging from 48.5 °F to 90.8 °F.

Table G-68. City of Calabasas 2004 to 2008 Monitoring Data

Date	NO_3-N (mg/l)	PO_4–P (mg/l)	Secchi Depth (m)	pH	Dissolved Oxygen (mg/l)	Water Temperature (°F)	Water Hardness (mg/l)	Total Dissolved Solids (mg/l)	Salinity (ppt)
1/20/2004	0.34	0.08	1.5	8.13	8.2	66.46	140	625	0.23
2/17/2004	0.30	0.11	1.8	8.01	8.12	65.23	135	640	0.28
3/24/2004	0.20	0.16	2.1	7.96	7.94	69.5	145	646	0.31
4/20/2004	0.15	0.20	2.4	7.91	7.71	70.5	145	652	0.45
5/19/2004	0.18	0.13	1.2	8.5	8.33	74.9	140	658	0.47
6/29/2004	0.23	0.11	0.5	9.07	9.8	78.08	140	661	0.5
7/28/2004	0.25	0.06	0.6	8.99	9.18	83.7	145	674	0.51
8/6/2004	0.23	0.19	1.1	8.12	9.8	81.18	145	694	0.52
9/28/2004	0.09	0.08	0.9	9.29	8.79	73.64	150	699	0.53
10/26/2004	0.16	0.03	0.9	9.29	11.29	64.79	145	633	0.48
11/24/2004	0.14	0.08	1.5	8.92	7.79	55.96	140	616	0.47
12/8/2004	0.48	0.09	1.8	7.91	8.74	49.69	140	717	0.49
1/11/2005	0.36	0.29	1.5	8.14	10.15	53.54	145	438	0.33
2/10/2005	0.36	0.07	2.1	8.83	11.26	56.22	145	445	0.34
3/31/2005	0.45	0.38	1.7	8.85	8.67	64.4	135	362	0.27
4/19/2005	0.50	0.20	1.2	8.98	8.59	68.03	140	402	0.3
5/3/2005	0.18	0.27	1.1	9.02	9.54	73.67	140	411	0.32
6/21/2005	0.48	0.07	0.8	8.85	7.95	79.77	145	451	0.34
7/27/2005	0.50	0.08	0.9	8.8	7.21	86.42	150	485	0.36
8/24/2005	0.34	0.05	0.8	8.81	11.45	81.82	155	521	0.39
9/19/2005	0.55	0.12	1.1	8.08	8.59	74.03	155	574	0.4

Date	NO_3-N (mg/l)	PO_4–P (mg/l)	Secchi Depth (m)	pH	Dissolved Oxygen (mg/l)	Water Temperature (℉)	Water Hardness (mg/l)	Total Dissolved Solids (mg/l)	Salinity (ppt)
10/10/2005	0.57	0.17	1.1	8.24	9.74	68	155	550	0.42
11/17/2005	0.59	0.09	0.8	8.64	14.2	62.96	155	551	0.42
12/22/2005	0.36	0.04	0.9	8.91	14.14	55.62	155	551	0.42
1/18/2006	0.66	0.05	0.9	9.1	10.86	52.9	155	501	0.39
2/6/2006	0.68	0.08	0.7	9.69	11.43	59.68	155	529	0.4
3/1/2006	0.52	0.15	0.8	9.32	8.48	56.31	145	539	0.41
4/26/2006	0.25	0.12	1.8	8.65	6.22	65.51	130	527	0.4
5/22/2006	0.66	0.06	1.8	9.03	6.18	74.46	130	546	0.41
6/22/2006	0.41	0.07	1.2	9.03	10.76	85.13	140	579	0.43
7/24/2006	1.61	0.12	1.1	8.95	7.38	90.82	140	615	0.46
8/16/2006	0.43	0.06	1.2	8.83	9.77	81.51	145	619	0.47
9/25/2006	0.48	0.06	1.2	9.57	10.41	76.44	145	647	0.49
10/18/2006	0.23	0.07	1.5	8.5	13.35	67.27	150	654	0.5
11/28/2006	0.50	0.06	0.6	9.38	6.69	58.46	150	653	0.5
12/27/2006	0.52	0.09	0.6	9.16	7.96	54.43	155	658	0.51
1/9/2007	0.61	0.12	0.8	8.92	6.57	50.01	160	661	0.51
2/22/2007	0.48	0.09	0.9	8.23	7.86	55.21	150	623	0.49
3/15/2007	0.18	0.12	1.2	8.03	4.96	68.6	145	593	0.45
4/26/2007	0.25	0.06	1.5	9.03	14.62	71.32	145	588	0.44
5/18/2007	0.30	0.77	1.4	8.84	8.13	71.56	150	610	0.46
6/22/2007	0.20	0.12	0.9	8.95	8.19	79.56	155	643	0.49
7/27/2007	0.27	0.05	0.9	9.17	7	82.78	160	661	0.5
8/17/2007	0.41	0.07	0.9	9.02	7.59	84.23	165	672	0.51
9/21/2007	0.39	0.07	0.9	8.96	8.67	76.7	155	643	0.49
10/5/2007	0.45	0.04	0.9	9.18	12	71.15	155	653	0.5
11/1/2007	0.41	0.04	1.5	8.79	6.06	67.72	155	660	0.5
12/19/2007	0.39	0.04	1.5	8.97	4.8	52.79	155	653	0.5
1/31/2008	0.43	0.03	1.8	9.53	11.78	54.35	140	527	0.4
2/29/2008	0.27	0.08	1.8	9.2	10.1	58.7	135	526	0.4
3/7/2008	0.32	0.47	1.5	9.07	15.82	62.02	130	524	0.4
4/8/2008	0.25	0.12	1.8	8.76	9.23	66.71	140	551	0.42

Date	NO_3-N (mg/l)	PO_4–P (mg/l)	Secchi Depth (m)	pH	Dissolved Oxygen (mg/l)	Water Temperature (℉)	Water Hardness (mg/l)	Total Dissolved Solids (mg/l)	Salinity (ppt)
5/6/2008	0.30	0.14	>2.7	8.48	6.05	70.95	145	588	0.45
6/19/2008	0.16	0.14	1.8	9	12.64	84.9	150	649	0.49
7/10/2008	0.41	0.18	1.8	8.97	8.88	86.3	145	658	0.49
8/21/2008	0.09	0.09	1.2	9.45	9.48	80.63	150	680	0.51
9/11/2008	0.07	0.11	1.2	9.25	8.6	79.53	155	699	0.53
10/23/2008	0.09	0.23	1.2	9.22	12.22	66.78	160	711	0.54
11/26/2008	0.11	>0.65	2.1	8.95	12.1	59.7	155	702	0.54
12/19/2008	0.05	0.18	2.1	8.82	11.76	48.56	150	695	0.52

The Regional Board sampled Lake Calabasas from three shoreline sites on January 15, 2009. Data are presented in Table G-69.

Table G-69. Analytical Data for the January 15, 2009 Lake Calabasas Sampling Event

Station	NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Ortho-phosphate (mg/L)	Total Phosphate (mg/L)	Total Dissolved Solids (mg/L)	Suspended Solids (mg/L)
CL1	0.391	1.92	0.154	<0.1	<0.4	<0.5	479	<10
CL1 (duplicate)	0.431	1.64	0.157	<0.1	<0.4	<0.5	623	<10
CL2	0.453	1.79	0.192	<0.1	<0.4	<0.5	610	<10
CL3	0.42	1.7	0.164	<0.1	<0.4	<0.5	622	<10

Total depth and Secchi depth were not measured during this event. Dissolved oxygen concentrations were not measured either. Temperature ranged from 11 ºC to 12 ºC and pH ranged from 8.2 to 8.7 (Table G-70).

Table G-70. Field Data for the January 15, 2009 Lake Calabasas Sampling Event

Site	Time	Depth (m)	Temp (C)	pH
CL1	10:20	surface	12.0	8.7
CL2	11:25	Surface	11.5	8.2
CL3	12:05	surface	11.0	8.3

USEPA sampled Lake Calabasas from two in-lake sites and the potable water input (CL-PW) on August 6, 2009 (Table G-71). In-lake ammonia concentrations were less than or equal to 0.03 mg-N/L; TKN ranged from 1.17 mg-N/L to 1.23 mg-N/L. Nitrate and nitrite samples were less than the detection limit of 0.01 mg-N/L. Orthophosphate ranged from 0.0129 mg-P/L to 0.0453 mg-P/L and total phosphorus ranged from 0.152 mg-P/L to 0.221 mg-P/L. Chlorophyll *a* ranged from 35 to 81 µg/L.

Table G-71. Analytical Data for the August 6, 2009 Lake Calabasas Sampling Event

Station	NH_3-N (mg/L)	TKN (mg/L)	Nitrate (mg/L)	Nitrite (mg/L)	Ortho-phosphate (mg/L)	TP (mg/L)	Chlorophyll *a* (ug/L)	Suspended Solids (mg/L)
CL-4	<0.03	1.227	<0.01	<0.01	0.0452	0.221	35.35	8.65
CL-5	0.03	1.166	<0.01	<0.01	0.0129	0.152	81.40	8.00
CL-PW	0.35	0.464	1.13	<0.01	0.0208	<0.016	NA	<0.5

Temperature and dissolved oxygen were not measured in the lake during this event. The total depth at CL-4 was 2.51 m and the depth at CL-5 was 2.06 m. The Secchi readings at CL-4 and CL-5 were 0.737 m and 0.660 m, respectively. Field data for the in-lake sites are presented in Table G-72. Field data were collected for the potable water source on August 6, 2009. After purging the line for approximately 10 minutes, the pH was 7.93 and the temperature was 18.2 ºC.

Table G-72. Field Data for the August 6, 2009 Lake Calabasas Sampling Event

Site	Time	Depth (m)	Secchi Depth (m)
CL-4	9:15	2.51	0.737
CL-5	9:40	2.06	0.660

Profile data were collected at Stations CL-4 and CL-5 on the morning of August 6, 2009 between 9:00 and 9:50. The depth at CL-4 was 2.51 meters, and the Secchi depth was 0.74 meters. The temperature in the lake ranged from 25.6 and 26.1°C. The specific conductivity was constant with depth, around 1.22 mS/cm. The DO ranged from 6.37 to 7.56 mg/L and pH ranged from 7.57 to 8.77. These profile data are shown in Figure G-15.

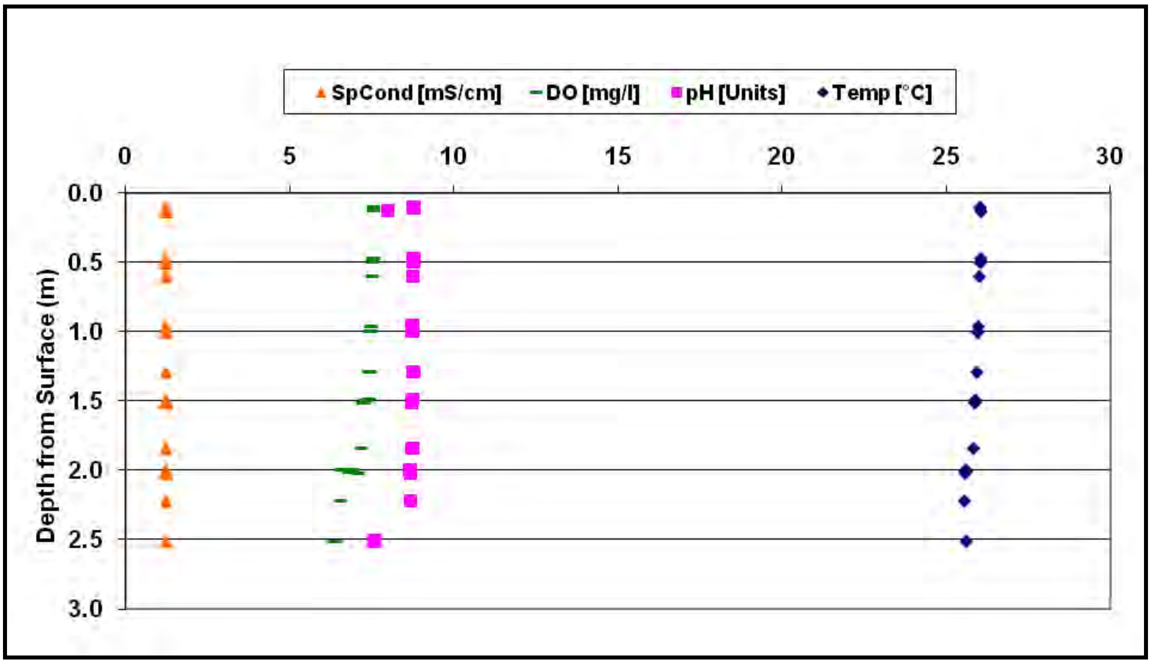


Figure G-15. Profile Data Collected in Lake Calabasas at CL-4 on August 6, 2009

The profile data collected at CL-5 is shown in Figure G-16. The temperature at this station was between 25.82 and 26.45°C. The pH ranged from 9.04 to 9.20. Dissolved oxygen ranged from 8.71 to 9.74 mg/L. The conductivity was between 1.04 to 1.05 mS/cm. The field team observed that this location was close to the tap water inlet and likely affected the conductivity levels, which were lower than those at CL-4. The depth at this station was 1.75 meters and the Secchi depth was 0.66 meters.

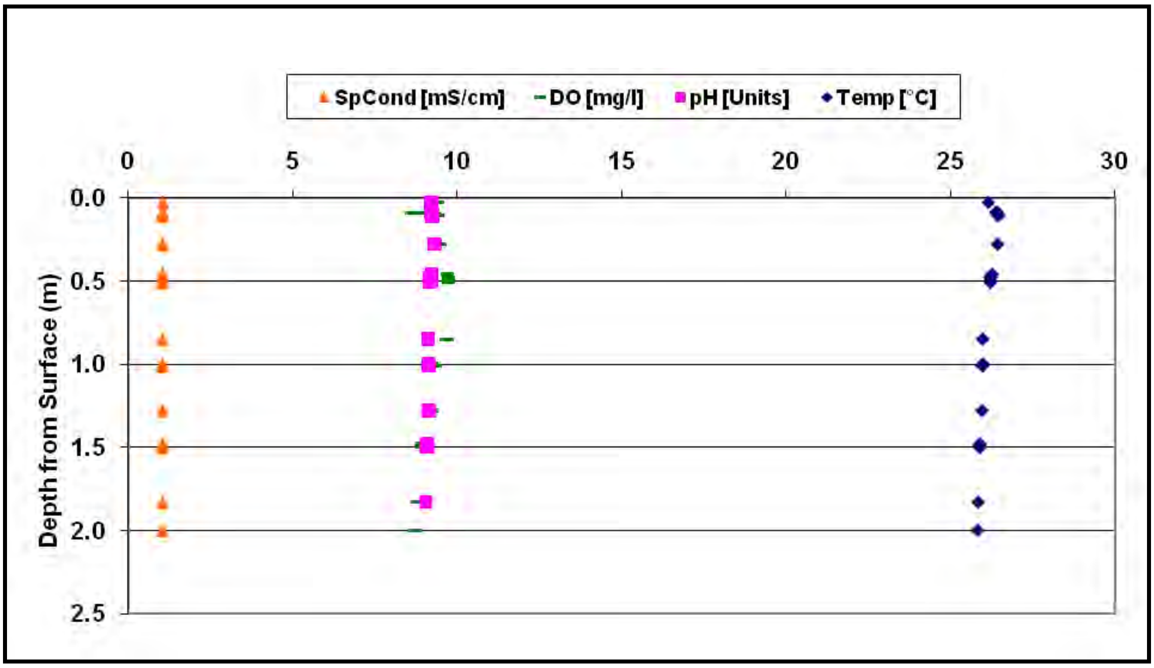


Figure G-16. Profile Data Collected in Lake Calabasas at CL-5 on August 6, 2009

(This page left intentionally blank.)

G.8 Monitoring Data for El Dorado Park Lakes

Monitoring data relevant to the impairments of El Dorado Park lakes are available from 1992, 1993, and 2008 through 2010. In addition, fish tissue data are available for 1991, 1992, 1998, and 2007. Figure G-17 shows the historical and recent monitoring locations for El Dorado Park lakes.

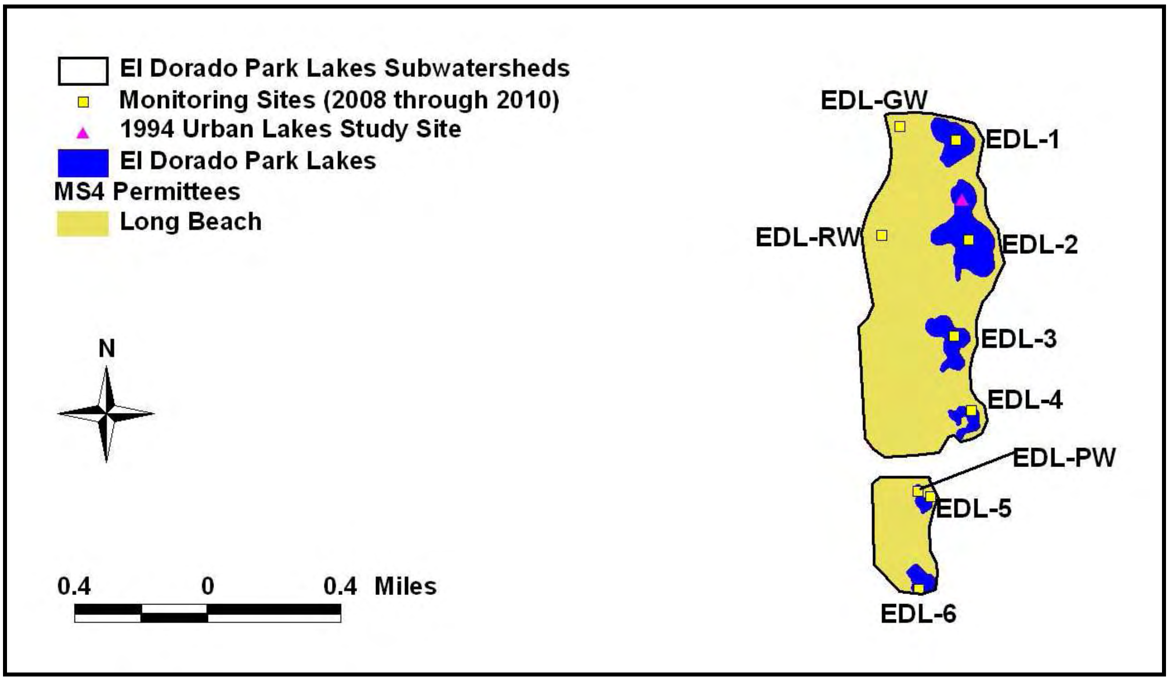


Figure G-17. Monitoring Sites in the El Dorado Park Lakes Watershed

G.8.1 MONITORING RELATED TO NUTRIENT IMPAIRMENTS

The El Dorado Parks lakes were included in the 1992/1993 sampling effort to support the Urban Lakes Study (Table G-73). Data were collected from the north end of Lake 2 shown in Figure G-17 (pink triangle). TKN concentrations ranged from 1.2 mg/L to 4.2 mg/L. Nineteen of 45 samples for ammonium were less than the reporting limit, and the maximum concentration observed was 1.9 mg/L, the upper end of this range is below the acute target, but above the chronic target (for assessment purposes, we are assuming the analysis methodology converted all ammonia to ammonium). Nitrite samples were consistently less than the reporting limit, as were the majority of nitrate concentrations. Measurable amounts of nitrate were only observed in January and February of 1993 when concentrations ranged from 0.1 mg/L to 0.3 mg/L. Orthophosphate concentrations ranged from 0.2 mg/L to 0.9 mg/L, and total phosphorus ranged from 0.3 mg/L to 1.1 mg/L. pH ranged from 8.2 to 9.4, and TOC ranged from 7.1 mg/L to 10.7 mg/L. The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from 5 μg/L to 133 μg/L with an average of 48 μg/L.

Although the 1996 Water Quality Assessment Database does not contain monitoring data for the El Dorado Park lakes, the summary table in the Report does include a synopsis of monitoring data and related impairments. pH was listed as partially supporting the aquatic life use and not supporting the contact recreation use: 116 measurements of pH were collected with values ranging from 6.9 to 9.4. Ammonium was not supporting the aquatic life or contact recreation uses; 45 ammonium samples were collected with concentrations ranging from non-detect to 1.92 mg/L , the upper end of this range is below the acute target, but above the chronic target (for assessment purposes, we are assuming the analysis methodology converted all ammonia to ammonium). Raw data are not available to assess location, date, time, depth, temperature, or pH with regard to these samples. Algae were listed as not supporting the contact and non-contact recreation uses. Eutrophication was listed as not supporting the aquatic life use.

Table G-73. El Dorado Park Lakes 1992/1993 Monitoring Data

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
7/16/1992	0	1.5	0.2	<0.01	<0.01	0.2	0.3	9.2	9.6	461
	3	1.4	<0.01	<0.01	<0.01	0.2	0.4	9.2	10.7	459
	4.5	4.2	1.9	<0.01	<0.01	0.9	1.1	8.4	10.3	470
7/16/1992	0	1.6	<0.01	<0.01	<0.01	0.2	0.3	9.3	10.3	459
	3	1.5	<0.01	<0.01	<0.01	0.2	0.4	9.2	10	471
	4.5	1.9	<0.01	<0.01	<0.01	0.3	0.5	9.2	9.3	476
7/16/1992	0	1.5	<0.01	<0.01	<0.01	0.2	0.3	9.1	9.7	488
	3	1.5	0.1	<0.01	<0.01	0.2	0.3	9.2	9.9	449
	4.5	1.7	<0.01	<0.01	<0.01	0.3	0.5	9.2	9.9	475
8/20/1992	0	1.55	<0.01	<0.01	<0.01	0.2	0.3	9.3	10.3	461
	2	1.6	<0.01	<0.01	<0.01	0.2	0.3	9.4	10.4	475
	4	2.3	0.5	<0.01	<0.01	0.3	0.4	9.2	10.7	466
9/24/1992	0	1.5	<0.01	<0.01	<0.01	0.2	0.3	9.3	9.9	442
	2	1.4	<0.01	<0.01	<0.01	0.3	0.3	9.3	9.6	443
	4	1.8	0.3	<0.01	<0.01	0.3	0.4	8.9	9.4	445
10/20/1992	0	2.4	0.3	<0.01	<0.01	0.3	0.5	9.2	10.2	435
	2	2	0.3	<0.01	<0.01	0.4	0.5	9.1	10.1	474
	3.5	2.3	0.3	<0.01	<0.01	0.4	0.5	9.1	10.5	493
11/12/1992	0	2.3	0.5	<0.01	<0.01	0.4	0.5	9	9.9	450
	2.5	2.4	0.4	<0.01	<0.01	0.4	0.5	9	10.3	450
	3.5	2.1	0.5	<0.01	<0.01	0.4	0.4	9.1	9.4	450
12/15/1992	0	3.4	1.5	<0.01	<0.01	0.6	0.5	8.5	8.7	449
	2.5	3.3	1.5	<0.01	<0.01	0.6	0.5	8.5	10.1	452
	3.5	NA	1.5	<0.01	<0.01	0.6	0.6	8.5	8.5	451
1/21/1993	0	2.4	1.1	<0.01	0.3	0.4	0.5	8.3	7.4	380
	2.5	2.4	1.1	<0.01	0.3	0.4	0.4	8.3	7.2	417
	3.5	2.3	1.2	<0.01	0.3	0.4	0.4	8.3	7.1	417
2/10/1993	0	2	<0.01	<0.01	0.2	0.3	0.5	8.9	8.3	407

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
	2.5	2.5	<0.01	<0.01	0.1	0.3	0.4	8.9	8.4	496
	3.5	2.1	<0.01	<0.01	0.2	0.3	0.4	8.9	7.7	439
3/8/1993	0	3.7	0.1	<0.01	<0.01	0.3	0.4	9.4	9.6	419
	1.5	4	<0.01	<0.01	<0.01	0.2	0.4	8.8	9.6	423
	2.5	2.2	0.1	<0.01	<0.01	0.2	0.4	8.6	9.4	413
	3.5	1.5	0.1	<0.01	<0.01	0.3	0.4	8.2	7.7	407
4/8/1993	0	1.4	<0.01	<0.01	<0.01	0.3	0.3	9	7.8	413
	1.5	1.3	<0.01	<0.01	<0.01	0.3	0.3	9	7.8	431
	2.5	1.2	<0.01	<0.01	<0.01	0.3	0.3	9	7.6	429
	3.5	1.5	0.3	<0.01	<0.01	0.4	0.3	8.6	7.4	412
5/12/1993	0	1.3	<0.01	<0.01	<0.01	0.3	0.3	8.8	8	459
	2.5	1.2	<0.01	<0.01	<0.01	0.3	0.3	8.8	7.6	460
	3.5	1.5	0.1	<0.01	<0.01	0.3	0.3	8.8	8.2	450
6/15/1993	0	1.9	0.3	<0.01	<0.01	0.4	0.4	9.2	10	468
	1.5	2	0.3	<0.01	<0.01	0.4	0.4	9.2	9.5	487
	2.5	2.1	0.3	<0.01	<0.01	0.4	0.3	9.2	9.1	478
	3.5	1.6	0.3	<0.01	<0.01	0.4	0.4	9	8.6	465

In May 2008, Marine Biochemists sampled water quality in the El Dorado Park lakes system. The data report does not specify who sponsored the sampling. On May 8, 2008 water quality data were collected in the upper four lakes (Table G-74). DO concentrations ranged from 7.36 mg/L to 8.63 mg/L, and pH ranged from 7.37 to 8.76. Temperature was fairly consistent in all four lakes and was approximately 69 ºF. The concentrations of nitrates were highly variable and ranged from 0.3 mg/L to 3.0 mg/L; phosphates ranged from 0.09 mg/L to 0.58 mg/L. It is not clear from the report if the units on the nitrate samples were "as N" or "as NO_3" or if the units on the phosphate samples were "as P" or "as PO_4." Sampling depth and time and analysis methodologies were not included with the hard copy data report Tetra Tech received, nor were sampling locations.

Table G-74. May 2008 Water Quality Monitoring Data for El Dorado Park Lakes

Lake Number	Dissolved Oxygen (mg/L)	pH	Temperature (ºF)	Nitrates (mg/L)	Phosphates (mg/L)
1	8.63	7.37	69.09	3.0	0.09
2	7.76	8.76	68.84	0.9	0.13
3	7.36	7.94	69.11	0.3	0.19
4	7.90	8.32	69.42	1.5	0.58

The El Dorado Park lakes were sampled February 26, 2009 and July 15, 2009 by USEPA and the Regional Board. The field notes from the event indicate that the top four lakes are supplied primarily by groundwater. Water flows from Lake 1 to 2 to 3 to 4; excess water is pumped out of Lake 4 and discharged to a storm drain. Lakes 5 and 6 are not naturally or artificially connected to Lakes 1 through 4; they are connected to each other. Water is supplied to these two lakes by a pipe that continuously discharges potable water to Lake 5 (Valentina Cabrera-Stagno, USEPA Region IX, personal communication, February 3, 2009). Lakes 1, 2, 5, and 6 were sampled in February and Lakes 1, 2, 3, 4, and 6 were sampled during the July monitoring event (nutrients were not analyzed at Lake 4). Lakes 4, 5, and 6 were treated with algaecides in mid-June (personal communication, Ed Gahafer, July 15, 2009), which may have reduced chlorophyll *a* concentrations during the July sampling event.

Table G-75 presents the in-lake water quality measurements for the February and July 2009 sampling events. During the February event, Lakes 1 and 2 were sampled from a depth of 0.76 meters and the total depth at each station was approximately 4.4 meters. The Secchi depth in Lake 1 was 1.31 meters and in Lake 2 was 1.37 meters. Lake 5 was sampled from a depth of 0.46 meters and the total depth at the sampling location was approximately 4.5 meters; Secchi depth was not measured at this site. Lake 6 was sampled from a depth of 0.92 meters and the total depth at the sampling location was 2.5 meters; the Secchi depth was 1.83 meters. The main source of water to Lakes 1 through 4, water pumped from groundwater, was also sampled. These data are also included for comparison as the nutrient loading from this source may be significant relevant to the upland sources represented by the LSPC/EMC approach (Appendix D).

During the July event, Lake 1 was sampled from a depth of 0.58 meters. The total depth at the Lake 1 station was 3.5 meters, and the Secchi depth was 1.17 meters. Lake 2 was sampled at 0.33 meters below the surface. The Secchi depth at Lake 2 was 0.69 meters and the total lake depth was 3.9 meters. Lake 6 was sampled at a depth of 0.97 meters. The total depth of Lake 6, as measured by the sampling probe, was 2.2 meters. Samples take at Lake 5 were approximately 0.46 meters below the surface. The total depth and Secchi depth were not measured at Lake 5 during this monitoring event. The Secchi depth reading at Lake 6 was 1.96 meters. The groundwater and potable water were also sampled during this event. The results of these efforts are shown in Table G-75. Temperature, DO, pH, and conductivity were not measured during either of the monitoring events.

Table G-75. 2009 In-lake Water Column Measurements for the El Dorado Park Lakes

Date	Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	Chl *a* (µg/L)	Secchi Depth (m)
2/26/2009	EDL-1	8:30	1.8	<0.03	<0.01	<0.01	0.028	0.160	48.7	1.31
	EDL-2	11:15	2.1	0.03	<0.01	<0.01	0.016	0.094	19.2	1.37
	EDL-2 (dup.)	11:30	2.2	<0.03	<0.01	<0.01	0.016	0.102	18.7	1.37
	EDL-5	12:15	1.1	<0.03	<0.01	<0.01	0.015	0.030	5.3	NA
	EDL-6	13:20	1.1	0.03	<0.01	<0.01	0.016	0.031	5.9	1.83
	EDL-GW[1]	9:40	0.84	0.33	<0.01	<0.01	0.074	0.190	NA	NA
7/15/2009	EDL-1	11:15	0.91	<0.03	<0.01	<0.01	<0.0075	0.047	22.9	1.17
	EDL-2	9:40	1.0	<0.03	<0.01	<0.01	<0.0075	0.1605	39.38	0.69
	EDL-2D	9:40	0.84	<0.03	NS	NS	NS	0.151	NS	0.69

Date	Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	Chl *a* (µg/L)	Secchi Depth (m)
	EDL-5	15:10	NS	0.1	<0.01	0.12	<0.0075	0.139	1.3	NA
	EDL-6	15:50	0.98	0.04	<0.01	0.09	<0.0075	0.138	6.2	1.96
	EDL-GW[1]	11:30	1.1	0.28	<0.01	<0.01	0.07095	0.291	NA	NA
	EDL-PW[1]	14:40	0.84	0.365	<0.01	0.37	<0.0075	0.1085	NA	NA

[1] EDL-GW represents the groundwater input to lakes 1 through 4 and EDL-PW represents the potable water input to Lake 5. These are not in-lake samples.

Additional water quality samples were collected from the El Dorado Park lakes. Table G-76 presents the chloride, sulfate, total alkalinity, total dissolved solids, and total organic carbon data collected from Lakes 1, 2, 5, and 6 during both monitoring events. Duplicate samples were not collected at Lake 2 in the July 2009 monitoring. Lakes 3 and 4 were only measured for hardness in July. The July solids monitoring data were reported percent solids for EDL-1, EDL-2, and EDL-6, while EDL-5 had TSS data reported in mg/L. Again, measurements collected from the groundwater source and potable water are included for comparison to in-lake samples.

Table G-76. Supplemental Water Quality Monitoring for In-lake Samples in the El Dorado Park Lakes

Date	Location	Time	Chloride (mg/L)	Sulfate (mg/L)	Total Alka-linity (mg/L)	Hardness (mg/L as $CaCO_3$)	TDS (mg/L)	TSS (mg/L)	DOC (mg/L)	TOC (mg/L)
2/26/2009	EDL-1	8:30	33.70	25.04	203	100.2	380	4.6	5.2	4.1
	EDL-2	11:15	80.92	30.02	274	113.4	532	4.1	9.3	7.1
	EDL-5	12:15	55.65	88.82	126	NA	406	2.6	5.0	3.9
	EDL-6	13:20	56.07	88.83	126	NA	370	3.5	5.0	6.2
	EDL-GW[1]	9:40	18.20	22.10	194	131.4	304	0.7	0.7	0.4
7/15/2009	EDL 1	11:15	25.56	20.47	210	117.3	350	24 (%)	4.3	4.8
	EDL 2	09:40	80.505	33.28	274	122.5	532	24 (%)	9.2	11.3
	EDL 2D	09:40	81.64	33.71	280	NS	NS	NS	9.1	NS
	EDL 3	13:35	NS	NS	NS	88.1	NS	NS	NS	NS
	EDL 4	14:30	NS	NS	NS	87.4	NS	NS	NS	NS
	EDL 5	15:10	57.77	82.7	120	85.6	388	2.15	3.7	4.1
	EDL 6	15:50	59.25	87.6	118	84.6	400	12.5 (%)	9.9	4.8
	EDL GW[1]	11:30	15.6	20.2	210	155.9	356	NS	0.2	0.4
	EDL PW[1]	14:40	51.53	52.245	116	81.65	345	NS	1.4	1.3

[1] EDL-GW represents the groundwater input to lakes 1 through 4 and EDL-PW represents the potable water input to Lake 5. These are not in-lake samples.

Profile data were also collected for specific conductivity, dissolved oxygen, pH, and temperature in each of the monitored lakes. Figure G-18 through Figure G-21 show the profile data collected on February 26, 2009 at Stations EDL-1, EDL-2, EDL-5, and EDL-6, respectively. Specific conductivity is constant with depth at each station. DO decreases from 5.1 mg/L to 8.7 mg/L near the surface to approximately 3.5 mg/L at the bottom of each lake. pH ranges from 7.3 to 8.4, and temperature ranges from 13.8 °C to 17.6 °C at each station.

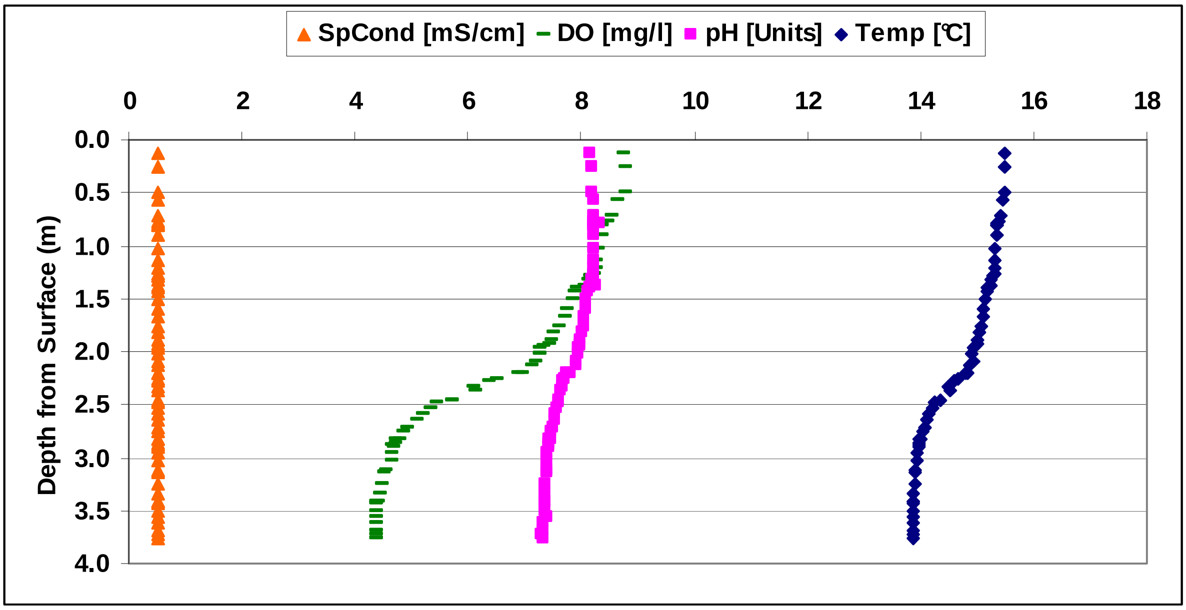


Figure G-18. Profile Data Collected in El Dorado Park Lake 1 on February 26, 2009

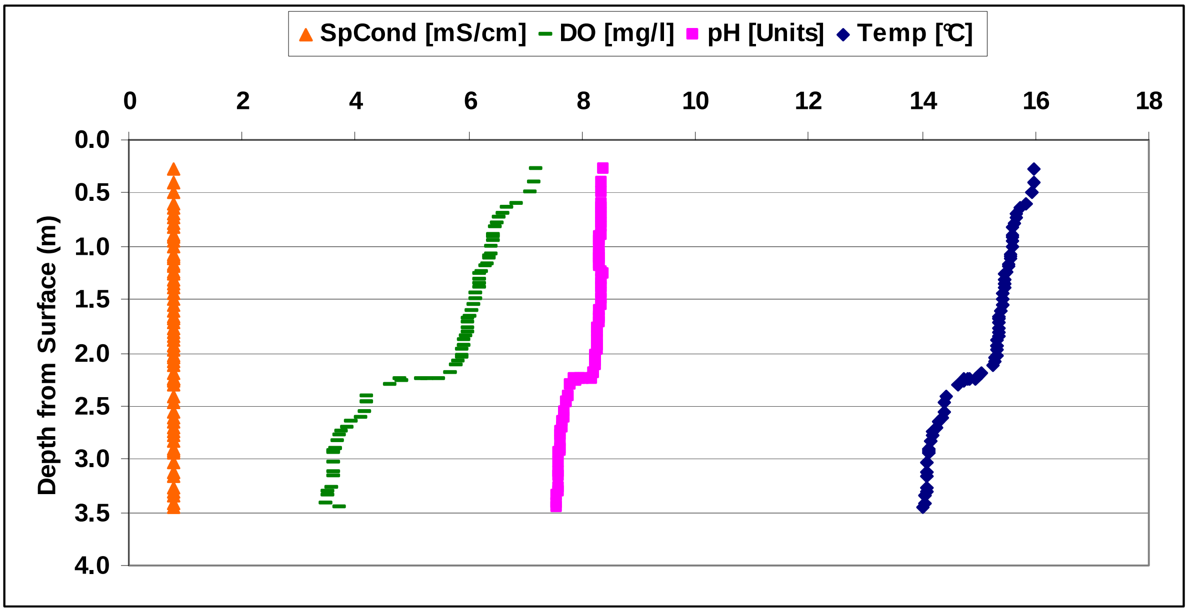


Figure G-19. Profile Data Collected in El Dorado Park Lake 2 on February 26, 2009

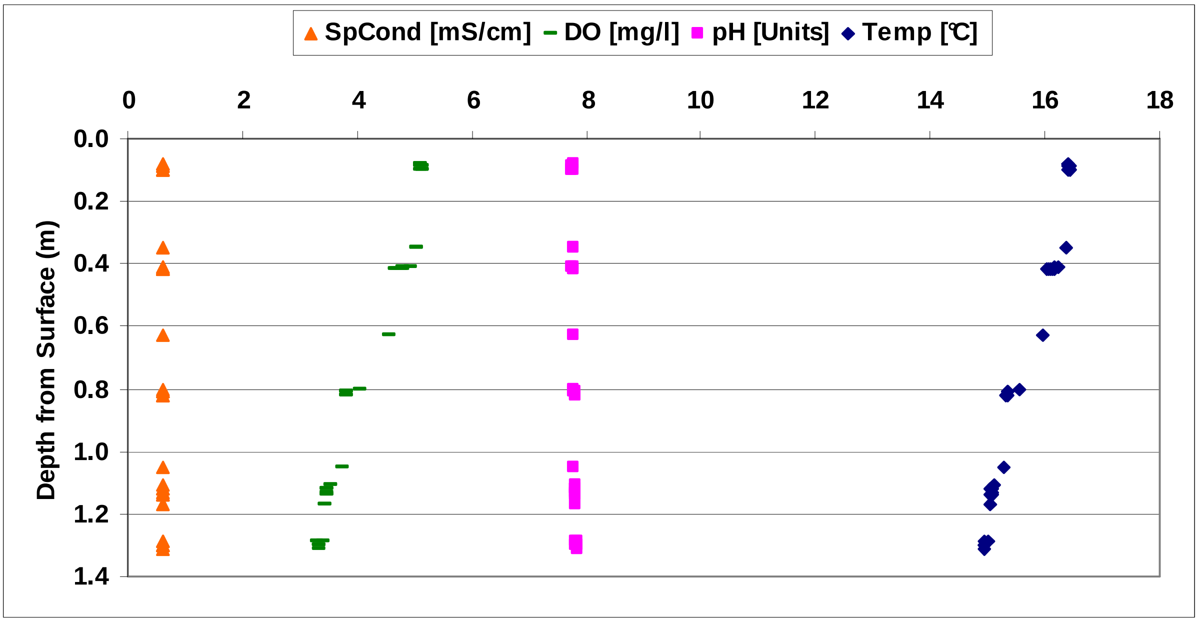


Figure G-20. Profile Data Collected in El Dorado Park Lake 5 on February 26, 2009

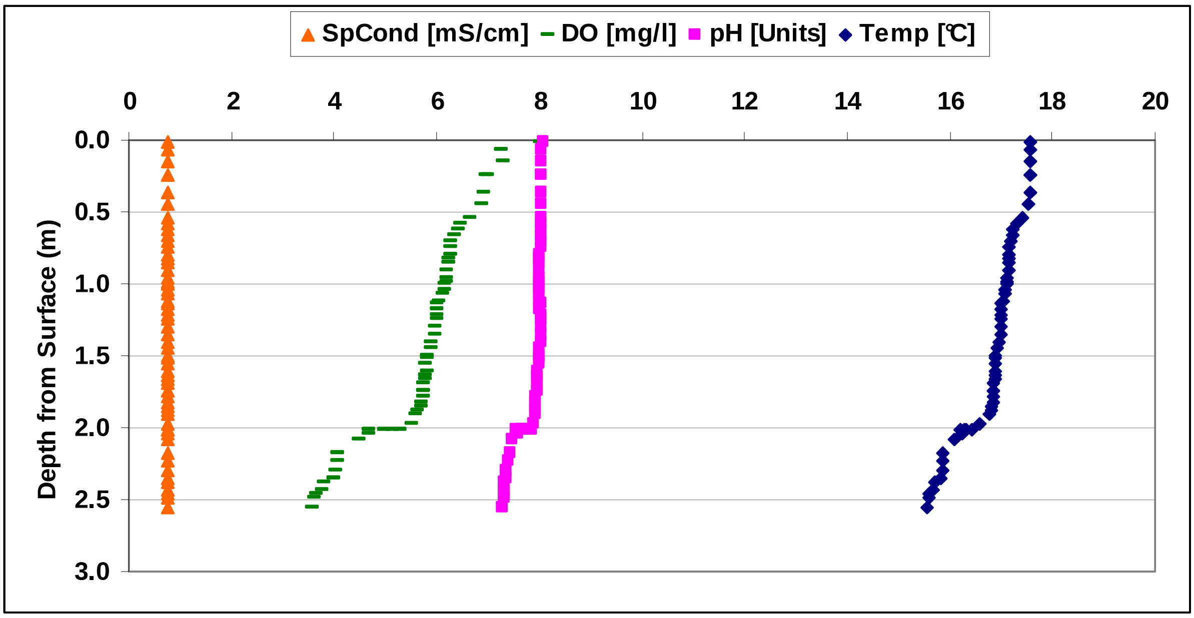


Figure G-21. Profile Data Collected in El Dorado Park Lake 6 on February 26, 2009

Profile data were also collected on July 15, 2009 for Stations EDL-1, EDL-2, and EDL-6. Summer temperatures range from 23.6 to 30.2 ºC at each station. The summer pH range is similar to the winter pH range, from 7.2 to 8.4. The DO ranges from 1.65 mg/L near the bottom of the lakes and up to 9.57 mg/L near the surface of the lakes. Specific conductivity is constant with depth at each station. The July profile data are displayed in Figure G-22 through Figure G-24.

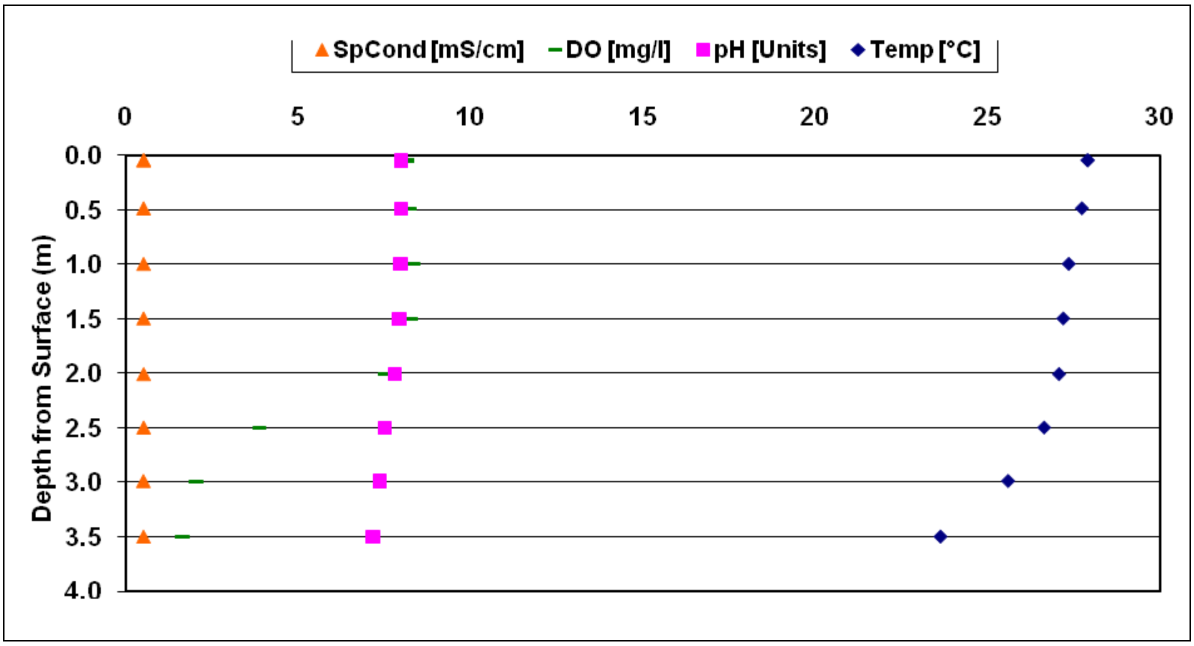


Figure G-22. Profile Data Collected in El Dorado Park Lake 1 on July 15, 2009

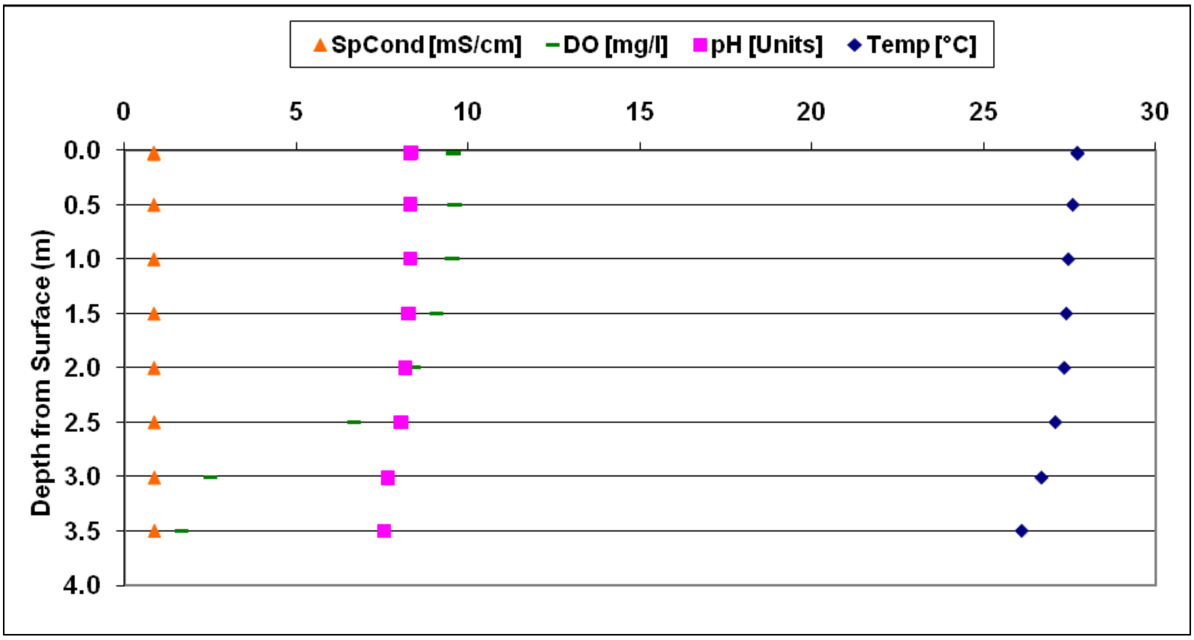


Figure G-23. Profile Data Collected in El Dorado Park Lake 2 on July 15, 2009

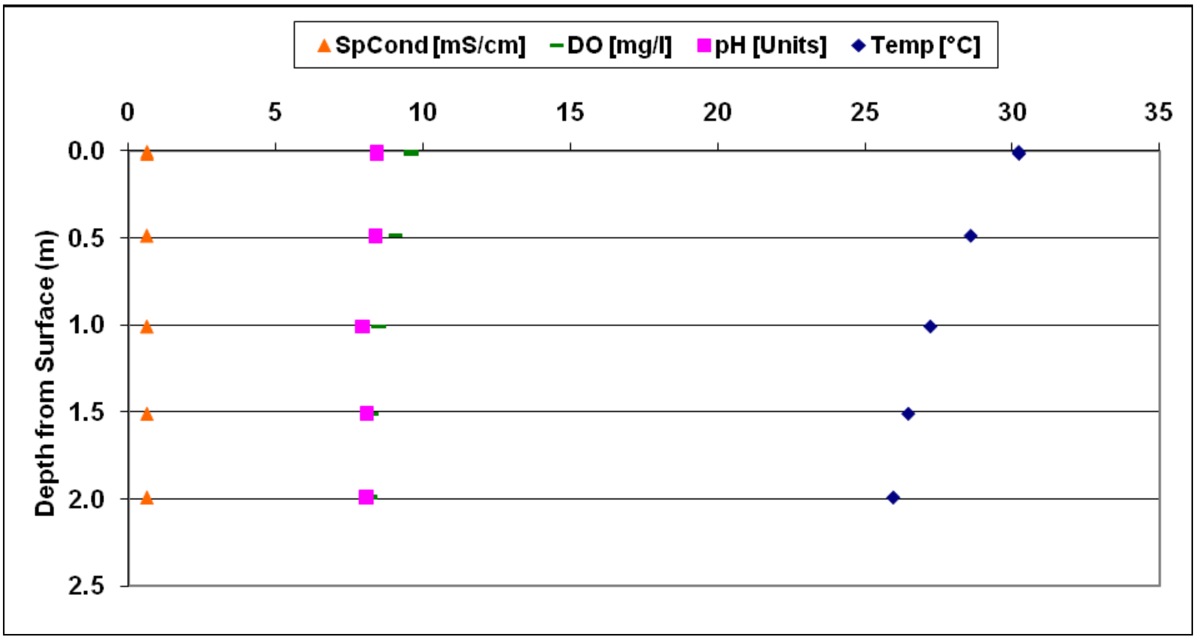


Figure G-24. Profile Data Collected in El Dorado Park Lake 6 on July 15, 2009

Field data were also collected at the potable water source at El Dorado Park during the July sampling event. At 14:40, the temperature was 27 ºC (pH measurements were not taken with a faulty meter and were not considered reliable).

Reclaimed water, used as irrigation on land surrounding the lake, was also sampled on December 1, 2009. Table G-77 presents the December 1, 2009 sampling results collected at EDLRW and ELDRWD (duplicate for ELDRW). In general, total phosphorus averaged 0.166 mg-P/L, and total nitrogen averaged 5.74 mg-N/L. EDLRW was also monitored for chloride, sulfate, alkalinity, hardness, total dissolved solids, dissolved organic carbon, and total organic carbon; results are presented in Table G-78.

Table G-77. Reclaimed Water Measurements for the El Dorado Park Lakes

Date	Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)
12/1/2009	EDLRW	14:30	1.30	0.61	0.05	4.04	0.07	0.164
	EDLRWD (dup.)	14:30	1.15	0.63	0.05	4.9	0.10	0.168

Table G-78. Supplemental Water Quality Monitoring for Reclaimed Water at the El Dorado Park Lakes

Date	Location	Time	Chloride (mg/L)	Sulfate (mg/L)	Total Alka-linity (mg/L)	Hardness (mg/L as $CaCO_3$)	TDS (mg/L)	DOC (mg/L)	TOC (mg/L)
12/1/2009	EDLRW	14:30	110.53	79.62	198	133.1	583	5.8	5.7

Field data were also collected at shoreline stations in El Dorado Park during the December 1, 2009 sampling event. At EDL-2S (total depth 1 foot) at 1:45 p.m., the temperature was 17.01 ºC and the pH was 8.31. EDL-1S was sampled at 2:15 pm (total depth 2 feet) with a temperature of 16.92 ºC and a pH of 8.23. Temperature at 3:40 p.m. at EDL-6S (total depth 1.5 feet) was 15.34 ºC and the pH was 8.12, while temperature was 14.94 ºC and pH was 8.17 at EDL-5S (total depth 2 feet) about 15 minutes later. The last two sites (EDL-4S [total depth 1 foot] and EDL-3S [total depth 2 feet]) both had a pH reading of 9.20 at 4:10 p.m. and 4:20 p.m., respectively. Temperatures at these sites were 15.92 ºC and 14.71 ºC, respectively.

The southern two lakes at El Dorado Park were resampled for nutrients on August 10, 2010. Table G-79 summarizes the nutrient data collected in each lake as well as the potable water source. TKN concentrations ranged from 0.67 to 1.03 mg-N/L. Ammonia concentrations ranged from 0.03 mg-N/L to 0.05 mg-N/L. Nitrite was approximately 0.05 mg-N/L in both lakes, and nitrate ranged from 0.23 mg-N/L to 0.24 mg-N/L. Orthophosphate ranged from 0.022 mg-P/L to 0.027 mg-P/L, and total phosphorus ranged from 0.027 mg-P/L to 0.038 mg-P/L. Chlorophyll *a* ranged from 4.81 µg/L to 6.23 µg/L.

Table G-79. August 10, 2010 In-lake Water Column Measurements for the Nature Center Lakes at El Dorado Park

Location	**Time**	**TKN (mg/L)**	**NH_3-N (mg/L)**	**NO_2-N (mg/L)**	**NO_3-N (mg/L)**	**PO_4-P (mg/L)**	**Total P (mg/L)**	**Chl *a* (µg/L)**	**Secchi Depth (m)**
EDL-5	13:00	1.03	0.0493	0.051	0.244	0.027	0.038	6.23	>1.25
EDL-6	15:00	0.67	0.0328	0.052	0.233	0.022	0.0271	4.81	1.5
EDL-PW	13:40	0.48	0.0359	0.054	0.173	0.026	<0.0165	<1.2	NA

During the August 2010 event, two continuous monitoring probes were deployed in each southern lake over a 24-hour period at depths of about 0.7 to 1.3 meters below the surface. DO concentrations ranged from 8.3 mg/L to 9.5 mg/L in Nature Center North Lake (Figure G-25) and from 9.5 mg/L to 12.6 mg/L in Nature Center South Lake (Figure G-26). pH ranged from 8.5 to 9.0 in both lakes.

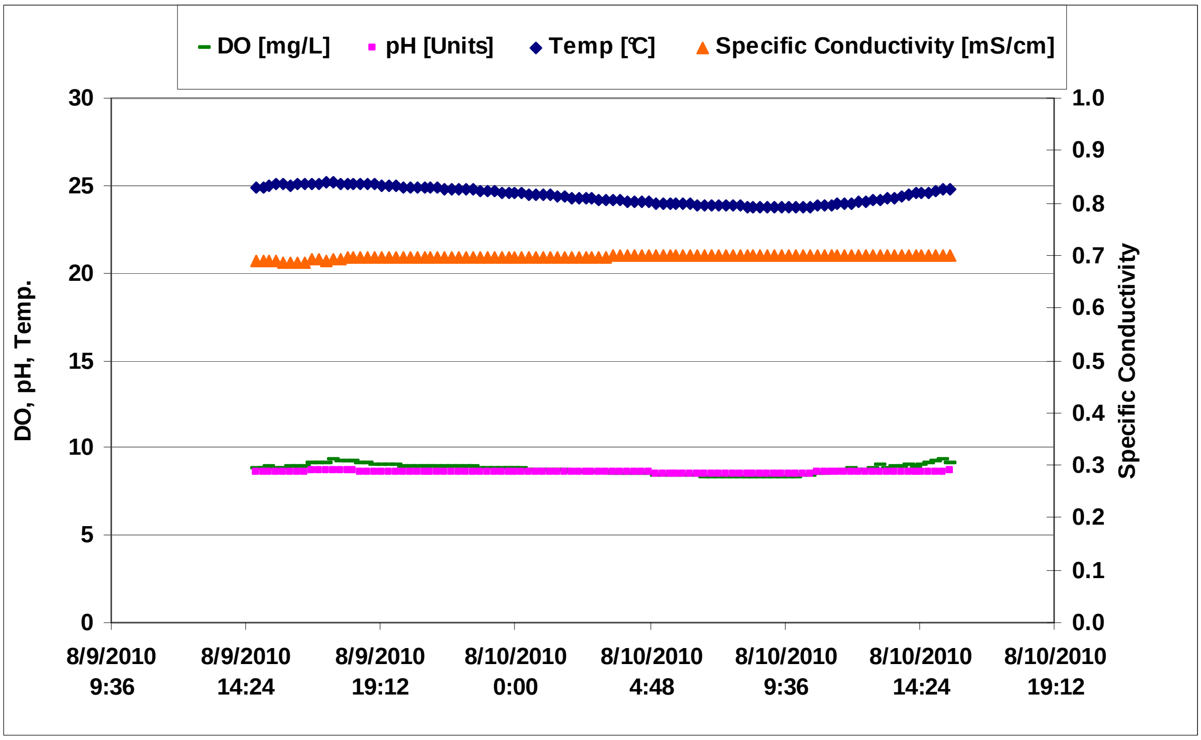


Figure G-25. 24-Hour Probe Data Collected in El Dorado Park Lake 5 on August 9, 2010

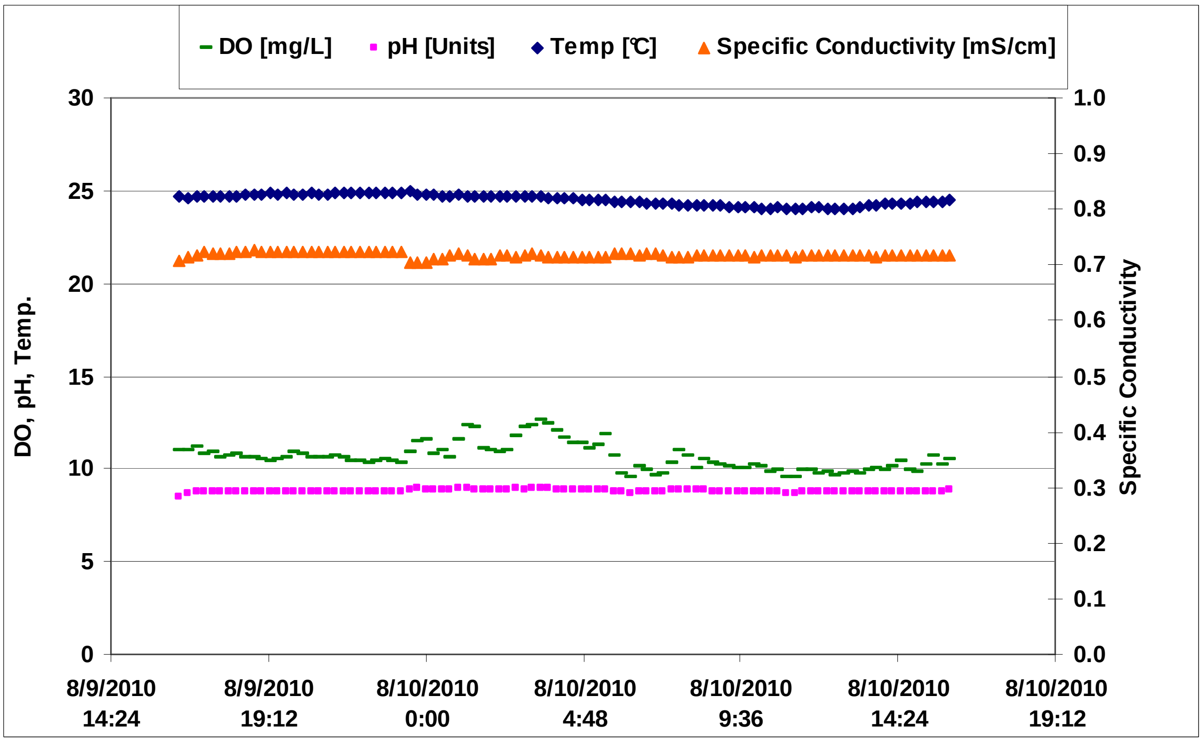


Figure G-26. 24-Hour Probe Data Collected in El Dorado Park Lake 6 on August 9, 2010

On August 10, 2010, depth-profile data were also collected during this water sampling event. Table G-80 summarizes the depth-profile data collected at ELD-5 and ELD-6. DO measurements collected from the surface to 0.3 meters above the bottom of Nature Center North Lake ranged from 8.4 mg/L to 8.5 mg/L. In Nature Center South Lake, DO ranged from 11.8 mg/L at the surface to 9.9 mg/L at 0.3 meters above the bottom of the lake. Figure G-27 and Figure G-28 show the profile data collected on August 10, 2010, 2010 at stations ELD-5 and ELD-6 respectively.

Table G-80. Profile Data Collected in the Nature Center Lakes at El Dorado Park on August 10, 2010

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Specific Conductivity (mS/cm)	Orp (mV)
EDL-5	12:52	0.1	24.48	8.59	8.40	0.707	162
		0.55	24.16	8.53	8.44	0.706	164
		1.02	24.04	8.57	8.53	0.705	161
EDL-6	14:39	0.11	27.22	8.95	11.78	0.707	279
		0.54	25.26	8.75	11.08	0.713	268
		1.04	24.63	8.60	10.28	0.715	265
		1.49	24.24	8.55	9.89	0.713	262
		2.03	23.57	8.60	9.96	0.712	259

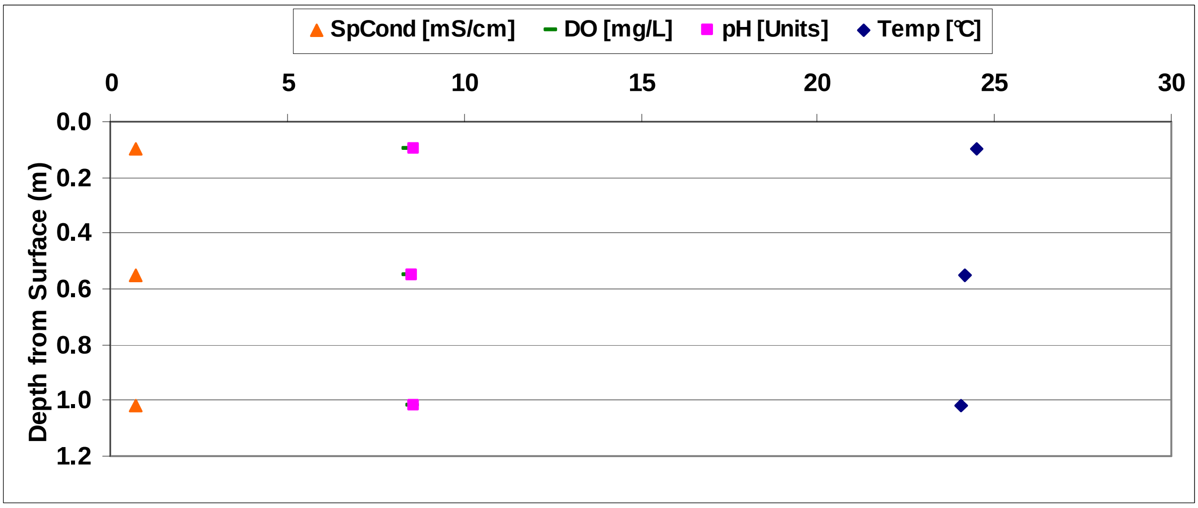


Figure G-27. Profile Data Collected in El Dorado Park Lake 5 on August 10, 2010

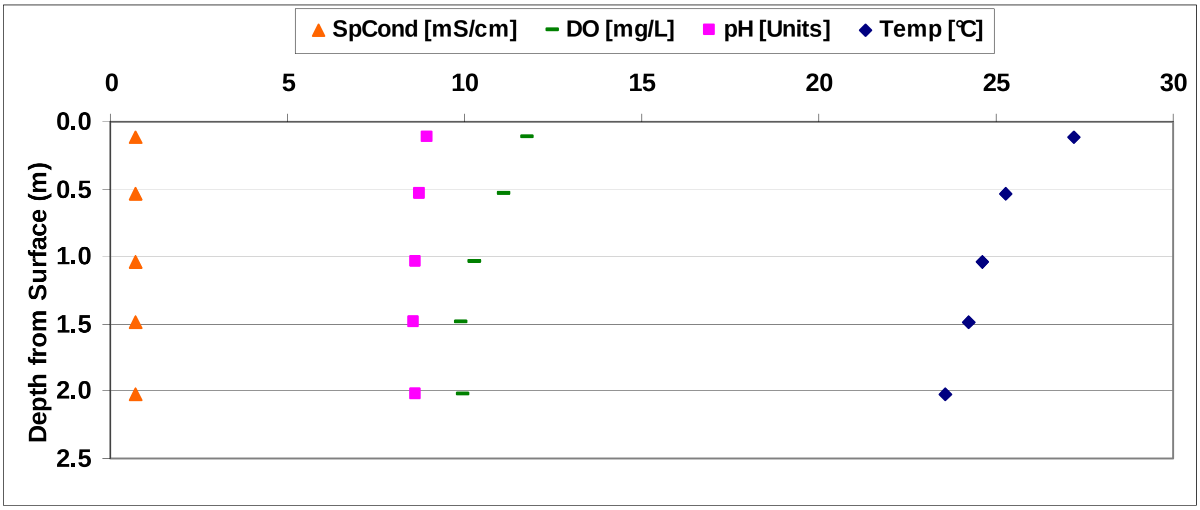


Figure G-28. Profile Data Collected in El Dorado Park Lake 6 on August 10, 2010

Sediment samples were also collected during the August 2010 monitoring event. Table G-81 summarizes these data.

Table G-81. August 10, 2010 Sediment Monitoring Data for the Nature Center Lakes (5 and 6) at El Dorado Park

Location	Time	TKN (mg/kg)	NH_3-N (mg/kg)	NO_2-N (mg/kg)	NO_3-N (mg/kg)	PO_4-P (mg/kg)	Total P (mg/kg)	Total Organic Carbon (% by wt.)	Acid Volatile Sulfides (mg/kg)	Percent Solids	Total Hardness (mg/kg)
EDL-5	13:20	2,570	15.5	1.54	2.81	1.56	1,210	7.34	5.33	19.8	12,900
EDL-6	15:15	4,950	41.9	2.71	4.86	1.73	1,050	11.0	3.63	12.5	9,370

In addition to the August 2010 sample, the southern two lakes at El Dorado Park were resampled for nutrients on September 28, 2010. Table G-82 summarizes the nutrient data collected in each lake as well as the potable water source. TKN concentrations ranged from 0.79 to 0.86 mg-N/L. Ammonia concentrations ranged from <0.03 mg-N/L to 0.05 mg-N/L. Nitrite was approximately 0.05 mg-N/L in both lakes, and nitrate ranged from 0.36 mg-N/L to 0.41 mg-N/L. Orthophosphate ranged from 0.008 mg-P/L to 0.017 mg-P/L. Total phosphorus was measured as below the detection limit of 0.0165 mg-P/L in both lakes. Chlorophyll *a* ranged from 6.01 µg/L to 6.68 µg/L.

Table G-82. August 9, 2010 In-lake Water Column Measurements for the Nature Center Lakes (5 and 6) at El Dorado Park

Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	Chl *a* (µg/L)	Secchi Depth (m)
EDL-5A	15:10	0.864	0.0475	0.0470	0.378	0.0100	<0.0165	6.68	1.6
EDL-5A (duplicate)	15:10	0.808	0.0490	0.0480	0.364	0.0170	<0.0165	6.01	1.6
EDL-6	15:10	0.792	<0.0300	0.0540	0.409	0.00800	<0.0165	6.01	1.4
EDL-PW	12:45	0.672	0.292	0.0600	0.173	0.00900	<0.0165	<1.00	NA

Similar to the August 2010 event, two continuous monitoring probes were deployed September 27, 2010 in each southern lake over a 24-hour period at depths of about 1 to 1.3 meters below the surface. DO concentrations ranged from 7.4 mg/L to 8.2 mg/L in Nature Center North Lake (Figure G-29) and from 6.6 mg/L to 9.7 mg/L in Nature Center South Lake (Figure G-30). pH ranged from about 7.6 to 8.1 in both lakes.

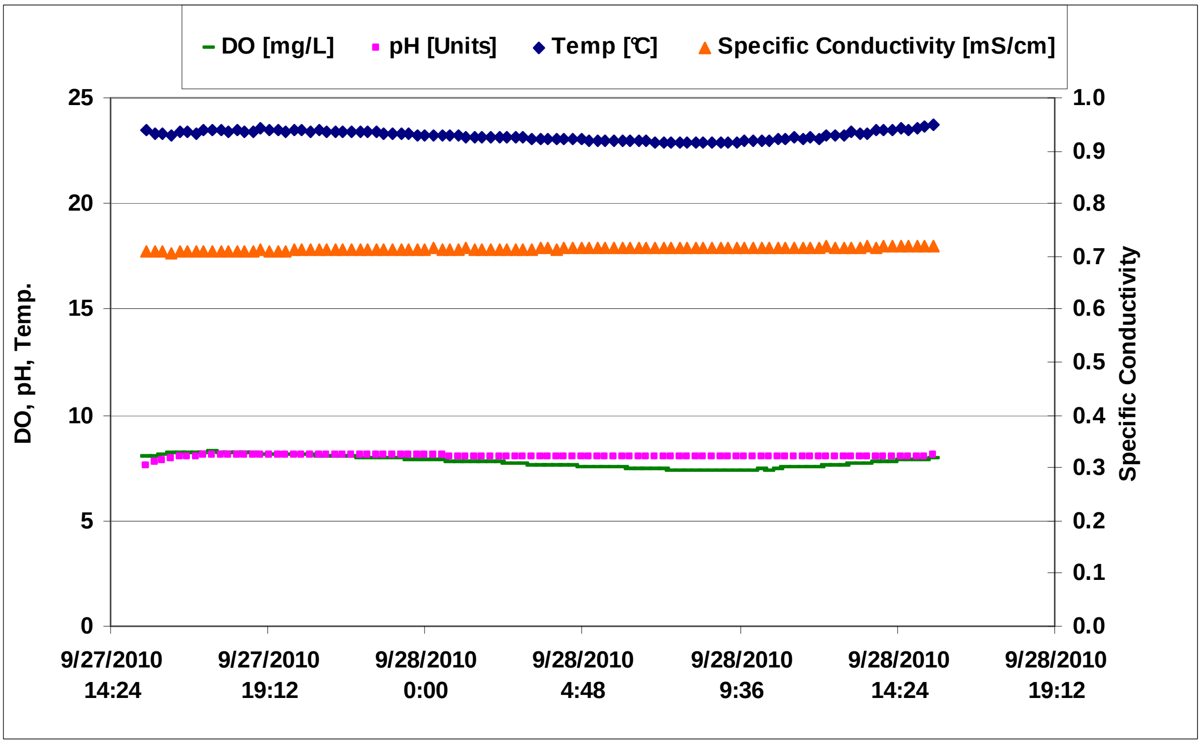


Figure G-29. 24-Hour Probe Data Collected at in El Dorado Park Lake 5 on September 27, 2010

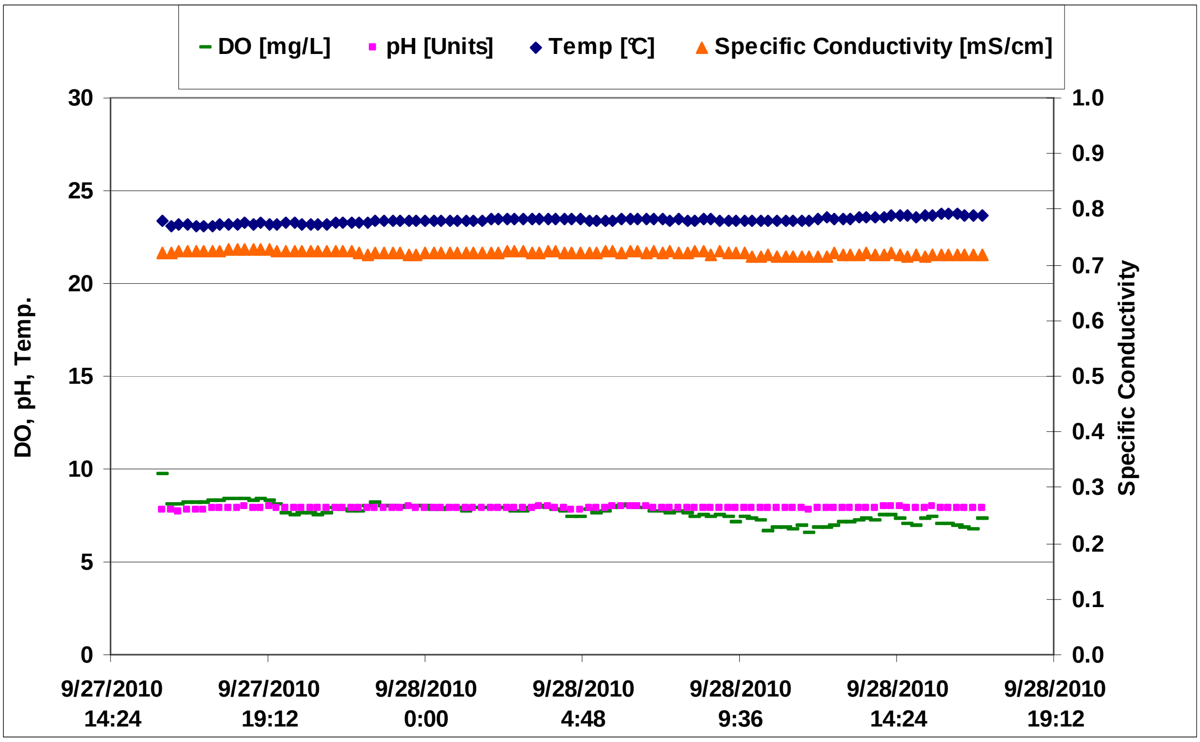


Figure G-30. 24-Hour Probe Data Collected in El Dorado Park Lake 6 on September 27, 2010

On September 28, 2010, depth-profile data were collected for Nature Center North Lake (EDL-5A) during this sampling event, which are summarized in Table G-83. These data were not collected at Nature Center South Lake due to time constraints. DO measurements collected from the surface of Nature Center North Lake ranged from 9.2 mg/L to 10.9 mg/L. At 0.4 meters above the bottom, DO was measured as 9.2 mg/L. Figure G-31 shows the profile data collected on September 28, 2010 at station EDL-5A.

Table G-83. Profile Data Collected in the Nature Center Lakes at El Dorado Park on September 28, 2010

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Specific Conductivity (mS/cm)	Orp (mV)
EDL-5A	15:10	0.5	23.86	7.8	9.21-10.85	0.668	127-104
		1	23.75	7.78	9.17	0.668	99
		1.5	23.82	7.79	9.14	0.668	92
		2	23.76	7.79	9.15	0.669	87
		2.5	23.71	7.78	9.18	0.667	82

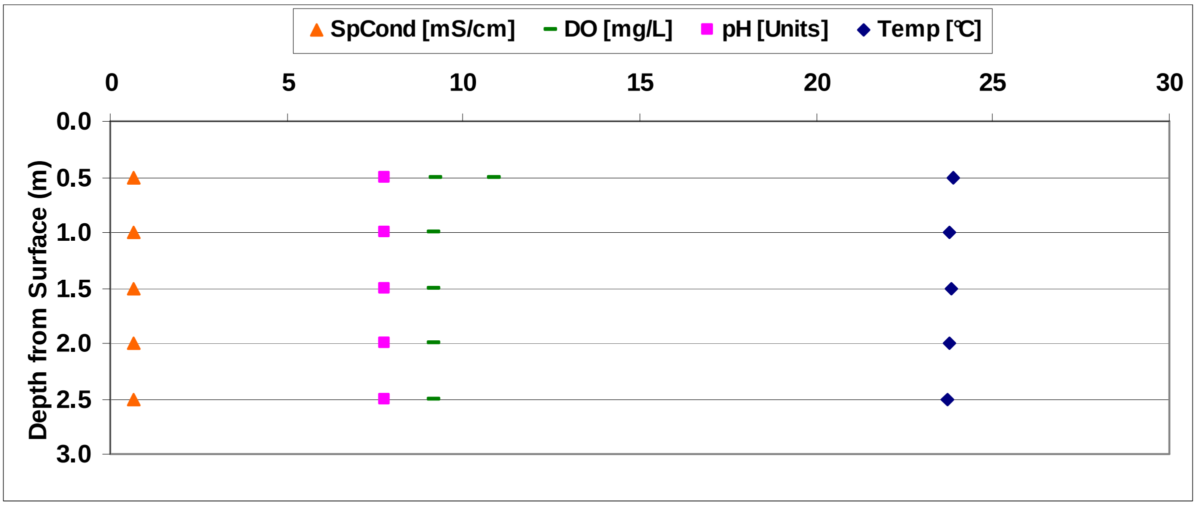


Figure G-31. Profile Data Collected in El Dorado Park Lake 5 on September 28, 2010

Sediment samples were also collected during the September 2010 monitoring event. Table G-84 summarizes these data.

Table G-84. September 28, 2010 Sediment Monitoring Data for the Nature Center Lakes (5 and 6) at El Dorado Park

Location	Time	TKN (mg/kg)	NH_3-N (mg/kg)	NO_2-N (mg/kg)	NO_3-N (mg/kg)	PO_4-P (mg/kg)	Total P (mg/kg)	Total Organic Carbon (% by wt.)	Acid Volatile Sulfides (mg/kg)	Percent Solids	Total Hardness (mg/kg)
EDL-5A	15:50	1,140	5.16	0.922	1.34	0.0264	253	2.00	15.4	43.4	11,600
EDL-5B	14:30	4,840	11.5	2.08	2.89	<0.00750	189	5.06	70.6	19.8	10,800
EDL-5C	14:40	4,530	24.0	1.85	2.64	0.0191	435	4.80	25.2	21.6	11,700
EDL-6	17:10	23,200	37.1	2.98	4.17	0.00891	281	8.60	118	13.4	9,610

G.8.2 MONITORING RELATED TO MERCURY IMPAIRMENT

Mercury data have been collected in the El Dorado Park lakes watershed since 1991. Fish tissue concentrations were measured three times under the Toxic Substances Monitoring Program (TSMP, 2009) from 1991 to 1998 and by the Regional Board in 2007 (Davis et al., 2008) and 2010. In-lake water column concentrations were measured as part of the Urban Lakes Study (UC Riverside, 1994) in 1992. USEPA and the Regional Board sampled in-lake and tributary water column and sediment mercury concentrations during two events in 2009.

G.8.2.1 In-Lake Sampling

G.8.2.1.1 Water Column Measurements

Mercury concentrations were measured in the water column of Lake 2 (pink triangle, Figure G-17) as part of the Urban Lakes Study (UC Riverside, 1994) in July and August of 1992. The detection limit of this dataset was relatively high (500 ng/L) and all 12 samples were less than detection.

In February 2009, the Regional Board and USEPA sampled mercury concentrations at Stations EDL-1 and EDL-2. Both samples were collected from a depth of 0.76 meters and the total depth at each location was approximately 4.4 meters. A duplicate sample was collected at EDL-2 and analyzed for total mercury. In July 2009, the Regional Board and USEPA sampled Lakes 1, 2, and 6 for mercury with a duplicate sample collected in Lake 2. Sampling depths were 0.58 m, 0.33 m, and 0.96 m, respectively. Total mercury was analyzed with EPA Method 1631 with a detection limit of 0.15 ng/L. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.021 ng/L.

Table G-85 presents the in-lake mercury and TSS measurements for the two sampling events. Methylmercury concentrations ranged from 0.020 ng/L to 0.072 ng/L. Total mercury concentrations were consistently below the water quality standard (50 ng/L) and ranged from 0.41 to 1.17 ng/L.

Table G-85. In-lake Water Column Measurements for the El Dorado Park Lakes

Location	Date	Time	MeHg (ng/L)	Total Hg (ng/L)	TSS (mg/L)
EDL-1	2/26/2009	8:30	0.046	0.89	4.6
EDL-2		11:15	0.041	1.08	4.0
EDL-2 (duplicate)		11:30	NA	1.17	NA
EDL-1	7/15/2009	11:15	0.063	0.50	5.9
EDL-2		9:40	0.072	0.41	9.6
EDL-2 (duplicate)		9:40	NA	0.42	NA
EDL-6		15:50	0.020	1.03	2.9

Additional water quality samples were collected from the El Dorado Park lakes. Table G-86 presents the chloride, sulfate, total alkalinity, total dissolved solids, and total organic carbon data collected from Lakes 1 and 2 on February 26, 2009 and July 15, 2009.

Table G-86. Supplemental Water Quality Monitoring for In-lake Samples in the El Dorado Park Lakes

Location	Date	Time	Chloride (mg/L)	Sulfate (mg/L)	Total Alkalinity (mg/L)	Total Dissolved Solids (mg/L)	Total Organic Carbon (mg/L)
EDL-1	2/26/2009	8:30	33.7	283.3	203	380	3.9
EDL-2		11:15	80.9	80.2	274	532	6.0
EDL-5		12:15	55.65	88.2	126	406	3.9
EDL-6		13:20	56.1	88.3	126	370	6.2
EDL-1	7/15/2009	11:15	25.56	20.47	210	350	4.8
EDL-2		9:40	80.51	33.28	274	532	11.3
EDL-2 (duplicate)		9:40	81.64	33.71	274	NA	NA

G.8.2.1.1 Sediment Samples

During the February and July sampling events, USEPA and the Regional Board also collected sediment samples at each station to measure total and methylmercury concentrations in sediment. Total mercury was analyzed with EPA Method 1631 with detection limits ranging from 3.51 µg/kgto 6.96 µg/kg. Methylmercury was analyzed with EPA Method 1630 with detection limits ranging from 0.023 µg/kg to 0.049 µg/kgg. Detection limits were adjusted to account for sample aliquot size.

Table G-87 presents the sediment mercury concentrations measured in the El Dorado Park lakes. Concentrations are reported on a dry weight basis. The methylmercury concentrations measured in sediments at these three stations ranged from approximately 0.1 µg/kg to 0.2 µg/kg; the total mercury concentration ranged from 78 µg/kg to 188 µg/kg.

Table G-87. In-lake Sediment Concentrations for the El Dorado Park Lakes

Location	Date	Time	MeHg (µg/kg)	Total Hg (µg/kg)	TSS (%)	Sulfate (mg/kg)
EDL-1	2/26/2009	8:30	0.198	123	28.5	541.5
EDL-2		11:15	0.202	86.8	36.74	130.4
EDL-2 (duplicate)		11:30	NA	89.5	35.88	Not sampled
EDL-1	7/15/2009	11:30	0.167	126	28.28	219.1
EDL-2		9:40	0.102	78.0	28.36	192.98
EDL-2 (duplicate)		9:40	0.121	94.8	29.04	NA
EDL-6		15:50	0.113	188	18.26	822.89

G.8.2.2 Fish Tissue Sampling

Mercury concentrations in the fish tissue of largemouth bass have been measured in the El Dorado Park lakes since 1991. Lake 1 was sampled by the TSMP in the 1990s as composite samples: the number in each composite was not provided. The Surface Water Ambient Monitoring Program (SWAMP) sampled individual fish from Lake 2 during the summer of 2007 (Davis et al., 2008) and March 2010. Table G-88 presents the fish tissue mercury concentrations on a wet weight basis. Concentrations range from 0.131 ppm to 0.678 ppm. The applicable fish tissue target for mercury measured as a wet weight concentration is 0.22 ppm.

Table G-88. Fish Tissue Mercury Concentrations Measured in the El Dorado Park Lakes

Program	Date	Fish Length (mm)	Total Mercury Concentration (ppm wet weight)
TSMP	4/21/1991	382	0.470
TSMP	4/26/1992	378	0.550
TSMP	6/23/1998	350	0.602
SWAMP	Summer 2007	537	0.318
SWAMP	Summer 2007	479	0.672

Program	Date	Fish Length (mm)	Total Mercury Concentration (ppm wet weight)
SWAMP	Summer 2007	386	0.432
SWAMP	Summer 2007	391	0.408
SWAMP	Summer 2007	380	0.480
SWAMP	Summer 2007	386	0.351
SWAMP	Summer 2007	400	0.310
SWAMP	Summer 2007	387	0.559
SWAMP	Summer 2007	391	0.500
SWAMP	Summer 2007	378	0.491
SWAMP	Summer 2007	370	0.446
SWAMP	Summer 2007	304	0.190
SWAMP	Summer 2007	294	0.188
SWAMP	Summer 2007	206	0.150
SWAMP	Summer 2007	219	0.131
SWAMP	3/30/2010	409	0.678
SWAMP	3/30/2010	348	0.259
SWAMP	3/30/2010	345	0.199
SWAMP	3/30/2010	343	0.235
SWAMP	3/30/2010	352	0.151

Piscivorous fish tend to have increased mercury tissue concentrations with age. Figure G-32 shows the mercury concentrations in largemouth bass plotted against length, which is an approximate surrogate for age. For composite fish samples, concentration is plotted against mean length. As expected, fish tissue mercury concentrations increase with length. All fish specimens with a mean or individual length greater than 350 mm exceed the fish tissue target of 0.22 mg/kg. Eleven individual and three composite samples had fish tissue concentrations greater than the target, while four individual samples had concentrations less than the target.

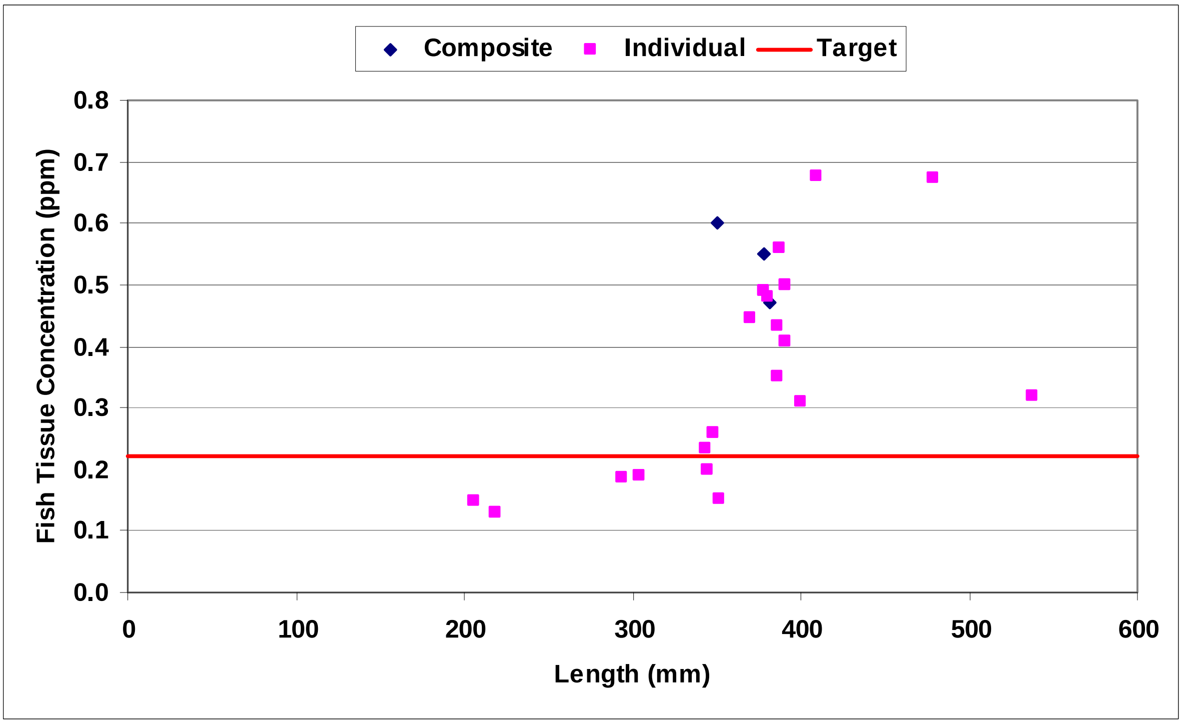


Figure G-32. Mercury Concentrations in Largemouth Bass in the El Dorado Park Lakes

G.8.2.3 Tributary/Inflow Monitoring

G.8.2.1.3 Water Column Measurements

During both the February and July 2009 sampling events, the only visible inputs to the El Dorado Park lakes were the groundwater (GW) input to the most upstream lake in the northern four lakes and the potable water (PW) input to the most upstream lake in the southern two lakes. No culverts in the park area were discharging. Concentrations of methyl and total mercury observed in these inputs are reported in Table G-89. Total mercury was analyzed with EPA Method 1631 with a detection limit of 3.03 ng/L. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.020 ng/L. Detection limits for the groundwater analyses were adjusted to account for sample aliquot size.

The groundwater input was sampled during both events near the pump house after allowing the line to purge for at least ten minutes. Methylmercury concentrations ranged from 0.109 ng/L to 0.215 ng/L; total mercury ranged from 131 ng/L to 142 ng/L. The concentration of total mercury in these samples were almost three times higher than the water quality standard of 50 ng/L and 100 to 200 times higher than the concentrations observed in the water columns of the northern lakes (Section G.8.2.1.1). The portion of mercury in the methyl form ranged from 0.08 to 0.15 percent; the methylmercury concentrations were two to five times higher than the average measured in the northern lakes. The potable water input was only sampled during the July event. Concentrations of methyl and total mercury were 0.020 ng/L and 2.84 ng/L, respectively.

Reclaimed water (RW) is used at the park for irrigation. This source was sampled in December 2009. Total mercury was analyzed with EPA Method 1631 with a detection limit of 0.15 ng/L. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.020 ng/L. These values are similar to the potable water results.

Table G-89. Tributary/Inflow Water Column Measurements for the El Dorado Park Lakes

Location	Date	Time	MeHg (ng/L)	Total Hg (ng/L)	TSS (mg/L)
EDL-GW	2/26/2009	9:40	0.215	142	0.7
EDL-GW	7/15/2009	11:30	0.109	131	1.7
EDL-PW	7/15/2009	14:40	0.020	2.84	0.3
EDL-RW	12/1/2009	14:30	0.021	1.46	0.8

The Long Beach Water Department samples five wells in the vicinity of the El Dorado Park Lakes. However, the analysis employed has relatively high detection limits (200 ng/L), and all samples have been less than detection.

G.8.2.1.3 Sediment Samples

During both the February and July 2009 monitoring events, the only inputs observed to the El Dorado Park lakes were the groundwater and potable water inputs. Neither of these inputs has a sediment-transport capacity.

G.8.3 MONITORING RELATED TO METALS IMPAIRMENTS

In 1996 El Dorado Park lakes was deemed impaired by copper and lead. Monitoring data for cadmium, copper, lead, and zinc are presented in this section. El Dorado Park lakes is not listed for cadmium or zinc, but those data are presented here for completeness because other waterbodies in the region are affected by some of these contaminants.

Metal samples were collected from the north end of Alamo Lake at El Dorado Park lakes (shown in Figure G-17 (pink triangle)), as part of the 1992-1993 Urban Lakes Study (UC Riverside, 1994). Results are shown in Table G-90. Specifically, sampling included dissolved copper and dissolved lead. Dissolved copper samples were collected throughout the water column at depths from the surface to 4.5 meters. The range of the 45 dissolved copper samples was between less than 10 µg/L and 99 µg/L. Similarly, dissolved lead samples were also collected throughout the water column, again at depths from the surface to 4.5 meters. The 45 samples collected ranged in concentration from less than 1 µg/L to 108 µg/L.

The Regional Board completed its Water Quality Assessment and Documentation Report for waterbodies in the Los Angeles Region in 1996 (LARWQCB, 1996). The summary table for El Dorado Park lakes states that copper and lead were not supporting their assessed uses: 45 measurements had a maximum lead concentration of 108 µg/L, a maximum copper concentration of 99 µg/L, and a maximum zinc concentration of 21 µg/L (raw data were not provided, but it is assumed that most of these samples are associated with the Urban Lake Study [UC Riverside, 1994]).

Unfortunately, metal levels were analyzed at relatively high detection limits compared to current detection limits; dissolved copper minimum detection 10 µg/L while dissolved lead was 1 µg/L. No hardness data were collected as part of the Urban Lakes Study, thus it cannot be compared to the hardness-based water quality objectives.

Table G-90. El Dorado Park Lakes 1992/1993 Monitoring Data for Metals

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
7/16/1992	0	23	<1
	3	21	<1
	4.5	17	2
7/16/1992	0	N/A	<1
	3	25	1
	4.5	27	2
7/16/1992	0	40	14
	3	28	8
	4.5	29	<1
8/20/1992	0	31	2
	2	21	<1
	4	18	1
9/24/1992	0	13	2
	2	<10	3
	4	<10	3
10/20/1992	0	16	<1
	2	21	<1
	3.5	24	<1
11/12/1992	0	21	2
	2.5	19	2
	3.5	34	3
12/15/1992	0	<10	1
	2.5	<10	1
	3.5	<10	1
1/21/1993	0	<10	2
	2.5	<10	<1
	3.5	<10	<1
2/10/1993	0	<10	<1
	2.5	18	<1
	3.5	99	<1
3/8/1993	0	<10	17

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
	1.5	19	<1
	2.5	N/A	N/A
	3.5	36	<1
4/8/1993	0	16	37
	1.5	14	3
	2.5	12	<1
	3.5	11	<1
5/12/1993	0	15	28
	2.5	13	4
	3.5	12	1
6/15/1993	0	<10	82
	1.5	<10	34
	2.5	<10	86
	3.5	<10	108

Table G-91 presents38 additional metals samples that were collected by the USEPA and Regional Board between February 2009 and September 2010 at the El Dorado Park lakes. Samples were collected at locations EDL-1, EDL-2, EDL-3, EDL-4, EDL-5, EDL-6, and shoreline samples at EDL-1S, EDL-2S, EDL-3S and EDL-6S. Sites were analyzed for dissolved cadmium, copper, lead, and zinc.

Detection limits were lower than the 1992-1993 study with a cadmium detection limit of 0.2 µg/L, dissolved copper detection limit of 0.4 µg/L, dissolved lead detection limit of 0.05 µg/L, and dissolved zinc detection limit of 0.2 µg/L. All dissolved cadmium concentrations were less than 0.6 µg/L; copper concentrations were between 0.4 µg/L and 6.7 µg/L; lead concentrations ranged from <0.1 µg/L to 0.4 µg/L; and zinc concentrations were <0.1 µg/L to 22.7 µg/L. Metals toxicity is affected by hardness; therefore, each sample was also analyzed for hardness. The 2009-2010 sampling resulted in a hardness range of 56 mg/L to 138.7 mg/L. Since dissolved results pertain to the applicable standard and recent data more closely represents current conditions, data in Table G-91 were weighted more heavily in the assessment.

Table G-91. Water Column Metals Data for the 2008-2010 El Dorado Park Lakes Sampling Events

Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
2/26/2009	EDL 1	100.2	<0.2	1	<0.1	0.3	
2/26/2009	EDL 2	113.4	<0.2	1.9	0.1	0.4	average of duplicate
7/15/2009	EDL 1	117.3	<0.2	1.2	0.1	1.6	
7/15/2009	EDL 2	122.5	<0.2	2.5	0.1	2.3	average of duplicate

Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
7/15/2009	EDL 3	88.1	<0.2	2.7	0.1	<0.1	
7/15/2009	EDL 4	87.4	<0.2	3.8	0.2	0.6	
7/15/2009	EDL 6	84.6	<0.2	2.5	0.1	4.3	
7/15/2009	EDL 5	85.6	<0.2	2.7	0.1	3.9	
12/1/2009	EDL1S	112.7	<0.2	0.5	<0.1	0.6	average of replicates
12/1/2009	EDL2S	132.2	<0.2	0.9	0.1	2.6	
12/1/2009	EDL3S	94.3	<0.2	1.6	0.2	1.6	
12/1/2009	EDL5	125	<0.2	2.9	0.1	11.5	
12/1/2009	EDL6S	120.8	<0.2	2.9	0.3	13	average of duplicate
12/1/2009	EDL4S	93.1	<0.2	1.4	0.2	2.3	
12/15/2009	EDL1	124.3	<0.2	0.4	<0.1	<0.1	average of replicates
12/15/2009	EDL2	138.7	<0.2	1.1	0.1	<0.1	
12/15/2009	EDL3	97.7	<0.2	1.8	0.3	2.5	average of duplicates
12/15/2009	EDL4	97.9	<0.2	2.5	0.4	1.1	
12/15/2009	EDL5	120.3	<0.2	2.8	0.2	14.2	
12/15/2009	EDL6	124.4	<0.2	2.7	0.3	10.6	
1/26/2010	EDL1S	107.8	<0.2	1.2	<0.1	1.4	average of replicates & duplicate
1/26/2010	EDL2S	123.8	<0.2	1.7	0.1	1.3	
1/26/2010	EDL3S	95.2	<0.2	2.5	0.2	<0.1	
1/26/2010	EDL4S	94.9	<0.2	3	0.2	1.6	
1/26/2010	EDL5	81.2	<0.2	3.4	0.2	13.9	
1/26/2010	EDL6S	103.9	<0.2	3.7	0.2	22.7	
8/10/2010	EDL1	NA	0.585	0.509	<0.05	<0.1	Hardness not analyzed
8/10/2010	EDL2	NA	0.502	0.915	<0.05	<0.1	Hardness not analyzed
8/10/2010	EDL3	NA	0.516	1.76	<0.05	<0.1	Hardness not analyzed
8/10/2010	EDL4	NA	0.525	2.16	<0.05	3.60	Hardness not analyzed
8/10/2010	EDL5	60.5	0.493	3.70	<0.05	5.21	
8/10/2010	EDL6	58.1	0.495	3.66	<0.05	10.4	
9/27/2010	EDL 1S	61	<0.2	<0.4	<0.05	<0.1	

Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
9/27/2010	EDL 2S	77	<0.2	1.14	<0.05	<0.1	
9/27/2010	EDL 3S	71	<0.2	4.51	<0.05	2.51	
9/27/2010	EDL 4S	72	<0.2	4.29	<0.05	1.25	
9/27/2010	EDL 5	56	<0.2	5.895	<0.05	12.25	
9/27/2010	EDL 6	57	<0.2	6.7	<0.05	12.2	

Note: All data collected by the Regional Board or USEPA.

USEPA collected eight sediment samples between August and September 2010 to further evaluate lake conditions. Table G-92 summarizes the copper and lead concentrations measured in these samples. There were zero sediment lead exceedances of the 128 ppm freshwater (Probable Effect Concentrations) sediment target. There were four sediment copper exceedances of the 149 ppm freshwater (Probable Effect Concentrations) sediment target.

Table G-92. Sediment Metals Data for the August 2010 El Dorado Park Lakes Sampling Event

Organization	Date	Station ID	Copper mg/kg)	Lead (mg/kg)	Notes
EPA	8/10/2010	EDL1	101	18.7	
EPA	8/10/2010	EDL2	109	19.8	
EPA	8/10/2010	EDL3	97.6	16.1	
EPA	8/10/2010	EDL4	121	16.0	
EPA	8/10/2010	EDL5	533	47.2	
EPA	8/10/2010	EDL6	278	34.6	
EPA	09/28/2010	EDL5	237.3	23.7	Average of field replicates
EPA	09/28/2010	EDL6	466	55.7	

(This page left intentionally blank.)

G.9 Monitoring Data for North, Center, and Legg Lakes

Monitoring data relevant to the impairments of North, Center, and Legg lakes are available from 1992, 1993, 2009, and 2010. Figure G-33 shows the historical and recent monitoring locations for these lakes.

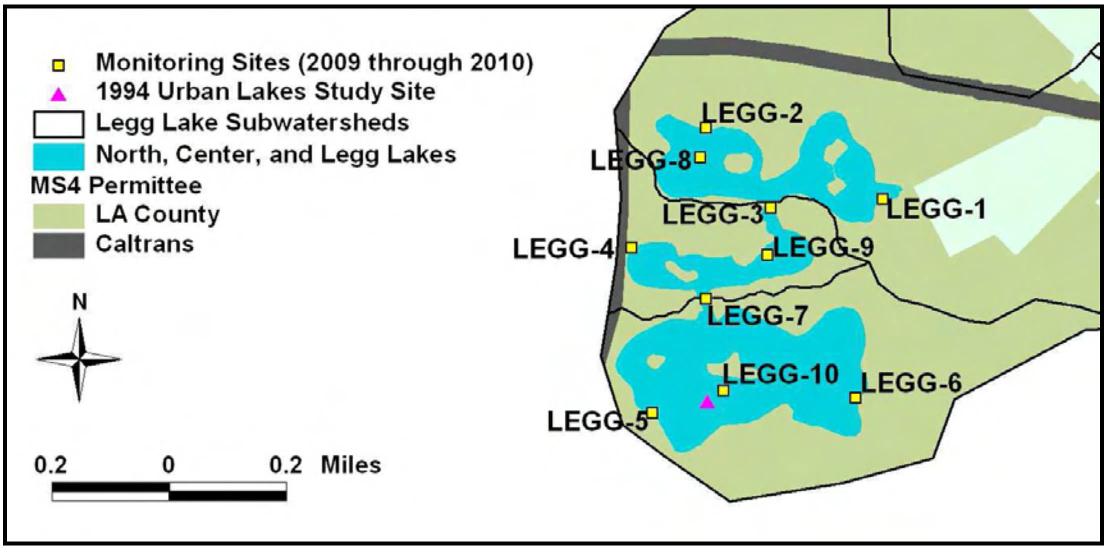


Figure G-33. North, Center, and Legg Lakes Monitoring Sites

G.9.1 MONITORING RELATED TO NUTRIENT IMPAIRMENTS

Legg Lake was monitored in 1992 and 1993 for water quality as part of the Urban Lakes Study from the lower section of the lake on the western side (pink triangle, Figure G-33) (Table G-93). TKN generally ranged from 0.6 mg/L to 1.0 mg/L although three samples were less than the reporting limit and one outlier had a concentration of 37 mg/L. The majority of the ammonium samples (33 of 43) were less than the reporting limit; ammonium concentrations as high as 0.4 mg/L were observed, which are above both the chronic and acute targets (for assessment purposes, we are assuming that the analysis methodology converted all ammonia to ammonium). All nitrite samples were less than the reporting limit, and nitrate concentrations did not exceed 0.2 mg/L. Both phosphate and total phosphorus were less than the reporting limit in all 43 samples. pH ranged from 8.0 to 8.9, and TOC ranged from 2.3 mg/L to 6.6 mg/L. The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from 2 μg/L to 27 μg/L with an average of 15 μg/L.

Table G-93. Legg Lake 1992/1993 Monitoring Data

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
7/6/1992	0	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	5.4	200
	1.5	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	4.8	197
	2.1	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	4.8	209
7/6/1992	0	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	5	199

Date	**Depth (m)**	**TKN (mg/L)**	**NH_4-N (mg/L)**	**NO_2-N (mg/L)**	**NO_3-N (mg/L)**	**PO_4-P (mg/L)**	**Total P (mg/L)**	**pH**	**TOC (mg/L)**	**TDS (mg/L)**
	1.3	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	4.8	202
	1.6	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	5.6	200
7/6/1992	0	0.8	0.1	<0.01	<0.01	<0.01	<0.01	8.9	4.6	206
	1.4	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.9	4.7	193
	1.8	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.9	5.1	201
8/12/1992	0	0.9	0.1	<0.01	<0.01	<0.01	<0.01	8.6	5.5	248
	1.5	0.7	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	5.5	196
	2.5	0.7	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	4.8	217
8/12/1992	0	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	6.3	204
	2	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	5.3	207
8/12/1992	0	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	6.6	191
	1.5	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	5.2	218
9/21/1992	0	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	5.1	201
	1.5	0.7	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	4.8	192
	2.5	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	4.7	190
10/8/1992	0	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	4.9	206
	1.5	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	4.2	212
	2.5	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	4.1	211
11/3/1992	0	37	<0.01	<0.01	<0.01	<0.01	<0.01	8.8	4.9	179
	1.5	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	5	200
	3	1	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	4.7	244
12/15/1992	0	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	8.4	3	228
	2	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	8.4	3.2	235
1/13/1993	0	0.8	0.1	<0.01	<0.01	<0.01	<0.01	8	3.3	191
	2	0.9	0.1	<0.01	<0.01	<0.01	<0.01	8.1	3.5	190
2/3/1993	0	0.7	<0.01	<0.01	0.2	<0.01	<0.01	8.7	4.4	215
	2	0.9	<0.01	<0.01	0.2	<0.01	<0.01	8.7	3.5	222
3/4/1993	0	0.8	0.4	<0.01	0.2	<0.01	<0.01	8	3	199
	1.5	0.8	0.3	<0.01	0.2	<0.01	<0.01	8.1	2.8	197
	2.5	1	0.3	<0.01	0.1	<0.01	<0.01	8.1	2.9	195
4/13/1993	0	0.7	0.2	<0.01	<0.01	<0.01	<0.01	8.4	2.9	227
	1.5	0.8	0.2	<0.01	<0.01	<0.01	<0.01	8.4	2.5	228

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
	2.5	0.9	0.2	<0.01	0.1	<0.01	<0.01	8.5	2.7	223
5/5/1993	0	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	8.2	2.8	202
	2	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	8.3	2.4	198
	3	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	8.4	2.3	192
6/8/1993	0	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	8.6	2.9	215
	1.5	0.7	<0.01	<0.01	<0.01	<0.01	<0.01	8.6	3.1	215
	2.5	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	8.6	2.9	214

The Regional Board's 1996 Water Quality Assessment Database does not include data for Legg Lake or its watershed. The Assessment Report does include summary information for the impairments. Ammonia was partially supporting the aquatic life use; 43 ammonium samples were collected with concentrations ranging from non-detect to 0.35 mg/L, the upper end of this range is below the acute target, but above the chronic target (for assessment purposes, we are assuming that the analysis methodology converted all ammonia to ammonium). Raw data are not available to assess location, date, time, depth, temperature, or pH with regard to these samples. pH was partially supporting the aquatic life use and not supporting the secondary drinking water use. Eighty-four measurements of pH ranged from 7.6 to 8.9. Odor was listed as not supporting the contact and non-contact recreation uses.

The Legg Lake system was sampled multiple times during May, June, and July 2007 (Table G-94; data provided by the county of Los Angeles). Nineteen of 21 samples of ammonia had concentrations ranging from less than the detection limit of 0.01 mg-N/L to 0.36 mg-N/L; two samples had ammonia concentrations of 0.51 mg-N/L and 0.53 mg-N/L (both were collected from Center Lake in May). Nitrate concentrations ranged from less than the detection limit of 0.02 mg-N/L to 0.59 mg-N/L. Orthophosphate ranged from less than the detection limits (either 0.01 mg-P/L or 0.02 mg-P/L, depending on the sampling event) to 0.07 mg-P/L.

Table G-94. 2007 County of Los Angeles Water Quality Data for the Legg Lake System

Monitoring Location	Date	Ammonia (mg-N/L)	Nitrate (mg-N/L)	Orthophosphate (mg-P/L)
Center Lake open water	5/18/07	0.23	0.02	0.02
	5/25/07	0.51	0.02	0.02
	5/31/07	0.53	0.02	0.07
	6/18/07	0.06	0.12	0.02
	6/21/07	0.1	0.15	0.02
	6/29/07	0.36	0.18	0.02
	7/5/07	0.25	0.07	0.01
North Lake east storm drain inlet	5/18/07	0.61	0.02	0.02
	5/25/07	0.01	0.04	0.02
	5/31/07	0.04	0.02	0.07

Monitoring Location	**Date**	**Ammonia (mg-N/L)**	**Nitrate (mg-N/L)**	**Orthophosphate (mg-P/L)**
	6/18/07	0.22	0.02	0.02
	6/21/07	0.33	0.02	0.02
	6/29/07	0.17	0.02	0.02
	7/5/07	0.33	0.03	0.01
North Lake west storm drain inlet	5/18/07	0.35	0.02	0.02
	5/25/07	0.01	0.05	0.02
	5/31/07	0.04	0.02	0.07
	6/18/07	0.12	0.02	0.02
	6/21/07	0.01	0.02	0.02
	6/29/07	0.99	0.02	0.02
	7/5/07	0.25	0.03	0.01
North Lake open water	5/18/07	0.01	0.02	0.02
	5/25/07	0.01	0.17	0.02
	5/31/07	0.02	0.02	0.07
	6/18/07	0.01	0.02	0.02
	6/21/07	0.04	0.02	0.02
	6/29/07	0.2	0.02	0.02
	7/5/07	0.03	0.05	0.01
North PVC irrigation pipe outlet	7/5/07	0.07	0.02	0.01
South Lake open water	5/18/07	0.01	0.1	0.02
	5/25/07	0.01	0.06	0.02
	5/31/07	0.1	0.02	0.07
	6/18/07	0.1	0.31	0.02
	6/21/07	0.12	0.59	0.02
	6/29/07	0.05	0.49	0.02
	7/5/07	0.07	0.21	0.01
South Lake near EPA treatment plant	5/18/07	0.01	0.17	0.02
	5/25/07	0.01	0.25	0.02
	5/31/07	0.12	0.34	0.07
	6/18/07	0.05	0.86	0.02
	6/21/07	0.08	0.65	0.02
	6/29/07	5.76	0.59	0.02

Monitoring Location	Date	Ammonia (mg-N/L)	Nitrate (mg-N/L)	Orthophosphate (mg-P/L)
	7/5/07	0.01	0.22	0.01
South Lake near well water inlet	5/18/07	0.01	0.11	0.02
	5/25/07	0.01	0.19	0.02
	5/31/07	0.1	0.12	0.07
	6/18/07	0.11	0.46	0.02
	6/21/07	0.19	0.5	0.02
	6/29/07	0.06	0.37	0.02
	7/5/07	0.15	0.22	0.01

On February 3, 2009, the Regional Board sampled water quality around the shoreline of Legg Lake (stations LEGG-5 and LEGG-6) as well as the two smaller lakes to the north (stations LEGG-1 through LEGG-4) and the connecting channel to Legg Lake (LEGG-7). Site LEGG-44 is a field duplicate site for LEGG-4. Note that the 2006 303(d) lakes coverage shows only Legg Lake proper. Table G-95 presents these monitoring results. As expected with shoreline monitoring, nutrients and chlorophyll *a* concentrations were relatively high (see Section 6 in main document). TKN ranged from 0.63 mg/L to 2.6 mg/L. Ammonia ranged from non-detect to 0.07 mg/L. Nitrite ranged from 0.04 mg/L to 0.05 mg/L, and nitrate ranged from 0.04 mg/L to 0.74 mg/L. Dissolved orthophosphate was only greater than the detection limit at LEGG1 with a concentration of 0.0106 mg/L. Total phosphorus concentrations ranged from 0.017 mg/L to 0.089 mg/L. TOC ranged from 3.0 mg/L to 5.9 mg/L. TDS ranged from 46 mg/L to 476 mg/L; TSS ranged from 5.7 mg/L to 16.6 mg/L. Chlorophyll *a* concentrations ranged from 26.7 µg/L to 115 µg/L. In general, concentrations were lower at the two Legg Lake shoreline sites relative to the other sites. This was particularly true of chlorophyll *a* concentrations, which ranged from 26.7 µg/L to 38.3 µg/L at the two locations. Secchi depths were not measured at these shoreline sites.

Table G-95. February 2009 Water Quality Monitoring Around the Shoreline of Legg Lake

Sample Location	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	TOC (mg/L)	TDS (mg/L)	TSS (mg/L)	Chl *a* (µg/L)
LEGG1	2.2	<0.03	0.04	0.185	0.01060	0.08	5.55	121	16.6	115
LEGG2	2.6	0.03	0.04	0.04	<0.0075	0.089	5.9	46	15.8	103.6
LEGG3	1.4	0.06	0.05	0.39	<0.0075	0.087	5.1	256	8.8	37.4
LEGG4	0.63	0.07	0.04	0.45	<0.0075	0.047	3.8	374	6.7	27.6
LEGG44	1.5	0.06	0.04	0.45	<0.0075	0.03	5.4	436	5.7	29.4
LEGG5	1.4	<0.03	0.04	0.64	<0.0075	0.033	4.4	444	10.8	38.3
LEGG6	0.70	<0.03	0.04	0.74	<0.0075	0.03	3.5	476	6.7	26.7
LEGG7	1.4	<0.03	0.04	0.63	<0.0075	0.017	3	434	10	32

Field data for the February 2009 monitoring event are summarized in Table G-96. At the two Legg Lake shoreline sites, temperature ranged from 16 ºC to 16.5 ºC, and pH ranged from 8.1 to 8.3. Across all sites, temperature ranged from 12.5 ºC to 16.5 ºC, and pH ranged from 8.0 to 9.0.

Table G-96. February 2009 Field Data for the Legg Lake Monitoring Event

Sample Location	Temperature ºC	pH
LEGG1	12.5	9.0
LEGG2	14.5	8.8
LEGG3	12.5	8.0
LEGG4	14.5	8.2
LEGG44	15.0	8.3
LEGG5	16.5	8.1
LEGG6	16.0	8.3
LEGG7	12.5	9.0

The North, Center and Legg lakes were sampled for summer conditions on July 14, 2009. In-lake samples were taken at Legg Lake sites 8, 9, and 10. A duplicate was performed at Legg Lake 10 as a quality control measure. Site 7 is in the channel that connects Legg Lake and Center Lake. The nutrients measured during this monitoring event are shown in Table G-97. Groundwater was also sampled from a pump. The groundwater pump provides flow to the North Lake and the South/Legg Lake via two cascading waterfall areas. Water flowing in North Lake at station Legg-3 was sampled from the pipe on the center lakeside flowing towards the north lake. The total depth at the sampling location and entrance of the pipe was 0.25 m. The samples at Legg-7 were taken at a depth of 0.20 m. The total depth of Legg-7 is 0.61 meters and has a Secchi depth of 0.41 meters. Legg-9 was sampled at 0.30 meters and has a total depth of 0.88 meters. The Secchi depth at Legg-9 was 0.61 meters. Samples at Legg-8 and Legg-10 (including the duplicate) were taken at approximately 0.20 meters. The depth of Legg-8 and Legg-10 are 2.2 and 2.5 meters, respectively. The Secchi depth at Legg-8 was 0.38 meters and the Secchi depth at Legg-10 was 0.48 meters.

Table G-97. July 2009 Water Column Measurements for the Legg Lakes

Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	Chl *a* (µg/L)	Secchi Depth (m)
LEGG-7[1]	11:05	1.4	<0.03	<0.01	<0.01	<0.0075	0.043	64.1	0.41
LEGG-8	12:15	1.7	<0.03	<0.01	<0.01	<0.0075	0.066	63.1	0.38
LEGG-9	9:30	1.4	<0.03	<0.01	<0.01	<0.0075	0.043	37.4	0.61
LEGG-10	10:45	1.47	<0.03	<0.01	<0.01	<0.0075	0.046	93.45	0.48
LEGG-10D	10:45	1.4	<0.03	<0.01	<0.01	<0.0075	0.089	NS	0.48
LEGG-GW[2]	8:18	<0.46	0.03	<0.01	1.26	<0.0075	0.036	NS	NA
LEGG-3[3]	13:30	1.5	0.03	<0.01	<0.01	<0.0075	0.185	26.7	NA

[1] LEGG-7 represents a channel sample location.

[2] LEGG-GW represents the groundwater input to the North and South Lakes, not an in-lake sample.

[3] LEGG-3 represents input from Center Lake to the North Lake, sampled from a pipe, not an in-lake sample.

The July 2009 sampling event also monitored for chloride, sulfate, total alkalinity, hardness, TDS, TSS, DOC and TOC. These samples were taken at Legg-7, 8, 9, and 10. No duplicate was performed for these parameters. These data are shown in Table G-98.

Table G-98. July 2009 Supplemental Water Quality Monitoring for In-lake Samples in the Legg Lakes

Location	Time	Chloride (mg/L)	Sulfate (mg/L)	Total Alkalinity (mg/L)	Hardness (mg/L as $CaCO_3$)	TDS (mg/L)	TSS (mg/L)	DOC (mg/L)	TOC (mg/L)
LEGG-7[1]	11:05	68.36	117.13	56	138.6	500	25	16.4	17.6
LEGG-8	12:15	42.12	62.38	50	97.6	302	26	12.1	13.3
LEGG-9	9:30	69.82	119.86	76	158.7	434	10.5	16.4	16.9
LEGG-10	10:45	66.18	114.21	54	136.2	440	21	14.85	18.4
LEGG-GW[2]	8:18	36.3	64.13	160	187.8	394	1	0.3	0.2
LEGG-3[3]	13:30	68.4	118.34	78	165.7	512	14.15	17.6	17

[1] LEGG-7 represents a channel sample location.

[2] LEGG-GW represents the groundwater input to the North and South Lakes, not an in-lake sample.

[3] LEGG-3 represents input from Center Lake to the North Lake, sampled from a pipe, not an in-lake sample.

Profile data were collected at LEGG-7, LEGG-8, LEGG-9, and LEGG-10 during the July 14, 2009 sampling event by USEPA and the Regional Board. These data are presented in Table G-99. The North Lake depth was 2.20 meters and the Secchi depth was 0.38 meters. The temperature in this lake was between 26.3 and 27.1 °C. The average DO is 12.7 mg/L, excluding the much lower bottom DO measurement, which was 7.9 mg/L. The DO maximum in Center Lake, LEGG-9, occurred at 1 meter of depth (11.33 mg/L) and the DO below 2.5 meters of depth was less than 2.0 mg/L. The temperature was between 25.3 and 28.6 °C. Center Lake had a depth of 2.9 meters and a Secchi depth reading of 0.61 meters. A reading was taken in the channel between the Center and South lakes at LEGG-7. The depth was 0.61 meters and the Secchi depth was 0.41 meters. The DO at the Secchi depth was 12.4 mg/L and the temperature was 28.2 °C. The DO in the South Lake, LEGG-10, was as high as 12.9 mg/L in the upper water column to declines to 2.8 mg/L at the bottom off the lake. Based on this data, the lakes appear to be stratified and have a euphotic zone of greater production, occurring just before or around the first meter of depth in each lake.

Table G-99. Data Collected in Legg Lakes on July 14, 2009

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	S Cond (mS/cm)	Secchi Depth (m)	Total Depth (m)
LEGG-7	11:20	0.4	28.2	9.1	12.4	0.633	0.41	0.61
LEGG-8	9:00	Surface	27.1	8.1	13.0	0.381	0.38	2.20
		0.5	27.1	9.1	13.6	0.381		
		0.99	26.8	8.9	13.1	0.383		
		1.01	26.8	8.9	13.1	0.383		
		1.5	26.7	8.5	10.6	0.402		

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	S Cond (mS/cm)	Secchi Depth (m)	Total Depth (m)
		2.0	26.3	7.9	7.2	0.435		
LEGG-9	12:00	Surface	28.6	8.7	10.8	0.677	0.61	2.90
		0.03	28.6	8.7	10.8	0.677		
		0.5	28.4	8.8	11.0	0.677		
		1.0	27.6	8.8	11.3	0.677		
		1.5	27.3	8.9	11.2	0.678		
		2.0	26.7	8.1	6.7	0.697		
		2.5	26.3	7.7	1.9	0.707		
		2.8	25.3	7.7	1.7	0.748		
LEGG-10	10:30	Surface	27.7	9.1	12.7	0.631	0.48	2.50
		0.1	27.6	9.1	12.7	0.631		
		0.5	27.4	9.2	12.9	0.630		
		1.0	26.8	9.1	12.8	0.630		
		1.5	26.4	8.8	12.5	0.643		
		2.0	25.9	8.1	7.2	0.671		
		2.4	25.2	8.0	2.8	0.716		

The South Lake was measured again at 15:00 during the July sampling event. The DO in the afternoon was much higher in the first meter of depth. The maximum DO in the afternoon is 16.3 mg/L and the maximum DO in the morning is 12.9 mg/L. The afternoon temperature was slighter higher than the morning temperature in the first meter of the lake. At the surface of the lake, the temperature in the morning was 27.7 °C and rose to 29.4 °C in the afternoon. These data are displayed in Figure G-34.

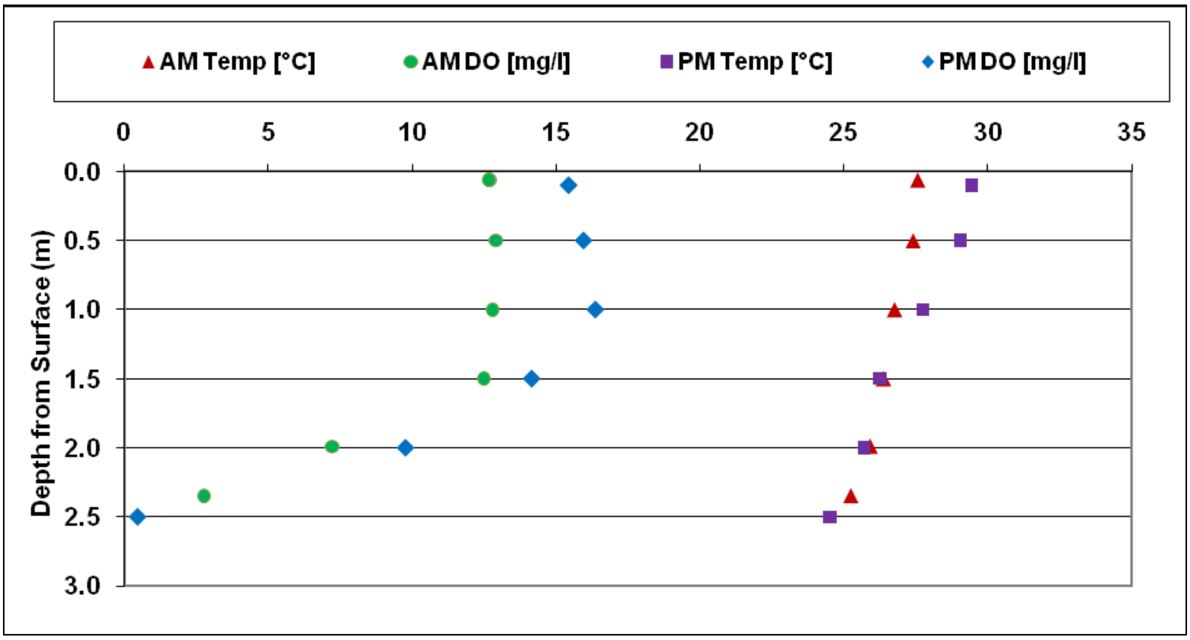


Figure G-34. Profile Data Collected in Lake Legg 10 at approximately 10:30 a.m. and 3:00 p.m. on July 14, 2009

The increase of dissolved oxygen in the afternoon indicate algal productivity in the lake. The saturation level of DO is used to give insight into the impacts of this productivity. The percent saturation in the morning and afternoon are shown in Figure G-35 and listed in Table G-100. The dissolved oxygen saturation is highest in the photic zone in the afternoon at 210 percent. In the morning, the DO saturation in this zone is 165 percent. Saturation above 100 percent indicates DO input from the algal production. The saturation at the bottom of the lake is 34 percent in the morning and 5 percent in the afternoon.

The algae have produced DO and caused the saturation level to exceed 100 percent. The algal productivity is higher in the afternoon when the light intensity is greater. The DO in the bottom of the lake has also been deleted by the increased producvitiy. The DO saturation at the bottom of the lake was 34 percent in the morning and 5 percent in the afternoon.

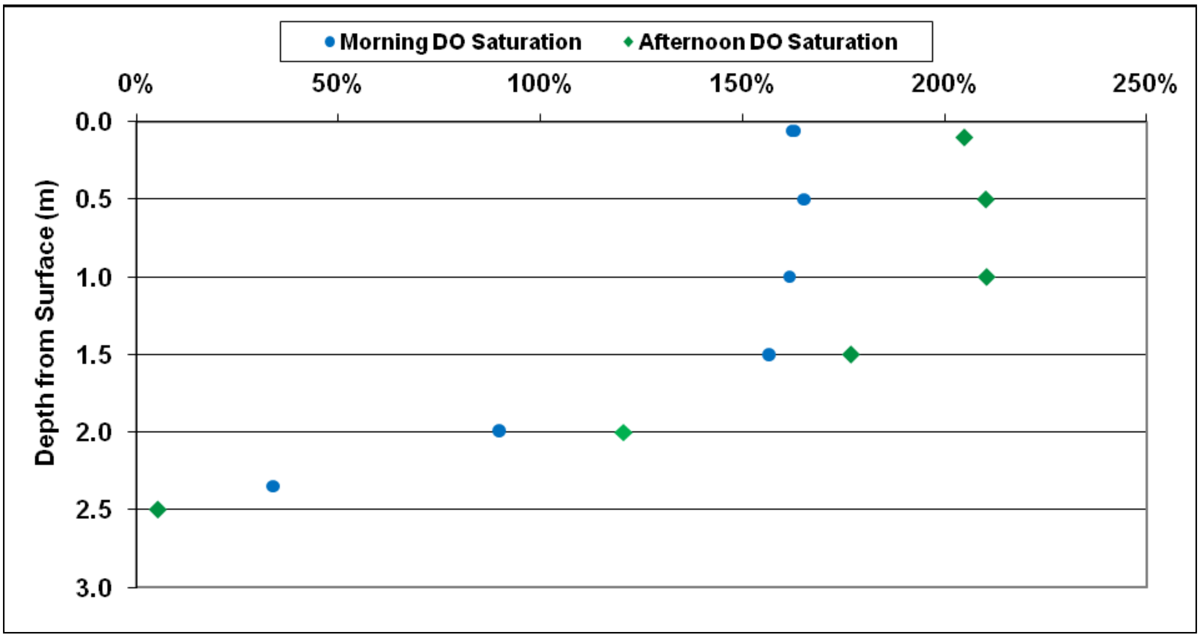


Figure G-35. Dissolved Oxygen Saturation in Lake Legg 10 at approximately 10:30 a.m. and 3:00 p.m. on July 14, 2009

Table G-100. Calculated DO Saturation from Data Collected in Legg Lake 10 on July 14, 2009

Depth (m)	DO Saturation at 10:30 a.m.	DO Saturation at 3:00 p.m.
~0.1	162%	205%
0.5	165%	210%
1.0	162%	210%
1.5	157%	177%
2.0	90%	121%
~2.5	34%	5%

USEPA sampled North, Center, and Legg lakes on June 8, August 11, and September 29, 2010 (Table G-101). Secchi depth ranged from 0.4 m to 1.3 m. In-lake samples of TKN ranged from 0.57 to 1.4 mg-N/L. Ammonia samples ranged from 0.03 to 0.082 mg-N/L. Nitrate-nitrite concentrations were below the detection limit of 0.015 mg-N/L during the June event for all stations and the September events at all Legg 9 and 10 stations; nitrate-nitrite of 0.059 to 0.081 mg-N/L was observed at Legg 8 in September. During the August event, nitrate ranged from below the detection limit of 0.05 mg-N/L to 0.29 mg-N/L, and nitrite samples were below detection limits of 0.25 mg-N/L. All 2010 orthophosphate measurements were below the detection limit of 0.5 mg-P/L; total phosphorus concentrations ranged from 0.02 mg-P/L to 0.06 mg-P/L. Chlorophyll *a* concentrations ranged from 11 µg/L to 44 µg/L. The August and September chlorophyll *a* data represent estimated values as the samples were held past the holding times.

Table G-101. 2010 In-lake Water Column Measurements for North, Center, and Legg Lakes

Date	Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	NO_2-NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	Chloro-phyll a (µg/L)	Secchi Depth (m)
6/8/2010	LEGG-8	10:30	0.9	0.046	NM	NM	<0.015	<0.5	0.05	15	1.1
6/8/2010	LEGG-9	9:45	0.64	0.03	NM	NM	<0.015	<0.5	0.03	11	1.3
6/8/2010	LEGG-10	8:45	0.57	0.05	NM	NM	<0.015	<0.5	0.04	13	0.8
8/11/2010	LEGG-8	13:00	1.4	0.062	<0.25	0.29	NM	<0.5	0.03	36	NM
8/11/2010	LEGG-8 (Duplicate)	13:00	NM	NM	<0.25	0.28	NM	<0.5	NM	36[2]	NM
8/11/2010	LEGG-9	11:45	0.8	0.046	<0.25	<0.05	NM	<0.5	0.03	18[2]	NM
8/11/2010	LEGG-10	9:15	1.1	0.056	<0.25	0.09	NM	<0.5	0.02	44[2]	0.5
9/29/2010	LEGG-8	10:20	1.1	0.082	<0.25	<0.05	0.081	<0.5	0.06	40	0.5
9/29/2010	LEGG-8 (Duplicate)	10:20	1.2	0.080	<0.25	0.05	0.059	<0.5	0.06	35	0.5
9/29/2010	LEGG-9	11:00	0.99	0.068	<0.25	<0.05	<0.015	<0.5	0.06	24	0.5
9/29/2010	LEGG-10	8:45	1.3	0.082	<0.25	<0.05	<0.015	<0.5	0.05	42	0.4

[1] NM indicates that this value was not measured.
[2] The August chlorophyll *a* data represent estimated values as the samples were held past the holding times.

Ground water quality data for June 8, August 11, and September 29, 2010 are shown in Table G-102. These data represent groundwater quality after being treated and before entering the lake. Ammonia and nitrite concentrations in the groundwater input were similar to those in the lake. TKN in the groundwater samples ranged from below the detection limit of 0.05 to 0.14 mg-N/L, and nitrate ranged from 2.1 to 2.6. Nitrate-nitrite ranged from below the detection of 0.015 to 2.2 mg-N/L. Orthophosphate concentration of the groundwater was below the detection limit of 0.5 mg-P/L; total phosphorus ranged from 0.02 to 0.05 mg/L.

Supplemental water quality data were also collected during the three 2010 sampling events for the in-lake and groundwater sites. These data are shown in Table G-103.

Table G-102. 2010 Ground Water Quality Measurements for North, Center, and Legg Lakes

Date	Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	NO_2-NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)
6/8/2010	EPA-GW	11:15	<0.05	0.059	NM	NM	<0.015	<0.5	0.05
8/11/2010	EPA-GW	10:12	0.05	0.040	<0.25	2.6	NM	<0.5	0.02
	EPA-GW (Duplicate)		0.06	0.042	<0.25	2.6	NM	<0.5	0.02
9/29/2010	EPA-GW	14:40	<0.05	0.067	<0.25	2.1	2.2	<0.5	0.03
	EPA-GW (Duplicate)		0.14	0.060	<0.25	2.1	2.2	<0.5	0.03

Table G-103. 2010 Supplemental Water Quality Monitoring for In-lake Samples in the Legg Lakes

Date	Location	Time	Chloride (mg/L)	Sulfate (mg/L)	Total Alkalinity (mg/L)	Hardness (mg/L as $CaCO_3$)	TDS (mg/L)	TSS (mg/L)	DOC (mg/L)	TOC (mg/L)
6/8/2010	LEGG-8	10:30	62	74	130	NM	360	5	4.5	5.8
6/8/2010	LEGG-9	9:45	83	92	150	NM	450	<5	4.3	4.5
6/8/2010	LEGG-10	8:45	83	94	120	NM	420	<5	4.4	3.8
6/8/2010	EPA-GW[2]	11:15	82	98	180	NM	480	<5	1.6	1.1
8/11/2010	LEGG-8	13:00	95	140	150	270	510	7	4.1	3.9
8/11/2010	LEGG-8 (Duplicate)	13:00	95	140	150	NM	NM	NM	NM	NM
8/11/2010	LEGG-9	11:45	100	130	150	250	490	ND	5.1	5.8
8/11/2010	LEGG-10	9:15	99	140	120	240	480	10	3.9	9.1
8/11/2010	EPA-GW	10:12	93	140	200	NM	650	<5	0.78	0.70
	EPA-GW (Duplicate)	10:12	93	140	NM	NM	NM	NM	NM	NM
9/29/2010	LEGG-8	10:20	95	140	150	270	540	6	4.1	3.9
9/29/2010	LEGG-8 (Duplicate)	10:20	95	140	150	250	NM	NM	NM	NM
9/29/2010	LEGG-9	11:00	100	130	150	240	530	5	5.1	5.8
9/29/2010	LEGG-10	8:45	99	140	120	250	500	11	3.9	9.1
9/29/2010	EPA-GW	14:40	93	140	200	280	580	<5	0.78	0.70
	EPA-GW (Duplicate)	14:40	93	140	NM	240	NM	NM	NM	NM

[1] NM indicates that this value was not measured.
[2] EPA-GW represents the groundwater input after treatment by the EPA facility, not an in-lake sample.

Depth-profile data were also collected during the three 2010 sampling events. As shown in Table G-104 and Table G-105, depth-profile data were collected during the morning and afternoon hours on June 8 and August 11, 2010. On September 29, depth-profile data were collected in the morning hours only due to equipment malfunction (Table G-106); specific conductivity was not measured during this sampling event.

Table G-104. Profile Data Collected in North, Center, and Legg Lakes (6/8/2010)

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Specific Conductivity (mS/cm)	Orp (mV)
Legg-8	10:30	0.32	25.5	8.5	9.9	0.604	134
		0.19	25.6	8.5	10.1	0.607	134
		0.50	25.6	8.5	10.4	0.609	134

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Specific Conductivity (mS/cm)	Orp (mV)
		1.01	25.5	8.5	10.0	0.608	135
		1.50	25.3	8.3	9.3	0.617	138
		1.89	24.8	7.7	3.4	0.633	85
Legg-8	16:00	0.16	27.7	8.7	11.0	0.607	-4
		0.50	27.7	8.7	11.0	0.607	2
		1.01	26.3	8.6	11.3	0.637	10
		1.49	25.7	8.3	9.6	0.625	19
Legg-9	AM	0.15	25.9	8.2	7.6	0.768	172
		0.50	26.0	8.2	7.7	0.767	173
		1.00	26.0	8.3	8.0	0.766	174
		1.51	26.0	8.3	8.2	0.766	174
		2.00	25.6	8.2	7.8	0.767	175
		2.49	24.7	8.0	6.0	0.771	179
Legg-9	PM	0.19	27.7	8.5	8.9	0.765	-2
		0.50	27.7	8.5	9.1	0.765	0
		0.98	27.2	8.4	8.6	0.768	4
		1.50	26.2	8.3	8.3	0.767	8
		1.95	25.7	8.2	7.6	0.768	13
		2.39	25.1	7.9	5.8	0.771	0
Legg-10	8:45	0.16	25.6	8.4	9.3	0.726	216
		-0.01	24.4	13.2	8.5	0.002	-60
		0.51	25.6	8.5	9.2	0.726	215
		1.00	25.6	8.5	9.3	0.727	214
		1.50	25.6	8.5	9.3	0.727	213
		1.97	25.6	8.5	9.3	0.728	212
		2.08	25.3	7.9	5.1	0.738	216
Legg-10	14:45	0.08	27.3	8.6	9.7	0.73	120
		0.51	27.1	8.7	10.0	0.729	115
		1.02	26.2	8.7	10.4	0.732	116
		1.53	25.8	8.7	10.4	0.726	116
		2.02	25.2	8.2	7.5	0.748	122
		2.47	24.2	7.7	1.2	0.755	-144

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Specific Conductivity (mS/cm)	Orp (mV)
		2.61	23.8	7.3	1.2	0.732	Over Range
		2.31	24.5	8.0	3.5	0.753	-137
EPA-GW[1]	11:30	0.01	21.3	7.8	5.0	0.844	105
		0.01	21.4	7.7	4.7	0.846	105
		0.00	21.2	7.6	5.8	0.842	106

[1] EPA-GW represents the groundwater input after treatment by the EPA facility, not an in-lake sample.

Table G-105. Profile Data Collected in North, Center, and Legg Lakes (8/11/2010)

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Specific Conductivity (mS/cm)	Orp (mV)
LEGG8	11:15	0.06	25.8	7.75	7.48	0.815	180
		0.46	25.77	7.73	7.43	0.816	180
		1.05	25.63	7.71	7.34	0.816	178
		1.52	25.32	7.65	6.59	0.818	176
		2.08	25.32	7.49	4.5	0.818	175
		2.53	25.26	7.66	5.57	0.817	71
LEGG8	16:45	0.13	28.43	8.16	8.79	0.818	161
		0.48	28.23	8.15	8.86	0.819	160
		1.01	27.25	8.13	8.95	0.816	159
		1.5	25.66	7.98	8.67	0.816	158
		2.04	25.38	7.81	7.37	0.816	158
		2.47	25.26	7.73	6.54	0.817	157
		2.74	25.16	7.59	4.54	0.818	-10
LEGG9	11:15	0.06	25.8	7.75	7.48	0.815	180
		0.46	25.77	7.73	7.43	0.816	180
		1.05	25.63	7.71	7.34	0.816	178
		1.52	25.32	7.65	6.59	0.818	176
		2.08	25.32	7.49	4.5	0.818	175
		2.53	25.26	7.66	5.57	0.817	71
LEGG9	16:45	0.13	28.43	8.16	8.79	0.818	161
		0.48	28.23	8.15	8.86	0.819	160
		1.01	27.25	8.13	8.95	0.816	159

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Specific Conductivity (mS/cm)	Orp (mV)
		1.5	25.66	7.98	8.67	0.816	158
		2.04	25.38	7.81	7.37	0.816	158
		2.47	25.26	7.73	6.54	0.817	157
		2.74	25.16	7.59	4.54	0.818	-10
LEGG-10	10:10	0.06	24.74	8.22	8.98	0.809	181
		0.44	24.75	8.17	9.01	0.809	183
		0.99	24.7	8.17	9.02	0.809	183
		1.49	24.65	8.15	9	0.808	184
		1.98	24.62	8.15	8.99	0.808	183
		2.5	24.54	7.47	4.07	0.834	188
		2.21	24.56	7.75	6.36	0.825	66
LEGG-10	16:00	0.04	26.7	8.39	10.52	0.808	156
		0.49	26.62	8.44	10.85	0.808	152
		0.99	24.94	8.23	10.48	0.813	152
		1.52	24.7	8.24	9.85	0.807	149
		1.98	24.62	8.14	9.31	0.808	149
		2.5	24.57	7.71	6.6	0.829	153

Table G-106. Profile Data Collected in North, Center, and Legg Lakes (9/292010)

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Orp (mV)
Legg-8	10:20	0.5	24.2	8.6	9.9	74.0
		1.0	23.9	8.3	9.9	76.0
		1.5	23.4	8.2	10.0	77.3
		2.0	22.7	7.9	9.7	81.6
		2.5	22.4	7.7	5.1	18.0
Legg-9	11:00	0.5	24.6	8.9	10.0	28.4
		1.0	23.4	8.8	9.7	31.0
		1.5	23.3	8.1	8.5	45.0
		2.0	22.6	7.9	6.5	42.7
Legg-10	8:45	0.5	23.8	8.8	10.6	119.5

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Orp (mV)
		1.0	23.8	8.8	10.5	117.4
		1.5	22.7	8.0	9.0	126.7
		2.0	22.2	7.8	6.2	119.0
		2.5	22.0	7.9	3.0	111.2

Figure G-36 through Figure G-38 display the depth-profile data for Legg 10 during these events. Dissolved oxygen saturation for Legg 10 is displayed in Figure G-39 through Figure G-41. Similar trends were observed compared to earlier profile sampling.

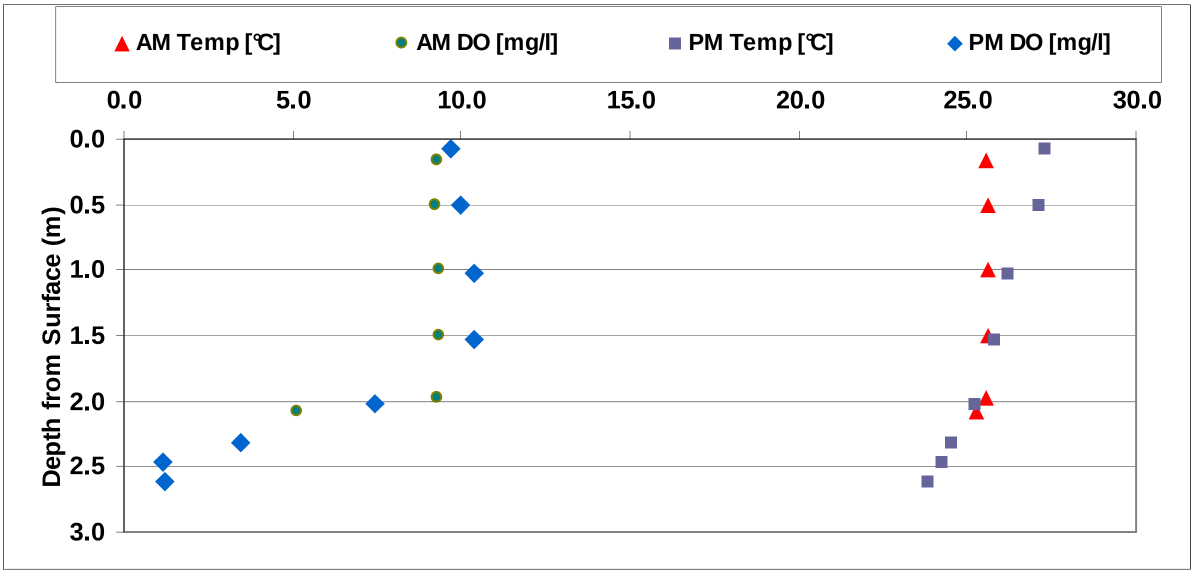


Figure G-36. Profile Data Collected in Lake Legg 10 at approximately 8:45 a.m. and 2:45 p.m. on June 8, 2010

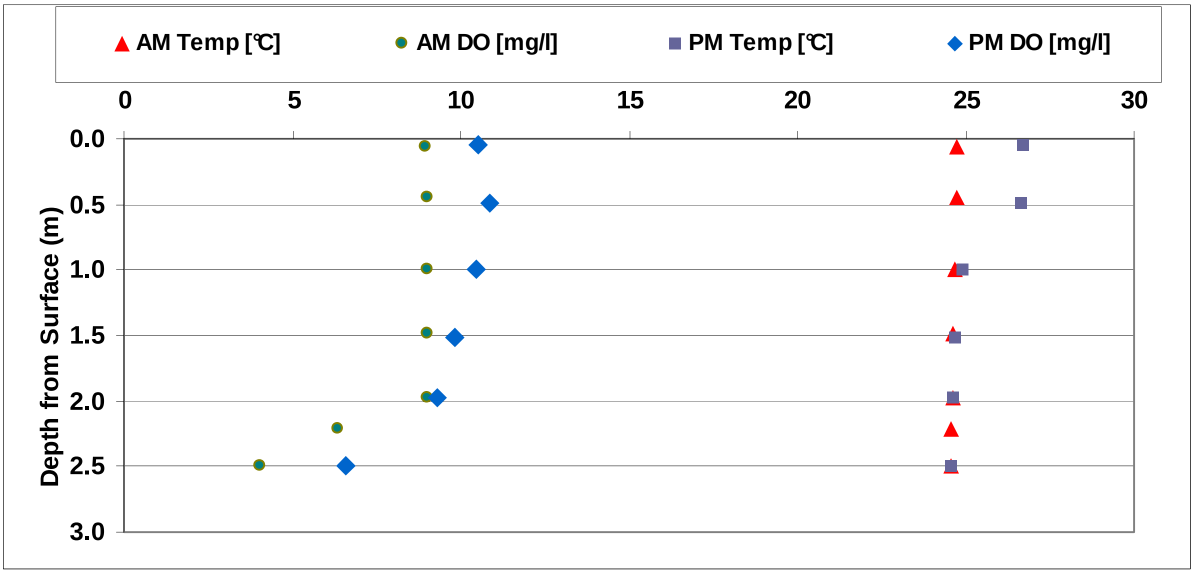


Figure G-37. Profile Data Collected in Lake Legg 10 at approximately 10:10 a.m. and 3:00 p.m. on August 11, 2010

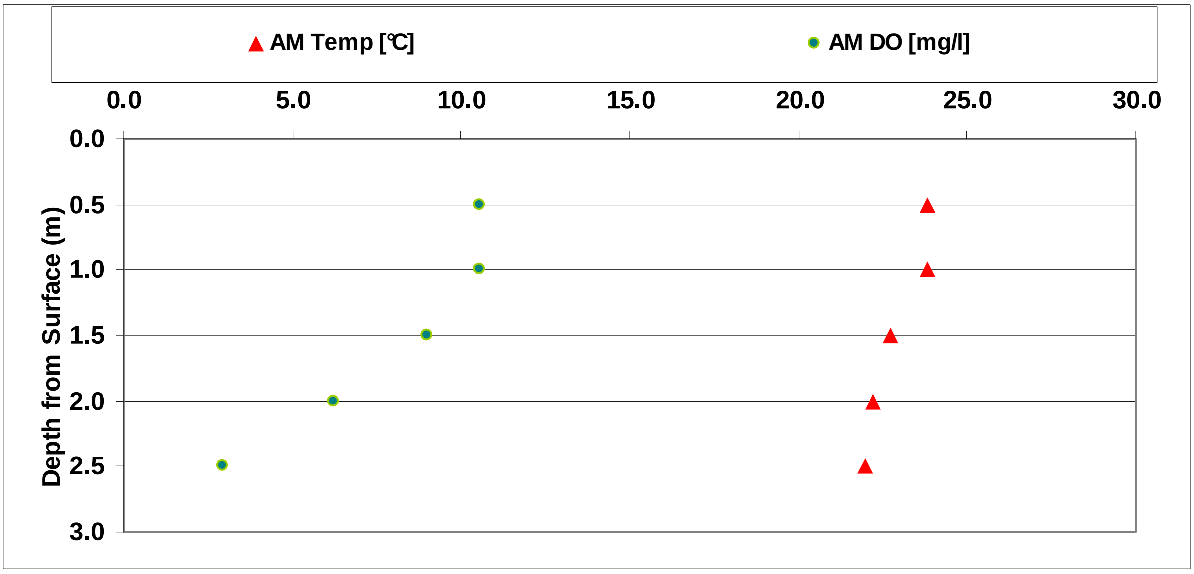


Figure G-38. Profile Data Collected in Lake Legg 10 at approximately 8:45 a.m. on September 29, 2010

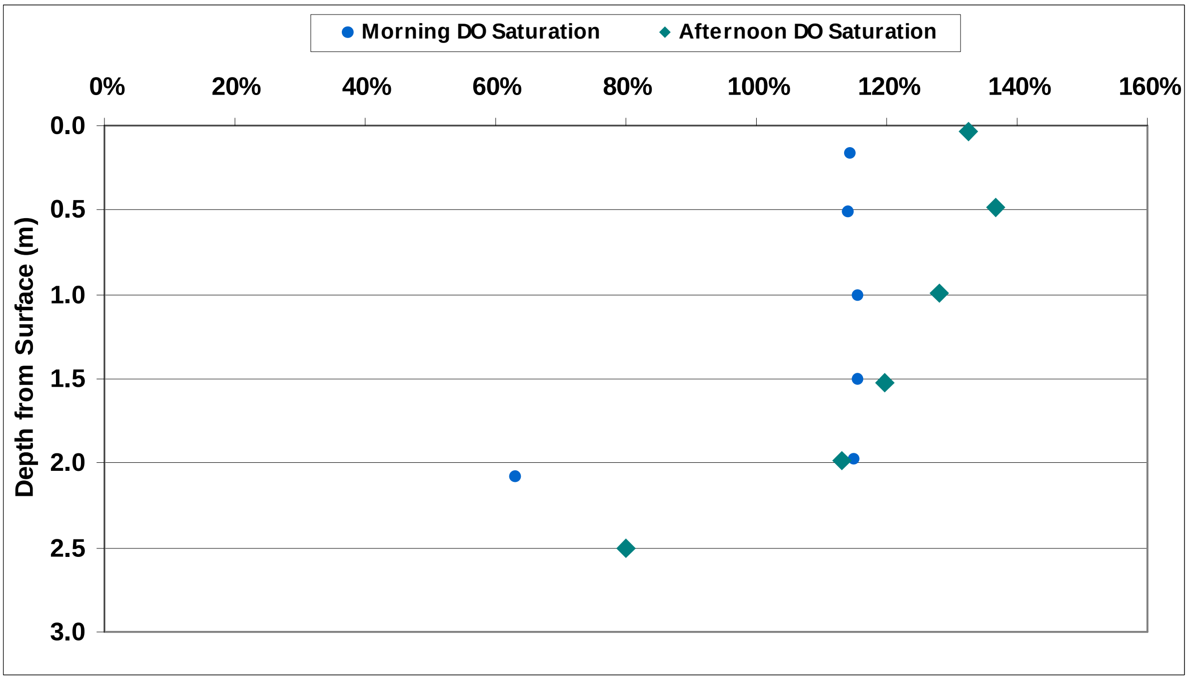


Figure G-39. Dissolved Oxygen Saturation in Lake Legg 10 at approximately 8:45 a.m. and 2:45 p.m. on June 8, 2010

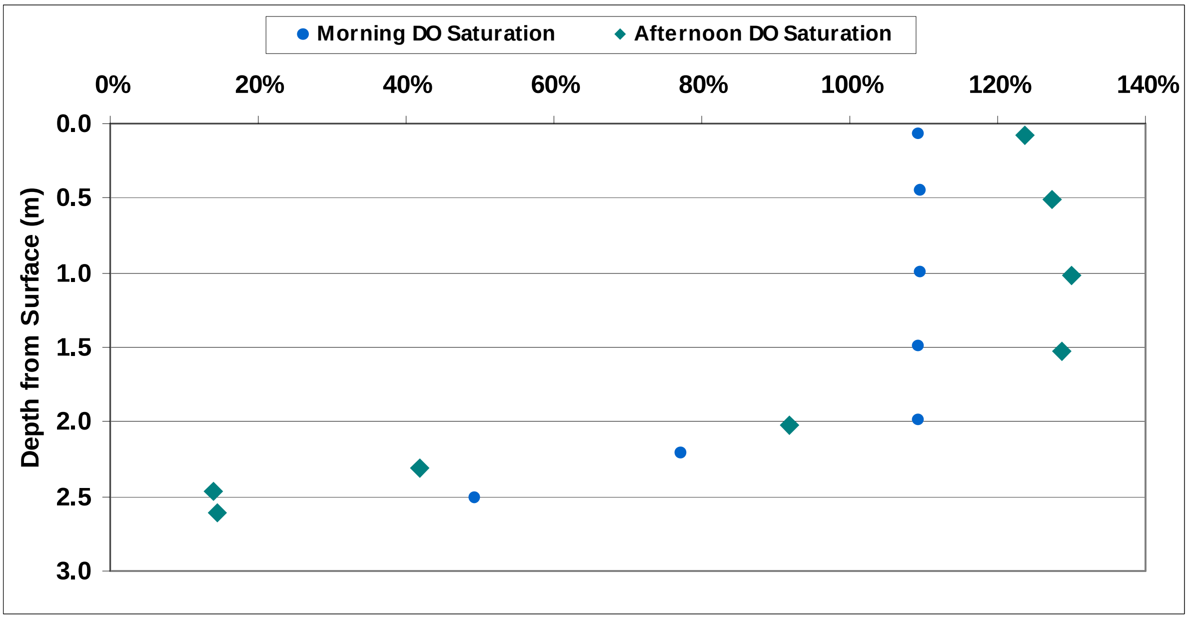


Figure G-40. Dissolved Oxygen Saturation in Lake Legg 10 at approximately 10:10 a.m. and 3:00 p.m. on August 11, 2010

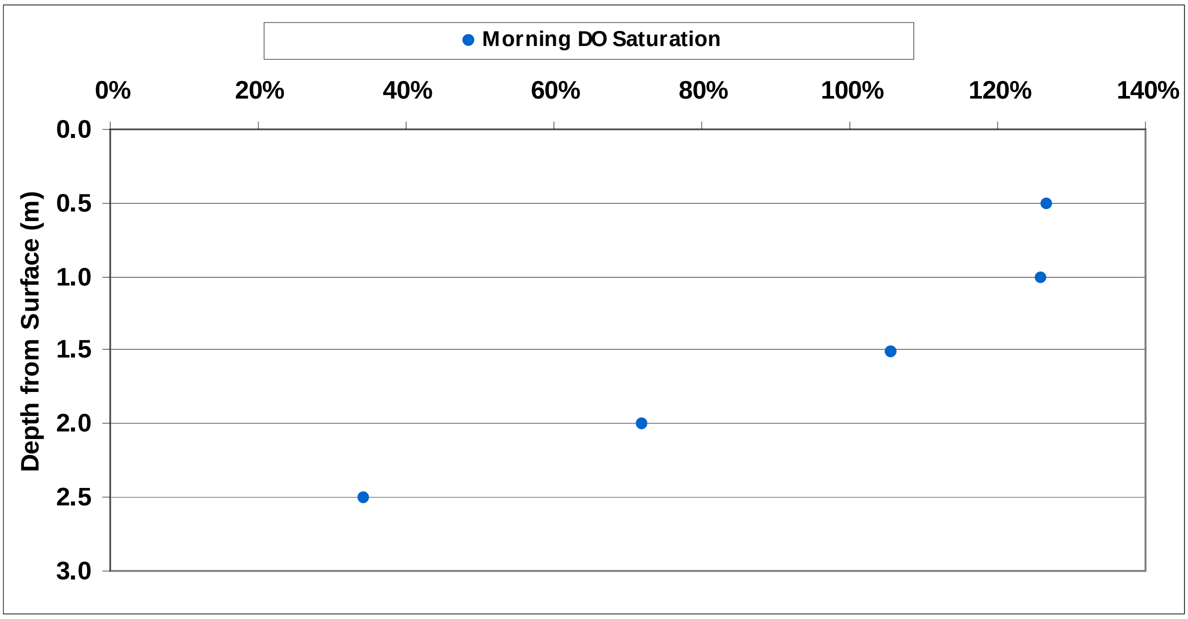


Figure G-41. Dissolved Oxygen Saturation in Lake Legg 10 at approximately 8:45 a.m. on September 29, 2010

G.9.2 MONITORING RELATED TO METALS IMPAIRMENTS

Metals data collected at Legg Lake (pink triangle, Figure G-33), as part of the 1992-1993 Urban Lakes Study (UC Riverside, 1994), are shown in Table G-107. Specifically, sampling included dissolved copper and dissolved lead. Dissolved copper samples were collected throughout the water column at depths from the surface to three meters. The range of the 43 dissolved copper samples was between less than 10 µg/L and 97 µg/L. Similarly, dissolved lead samples were also collected throughout the water column, again at depths from the surface to three meters. The 43 samples collected ranged in concentration from less than 1 µg/L to 70 µg/L.

The Regional Board completed its Water Quality Assessment and Documentation Report for waterbodies in the Los Angeles Region in 1996 (LARWQCB, 1996). The summary table for Legg Lake states that copper and lead were not supporting the assessed uses: 43 measurements had a maximum lead concentration of 70 µg/L, a maximum copper concentration of 97 µg/L, and a maximum zinc concentration of 134 µg/L (raw data were not provided, but it is assumed that most of these samples are associated with the Urban Lake Study [UC Riverside, 1994]).

Unfortunately, metals levels were analyzed at relatively high detection limits compared to current detection limits; dissolved copper minimum detection 10 µg/L while dissolved lead was 1 µg/L. No hardness data were collected as part of the Urban Lakes Study, thus it cannot be compared to the hardness-based water quality objectives.

Table G-107. Legg Lake 1992/1993 Monitoring Data for Metals

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
7/6/1992	0	17	<1
	1.5	35	<1
	2.1	41	1
7/6/1992	0	16	N/A
	1.3	22	<1
	1.6	27	2
7/6/1992	0	18	1
	1.4	42	1
	1.8	26	<1
8/12/1992	0	27	5
	1.5	<10	1
	2.5	30	1
8/12/1992	0	32	7
	2	27	8
8/12/1992	0	41	14
	1.5	37	2
9/21/1992	0	<10	<1
	1.5	<10	<1
	2.5	<10	<1
10/8/1992	0	<10	<1
	1.5	<10	<1
	2.5	<10	<1
11/3/1992	0	<10	1
	1.5	15	1
	3	32	2
12/15/1992	0	<10	<1
	2	<10	<1
1/13/1993	0	<10	<1
	2	12	<1
2/3/1993	0	<10	<1
	2	<10	<1

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
3/4/1993	0	14	<1
	1.5	<10	18
	2.5	<10	27
4/13/1993	0	97	5
	1.5	84	<1
	2.5	78	<1
5/5/1993	0	<10	21
	2	<10	22
	3	<10	<1
6/8/1993	0	<10	70
	1.5	<10	28
	2.5	<10	17

On July 18, 2007, the county of Los Angeles contracted with AquaBio Environmental Technologies to perform sediment sampling near two storm drain inlets in North Lake (Table G-108).

Table G-108. July 18, 2007 County of Los Angeles Sediment Monitoring Data in North Lake

Station	Copper (mg/kg)	Lead (mg/kg)	Zinc (mg/kg)
North Lake near the east storm drain	<5.0	5.7	24
North Lake near the west storm drain	11	<5.0	31

Table G-109 presents 45 additional metal samples that were collected by the USEPA, Regional Board, and/or the County of Los Angeles between February 2009 and September 2010 in North, Center, and Legg lakes. Samples were collected at locations LEGG-1, LEGG -2, LEGG -4, LEGG -5, LEGG -6, LEGG -8, LEGG -9 and LEGG -10. Sites were analyzed for dissolved cadmium, copper, lead, and zinc.

Detection limits were lower than the 1992-1993 study with a cadmium detection limit of 0.2 µg/L, dissolved copper detection limit of 0.4 µg/L, dissolved lead detection limit of 0.05 µg/L, and dissolved zinc detection limit of 0.2 µg/L. All dissolved cadmium concentrations were < 0.2 µg/L to 0.2 µg/L; copper concentrations were between <0.4 µg/L and 3.5 µg/L; lead concentrations ranged from <0.1 µg/L to 1.0 µg/L; and zinc concentrations were <0.1 µg/L to 14.5 µg/L. Metals toxicity is affected by hardness; therefore, each sample was also analyzed for hardness. The 2009-2010 sampling resulted in a hardness range of 68.05 mg/L to 280 mg/L. Since dissolved results pertain to the applicable standard and recent data more closely represents current conditions, data in Table G-109 were weighted more heavily in the assessment.

Table G-109. Metals Data for the 2008-2010 Legg Lake Sampling Events

Organi-zation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
RB	2/3/2009	LEGG 1/2	106.7	0.2	0.9	<0.1	0.4	average of Legg 1 replicates and Legg 2
RB	2/3/2009	LEGG 4	174.3	<0.2	1.1	0.2	1.2	average of duplicates
RB	2/3/2009	LEGG 5 / 6	181.55	<0.2	1.6	0.2	0.6	average of sites 5 & 6
RB/EPA	7/14/2009	LEGG 1/2	96	<0.3	0.6	<0.1	2.1	
RB/EPA	7/14/2009	LEGG 4	155.7	<0.2	0.6	0.1	3.3	
RB/EPA	7/14/2009	LEGG 5	138.5	<0.2	0.6	0.1	1.6	
RB/EPA	7/14/2009	LEGG 8	97.6	<0.2	0.5	0.1	<0.1	
RB/EPA	7/14/2009	LEGG 9	158.7	<0.2	0.5	0.1	<0.1	
RB/EPA	7/14/2009	LEGG 10	136.2	<0.2	0.5	0.1	<0.1	average of replicates and duplicate
County	12/8/2009	LEGG-1	133.75	<0.2	3.5	0.2	14.5	average of replicates
County	12/8/2009	LEGG-10	198.5	<0.2	0.5	<0.1	<0.1	
County	12/8/2009	LEGG-4	195.9	<0.2	<0.4	<0.1	<0.1	
County	12/8/2009	LEGG-6	202.1	<0.2	1.2	<0.1	<0.1	
County	12/8/2009	LEGG-8	155.4	<0.2	0.7	<0.1	0.9	
County	12/8/2009	LEGG-9	188	<0.2	0.5	<0.1	<0.1	average of duplicates
EPA	12/16/2009	LEGG-1	117.55	<0.2	2.3	0.2	11.3	average of replicates
EPA	12/16/2009	LEGG-10	211.1	<0.2	1.4	0.2	9.2	average of duplicates
EPA	12/16/2009	LEGG-4	166.2	<0.2	1.4	0.1	6.6	
EPA	12/16/2009	LEGG-6	200.6	<0.2	0.6	0.2	<0.1	
EPA	12/16/2009	LEGG-8	140.5	<0.2	1.1	0.1	2.7	
EPA	12/16/2009	LEGG-9	170.5	<0.2	1.3	0.1	10.9	
County	1/28/2010	LEGG-1	68.05	<0.2	1.8	0.1	<0.1	average of replicates
County	1/28/2010	LEGG-10	188	<0.2	0.5	0.1	<0.1	
County	1/28/2010	LEGG-4	118.6	<0.2	1.1	<0.1	<0.1	

Organi-zation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
County	1/28/2010	LEGG-6	190.2	<0.2	0.6	0.1	<0.1	
County	1/28/2010	LEGG-8	82.4	<0.2	2	<0.1	<0.1	average of duplicates
County	1/28/2010	LEGG-9	121.8	<0.2	1	0.1	<0.1	
County	2/17/2010	LEGG-1	103.8	<0.2	0.9	0.06	0.2	
County	2/17/2010	LEGG-10	184.2	<0.2	0.6	0.16	0.8	
County	2/17/2010	LEGG-4	128.1	<0.2	0.8	0.06	3.55	average of replicates
County	2/17/2010	LEGG-6	184.9	<0.2	0.6	0.16	2.1	
County	2/17/2010	LEGG-8	79.1	<0.2	1.15	0.055	1.05	average of duplicates
County	2/17/2010	LEGG-9	129	<0.2	0.8	0.05	4	
EPA / RB	8/11/2010	LEGG-1	260	NA	<1	<1	NA	
EPA / RB	8/11/2010	LEGG-10	220	NA	<1	<1	NA	
EPA / RB	8/11/2010	LEGG-4	220	NA	<1	<1	NA	
EPA / RB	8/11/2010	LEGG-6	220	NA	2	<1	NA	
EPA / RB	8/11/2010	LEGG-8	240	NA	<1	<1	NA	
EPA / RB	8/11/2010	LEGG-9	220	NA	<1	<1	NA	
EPA / RB	9/29/2010	LEGG-1	240	NA	1.2	<1	NA	
EPA / RB	9/29/2010	LEGG-10	240	NA	1.7	<1	NA	
EPA / RB	9/29/2010	LEGG-4	250	NA	2.4	<1	NA	
EPA / RB	9/29/2010	LEGG-6	280	NA	1.9	<1	NA	
EPA / RB	9/29/2010	LEGG-8	270	NA	2.15	<1	NA	
EPA / RB	9/29/2010	LEGG-9	250	NA	2.4	<1	NA	

RB = Regional Board
EPA = USEPA
County = County Los Angeles

USEPA also collected three sediment samples during August 2010 to further evaluate lake conditions. Table G-110summarizes the lead and copper concentrations measured in these samples. There were zero sediment lead exceedances of the 128 ppm freshwater (Probable Effect Concentrations) sediment target and zero sediment copper exceedances of the 149 ppm freshwater (Probable Effect Concentrations) sediment target.

Table G-110. Sediment Metals Data for the August 2010 Legg Lakes Sampling Event

Organi zation	Date	Station ID	Copper (µg/g)	Lead (µg/g)	Notes
EPA	08/11/2010	LEGG-8	135	76	Average of duplicates
EPA	08/11/2010	LEGG-9	110	60	
EPA	08/11/2010	LEGG-10	52	20	

G.10 Monitoring Data for Puddingstone Reservoir

Sampling has occurred intermittently from 1992 to 2009. In addition, fish tissue data are available for 1986 to 2007. Figure G-42 shows the locations of historic and recent monitoring.

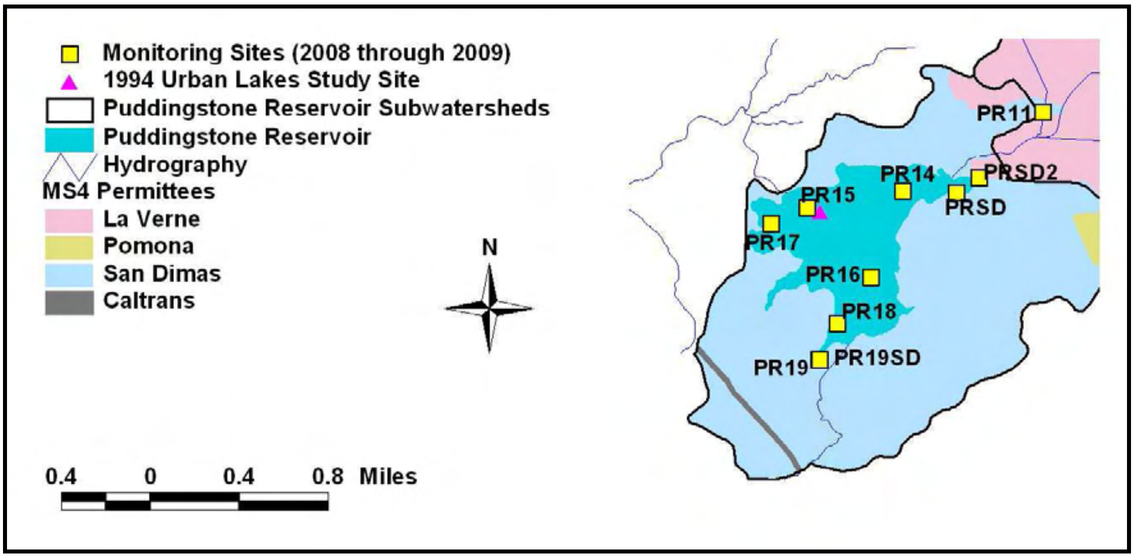


Figure G-42. Puddingstone Reservoir Monitoring Sites

G.10.1 MONITORING RELATED TO NUTRIENT IMPAIRMENTS

Puddingstone Reservoir was monitored for water quality in 1992 and 1993 in support of the Urban Lakes Study near the center of the northern half of the lake (pink triangle, Figure G-42) (Table G-111). TKN ranged from 0.3 mg/L to 6.9 mg/L although concentrations greater than 1.2 mg/L only occurred at depths greater than or equal to 8 meters. Ammonium ranged from 0.1 mg/L to 5.3 mg/L with 39 measurements less than the reporting limit; concentrations did not exceed 0.2 mg/L except at depths greater than or equal to 8 meters. The upper range of these concentrations are above both the chronic and acute targets (for assessment purposes, we are assuming that the analysis methodology converted all ammonia to ammonium). Each of the 75 measurements of nitrite was less than the reporting limit, and 23 nitrate samples were less than the reporting limit. The maximum concentration of nitrate observed was 2 mg/L. Forty-nine of 75 samples of orthophosphate were less than the reporting limit, and the maximum concentration observed was 1.7 mg/L. Total phosphorus was similar with 45 measurements less than the reporting limit and a maximum observed concentration of 1.3 mg/L. Concentrations of neither orthophosphate nor total phosphorus exceeded 0.2 mg/L except at depths greater than or equal to 14 meters. pH ranged from 7.4 to 9.0, and TOC ranged from 2.8 mg/L to 8.2 mg/L. The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from 4 µg/L to 22 µg/L with an average of 13 µg/L; however, the raw data have not been located.

Table G-111. Puddingstone Reservoir 1992/1993 Monitoring Data

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
7/28/1992	0	0.8	<0.01	<0.01	0.2	<0.01	<0.01	7.5	6.7	225
	3	0.8	<0.01	<0.01	0.1	<0.01	<0.01	8.1	5.4	224
	6.5	0.6	<0.01	<0.01	0.1	<0.01	<0.01	7.9	8.2	228
	10	0.9	0.2	<0.01	0.1	<0.01	<0.01	7.8	5.9	231
	13.5	1.3	0.8	<0.01	0.1	0.2	0.2	7.7	5.9	222
	17	2.9	2.5	<0.01	0.1	0.9	0.8	7.5	5.6	235
7/28/1992	0	1	0.1	<0.01	0.1	<0.01	<0.01	8.6	6.4	228
	2	1.1	<0.01	<0.01	0.1	<0.01	<0.01	8.7	5.6	225
	4.5	0.8	<0.01	<0.01	0.1	<0.01	<0.01	8.4	6.9	236
	7	0.7	<0.01	<0.01	0.1	<0.01	<0.01	8	6.6	223
7/28/1992	0	1	<0.01	<0.01	0.1	<0.01	<0.01	8.7	6.1	229
	2.5	1.2	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	6.3	236
	5.5	0.7	<0.01	<0.01	0.1	<0.01	<0.01	8.2	6.4	229
	7.5	0.5	<0.01	<0.01	<0.01	<0.01	<0.01	8	8.2	239
	9.5	0.9	0.3	<0.01	<0.01	0.1	<0.01	7.9	6.7	238
9/1/1992	0	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.1	6.2	190
	4	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.2	6.3	181
	8	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.2	6.2	204
	11	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.1	6.1	197
	14	2.8	1.9	<0.01	<0.01	0.6	0.5	7.6	6.1	211
	17	5.1	4	<0.01	<0.01	1.3	1.3	7.4	6.4	201
10/6/1992	0	0.5	<0.01	<0.01	<0.01	<0.01	<0.01	8.3	6.3	185
	4	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	8.3	6.1	216
	8	0.7	<0.01	<0.01	<0.01	<0.01	<0.01	8.2	6.2	202
	11	0.7	<0.01	<0.01	<0.01	<0.01	<0.01	8.2	6.2	185
	14	0.8	0.3	<0.01	<0.01	<0.01	<0.01	7.8	6.6	220
	17	6.9	5.3	<0.01	<0.01	1.7	1.1	7.4	6.7	193
11/5/1992	0	0.8	0.2	<0.01	0.4	0.1	0.1	7.9	6	197
	3	0.8	0.1	<0.01	0.4	<0.01	0.1	7.9	6	188
	6	0.7	0.2	<0.01	0.4	<0.01	0.1	7.9	5.9	204
	9	1.1	0.2	<0.01	0.4	0.1	0.2	7.9	6.1	186

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
	12	0.9	0.2	<0.01	0.4	0.1	0.1	7.9	5.9	195
	15	0.9	0.3	<0.01	0.4	0.1	0.1	7.9	5.9	200
12/17/1992	0	1.1	0.1	<0.01	0.6	<0.01	<0.01	7.9	5.7	211
	3.5	0.9	<0.01	<0.01	0.6	<0.01	<0.01	7.9	5.7	209
	6.5	0.9	<0.01	<0.01	0.6	<0.01	<0.01	7.9	6.3	205
	9.8	0.9	<0.01	<0.01	0.6	<0.01	<0.01	7.9	5.7	210
	12.5	0.7	<0.01	<0.01	0.5	<0.01	<0.01	7.9	5.8	209
	16.5	0.9	<0.01	<0.01	0.6	<0.01	<0.01	7.9	5.7	207
1/27/1993	0	0.9	0.2	<0.01	2	0.1	0.2	7.9	4.8	196
	5	0.9	0.2	<0.01	2	0.1	0.2	7.9	4.6	198
	10	0.8	0.2	<0.01	2	0.1	0.2	7.9	4.8	199
	15	0.8	0.2	<0.01	2	0.1	0.2	7.9	4.5	208
	20	1	0.3	<0.01	1.9	0.1	0.2	7.9	4.7	216
	25	1.2	<0.01	<0.01	1.8	0.2	0.2	7.9	5.1	194
2/11/1993	0	1.2	<0.01	<0.01	1.7	0.1	0.2	8.1	4.4	181
	3	0.9	0.1	<0.01	1.8	0.2	0.2	8.1	3.9	177
	6	0.6	0.1	<0.01	1.7	0.1	0.2	8.1	4.1	187
	9	0.6	0.1	<0.01	1.7	0.1	0.1	8.1	4.4	186
	12	0.7	0.1	<0.01	1.7	0.1	-	8.1	4.2	208
	15	0.6	0.2	<0.01	1.7	0.2	0.2	8.1	4.4	193
3/11/1993	0	0.6	<0.01	<0.01	1.4	<0.01	<0.01	8.6	3.6	221
	2	0.6	<0.01	<0.01	1.4	<0.01	<0.01	8.7	3.3	225
	5	0.4	<0.01	<0.01	1.7	<0.01	<0.01	8.3	3.2	228
	10	0.5	0.1	<0.01	1.7	<0.01	<0.01	8.2	3	224
	15	0.3	0.2	<0.01	1.6	<0.01	<0.01	8.1	2.8	223
	19	0.3	0.2	<0.01	1.6	<0.01	<0.01	8.2	2.9	233
4/14/1993	0	1.1	<0.01	<0.01	0.7	<0.01	<0.01	9	4.4	236
	2	1	<0.01	<0.01	0.7	<0.01	<0.01	8.9	3.5	230
	6	0.3	<0.01	<0.01	1.1	<0.01	<0.01	8.1	2.8	237
	10	0.5	<0.01	<0.01	1.5	<0.01	<0.01	8	2.8	245
	14	0.4	<0.01	<0.01	1.6	<0.01	<0.01	7.9	2.9	224
	17	0.5	<0.01	<0.01	1.8	<0.01	<0.01	7.9	2.9	211

Date	**Depth (m)**	**TKN (mg/L)**	**NH_4-N (mg/L)**	**NO_2-N (mg/L)**	**NO_3-N (mg/L)**	**PO_4-P (mg/L)**	**Total P (mg/L)**	**pH**	**TOC (mg/L)**	**TDS (mg/L)**
5/11/1993	0	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.6	4.1	213
	5	1.1	0.1	<0.01	<0.01	<0.01	<0.01	8.2	3.6	218
	8	1.3	0.3	<0.01	<0.01	<0.01	<0.01	7.9	3.9	231
	11	1.4	0.2	<0.01	0.8	<0.01	<0.01	7.9	3.4	235
	14	0.8	0.3	<0.01	0.9	0.1	<0.01	7.9	3.6	231
	18	1.3	0.7	<0.01	0.3	0.3	0.3	7.9	3.8	227
6/10/1993	0	1.2	<0.01	<0.01	<0.01	<0.01	0.2	8.9	4.6	238
	3.5	1.1	<0.01	<0.01	<0.01	<0.01	0.1	8.5	3.7	218
	7.5	0.8	<0.01	<0.01	<0.01	<0.01	0.1	8.3	4	224
	10	0.8	<0.01	<0.01	<0.01	<0.01	0.1	8.2	3.6	242
	14	0.5	0.4	<0.01	0.3	0.2	0.1	8	3.7	236
	17	1.5	1.1	<0.01	<0.01	0.4	0.4	8	3.7	234

The 1996 Water Quality Assessment Report does contain summary information regarding the DO impairment which was listed as not supporting the aquatic life use. DO was measured 187 times with concentrations ranging from 0.1 mg/L to 14.9 mg/L. However, the accompanying database does not contain these measurements so no information regarding location, time, depth, or temperature can be compared. There are some temperature and pH measurements in the database that were collected from December 1977 through March 1978. Temperature ranged from 11.1 ºC to 11.7 ºC, and pH ranged from 6.6 to 7.6.

More recent monitoring of nutrients in Puddingstone Reservoir occurred on November 18, 2008 at four locations as well as one site located on Live Oak Wash above the mouth (Figure G-42). All samples of ammonia, TKN, nitrate, nitrite, orthophosphate, total phosphate, and total suspended solids collected at the four lake stations were below the reporting limits of 0.1 mg/L, 1 mg/L, 0.1 mg/L, 0.1 mg/L, 0.4 mg/L, 0.5 mg/L, and 10 mg/L, respectively. Total dissolved solids in the lake ranged from 217 mg/L to 251 mg/L. At the Live Oak Wash site (PR11), the following concentrations were observed: ammonia 0.215 mg/L, TKN 1.87 mg/L, TSS 50 mg/L, TDS 761 mg/L, nitrate 3.31 mg/L, and nitrite 0.131 mg/L. Samples of orthophosphate and total phosphate at this site were less than the reporting limit. Chlorophyll *a* ranged from 11.3 μg/L to 21.4 μg/L.

Field data for the November 2008 monitoring event are summarized in Table G-112. The sampling pump broke after sampling at site PR-15. Water quality samples at the other three sites were collected approximately 4 inches below the surface of the water.

Table G-112. Field Data for the November 18, 2008 Monitoring Event at Puddingstone Reservoir

Station	Time	Depth (m)	Temperature ºC	pH	Secchi Depth (m)
PR-14	13:15	Surface	18.5	8.3	1.2
PR-15	11:00	0.4	18.5	8.5	5.6
		3.0	18.5	8.2	
		6.1	18.0	8.0	
PR-16	15:20	Surface	18.0	8.3	Not reported
PR-17	16:30	Surface	17.5	8.3	Not reported

Puddingstone Reservoir was sampled in February and July in 2009 by USEPA and the Regional Board. The field notes report that approximately 300 gallons of chlorine are pumped into the swim beach area each week during the summer. The edges of the lake are sometimes treated for weeds. The location of the swim beach is not reported. Table G-113 lists the nutrient related measurements collected at stations PR-15 and PR-16 in Puddingstone Reservoir. Samples were collected from a depth of 1.5 meters at locations PR-15 and PR-16 in the winter, and sampled at a depth of 0.35 meters at both locations in the summer. Secchi depths were 0.76 meters at all locations in the winter and 0.71 meters at all locations sampled in the summer. Ideally, samples would have been collected from half the Secchi depth, rather than twice the Secchi depth to reflect average conditions over the photic zone. The summer samples were collected in this area, but winter samples were collected below the photic zone. Nitrogen species had relatively low concentrations at both locations in both seasons. Total phosphorus was slightly elevated with an average concentration of 0.11 mg/L in February and 0.08 mg/L in July. Chlorophyll *a* measurements were relatively high and ranged from 66.1 µg/L to 113.5 µg/L in February. The summer chlorophyll *a* levels were much less, at an average of 26.2 µg/L.

Table G-113. 2009 Water Quality Monitoring at Puddingstone Reservoir

Date	Sample Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	Chl *a* (µg/L)	Secchi Depth (m)
2/24/2009	PR-15	10:10	1.7	0.04	0.05	0.26	0.062	0.121	113.5	0.76
	PR-15 (dup.)	10:30	1.3	0.03	0.02	0.02	0.016	0.114	94.8	0.76
	PR-16	12:15	1.3	0.03	0.02	0.02	0.016	0.098	66.1	0.76
7/16/2009	PR-15	9:00	0.98	<0.03	<0.01	<0.01	<0.0075	0.041	27.3	0.66
	PR-16	10:00	1.1	<0.03	<0.01	<0.01	<0.0075	0.164	25.1	0.71
	PR-16 (dup.)	10:00	1.1	<0.03	NA	NA	NA	0.048	NA	0.71

Supplemental water quality samples were collected from Puddingstone Reservoir. Table G-114 presents the chloride, sulfate, total alkalinity, total dissolved solids, and total organic carbon data collected from Sites 15 and 16.

Table G-114. Supplemental Water Quality Monitoring for In-lake Samples in Puddingstone Reservoir

Date	Location	Time	Chloride (mg/L)	Sulfate (mg/L)	Total Alka-linity (mg/L)	Bicarb-onate (mg/L)	TDS (mg/L)	TSS (mg/L)	DOC (mg/L)	TOC (mg/L)
2/24/2009	PR-15	10:10	40.1	30.2	114	102	224	10.3	5.8	5.8
	PR-15 (dup.)	10:30	40.0	30.3	120	112	118	NA	5.7	6.6
	PR-16	12:15	40.3	30.8	114	98	196	12.7	6.0	6.2
7/16/2009	PR-15	9:00	47.1	34.6	90	NA	258	NA	6.0	7.6
	PR-16	10:00	47.0	34.4	92	NA	246	NA	6.1	8.0

Profile data collected at stations PR-15 and PR-16 on February 24, 2009 are shown in Figure G-43 and Figure G-44, respectively. Measurement depths were limited by cable length to approximately 13 meters. Specific conductivity is constant with depth at both locations. Over 3 to 4 meters of depth, DO decreases from over 6 mg/L at the surface to 0 mg/L. pH ranges from 7.6 to 9.4 at each station. Temperature at these two stations ranges from 11.3 °C to 14.6 °C. **Note that field operators found DO readings suspicious and have since sent meter off for repair (Greg Nagle, USEPA Region IX, personal communication, 5/22/09).**

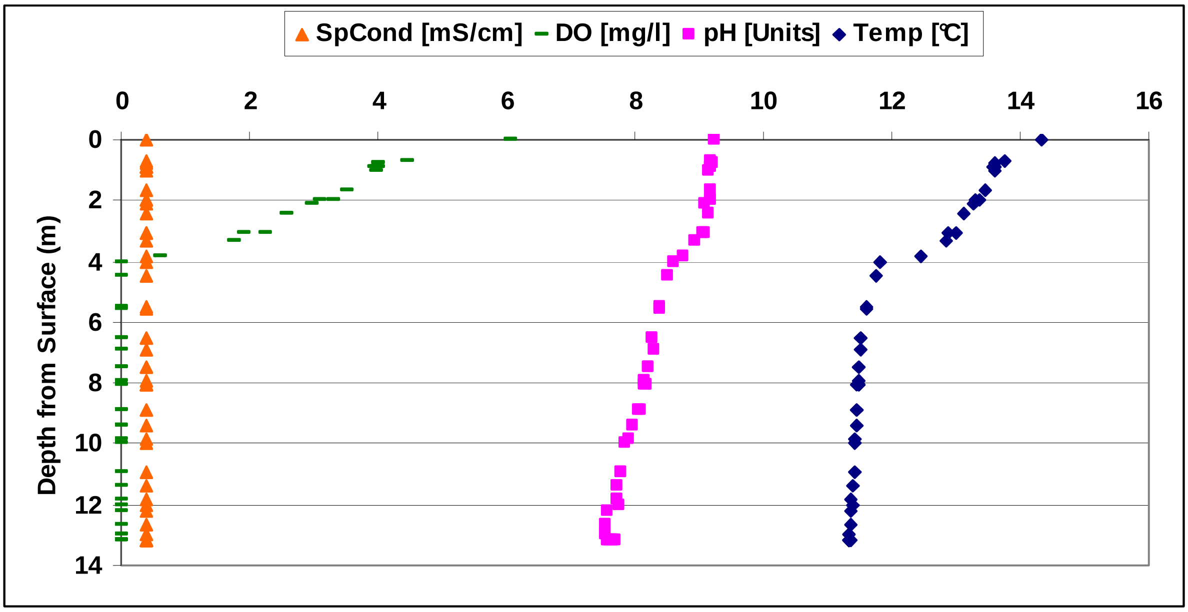


Figure G-43. Profile Data Collected at PR-15 on February 24, 2009

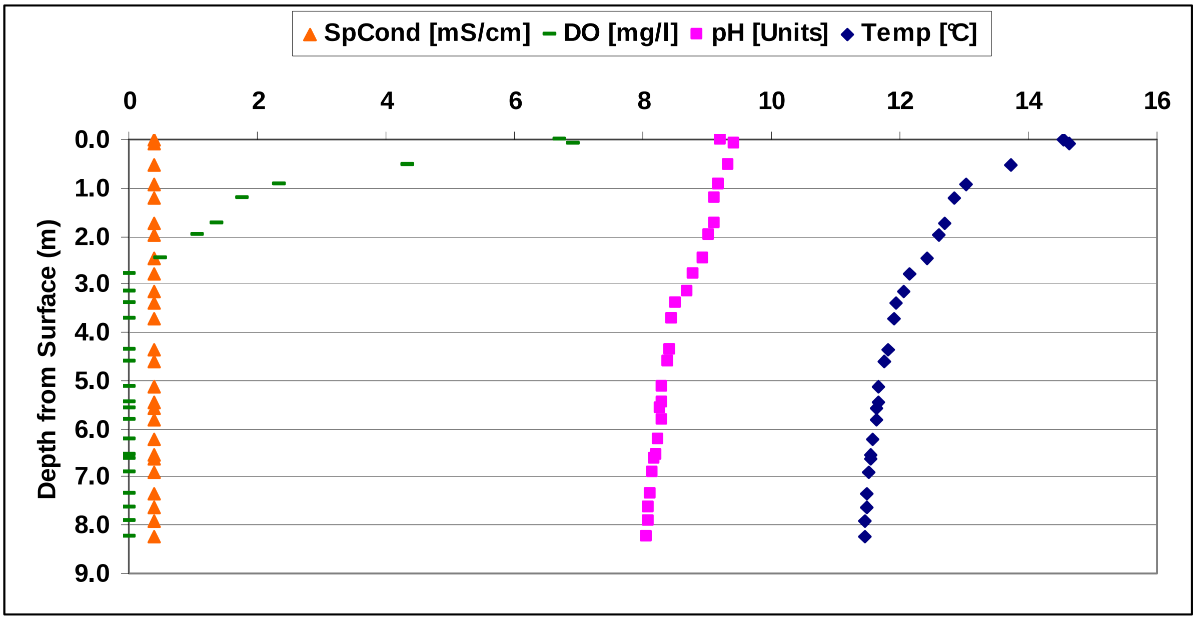


Figure G-44. Profile Data Collected at PR-16 on February 24, 2009

Profile data were also collected on the stations on July 16, 2009, shown in Figure G-45 and Figure G-46. Specific conductivity is constant with depth at both locations, similar to the data collected in January. The DO decreased between 3 and 4 meters of depth, although it remained around 1.8 mg/L instead of dropping to 0 mg/L as it did in January. Temperature at these two stations ranges from 13.0 ºC to 27.1 ºC. The pH ranges from 7.6 to 8.9 at each station.

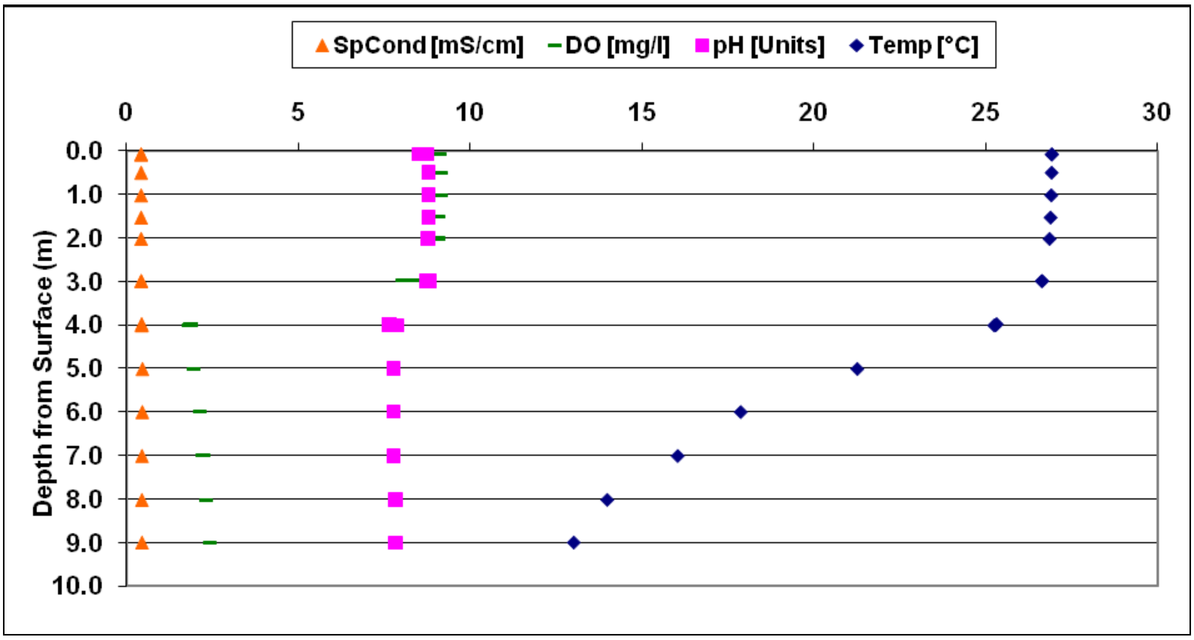


Figure G-45. Profile Data Collected at PR-15 on July 16, 2009

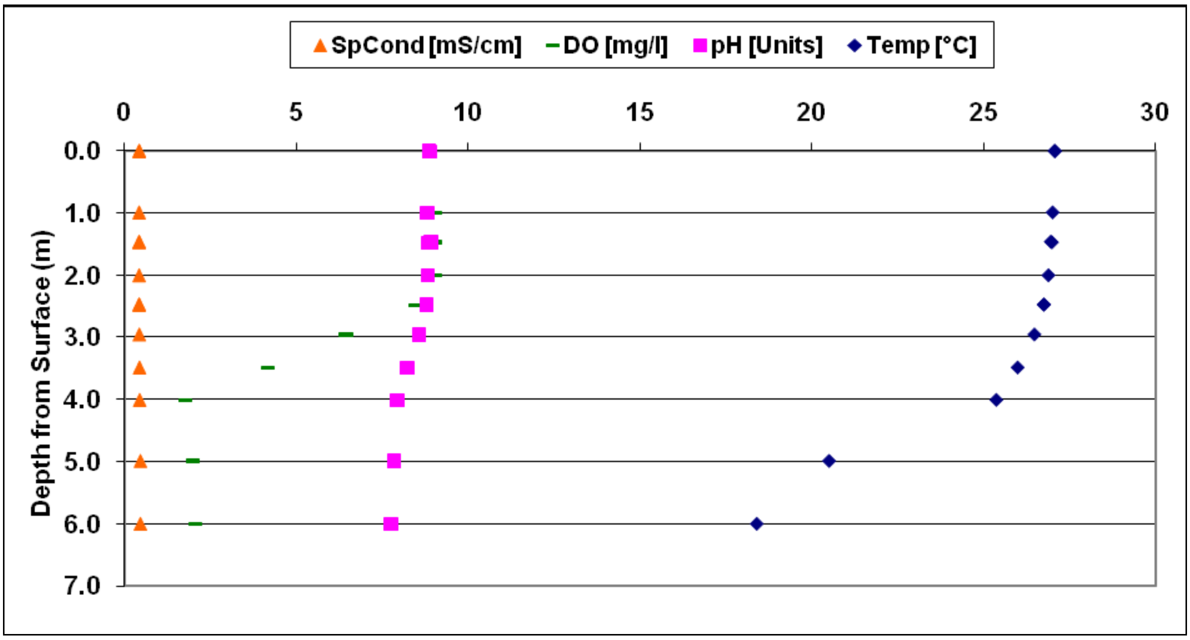


Figure G-46. Profile Data Collected at PR-16 on July 16, 2009

G.10.2 MONITORING RELATED TO MERCURY IMPAIRMENT

Mercury data have been collected in the Puddingstone Reservoir watershed since 1986. Fish tissue concentrations were measured four times under the TSMP from 1986 to 1999. The San Gabriel Watershed Council (SGWC) collected fish tissue measurements in 2006 and 2007, and the Regional Board collected samples in 2004 and 2007 (Davis et al., 2008). In-lake water column concentrations were measured as part of the Urban Lakes Study (UC Riverside, 1994) in 1992. The Regional Board sampled in-lake and tributary water column and sediment mercury concentrations in 2008 and 2009.

G.10.2.1 In-Lake Water Quality Monitoring

G.10.2.1.1 Water Column Measurements

In-lake water column mercury concentrations were measured in July and September 1992 as part of the Urban Lakes Study. All 21 measurements were less than the detection limit of 0.5 µg/L (500 ng/L). As the detection limit of this dataset is 10 times higher than the water quality criterion for mercury (50 ng/L), it is difficult to assess compliance in terms of a water column concentration.

In November 2008, the Regional Board sampled Puddingstone Reservoir for total mercury concentrations in the water column. Water column concentrations ranged from 1.2 ng/L to 1.6 ng/L and were more than one order of magnitude less than the water quality standard. Duplicates were measured at two sites. Samples were processed with EPA method 1631Em, which has a minimum detection limit of 0.5 ng/L.

In February 2009, the Regional Board and USEPA sampled two lake stations for both total and methylmercury. Station PR15 was sampled at a depth of 15.24 meters; the total depth at this location was 17.68 meters. Station PR16 was sampled at a depth of 1.5 meters; the total depth was 8.23 meters. Total mercury was analyzed with EPA Method 1631 with a detection limit of 0.15 ng/L; concentrations ranged from 1.67 ng/L to 2.52 ng/L. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.020 ng/L; concentrations ranged from 0.081 ng/L to 0.127 ng/L. The percent of mercury in the methyl form ranged from 4.8 to 5.2.

These two stations were resampled in July 2009. Both samples were collected from a depth of approximately 0.34 m. Concentrations of methyl and total mercury at both stations were less than those observed during the winter sampling event. Methylmercury ranged from 0.025 ng/L to 0.027 ng/L and total mercury ranged from 0.34 ng/L to 0.35 ng/L. Total mercury was analyzed with EPA Method 1631 with a detection limit of 0.15 ng/L; methyl mercury was analyzed with EPA Method 1630 with a detection limit of 0.020 ng/L.

Table G-115 presents the water column sampling results for Puddingstone Reservoir.

Table G-115. In-lake Water Column Measurements for Puddingstone Reservoir

Location	Date	Time	MeHg (ng/L)	Total Hg (ng/L)	TSS (mg/L)
PR14	11/18/2008	13:15	NA	1.5	NA
PR15		11:00	NA	1.6	NA
PR16		15:20	NA	1.2	NA
PR17		16:30	NA	1.4	NA
PR17 (duplicate)		16:30	NA	1.4	NA
PR15	2/24/2009	10:10	0.127	2.44	10.3
PR15 (duplicate)		10:10	NA	2.52	NA
PR16		12:15	0.081	1.67	12.6
PR15	7/16/2009	9:00	0.027	0.35	9.3
PR16		10:15	0.025	0.34	9.3
PR16 (duplicate)		10:15	NA	0.26	NA

Supplemental water quality data were also collected during the February 2009 event. Table G-116 summarizes the results. Note that the sampling depth for the supplemental data collected at PR-15 was 5 ft, which is different than the sampling depth used to obtain the mercury and TSS measurements.

Table G-116. Supplemental Water Quality Monitoring for In-lake Samples in Puddingstone Reservoir

Date	Location	Time	Chloride (mg/L)	Sulfate (mg/L)	Total Alka-linity (mg/L)	Bicarb-onate (mg/L))	TDS (mg/L)	DOC (mg/L)	TOC (mg/L)
2/24/2009	PR15	10:00	40.1	441.8	117	107	221	5.75	4.20
	PR16	12:15	40.3	4,233.5	114	98	196	6.0	4.95
	PRSD	13:10	252.9	162.1	218	218	NA	12.5	10.80
	PR11	14:30	166.9	88.6	122	66	610	6.5	2.96

G.10.2.1.1 Sediment Samples

In February 2009, the Regional Board and USEPA sampled sediment mercury concentrations at two locations in Puddingstone Reservoir. Total mercury concentration in the sediment samples ranged from 125 µg/kg to 165 µg/kg on a dry weight basis. Methylmercury concentrations ranged from 0.263 µg/kg to 0.502 µg/kg in the sediments. Total mercury was analyzed with EPA Method 1631 with detection limits ranging from 3.61 µg/kg to 4.72 µg/kg. Methylmercury was analyzed with EPA Method 1630 with detection limits ranging from 0.020 µg/kg to 0.021 µg/kg. Detection limits were adjusted to account for sample aliquot size.

In July 2009, the Regional Board and USEPA resampled sediment mercury concentrations at these two locations. Total mercury concentration in the sediment samples ranged from 121 µg/kg to 145 µg/kg on a dry weight basis. Methylmercury concentrations ranged from 0.246 µg/kg to 0.330 µg/kg in the sediments. Total mercury was analyzed with EPA Method 1631 with detection limits ranging from 3.24 µg/kg to 4.12 µg/kg. Methylmercury was analyzed with EPA Method 1630 with detection limits ranging from 0.030 µg/kg to 0.037 µg/kg. Detection limits were adjusted to account for sample aliquot size.

Table G-117 shows the sediment mercury concentrations measured in Puddingstone Reservoir.

Table G-117. In-lake Sediment Concentrations for Puddingstone Reservoir

Location	Date	Time	MeHg (µg/kg)	Total Hg (µg/kg)	TSS (%)	Sulfate (mg/kg)
PR15	2/24/2009	11:00	0.502	165	42.46	859.96
PR15 (duplicate)		11:00	NA	136	26.65	846.61
PR16		12:45	0.263	125	39.73	816.26
PR15	7/16/2009	9:00	0.246	121	23.48	34.56
PR16		10:15	0.330	125	28.78	34.42
PR16 (duplicate)		10:15	NA	145	31.39	NA

G.10.2.2 Fish Tissue Sampling

Mercury concentrations in the fish tissue of largemouth bass have been measured in Puddingstone Reservoir since 1986 by the TSMP, SGWC, and SWAMP. Table G-118 presents the fish tissue mercury concentrations on a wet weight basis. Concentrations range from 0.114 ppm to 0.744 ppm. Twelve individual common carp ranging in length from 395 mm to 687 mm were also analyzed for mercury during the 2004 sampling. Mercury concentrations ranged from 0 ppm to 0.092 ppm and were not considered in the fish tissue versus length mercury regression analysis as a conservative assumption. The applicable fish tissue guideline for mercury measured as a wet weight concentration is 0.22 ppm.

Table G-118. Largemouth Bass Fish Tissue Mercury Concentrations Measured in Puddingstone Reservoir

Program	Date	Number in Sample	Fish Length (mm)	Total Mercury Concentration (ppm wet weight)
TSMP	5/6/1986	6	302	0.200
TSMP	6/11/1991	6	380	0.510
TSMP	4/28/1992	6	386	0.420
TSMP	8/10/1999	6	345	0.371
SWAMP	9/22/2004	1	465	0.449
SWAMP	9/22/2004	1	349	0.365
SWAMP	9/22/2004	1	390	0.311
SWAMP	9/22/2004	1	429	0.39
SWAMP	9/22/2004	1	355	0.384
SWAMP	9/22/2004	1	380	0.369
SWAMP	9/22/2004	1	311	0.152
SWAMP	9/22/2004	1	324	0.271
SWAMP	9/22/2004	1	326	0.149
SWAMP	9/22/2004	1	374	0.228
SWAMP	9/22/2004	1	430	0.292
SWAMP	9/22/2004	1	520	0.499
SGWC	11/2/2006	5	150	0.328
SGWC	6/6/2007	16	350	0.224
SWAMP	Summer 2007	1	365	0.744
SWAMP	Summer 2007	1	375	0.451
SWAMP	Summer 2007	1	385	0.713
SWAMP	Summer 2007	1	351	0.346
SWAMP	Summer 2007	1	370	0.417
SWAMP	Summer 2007	1	367	0.463
SWAMP	Summer 2007	1	387	0.623
SWAMP	Summer 2007	1	371	0.311
SWAMP	Summer 2007	1	317	0.229
SWAMP	Summer 2007	1	365	0.532
SWAMP	Summer 2007	1	432	0.723
SWAMP	Summer 2007	1	598	0.535

Program	Date	Number in Sample	Fish Length (mm)	Total Mercury Concentration (ppm wet weight)
SWAMP	Summer 2007	1	258	0.253
SWAMP	Summer 2007	1	255	0.158
SWAMP	Summer 2007	1	220	0.114
SWAMP	Summer 2007	1	200	0.115

Figure G-47 shows the mercury concentrations in largemouth bass plotted against length, which is an approximate surrogate for age. For composite fish samples, concentration is plotted against mean length. As expected, fish tissue mercury concentrations increase with length. Concentrations exceed 0.22 ppm in all individual or composite samples greater than 345 mm. Twenty-three individual and five composite samples exceed the fish tissue target; five individual samples and one composite had concentrations less than the target.

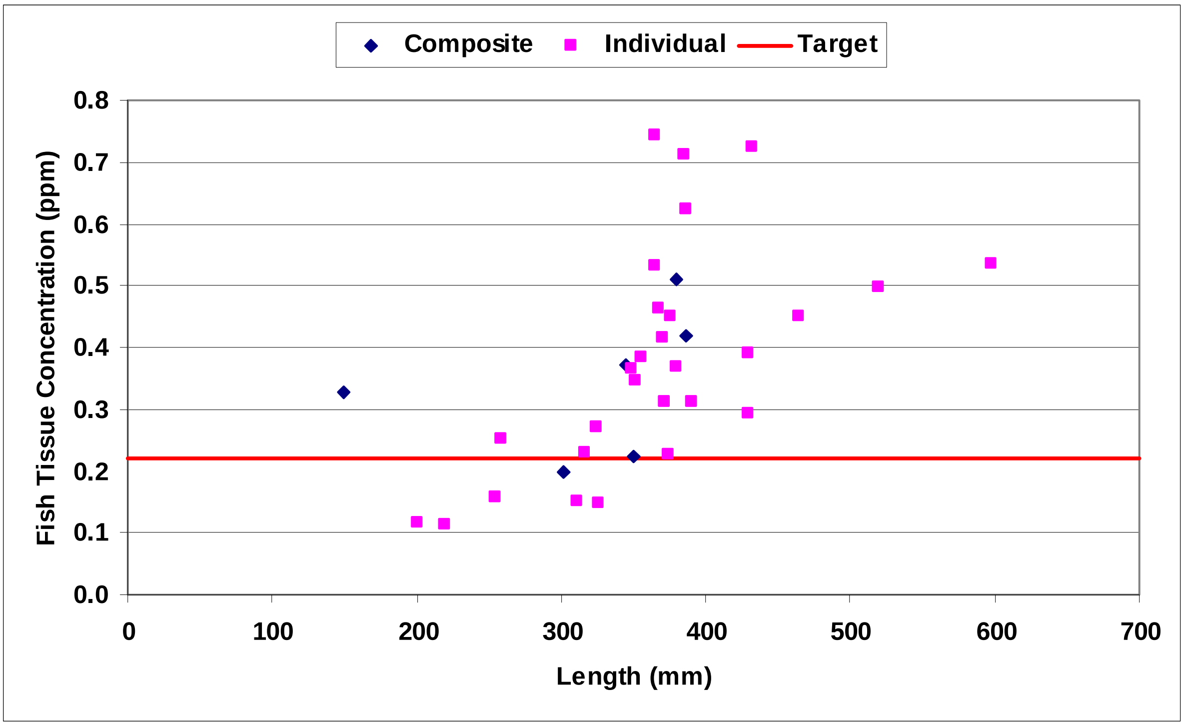


Figure G-47. Mercury Concentrations in Largemouth Bass in Puddingstone Reservoir

G.10.2.3 Tributary/Inflow Monitoring

G.10.2.1.3 Water Column Measurements

In February 2009, USEPA and the Regional Board sampled Live Oak Wash and the storm drain near the campground for total and methylmercury in the water column. The total mercury concentration measured from these two inputs ranged from 2.65 ng/L to 3.52 ng/L. Total mercury was analyzed with EPA Method 1631 with a detection limit of 3.03 ng/L. Methylmercury concentrations ranged from less than the detection limit to 0.043 ng/L. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.020 ng/L. The percent of mercury in the methyl form was 1.2 percent in the sample where methylmercury was greater than the detection limit.

Inflow water column measurements were collected again in the summer of 2009. Total mercury was analyzed with EPA Method 1631 with a detection limit of 0.15 ng/L; methyl mercury was analyzed with EPA Method 1630 with a detection limit of 0.020 ng/L. Concentrations of methyl and total mercury in Live Oak Wash were 0.553 ng/L and 4.24 ng/L, respectively. Concentrations measured in PRSD2 were 0.046 ng/L and 7.55 ng/L, respectively. [Note that storm drain PRSD was not flowing during this sampling event.]

Table G-119 shows the tributary and storm drain water column measurements for Puddingstone Reservoir.

Table G-119. Tributary/Inflow Water Column Measurements for Puddingstone Reservoir

Location	Date	Time	MeHg (ng/L)	Total Hg (ng/L)	TSS (mg/L)
PR11	2/24/2009	14:30	0.043	3.52	5.8
PRSD		13:10	<0.020	2.65	5.7
PR11	7/16/2009	11:45	0.553	4.24	3.6
PRSD2		13:10	0.046	7.55	3.8

G.10.2.1.3 Sediment Samples

During the February 2009 monitoring event, a sediment sample was collected from Live Oak Wash (PR11). The storm drain near the campground (PRSD) was not sampled for sediment because the only solid material evident at the discharge was leaves. The total mercury concentration of the Live Oak Wash sample was 52.9 µg/kg. The methylmercury concentration was less than the detection limit . Total mercury was analyzed with EPA Method 1631 with a detection limit of 2.61 µg/kg. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.011 µg/kg.

In July 2009, sediment samples were collected from four inlet locations. Concentrations of methyl and total mercury at Live Oak Wash were 1.71 µg/kg and 73.1 µg/kg, respectively. Concentrations were also measured in the overland flow ditch (PR19) and storm drain (PR19SD) present on the south side of the reservoir. Concentrations of methyl and total mercury from the ditch were 0.068 µg/kg and 34.3 µg/kg. the storm drain had concentrations of 0.940 µg/kg and 66.2 µg/kg, respectively. Concentrations measured at PRSD2 were 1.14 µg/kg and 50.4 µg/kg, respectively. Total mercury was analyzed with EPA Method 1631 with detection limits ranging from 1.28 µg/kg to 3.03 µg/kg. Methylmercury was analyzed with EPA Method 1630 with detection limits ranging from 0.011 µg/kg to 0.025 µg/kg. Detection limits were adjusted to account for sample aliquot size.

Table G-120 presents the sediment concentrations measured in the inputs to Puddingstone Reservoir. Concentrations are reported on a dry weight basis.

Table G-120. Inflow Sediment Concentrations for Puddingstone Reservoir

Location	Date	Time	MeHg (µg/kg)	Total Hg (µg/kg)	TSS (%)	Sulfate (mg/kg)
PR11	2/24/2009	14:30	<0.011	52.9	74.63	79.95
PR11	7/16/2009	11:45	1.71	73.1	35.59	98.92
PR19		14:05	0.068	34.3	81.19	73.02

Location	Date	Time	MeHg (µg/kg)	Total Hg (µg/kg)	TSS (%)	Sulfate (mg/kg)
PR19SD		14:10	0.940	66.2	37.62	138.93
PRSD2		13:10	1.14	50.4	34.58	163.86

G.10.3 Monitoring Related to Organochlorine Pesticides and PCBs Impairments

An OC Pesticides and PCBs TMDL has been developed for Puddingstone Reservoir. The reservoir is impaired by DDT, chlordane, and PCBs. The Regional Board, UCLA, SWAMP, and TSMP report organic data for Puddingstone from several different media. Levels of OC Pesticides and PCBs have been analyzed in the water column, lake sediment, suspended sediments, fish, porewater and suspended sediment in the porewater. The existing data for chlordane, DDT, dieldrin, and PCBs are summarized in this section. Puddingstone Reservoir is not listed for a dieldrin impairment, however dieldrin data are included for potential future needs and because nearby lakes (Echo and Peck Road Park Lakes) are impaired by this pesticide.

G.10.3.1 Water Column Data Observed in Puddingstone Reservoir

Water sampling was conducted for the UCLA study in the fall of 2008 at PR11, PR-14, and PR-15. The only analyte quantified was PCB-5 (17.95 ng/L) at PR-15. Results are shown in Table G-121.

Table G-121. Water Column Measurements at Puddingstone Reservoir in Fall 2008

	PR-11			PR-11 (field dup)			PR-14			PR-15		
	DL	RL	Result	DL	RL	Result	DL	RL	Result	DL	RL	Result
Contaminant	(ng/L)											
Chlordane-gamma	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
Chlordane-alpha	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
4,4'-DDE	3.00	30.00	ND	3.05	30.46	ND	3.05	30.46	ND	3.14	31.41	ND
4,4'-DDD	3.00	30.00	ND	3.05	30.46	ND	3.05	30.46	ND	3.14	31.41	ND
4,4'-DDT	3.00	30.00	ND	3.05	30.46	ND	3.05	30.46	ND	3.14	31.41	ND
Dieldrin	3.00	30.00	ND	3.05	30.46	ND	3.05	30.46	ND	3.14	31.41	ND
PCB 5	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	17.95
PCB 18	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 31	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 52	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 44	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 66	1.50	15.00	ND	1.52	15.23	3.66*	1.52	15.23	ND	1.57	15.71	ND
PCB 101	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 87	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 151	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 110	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 153	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 141	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 138	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 187	1.50	15.00	ND	1.52	15.23	5.72*	1.52	15.23	ND	1.57	15.71	ND

Contaminant	PR-11			PR-11 (field dup)			PR-14			PR-15		
	DL	RL	Result	DL	RL	Result	DL	RL	Result	DL	RL	Result
	(ng/L)											
PCB 183	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 180	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 170	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND
PCB 206	1.50	15.00	ND	1.52	15.23	ND	1.52	15.23	ND	1.57	15.71	ND

*Results above detection limit but below reporting limit.

The Regional Board collected water samples from several stations on November 18, 2008 and collaborated in sampling efforts with USEPA on February 24, 2009 and July 16, 2009. On November 18, 2008 samples were collected at PR-11, PR-14, PR-15, PR-16 and PR-17. A duplicate sample was taken at Station PR-17. The collected samples were analyzed for Aroclor PCBs, PCBs, and chlorinated pesticides. The Aroclor PCBs tested for included the following congeners: 1016, 1221, 1232, 1242, 1248, 1254, and 1260. No Aroclor PCBs or chlorinated pesticides were detected at any of the sampled locations. Only one PCB congener was quantified in the water samples; PCB-201 was detected at 555.1 ng/L at PR-15. This concentration is well above the criteria for the protection of aquatic life and human health. The results of the November 18th monitoring are shown in Table G-122.

Table G-122. Water Column Measurements at Puddingstone Reservoir on November 18, 2008

Contaminant (ng/L)	PR 11	PR 14	PR 15	PR 16	PR 17		MDL
					Result	Duplicate	
Chlordane-alpha	ND	ND	ND	ND	ND	ND	1.0
Chlordane-gamma	ND	ND	ND	ND	ND	ND	1.0
cis-Nonachlor	ND	ND	ND	ND	ND	ND	1.0
trans-Nonachlor	ND	ND	ND	ND	ND	ND	1.0
Oxychlordane	ND	ND	ND	ND	ND	ND	1.0
2-4’DDD	ND	ND	ND	ND	ND	ND	1.0
2-4’DDE	ND	ND	ND	ND	ND	ND	1.0
2-4’DDT	ND	ND	ND	ND	ND	ND	1.0
4-4’DDD	ND	ND	ND	ND	ND	ND	1.0
4-4’DDE	ND	ND	ND	ND	ND	ND	1.0
4-4’DDT	ND	ND	ND	ND	ND	ND	1.0
Dieldrin	ND	ND	ND	ND	ND	ND	1.0
Aroclor 1016	ND	ND	ND	ND	ND	ND	10.0
Aroclor 1221	ND	ND	ND	ND	ND	ND	10.0
Aroclor 1232	ND	ND	ND	ND	ND	ND	10.0
Aroclor 1242	ND	ND	ND	ND	ND	ND	10.0
Aroclor 1248	ND	ND	ND	ND	ND	ND	10.0
Aroclor 1254	ND	ND	ND	ND	ND	ND	10.0

Contaminant (ng/L)	PR 11	PR 14	PR 15	PR 16	PR 17		MDL
					Result	Duplicate	
Aroclor 1260	ND	ND	ND	ND	ND	ND	10.0
PCB003	ND	ND	ND	ND	ND	ND	1.0
PCB008	ND	ND	ND	ND	ND	ND	1.0
PCB018	ND	ND	ND	ND	ND	ND	1.0
PCB028	ND	ND	ND	ND	ND	ND	1.0
PCB031	ND	ND	ND	ND	ND	ND	1.0
PCB033	ND	ND	ND	ND	ND	ND	1.0
PCB037	ND	ND	ND	ND	ND	ND	1.0
PCB044	ND	ND	ND	ND	ND	ND	1.0
PCB049	ND	ND	ND	ND	ND	ND	1.0
PCB052	ND	ND	ND	ND	ND	ND	1.0
PCB056/060	ND	ND	ND	ND	ND	ND	1.0
PCB066	ND	ND	ND	ND	ND	ND	1.0
PCB070	ND	ND	ND	ND	ND	ND	1.0
PCB074	ND	ND	ND	ND	ND	ND	1.0
PCB077	ND	ND	ND	ND	ND	ND	1.0
PCB081	ND	ND	ND	ND	ND	ND	1.0
PCB087	ND	ND	ND	ND	ND	ND	1.0
PCB095	ND	ND	ND	ND	ND	ND	1.0
PCB097	ND	ND	ND	ND	ND	ND	1.0
PCB099	ND	ND	ND	ND	ND	ND	1.0
PCB101	ND	ND	ND	ND	ND	ND	1.0
PCB105	ND	ND	ND	ND	ND	ND	1.0
PCB110	ND	ND	ND	ND	ND	ND	1.0
PCB114	ND	ND	ND	ND	ND	ND	1.0
PCB118	ND	ND	ND	ND	ND	ND	1.0
PCB119	ND	ND	ND	ND	ND	ND	1.0
PCB123	ND	ND	ND	ND	ND	ND	1.0
PCB126	ND	ND	ND	ND	ND	ND	1.0
PCB128	ND	ND	ND	ND	ND	ND	1.0
PCB138	ND	ND	ND	ND	ND	ND	1.0
PCB141	ND	ND	ND	ND	ND	ND	1.0

Contaminant (ng/L)	PR 11	PR 14	PR 15	PR 16	PR 17		MDL
					Result	Duplicate	
PCB149	ND	ND	ND	ND	ND	ND	1.0
PCB151	ND	ND	ND	ND	ND	ND	1.0
PCB153	ND	ND	ND	ND	ND	ND	1.0
PCB156	ND	ND	ND	ND	ND	ND	1.0
PCB157	ND	ND	ND	ND	ND	ND	1.0
PCB158	ND	ND	ND	ND	ND	ND	1.0
PCB167	ND	ND	ND	ND	ND	ND	1.0
PCB168+132	ND	ND	ND	ND	ND	ND	1.0
PCB169	ND	ND	ND	ND	ND	ND	1.0
PCB170	ND	ND	ND	ND	ND	ND	1.0
PCB174	ND	ND	ND	ND	ND	ND	1.0
PCB177	ND	ND	ND	ND	ND	ND	1.0
PCB180	ND	ND	ND	ND	ND	ND	1.0
PCB183	ND	ND	ND	ND	ND	ND	1.0
PCB187	ND	ND	ND	ND	ND	ND	1.0
PCB189	ND	ND	ND	ND	ND	ND	1.0
PCB194	ND	ND	ND	ND	ND	ND	1.0
PCB195	ND	ND	ND	ND	ND	ND	1.0
PCB200	ND	ND	ND	ND	ND	ND	1.0
PCB201	ND	ND	555.1	ND	ND	ND	1.0
PCB203	ND	ND	ND	ND	ND	ND	1.0
PCB206	ND	ND	ND	ND	ND	ND	1.0
PCB209	ND	ND	ND	ND	ND	ND	1.0

A water sample was collected during field monitoring by the Regional Board on February 24, 2009 at a storm drain flowing to Puddingstone River (PR-SD). The sample was tested for PCBs only (not chlordane, DDTs, or dieldrin). No PCBs were detected in the sample. The detection limit for each PCB congener was 1 ng/L.

On July 16, 2009, water samples were collected at PR-11, PR-15, PR-16, PR-SD2. A duplicate sample was collected at PR-16. Samples were analyzed for chlorinated pesticides and PCB congeners. No analytes were detected in any of the samples (Table G-123).

Table G-123. Water Column Measurements at Puddingstone Reservoir on July 16, 2009

Contaminant (ng/L)	PR-11	PR-15	PR-16		PR-SD2	MDL
			Results	Duplicate		
Chlordane-alpha	ND	ND	ND	ND	ND	1.0
Chlordane-gamma	ND	ND	ND	ND	ND	1.0
cis-Nonachlor	ND	ND	ND	ND	ND	1.0
trans-Nonachlor	ND	ND	ND	ND	ND	1.0
Oxychlordane	ND	ND	ND	ND	ND	1.0
2-4’DDD	ND	ND	ND	ND	ND	1.0
2-4’DDE	ND	ND	ND	ND	ND	1.0
2-4’DDT	ND	ND	ND	ND	ND	1.0
4-4’DDD	ND	ND	ND	ND	ND	1.0
4-4’DDE	ND	ND	ND	ND	ND	1.0
4-4’DDT	ND	ND	ND	ND	ND	1.0
Dieldrin	ND	ND	ND	ND	ND	1.0
PCB003	ND	ND	ND	ND	ND	1.0
PCB008	ND	ND	ND	ND	ND	1.0
PCB018	ND	ND	ND	ND	ND	1.0
PCB028	ND	ND	ND	ND	ND	1.0
PCB031	ND	ND	ND	ND	ND	1.0
PCB033	ND	ND	ND	ND	ND	1.0
PCB037	ND	ND	ND	ND	ND	1.0
PCB044	ND	ND	ND	ND	ND	1.0
PCB049	ND	ND	ND	ND	ND	1.0
PCB052	ND	ND	ND	ND	ND	1.0
PCB056/060	ND	ND	ND	ND	ND	1.0
PCB066	ND	ND	ND	ND	ND	1.0
PCB070	ND	ND	ND	ND	ND	1.0
PCB074	ND	ND	ND	ND	ND	1.0
PCB077	ND	ND	ND	ND	ND	1.0
PCB081	ND	ND	ND	ND	ND	1.0
PCB087	ND	ND	ND	ND	ND	1.0
PCB095	ND	ND	ND	ND	ND	1.0
PCB097	ND	ND	ND	ND	ND	1.0
PCB099	ND	ND	ND	ND	ND	1.0
PCB101	ND	ND	ND	ND	ND	1.0
PCB105	ND	ND	ND	ND	ND	1.0
PCB110	ND	ND	ND	ND	ND	1.0
PCB114	ND	ND	ND	ND	ND	1.0
PCB118	ND	ND	ND	ND	ND	1.0

Contaminant (ng/L)	PR-11	PR-15	PR-16		PR-SD2	MDL
			Results	Duplicate		
PCB119	ND	ND	ND	ND	ND	1.0
PCB123	ND	ND	ND	ND	ND	1.0
PCB126	ND	ND	ND	ND	ND	1.0
PCB128	ND	ND	ND	ND	ND	1.0
PCB138	ND	ND	ND	ND	ND	1.0
PCB141	ND	ND	ND	ND	ND	1.0
PCB149	ND	ND	ND	ND	ND	1.0
PCB151	ND	ND	ND	ND	ND	1.0
PCB153	ND	ND	ND	ND	ND	1.0
PCB156	ND	ND	ND	ND	ND	1.0
PCB157	ND	ND	ND	ND	ND	1.0
PCB158	ND	ND	ND	ND	ND	1.0
PCB167	ND	ND	ND	ND	ND	1.0
PCB168+132	ND	ND	ND	ND	ND	1.0
PCB169	ND	ND	ND	ND	ND	1.0
PCB170	ND	ND	ND	ND	ND	1.0
PCB174	ND	ND	ND	ND	ND	1.0
PCB177	ND	ND	ND	ND	ND	1.0
PCB180	ND	ND	ND	ND	ND	1.0
PCB183	ND	ND	ND	ND	ND	1.0
PCB187	ND	ND	ND	ND	ND	1.0
PCB189	ND	ND	ND	ND	ND	1.0
PCB194	ND	ND	ND	ND	ND	1.0
PCB195	ND	ND	ND	ND	ND	1.0
PCB200	ND	ND	ND	ND	ND	1.0
PCB201	ND	ND	ND	ND	ND	1.0
PCB203	ND	ND	ND	ND	ND	1.0
PCB206	ND	ND	ND	ND	ND	1.0
PCB209	ND	ND	ND	ND	ND	1.0

G.10.3.2 Porewater Data Observed in Puddingstone Reservoir

Porewater and TSS from porewater were analyzed in Puddingstone Reservoir in fall 2008 as part of the UCLA study. PR-14 and PR-15 were sampled, as shown in Table G-124 (see Stenstrom et al., 2009 for raw data). Chlordane, DDT, and dieldrin were not detected in any of the samples. PCB-31 was detected in the porewater at PR-14 and PR-15 and in the suspended sediment at PR-14, but not at reportable levels (DNQ).

Table G-124. Porewater Measurements at Puddingstone Reservoir in Fall 2008

Contaminant (ng/L)	Porewater (ng/L)			TSS in Porewater (µg/kg)		
	PR-14	PR-15	MDL	PR-14	PR-15	MDL
Chlordane	ND	ND	15	ND	ND	0.2-0.53
DDT	ND	ND	30	ND	ND	0.4-1.06
Dieldrin	ND	ND	30	ND	ND	0.4-1.06
Total PCBs	DNQ[1]	DNQ[1]	15	DNQ[1]	ND	0.2-0.53

[1] PCB-31 was detected at less than reporting level (150 ng/L for porewater and 3.01 µg/kg for TSS in porewater).

G.10.3.3 Fish Tissue Levels Observed in Puddingstone Reservoir

Concentrations of the organochlorides and PCBs in fish from Puddingstone Reservoir are shown below in Table G-125. The common carp in Puddingstone Reservoir had the highest average concentrations of Aroclor PCBs, chlordane, DDT, and dieldrin. The chlordane and DDT average concentrations for all fish species were above the FCGs. In common carp samples, the average chlordane concentration was 119.6 ppb and the average DDT level was 232.8 ppb. The average concentration in bullhead and largemouth bass for chlordane was 46.5 and 10.5 ppb, and 71.0 and 20.9 ppb for DDTs, respectively. Levels of PCBs were 60.2, 125.5, and 17.2 ppb for bullhead, common carp, and largemouth bass, respectively. Dieldrin concentrations were non-detect for bullhead and 4.6 and 1.2 ppb for common carp and largemouth bass, respectively.

Table G-125. OC Pesticides and PCBs Fish Tissue Data for Puddingstone Reservoir

Agency	Pollutant	Sample Date	Common Name	Concentration (ppb, w wt)	Mean Length (mm)	Mean Weight (g)
TSMP	Aroclor PCBs	4/28/1992	Largemouth Bass	65	386	1,268.7
TSMP	Aroclor PCBs	5/6/1986	Common Carp	590	566	4474
TSMP	Aroclor PCBs	8/10/1999	Largemouth Bass	13	345	816.6
TSMP	Aroclor PCBs	6/16/1987	Common Carp	160	557	362.2
TSMP	Aroclor PCBs	6/22/1988	Brown Bullhead	66	315	538.7
TSMP	Aroclor PCBs	6/16/1987	Bullhead	ND	282	350.1
TSMP	Aroclor PCBs	6/11/1991	Largemouth Bass	54	380	1,030.4
TSMP	Aroclor PCBs	5/6/1986	Largemouth Bass	ND	302	509.7
TSMP	Chlordane	4/28/1992	Largemouth Bass	31.7	386	1,268.7
TSMP	Chlordane	5/6/1986	Common Carp	460	566	4474
TSMP	Chlordane	8/10/1999	Largemouth Bass	2.8	345	816.6
TSMP	Chlordane	6/16/1987	Common Carp	193.5	557	362.2
TSMP	Chlordane	6/22/1988	Brown Bullhead	48.5	315	538.7
TSMP	Chlordane	6/16/1987	Bullhead	44.4	282	350.1
TSMP	Chlordane	6/11/1991	Largemouth Bass	16.1	380	1,030.4
TSMP	Chlordane	5/6/1986	Largemouth Bass	10.4	302	509.7

Agency	Pollutant	Sample Date	Common Name	Concentration (ppb, w wt)	Mean Length (mm)	Mean Weight (g)
SWAMP	Chlordane	Summer 2007	Largemouth Bass	9.29	366	NA
SWAMP	Chlordane	Summer 2007	Largemouth Bass	4.97	365	NA
SWAMP	Chlordane	9/22/2004	Largemouth Bass	12.43	397.6	799.6
SWAMP	Chlordane	9/22/2004	Largemouth Bass	5.95	343	563.1
SWAMP	Chlordane	9/22/2004	Largemouth Bass	13.55	456.6	1,464.6
SWAMP	Chlordane	9/22/2004	Largemouth Bass	7.31	342.6	581.6
SWAMP	Chlordane	9/22/2004	Common Carp	1.17	420.7	1,203.9
SWAMP	Chlordane	9/22/2004	Common Carp	27.25	632.7	3,795
SWAMP	Chlordane	9/22/2004	Common Carp	19.98	593.7	2,631
SWAMP	Chlordane	9/22/2004	Common Carp	15.60	669	4,354.7
TSMP	DDT	4/28/1992	Largemouth Bass	36	386	1,268.7
TSMP	DDT	5/6/1986	Common Carp	880	566	4474
TSMP	DDT	8/10/1999	Largemouth Bass	10.7	345	816.6
TSMP	DDT	6/16/1987	Common Carp	358	557	362.2
TSMP	DDT	6/22/1988	Brown Bullhead	72	315	538.7
TSMP	DDT	6/16/1987	Bullhead	70	282	350.1
TSMP	DDT	6/11/1991	Largemouth Bass	25	380	1,030.4
TSMP	DDT	5/6/1986	Largemouth Bass	16	302	509.7
SWAMP	DDT	Summer 2007	Largemouth Bass	10.8	365	NA
SWAMP	DDT	Summer 2007	Largemouth Bass	30.77	366	NA
SWAMP	DDT	9/22/2004	Largemouth Bass	33.72	397.6	799.6
SWAMP	DDT	9/22/2004	Largemouth Bass	15.561	343	563.1
SWAMP	DDT	9/22/2004	Largemouth Bass	35.34	456.6	1,464.6
SWAMP	DDT	9/22/2004	Largemouth Bass	19.42	342.6	581.6
SWAMP	DDT	9/22/2004	Common Carp	2.51	420.7	1,203.9
SWAMP	DDT	9/22/2004	Common Carp	69.357	632.7	3,795
SWAMP	DDT	9/22/2004	Common Carp	47.66	593.7	2,631
SWAMP	DDT	9/22/2004	Common Carp	39.082	669	4,354.7
TSMP	Dieldrin	4/28/1992	Largemouth Bass	ND	386	1,268.7
TSMP	Dieldrin	5/6/1986	Common Carp	12	566	4474
TSMP	Dieldrin	8/10/1999	Largemouth Bass	ND	345	816.6
TSMP	Dieldrin	6/16/1987	Common Carp	5	557	362.2
TSMP	Dieldrin	6/22/1988	Brown Bullhead	ND	315	538.7
TSMP	Dieldrin	6/16/1987	Bullhead	ND	282	350.1
TSMP	Dieldrin	6/11/1991	Largemouth Bass	ND	380	1,030.4
TSMP	Dieldrin	5/6/1986	Largemouth Bass	ND	302	509.7
SWAMP	Dieldrin	9/22/2004	Largemouth Bass	1.73	397.6	799.6
SWAMP	Dieldrin	9/22/2004	Largemouth Bass	0.858	343	563.1

Agency	Pollutant	Sample Date	Common Name	Concentration (ppb, w wt)	Mean Length (mm)	Mean Weight (g)
SWAMP	Dieldrin	9/22/2004	Largemouth Bass	1.58	456.6	1,464.6
SWAMP	Dieldrin	9/22/2004	Largemouth Bass	1.16	342.6	581.6
SWAMP	Dieldrin	9/22/2004	Common Carp	0.704	420.7	1,203.9
SWAMP	Dieldrin	9/22/2004	Common Carp	4.34	632.7	3,795
SWAMP	Dieldrin	9/22/2004	Common Carp	3.35	593.7	2,631
SWAMP	Dieldrin	9/22/2004	Common Carp	2.48	669	4,354.7
SWAMP	Dieldrin	Summer 2007	Largemouth Bass	0.68	366	NA
SWAMP	Dieldrin	Summer 2007	Largemouth Bass	ND	365	NA
SWAMP	PCB	9/22/2004	Largemouth Bass	29.108	397.6	799.6
SWAMP	PCB	9/22/2004	Largemouth Bass	16.024	343	563.1
SWAMP	PCB	9/22/2004	Largemouth Bass	35.87	456.6	1,464.6
SWAMP	PCB	9/22/2004	Largemouth Bass	17.85	342.6	581.6
SWAMP	PCB	9/22/2004	Common Carp	6.461	420.7	1203.9
SWAMP	PCB	9/22/2004	Common Carp	49.304	632.7	3,795
SWAMP	PCB	9/22/2004	Common Carp	36.799	593.7	2,631
SWAMP	PCB	9/22/2004	Common Carp	28.314	669	4,354.7
SWAMP	PCB	Summer 2007	Largemouth Bass	19	366	NA
SWAMP	PCB	Summer 2007	Largemouth Bass	8	365	NA
Regional Board	Total Detectable DDTs	11/3/2006	Bass	25.6	NA	NA
Regional Board	Total Detectable DDTs	11/3/2006	Bass	10.1	NA	NA
Regional Board	Total Detectable PCBs	11/3/2006	Bass	ND	NA	NA
Regional Board	Total Detectable PCBs	11/3/2006	Bass	3.3	NA	NA
Regional Board	Chlordane	11/3/2006	Bass	1.1	NA	NA
Regional Board	Chlordane	11/3/2006	Bass	ND	NA	NA
Regional Board	Dieldrin	11/3/2006	Bass	ND	NA	NA
Regional Board	Dieldrin	11/3/2006	Bass	ND	NA	NA

ND = non-detect

G.10.3.4 Sediment Data Observed in Puddingstone Reservoir

Sediment samples from Puddingstone Reservoir were collected in the fall of 2008 by UCLA at PR-14 and PR-15. A field duplicate was collected at PR-14 and laboratory duplicates were performed for each sample. Chlordane-gamma was detected at unreportable levels at PR-14 (laboratory duplicate of field duplicate). DDT and dieldrin were not detected in any sample. Four PCB congeners were detected at PR-14 (laboratory duplicate of field duplicate): PCB-5, PCB-31, PCB-66, and PCB-138. The concentration of PCB-5 was 6.78 µg/kg dry weight, and the concentration of PCB-31 was 12.67 µg/kg dry weight, these were the only reportable PCB congeners. PCBs were not detected at PR-15. The results and detection and reporting limits for each contaminant are shown in Table G-126.

Table G-126. OC Pesticides and PCBs Measurements in Sediment at Puddingstone Reservoir in Fall 2008

	PR-14			**PR-14 (lab dup)**			**PR-14 (field dup)**			**PR-14 (lab dup of field dup)**			**PR-15**			**PR-15 (lab dup)**		
	DL	**RL**	**Result**	**DL**	**RL**	**Result**	**DL**	**RL**	**Result**	**DL**	**RL**	**Result**	**DL**	**RL**	**Result**	**DL**	**RL**	**Result**
Contaminant	**µg/kg dry suspended solids**																	
Chlordane-gamma	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	0.14*	1.58	15.83	ND	1.58	15.84	ND
Chlordane-alpha	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
4,4'-DDE	0.99	9.91	ND	1.00	9.96	ND	0.79	7.85	ND	0.77	7.74	ND	3.17	31.67	ND	3.17	31.68	ND
4,4'-DDD	0.99	9.91	ND	1.00	9.96	ND	0.79	7.85	ND	0.77	7.74	ND	3.17	31.67	ND	3.17	31.68	ND
4,4'-DDT	0.99	9.91	ND	1.00	9.96	ND	0.79	7.85	ND	0.77	7.74	ND	3.17	31.67	ND	3.17	31.68	ND
Dieldrin	0.99	9.91	ND	1.00	9.96	ND	0.79	7.85	ND	0.77	7.74	ND	3.17	31.67	ND	3.17	31.68	ND
PCB 5	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	6.78	1.58	15.83	ND	1.58	15.84	ND
PCB 18	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 31	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	12.67	1.58	15.83	ND	1.58	15.84	ND
PCB 52	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 44	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 66	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	1.03*	1.58	15.83	ND	1.58	15.84	ND
PCB 101	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 87	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 151	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 110	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 153	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 141	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 138	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	0.97*	1.58	15.83	ND	1.58	15.84	ND
PCB 187	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 183	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 180	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 170	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND
PCB 206	0.50	4.95	ND	0.50	4.98	ND	0.39	3.93	ND	0.39	3.87	ND	1.58	15.83	ND	1.58	15.84	ND

*Results above detection limit, but below reporting limit.

Sediment samples were collected by the Regional Board and USEPA on July 16, 2009 at PR-11, PR-15, PR-16, PR-19, PR-19SD, and PR-SD2. Chlordane was quantified at all stations, between 1.1 and 6.5 µg/kg dry weight. The chlordane levels at PR-11 and PR-15 were above the TEC CBSQG (3.24 µg/kg dry weight). DDT was only detected at PR-19; however, DDE was detected at almost all stations. PCBs were also detected at all locations except PR-19. Table G-127 shows the results of the July 16, 2009 monitoring.

Table G-127. OC Pesticides and PCBs Measurements in Sediment at Puddingstone Reservoir on July 16, 2009

Contaminant (µg/kg dry weight)	PR-11	PR-15	PR-16	PR-19	PR-19SD	PR-SD2	MDL
Chlordane	6.5	4.1	2.4	1.1	2.6	2.2	1
DDE	5.2	18.6	11.8	6.1	8.5	ND	1
DDT	ND	ND	ND	1.7	ND	ND	1
Dieldrin	ND	ND	ND	ND	ND	ND	1
PCB099	1	1.3	1.6	ND	ND	ND	1
PCB101	1.45	1.45	1.8	ND	ND	ND	1
PCB110	1.4	1.2	1.3	ND	ND	1	1
PCB118	ND	1.2	ND	ND	ND	ND	1
PCB119	ND	ND	ND	ND	193.7	ND	1
PCB138	ND	1.8	ND	ND	1	ND	1
PCB153	1.4	1.8	ND	ND	ND	ND	1
PCB174	ND	ND	1.1	ND	ND	ND	1
PCB180	2.1	1.7	1.5	ND	ND	ND	1
Total PCBs	7.4	10.5	7.3	ND	194.7	1.0	1

G.10.3.5 Suspended Sediment Data Observed in Puddingstone Reservoir

Samples of suspended solids were collected at PR-11, PR-14, and PR-15 in the fall of 2008 by UCLA. Chlordane-gamma was detected at unreportable levels at PR-11. In each sample except one of the PR-11 duplicates, PCBs were detected below reporting limits. The individual PCBs detected at each station and other results of the fall 2008 TSS analysis are shown in Table G-128.

Table G-128. OC Pesticides and PCBs Measurements in Suspended Sediment at Puddingstone Reservoir in Fall 2008

	PR-11			PR-11 (dup 1)			PR-11 (dup 2)			PR-14			PR-15		
	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.	DL	RL	Res.
Contaminant	µg/kg dry suspended solids														
Chlordane-gamma	29.07	290.70	ND	2.38	23.79	10.30*	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
Chlordane-alpha	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
4,4'-DDE	58.14	581.40	ND	4.76	47.59	ND	4.23	42.27	ND	44.23	442.26	ND	72.46	724.64	ND
4,4'-DDD	58.14	581.40	ND	4.76	47.59	ND	4.23	42.27	ND	44.23	442.26	ND	72.46	724.64	ND

	PR-11			**PR-11 (dup 1)**			**PR-11 (dup 2)**			**PR-14**			**PR-15**		
	DL	**RL**	**Res.**	**DL**	**RL**	**Res.**	**DL**	**RL**	**Res.**	**DL**	**RL**	**Res.**	**DL**	**RL**	**Res.**
Contaminant	**µg/kg dry suspended solids**														
4,4'-DDT	58.14	581.40	ND	4.76	47.59	ND	4.23	42.27	ND	44.23	442.26	ND	72.46	724.64	ND
Dieldrin	58.14	581.40	ND	4.76	47.59	ND	4.23	42.27	ND	44.23	442.26	ND	72.46	724.64	ND
PCB 5	29.07	290.70	132.56*	2.38	23.79	7.77*	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 18	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 31	29.07	290.70	ND	2.38	23.79	14.75*	2.11	21.14	ND	22.11	221.13	61.18*	36.23	362.32	ND
PCB 52	29.07	290.70	256.22*	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 44	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 66	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	62.36*
PCB 101	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 87	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 151	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 110	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 153	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 141	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 138	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	98.23*
PCB 187	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 183	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 180	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 170	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND
PCB 206	29.07	290.70	ND	2.38	23.79	ND	2.11	21.14	ND	22.11	221.13	ND	36.23	362.32	ND

*Results are above detection levels but below reporting levels.

In the fall of 2008, a TSS sample was collected at PR-11 during a wet weather event (Table G-129). A composite sample from the event did not detect any of the pollutants. A grab sample from PR-11 was collected 90 minutes into the wet weather event also had no detectable results. Water column samples were also collected during this event (a time series composite and a single time point sample), but not analyzed.

Table G-129. Wet Weather OC Pesticides and PCBs Measurements in Suspended Sediment at Puddingstone Reservoir in Fall 2008

Contaminant (µg/kg dry suspended solids)	**PR-11 Storm Composite**	**Composite MDL**	**PR-11 Storm @ 1.5 hours**	**Grab Sample MDL**
Chlordane	ND	1.57	ND	2.70
DDT	ND	3.14	ND	5.39
Dieldrin	ND	3.14	ND	5.39
Total PCBs	ND	1.57	ND	2.70

(This page left intentionally blank.)

G.11 Monitoring Data for Santa Fe Dam Park Lake

Monitoring data relevant to the impairments of Santa Fe Dam Park Lake are available from 1992, 1993, 2009, and 2010. Figure G-48 shows the historical and recent monitoring locations for these lakes.

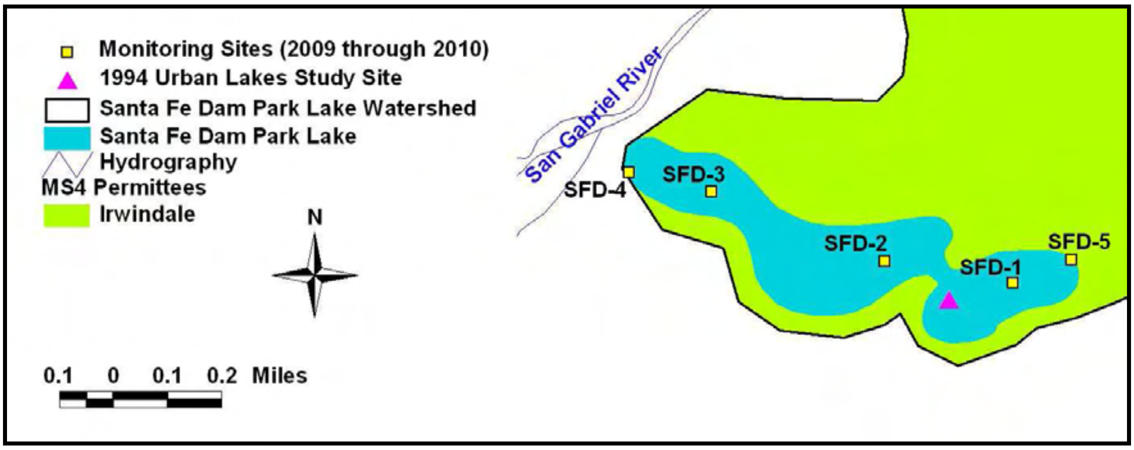


Figure G-48. Santa Fe Dam Park Lake Monitoring Sites

G.11.1 MONITORING RELATED TO NUTRIENT IMPAIRMENTS

In 1992 and 1993, Santa Fe Dam Park Lake was monitored for water quality as part of the Urban Lakes Study (Table G-130). The station was located in the southeast end of the lake near the spillway (pink triangle, Figure G-48) (UC Riverside, 1994). TKN ranged from 0.3 mg/L to 1.1 mg/L. Ammonium generally ranged from 0.1 mg/L to 0.2 mg/L with 21 measurements less than the reporting limit and one measurement of 0.4 mg/L collected at a depth of 2 meters. The upper range of these concentrations are above the chronic target, but below the acute target (for assessment purposes, we are assuming that the analysis methodology converted all ammonia to ammonium). All 37 samples of nitrite were less than the reporting limit, and the majority of nitrate samples (32) were less than the reporting limit; the maximum observed nitrate concentration was 0.2 mg/L. All orthophosphate and total phosphorus concentrations were less than the reporting limit except one total phosphorus concentration which measured 0.1 mg/L. pH ranged from 8.0 to 9.6, and TOC ranged from 2.3 mg/L to 3.4 mg/L. The summary table from the 1994 Lakes Study Report (UC Riverside, 1994) lists chlorophyll *a* concentrations ranging from 1 µg/L to 29 µg/L with an average of 13 µg/L; the raw data were not available.

Table G-130. Santa Fe Dam Park Lake 1992/1993 Monitoring Data

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
8/10/1992	0	0.8	0.1	<0.01	<0.01	<0.01	<0.01	9.1	3.3	256
	2	0.9	0.4	<0.01	<0.01	<0.01	<0.01	9.2	3.3	279
	3.5	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	9.1	3	296
8/10/1992	0	0.9	0.2	<0.01	<0.01	<0.01	<0.01	8.9	2.7	274
	2.5	0.9	0.1	<0.01	0.1	<0.01	<0.01	8.7	2.9	346
8/10/1992	0	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	9.1	3.3	268
	2.5	0.8	<0.01	<0.01	<0.01	<0.01	<0.01	9.1	3.3	309
9/10/1992	0	1.1	0.1	<0.01	<0.01	<0.01	<0.01	8.9	3.1	284
	2	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.9	3	287
	3.5	0.9	<0.01	<0.01	<0.01	<0.01	<0.01	8.9	2.9	281
10/13/1992	0	0.9	0.1	<0.01	<0.01	<0.01	<0.01	8.7	2.6	286
	2	0.8	0.1	<0.01	<0.01	<0.01	<0.01	8.7	2.7	316
	3.5	0.8	0.1	<0.01	<0.01	<0.01	<0.01	8.6	2.8	301
11/3/1992	0	0.7	0.1	<0.01	0.2	<0.01	<0.01	8.7	3.1	231
	1.5	0.7	0.1	<0.01	0.1	<0.01	<0.01	8.8	3.1	252
	2.5	0.7	<0.01	<0.01	0.1	<0.01	<0.01	8.7	3.2	282
12/10/1992	0	0.6	<0.01	<0.01	0.1	<0.01	<0.01	8.7	2.6	286
	2.5	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	8.6	2.8	284
	3.5	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	8.6	2.8	327
1/14/1993	0	0.5	<0.01	<0.01	<0.01	<0.01	<0.01	8.3	2.7	181
	2	0.5	<0.01	<0.01	<0.01	<0.01	<0.01	8.4	2.8	183
	3.5	0.5	<0.01	<0.01	<0.01	<0.01	<0.01	8.4	2.8	189
2/3/1993	0	0.7	0.2	<0.01	<0.01	<0.01	<0.01	8	2.4	221
	2	0.6	0.2	<0.01	<0.01	<0.01	<0.01	8.2	2.6	251
	3	0.7	0.2	<0.01	<0.01	<0.01	<0.01	8.2	2.3	229
3/9/1993	0	0.6	0.1	<0.01	<0.01	<0.01	<0.01	8.1	2.7	212
	2	0.7	0.1	<0.01	<0.01	<0.01	<0.01	8.3	2.5	201
	3.5	0.7	0.1	<0.01	<0.01	<0.01	<0.01	8.3	2.6	223
4/14/1993	0	0.7	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	3.4	247
	1.5	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	2.5	235
	2.5	0.7	<0.01	<0.01	<0.01	<0.01	<0.01	8.7	2.7	256

Date	Depth (m)	TKN (mg/L)	NH_4-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	pH	TOC (mg/L)	TDS (mg/L)
5/25/1993	0	0.4	<0.01	<0.01	<0.01	<0.01	<0.01	8.9	3	257
	1.5	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	8.9	3.2	248
	2.5	0.6	<0.01	<0.01	<0.01	<0.01	<0.01	8.9	3.3	246
6/21/1993	0	0.3	<0.01	<0.01	<0.01	<0.01	0.1	9.5	2.7	249
	1.5	0.3	<0.01	<0.01	<0.01	<0.01	<0.01	9.6	2.9	252
	2.5	0.3	<0.01	<0.01	<0.01	<0.01	<0.01	9.6	3.1	242

The 1996 Water Quality Assessment Report states that pH was partially supporting the aquatic life use and not supporting the contact recreation and secondary drinking water uses. Ninety-five measurements of pH were taken, ranging from 7.5 to 9.6. The associated database did not contain the raw data for these samples.

On March 3 and August 3, 2009, USEPA and the Regional Board sampled water quality in the Santa Fe Dam Park Lake (Table G-131). The field notes indicate that water is pumped from an underground well to fill the lake every night. The well water enters the lake via a rock stream about 50 ft from SFD-4. Potable water is also input at SFD-3 from the Valley County Water District. During the swimming season, lake water is treated with chlorine several days a week. The chlorine is mixed with lake water in a pump house. Three samples were collected in the lake during both sampling events. During the winter sampling, two shoreline samples were collected on the western and eastern ends of the lakes. In the summer, the well water was sampled. Overall, both nitrogen and phosphorus levels were very low. Chlorophyll *a* concentrations in the lake did not exceed 20.5 µg/L; a shoreline sample had a chlorophyll *a* concentration of 25.8 µg/L. In August, chlorophyll *a* was below the detection level. The field notes report that the lake was very green in August and the Secchi depth readings were shallow, indicative of algal production. A less common chlorophyll structure, e.g. Chlorophyll *b*, could be present in the lake. The average depths at SFD-1 and SFD-2 were 2.93 and 3.02 meters, respectively. The depth at SFD-3 averaged 2.5 meters.

Table G-131. 2009 In-lake and Shoreline Water Column Measurements for Santa Fe Dam Park Lake

Date	Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	Chloro-phyll *a* (µg/L)	Secchi Depth (m)
3/3/2009	**In-lake Samples**									
	SFD-1	10:00	1.1	<0.03	0.04	0.1	<0.0075	0.025	20.5	0.61
	SFD-2	10:40	1.1	0.05	0.04	0.08	<0.0075	0.021	14.4	0.84
	SFD-3	11:10	0.84	0.03	0.04	0.06	<0.0075	0.03	16.7	0.84
	Shoreline Samples									
	SFD-4	12:40	0.98	0.03	0.03	0.04	<0.0075	0.028	14.0	NA
	SFD-4 (duplicate)	13:00	1.1	<0.03	0.03	0.04	<0.0075	0.028	11.6	NA
	SFD-5	13:30	0.98	0.03	0.03	0.14	<0.0075	0.036	25.8	NA

Date	Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	Chlorophyll *a* (µg/L)	Secchi Depth (m)
8/3/2009	**In-lake Samples**									
	SFD-1	10:45	0.58	<0.03	0.04	<0.01	<0.0075	0.027	<1	0.61
	SFD-2	9:20	0.47	<0.03	<0.01	<0.01	<0.0075	0.036	<1	0.56
	SFD-3	8:40	<0.46	<0.03	<0.01	<0.01	<0.0075	0.050	<1	0.46
	Well sample									
	Well SFD-1	13:30	<0.456	<0.03	<0.01	2.985	0.016	NS	NS	NA

Supplemental water quality samples were collected from Santa Fe Dam Park Lake. Table G-132 presents the chloride, sulfate, total alkalinity, total dissolved solids, and total organic carbon data measured in the lake. Temperature and pH measurements reported in the field notes are also shown in this table. The additional chloride added to the lake in the summer is apparent in the higher concentrations measured during the August sampling event. The average chloride concentration in the lake during August was 35.4 mg/L and 28.1 mg/L in March. Temperature, alkalinity, total hardness, total dissolved solids, and TSS also increased during the summer. The pH range remained similar for both sampling periods. The TOC was slightly lower in the summer, between 3.5 and 3.7 mg/L; the winter range of TOC was 4.0 to 5.2 mg/L.

Table G-132. 2009 Supplemental Water Quality Monitoring for Santa Fe Dam Park Lake

Date	Location	Time	Chloride (mg/L)	Temperature (^oC)	pH	Total Alkalinity (mg/L)	Total Hardness as $CaCO_3$ (mg/L)	TDS (mg/L)	TSS (mg/L)	TOC (mg/L)
3/3/2009	**In-lake Samples**									
	SFD-1	10:00	28.18	15.0	8.6	118	107.5	284	6.9	4.0
	SFD-2	10:40	27.76	15.0	8.7	118	104.9	314	5.2	4.6
	SFD-3	11:10	28.30	15.0	8.7	114	103.0	314	6.7	5.0
	Shoreline Samples									
	SFD-4	12:40	27.88	16.0	8.7	114	103.1	290	8.5	5.2
	SFD-4 (duplicate)	13:00	27.88	16.0	8.7	112	102.0	292	8.2	4.5
	SFD-5	13:30	27.79	16.0	8.7	116	101.8	286	10.5	4.7
8/3/2009	**In-lake Samples**									
	SFD-1	10:45	35.63	28.5	8.8	126	131.3	286	9.5	3.5
	SFD-2	9:20	35.23	27.5	8.7	124	133.1	306	9.6	3.7
	SFD-3	8:40	35.23	27.2	8.7	122	131.3	316	14.8	3.5

On May 4, 2009, Clean Lakes Inc. was contracted by the Los Angeles County Department of Parks and Recreation to conduct baseline water quality monitoring (Table G-133) of Santa Fe Dam Park Lake to determine if aquatic weed or algal growth controls were needed. Three locations were sampled in the lake at a depth of approximately 1 ft below the water surface. The location numbering and locations correspond to SFD-1, SFD-2, and SFD-3 monitored by the Regional Board and USEPA.

Table G-133. In-lake Water Column Measurements for Santa Fe Dam Park Lake (5/4/09)

Date	Location	Time	NH_3-N (mg/L)	NO_{2+3}-N (mg/L)	Total P (mg/L)	COD (mg/L)	TSS (mg/L)	Secchi Depth (m)
5/4/2009	SFD-1	9:55	0.24	0.29	<0.01	14	<5	1.7
	SFD-2	10:13	0.47	0.21	<0.01	16	<5	1.4
	SFD-3	10:24	0.17	0.18	<0.01	17	<5	1.5

Four types of alkalinity were also monitored at these locations (Table G-134). Total alkalinity at each station was approximately 140 mg/L in the bicarbonate form.

Table G-134. Alkalinity Measurements for Santa Fe Dam Park Lake (5/4/09)

Date	Location	Time	Total Alkalinity (mg/L)	Carbonate Alkalinity (mg/L)	Bicarbonate Alkalinity (mg/L)	Hydroxide Alkalinity (mg/L)
5/4/2009	SFD-1	9:55	142	< 1	142	< 1
	SFD-2	10:13	142	< 1	142	< 1
	SFD-3	10:24	140	< 1	140	< 1

Clean Lakes, Inc. conducted depth profiles at each location in Santa Fe Dam Park Lake on May 4, 2009 (Table G-135). The pH ranged from 7.39 to 7.96 at all locations and depths. DO ranged from 5.54 mg/L to 8.27 mg/L at all stations and depths with the exception of the bottom reading at station SFD-1 where the DO was 3.72 mg/L. Depth measurements were between 3.18 and 3.25 meters, and Secchi depth readings were between 1.35 and 1.65 meters.

Table G-135. Profile Data Collected in Santa Fe Dam Park Lake (5/4/09)

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Secchi Depth (m)	Total Depth (m)
SFD-1	9:50	0.27	22.44	7.96	8.27	1.65	3.20
		0.67	22.29	7.86	8.17		
		1.35	22.01	7.78	8.03		
		2.01	21.91	7.73	7.73		
		2.66	21.25	7.64	7.18		
		3.29	21.18	7.58	5.54		
		3.48	21.21	7.55	3.72		
SFD-2	9:30	0.30	22.27	7.67	8.19	1.35	3.18

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Secchi Depth (m)	Total Depth (m)
		0.69	22.16	7.64	8.04		
		1.31	21.97	7.6	7.94		
		1.97	24.54	7.59	7.87		
		2.66	21.43	7.52	7.29		
		3.31	21.16	7.43	6.21		
		3.44	24.15	7.39	5.82		
SFD-3	10:20	0.31	21.86	7.59	7.97	1.52	3.25
		0.67	21.81	7.56	7.73		
		1.34	21.45	7.53	7.62		
		2.02	21.3	7.52	7.47		
		2.67	21.18	7.51	7.56		
		3.30	21.17	7.51	7.44		
		3.55	21.17	7.49	7.35		

Profile data for these three sites was also collected by the Regional Board on August 3, 2009 and is listed in Table G-136. The profile data for SFD-1 is shown in Figure G-49. The temperature at this site ranged from 26.3 to 28.5 °C and the pH ranged from 7.45 to 8.75. The DO was greatest at one meter of depth, it ranged from 1.75 to 12.24 mg/L. Morning and afternoon data were collected from SFD-2 and SFD-3, shown in Figure G-50 and Figure G-51. At both sites, the afternoon temperature and DO were slightly higher, especially at the surface. At both stations, below two meters of depth there was no temperature difference between morning and afternoon. Field data were also collected for the groundwater source during this event. After purging the line for approximately 10 minutes, the pH was 7.59 and the temperature was 18.4 ºC.

Table G-136. Profile Data Collected in Santa Fe Dam Park Lake (8/3/09)

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Secchi Depth (m)	Total Depth (m)
SFD-1	10:45	0.10	28.50	8.75	10.74	0.61	2.95
		0.50	28.36	8.73	11.83		
		1.00	28.03	8.73	12.24		
		1.50	27.67	8.55	9.75		
		2.00	27.23	8.39	8.20		
		2.50	26.72	7.83	3.79		
		3.00	26.33	7.45	1.75		
SFD-2	9:30	0.07	27.52	8.59	8.58	0.56	2.90
		0.49	27.53	8.74	9.94		
		0.99	27.50	8.74	10.06		

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Secchi Depth (m)	Total Depth (m)
		1.50	27.46	8.71	10.05		
		1.99	27.44	8.68	9.80		
		2.50	27.33	8.28	6.24		
		2.99	26.89	7.98	4.56		
		2.98	26.91	7.94	3.96		
SFD-2	16:30	0.09	29.09	8.96	11.83	0.56	2.90
		0.50	29.12	8.85	11.87		
		1.01	29.01	8.86	11.84		
		1.50	28.43	8.78	11.12		
		1.99	27.39	8.50	8.46		
		2.49	27.13	8.09	5.11		
		3.01	26.85	7.93	3.13		
SFD-3	8:45	0.48	27.17	8.55	9.59	0.46	2.36
		0.98	27.18	8.65	10.01		
		2.00	27.15	8.64	9.98		
		0.10	27.20	8.73	10.21		
		1.50	27.18	8.64	10.09		
SFD-3	16:00	0.09	28.91	9.02	12.00	0.46	2.36
		0.09	28.90	8.99	12.00		
		0.49	28.82	9.00	12.11		
		1.00	28.50	9.01	12.19		
		1.51	27.85	8.95	12.34		
		2.01	27.28	8.74	10.89		

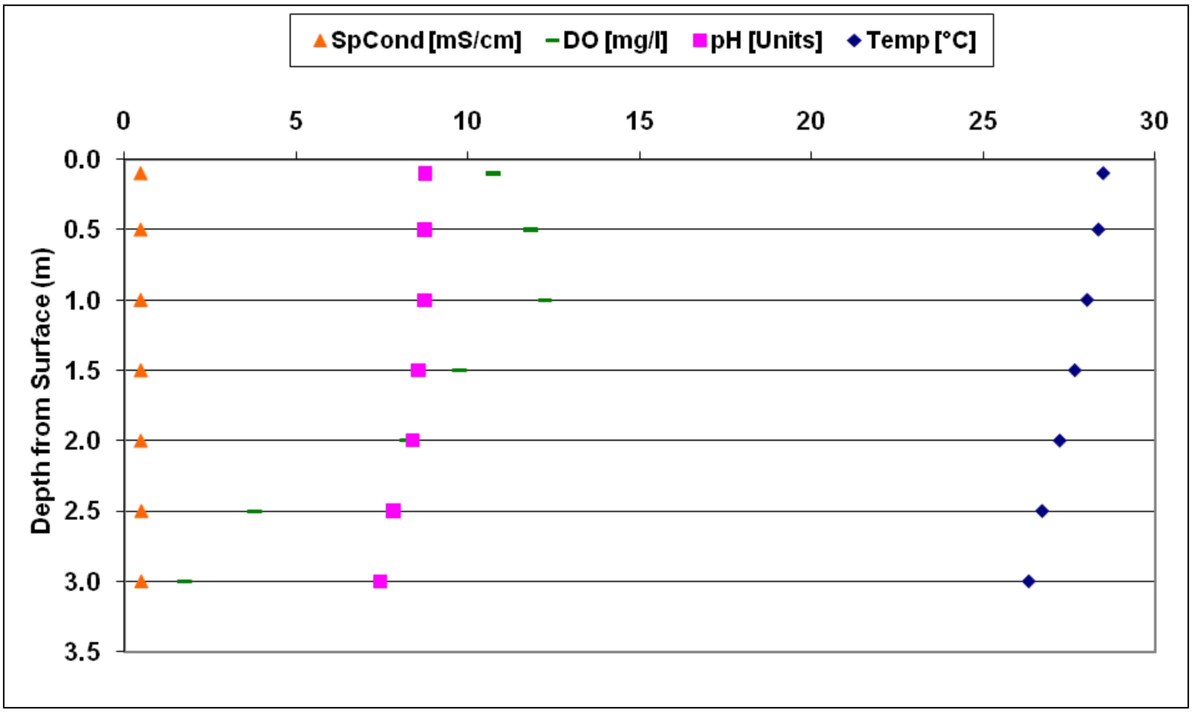


Figure G-49. Profile Data Collected at SFD-1 on August 3, 2009

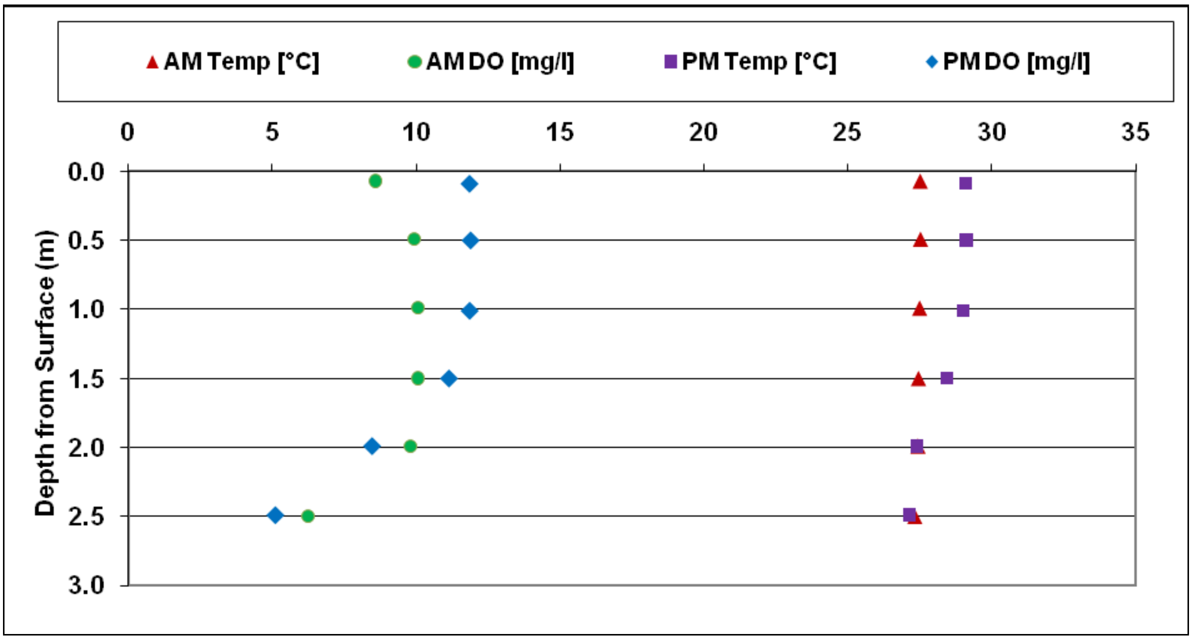


Figure G-50. Profile Data Collected at SFD-2 on August 3, 2009

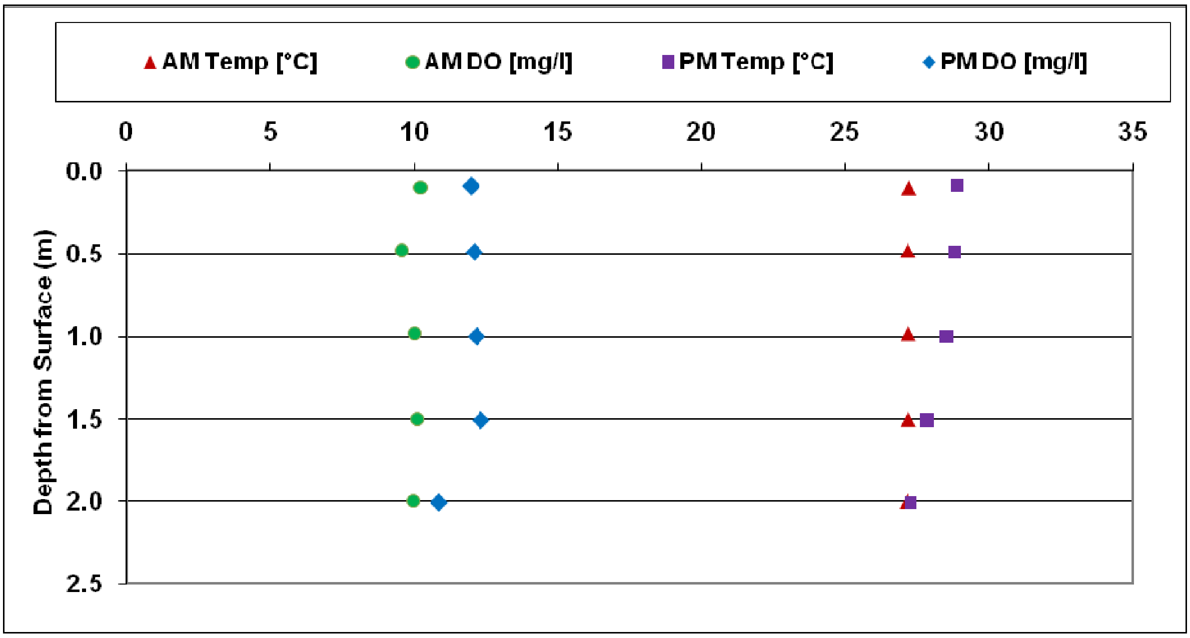


Figure G-51. Profile Data Collected at SFD-3 on August 3, 2009

The DO differences from morning and afternoon can further be analyzed by the DO saturation levels. The saturation at SFD-2 was highest in the afternoon, reaching a maximum of 157 percent at the surface. In the morning, the surface DO was at 110 percent and 129 percent between 0.5 and 2 meters of depth. The DO below 2 meters of depth was between 80 and 40 percent in the morning and afternoon. The DO saturation profile for SFD-2 is shown in Figure G-52.

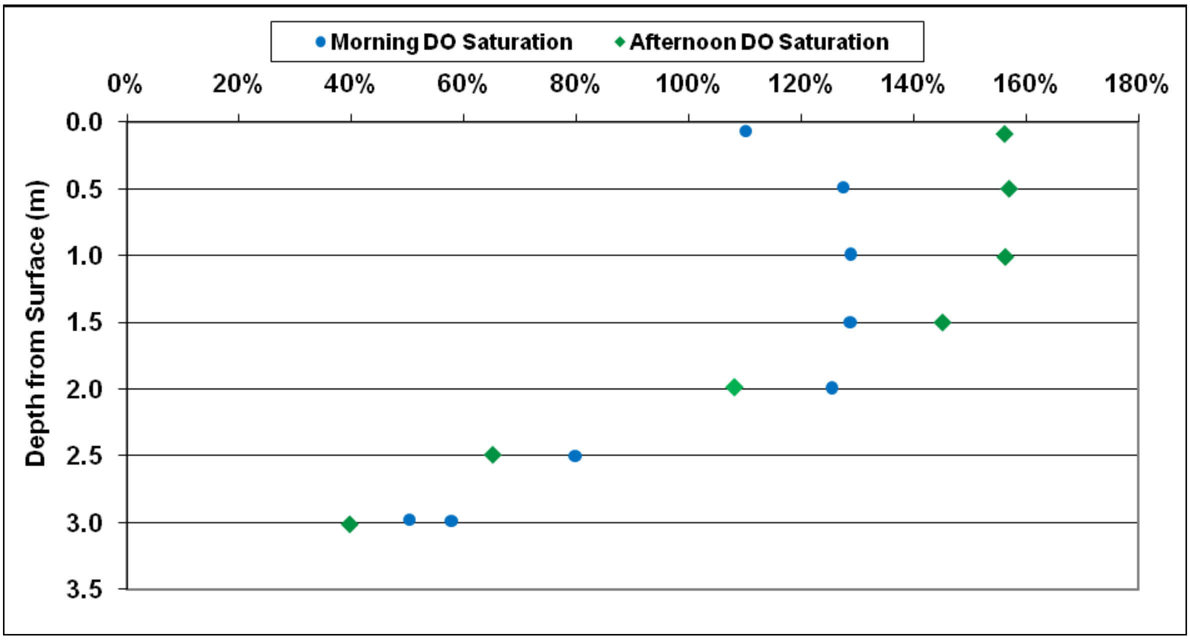


Figure G-52. DO Saturation from Profile Data Collected at SFD-2 on August 3, 2009

The DO saturation at SFD-3 had spatial and temporal patterns similar to those at SFD-2, shown in Figure G-53. The maximum DO was in the afternoon near the surface, 159 percent. The DO saturation in the morning had very little variation with depth and ranged from 122-130 percent. The DO saturation in the afternoon was around 159 percent between 1.5 meters of depth and the surface. The lowest saturation percent in the afternoon was 139, measured at 2.0 meters of depth.

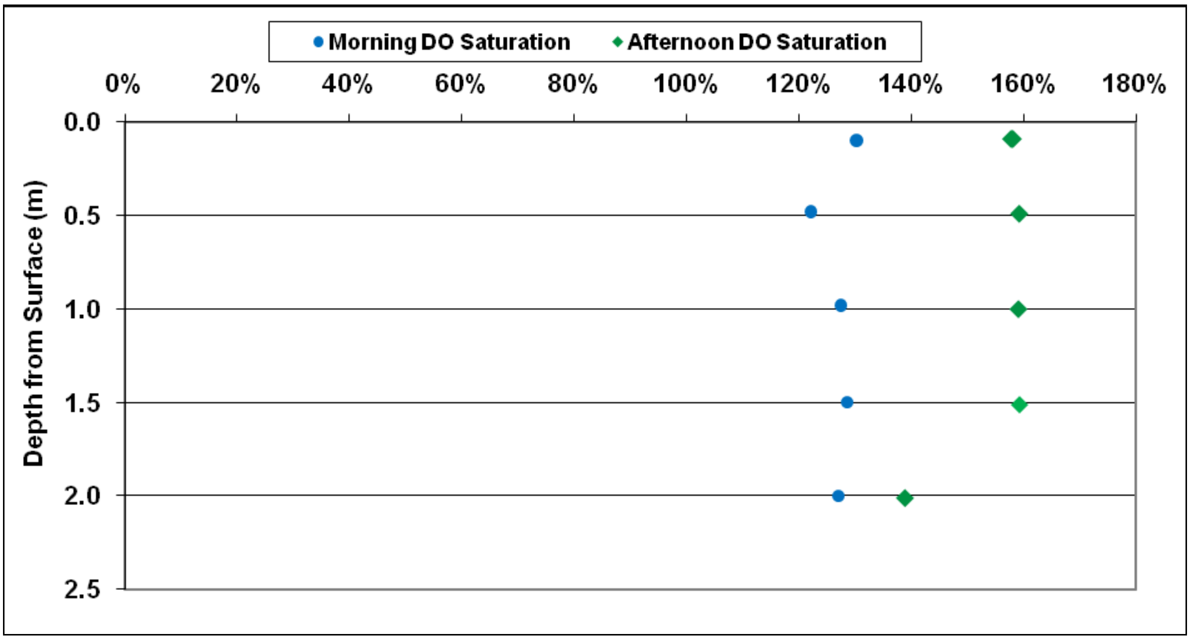


Figure G-53. DO Saturation from Profile Data Collected at SFD-3 on August 3, 2009

During the December 2009 monitoring event, profile measurements were collected at two in-lake stations. Additional single measurements were collected at one shoreline site. Table G-137 summarizes the field data collected during this event.

Table G-137. Field Data Collected At Santa Fe Dam Park Lake on December 14, 2009

Station	Depth (m)	pH	ORP (mV)	Temp (C)	DO (mg/L)	Cond (mS/cm2)
SFD-1	.5	8.63	105.6	12.75	10.2	0.537
	1	8.70	106.0	12.72	9.8	0.538
	1.5	8.73	105.4	12.65	9.2	0.541
	2	8.73	107.8	12.54	8.15	0.544
	2.5	8.63	108.3	12.41	6.1	0.550
SFD-3	.5	8.82	82.4	12.24	9.66	0.542
	1	8.89	85.0	12.22	9.55	0.542
	1.5	8.90	87.0	12.08	9.17	0.543
	2	8.90	87.4	12.06	8.98	0.543
	2.5	8.88	88.9	11.99	8.45	0.544
	2.8	8.87	88.9	12.00	8.07	0.546

Station	Depth (m)	pH	ORP (mV)	Temp (C)	DO (mg/L)	Cond (mS/cm2)
SFD-4	Surface	8.08	97.0	13.24	10.8	0.542
SFD-5	Surface	8.65	86.7	14.15	11.0	0.540

USEPA sampled Santa Fe Dam Park Lake again on August 12, 2010 (Table G-138). Secchi depth ranged from 0.61 m to 0.762 m. In-lake samples of TKN ranged from less than the detection limit of 0.47 mg-N/L to 0.594 mg-N/L. Ammonia samples at SFD-1 and SFD-3 were less than the detection limit of 0.03 mg-N/L, and nitrite samples were both detected at 0.035 mg-N/L. Nitrate concentrations were less than the detection limit (0.01 mg-N/L) at SFD-3 and 0.097 mg-N/L at SFD-1. Orthophosphate measurements at both sites were less than the detection limit of 0.0075 mg-P/L; total phosphorus concentrations ranged from 0.023 mg-P/L to 0.129 mg-P/L. Chlorophyll *a* concentrations ranged from 18.4 µg/L to 22.7 µg/L. Ammonia and nitrite concentrations in the groundwater were similar to those in the lake. TKN and nitrate in the groundwater sample were 1.11 mg-N/L and 1.62 mg-N/L, respectively. Orthophosphate concentration of the groundwater was 0.036 mg-P/L; total phosphorus was less than the detection limit of 0.0165 mg-P/L. Chlorophyll *a* concentration was less than the detection limit of 1.2 µg/L.

Table G-138. 2010 In-lake Water Column Measurements for Santa Fe Dam Park Lake

Date	Location	Time	TKN (mg/L)	NH_3-N (mg/L)	NO_2-N (mg/L)	NO_3-N (mg/L)	PO_4-P (mg/L)	Total P (mg/L)	Chlorophyll *a* (µg/L)	Secchi Depth (m)
8/12/2010	SFD-1	11:00	0.594	<0.03	0.035	0.097	<0.0075	0.129	22.7	0.762
8/12/2010	SFD-3	11:40	<0.47	<0.03	0.035	<0.01	<0.0075	0.0228	18.4	0.61
8/12/2010	SFD Well	12:40	1.11	<0.03	0.036	1.62	0.036	<0.0165	<1.2	NA

Supplemental water quality data for the August 2010 sampling event are shown in Table G-139.

Table G-139. 2010 Supplemental Water Quality Measurements for Santa Fe Dam Park Lake

Date	Location	Time	Chloride (mg/L)	Temperature (°C)	pH	Total Alkalinity (mg/L)	Total Hardness as $CaCO_3$ (mg/L)	TDS (mg/L)	TSS (mg/L)	TOC (mg/L)
8/12/2010	SFD-1	11:00	35.1	25.85	8.72	156	92.5	260	8.50	4.32
8/12/2010	SFD-3	11:40	36.4	25.93	8.73	150	92.9	222	10.8	4.11
8/12/2010	SFD Well	12:40	19.7	18.71	7.81	162	NA	228	<0.5	<2.0

During the August 2010 monitoring event, 24-hr temperature/pH/DO/conductivity probes were deployed at SFD-1 and SFD-3 (Figure G-54 and Figure G-55, respectively). The diurnal sampler placed at SFD-1 measured pH values ranging from 8.75 to 8.97 and DO concentrations ranging from 8.3 mg/L to 9.9 mg/L. At SFD-3, diurnal measurements of pH ranged from 8.82 to 8.97, and DO concentrations ranged from 8.9 mg/L to 11.3 mg/L.

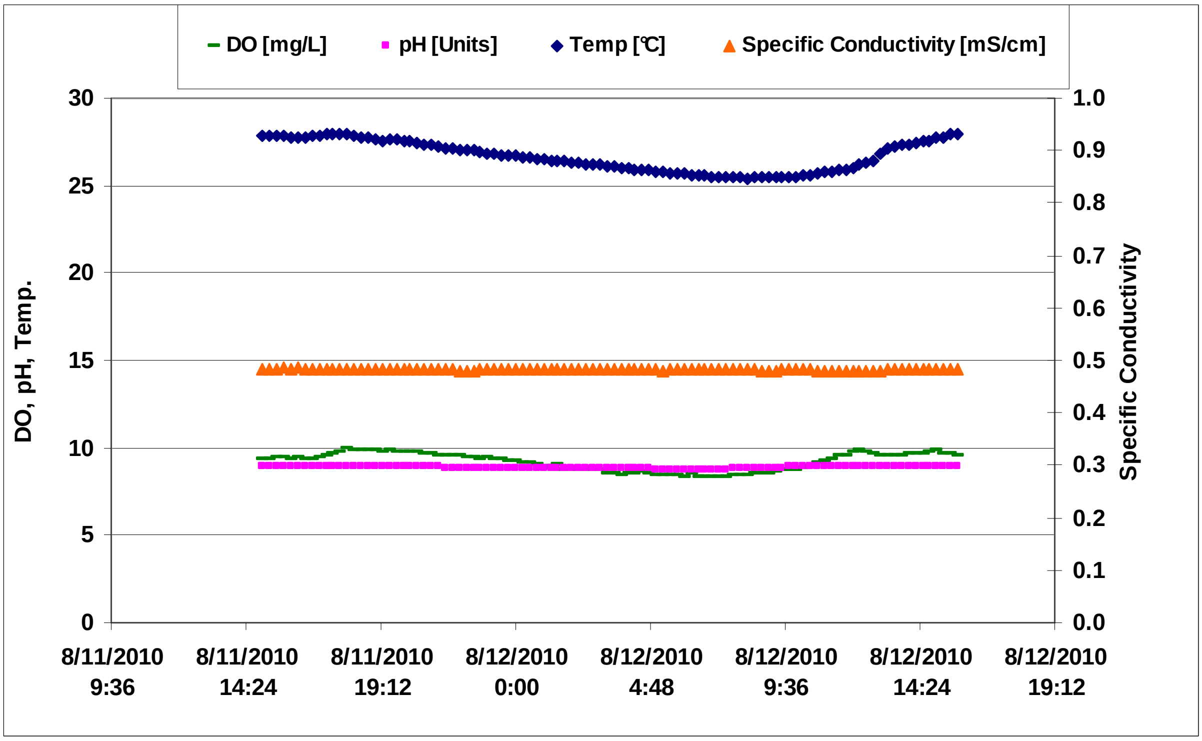


Figure G-54. Profile Data Collected at SFD-1 on August 12, 2010

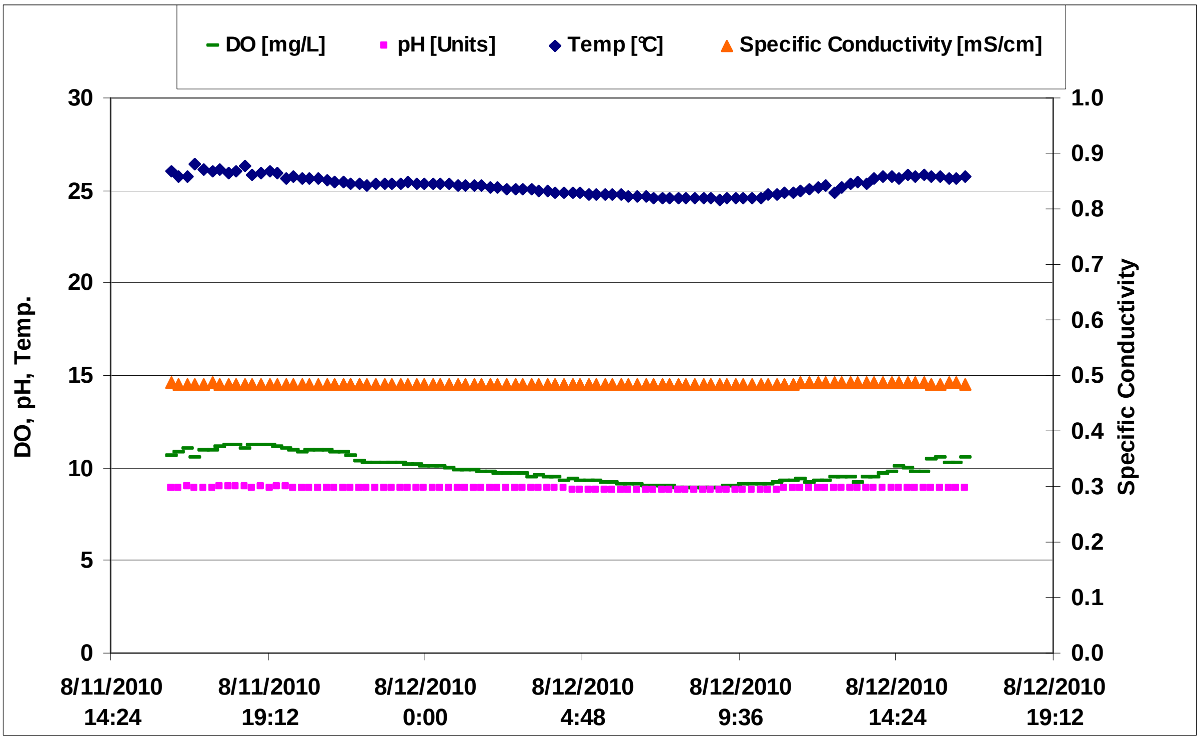


Figure G-55. Profile Data Collected at SFD-3 on August 12, 2010

Depth-profile data were also collected during this water sampling event. Table G-140 summarizes the depth-profile data collected at SFD-1 and SFD-3.

Table G-140. Profile Data Collected in Santa Fe Dam Park Lake (8/12/2010)

Site	Time	Depth (m)	Temp (C)	pH	DO (mg/L)	Specific Conductivity (mS/cm)	Orp (mV)
SFD-1	10:40	0.03	26.13	8.49	8.29	0.488	157
		0.53	25.85	8.72	8.62	0.487	158
		0.97	25.54	8.71	8.75	0.488	158
		1.45	25.41	8.69	8.66	0.488	158
		1.96	25.32	8.67	8.52	0.489	157
		2.54	24.3	8.56	8.29	0.488	157
SFD-2	11:25	0.06	26.07	8.73	8.33	0.488	145
		0.46	25.93	8.73	8.49	0.488	144
		1	24.85	8.75	8.93	0.486	144
		1.59	24.64	8.74	8.97	0.485	144
		1.97	24.52	8.73	8.87	0.485	143

Sediment samples were also collected during the August 2010 monitoring event. Table G-141 summarizes these data.

Table G-141. August 12, 2010 Sediment Monitoring Data for Santa Fe Dam Park Lake

Location	Time	TKN (mg/kg)	NH$_3$-N (mg/kg)	NO$_2$-N (mg/kg)	NO$_3$-N (mg/kg)	PO$_4$-P (mg/kg)	Total P (mg/kg)	Total Organic Carbon (% by wt.)	Acid Volatile Sulfides (mg/kg)	Percent Solids	Total Hardness (mg/kg)
SFD-1	11:00	903	7.21	1.79	1.90	0.621	739	2.89	1.08	25.0	48,600
SFD-1D	11:40	1,150	10.4	1.79	1.90	0.584	750	2.86	0.308	24.6	36,200
SFD-3	12:40	855	8.28	1.51	1.60	0.461	842	2.31	0.371	29.9	36,000

G.11.2 MONITORING RELATED TO METALS IMPAIRMENTS

In 1996 Santa Fe Dam Park Lake was deemed impaired by copper and lead. Monitoring data for cadmium, copper, lead, and zinc are presented in this section. Santa Fe Dam Park Lake is not listed for cadmium or zinc, but those data are presented here for completeness because other waterbodies in the region are affected by some of these contaminants.

Metals data collected at Santa Fe Dam Park Lake, as part of the 1992-1993 Urban Lakes Study (UC Riverside, 1994), are shown in Table G-142. The station was located in the southeast end of the lake near the spillway (pink triangle, Figure G-48) (UC Riverside, 1994). Sampling included dissolved copper and dissolved lead. Dissolved copper samples were collected throughout the water column at depths from the surface to 3.5 meters. The range of the 34 dissolved copper samples was between less than 10 µg/L and 56 µg/L. Similarly, dissolved lead samples were also collected throughout the water column, again at depths from the surface to 3.5 meters. The 34 samples collected ranged in concentration from less than 1 µg/L to 51 µg/L.

The Regional Board completed its Water Quality Assessment and Documentation Report for waterbodies in the Los Angeles Region in 1996 (LARWQCB, 1996). The summary table for Santa Fe Dam Park Lake states that copper and lead were not supporting the assessed uses: 37 measurements had a maximum lead concentration of 51 µg/L, a maximum copper concentration of 56 µg/L, and a maximum zinc concentration of 65 µg/L (raw data were not provided, but it is assumed that most of these samples are associated with the Urban Lake Study [UC Riverside, 1994]).

Unfortunately, metals levels were analyzed at relatively high detection limits compared to current detection limits; dissolved copper minimum detection 10 µg/L while dissolved lead was 1 µg/L. No hardness data were collected as part of the Urban Lakes Study, thus it cannot be compared to the hardness-based water quality objectives.

Table G-142. Santa Fe Dam Park Lake 1992/1993 Monitoring Data for Metals

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
8/10/1992	0	18	<1
	2	18	10
	3.5	13	3
8/10/1992	0	18	2
	2.5	19	2
8/10/1992	0	22	2
	2.5	21	2
9/10/1992	0	<10	2
	2	<10	<1
	3.5	<10	<1
10/13/1992	0	<10	15
	2	<10	4
	3.5	<10	<1
11/3/1992	0	27	3
	1.5	20	2
	2.5	56	2
1/14/1993	0	<10	1
	2	<10	<1

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
	3.5	<10	<1
2/3/1993	0	<10	<1
	2	<10	<1
	3	<10	<1
3/9/1993	0	<10	8
	2	<10	23
	3.5	<10	2
4/14/1993	0	<10	9
	1.5	<10	37
	2.5	<10	<1
5/25/1993	0	<10	18
	1.5	<10	36
	2.5	<10	12
6/21/1993	0	<10	<1
	1.5	<10	8
	2.5	<10	51

Table G-143 presents 32 additional water column metals samples that were collected by the USEPA, Regional Board, and/or the County of Los Angeles between March 2009 and August 2010. Samples were collected at locations SFD-1, SFD -2, SFD -3, SFD -4, and SFD-5. Sites were analyzed for dissolved cadmium, copper, lead, and zinc.

Detection limits were lower than the 1992-1993 study with a cadmium detection limit of 0.2 µg/L, dissolved copper detection limit of 0.4 µg/L, dissolved lead detection limit of 0.05µg/L, and dissolved zinc detection limit of 0.1 µg/L to 0.2 µg/L. All dissolved cadmium concentrations were < 0.2 µg/L; copper concentrations were between 0.6 µg/L and 2.76 µg/L; lead concentrations ranged from < 0.05 µg/L to 0.1 µg/L; and zinc concentrations were <0.1 µg/L to 2.9 µg/L. Metals toxicity is affected by hardness; therefore, each sample was also analyzed for hardness. The 2009-2010 sampling resulted in a hardness range of 86 mg/L to 133.2 mg/L. Since dissolved results pertain to the applicable standard and recent data more closely represents current conditions, data in Table G-143 were weighted more heavily in the assessment.

Table G-143. Water Column Metals Data for the 2008-2010 Santa Fe Dam Park Lake Sampling Events

Organi-zation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
RB	3/3/2009	SFD-2 / 3	103.95	<0.2	1.7	<0.1	0.1	average of stations 2 and 3
RB	3/3/2009	SFD-1	106.8	<0.2	1.8	<0.1	0.1	average of replicates
RB	3/3/2009	SFD-4	102.55	<0.2	1.5	<0.1	<2.4	average of duplicates
RB	3/3/2009	SFD-5	101.8	<0.2	1.9	<0.1	0.1	
RB/EPA	8/3/2009	SFD 1	131.3	<0.2	1.9	<0.1	1.9	
RB/EPA	8/3/2009	SFD 2 / 3	132.175	<0.2	1	0.1	0.1	average of replicates and then of sites 2 and 3
RB/EPA	8/3/2009	SFD 4	133.2	<0.2	1.1	0.1	1.1	
RB/EPA	8/3/2009	SFD 5	132.7	<0.2	1.8	0.1	2	
EPA/County	11/17/2009	SFD 4	89.9	<0.2	0.9	<0.1	1.1	
EPA/County	11/17/2009	SFD 5	92.5	<0.2	0.9	<0.1	1.4	
EPA/County	11/17/2009	SFD 3	91.6	<0.2	1	<0.1	1.5	averaged with dup & field filtered
EPA/County	11/17/2009	SFD 1	91.8	<0.2	0.8	<0.1	<0.1	average of replicates
County	12/8/2009	SFD 1	93.55	<0.2	1.4	<0.1	<0.1	average of replicates
County	12/8/2009	SFD 3	89.7	<0.2	1	<0.1	<0.1	average of replicates
County	12/8/2009	SFD 4	91.4	<0.2	0.6	<0.1	<0.1	
County	12/8/2009	SFD 5	87.8	<0.2	1.5	<0.1	0.7	
EPA	12/14/2009	SFD 1	89.35	<0.2	0.7	<0.1	<0.1	average of replicates
EPA	12/14/2009	SFD 3	88.3	<0.2	0.7	<0.1	<0.1	average of replicates
EPA	12/14/2009	SFD 4	90.2	<0.2	0.8	<0.1	<0.1	
EPA	12/14/2009	SFD 5	86	<0.2	0.7	<0.1	<0.1	
County	1/28/2010	SFD 1	101.4	<0.2	0.9	<0.1	<0.1	average of replicates & duplicate

Organi-zation	Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
County	1/28/2010	SFD 3	100.2	<0.2	0.9	<0.1	<0.1	
County	1/28/2010	SFD 4	100	<0.2	1	<0.1	<0.1	
County	1/28/2010	SFD 5	103.5	<0.2	0.9	<0.1	<0.1	
County	2/17/2010	SFD 1	109.1	<0.2	1.1	0.065	0.65	average of duplicate
County	2/17/2010	SFD 3	110.5	<0.2	1.1	0.07	2.7	
County	2/17/2010	SFD 4	113.1	<0.2	1.15	0.06	1.95	average of replicates
County	2/17/2010	SFD 5	112	<0.2	1.2	0.06	2.9	
EPA	8/12/2010	SFD 1	92.5	<0.2	1.03	<0.05	<0.1	
EPA	8/12/2010	SFD 3	92.9	<0.2	2.76	<0.05	<0.1	
EPA	8/12/2010	SFD 4	NA	<0.2	0.879	<0.05	2.06	
EPA	8/12/2010	SFD 5	NA	<0.2	1.05	<0.05	<0.1	

RB = Regional Board

EPA = USEPA

County = County of Los Angeles

USEPA also collected two sediment samples during the month of August 2010 to further evaluate lake conditions. Table G-144 summarizes the copper and lead concentrations measured in these samples. There were zero sediment lead exceedances of the 128 ppm freshwater (Probable Effect Concentrations) sediment target and zero sediment copper exceedances of the 149 ppm freshwater (Probable Effect Concentrations) sediment target.

Table G-144. Sediment Metals Data for the August 2010 Santa Fe Dam Park Lake Sampling Event

Organization	Date	Station ID	Copper (mg/kg)	Lead (mg/kg)	Notes
EPA	8/12/2010	SFD 1	14.7	1.76	Average of duplicates
EPA	8/12/2010	SFD 3	5.92	1.49	

(This page left intentionally blank.)

G.12 Monitoring Data for Lake Sherwood

Fish tissue monitoring data relevant to the impairments of Lake Sherwood are available from 1991 top 2007, while water and sediment quality data are available for 2009. Figure G-56 shows the historical and recent monitoring locations for Lake Sherwood.

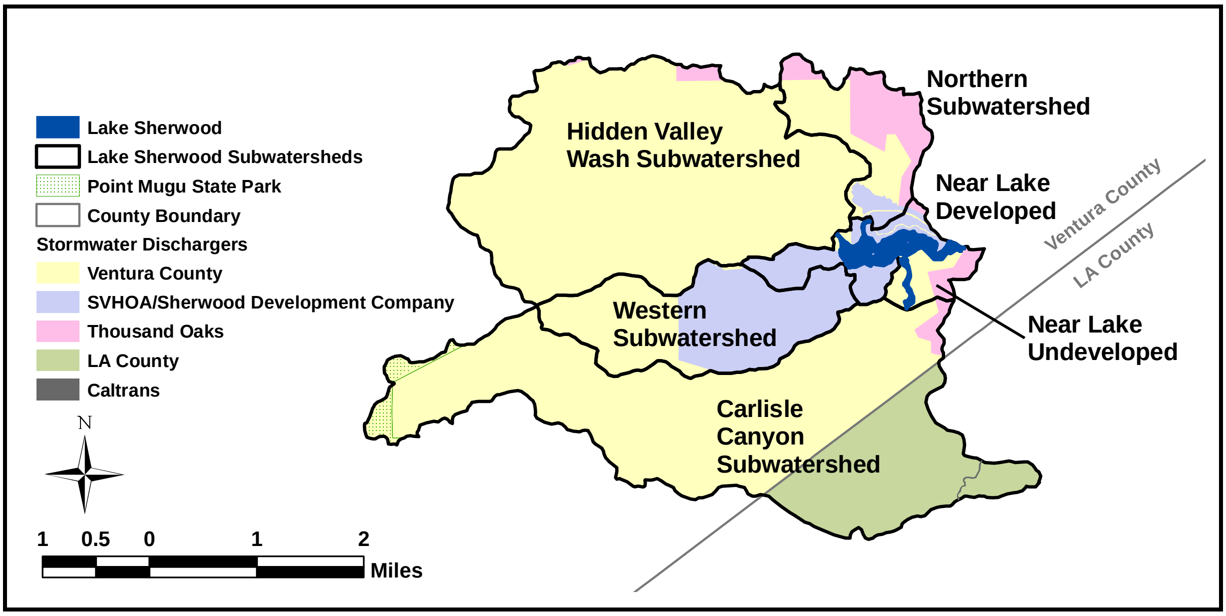


Figure G-56. Lake Sherwood Monitoring Sites

G.12.1 MONITORING RELATED TO MERCURY IMPAIRMENT

Mercury data have been collected in the Lake Sherwood watershed since 1991. Fish tissue concentrations were measured three times under the Toxic Substances Monitoring Program (TSMP) from 1991 to 1997 and by the Regional Board in 2007 (Davis et al., 2008). USEPA and the Regional Board also sampled in-lake and tributary water column and sediment mercury concentrations during two events in 2009. Figure G-56 shows the locations of the water quality monitoring stations.

G.12.1.1 In-Lake Water Quality Monitoring

G.12.1.1.1 Water Column Measurements

USEPA and the Regional Board sampled one station in Lake Sherwood for total and methylmercury in February and July 2009. During the February event, the total depth at this location was 5 meters; samples were collected from 3 meters below the surface. A representative of the Lake Sherwood home owner's association (HOA) provided a boat and accompanied the sampling team. The HOA representative would not allow the sampling team to anchor the boat during sampling, so the engine was left running. The in-lake February sample may therefore be contaminated from the exhaust of the outboard motor. During the July event, samples were collected from a depth of 1 m, and the total depth at this site was 7.8 m. The boat was anchored during this event with the engine turned off.

Table G-145 compares the February and July 2009 water column concentrations observed in Lake Sherwood. In February, the total mercury concentration was 3.32 ng/L, and the methylmercury concentration was 0.189 ng/L or 5.7 percent. In July, the total mercury concentration was 0.75 ng/L and the methylmercury concentration was 0.329 ng/L. The percent of mercury in the methyl form in July was 44 percent. Total mercury was analyzed with EPA Method 1631 with a detection limit of 0.15 ng/L. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.020 ng/L.

Supplemental water quality data are included in Table G-146.

Table G-145. In-lake Water Column Measurements for Lake Sherwood

Location	Date	Time	MeHg (ng/L)	Total Hg (ng/L)	TSS (mg/L)
SL-In-lake	2/25/2009	10:00	0.189	3.32	7.1
SL-In-lake	7/13/2009	9:00	0.329	0.75	5.3

Table G-146. Supplemental Water Quality Monitoring for In-lake Samples in Lake Sherwood

Location	Date	Time	Chloride (mg/L)	Sulfate (mg/L)	Total Alkalinity (mg/L)	Total Dissolved Solids (mg/L)	Total Organic Carbon (mg/L)
SL-In-lake	2/25/2009	10:00	73.73	180.03	202	664	6.0
SL-In-lake	7/13/2009	9:00	73.23	200.06	240	752	6.75

Profile data were collected at station SL-In-lake on February 25, 2009 (Figure G-57). Specific conductivity is constant with depth. DO decreases from over 9 mg/L at the surface to 0 mg/L at a depth of 3 meters. pH ranges from 8.0 to 8.6, and temperature ranges from 11.1 °C to 13.0 °C. **Note that field operators found DO readings suspicious and have since sent meter off for repair (Greg Nagle, USEPA Region 9, personal communication, 5/22/09).**

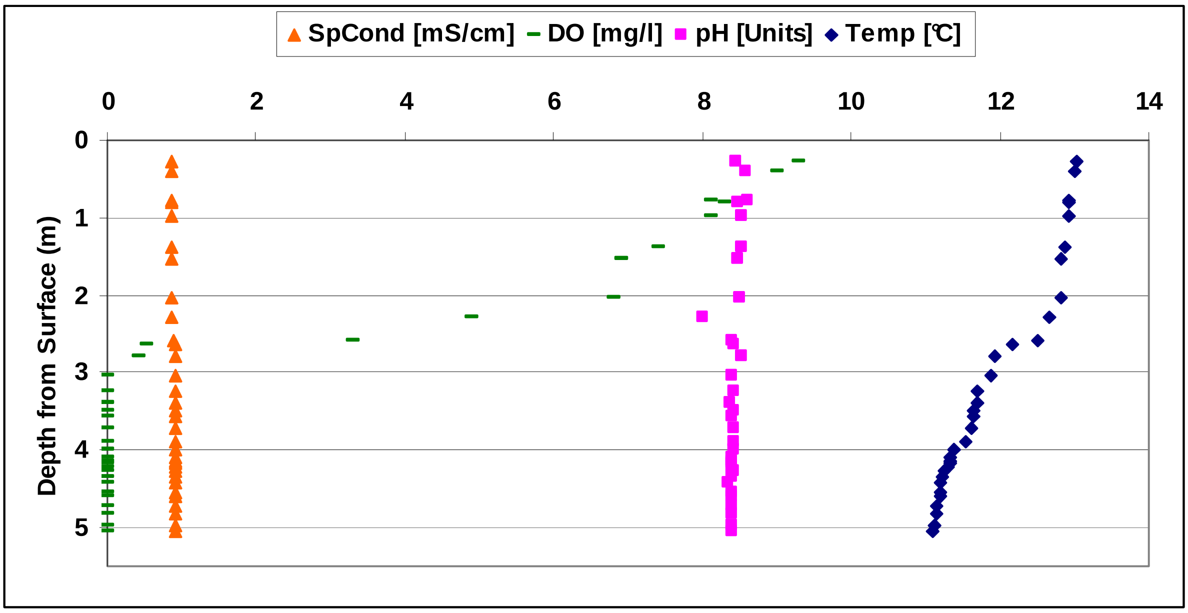


Figure G-57. Profile Data Collected at SL-In-lake on February 25, 2009

Profile data were also collected at station SL-In-lake on July 13, 2009 (Figure G-58). Specific conductivity remained constant with depth. DO decreases from over 10 mg/L at the surface to almost 0 mg/L at a depth of 8 meters. The DO meter was repaired for these readings. pH ranges from 7.5 to 8.8, and temperature ranges from 21.1 ºC to 25.9 ºC

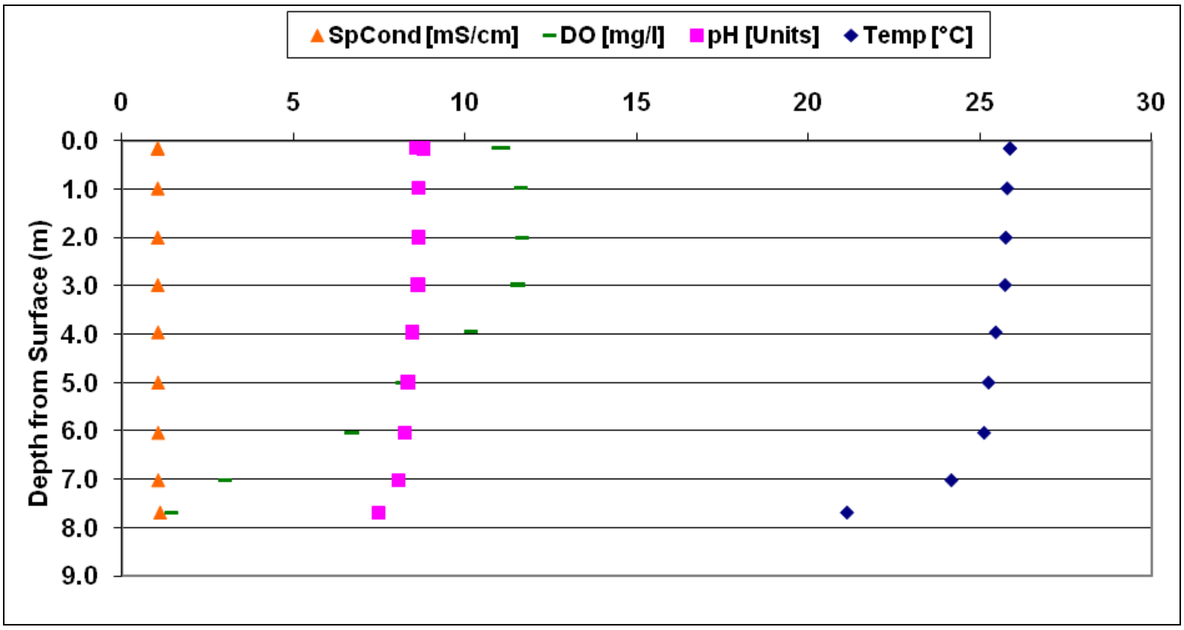


Figure G-58. Profile Data Collected at SL-In-lake on July 13, 2009

G.12.1.1.1 Sediment Samples

USEPA and the Regional Board collected sediment samples from Lake Sherwood to measure total and methylmercury concentrations in sediment. In February, total mercury was analyzed with EPA Method 1631 with a detection limit of 4.96 µg/kg. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.022 µg/kg. The concentrations of total and methylmercury were 470 µg/kg and 0.685 µg/kg, respectively. In July, total mercury was analyzed with EPA Method 1631 with a detection limit of 15.9 µg/kg. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.025 µg/kg. The concentrations of total and methylmercury were 388 µg/kg and 0.599 µg/kg, respectively.

In-lake sediment mercury concentrations for Lake Sherwood are presented in Table G-147. Supplemental data are presented in Table G-148. Concentrations are reported on a dry weight basis.

Table G-147. In-lake Sediment Concentrations for Lake Sherwood

Location	Date	Time	MeHg (µg/kg)	Total Hg (µg/kg)	TSS (%)
SL-In-lake	2/25/2009	10:00	0.685	470	36.70
SL-In-lake	7/13/2009	9:00	0.599	388	33.96

Table G-148. Supplemental Sediment Data for In-lake Samples in Lake Sherwood

Location	Date	Time	Sulfate (mg/kg)	Total Organic Carbon (percent of dry weight)
SL-In-lake	2/25/2009	10:00	481.93	3.38
SL-In-lake	7/13/2009	9:00	218.99	5.15

G.12.1.2 Fish Tissue Sampling

Mercury concentrations in the fish tissue of largemouth bass have been measured in Lake Sherwood since 1991. The TSMP sampled individual fish three times. The SWAMP sampled individual fish during the summer of 2007 and April 2010. The Sherwood Valley HOA sampled five individual fish in 2007 as well (Weston Solutions, 2007); length data were not retained during analysis. Fillet and liver tissue were analyzed. Table G-149 presents the fish tissue mercury concentrations on a wet weight basis; liver concentrations are not included. Concentrations range from 0.214 ppm to 1.6 ppm. The applicable fish tissue guideline for mercury measured as a wet weight concentration is 0.22 ppm.

Table G-149. Fish Tissue Mercury Concentrations Measured in Lake Sherwood Large Mouth Bass

Program	Date	Fish Length (mm)	Total Mercury Concentration (ppm wet weight)
TSMP	4/22/1991	356	0.700
TSMP	4/21/1992	286	1.600
TSMP	7/17/1997	349	0.214
SWAMP	Summer 2007	205	0.219
SWAMP	Summer 2007	242	0.239
SWAMP	Summer 2007	261	0.325
SWAMP	Summer 2007	284	0.236
SWAMP	Summer 2007	305	0.362
SWAMP	Summer 2007	321	0.322
SWAMP	Summer 2007	365	0.802
SWAMP	Summer 2007	345	0.751
SWAMP	Summer 2007	353	0.601
SWAMP	Summer 2007	318	0.444
SWAMP	Summer 2007	328	0.464
SWAMP	Summer 2007	349	0.504
SWAMP	Summer 2007	339	0.607
SWAMP	Summer 2007	386	0.552
SWAMP	Summer 2007	418	0.802

Program	Date	Fish Length (mm)	Total Mercury Concentration (ppm wet weight)
SWAMP	Summer 2007	452	0.665
Sherwood HOA	Summer 2007	Length data not available	0.465
Sherwood HOA	Summer 2007	Length data not available	0.670
Sherwood HOA	Summer 2007	Length data not available	0.319
Sherwood HOA	Summer 2007	Length data not available	0.284
Sherwood HOA	Summer 2007	Length data not available	0.409
SWAMP	4/19/2010	417	1.02
SWAMP	4/19/2010	385	0.664
SWAMP	4/19/2010	374	0.824
SWAMP	4/19/2010	368	0.994
SWAMP	4/19/2010	357	1.09

Piscivorous fish tend to have increased mercury tissue concentrations with age. Figure G-59 shows the mercury concentrations in largemouth bass plotted against length, which is an approximate surrogate for age. As expected, fish tissue mercury concentrations increase with length. All fish specimens with a mean or individual length greater than 205 mm exceed the fish tissue target of 0.22 mg/kg, with the exception of one sample that had a concentration of 0.214 ppm and a length of 349 mm. Of the samples with corresponding length data, 22 exceeded the fish tissue target.

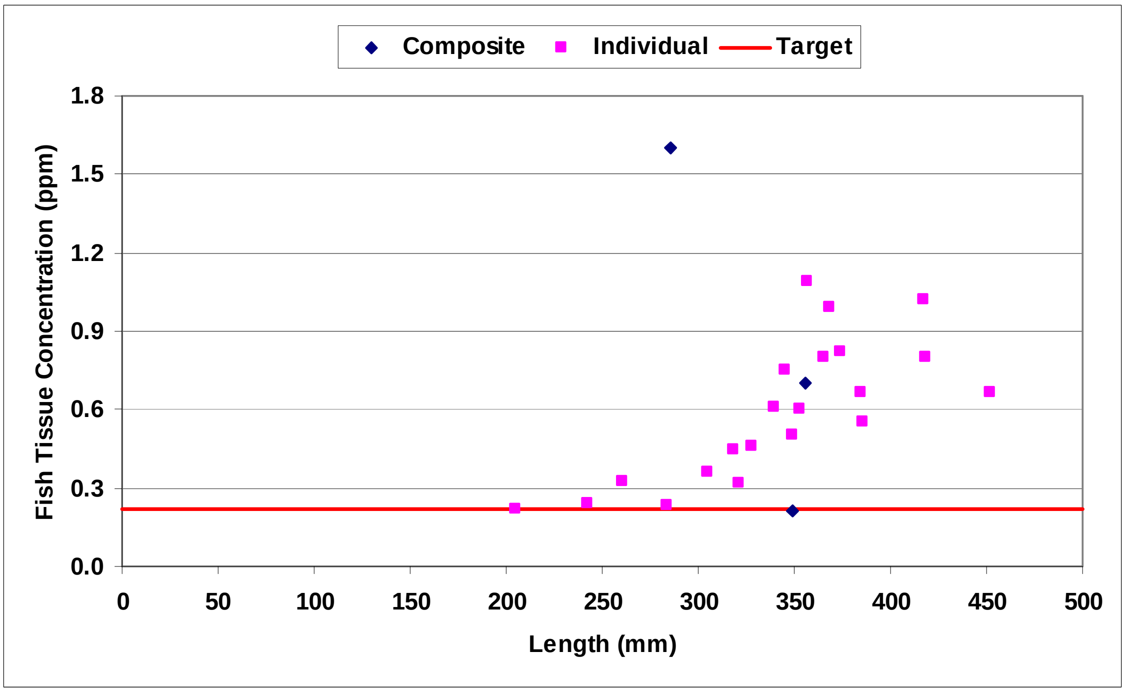


Figure G-59. Mercury Concentrations in Largemouth Bass in Lake Sherwood

SWAMP also collects data on mercury concentration in redear sunfish. Table G-150 provides composite results for redear sunfish collected in April 2010.

Table G-150. Composite Fish Tissue Mercury Concentrations Measured in Lake Sherwood Redear Sunfish

Program	Date	Average Fish Length (mm)	Number of Fish per Composite	Total Mercury Concentration (ppm wet weight)
SWAMP	4/19/2010	289	5	0.140
SWAMP	4/19/2010	291	5	0.185
SWAMP	4/19/2010	291	5	0.169

G.12.1.3 Tributary/Inflow Monitoring

G.12.1.1.3 Water Column Measurements

In February 2009, USEPA and the Regional Board sampled water column total and methylmercury concentrations from two tributaries and one storm drain. However, the temperature requirements for the methylmercury sample collected at the storm drain (SL-8) were not met, so the measured methylmercury concentration may be compromised. Total mercury was analyzed with EPA Method 1631 with a detection limit of 0.15 ng/L. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.020 ng/L. The two tributary samples (SL-3 and SL-6) had total mercury concentrations ranging from 2.96 ng/L to 6.00 ng/L. The storm drain (SL-8) had a higher total mercury concentration of 23.9 ng/L. Methylmercury concentrations in the tributary samples ranged from 0.157 ng/L to 0.216 ng/L. Methylmercury in the storm drain sample was an order of magnitude lower, but this sample was compromised and may not be accurate.

Inflow water column measurements were collected again in the summer of 2009. The tributary at SL-6 was not flowing, so a sample was not collected during the July event. The storm drain at SL-8 had methyl and total mercury concentrations of 0.096 ng/L and 54.0 ng/L, respectively. The creek flowing through the golf course community was sampled at SL-3; a duplicate sample was analyzed for total mercury. The forebay at the outlet of Hidden Valley Wash (SL-7) had methyl and total mercury concentrations of 3.41 ng/L and 11.3 ng/L, respectively. Total mercury was analyzed with EPA Method 1631 with detection limits ranging from 0.15 ng/L to 0.73 ng/L. Methylmercury was analyzed with EPA Method 1630 with a detection limit of 0.020 ng/L.

Table G-151 presents the results of the water column mercury and TSS concentrations measured in the tributaries and storm drains to Lake Sherwood. The tributary flowing through the mountainous subwatershed that discharges to the south side of the lake had the lowest concentrations of methyl and total mercury during the winter sampling event; this tributary was not flowing during the summer event. The highest concentrations of total mercury were observed in storm drain SL-8. Methylmercury concentrations were highest in the forebay at the outlet of Hidden Valley Wash. This site (SL-7) was identified as a potential methylation hot spot based on sediment samples collected in February 2009 (see discussion in Section G.12.1.1.3. Table G-152 presents the supplemental water quality data.

Table G-151. Tributary/Inflow Water Column Measurements for Lake Sherwood

Location	Date	Time	MeHg (ng/L)	Total Hg (ng/L)	TSS (mg/L)
SL-3	2/25/2009	13:00	0.157	6.00	1.0
SL-6		11:00	0.216	2.96	2.9
SL-8		11:45	0.025[1]	23.9	2.2
SL-8	7/13/2009	10:00	0.096	54.0	5.1
SL-3		8:55	0.536	4.58	2.1
SL-3D		8:55	NA	4.63	NA
SL-7		10:15	3.41	11.3	20.3

[1] Temperature requirements for methylmercury analysis not met.

Table G-152. Supplemental Water Quality Monitoring for Inflow Samples for Lake Sherwood

Location	Date	Time	Chloride (mg/L)	Sulfate (mg/L)	Total Alkalinity (mg/L)	Total Dissolved Solids (mg/L)	Total Organic Carbon (mg/L)
SL-3	2/25/2009	13:00	134.61	384.7	262	1,094	4.7
SL-6		11:00	55.68	146.65	206	578	4.1
SL-8		11:45	180.97	488.44	238	1,310	5.6
SL-8	7/13/2009	10:00	156.33	271.61	248	1036	5.5
SL-3		8:55	190.1	573.81	346	1628	5.2
SL-7		10:15	76.46	181.58	192	714	7.7

G.12.1.1.3 Sediment Samples

Sediment samples were collected from three tributaries (SL-3, SL-6, and SL-7) and one storm drain (SL-5) during the February 2009 monitoring event. Samples SL-3 and SL-6 represented flowing water through developed and undeveloped areas, respectively. Total and methylmercury sediment concentrations at these two sites were much lower than site SL-7, which was intended to represent the Hidden Valley Wash tributary. This tributary appears to be piped under Janss Road prior to discharging to Lake Sherwood. The outlet of the pipe is beneath the surface of a stagnant backwater area adjacent to the Lake. The sediment mercury concentration of this sample may be more reflective of a wetland area than the sediment being delivered from the upland areas draining to Hidden Valley Wash. This is particularly true of the methylmercury sediment concentration which is an order of magnitude greater than those measured in the other inputs or Lake Sherwood itself. Though the methyl and total mercury concentrations at site SL-7 may not be accurate for estimating loading from Hidden Valley Wash, they do identify a potential location of high rates of methylation that may be increasing the bioavailability of mercury to the aquatic life in Lake Sherwood. Typical hotspots for methylation include wetlands, where sediments alternate between wet and dry conditions. Based on two reconnaissance events conducted for Lake Sherwood, this backwater area undergoes both dry (January 2009) and wet/stagnant (February 2009) periods.

In July 2009, sediment samples were collected from four locations. Duplicate total mercury samples were collected at SL-3. The lowest total mercury concentrations (approximately 60 µg/kg) were observed at

SL-PR (upstream of SL-7 on Hidden Valley Wash at Potrero Road) and SL-5. The highest total mercury concentrations were measured at SL-3 and SL-7. Methylmercury concentrations ranged from 0.397 µg/kg to 0.657 µg/kg at SL-3, SL-5, and SL-7 with the highest concentration (0.696) measured at SL-8. Concentrations were much lower at SL-PR and were equivalent to the detection limit for that sample.

Sediment mercury concentrations collected from the inputs and adjacent area of Lake Sherwood are presented in Table G-153. Concentrations are reported on a dry weight basis. Table G-154 presents the supplemental sediment quality data.

Table G-153. Inflow Sediment Concentrations for Lake Sherwood

Location	Date	Time	MeHg (µg/kg)	Total Hg (µg/kg)	TSS (%)
SL-3	2/25/2009	13:00	0.269	92.7	77.70
SL-6		11:00	0.136	129	75.90
SL-5		13:15	0.145	51.0	82.62
SL-7[1]		08:30	2.53	243	74.25
SL-3	7/13/2009	8:55	0.397	392	30.30
SL-3D		8:55	NA	265	34.45
SL-5		9:45	0.657	62.9	96.80
SL-7		10:15	0.453	275	73.18
SL-PR		10:50	0.009	60.3	98.82
SL-8		10:00	0.696	63.3	74.52

[1] This sample is likely not representative of the sediment methylmercury concentrations delivered from Hidden Valley Wash.

Table G-154. Supplemental Sediment Data for Inflow Samples to Lake Sherwood

Location	Date	Time	Sulfate (mg/kg	Total Organic Carbon (percent of dry weight)
SL-3	2/25/2009	13:00	157.54	0.24
SL-6		11:00	125.09	0.58
SL-5		13:15	92.76	1.44
SL-7		08:30	108.98	1.67
SL-3	7/13/2009	8:55	1,106.74	10.15
SL-5		9:45	903.53	3.93
SL-7		10:15	93.23	0.68
SL-PR		10:50	9.3	1.64
SL-8		10:00	41.69	2.35

G.13 Monitoring Data for Westlake

Monitoring data relevant to the impairments of Westlake Lake are available for 1992, 1993, 2009, and 2010. Figure G-60 shows the historical and recent monitoring locations for Westlake Lake.

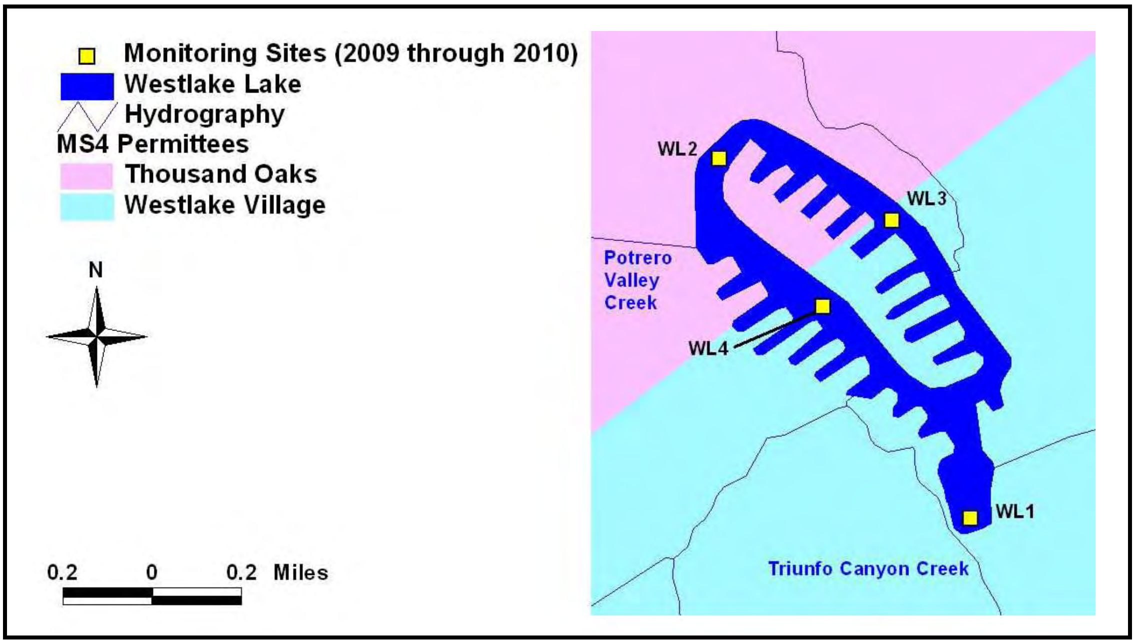


Figure G-60. Westlake Lake Monitoring Sites

G.13.1 MONITORING RELATED TO METALS IMPAIRMENT

In 1996 Westlake Lake was impaired by lead. Monitoring data for cadmium, copper, lead, and zinc are presented in this section. Westlake Lake is not listed for cadmium, copper, or zinc, but those data are presented here for completeness because other waterbodies in the region are affected by some of these contaminants.

Metals data collected at Westlake Lake, as part of the 1992-1993 Urban Lakes Study (UC Riverside, 1994), are presented in Table G-155. Samples were collected near the outlet of the lake (WL1) and included dissolved copper and dissolved lead. Dissolved copper samples were collected throughout the water column at depths from the surface to six meters. The range of the 52 dissolved copper samples was between less than 10 µg/L and 56 µg/L. Similarly, dissolved lead samples were also collected throughout the water column, again at depths from the surface to six meters. The 52 samples collected ranged in concentration from less than 1 µg/L to 91 µg/L.

The Regional Board completed its Water Quality Assessment and Documentation Report for waterbodies in the Los Angeles Region in 1996 (LARWQCB, 1996). The summary table for Westlake Lake states that copper and lead were not supporting the assessed uses (copper has since been delisted): 52 measurements had a maximum lead concentration of 91 µg/L, a maximum copper concentration of 56 µg/L, and a maximum zinc concentration of 12 µg/L (raw data were not provided, but it is assumed that most of these samples are associated with the Urban Lake Study [UC Riverside, 1994]).

Unfortunately, metals levels were analyzed at relatively high detection limits compared to current detection limits; dissolved copper minimum detection 10 µg/L while dissolved lead was 1 µg/L. No hardness data were collected as part of the Urban Lakes Study, thus it cannot be compared to the hardness-based water quality objectives.

Table G-155. Westlake Lake 1992/1993 Monitoring Data for Metals

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
8/3/1992	0	21	<1
	2	21	<1
	4	19	1
	6	13	4
8/3/1992	0	25	2
	2	21	<1
8/3/1992	0	42	<1
	2.5	28	2
8/18/1992	0	47	<1
	2.5	47	<1
	4	36	<1
	6	21	<1
9/23/1992	0	55	22
	1.5	33	6
	4	25	4
	6	21	2
10/14/1992	0	48	<1
	2	46	<1
	4	46	<1
	6	44	<1
11/10/1992	0	24	1
	2	34	1
	4	37	1
	6	54	2
12/14/1992	0	31	11
	1.5	29	5
	3	42	6
	6	56	13
1/20/1993	0	<10	<1

Date	Depth (m)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)
	2	<10	<1
	4	<10	<1
	6	<10	<1
2/24/1993	0	<10	<1
	2	<10	<1
	4	12	<1
	6	<10	<1
3/10/1993	0	<10	<1
	2	<10	<1
	4	<10	<1
	6	<10	<1
4/19/1993	0	26	1
	2.5	30	33
	3.5	27	8
	4.5	18	6
5/19/1993	0	30	91
	2.5	31	27
	4.5	26	9
	6.5	19	17
6/28/1993	0	36	19
	2	33	2
	4	29	<1
	6	29	<1

Table G-156 presents 24 additional metals samples that were collected by USEPA and the Regional Board between March 2009 and October 2010. Samples were collected at locations WL-1, WL-2, WL-3, and WL-4. Sites were analyzed for dissolved cadmium, copper, lead, and zinc.

Detection limits were lower than the 1992-1993 study with a cadmium detection limit of 0.2 µg/L, dissolved copper detection limit of 0.4 µg/L, dissolved lead detection limit of 0.05 µg/L, and dissolved zinc detection limit of 0.2 µg/L. All dissolved cadmium concentrations were less than 0.4 µg/L; copper concentrations ranged from 2.5 µg/L to 8.9 µg/L; lead concentrations were between <0.05 µg/L and 0.065 µg/L; and zinc concentrations ranged from <0.1 µg/L to 5.45 µg/L. Metals toxicity is affected by hardness; therefore, each sample was also analyzed for hardness. The 2009-2010 sampling resulted in a hardness range of 231 mg/L to 477 mg/L. Since dissolved results pertain to the applicable standard and recent data more closely represents current conditions, data in Table G-156 were weighted more heavily in the assessment.

Table G-156. Metals Data for the 2009-2010 Westlake Lake Sampling Events

Date	Station ID	Hardness (mg/L)	Dissolved Cadmium (µg/L)	Dissolved Copper (µg/L)	Dissolved Lead (µg/L)	Dissolved Zinc (µg/L)	Notes
3/26/2009	WL 1	348.58	<0.2	5.78	<0.05	1.18	average of duplicates and replicates
3/26/2009	WL 2	353.60	<0.2	5.50	<0.05	1.40	
3/26/2009	WL 3	343.80	<0.2	6.00	<0.05	2.20	
3/26/2009	WL 4	347.70	<0.2	5.60	<0.05	0.80	
7/17/2009	WL 1	469.77	<0.2	6.87	0.05	0.57	average of duplicates and replicates
7/17/2009	WL 2	477.00	<0.2	8.90	<0.05	<0.10	
7/17/2009	WL 3	466.00	<0.2	7.30	<0.05	0.40	
7/17/2009	WL 4	469.25	<0.2	8.15	<0.05	<0.10	average of replicates
12/17/2009	WL 1	382.6	<0.2	4.9	0.055	5.2	average of replicates
12/17/2009	WL 2	382.9	<0.2	4.4	0.065	5.45	average of duplicates
12/17/2009	WL 3	351	<0.2	4.8	0.05	4.7	
12/17/2009	WL 4	388.2	<0.2	4.1	0.05	1.3	
1/26/2010	WL 1	246.1	<0.2	3.4	<0.05	1.55	average of replicates
1/26/2010	WL 2	243.3	<0.2	2.5	<0.05	4.05	average of duplicates
1/26/2010	WL 3	231.5	<0.2	3.25	<0.05	1.6	
1/26/2010	WL 4	256.3	<0.2	2.8	<0.05	0.6	
8/13/2010	WL 1	333	0.409	4.46	<0.05	2.61	
8/13/2010	WL 2	334	<0.2	4.10	<0.05	<0.1	
8/13/2010	WL 2D	334	0.407	4.10	<0.05	<0.1	
8/13/2010	WL 3	332	ND	4.08	<0.05	0.748	
8/13/2010	WL 4	331	ND	4.14	<0.05	<0.1	
10/1/2010	WL 1	337	<0.2	5.96	<0.05	<0.1	
10/1/2010	WL 2	335	<0.2	5.95	<0.05	<0.1	
10/1/2010	WL 3	328	<0.2	4.99	<0.05	<0.1	
10/1/2010	WL 4	335	<0.2	6.34	<0.05	<0.1	

Note: all sampling performed by the Regional Board and/or USEPA.

USEPA also collected two sediment samples during August 2010 to further evaluate lake conditions. Table G-157 summarizes the lead concentrations measured in these samples. There were zero sediment lead exceedances of the 128 ppm freshwater (Probable Effect Concentrations) sediment target.

Table G-157. Sediment Metals Data for August 2010 West Lake Sampling Event

Organization	Date	Station ID	Lead (mg/kg)	Notes
EPA	08/13/2010	WL1	31.1	
EPA	08/13/2010	WL2	83.1	Average of duplicates

(This page left intentionally blank.)

G.14 References

Davis, J.A., A.R. Melwani, S.N. Bezalel, G. Ichikawa, A. Bonnema, C. Lamerdin, W.A. Heim, D. Crane, and M. Stephenson. 2008. DRAFT. Technical Report on Year One of a Two-Year Screening Study of Bioaccumulation in California Lakes and Reservoirs. Prepared for the Surface Water Ambient Monitoring Program. October 2008.

LARWQCB. 1996. LA Regional Water Quality Control Board 1996 Water Quality Assessment & Documentation – 305(b) Report Supporting Documentation for Los Angeles Region. Developed by the Los Angeles Regional Water Quality Control Board.

Stenstrom, M.K., I.H. Suffet, and V. Vasquez. 2009. Final Data Evaluation Report Field Studies for the Development of Total Maximum Daily Loads for Organochlorine Pesticides and Polychlorinated Biphenyls in Three Los Angeles County Lakes. Prepared by the Institute of the Environment University of California, Los Angeles. Prepared for Regional Water Quality Control Board, Los Angeles. March 2009.

Surface Water Ambient Monitoring Program (SWAMP). 2009. Water quality data available at: http://www.waterboards.ca.gov/water_issues/programs/swamp/. Accessed in 2009.

Toxic Substances Monitoring Program (TSMP). 2009. Fish tissue data available at: http://www.swrcb.ca.gov/water_issues/programs/swamp/mussel_watch.shtml. Accessed in 2009.

UC Riverside. 1994. Evaluation of water quality for selected lakes in the Los Angeles hydrologic basin. Submitted to LARWQCB, December 1994.

Weston Solutions (2007). Sherwood Valley HOA Fish Tissue Study 2007, Final Report. August 2007.

(This page left intentionally blank.)

Appendix H. Methodology for Organochlorine Pesticides and PCBs TMDL Development

(This page left intentionally blank.)

H.1 Introduction

USEPA Region IX is establishing Total Maximum Daily Loads (TMDLs) for impairments in nine lakes in the Los Angeles Region (Figure H-1). USEPA was assisted in this effort by the Los Angeles Water Quality Control Board (Regional Board). Impairments of these waterbodies include low dissolved oxygen/organic enrichment, odor, ammonia, eutrophication, algae, pH, mercury, lead, copper, chlordane, DDT, dieldrin, PCBs, and trash.

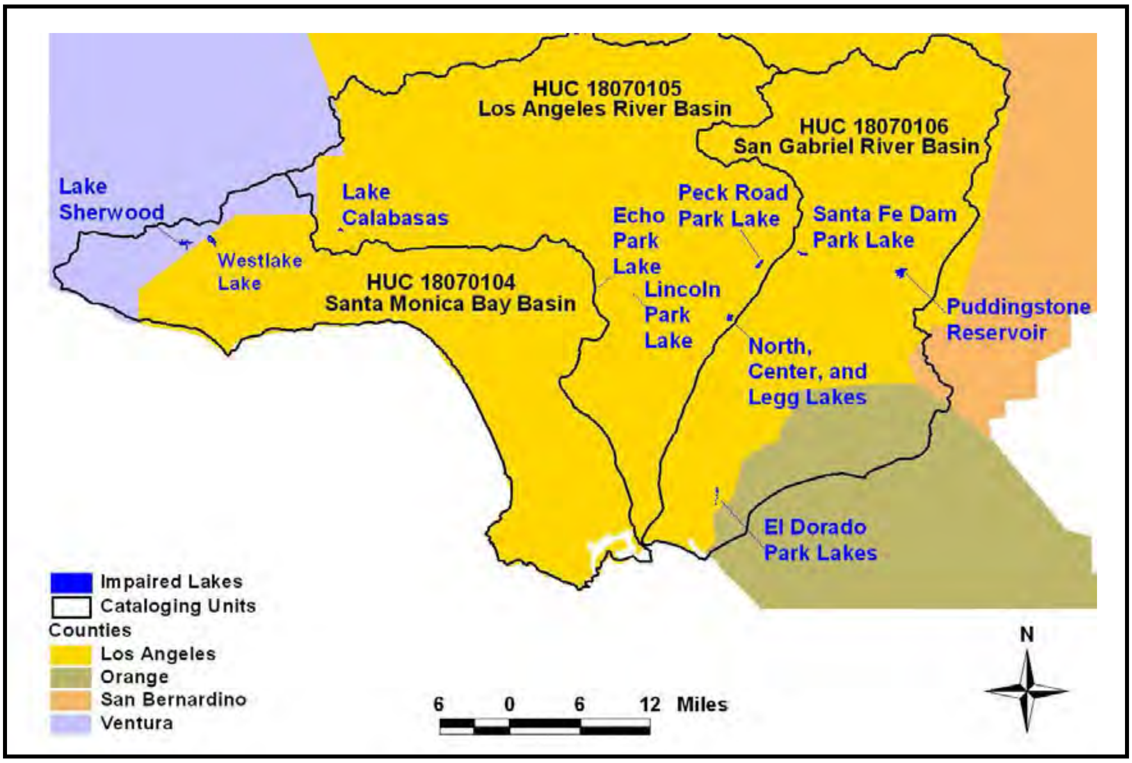


Figure H-1. Location of 10 TMDL Lakes in the Los Angeles Region

Three of these waterbodies are listed as impaired by Organochlorine (OC) Pesticides and PCBs due to elevated fish tissue concentrations: Echo Park Lake, Peck Road Park Lake, and Puddingstone Reservoir. Puddingstone Reservoir was listed for fish tissue concentrations of chlordane, DDT, and PCBs in 1996 and 1998 based on data collected by the Toxic Substance Monitoring Program (TSMP). The listings were carried over to the 2008-2010 303(d) list. The TSMP fish data were also used as the basis for listing PCBs in Echo Park Lake and chlordane and DDT in Peck Road Park Lake. These listings began in 1996 and were also listed on the 1998, 2002, 2006, and 2008-2010 303(d) lists. Recently collected data revealed other impairments not included in the 2008-2010 303(d) listings, but requiring remedial efforts. PCB and dieldrin impairments were identified in Peck Road Park Lake, a dieldrin impairment was identified in Puddingstone Reservoir, and chlordane and dieldrin impairments were found in Echo Park Lake based on fish tissue contamination found in 2004, 2007, and/or 2010 data collected for the Surface Water Ambient Monitoring Program (SWAMP) study. The basis for listings in each lake is shown in Table H-1.

The TMDLs developed for fish tissue contaminations will also reduce OC Pesticides and PCBs in the sediment and water. This appendix discusses the methods used to calculate TMDLs based on the measured tissue concentrations observed in each waterbody. The lake-specific chapters describe data, results, and allocations associated with Echo Park Lake, Peck Road Park Lake, and Puddingstone Reservoir.

Table H-1. OC Pesticides and PCBs Impairments in Los Angeles Region Lakes

Lake	Chlordane	DDT	Dieldrin	Total PCBs
Echo Park Lake	○		○	●
Peck Road Park Lake	●	●	○	○
Puddingstone	●	●	○	●

● Impairment included in both the consent decree and 2008-2010 303(d) list.

○ Impairment identified by new data analyses (after the 2008-2010 303(d) list data cutoff).

H.2 Conceptual Model

Storage in the sediment accounts for the major fraction of OC Pesticides and PCBs in most lake systems. The cycling of OC Pesticides and PCBs between sediment, benthic biota, and aquatic organisms is illustrated in the conceptual model in Figure H-2. The figure illustrates the direct uptake of pollutants by filter feeders and benthic organisms (via adsorption or ingestion) and the indirect uptake of pollutants in fish by consumption of contaminated benthic organisms. Most of the OC Pesticides and PCBs mass that is not incorporated in the aquatic lifecycle will travel through a settling-resuspension cycle of lake particulates. Other transport cycles are also shown in Figure H-2 or described in the Linkage Analysis discussion (Section H.4).

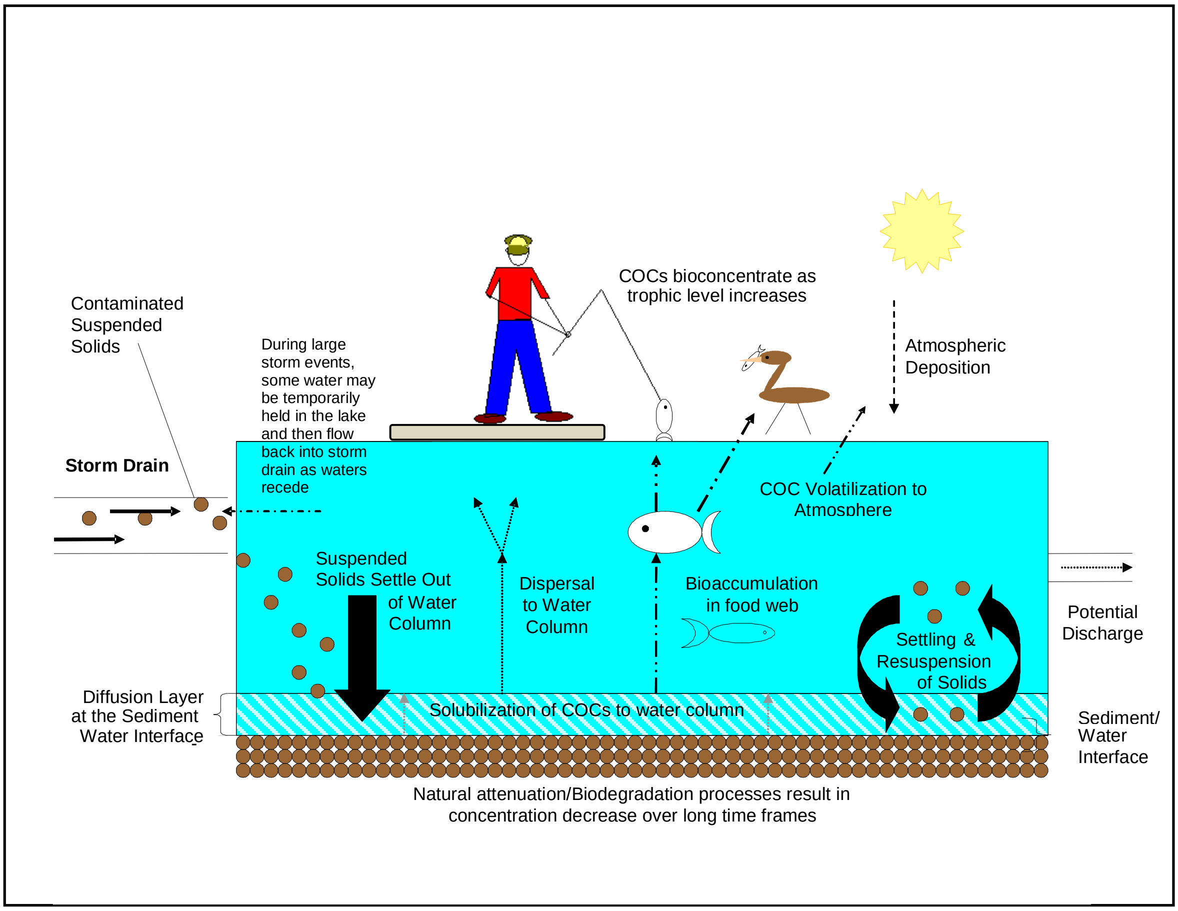


Figure H-2. Conceptual Model for OC Pesticides and PCBs Mobilization

The remainder of this section provides a summary of brief background information on the pollutants addressed in the TMDLs for organic compounds impairments. For the OC Pesticides and PCBs, the general uses and sources of the chemical are explained. There is an abundance of literature for each contaminant's history, chemical characteristics, and toxicological effects, which are rudimentarily summarized for this background.

H.2.1 ORGANOCHLORINE PESTICIDES

Organochlorine (OC) pesticides describes a large collection of pesticides synthetically generated and composed of an organic chemical with at least one chlorine atom. OC pesticides include aldrin, chlordane, DDT, dicofol, dieldrin, endosulfan, endrin, heptachlor, mirex, and toxaphene. The use of OC pesticides was widespread between 1940-1980, at which point, most OCs were banned in the United States (Kalkhoff and Van Metre, 2009). This group of pesticides is often referred to as legacy pesticides, as they continue to persist in the environment long after their initial entry. The OC pesticides addressed for the lake TMDLs are chlordane, dichlorodiphenyltrichloroethane (DDT), and dieldrin. Many of the OC pesticides, including the chemicals of concern here, are nonpolar and highly lipophilic (Connell, 2005), giving them a propensity to bioaccumulate in fats (lipids) in fish tissue.

H.2.1.1 DDT

Dichlorodiphenyltrichloroethane (DDT) is a synthetic organochlorine insecticide once used throughout the world to control insects. Technically DDT consists of two isomers, 4,4’-DDT and 2,4’-DDT, of which the former is the most toxic. In the environment, DDT breaks down to form two related compounds: DDD (tetrachlorodiphenylethane) and DDE (dichlorodiphenyl-dichloroethylene). The sum of DDT, DDD, and DDE is referred to as total DDTs. DDT and its degradation products are colorless crystalline solids and exhibit physical properties of low water solubility and high lipophilicity, which play a key role in its environmental fate (LARWQCB, 2009a; LARWQCB, 2009b). DDT became widely used as a pesticide in 1939. During World War II, its use was focused on controlling disease-carrying insects, such as mosquitoes and lice (USEPA, 1975). DDT for agricultural and commercial uses started after 1945. Use of DDT peaked in 1959, at which time approximately 80 million pounds were being applied annually. In California, DDT was widely used for control of both agricultural and disease-carrying pests (Mischke et al., 1985). In 1963 the California Department of Food and Agriculture (CDFA) declared DDT a restricted material. The last year that substantial amounts of DDT were applied in California was 1970, when roughly 1.2 million pounds of DDT were applied, primarily to agricultural areas (Mischke et al., 1985).

The overall use of DDT started to decline in the early 1970s because of restrictions and reporting uses, in addition to the developed resistance of the pests that were previously sensitive to DDT (USEPA, 1975). Furthermore, new more effective pesticides had been developed, and there was growing public concern over adverse human and environmental health effects from DDT exposure (USEPA, 1975). Even though domestic usage of DDT has been banned for more than 30 years, there are still widespread environmental impairments caused by DDT and DDT-associated degradation products.

Because DDT exhibits such low water solubility, it is mainly concentrated in soils and will bind strongly to the organic fraction of sediments (Walker et al., 2001). DDT has an estimated half-life in soil of two to sixteen years (Connell, 2005). DDT is transported to surface waterbodies through the sediment and erosion runoff. DDT in the water column will remain partitioned to sediment or other organic mediums (living organisms).

DDT is also highly lipophilic and will accumulate in the fatty tissues of exposed wildlife and biomagnify as it moves through the food chain to reach the primary predator (NPIC, 1999). The ability of DDT to biomagnify is one of the primary environmental concerns of this pollutant because the exposure increases from one trophic level to another.

H.2.1.2 Chlordane

Chlordane is a white solid pesticide that was first registered and approved for agricultural and non-agricultural uses in the United States in 1948. Chlordane is actually a generally encompassing term used to describe technical chlordane, the common pesticide formula which is composed of over 50 different closely-related compounds. Technical chlordane includes heptachlor, nonachlor, chlordane and similar chemicals. The true chlordane compound composes roughly 40 percent of the technical mixture in two isomers: alpha-chlordane and gamma-chlordane (NPIC, 2001).

Non-agricultural uses of chlordane included treating pests in residential lawns and gardens as well as structural pests such as termites. Chlordane was used on a variety of agricultural crops including corn, citrus, deciduous fruits and nuts, and vegetables. USEPA banned the use of chlordane on all food crops, lawns, and gardens in 1978. It was still registered as a termiticide until 1988, when USEPA expanded the chlordane ban to all uses (USEPA, 2009a).

As an organochlorine pesticide, chlordane has similar properties to DDT. It has low water solubility, a strong binding affinity to soil particles, and is persistent in the environment, with a half-life in soils of approximately four years (EXTOXNET, 1996). Soils historically treated with chlordane can continue to be a present source of chlordane in the environment and contaminated soils can be transported to waterbodies via runoff. Moreover, chlordane will bioaccumulate in the fat tissue of exposed organisms and is considered highly toxic to fish and freshwater invertebrates (NPIC, 2001; EXTOXNET, 1996).

H.2.1.3 Dieldrin

Dieldrin is a man-made organochlorine pesticide product, but can also be produced through the natural and metabolic degradation of aldrin, another organochlorine pesticide (USEPA, 2008). Dieldrin was originally developed as an alternative to DDT and mainly used between 1950 and 1970. It was applied to structures for termite control and used in agriculture for control of soil insects such as corn rootworms, cutworms, and locusts in citrus, corn, and cotton crops (ATSDR, 2002; USEPA, 2008). Use of dieldrin peaked in 1966 at one million pounds and dropped to 670,000 pounds in 1970, during the same period the use of aldrin dropped from 19 million pounds to 10.5 million pounds (USEPA, 1980).

In 1970, all registered uses for both pesticides were cancelled by the US Department of Agriculture (USDA), but the USEPA lifted the cancellation under the authority of the Federal Insecticide, Fungicide and Rodenticide Act (FIFRA) in 1972 for deep ground insertions for termite control, nursery clippings of roots and tops of non-food plants, and moth-proofing. In 1974, the manufacturing of aldrin and dieldrin was suspended, and in 1987 all uses of dieldrin were cancelled (USEPA, 2008).

Dieldrin is resistant to biotic and abiotic degradation, becomes sequestered in the soil with time, and therefore persists in the environment. The half life in soils is between six months and three years (Connell, 2005; Alexander, 1999). Similar to the other organochlorine pesticides, dieldrin also has a strong affinity to soil particles and lipids and a low solubility in water. The most common exposure routes of dieldrin are from living in houses treated with dieldrin to control termites and consumption of root crops, fish, and seafood. Dieldrin has a wide range of suspected negative effects in living organisms. Most often in humans, dieldrin damages functions of the nervous system (ASTDR, 2002).

H.2.2 POLYCHLORINATED BIPHENYLS

Polychlorinated biphenyls (PCBs) consist of two phenyl rings with from one to ten chlorine atoms attached. Individual PCB compounds, referred to as congeners, vary in the number and placement of the chlorine atoms. There are a total of 209 possible congeners, which vary in physical properties and toxicity (ATSDR, 2001). Some commercial mixtures of PCBs are known by the trade name Aroclor. Most PCBs are oily liquids or waxy solids, and some can exist as a vapor in air (ATSDR, 2001; USEPA, 2009b). There are no natural sources of PCBs.

PCBs were manufactured in the U.S. from 1929 until production was banned in 1979. The cumulative production of PCBs in the United States from 1930 to 1985 is estimated at 1.4 billion pounds (USEPA, 2010). PCBs were used for a variety of applications and functions, including coolants and lubricants in transformers, capacitors, and other electrical equipment; heat transfer and hydraulic fluids; fluorescent light ballasts; cable insulation and thermal insulation; adhesives and tape; varnishes, surface coatings and paints; caulking; plastics; and carbonless copy paper (USEPA, 2009b). Useful characteristics, such as non-flammability, chemical stability, and insulating ability, resulted in use of PCBs for myriad industrial and commercial purposes (USEPA, 2009b).

Prior to the 1979 ban on manufacturing, PCBs were released into the environment during their production and various uses (USEPA, 2009b). USEPA regulates PCBs under the Toxic Substances Control Act (TSCA), which generally bans the manufacture, use, and distribution in commerce of the chemicals in products at concentrations of 50 parts per million or more. TSCA allows USEPA to authorize certain continued uses of PCBs, such as to rebuild existing electrical transformers during the transformers' useful life, which may be 30 years or more. PCBs are also still present in older materials made prior to 1979, such as paint, and caulking (USEPA, 1999).

PCBs enter the environment through improper disposal of industrial waste; releases or leachate from abandoned manufacturing areas and waste sites; and leaks and/or improper dumping of materials containing PCBs. Global cycling of PCBs occurs when they volatilize from soils and/or surface waters, are transported into the atmosphere, and are then redeposited to land and surface waters (USEPA, 1999; ATSDR, 2001). This process plays an important role in the transport and deposition of PCBs to surface waters (USEPA, 1999; USEPA, 2009b).

PCBs have low water solubility and are highly lipophilic, with variation dependent on the characteristics of the individual congeners (USEPA, 1999). PCBs bind strongly to soils and natural organic matter, which can be transported to surface waters through runoff (USEPA, 1999). Because of their high lipophilicity, PCBs are stored in the fat tissue of exposed organisms and bioaccumulate through the food chain. Bioconcentration factors generally increase with chlorine content of the congeners. Because PCBs concentrate in the food chain, a small concentration in water or sediment can produce a significant environmental impact.

PCBs are resistant to abiotic and biotic degradation and the resistance increases as the chlorination of the compound increases. Historical loads of PCBs, stored in lake sediments, can continue to contaminate the aquatic food chain for many decades.

H.3 Source Assessment

The OC Pesticides and PCBs addressed in these TMDLs are no longer in production and use is either banned or strictly limited. For this reason, loading to the lakes is expected to have declined over time, and the historic loads that are stored in lake sediment appear to be the major source of bioaccumulation in fish. Nonetheless, any ongoing loads must also be addressed in the TMDL.

Sources of chlordane, DDT, dieldrin, and PCBs that cause contamination in a waterbody may include both point and nonpoint sources. Federal regulations distinguish between allocations for point sources regulated under NPDES permits (for which waste load allocations are established) and nonpoint sources that are not regulated through NPDES permits (for which load allocations are established) (see 40 CFR 130.2). Continuing loads of OC Pesticides and PCBs into the lakes is from permitted stormwater discharges by municipalities. Chlordane, DDT and dieldrin are expected to be in highest concentrations near agricultural land, on which pesticides and insecticides were used heavily. Older industrial sites are more likely to contain PCBs where they were used or integrated into substances such as coolants, lubricants, and surface coatings. Older residential areas are also potential sources of PCBs and organochlorine pesticides.

H.3.1 POINT SOURCES

Discharges that occur at one or more defined points, such as a pipe or storm drain outlet, are defined as point sources. Most point sources are regulated through the NPDES permitting process.

H.3.1.1 MS4 Permittees

In 1990, USEPA developed rules establishing Phase 1 of the NPDES stormwater program, designed to prevent pollutants from being washed into the Municipal Separate Storm Sewer Systems (MS4) by stormwater runoff, or from being directly discharged into the MS4 and then discharged into local waterbodies. Phase I of the program required operators of medium and large MS4s (those generally serving populations of 100,000 or more) to implement a stormwater management program as a means to control polluted discharges. Phase II of the program extends the requirements to operators of small MS4 systems, which must reduce pollutants in stormwater to the maximum extent practicable (MEP) to protect water quality.

OC pesticides and PCB loads from urban stormwater runoff and associated sediment are estimated from monitoring data collected from the lake sediments near drainage inputs (Appendix G, Monitoring Data) and simulated sediment loads from a previously developed LSPC model of the San Gabriel and Los Angeles river basins (Appendix D, Wet Weather Loading) (Tetra Tech, 2004; Tetra Tech, 2005). To estimate runoff volumes and sediment loads, average monthly areal flow rates have been extracted for each land use and applied to the land use composition that drains to an MS4 for each lake. Sediment event mean concentrations for each land use are used to estimate sediment loads. The LSPC model results and estimated sediment loading for each contributing MS4 system are described in further detail in Appendix D (Wet Weather Loading).

Because OC Pesticides and PCBs are strongly sorbed to sediment, loading and transport during dry weather flow is assumed to be insignificant. Therefore, loading estimates are based on sediment delivery and no separate load calculation is performed for dry weather flows.

H.3.1.2 Other NPDES Discharges

In addition to MS4 stormwater dischargers, the NPDES program regulates stormwater discharges associated with industrial and construction activities and non-stormwater discharges (individual and general permits). Loading of OC Pesticides and PCBs from non-MS4 NPDES discharges is expected to be negligible because the contaminants addressed in these TMDLs are no longer in use and have been banned for over 20 years. To quantify OC Pesticides and PCBs loading from non-MS4 discharges, the permit databases maintained by the Los Angeles Regional Board were downloaded for the Los Angeles and San Gabriel river basins. Geographic information listed for each permit was used to determine which facilities are located in the watersheds of the three OC Pesticides and PCBs-impaired lakes. OC Pesticides and PCBs loading from each facility was estimated based on the reported disturbed area. The facilities and estimated loads are described in more detail in the lake-specific sections of this report.

H.3.1.3 Additional Inputs

One of the lakes addressed by these TMDLs has supplemental water additions from groundwater wells or potable water that maintain its lake level. Access and monitoring data for these inputs are limited and no specific OC Pesticides and PCBs analyses are available. OC Pesticides and PCBs loading from unknown inputs are encompassed in the calculated loading because the loadings are based on the observed data, which capture all sources upstream of the monitoring station.

H.3.2 NONPOINT SOURCES

OC Pesticides and PCBs loading from nonpoint sources originates from sources that do not discharge at a defined point. This section describes the methods used to estimate loading from nonpoint sources.

H.3.2.1 Watershed Loading

OC Pesticides and PCBs loads from areas that do not drain to an MS4 system are also estimated from monitoring data collected from the lake sediments near drainage inputs (Appendix G, Monitoring Data) and simulated sediment loads (Appendix D, Wet Weather Loading). Two flow-calibrated LSPC models were previously developed for the San Gabriel and Los Angeles river basins (Tetra Tech, 2004; Tetra Tech, 2005). To estimate runoff volumes and sediment loads, average monthly areal flow rates have been extracted for each land use and applied to the land use composition that does not drain to an MS4 for each lake. Sediment event mean concentrations for each land use are used to estimate sediment loads. Appendix D (Wet Weather Loading) describes the LSPC model output and estimated sediment loading for areas that do not discharge to an MS4.

H.3.2.2 Atmospheric Deposition

The atmospheric deposition of OC Pesticides and PCBs on the watershed is accounted for in the annual runoff loads. The direct net deposition of OC Pesticides and PCBs (on the lake surfaces) is estimated to be minimal in comparison to the indirect loading. The surface area of each impaired lake is only a small portion of the total draining area for each lake. The lake surface area of Peck Road Park Lake represents only 0.37 percent of the total surface area draining to the lake. The atmospheric deposition from the remaining drainage area for the lake (99.63 percent) is accounted for in the annual runoff loads collected in the MS4 system. The area for direct deposition for Echo Park Lake and Puddingstone Reservoir is 1.8 percent and 3.1 percent of the total draining surface area (respectively). Moreover, research of OC Pesticides and PCBs exchange between waterbodies and atmosphere has demonstrated a recent shift in equilibriums that causes OC Pesticides and PCBs to be expelled from the lake and into the atmosphere under conditions of declining loads. The volatilization of OC Pesticides and PCBs may be greater than the direct atmospheric deposition into the lake (Manodori, et al., 2007; Thomann and Di Toro, 1983). Thus, direct deposition of OC Pesticides and PCBs to the lake surface is not evaluated as a loading source in these TMDLs.

H.3.2.3 OC Pesticides and PCBs Stored in Lake Sediment

Historical loading of OC Pesticides and PCBs has resulted in storage of these contaminants in lake sediment. In most cases, this legacy storage appears to be the major source of OC Pesticides and PCBs in the food chain. Benthic macroinvertebrates accumulate OC Pesticides and PCBs from the sediment and are consumed by sediment foraging fish, which in turn are consumed by higher trophic level fish, resulting in bioconcentration of OC Pesticides and PCBs.

The sediment stores of OC Pesticides and PCBs do not constitute an ongoing load and are thus not amenable to a traditional load allocation in mass per time units. Instead, a target concentration is assigned to achieve FCGs based on a BSAF analysis (see Section H.4.1).

(This page left intentionally blank.)

H.4 Linkage Analysis

The linkage analysis provides the quantitative basis for determining the loading capacity of each impaired lake. The loading capacity is used to estimate the TMDL, and allocate that load to permitted, point sources (wasteload allocations) and nonpoint sources (load allocations). The TMDL also contains a Margin of Safety.

The OC Pesticides and PCBs TMDLs for the three lakes assess watershed loading into the lakes using monitoring data and sediment loads simulated by a previously developed LSPC model calibrated for the Los Angeles and San Gabriel river basins. The simulated sediment loads are based on the characteristics of the watershed land uses and incorporate dry, normal, and wet conditions for the Los Angeles area. The LSPC model is discussed in further detail in Appendix D (Wet Weather Loading) and by Tetra Tech (2004, 2005).

For many of the OC Pesticides and PCBs impairments, concentrations in water and sediment meet applicable criteria for those media, but concentrations in fish exceed FCGs. The CTR criteria for the protection of human health are designed to protect against elevated fish tissue concentrations due to bioaccumulation from the water column, but do not address bioaccumulation from the sediment. The consensus-based TEC targets are designed to protect against direct toxicity to benthic organisms, but explicitly do not consider food chain bioaccumulation. Therefore, a separate linkage analysis is needed to determine the sediment exposure concentration that will achieve FCGs.

Lake sediments are often the predominant source of OC pesticides and PCBs in the water column. The bottom sediment serves as a sink for organic compounds that can be recycled through the aquatic life cycle. OC Pesticides and PCBs have long half-lives in sediment and water and decay will not be a significant mechanism of reduction. Incoming loads of OC Pesticides and PCBs will mainly be adsorbed to particulates in stormwater runoff (eroded sediments from legacy contamination sites or from atmospheric deposition).

H.4.1 REPRESENTATION OF BIOACCUMULATION FROM SEDIMENT

A linkage between the OC Pesticides and PCBs concentrations in sediment and the concentration in the impaired fish species is established using an empirical relationship based on a biota-sediment accumulation factor (BSAF).

Bioaccumulation of OC Pesticides and PCBs from contaminated sediment is described using biota-sediment accumulation factors (BSAFs). The BSAF describes the pollutant ratio between sediment and aquatic biota. The species and environmental factors are accounted for by normalizing the ratio to the fraction of organic carbon in sediments and fraction of lipids in the biota:

$$BSAF = \frac{C_{biota} / f_l}{C_{sed} / f_{OC}},$$

where C_{biota} is pollutant concentration in the benthic organism or benthic community, C_{sed} is the pollutant concentration in the sediment, f_l is the fraction of lipids in the biota, and f_{OC} is the fraction of organic carbon in the sediment.

Typical BSAF values are provided by Wong et al. (2001). Measurements of contaminants in sediment and fish from hundred of sites in the United States were compiled for data between 1992 and 1995 and analyzed by Wong et al. (2001). There were several different fish taxa included in the analysis; most (88 percent) of the samples were benthic species (carp, white sucker, channel catfish, etc.), but some pelagic fish were included (trout and bass) in the calculation. The BSAF values for the selected pollutants are shown in Table H-2.

Table H-2. Typical BSAF[1] Values

Pollutant	BSAF
Chlordane	2.9
DDT[2]	1.1
Dieldrin	3.4
Total PCBs	2.4

[1]Typical values from Wong et al. (2001).
[2]Based on o'p-DDT.

The BSAF can be used to determine the associated equilibrium sediment concentration, as shown below:

$$BSAF = \frac{C_{biota}/f_l}{C_{sed}/f_{OC}}\text{; therefore } C_{sed} = f_{OC} \cdot \frac{C_{biota}}{f_1} \div BSAF$$

The maximum allowable sediment concentration that can exist without causing impairment to the fish is determined using the FCGs for C_{biota}. The difference between the existing sediment concentration and the maximum allowable concentrations are compared to determine necessary reductions.

$$C_{biota} = FCG$$

$$C_{sed-target} = f_{OC} \cdot \frac{FCG}{f_1} \div BSAF$$

The allowable fraction of existing sediment concentration is then simply

$$\frac{C_{sed-target}}{C_{sed}} = \frac{FCG}{C_{biota}}$$

The sediment targets calculated from the BSAF analysis for each lake and the applicable OC Pesticides and PCBs are described in the lake-specific chapters.

H.4.2 EQUILIBRIUM MODEL FOR OC PESTICIDES AND PCBS IN LAKES

The linkage analysis also employs a model of in-lake processes, described in Butcher (1997), Chapra (1991), and Chapra and Reckhow (1983). In general, the steady-state model presented here uses the notation and solutions of the full steady-state model presented in Chapra and Reckhow (1983), which accounts for partitioning, losses, burial, and recycling from the sediment. This model idealizes the lake as three zones, representing the water column, mixed or active sediment layer, and deep sediment and derives mass balances for each layer. The equilibrium model can be used to determine the rate of external loading that would be required to account for current observed sediment concentrations under steady-state conditions. It can also be used to estimate concentrations in water and sediment when these are below analytical detection limits.

Chapra's steady-state solution for contaminant concentration in the mixed sediment layer, $c_{t,m}$(μg/m^3) is

$$c_{t,m} = \frac{F_{dw}}{F_{dp}} R_{df}\, c_{t,w},$$

where $c_{t,w}$(μg/m^3) is the steady-state concentration in the water column, F_{dw} is the dissolved fraction in the water column, F_{dp} is the ratio of sediment porewater pollutant concentration to the total concentration of contaminant in the sediment, and R_{df} is the diffusive feedback ratio, i.e., the ratio of contaminant concentration in porewater to that dissolved in the water column. These are defined as follows:

$$F_{dw} = \frac{1}{1 + K_{d,w}\, s_{t,w}}$$

$$F_{dp} = \frac{1}{\phi + (1-\phi)\rho_p\, K_{d,s}}$$

$$R_{df} = \frac{\phi\left(D_s \big/ z'_b\right) + K_{d,w}\, s_{t,w}\, v_w \left(A_w \big/ A_m\right)}{\phi\left(D_s \big/ z'_b\right) + s_{t,w}\, v_w \left(A_w \big/ A_m\right)\left[K_{d,s} + \phi \big/ \left[(1-\phi)\rho_p\right]\right] + \left(k_m\, z_m \big/ F_{dp}\right) - \phi D_s \lambda_2}$$

In these equations,

$K_{d,w}$ is the partition coefficient to solids in the water column (m^3/g),

$K_{d,s}$ is the partition coefficient to solids in the sediment (m^3/g),

$s_{t,w}$ is the solids concentration in the water column (g/m^3),

φ is the sediment porosity (unitless),

ρ_p is the density of solids (g/m^3),

D_s is the diffusion rate for the contaminant in porwater (m^2/yr),

z'_b is a thickness (m) defining the gradient between the mixed sediment layer and the overlying water – nominally the average of the mixed layer depth and the overlying laminar layer,

v_w is the settling velocity of solids (m/yr),

A_w is the water surface area (m^2),

A_m is the sediment surface area (m^2),

k_m is the first order decay rate for the contaminant in sediment (yr^{-1}), assumed equal for the mixed and deep sediment layers,

z_m is the sediment mixed layer depth (m), and

$$\lambda_2 = \frac{v_b}{2\phi D_s\, F_{dp}}\left[1 - \sqrt{1 + \frac{4\phi F_{dp} D_s k_m}{{v_b}^2}}\right],$$

with v_b being the resuspension velocity (m/yr), defined as

$$v_b = \frac{v_w A_w s_{t,w}}{A_m (1-\phi)\rho_p} .$$

The steady-state solution for the fully-mixed water column concentration is given by

$$c_{t,w} = \frac{W_c}{Q + k_w V_{t,w} + v_a A_w} ,$$

where

W_c is the mass loading rate of the contaminant (µg/yr),

Q is the outflow (m^3/yr),

k_w is the first-order decay coefficient in the water column (yr^{-1}),

$V_{t,w}$ is the volume of the water column (m^3), and

$$v_a = F_{pw} v_w + \phi \frac{D_s}{z'_b} \frac{A_m}{A_w} F_{dw} \left(1 - R_{df}\right) - v_r \frac{A_m}{A_w} \frac{F_{dw}}{F_{dp}} R_{df} .$$

Here, F_{pw} is the fraction of pollutant mass attached to particulate matter in the water column,

$$F_{pw} = \frac{K_{d,w} s_{t,w}}{1 + K_{d,w} s_{t,w}} ,$$

and v_r is the resuspension velocity (m/yr).

Chapra's steady-state toxicant formulation does not explicitly account for volatilization losses; however, these are readily included in the general water column decay coefficient (k_w, yr^{-1}) by inclusion of a term v_v/H, where v_v is a volatilization velocity (m/yr) and H is the average lake depth (m). Volatilization velocity may be estimated by the two-film method of Mackay (1981) as

$$\frac{1}{v_v} = \frac{1}{K_l} + \frac{1}{K_g H_e} ,$$

where K_l is the liquid side mass transfer coefficient (m/yr), K_g is the gas side mass transfer coefficient, and H_e is the dimensionless Henry's Law constant, a measure of volatility. Methods to approximate values of the transfer coefficients from wind speed (W, m/s) and molecular weight (M) are also given in Chapra and Reckhow (1983). In units of m/yr, these are:

$$K_l = 204.4 \frac{W^{3/2}}{\sqrt{M}} \text{ and } K_g = 43800 \frac{W}{\sqrt{M}} .$$

The rate loss of a toxicant in the waterbody depends on physical and chemical characteristics, such as volatility, degradability, and tendency to sorb to particulate matter. The chemical-specific parameters used for the simulations were selected from Brunner et al. (1990), Hansen et al. (1999), Leatherbarrow et al. (2006), Li et al. (1990), and Mackay et al. (1992). Henry's law coefficients are weighted for presence of individual congeners and a separate PCB coefficient was determined for each waterbody. These values are displayed in Table H-3.

Table H-3. Chemical-Specific Parameters for Simulation

Parameters for Model Input	Total Chlordane	Total DDTs	Dieldrin	Total PCBs
Molecular weight	409.6[e]	355[e]	381[e]	326[e]
Dimensionless Henry's Law Constant (H_e) at 20°C	1.18E-03 [c]	1.21E-04[c]	2.14E-04[c]	6.70E-03[a]
Partition coefficient (K_{OC}; l/kg)	38,000[d]	240,000[d]	12,000[d]	676,000[b]
Degradation rate in sediment (yr^{-1})	0.30[c]	0.08[c]	0.25[c]	2.99[e]
Degradation rate in water (yr^{-1})	0.73[c]	0.73[c]	0.84[c]	2.07[e]

Sources: (a) Brunner et al. (1990); (b) Hansen et al. (1999); (c) Leatherbarrow et al. (2006); (d) Li et al. (1990); (e) Mackay et al. (1992).

Loss rates are also dependent on the physical characteristics of the individual lakes; specifically surface area, volume, drainage area, annual runoff, and organic carbon fraction in the sediments. The lake-specific parameters were gathered mainly from SCAG 2005 land use data. The organic carbon fraction of the lake sediment was calculated for each lake using data collected by USEPA and the Regional Board during sampling events conducted in 2008 and 2009. For other lake characteristics (e.g., sediment solids density, settling velocity, resuspension velocity, active sediment thickness, and sediment porosity), the model uses typical or assumed values appropriate for these lakes, as reported by Chapra and Reckhow (1983). The presumed values are shown in Table H-4.

Table H-4. Assumed Parameters for Simulation

Parameters for Model Input	Assumed Value
Sediment solids density (g/cm^3)	1.38
Settling velocity (m/yr)	100
Resuspension velocity (m/yr)	0.007
Active sediment thickness (cm)	5.0
Sediment porosity (unitless)	0.8
Diffusion rate in sediment porewater (m^2/yr)	0.01

(This page left intentionally blank.)

H.5 TMDL Development

A TMDL is defined by the loading capacity. The loading capacity of a waterbody represents the maximum amount of pollutant loading that can be assimilated without violating water quality standards (40 CFR 130.2(f)). The OC Pesticides and PCBs TMDLs are calculated based on the maximum amount of organochlorine compound loading consistent with meeting the fish tissue goals.

H.5.1 LOADING CAPACITY AND ALLOCATIONS

The loading capacity for each lake and the applicable OC Pesticides and PCBs are determined using the target sediment concentration. Estimates of the existing sediment load to each lake are discussed in Appendix D (Wet Weather Loading) and the individual lake chapters. The loading capacity is expressed as a concentration in micrograms per dry kilogram (µg /kg dry weight). The loading capacity can be further broken down into the wasteload allocations (WLAs), load allocations (LAs), and Margin of Safety (MOS) using the general TMDL equation:

$$TMDL = Loading\ Capacity = \sum WLAs + LAs + MOS$$

Because the loading capacity is presented as a concentration, the WLAs and LAs are also shown as a concentration for each jurisdiction and subwatershed. The watershed areas associated with permitted Municipal Separate Storm Sewer Systems (MS4s) are assigned wasteload allocations, which are further broken down by jurisdiction and subwatershed. In addition, general industrial and general construction stormwater permittees are also assigned wasteload allocations. Load allocations are assigned to areas not draining to an MS4. The specific allocations for each lake are described by jurisdiction and subwatershed in further detail in their respective chapters.

H.5.2 MARGIN OF SAFETY

TMDLs must include a margin of safety (MOS) to account for any lack of knowledge concerning the relationship between load and wasteload allocations and water quality. The MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. This TMDL contains an implicit MOS based on conservative assumptions. The allocations are set based on the lower of either the BSAF-derived sediment target or the consensus-based TEC sediment target to ensure achievement of the OEHHA FCG target in fish tissue. The selected BSAF-derived target concentration in sediment is considerably lower than the consensus-based TEC target.

H.5.3 DAILY LOAD EXPRESSION

Sediment contamination and resulting bioaccumulation is a long-term process and annual loading rates are the most appropriate measure for the TMDL. However, USEPA recommends inclusion of a daily load expression for all TMDLs to comply with the 2006 D.C. Circuit Court of Appeals decision for the Anacostia River TMDL. The TMDLs developed here each include a daily maximum load estimate consistent with the guidelines provided by USEPA (2007). Because the majority of external OC Pesticides and PCBs loads occur during wet weather events that deliver sediment to the lakes, the maximum allowable daily load is calculated from the 99th percentile flow multiplied by the sediment concentration target of the OC Pesticides and PCBs and the sediment event mean concentration (annual average sediment load divided by annual average flow from the watershed).

(This page left intentionally blank.)

H.6 References

ATSDR. 2001. Agency for Toxic Substances and Disease Registry Division of Toxicology ToxFAQs Fact Sheet – Polychlorinated Biphenyls. Online [http://www.atsdr.cdc.gov/tfacts17.pdf]. Accessed January 2010.

ATSDR. 2002. Agency for Toxic Substances and Disease Registry (ATSDR). Toxicological profile for Aldrin/Dieldrin. Atlanta, GA: U.S. Department of Health and Human Services, Public Health Service.

Alexander, M. 1999. *Biodegradation and Bioremediation.* 2nd Edition. Academic Press, San Diego CA.

Brunner, S., E. Hornung, H. Santi, E. Wolff, O.G. Piringer, J. Altschuh, and R. Brueggemann. 1990. Henry's law constants for polychlorinated biphenyls: Experimental determination and structure-property relationships. *Environmental Science and Technology.* 24(11): 1751:1754.

Butcher, J.B. 1997. Toxics zoning for reservoir source water protection. *Journal of Lake and Reservoir Management,* 13(4): 281-291.

Chapra, S.C. 1991. Toxicant-loading concept for organic contaminants in lakes. *Journal of Environmental Engineering,* 117(5): 656-677.

Chapra, S.C. and K.H. Reckhow. 1983. *Engineering Approaches for Lake Managemen*t, Volume 2: Mechanistic Modeling. Butterworth Publishers, Boston.

Connell, D.W. 2005. *Basic Concepts of Environmental Chemistry.*_2nd Edition. CRC Press, Taylor and Francis Group, Boca Raton FL.

EXTOXNET. 1996. Extension Toxicology Network (EXTOXNET) Pesticide Information Profiles, General Fact Sheet Chlordane. Revised June 1996. http://extoxnet.orst.edu/pips/chlordan.htm, accessed on January 22, 2010.

Hansen, B.G., A.B. Paya-Perez, M. Rahman and B.R. Larsen. 1999. QSARs for K_{OW} and K_{OC} of PCB Congeners: A critical examination of data, assumptions, and statistical approaches. *Chemosphere:* 39(13): 2209-2228.

Kalkhoff, S.J. and P.C. Van Metre. 2009. Organochlorine Compounds in a Sediment Core From Coralville Reservoir, Iowa. US Geological Survey Fact Sheet FS-129-97. Online [http://ia.water.usgs.gov/nawqa/factsheets/fs-129/fs-129.html]. Accessed January, 2010.

LARWQCB. 2005. Calleguas Creek Watershed Toxicity TMDL. Los Angeles Regional Water Quality Control Board.

LARWQCB. 2009a. Colorado Lagoon Organochlorine Pesticides, PCBs, Sediment Toxicity, PAHs and Metals TMDL. Los Angeles Regional Water Quality Control Board.

LARWQCB. 2009b. McGrath Lake PCBs, Organochlorine Pesticides, and Sediment Toxicity TMDL. Los Angeles Regional Water Quality Control Board.

Leatherbarrow, J.E., N. David, B.K. Greenfield, J.J. Oram, and J.A. Davis. 2006. Organochlorine pesticide fate in San Francisco Bay. RMP Technical Report: SFEI Contribution #433. San Francisco Estuary Institute, Oakland. CA.

Li, W., D.E. Merrill, and D.A. Haith. 1990. Loading functions for pesticide runoff. *Research Journal WPCF*: 62(1): 16-26.

Mackay, D. 1981. Environmental and laboratory rates of volatilization of toxic chemicals from water. pp. 303-322 *in* J. Saxena and F. Fisher, eds., *Hazard Assessment of Chemical, Current Developments*, vol. I. Academic Press, New York.

Mackay, D., W.Y. Shiu and K.C. Ma. 1992. *Illustrated Handbook of Physical-Chemical Properties and Environmental Fate for Organic Chemicals: Monoaromatic Hydrocarbons, Chlorobenzenes, and PCBs.* Chelsea, MI: Lewis.

Manodori, L., A. Gambaro, I. Moret, G. Capodaglio, and P. Cescon. 2007. Air-sea gaseous exchange of PCB at the Venice lagoon (Italy). *Marine Pollution Bulletin*. 54(10): 1634-1644.

Mischke, T., K. Brunetti, V. Acosta, D. Weaver, and M. Brown. 1985. *Agricultural Sources of DDT Residues in California's Environment*. California Department of Food and Agriculture. Report prepared in response to House Resolution No. 53.

NPIC. 1999. National Pesticide Information Center - DDT General Fact Sheet. Online [http://npic.orst.edu/factsheets/ddtgen.pdf]. Accessed January, 2010.

NPIC. 2001. National Pesticide Information Center - Chlordane Technical Fact Sheet. Online [http://npic.orst.edu/npicfact.htm]. Accessed January, 2010.

Tetra Tech. 2004. Model Development for Simulation of Wet-Weather Metals Loading from the Los Angeles River Watershed. Prepared for USEPA Region IX and the Los Angeles Regional Water Quality Control Board.

Tetra Tech. 2005. Model Development for Simulation of Wet-Weather Metals Loading from the San Gabriel River Watershed. Prepared for USEPA Region IX and the Los Angeles Regional Water Quality Control Board.

Thomann, R.V. and D.M. Di Toro. 1983. Physico-Chemical Model of Toxic Substances in the Great Lakes. *Journal of Great Lakes Research*. 9(4):474-496.

USEPA. 1975. DDT Regulatory History: A Brief Survey (to 1975). U.S. Environmental Protection Agency. Online [http://www.epa.gov/history/topics/ddt/02.htm]. Accessed January 2010.

USEPA. 1980. Ambient Water Quality Criteria for Aldrin/Dieldrin. U.S. Environmental Protection Agency. Online [http://www.epa.gov/waterscience/criteria/library/ambientwqc/aldrindieldrin.pdf]. Accessed January 2010.

USEPA. 1999. Polychlorinated Biphenyls (PCBs) Update: Impact on Fish Advisories. U.S. Environmental Protection Agency. EPA-823-F-99-019.

USEPA. 2007. Options for Expressing Daily Loads in TMDLs. U.S. Environmental Protection Agency Office of Wetlands, Oceans & Watersheds, June 22, 2007 Draft.

USEPA. 2008. Aldrin/Dieldrin. Persistent Bioaccumulative and Toxic Chemical Program. U.S. Environmental Protection Agency. Online [http://www.epa.gov/pbt/pubs/aldrin.htm]. Accessed January 2010.

USEPA. 2009a. Ground Water and Drinking Water, Consumer Factsheet on Chlordane. U.S. Environmental Protection Agency. Online [http://www.epa.gov/OGWDW/contaminants/dw_contamfs/chlordan.html]. Accessed January 2010.

USEPA. 2009b. Polychlorinated Biphenyls, Basic Information. U.S. Environmental Protection Agency Online. [http://www.epa.gov/epawaste/hazard/tsd/pcbs/pubs/about.htm]. Accessed January 2010.

USEPA. 2010. Polychlorinated Biphenyls (PCBS) Environmental Occurrence. U.S. Environmental Protection Agency Technology Innovation Program. Online [http://www.clu-in.org/contaminantfocus/default.focus/sec/Polychlorinated_Biphenyls_(PCBs)/cat/Environmental_Occurrence/]. Accessed January 2010.

Walker, C.H., Hopkin, S. P., Sibly, R.M., and Peakall, D.B. 2001. *Principles of Ecotoxicology*, Taylor and Francis, Inc.

Wong, C.S., P.D. Capel, and L.H. Nowell. 2001. National-scale, field-based evaluations of the biota-sediment accumulation factor model. *Environmental Science and Technology*. 35: 1709-1715.

(This page left intentionally blank.)