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A field studies and modeling approach to develop organochlorine pesticide and PCB total maximum daily load calculations: Case study for Echo Park Lake, Los Angeles, CA

V.R. Vasquez [a,*], J. Curren [a], S.-L. Lau [b], M.K. Stenstrom [b], I.H. Suffet [a]

[a] Environmental Science and Engineering Program, University of California, Los Angeles, Los Angeles, CA 90095–1496, United States
[b] Department of Civil and Environmental Engineering, University of California, Los Angeles, Los Angeles, CA 90095–1496, United States

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ABSTRACT

Echo Park Lake is a small lake in Los Angeles, CA listed on the USA Clean Water Act Section 303(d) list of impaired water bodies for elevated levels of organochlorine pesticides (OCPs) and polychlorinated biphenyls (PCBs) in fish tissue. A lake water and sediment sampling program was completed to support the development of total maximum daily loads (TMDL) to address the lake impairment. The field data indicated quantifiable levels of OCPs and PCBs in the sediments, but lake water data were all below detection levels. The field sediment data obtained may explain the contaminant levels in fish tissue using appropriate sediment–water partitioning coefficients and bioaccumulation factors. A partition-equilibrium fugacity model of the whole lake system was used to interpret the field data and indicated that half of the total mass of the pollutants in the system are in the sediments and the other half is in soil; therefore, soil erosion could be a significant pollutant transport mode into the lake. Modeling also indicated that developing and quantifying the TMDL depends significantly on the analytical detection level for the pollutants in field samples and on the choice of octanol-water partitioning coefficient and bioaccumulation factors for the model.

1. Introduction

Echo Park Lake is a small lake in Los Angeles (LA), CA listed on the USA Clean Water Act (CWA) Section 303(d) list of impaired water bodies for elevated levels of organochlorine pesticides (OCPs) and polychlorinated biphenyls (PCBs) in fish tissue. These chemicals present human health risks from consumption of contaminated fish (Longnecker et al., 1997) and are also directly toxic to fish and other aquatic organisms (Mayer et al., 1977; Roche et al., 2000). Lund et al. (1994) had previously reported fish tissue data from Echo Park Lake showing elevated levels of OCPs and PCBs. The California State Water Resources Control Board (SWRCB) (2009) recently reported fish tissue data collected from Echo Park Lake that indicated total PCB tissue concentrations up to 100 ug/kg and total DDT tissue concentrations up to 20 ug/kg. Regulatory agencies responsible for enforcing the USA CWA are obligated to develop and adopt total maximum daily loads (TMDLs) to address impaired water bodies. The TMDL sets the upper limit for pollutant mass loading from all sources to an impaired water body.

PCBs and OCPs are among the persistent, bioaccumulative toxic chemicals targeted by the USEPA for human exposure reduction (USEPA, 2009). Although PCBs were banned in the US in 1977 and DDT in 1972 (USEPA, 2009), they still persist in the environment. DDT has a reported half-life of 20 years to degrade to DDD and eventually to DDE (Wolfe et al., 1977). PCB and DDT are hydrophobic and sorb well to soil particles, and consequently, the mechanism of transport is via stormwater suspended solids (Connell et al., 1998; Parker et al., 2000). Thus, stormwater runoff rarely contains soluble concentrations of PCBs and DDT (Curren et al., 2011).

Echo Park Lake (Fig. 1) is a lake behind a constructed dam located in a small park, 2 miles NW of downtown LA, within a 160 hectare urbanized watershed. The lake is 450 m in length and 120 m at its widest. The lake is used for non-contact recreation such as paddle boating and fishing and is periodically stocked with fish. The lake also provides an aquatic habitat that supports a freshwater ecosystem. Due to a modified hydrology in the lake watershed, drainage into Echo Park Lake can only occur during significant rainfall, and the City of LA augments the lake with potable water. Outflow from the lake is controlled, and evaporation is a major mechanism of water flux. Thus, the lake is an effective trap for suspended sediment transported within urban runoff.

* Corresponding author. Tel.: +1 916 464 4623.
E-mail addresses: vrvasquez@ucla.edu (V.R. Vasquez), janecurren@yahoo.com (J. Curren), simlin@ucla.edu (S.-L. Lau), stenstro@seas.ucla.edu (M.K. Stenstrom), msuffet@ucla.edu (I.H. Suffet).

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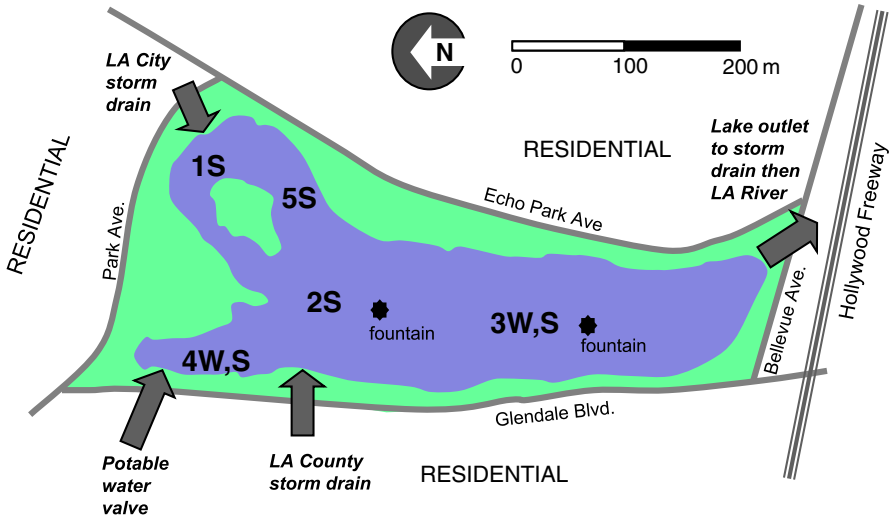

Fig. 1. Echo Park Lake showing sites 1–5 for water (W) and sediment (S) sampling.

The present study contributes knowledge regarding the fate and transport of OCPs and PCBs in small urban freshwater systems. The primary objectives of the present study are to assess the relationship between field data and the observed lake impairment and to demonstrate the use of field data within an environmental model to facilitate TMDL development. A field sampling program was conducted with the goal of determining for the first time the concentrations of OCPs and PCBs in Echo Park Lake in lake water, total suspended solids (TSS), sediment, pore water and pore water suspended solids to support the development of TMDLs by the LA Regional Water Quality Control Board. The field water and sediment data for total PCB and total DDT were then further analyzed using a modified Level I Fugacity Model (a partition-equilibrium model for the fate and transport of chemicals in environmental compartments) (Mackay and Paterson 1981, 1982). Fugacity models have been used previously to analyze sources of contaminants based on sediment data (Connell et al., 1998). A field study and modeling approach to support OCP and PCB TMDL calculations provides a way to evaluate any apparent relationship between fish tissue and sediment data as well as provides a first approximation of the sources and sinks of PCB and OCP for a lake system such as Echo Park Lake.

2. Materials and methods

2.1. Sampling

Water and sediment samples representative of different lake morphology and hydrology were collected from near stormwater drainages as well as the middle of Echo Park Lake, as shown in Fig. 1. Dry-weather water samples were collected from two sites and dry-weather sediment samples were collected from five sites in June 2008; a second round of dry-weather sediment samples were taken at two sites in December 2008. Water samples were collected and stored directly using water sample containers (i.e., acid washed, 1-gallon or 2.5-liter glass bottles with Teflon-lined lids). Sediment samples from sites deeper than 2 feet of water were collected using an Eckman Dredge. Sediment from shallower water sites were collected using clean scoops. Approximately the top 1 inch (2.54 cm) of sediment was collected. All water and sediment samples were stored in coolers, transported to UCLA and stored in a cold room at 4 °C until analyzed. As part of quality assurance and control (QA/QC) procedures, duplicate field samples, field water blanks, and Eckman Dredge rinsate water were collected.

2.2. Analytical procedures

Water samples were filtered to separate total suspended solids (TSS) from the aqueous phase using pre-weighed 142-mm, 0.7 um pure glass (no binder) TCLP filters (Whatman Inc., UK) and a Hazardous Waste Pressure Filter System (Millipore, Billerica, MA). The filtered water phase was extracted using solid phase extraction C-18 cartridges (Honeywell Burdick & Jackson, Morris Township, NJ) based on EPA Method 3535A. Filters containing the TSS were dried in 250 mL glass jars containing calcium chloride over 24 h, based on the method of Capangpangan and Suffet (1996), and then refrigerated at 4 °C until extracted. Filters were reweighed after drying to determine the amount of TSS collected. Microwave extraction of PCBs and OCPs from the TSS samples was done by EPA Method 3546.

Sediment samples were filtered using the same filtering procedures as for TSS to remove as much water as possible from the sediment. Approximately 5 g of the filtered sediment sample were then extracted using Soxhlet extraction in accordance with EPA Method 3540 C. When significant matrix interferences were observed by analysis, the Soxhlet extract was cleaned up in accordance with EPA Method 3630C. To remove excess water, some samples and extracts required additional drying steps such as pre-drying sediment samples prior to Soxhlet extraction in the same manner as drying of TSS samples or drying extracts further with additional sodium sulfate. Approximately 120 mL of sediment pore water were collected by decanting sediment samples and from sediment filtration, and TSS from pore water was collected in the same manner as lake water TSS. Extraction of pore water and pore water TSS were as described above for water and TSS samples.

Quantification of PCBs and OCPs in the extracts from lake water, TSS, sediments, pore water, and pore water TSS was conducted using gas chromatography (GC) with an electron capture detector. EPA Method 8081B was used to quantify for 19 OCPs; total DDT is the sum of results for 4,4′-DDE, 4,4′-DDD, 4,4′-DDT. EPA Method 8082A was used to quantify for 18 PCB congeners; total PCB is the sum these 18 compounds. Recovery studies demonstrated that the water, TSS and sediment analytical procedures used met USEPA QA/QC requirements. Blank solvent duplicates via the Soxhlet apparatus indicated the presence of the high molecular weight PCBs BZ180, BZ170 and BZ206

which could affect the sediment sample results from Site 1. Therefore, the solvent blank BZ180 concentration was subtracted from the BZ180 sediment sample result for Site 1.

Method detection limits (MDL) and Limits of Quantitation (LOQ) for field sample analyses were determined based on the lowest concentration of PCBs and OCPs that can be quantified in a volume of solvent extract injected into the GC, i.e., the instrument limit of quantitation (ILQ). The LOQ was calculated from the ILQ, and the MDL was set at 1/10th of the LOQ in lieu of a signal-to-noise determination. The MDLs and LOQs for field samples are given in terms of mass of OCP or PCB per unit volume of water sample or per unit mass of sediment sample. Field sample concentrations from the GC analysis results are reported as a numerical value if above the LOQ, as DNQ (Detected, Not Quantifiable) if between the MDL and LOQ, and ND (Not Detected) if below the MDL.

2.3. Partition-equilibrium modeling

The field water and sediment data were analyzed using the Level I Fugacity Model developed by Mackay and Paterson (1981, 1982), which was appropriate for the present study to provide a relatively simple model for initial TMDL development. The fugacity model considers the distribution of a compound in an environmental system consisting of six environmental compartments: air, water, soil (i.e., the land surface of the watershed), bottom sediment, suspended sediments, and biota. An Excel spreadsheet version of the Level I Fugacity Model was used for this study with the ability to specify each model parameter for a specific location. Physical parameters for DDT (MW 354.5) and Aroclor 1254 (MW 327.7) were used in the model (Mackay et al., 2006). Because the fugacity model used did not include environmental compartments for pore water and pore water TSS, and because the field data for these were limited, these were not considered in the modeling.

The parameter inputs for the fugacity model include chemical and physical properties of the chemical compound of interest (molecular weight, water solubility, vapor pressure, octanol-water partitioning coefficient), environmental compartment volumes, and soil and sediment density and organic fraction, as summarized in Table 1. For this study, Echo Park Lake was modeled as an elliptical lake with the same surface area and maximum and average depths as Echo Park Lake to estimate the water compartment volume for input into the fugacity model. Bottom sediment volume was estimated as the area of the lake multiplied by 2.54 cm, to correspond with the field sampling depth. Other environmental compartment volumes were estimated relative to the lake volume estimate based on previous modeling experience. The air compartment and soil compartment volumes used in the model were 500 times and 22 times the lake volume, respectively. Alternatively, the soil compartment volume used in the model can also be approximately related to the total land area, impervious area percentage, hydrology, and erodible soil depth of the Echo Park Lake watershed. Henry's Law constants, soil and sediment partitioning coefficients, bioaccumulation factors and other parameters may also be model inputs or may be estimated by the fugacity model from the chemical compound properties and other model inputs. The fugacity model is run with the total environmental system mass loading of a chemical compound as the input with the resulting mass loading of the chemical compound in the various environmental compartments as the output.

In this study, the fugacity model was used as a tool to estimate the total environmental system mass loading for total DDT and total PCB based on the bottom sediment concentration observed from the June 2008 field sampling. Because the fugacity model gives only a single result for each environmental compartment, it was necessary to average the field sample results from the five sediment sampling sites to obtain a single value that can be compared to the fugacity model output. The average sediment concentration at each of the five sediment sampling sites was calculated with a numeric value assigned to field sample results that were ND, based on the chemical analysis MDL as discussed in the Results and Discussion section. The GC analysis result was substituted for the single DNQ result in the June 2008 sediment data set. The five sampling site averages were then area-averaged by dividing the lake area into Thiessen polygons defined by a triangulated irregular network where the sampling site locations were the network nodes (Brennan et al., 2007). After calculating the area-averaged field sediment concentration, the value of the fugacity model input for total environmental system mass loading was varied until the model output for the bottom sediment concentration equaled the calculated area-averaged field sediment concentration.

Table 1
Summary of level I fugacity model inputs.

Parameter	Units	Value
a) Environmental compartment physical properties, all cases		
Air compartment volume	m^3	3.6 E+07
Water compartment volume	m^3	7.2 E+04
Soil compartment volume	m^3	3.2 E+03
Bottom sediment compartment volume	m^3	1542
Suspended sediment compartment volume	m^3	1.3
Biota compartment volume	m^3	0.072
Soil density	g/cm^3	1.5
Bottom sediment density	g/cm^3	1.5
Suspended sediment density	g/cm^3	1.5
Biota density	g/cm^3	1
b) Environmental compartment physical properties, base case		
f_{oc}	%	4

Parameter	Units	DDT	PCB (Aroclor 1254)
c) Pollutant chemical properties, all cases			
Molecular weight	g/mole	354.5	327.7
Water solubility	g/m^3	0.0017	0.0575
Vapor pressure	Pa	2.53 E−05	3.95 E−03
d) Pollutant chemical properties, base case			
log K_{ow}	–	6.914	6.45
BAF	–	5.591	5.131
Bottom sediment concentration	ug/kg	4.7	35

3. Results and discussion

3.1. Field samples

The results of chemical analysis for OCPs and PCBs in field samples are summarized in Table 2. From the June 2008 sampling, there were

Table 2
Summary of sample detection and quantification.

		June 2008 samples		December 2008 samples	
Sample	Result[a]	# OCP results	# PCB results	# OCP results	# PCB results
Lake water	ND	38	34	–	–
	DNQ	0	2	–	–
	>LOQ	0	0	–	–
Lake water TSS	ND	38	34	–	–
	DNQ	0	1	–	–
	>LOQ	0	1	–	–
Sediment	ND	89	81	34	33
	DNQ	0	1	1	2
	>LOQ	6	8	3	1
Pore water	ND	19	18	–	–
	DNQ	0	0	–	–
	>LOQ	0	0	–	–
Pore water TSS	ND	19	17	–	–
	DNQ	0	1	–	–
	>LOQ	0	0	–	–

[a] ND, not detected; DNQ, detected but not quantifiable; >LOQ, quantifiable result greater than limit of quantitation.

two lake water samples, two lake water TSS samples, five sediment samples, two pore water samples, and one pore water TSS sample analyzed. In addition, there were two sediment samples from the December 2008 sampling. Table 2 indicates that most of the various samples had non-detectable (ND) concentrations for OCPs and PCBs; however, there were detectable but not quantifiable (DNQ) levels of some PCBs in lake water, lake water TSS, sediment, and pore water TSS. More significantly, Table 2 shows that there was one quantifiable result for a PCB in lake water TSS and several quantifiable results for both PCB and OCP in sediments. This suggests that these chemical pollutants may be entering Echo Park Lake with the suspended solids and stored in the bottom sediments.

Table 2 indicates that there were no quantifiable results for either OCPs or PCBs in lake water. However, because the MDLs for the water sample OCP and PCB analyses were an order of magnitude higher than the allowable water quality concentrations for freshwater water bodies promulgated by USEPA in the California Toxics Rule (CTR) (Code of Federal Regulations, Title 40, Section 131.38) to avoid impairment, the ND and DNQ results for OCPs or PCBs in lake water may be significant. MDLs for both OCPs and PCBs were at least 2.4 ng/L whereas, for example, CTR water quality criteria is 0.6 ng/L for 4,4′-DDT and 0.17 ng/L for total PCBs. Therefore, the lake water analysis results indicate, but do not conclusively show, that OCPs and PCBs are not present in the lake waters at levels that could cause impairment.

Table 2 indicates that there were several DNQ and quantifiable results for OCPs and PCBs from both the June 2008 and December 2008 sediment samples. The detailed analytical results included one quantifiable result for gamma-chlordane of 8 ng/g dry sediment, which is above the Threshold Effect Level (TEL) sediment contamination level of 4.5 ng/g dry sediment (NOAA, 2008) for chlordane which could indicate impairment. The detailed results also included 3 quantifiable results for total PCBs at 3 of the 5 sediment sampling sites (120, 48 and 150 ng/g dry sediment), which exceed the TEL sediment contamination level of 34.1 ng/g dry sediment (NOAA, 2008). There were three sediment DNQ results for PCBs, and these do not indicate impairment because the TEL sediment contamination levels are above the sediment LOQs. However, there is one sediment DNQ result for 4,4′-DDT, and this result may suggest impairment because the TEL is below both the MDL and LOQ. Out of a total of 185 sediment sample analyses, 170 results were ND. For several OCPs (i.e., gamma-BHC, heptachlor epoxide, 4,4′-DDE and 4,4′-DDT), the ND results may still indicate possible impairment because the TELs were below the MDLs (NOAA, 2008).

Based on the quantifiable results from sediment samples and available sediment-water partitioning ratios, concentrations of contaminants in the aqueous phase might be significantly lower than can be detected using even the more sensitive analytical methods for PCBs used in this study. For example, total PCBs at Site 4 is 150 ng/g dry sediment; therefore, total PCB concentration in the water would be expected to be 0.15 ng/L using a sediment partition coefficient of 10^6 (Baker et al., 1986). This expected total PCB water concentration is of the same order of magnitude as the CTR value of 0.17 ng/L for water quality impairment but is lower than the MDL of 2 ng/L for PCBs. However, because sediment partition coefficients have been reported over a range (Mackay et al., 2006), predicted water concentrations would depend on the choice of partition coefficient.

The fish tissue data reported by the SWRCB (2009) may be explained by the DNQ results for lake water concentrations using appropriate bioconcentration factors (BCF) or bioaccumulation factors (BAF). Fish and other aquatic organisms are exposed to chemical pollutants through non-dietary and dietary processes. The BCF is a measure of absorption of pollutants through non-dietary routes such as respiration and dermal contact, and levels of exposure depend on the concentration of a pollutant in the water phase and the lipid content of the organism. The BAF is a measure of absorption of pollutants through both non-dietary routes and dietary processes such as ingestion of contaminated sediments and exposure through a contaminated food chain. The BCF and BAF are both calculated as the ratio of the concentrations in biological tissue and in ambient water with units of L/kg, and they have often been incorrectly used interchangeably, especially in the older literature. (Arnot and Gobas, 2006) For many chemical compounds, there is a range of BCF/BAF values reported in the literature; for example, the range of log BCFs for 4,4′-DDT has been reported to be 1.04 to 5.00 (Arnot and Gobas, 2006). Using a BAF of 126,000 for DDT (Mackay et al., 2006) and substituting the MDL of 2 ng/L as the minimum possible value for DNQ results, fish tissue concentrations of 250 ppb (ug/kg tissue) are predicted, which is 10 to 25 times the magnitude of observed fish tissue data for total DDT.

3.2. Partition-equilibrium model results

The range of possible BCF/BAF, the difference between BCF and BAF, and expected uncertainties associated with the partitioning coefficient of organic pollutants from solid surfaces to the aqueous phase have regulatory implications for the development of TMDLs and the restoration of impaired water bodies such as Echo Park Lake. Depending on the partitioning ratio and BCF/BAF used, a causal relationship may be established between observed field concentrations of pollutants in sediment and lake water and in fish tissue. The fugacity model was used to investigate these interdependent relationships as well as to obtain approximations of the sources and sinks of PCB and OCP for lakes such as Echo Park Lake.

Table 3 shows the fugacity model output for the predicted concentrations and the percent mass distribution of total DDT and total PCB in the six compartments for the base case where sediment ND values were set equal to the MDL, and octanol–water partitioning coefficient (K_{ow}), sediment and TSS organic carbon fraction (f_{oc}) and BAF were as given in the Table 3 footnote. These fugacity model outputs indicate that approximately half of the total system loading is stored in the soil compartment and the other half is stored in the bottom sediments. Consequently, the soil compartment may be a significant potential source of the pollutants to the lake through erosion. It is noteworthy that the model outputs in the base case for DDT and PCB concentrations in the biota compartment are 14 ug/kg and 102 ug/kg, respectively, which are closely similar to the observed fish tissue concentrations as discussed in the Introduction section.

The presence of ND values in water quality data sets have been shown to significantly affect calculated average concentrations and

Table 3
Fugacity model results for base case.[a]

Compartment	Conc. (mol / m^3)	Conc. (ppb, ug/kg)	% of total
			Mass loading
1) DDT			
Air	2.17E−13	–	0.01
Water	1.02E−10	3.6E−05	0.01
Soil	1.03E−05	2	51.17
Bottom sediment	2.05E−05	5	48.77
Suspended solids	2.05E−05	5	0.04
Biota	3.96E−05	14	0.004
Total			100.00
2) PCB (Aroclor 1254)			
Air	2.08E−11	–	0.15
Water	2.29E−09	7.5E−04	0.03
Soil	7.95E−05	17	51.09
Bottom sediment	1.59E−04	35	48.69
Suspended solids	1.59E−04	35	0.04
Biota	3.10E−04	102	0.004
Total			100.00

[a] Base case settings and parameters: ND = MDL; f_{oc} = 4%; log K_{ow} = 6.914 and log BAF = 5.591 for DDT; log K_{ow} = 6.45 and log BAF = 5.131 for PCB.

Table 4
Total system load as function of value assigned to ND results.

Value assigned to ND results	DDT		PCB	
	Sediment concentration [a] (ng/ g)	Total system load model output (kg)	Sediment concentration [a] (ng/ g)	Total system load model output (kg)
MDL	4.7	0.022	35	0.166
0.5 MDL	2.3	0.011	29	0.138
0.1 MDL	0.5	0.002	23	0.109

[a] Area-averaged concentration.

mass loadings (Kayhanian et al., 2002). In the present study, the majority of lake water and sediment quality results were ND, and as discussed previously, do not conclusively indicate that these results are not causing impairment. Therefore, to understand how the total system mass loading and other model outputs based on sediment data would change if the chemical analysis MDL were lower, the area-averaged sediment concentration based on the field sediment sample results was recalculated with ND results assigned values equal to the MDL, 0.5 MDL, or 0.1 MDL, and the fugacity model was rerun with those recalculated values. Table 4 gives the recalculated area-averaged sediment concentrations and corresponding total system mass loading. Table 4 shows that if the MDL were one-tenth of the actual MDL, the total system load for DDT and PCBs would be 90% and 34% lower, respectively. This indicates that the TMDL calculated from field studies would be greatly influenced by the sensitivity of chemical analysis procedures used.

Fugacity model outputs were determined for the case where ND values were assigned the value of the MDL while separately varying the K_{ow}, from which K_{oc} is estimated in the fugacity model; the sediment and TSS f_{oc}, which partly determines solid phase partitioning; and BAF. Table 5 summarizes the ratio of model outputs for total DDT and PCB concentrations in the different environmental compartments relative to the base case. The conclusions that can be made from the ratios of model predictions given in Table 5 are as follows: 1) concentrations in water and biota decrease by almost 3 orders of magnitude if K_{ow} increase by 3 orders of magnitude; 2) water and biota concentrations decrease by approximately 30% while sediment concentrations increase by about 34% as f_{oc} in sediments and TSS increase from 2% to 4%; 3) concentrations in biota increase by more than three orders of magnitude, but water and sediment concentrations are not affected, if BAF increase by three orders of magnitude. These results indicate that the choice for K_{ow} and BAF significantly affects model results. These results highlight the difficulties that regulatory agencies would face in developing TMDLs due to the uncertainties and wide range of values reported for K_{ow} and BAF.

Table 5
Ratios of fugacity model outputs.

1) Varying octanol–water partitioning coefficient, K_{ow}

DDT				PCB			
Log K_{ow}	Water	Sediment	Biota	Log K_{ow}	Water	Sediment	Biota
3.98	687.948	0.808	687.566	4.08	163.518	0.698	163.518
5.55	22.769	0.994	22.757	6.45	1.000	1.000	1.000
6.914	1.000	1.000	1.000	7.17	0.191	1.002	0.191

2) Varying fraction organic carbon, f_{oc}

DDT				PCB			
f_{oc} (%)	Water	Sediment	Biota	f_{oc} (%)	Water	Sediment	Biota
2	1.311	0.661	1.310	2	1.322	0.661	1.322
3	1.129	0.854	1.128	3	1.139	0.854	1.139
4	1.000	1.000	1.000	4	1.000	1.000	1.000

3) Varying Bioaccumulation Factor, BAF

DDT				PCB			
Log BAF	Water	Sediment	Biota	Log BAF	Water	Sediment	Biota
3.08	0.991	1.000	0.003	5.131	1.001	1.001	0.042
5.591	1.000	1.000	1.000	6.51	1.000	1.000	1.000
6.67	0.990	1.000	11.873	7.21	0.996	0.996	4.991

4. Conclusion

The objective of this study was to determine and interpret the concentrations of OCPs and PCBs in water, suspended solids, bottom sediment, pore water, and pore water suspended solids in Echo Park Lake to support the development of TMDLs to address elevated levels of these compounds detected in fish tissue. The field data indicated quantifiable levels of OCPs and PCBS in the lake sediments that were above sediment levels that could indicate impairment of the lake. Lake water data were all ND or DNQ; however, because much lower analytical detection levels are needed, these may still indicate water column concentrations that could result in impairment. The field sediment data obtained from this study may explain the observed contaminant levels in fish tissue if appropriate sediment-water phase partitioning coefficients and BCF/BAF are considered. A Fugacity Level 1 model of the whole lake system was used to interpret the field data and indicated that half of the total mass of the pollutants in the system are in the bottom sediments and the other half is in soil; therefore, erosion of soil into the lake could be a significant source of the pollutants for the lake. Additional modeling indicated that developing and quantifying the TMDL may depend significantly on the ability of analytical procedures to detect the pollutants in field samples and on the choice of K_{ow} and BAF.

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