# Water Resources Center
# Annual Technical Report
# FY 2001

## Introduction

The University of California Center for Water Resources is a multicampus research unit and special program within the UC Division of Agriculture and Natural Resources. The major function is to support research and extension activities which will contribute to the efficient management of water resources within the state. Meeting the needs of the urban, agricultural and wildlife sections from both water quality and quantity considerations is a goal of the Center. The Center has linkages to faculty of all nine campuses in the UC system and to extension personnel in each 58 counties. The Center can be reached by email at cwres@ucr.edu and can be viewed on the web at http://waterresources.ucr.edu

The Water Resources Center funded 9 new and continued 18 projects for a total of $693,891 with nearly every UC campus participating. 5 of these projects were selected to participate in the NIWR-USGS program. The research categories -- Hydrology, Climatology & Hydraulics; Aquatic Ecosystems; Water Quality; Water Development and Management Alternatives; Water Law, Institutions and Policy were all represented. Following is a list of newly funded projects (July 1, 2002).

## Research Program

# Dynamic Chemical Loads as a Function of Land-Use Changes in a Watershed

## Basic Information

| | |
|---|---|
| **Title:** | Dynamic Chemical Loads as a Function of Land-Use Changes in a Watershed |
| **Project Number:** | 2000CA6G |
| **Start Date:** | 9/1/2000 |
| **End Date:** | 8/31/2003 |
| **Funding Source:** | 104G |
| **Congressional District:** | 43 |
| **Research Category:** | Ground-water Flow and Transport |
| **Focus Category:** | Surface Water, Solute Transport, Management and Planning |
| **Descriptors:** | |
| **Principal Investigators:** | Arturo A Keller |

## Publication


1. Robinson, Timothy H., Al Leydecker, John M. Melack and Arturo A. Keller. 2002. Nutrient Concentrations in Southern California Streams related to landuse. Coastal Water Resources, AWRA 2002 Spring Specialty Conference Proceedings, Lesnick, John R. (Editor). American Water Resources Association, Middleburg, Virginia, TPS-02-1, pp 339-343.

## Progress Report as of 12/31/02 for:

## Agreement No. 00HQGR0089 from USDI

## USGS competitive grants program

# Dynamic Chemical Loads as a Function of Land-Use Changes in a Watershed

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## Summary

This report summarizes our progress to date on this project. The project involved the implementation of two watershed modeling frameworks for the Santa Ana River Basin, for a number of pollutants, to study the effect of land-use changes on the chemical loading that moves from the land surface to the rivers and creeks in the watershed. The project focused on the Lower Santa Ana River Basin and the Newport Bay area, given the extensive data sets available from the USGS, the Regional Water Quality Control Board and other local agencies. The pollutants of interest were nutrients (N and P) and two common organophosphate pesticides (Diazinon and Chlorpyrifos). All of these pollutants have distinctive applications in the land-uses of interest (residential and agriculture), which have been changing rapidly over the last few decades.

The watershed modeling frameworks used were BASINS (USEPA/TetraTech) and WARMF (EPRI/Systech), based on their ability to model land application and mechanistic processes to transport and transform these pollutants through the watershed. The models were implemented using land-use data available through the BASINS website as well as historical information, with meteorology from NOAA, and with hydrology and water quality information available through the USGS NWIS and NAWQA programs. The models were calibrated using the historical data up to 2000 and verified with data from 2001-2. In general, hydrology was easier to calibrate given the more complete data sets, while there is significantly larger uncertainty in the water quality calibration due to the sparseness of the data. However, for the purpose of this analysis, which is mostly to understand the relative

magnitude of the various sources and their temporal and spatial variation, it was deemed that the model calibration was adequate.

To date we have calibrated the BASINS and WARMF models for the Newport Bay area, in particular for pesticides loading. We are in the process of implementing the models for the Lower Santa Ana River basin, which is a much more extensive area. Results to date indicate a very good match between simulated and observed organophosphate concentrations. The loading varies temporally (we estimate different monthly application rates) and spatially (for the various land uses). However, the output signal to the receiving surface water bodies is strongly dominated by the periodic flushing of storm events, as observed by the USGS NAWQA study.

We have produced one Master's thesis to date and have directly trained 9 Master's level students and one doctoral student. The doctoral student continues to implement and use the models under supervision of Dr. Keller. Keller has also used the large amount of information generated by this project to develop a graduate level course (ESM 595D Watershed Quality Modeling and Management), which trains about 10 graduate students a year. We have also had fruitful discussions with John Izbicki from the USGS' NAWQA study in the Santa Ana River Basin. Part of the work was presented at the California OCEANS conference. We are currently in the process of preparing presentations for the American Water Resources Association's 2003 International Water Congress on Watershed Management for Water Supply Systems, to be held in New York. Manuscripts for peer-reviewed literature are in preparation at this time. We expect completion of final modeling and manuscript preparation in the summer and fall of 2003.

# 1. Introduction

The development of Total Maximum Daily Loads (TMDLs) is a pressing issue for the State Water Quality Control Board, Regional Water Quality Control Boards (RWQCB), local agencies, stakeholders in the watershed, and of course the U.S. Environmental Protection Agency (USEPA). The magnitude of the task is daunting, given the large number of TMDLs that have to be developed, nationally and even just statewide. As recent experience has shown, developing TMDLs is a complex matter, not only from the scientific perspective, but also from the more complex socioeconomic perspective. To increase the complexity, TMDLs have to take into account projected land-use changes.

This project focused on the region around the Santa Ana River Basin in Southern California, based on (1) high quality data on land-use, hydrology and water quality; (2) major water quality issues, since this watershed has been extensively developed; and (3) the likelihood that watershed will continue to experience major land-use changes in the next decades. Given the complexity of the system, sophisticated modeling tools were needed to address land-use changes around the Santa Ana River Basin. Since the SAR basin is quite extensive, the project focused on two important regions, namely the lower SAR, below Prado Dam, and Newport Bay, next to the lower SAR.

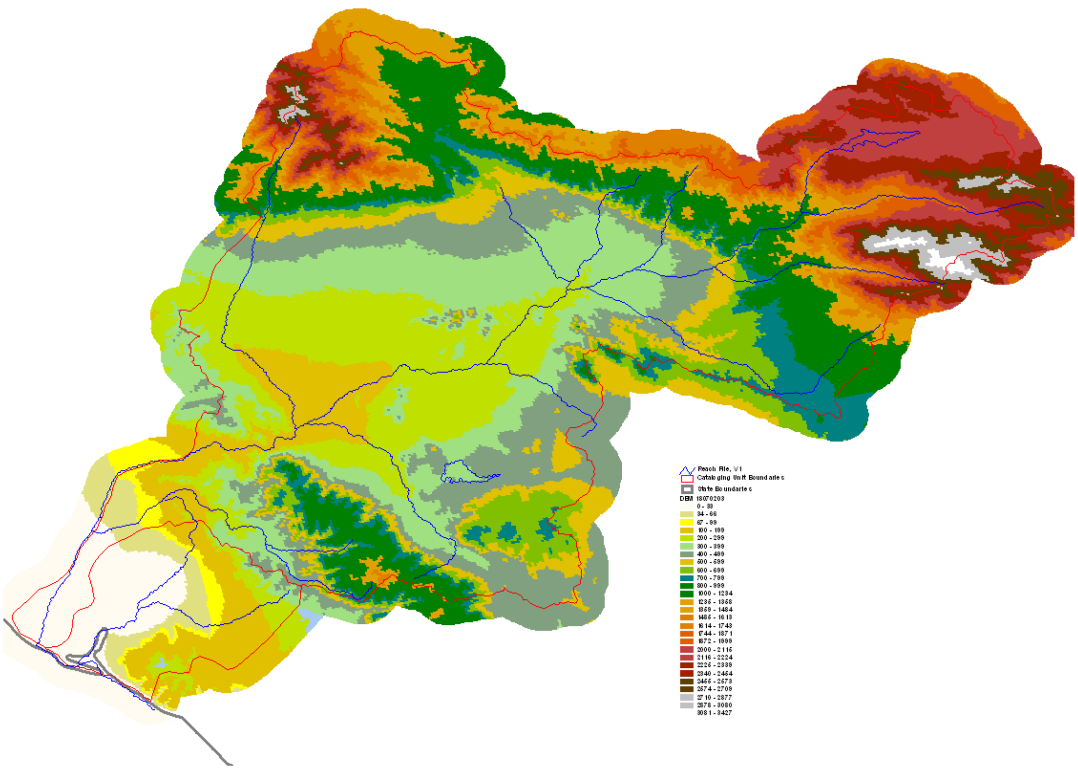
The main objective was to address the issue of land-use changes directly, using two watershed models, namely a watershed-scale modeling framework that has been developed by USEPA and TetraTech (BASINS), as well as a similar framework developed by the Electric Power Research Institute (EPRI) and Systech Engineering, Inc., denominated the Watershed Analysis Risk Management Framework (WARMF). Although very similar in their conceptual models, these two frameworks differ in their implementation, degree of user friendliness and the tools available to the users for developing TMDLs. Both of these tools allow the implementation of changing land-use scenarios over time, and can be adapted to consider BMPs.

The specific objectives were: (1) implement the BASINS and WARMF models for the Santa Ana River watershed; (2) develop temporally-variable response functions for the various catchments in the watershed, correlating chemical load with a number of catchment characteristics as well as external driving forces; (3) investigate the effect of projected land-use changes on chemical loading and water quality; (4) determine the effect of BMPs on chemical loading and water quality; and (5) develop a methodology, for characterizing other watersheds where TMDLs will be developed, which systematically analyses the role of land-use and BMPs.

## 1.1 SAR Basin

Water quality in the Santa Ana River and tributaries has been compromised in recent years by the large number of diversions, as well as point and non-point source loading, resulting in long episodes of very low water flow, low dissolved oxygen, nutrients, volatile organic compounds (VOCs) and pesticides contribute to water quality problems in the Santa Ana River (NAWQA, 2000). The Santa Ana River Basin covers an area of about 7,000 km$^2$, covering parts of four counties in Southern California (Los Angeles, Orange, Riverside, and San Bernardino). The Santa Ana River runs west from the San Bernardino Mountains, at elevations of up to 3500 m, down to the Pacific Ocean (Figure 1) by Huntington Beach, crossing over 150 km of diverse land uses.

Figure 1. Digital Elevation Model (30-m USGS) of the Santa Ana River Basin and Newport Bay areas.

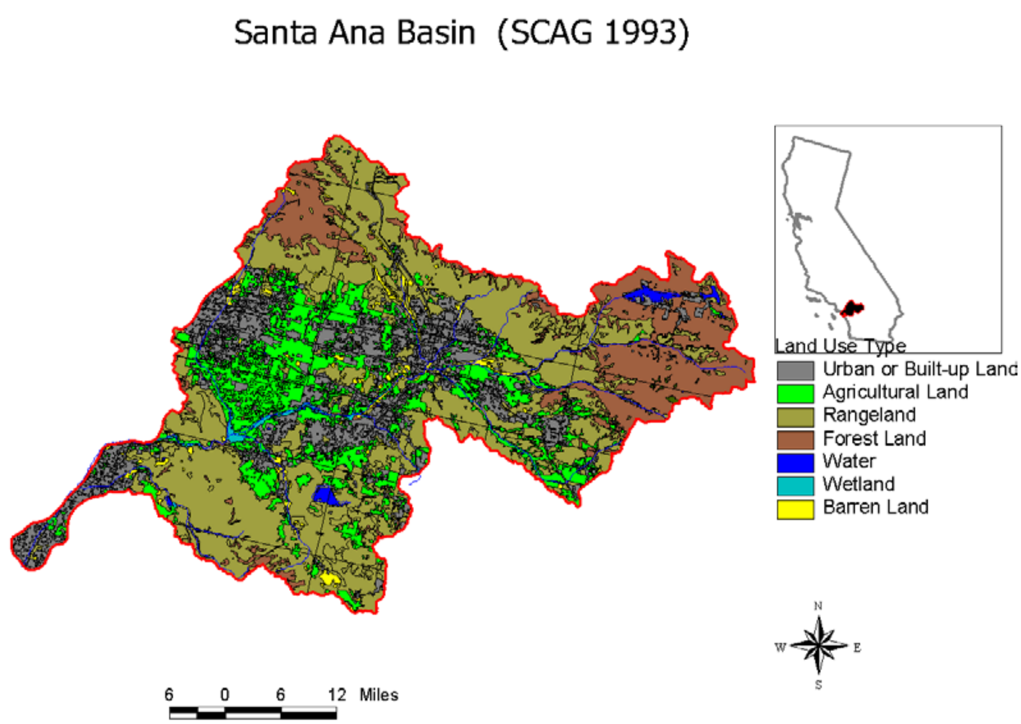



A significant fraction of the Santa Ana River basin is heavily developed, with about 21 percent of the land used for residential, commercial, or industrial activities (Figure 2). Not surprisingly, urbanization follows the major transportation corridors going east and south from Los Angeles. Agriculture represents about 17% of the land use, almost equally divided between orchards and croplands, with about 1% of the land devoted to confined feed lots that can be significant sources of pollutants. It has been estimated that these confined cattle lots hold more than 340,000 animals (SA RWQCB, 1996), mostly in facilities in Chino and Cucamonga Creeks above Prado Dam. Around 16% of the watershed is forestland, mostly in the headwaters of the San

Bernardino area. 43% of the land is rangeland, mostly shrubs and bushes. There are a number of small lakes and reservoirs that cover approximately 0.7% of the landscape. These reservoirs are heavily managed, controlling releases based on a number of criteria, including flood control and habitat management. Prado Dam, above the lower SAR, controls the release of water from wetlands covering more than 5000 acres. Barren land (e.g. beaches, exposed rock, strip mining, sandy areas) represents the balance, with about 2.3% of the total surface.

The lower SAR region is significantly more developed than the upper SAR, with over 26% of the land used for residential, commercial and industrial activities. In some sections, for example at the mouth of the SAR, up to 85% of the land is urbanized. This area has seen a number of land use changes over the last few decades, from the original natural habitats, followed by conversion to orchards and row crops early in the last century, followed by sprawling urbanization from the 1950's onward to its present state. Agriculture now represents only 6.1% of the lower SAR. Although about 64% of the land use is rangeland (60.5%), forests (2.8%), and wetlands (0.6%), these lands in the Santiago Creek region are less suitable for agriculture, and are in general not considered for immediate urbanization due to their geology and topography. The balance is mostly barren lands and water reservoirs.

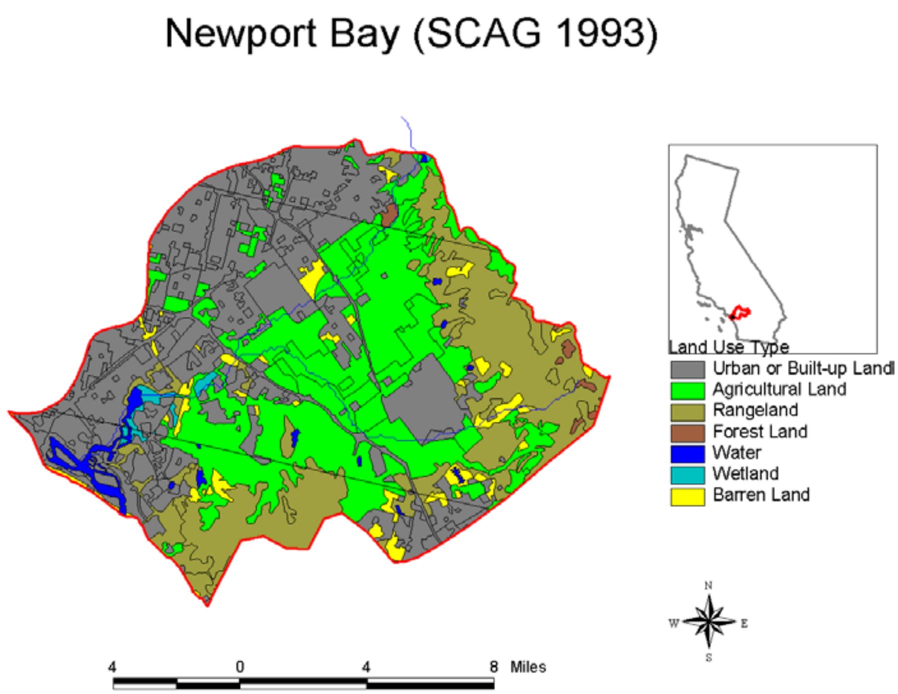
Figure 2. Land Use in the Santa Ana River Basin based on SCAG 1993 data.



The Newport Bay catchment is still undergoing significant land use changes, transitioning mostly from agriculture to residential, commercial and industrial land uses. As of 1993, 41% of the landscape was urban, with 20% residential, 11 % commercial and the balance a combination of industrial and transportation uses. 31% was still in agriculture, but with significant pressure to urbanize. Around 23% of the land is still used as rangeland or forests, generally in the higher altitudes (Figure 3). Only 2% of the landscape is occupied by wetlands and other small waterbodies, a marked change from its original state.

Given the relatively low precipitation (0.30 to 0.45 m/yr) and the short rainy season (late November to March, mostly concentrated in 2 months), the Santa Ana River has very low flows in the summer months under natural conditions. For water supply, the various water utilities depend on ground water (ca. 2/3$^{rd}$) and imported water from northern California and the Colorado River (ca. 1/4$^{th}$). In fact, on a typical year almost all the flow of the SAR, which is around 200,000 acre-feet [metric] is diverted by the Orange County Water District (OCWD) to the region below Imperial Highway where it infiltrates to the underlying aquifers through natural and artificial recharge ponds (OCWD, 1996). Base flow in the Santa Ana River is dominated by treated wastewater discharges, mostly tertiary treatment, through significant parts of the year (Burton et al., 1998). Storm and sanitary sewers are mostly separated in the communities within the SAR basin (Burton et al., 1998). Residual nitrate in the tertiary effluent is partially removed in a series of artificial wetlands upstream of Prado Dam (Izbicki et al., 2000).

Figure 3. Land Use in the Newport Bay area based on SCAG 1993 data.



There is concern that stormwater runoff (from suburban and urban areas) and discharge from the Santa Ana River are having a significant impact on surface and groundwater, as well as coastal ecosystems. Factors affecting water quality include large consumptive use, stormwater runoff from urban and suburban areas, non-point source pollution from agricultural activities, water high in dissolved solids that is introduced to the basin, recycling of water within the basin, and atmospheric inputs of nutrients, metals and Volatile Organic Compounds (VOCs) (NAWQA, 2000).

The rate of urbanization in the Santa Ana River Basin is likely to continue in the years ahead. The effect of these land-use changes, and the associated "management" practices, are going to play a pivotal role on water quantity and quality. Any attempt to develop TMDLs for this region without consideration for these rapid changes, and the effects of BMPs, is likely to result in an incorrect assessment of the situation.

## 1.2 Pollutants studied in this project

For this study, we focused on two sets of pollutants, with significant differences in biogeochemical characteristics and loading rates, namely nutrients (N as ammonia, nitrite and nitrate and P as phosphate, dissolved or total phosphate) and two organophosphate pesticides (Diazinon and Chlorpyrifos). The selection of these pollutants was based on (1) their wide availability and use; (2) reasonably good loading data for various land uses; and (3) potential for wide impact on human and ecological receptors. The nutrients can come from a variety of sources, including treated tertiary effluent, fertilizer application in agriculture and urban green areas, atmospheric deposition, septic tank systems and other minor sources.

The two pesticides we considered are among the most widely applied in this region, and although there is a partial phase-out under effect, this pertains mostly to small-scale residential uses, and not to other large uses as discussed in more detail in Section 3. The most relevant fate and transport characteristics of these two pesticides are presented in Table 1.

Both pesticides have significant water solubility, compared to the levels associated with toxicity, leading to significant mobility in runoff from irrigation or rainfall. Their vapor pressures are small, although Chlorpyrifos has a slightly greater vapor pressure and lower water solubility, leading to a noticeably greater Henry's Law Constant ($K_H$) than Chlorpyrifos. However, the $K_H$ of both compounds is rather low, indicating that they are not very volatile and tend to stay in soil or water rather than escape to the atmosphere. The two compounds tend to adhere to sediment and organic matter (U.S.EPA, 2000), based on their relatively high partitioning coefficients ($K_{ow}$ and $K_{oc}$). Transport in runoff either dissolved or adhered to sediments and organic matter is important for both pesticides.

Table 1. Physicochemical Properties of Diazinon and Chlorpyrifos at 25 °C

| | | Diazinon | Chlorpyrifos |
|---|---|---|---|
| Chemical Formula | | $C_{12}H_{21}N_2O_3PS$ | $C_9H_{11}C_{13}NO_3PS$ |
| Molecular Weight | g/mol | 304.36 | 350.57 |
| Density | $kg/m^3$ | 1,110 | 1,400 |
| Water Solubility | mg/L | 40 | 2 |
| Vapor Pressure | Pa | 0.0004 | 0.0027 |
| Henry's Constant, $K_H$ | atm $m^3$/mol | $1.09 \times 10^{-7}$ | $1.23 \times 10^{-5}$ |
| Octanol-water partitioning coefficient, $K_{ow}$ | - | $10^{3.81}$ | $10^{5.11}$ |
| Organic content-water partitioning coefficient, $K_{oc}$ | - | $10^{2.28}$ | $10^{3.73}$ |
| $t_{1/2}$ in air | d | 4.1 | N/A |
| $t_{1/2}$ in water | d | 43 | N/A |
| $t_{1/2}$ in soil | d | 50 | 42 (muck) |
| $t_{1/2}$ in biota | d | 32 | N/A |

N/A = not available Source: Watts (1998) and Lee (1998).

To some extent, these two Organophosphate pesticides (OPs) may evaporate from impervious surfaces or during spray applications. Atmospheric losses may occur, but studies show that most airborne OPs appear to redeposit locally (U.S.EPA, 2000), typically within the same watershed where they were applied. In this case, OP volatilization and deposition do not significantly alter their concentrations in urban creeks.

Diazinon and Chlorpyrifos may be adsorbed to the sediments of creek beds. These sediments may serve as a transport mechanism within a creek and may also be an important OP sink. Sediment leaching experiments performed by Alameda County found that Diazinon might also be re-suspended into the water column, making sediment a potential OP source. The process of leaching from sediments into the water column occurs more frequently during dry weather. This may be an important process in stagnant pools and ditches that have high concentrations of OPs in their sediment, or in creeks where water flows slowly over a long stretch of OP-laden sediment (URS, 1999).

Persistence of Diazinon and Chlorpyrifos at the application site and throughout the watershed depends on physical, chemical and biological factors, including temperature, humidity, light, soil and water pH, and microbial activity. OPs applied outdoors to soils, plants, and impervious surfaces generally reach biota mostly through surface water. Most of the OPs applied to soil and lawn surfaces break down in the soil before reaching storm drains and creeks. In soil, microbial degradation is the major route of OP decomposition. The degradation rate constants (kr) and half-lives (t1/2) for Diazinon and Chlorpyrifos are shown in Table 2.

Table 2. Degradation Constants for Diazinon and Chlorpyrifos

| Degradation Process | Diazinon $t_{1/2}$ (d) | Diazinon $k_r$ ($day^{-1}$) | Chlorpyrifos $t_{1/2}$ | Chlorpyrifos $k_r$ ($day^{-1}$) |
|---|---|---|---|---|
| Photolysis: | >150 | <0.0046 | >150 | <0.0046 |
| Hydrolysis: pH = 5 | 12 | 0.0578 | 73 | 0.0095 |
| pH = 7 | 138 | 0.0050 | 72 | 0.0096 |
| pH = 9 | 77 | 0.0090 | 16 | 0.0433 |
| Biodegradation.: aerobic | 37 | 0.0187 | 11-141 | 0.0630-0.0049 |
| anaerobic | 34 | 0.0204 | 37 | 0.0187 |

Source: USEPA, 2000

For Diazinon soil decomposition rates range from 2 to 4 weeks, though it may persist for up to six months or longer at low temperature, low moisture, high alkalinity, and under conditions where microbial degraders are absent (Sheipline, 1993). Because microbial degradation occurs much more slowly on impervious surfaces, Diazinon breakdown is considerably slower there. Diazinon will degrade in water, though this process is less important in surface water due to the relatively short residence time of storm water in urban creeks. When Diazinon does reach surface waters through storm water runoff, hydrolysis is the predominant degradation pathway. Hydrolysis is rapid under acidic conditions with a half-life of 12 days at pH 5. Under neutral and alkaline conditions, Diazinon hydrolyzes more slowly, with half-lives of 138 days at pH 7 and 77 days at pH 9. Diazoxon is the first degradate formed by oxidation and it rapidly oxidizes further to oxypyrimidine. Diazinon is stable to photolysis in water (USEPA, 2000).

Chlorpyrifos is moderately persistent in the environment. Chlorpyrifos generally has a half-life of less than 60 days in the field, degrading primarily by aerobic and anaerobic metabolism, and to a lesser degree through biodegradation. Photolysis, hydrolysis and volatilization also occur but are not believed to be major routes of dissipation (U.S.EPA, 2001). The majority of Chlorpyrifos partitions into soil, where it is quite persistent, especially when not exposed to light. Soil persistence of 60-120 days has been reported (Spectrum Laboratories, 2001). The major degradates of Chlorpyrifos are 3,5,6-trichloro-2-pyridinol (TCP), which is less toxic but more persistent and mobile than Chlorpyrifos, and 2-methoxy-3,5,6-trichloropyridine (EXTOXNET, 1996).

The generic biogeochemical properties of these two OPs were studied in the context of a watershed in which they may be released. To evaluate their steady state distribution in a watershed, the fugacity-based Environmental Equilibrium Partitioning Model (EEP) model version 1.01 (Canadian Environmental Modelling

Centre, 1997) was used to quantify the partitioning of these OPs within a hypothetical environment. The EEP model estimates a chemical's behavior based upon its physicochemical properties. The model we considered includes the following media: air, water, soil, sediment, biota, aerosols and suspended sediment. EEP simulates the major pathways of a substance released into the environment, including degradation, volatilization and sedimentation. Evaluative Level II was used, which, in addition to assuming that thermodynamic equilibrium is achieved, includes advection and reaction processes. Level II is a steady state model with a constant input rate, rather than single release of chemical. For the purposes of illustration, we considered a hypothetical loading of 10 kg/h into the watershed

The results of the EEP model are presented in Figures 4 and 5.According to EEP Level II model results, the majority of Diazinon partitions into the soil (98%), with much smaller fractions partitioning into the sediment (1.96%), atmosphere (0.01%), and to surface water (0.002%). The very low partitioning to surface waters reflects the relatively low volume of this compartment in these dry Southern California watersheds. The steady-state aqueous concentrations predicted for Diazinon by this model are in general agreement with typical values found in urban and suburban creeks, which are discussed in more detail later. Most of the applied Diazinon is lost via reaction in the soil (95%), followed by reaction in sediments (3%). The balance is lost via advection (mostly via air transport), and in reactions in suspended sediments, biota, air and water, which are not all shown in this diagram.

Chlorpyrifos partitions similarly to Diazinon, with more affinity for the gas phase and therefore a larger fraction in the atmosphere and greater air transport out of the watershed to surrounding areas. Still, 98% of the applied load remains in the soil and reactions in the soil account for about 88% of the loss. Given the lower water solubility of Chlorpyrifos, the aqueous concentrations are expected to be about an order of magnitude lower than for Diazinon, given equal application rates.

Diazinon and Chlorpyrifos act as pesticides by attacking the nervous system of insects. Diazinon acts as an inhibitor of acetylcholinesterase, an enzyme necessary for proper nervous system function, while Chlorpyrifos affects the central nervous system, the cardiovascular system and the respiratory system. The effects of Diazinon and Chlorpyrifos are not limited to insects; humans and animals are also affected by Diazinon and Chlorpyrifos toxicity. When taken in through dermal, oral or inhalation exposure, Diazinon and Chlorpyrifos can cause nervous system malfunction in humans and animals, leading to illness and possibly death (EXTOXNET, 1996).

Figure 4. Diazinon Partitioning in a watershed modeled using EEP Level II.

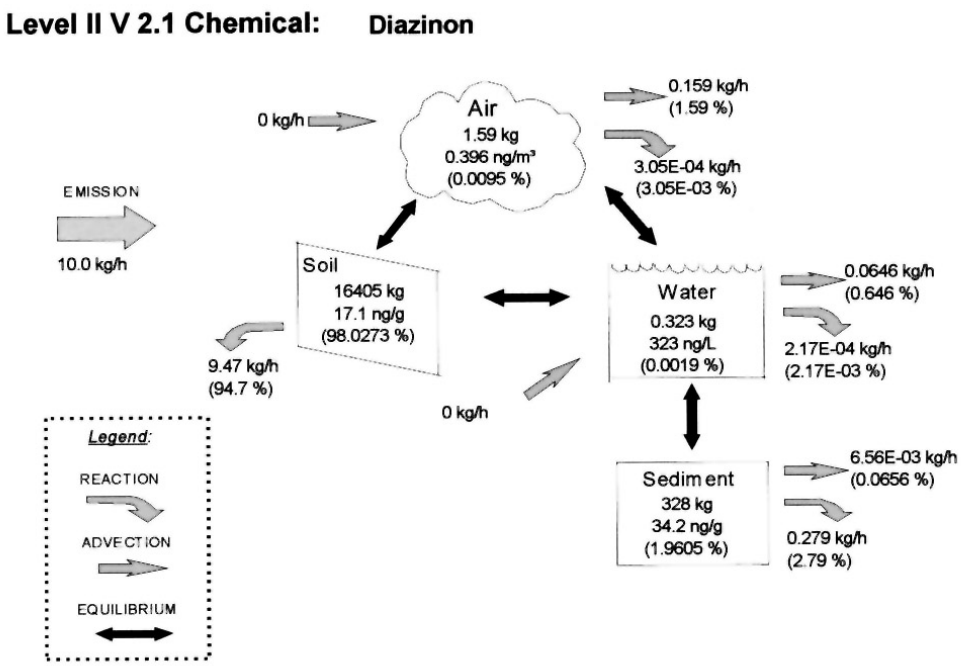
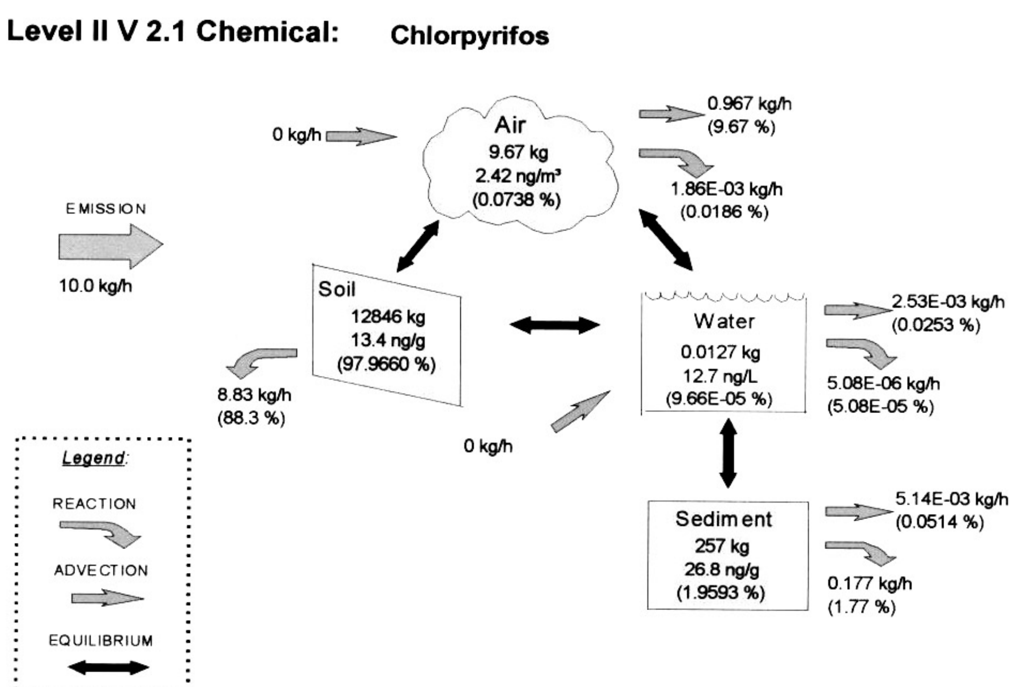



Figure 5. Chlorpyrifos Partitioning in a watershed modeled using EEP Level II.



Pollutants impact the natural environment through their effects on entire populations, but act by their effects on individual organisms (Moriarty, 1983). Thus, in order to understand the effects of Diazinon and Chlorpyrifos on the ecosystems in Newport Bay, it is necessary to look at the effects of the pesticides on individual organisms within those populations. Many studies have focused on the effects of Diazinon and Chlorpyrifos on individual species found in Newport Bay. Birds in the Bay have been an important focus of such studies. For example, studies have shown that the health and reproductive abilities of the snowy plover, an endangered species of bird found in the Bay, are adversely affected by the bioaccumulation of pesticides during their stay in nesting and wintering grounds in southern California. These effects are largely due to the fact that snowy plovers are primarily insectivorous, resulting in the ingestion of high levels of insecticides (Powell and Hothem, 1997).

Acute exposure to Diazinon accounts for the highest percentage (21%) of ecological incidents involving organophosphates (U.S.EPA, 2001). Birds are the most susceptible species to Diazinon poisoning; bird kills associated with Diazinon usage are reported year-round throughout the country (Ladaa et al, 1998). Broadcast application of Diazinon to turf poses one of the greatest pesticide risks to birds. Just one granule or seed treated with Diazinon is enough to kill a small bird. Diazinon had the highest number of reported bird kill incidents of any registered pesticide during 1994-1998. Birds of many species have been killed, including ducks, geese, hawks, songbirds, woodpeckers and others. Diazinon is also highly toxic to fish and aquatic invertebrates; around 11% of Diazinon related incidents involved aquatic species. Mammals are less sensitive than birds, although Diazinon can be highly toxic to mammals when taken in through dermal and inhalation routes (U.S.EPA, 2001).

Acute exposure to Chlorpyrifos also threatens species in the natural environment, such as fish, aquatic invertebrates and birds. For example, a study by the U.S. Fish and Wildlife Service found that a single aerial spray application to kill mosquito larvae resulted in the death of significant numbers of fishes and crustaceans. A study by Hurlbert et al. (1970) found that four applications of Chlorpyrifos to freshwater ponds at 2-week intervals resulted in a high mortality (>42%) of mallard (Anas platyrhynchos) ducklings. Mammals are comparatively tolerant, although smaller mammals, such as mice, are affected more than larger ones. Sublethal effects of acute exposure to Chlorpyrifos also threaten organisms through such outcomes as bioconcentration, cholinesterase activity reduction, reduced growth, impaired reproduction, motor incoordination, convulsions and depressed population densities of aquatic invertebrates (U.S. Fish and Wildlife Service, 1988).

Diazinon and Chlorpyrifos bioaccumulate rapidly in aquatic organisms as a result of their chemical properties. Due to the relatively high octanol-water partitioning coefficient, Kow, of Diazinon and Chlorpyrifos, these OP pesticides transfer readily to biota (EXTOXNET, 1996), with BioConcentration Factors (BCF) of 540 L/kg wet fish for Diazinon and 2700 L/kg wet fish for Chlorpyrifos (USEPA, 2001). Halibut or spotted sandbass from Newport Bay pose a special risk for humans due to the bottom-feeding nature of these fish. Bottom-feeding species are known to accumulate high concentrations of contaminants both from direct contact with contaminated sediments and consumption of organisms living in contaminated sediments (Alabama Department of Public Health, 1996).

Studies of pesticide bioconcentration suggest that high levels of pollutant may remain for several days in fish exposed to contamination. For example, a study prepared by the University of Wisconsin-Superior and the Great Lakes Environmental Center (2000) found that bioaccumulation of Diazinon in saltwater fish is rapid, reaching steady state within four days, and that the bioconcentration factors for fish exposed to 1.8, 3.5 and 6.5 ug/L were 147, 147 and 213, respectively. The study determined that it takes about seven days for most of the accumulated Diazinon residues to be eliminated from fish systems. Such results suggest that consumption of fish during periods of high pesticide loading could result in risks to birds and/or human health.

Humans may be exposed to Diazinon and Chlorpyrifos pesticides through oral, dermal and inhalation intake routes, creating a risk for human health. Studies conducted by the EPA name the endpoint of human dermal contact as being significant serum and brain cholinesterase inhibition, while inhalation results in significant plasma and cholinesterase inhibition (U.S.EPA, 2001). The creeks, streams, estuaries and bays in the Santa Ana River area offer opportunities for walking, boating, rowing, swimming and fishing, all of which create the potential for pesticide contact by people engaged in recreational activities. Resorts in Newport boast nearby water recreation in the Bay, including swimming, snorkeling, scuba diving, trophy fishing, canoeing and water skiing (U.S. Resort & Cottage Registry, 2002). Oral intake may also occur among residents of Newport Bay as a result of fish consumption, which has been reported among low-income residents of the area. Human consumption of halibut, spotted sandbass and mullet fish is not uncommon in this area (Skinner, 2001). These activities create pesticide intake pathways for humans who either consume fish from the Bay, which receives waters with OP pesticides from the surrounding Santa Ana River watersheds. Dermal contact with water or oral water intake during recreation also poses a risk.

Residents of these watersheds may be further exposed to Diazinon and Chlorpyrifos when the pesticides are applied at the home for pest control. Exposure occurs either at the time of application or when residents enter a recently treated site. Diazinon is one of the leading causes of acute insecticide poisoning for humans with the majority of incidents occurring in the home. Chlorpyrifos is also a source of acute insecticide poisoning for humans, with the majority of incidents also occurring in the home. Children are especially vulnerable to poisoning due to their smaller size and high contact with treated areas during play, such as lawns or sediments. Potential routes of exposure for children in the home include dermal contact and inhalation of vapors or airborne particles (U.S. Environmental Protection Agency, 2001). However, inhalation remains a lesser intake route for humans due to the low vapor pressures of Diazinon and Chlorpyrifos (U.S. Department of Labor, 2001).

## 1.3 Organization of the report

This report presents the modeling frameworks used for the study, their hydrological implementations, the determination of loading rates per pollutant and land use for non-point sources, the observed water quality available for calibrating the model, the results of our dynamic loading analysis, the work being conducted at this point and the conclusions we have drawn to date.

## 2. Modeling Approach

Our research addressed the influence of land-use changes in a watershed on nutrient and pollutant loading to the receiving water bodies. In addition to changes due to current development practices (e.g., expanding urban and suburban areas, new golf courses and ski areas, development of upland areas for second homes, conversion to agriculture), we addressed the potential implementation of Best Management Practices (BMPs) in the various land-uses (e.g. agriculture, pasture, suburban, urban, forests, etc.), for example the work by (Baker *et al.*, 1995; Barling and Moore, 1994; Earles, 1999; France *et al.*, 1998; Irvine and Ray, 1996; Moore *et al.*, 1992; Persson *et al.*, 1983; Weatherley *et al.*, 1993; Yoon and Disrud, 1997). These BMPs were studied separately and then concurrent with land-use changes. The strong influence of land-uses and management practices (or lack thereof) makes it imperative that these issues be adequately addressed when developing Total Maximum Daily Loads (TMDLs). Federal, state and local agencies are under considerable pressure to develop TMDLs in coordination with stakeholders, and in principle the TMDLs have to consider future changes in the watershed. To do so, it is necessary to understand the contribution of chemical loading from each subcatchment in the watershed, under current and future land-use and management practices, considering:

- Spatial and temporal variation of hydrologic and chemical loading from the atmosphere;
- Seasonal patterns of land-use practices (e.g. crop cultivation, pesticide application, fertilization, crop rotation, etc.);
- Seasonal processes in natural ecosystems that influence chemical fate and transport (e.g. primary productivity and nutrient uptake, litterfall, leaf-area coverage, etc.).

Since each subcatchment has its own characteristics and external driving forces (e.g., topography, micrometeorology, vegetation type and cover, soils, surface detention reservoirs, land-use combination, etc.), the analysis runs along many dimensions.

To evaluate all of these processes interactively, it is necessary to use integrated modeling frameworks that consider atmospheric, hydrologic, geomorphic and biogeochemical processes, and that realistically describe the spatial and temporal dimensions. For the current research, we implemented the Better Assessment Science Integrating Point and Nonpoint Sources (BASINS) model developed by USEPA and TetraTech (USEPA, 2000a), as well as the similar Watershed Analysis Risk Management Framework (WARMF) developed by the Electric Power Research Institute (EPRI) and Systech Engineering, Inc (EPRI, 1998). Although their respective peer reviews (Keller and al, 2000; USEPA, 1999) have shown that there are some deficiencies with either framework, there are few other options for developing TMDLs or studying all these watershed-scale processes. Both of these models rely on a mechanistic process description, and can be modified in terms of their spatial and temporal resolution to the appropriate scale.

Both modeling frameworks use well-known hydrologic, sediment transport and water quality (biogeochemical) models, such as ILWAS (Chen *et al.*, 1984; Gherini *et al.*, 1985), ANSWERS (Beasley, 1983; Beasley *et al.*, 1980; Dillaha *et al.*, 1982; McCain, 1980; Montas and Madramootoo, 1991; Parks *et al.*, 1982; Silburn and Connolly, 1995; USEPA, 1981), SWMM (Codner, 1991; Karkowski and Walters, 1994; Pandit and Gopalakrishnan, 1997), WASP5 (Suarez *et al.*, 1995; Tsiros and Ambrose, 1998; Tufford and McKellar, 1999; Warwick *et al.*, 1997), HSPF/NSPM (Al-Abed, 1998; Becknell *et al.*, 1993; Chen *et al.*, 1995; Codner, 1991; Fielland and Ross, 1991; Laroche *et al.*, 1996; Munson *et al.*, 1998), QUAL-2E (Thakar and Rogers, 1994) and TOXIROUTE. These models share many common processes and the corresponding mathematical representation(s). The basic principle is to have the meteorological input drive the hydrologic cycle, with the hydrologic model considering precipitation, canopy interception, throughfall, infiltration, groundwater transport, surface detention, surface runoff, discharge to receiving water bodies, flow in streams and rivers to the receiving lakes or reservoirs. Sediments are transported down the watershed, to stream and rivers, which in turn transport them to the lakes. Similarly, chemicals (from atmospheric deposition, biome processes, agricultural practices, etc.) are transported through the watershed to the receiving water bodies. Transformation of many chemicals is considered, as well as other sinks (sorption, burial in sediments, etc.). The key dependent variables are water quantity and quality (dissolved oxygen, temperature, nutrients, TDS, SS, pesticide concentrations, metals, organic compounds, etc.), which are the relevant to the development of TMDLs, either directly as water quality criteria, or indirectly as contributors to biogeochemical processes within the terrestrial or aquatic media. More details about BASINS is available through the model documentation (USEPA, 2000b); for WARMF, there are two documents which provide information about the model (EPRI, 1998; Systech, 2000), as well as a peer-reviewed publication (Chen *et al.*, 1996).

## 2.1 Hydrologic Model Implementation

We implemented the BASINS software, version 2.0, to simulate the fate and transport of Chlorpyrifos, Diazinon, and sediment. BASINS is a biogeochemical watershed model that integrates meteorological data, geographical information systems (GIS) data, specific chemical properties, hydrologic and soil characteristics, and landuse distribution to simulate the transport of constituents through a watershed. TetraTech, Inc. developed the BASINS software for the U.S. Environmental Protection Agency's Office of Water.

The BASINS physiographic data, monitoring data, and associated assessment tools, are integrated in a customized GIS environment. The GIS used is ArcView 3.0a, developed by Environmental Systems Research Institute, Inc. The simulation models are connected into this GIS environment through a link in which the input files for the watershed simulation model (NPSM/HSPF) are generated in the ArcView environment and then passed directly to the model (USEPA Office of Water, 1998). The BASINS system includes a variety of databases that facilitate watershed-based analysis and modeling. The databases were compiled from a wide range of federal sources including USGS, Bureau of the Census, USEPA, U.S. Department of Agriculture, and the National Oceanic and Atmospheric Administration. Individual data sources are listed in the BASINS User Manual (USEPA Office of Water, 1998).

WARMF consists of five modules: Engineering, Knowledge, Data, Consensus, and TMDL. The Engineering and Data module were used for this project. The Engineering module consists of three models: catchment, river, and reservoir, which are seamlessly integrated. The catchment module retrieves daily meteorological data and monthly atmospheric data, and then simulates canopy processes, snowpack, infiltration into the ground, surface runoff, and associated non-point source load (EPRI, 2000). The river model proceeds to accept any outflow from the catchments and routes the hydrology and water quality from one stream segment to the next. The river model can also route stream segments into reservoirs, which are simulated by the reservoir model. The reservoir model simulates deposition and thermal stratification of a lake and its outflow. The automatic integration of these three modules allows WARMF to output flow, chemistry, and sediment for the entire basin. The Data module is a database that stores a time series of the observed meteorology, air quality, hydrology, and water quality (EPRI, 2000).

GIS data for the Santa Ana River and Newport Bay watersheds was downloaded from the EPA website (http://www.epa.gov/OST/BASINS). This information included the following site-specific GIS data:

- USGS Hydrologic Cataloging Units #18070203 and 18070204 core data
- Base cartographic data including Hydrologic Unit Boundary, Major Roads, Populated Place Locations, Urbanized Areas, State and County Boundaries, EPA Regions, Ecoregions, National Water Quality Assessment Study Unit Boundaries, 1996 Clean Water Needs Survey, State Soil and Geographic Database, Managed Area Database, and Reach 1 Files (major channel network)
- 30-meter resolution Digital Elevation Map (DEM)
- Reach 3 Files (detailed tributary network)
- Landuse (based on SCAG 1993)

The Data module of the base watershed was updated with observed data applicable to our study area from the National Climatic Data Center (NCDC). Minimum temperature, maximum temperature, and precipitation data were used for both models. The Los Angeles International Airport Station (ID: 045114) was the only station within reasonable distance of the watershed that recorded daily dewpoint and wind speed. Meteorological data at 15 minutes, hourly or daily average was obtained from the National Climate Data Center site (http://lwf.ncdc.noaa.gov/oa/climate/research/cag3/cag3.html) spanning the following time periods:

Table 3. Meteorological stations available for the Santa Ana River Basin and Newport Bay

| COOP ID | WBAN ID | Station Name | Lat (N) | Long (W) | Elev. (m) | 15-min | Hourly | Daily |
|---|---|---|---|---|---|---|---|---|
| 40192 | | Anaheim | 33 52' | 117 51' | 102 | | | 1989-2002 |
| 41057 | | Brea Dam | 33 53' | 117 56' | 84 | 1984-1994 | 1970-2002 | |
| 41517 | | Carbon Canyon Dam | 33 55' | 117 50' | 122 | | 1972-1977 | |
| 41518 | | Carbon Canyon Gilman | 33 55' | 117 47' | 495 | 1984-1999 | 1970-2002 | |
| 42775 | | El Modena | 33 48' | 117 47' | 140 | | 1970-1977 | |
| 43285 | | Fullerton Dam | 33 54' | 117 53' | 104 | 1984-1994 | 1970-2002 | |
| 43288 | | Fullerton Hillcrest Res. | 33 53' | 117 55' | 101 | | | 1970-1976 |
| 44647 | | Laguna Beach | 33 33' | 117 47' | 11 | | | 1970-2002 |
| 44650 | | Laguna Beach 2 | 33 33' | 117 48' | 64 | 1971-2002 | 1970-2002 | |
| 46175 | 3107 | Newport Beach | 33 36' | 117 53' | 3 | | | 1970-2002 |
| 46473 | | Orange County Res. | 33 56' | 117 53' | 201 | 1984-2002 | 1970-2002 | |
| 47836 | | San Juan Canyon | 33 32' | 117 33' | 114 | | | 2001-2002 |
| 47837 | | San Juan Guard Station | 33 36' | 117 31' | 223 | 1978-2002 | 1970-2002 | |
| 47888 | | Santa Ana Fire Station | 33 45' | 117 52' | 41 | | | 1970-2002 |
| 47987 | | Santiago Dam | 33 47' | 117 43' | 261 | 1984-2001 | 1970-2001 | |
| 48243 | | Silverado | 33 45' | 117 39' | 334 | 1984-2002 | 1970-2002 | |
| 48992 | | Trabuco Canyon | 33 39' | 117 36' | 296 | 1971-2002 | 1970-2002 | |
| 49087 | | Tustin Irvine Ranch | 33 42' | 117 45' | 72 | | | 1970-2002 |
| 49847 | | Yorba Linda | 33 53' | 117 49' | 107 | | | 1970-2002 |
| | 93101 | El Toro Mcas | 33 40' | 117 44' | 116 | | | 1970-1999 |
| | 93114 | Tustin Mcaf | 33 42' | 117 50' | 18 | | | 1970-1998 |
| | 93117 | San Clemente Is | 33 01' | 118 35' | 52 | | | 1970-1989 |
| 046175 | 03107 | Newport Beach, CA | 33 36' | 117 53' | 3 | | | 1971-1995 |
| 045114 | 23174 | LAX Airport* | 33 56' | 118 24' | 30.5 | | | 1971-1995 |

*LAX data was used only to fill in missing data sets as needed.

The BASINS model averages geologic and hydrologic characteristics within each subcatchment, requiring that the watershed be discretized into individual subcatchments with homogeneous (spatially-averaged) properties. In order to facilitate model calibration, the watershed delineation was also determined by proximity to USGS flow gauging stations. Stream flow, groundwater elevation, and surface and ground water quality data was obtained from the USGS National Water Information System (NWIS) website (http://waterdata.usgs.gov/nwis-w/CA/).

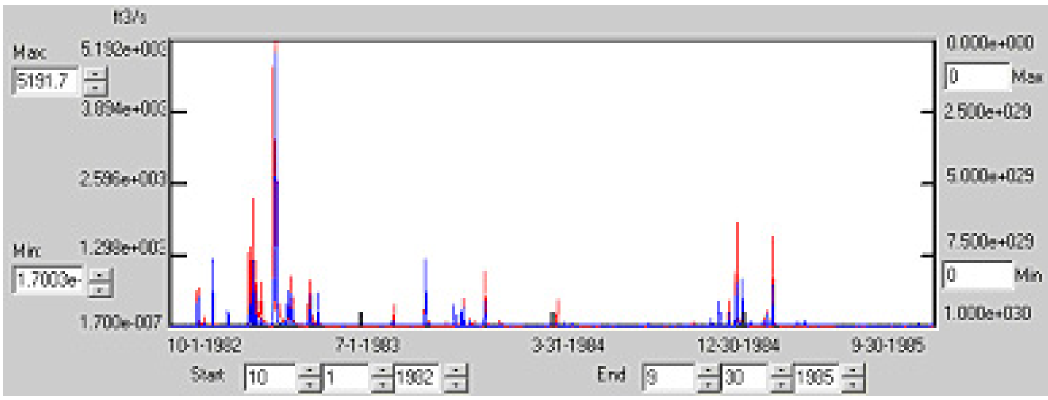
Other characteristics of the Newport Bay watershed were also input into the WARMF implementation including landuse, soil and physical information. Landuse designations and percent area of each catchment were queried from the landuse GIS layer in the BASINS model. The GIS layer was acquired from SCAG and represents the spatial pattern seen in 1993 at 30-meter (m) resolution. The 30 m resolution Digital Elevation Map (DEM) available in BASINS of Newport Bay watershed was queried in order to determine the length and slope of each stream segment. Soils information is also well developed, since basin wide models have been constructed for this region (NAWQA, 2000).

Once the watershed was characterized and appropriate initial and boundary conditions were set up, we proceeded with calibration. The procedure was to first calibrate the hydrologic submodels of each modeling framework, to ensure that there is a reasonably good match of the observed hydrologic data in all stream segments, lakes and reservoirs. Once appropriate meteorological information is entered, the cross-sectional characteristics of each river or tributary segment are adjusted to produce reasonable water flowrates compared to observed data. The other important hydrological calibration parameters are related to hydraulic conductivity, subsurface storage and initial soil moisture content. From previous work, it became clear that the most important calibration parameters were related to subsurface storage, which to a large extent controls the fate of precipitation (i.e. whether it infiltrates, moves on the landscape or is lost via evapotranspiration).

The model fit was measured by comparing total flow over the period of calibration between the simulated and actual flow. For example, for the Newport Bay area, the BASINS model over-predicted the total flow for Peter's Canyon Subcatchment by 49%, and under-predicted total flow for the Upper San Diego Subcatchment by 13%. The Intermediate San Diego Subcatchment represents a combination of flow from all subcatchments (Below the confluence of Peter's Canyon Channel and Upper San Diego Creek). The model simulation of total stream flow for the Intermediate San Diego Subcatchment was within 5.5% of the actual stream flow, recorded at gauging station #11048550 (Figure 6).

Sediment transport was calibrated by a combination of parameter value adjustments. The channel geometry for Peter's Canyon was made steep and narrow, and the channel geometries for Upper San Diego Creek and Intermediate San Diego Creek were made flat and wide to replicate channelization. We selected a sediment distribution of 20% sand, 60% silt, and 20% clay. The concentration of simulated suspended sediment remained lower than that found in the actual recorded data, probably given lack of information of construction projects that are believed to contribute significantly in parts of the watershed (ACE, 2000). Peaks in the recorded data had a maximum of approximately 10,000 mg/L, and the model simulated maximum peaks of approximately 1,900 mg/L. Possible sources of variance include undetected sources from watershed such as construction sites, and unstable stream banks.

Figure 6. Stream Flow Hydrograph. Model fit was determined by comparing simulated stream flow (cfs) in red to actual recorded stream flow in blue. Observed stream flow from gauging station #11048550, located on San Diego Creek just above where San Diego Creek enters the upper Newport Bay.




WARMF Model calibration concentrated on hydrology and total suspended sediment. Hydrology was calibrated by comparing the mean flow and accumulated flow of the simulated hydrology versus the observed hydrology from January 1, 1982 to December 30, 1995. Simulated flow was adjusted by correcting the initial moisture, field moisture, and conductivity of the soils for each catchment. While the mean flow value (Table 4) indicates the overall accuracy of the simulated flow when compared to observed flow, it is important to also evaluate the hydrographs for each river segment in order to determine if the system is responding to rain events. The hydrographs for each river segment revealed that the model was under predicting major stormflow events (Figure 7 through 9). WARMF also allows the user to examine the accumulated water volume over the time period simulated. The amount of water flowing through Peter's Canyon Wash and upper San Diego Creek became greater towards the end of the simulation period, however, lower San Diego Creek, below the confluence of the upper two segments, was a nearly perfect predictor of the observed data. In general, all three segments matched the accumulated volume of the observed data (Figures 10 through 12). Evaluating these three indicators of model fit show that while the model outputs the same amount of water as the actual watershed it runs using a higher baseflow and does not peak as high in storm events.

Table 4. Calibration Quantification for the Three Main Reaches of the Base WARMF Newport Bay Watershed Model.

| | **Upper San Diego Creek** | | **Peters Canyon Wash** | | **Lower San Diego Creek** | |
|---|---|---|---|---|---|---|
| | **Simulated** | **Observed** | **Simulated** | **Observed** | **Simulated** | **Observed** |
| Mean Flow ($m^3s^{-1}$) | 0.364 | 0.414 | 0.794 | 0.747 | 1.506 | 1.604 |
| Maximum Sediment Concentration (mg/L) | 5864 | 5880 | 12410 | 12500 | 9719 | 9960 |

Figure 7. Hydrograph for Upper San Diego Creek From January 1, 1982 Through December 30, 1985. Simulated Flow ($m^3s^{-1}$) Under Predicts Observed Stormflow.

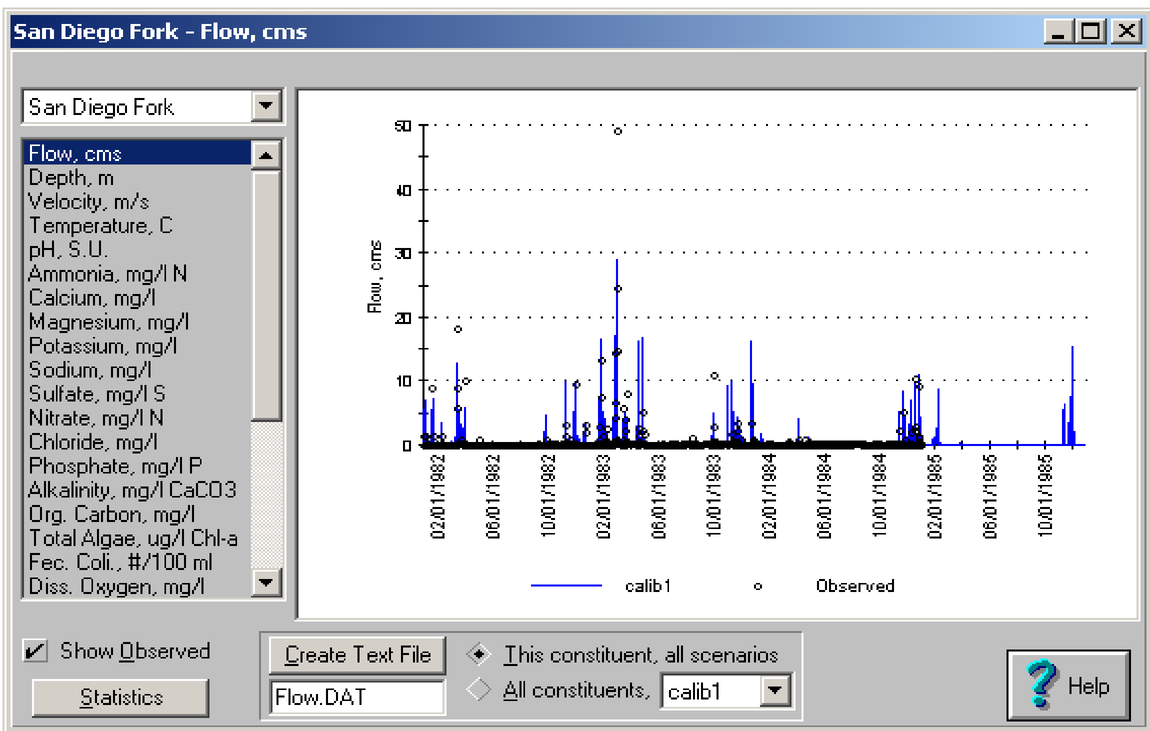


Figure 8. Hydrograph for Peter's Canyon Wash From January 1, 1982 Through December 30, 1985.

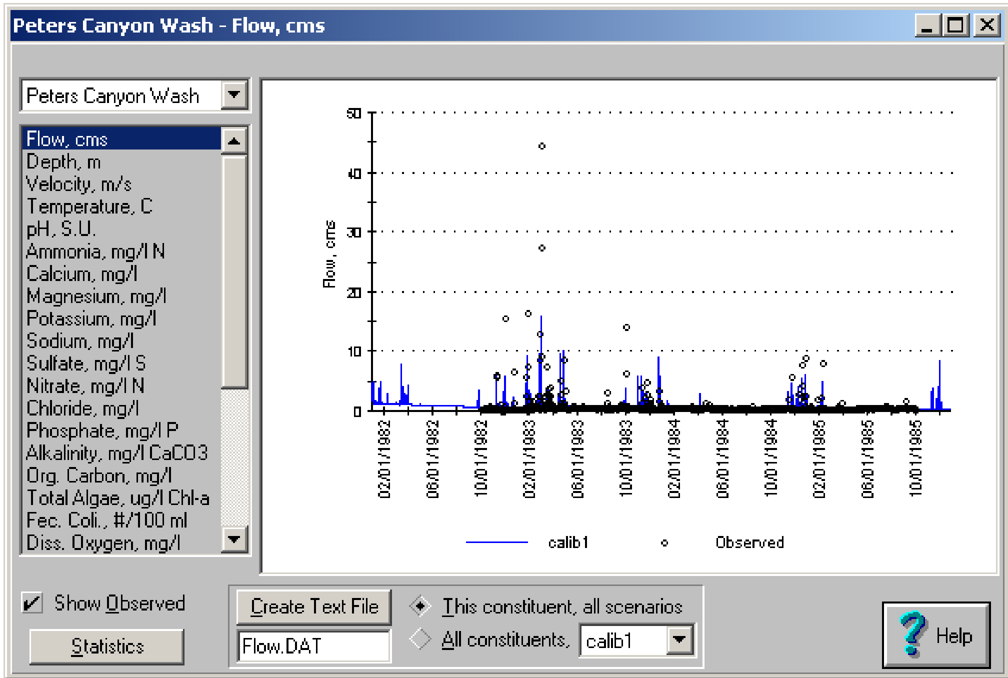



Figure 9. Hydrograph for Lower San Diego Creek From January 1, 1982 Through December 30, 1985.

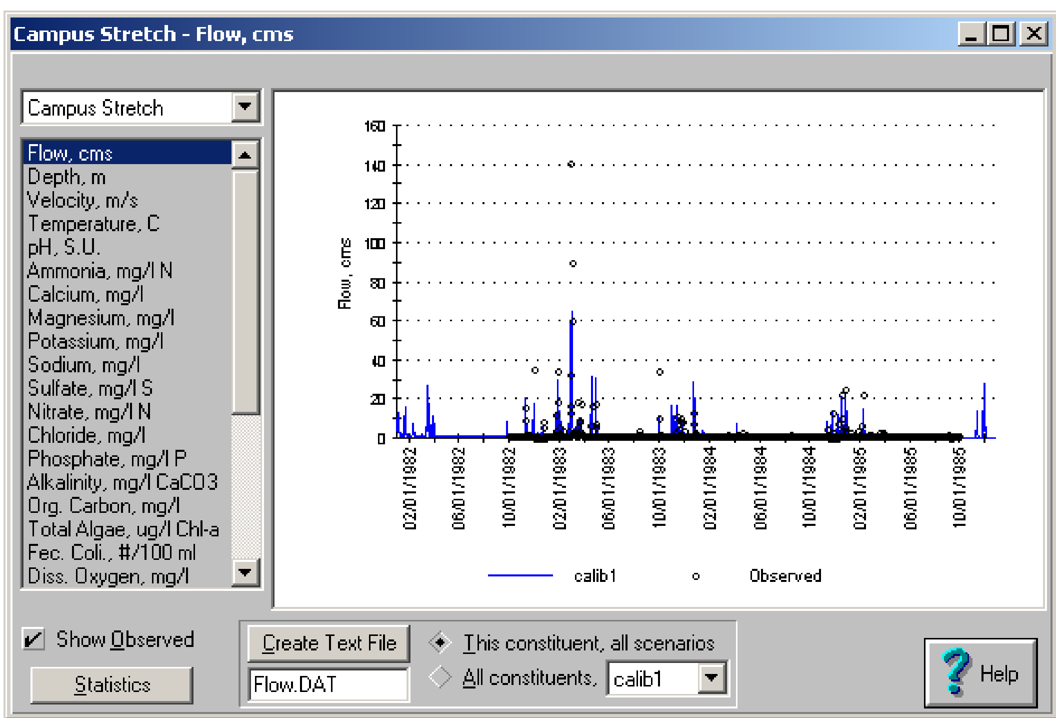
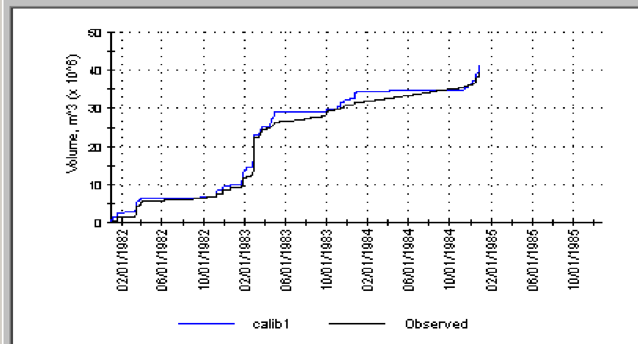


Figure 10. Simulated Flow (Blue) Cumulative Volume Versus Observed Flow (Black) Cumulative Volume For Upper San Diego Creek From January 1, 1982 Through December 30, 1985.

San Diego Fork - Flow, cms

| | calib1 | Observed |
|---|---|---|
| Mean | 0.364 | 0.414 |
| Minimum | 000000289 | 0 |
| Maximum | 28.86 | 49.27 |
| # Compare Pts | 1097 | 1097 |
| Relative Error | 0.0189 | 0 |
| Absolute Error | 0.355 | 0 |
| RMS Error | 1.373 | 0 |
| r squared | 0.62 | 1 |



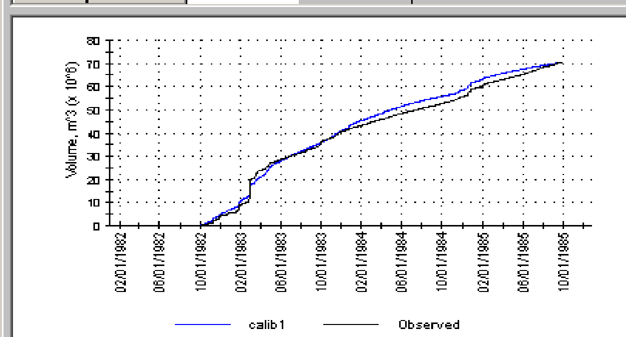

Help

Scatter Plot Frequency Distribution Cumulative Quantity

Figure 11. Simulated Flow (Blue) Cumulative Volume Versus Observed Flow (Black) Cumulative Volume For Peter's Canyon Wash From January 1, 1982 Through December 30, 1985.

Peters Canyon Wash - Flow, cms

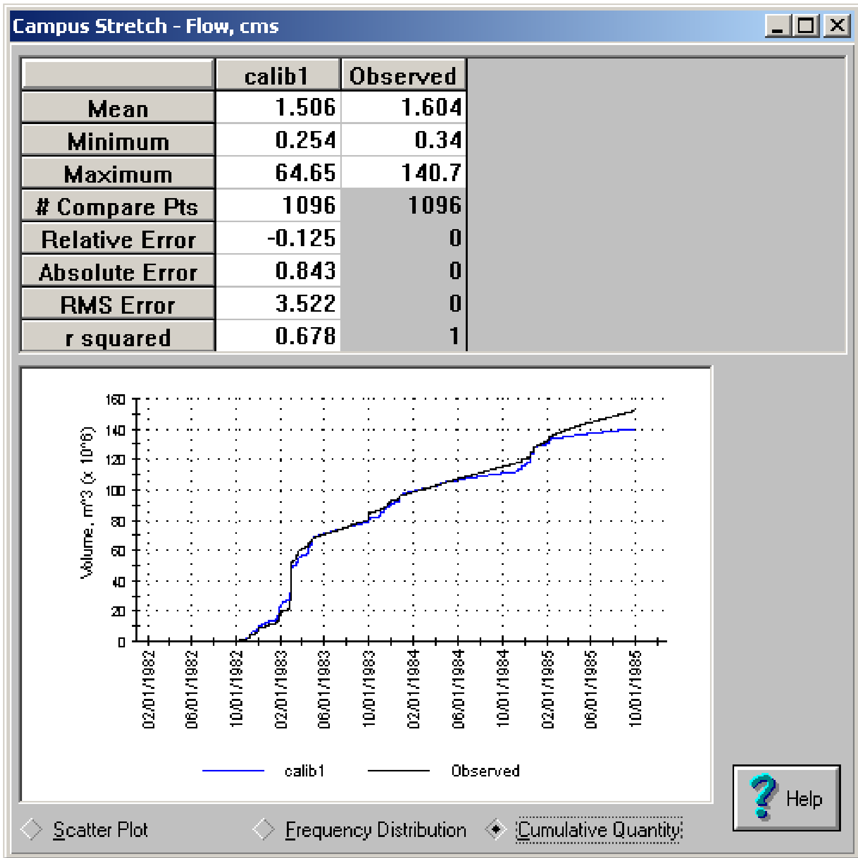
| | calib1 | Observed |
|---|---|---|
| Mean | 0.794 | 0.747 |
| Minimum | 0.251 | 0.113 |
| Maximum | 15.88 | 44.46 |
| # Compare Pts | 1096 | 1096 |
| Relative Error | -0.000821 | 0 |
| Absolute Error | 0.341 | 0 |
| RMS Error | 1.322 | 0 |
| r squared | 0.608 | 1 |




Help

Scatter Plot Frequency Distribution Cumulative Quantity

Figure 12. Simulated Flow (Blue) Cumulative Volume Versus Observed Flow (Black) Cumulative Volume for Lower San Diego Creek From January 1, 1982 Through December 30, 1985.




Total suspended sediment calibration was important because it is a key variable in the fate and transport of Diazinon and Chlorpyrifos. The total suspended sediment fit was determined by comparing the maximum simulated amounts versus the observed peaks (Table 4) during the period from January 1, 1982 through January 1, 1985. Calibration of the sediment was achieved by adjusting the bank stability factor associated with the stream channels. The stage-width curve of each major tributary was also changed in order to portray the narrow concrete channels found in the Newport Bay watershed. Figures 13 through 15 illustrates the total suspended sediment simulated output versus observed measurements for each catchment. The model outputs a number of simulated peaks that concur in time and magnitude with observed values.

Figure 13. Simulated Total Suspended Sediment (mg/L) for Upper San Diego Creek From January 1, 1982 Through January 1, 1985.

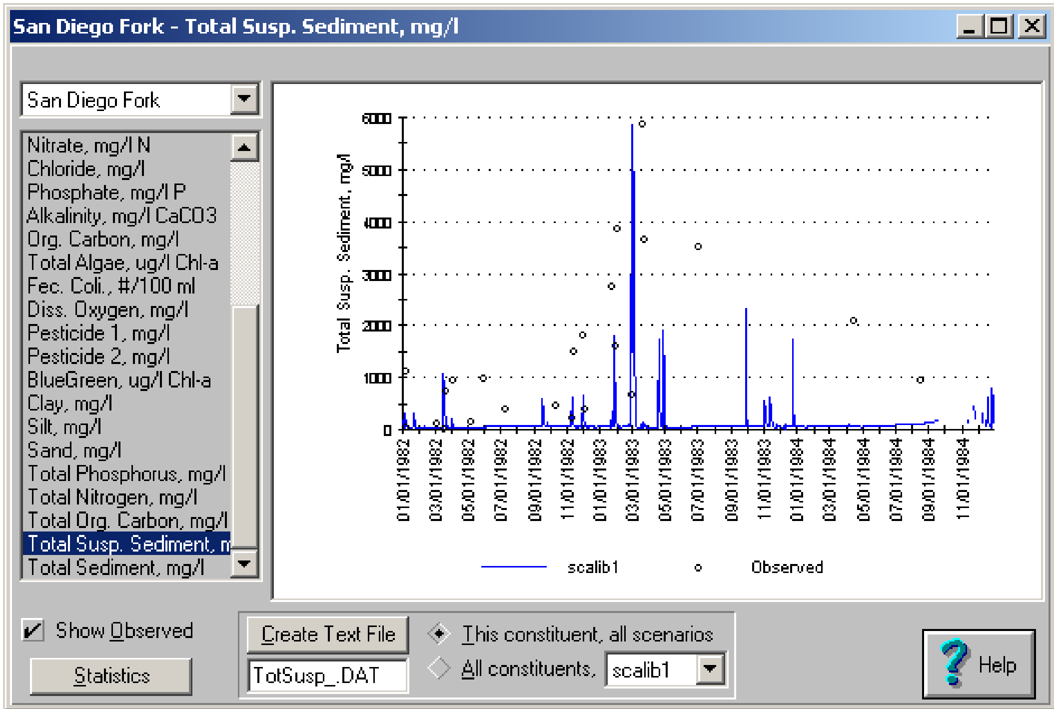



Figure 14. Simulated total suspended sediment (mg/L) for Peter's Canyon Wash from January 1, 1982 through January 1, 1985. Peter's Canyon Wash produces the most sediment within the watershed.

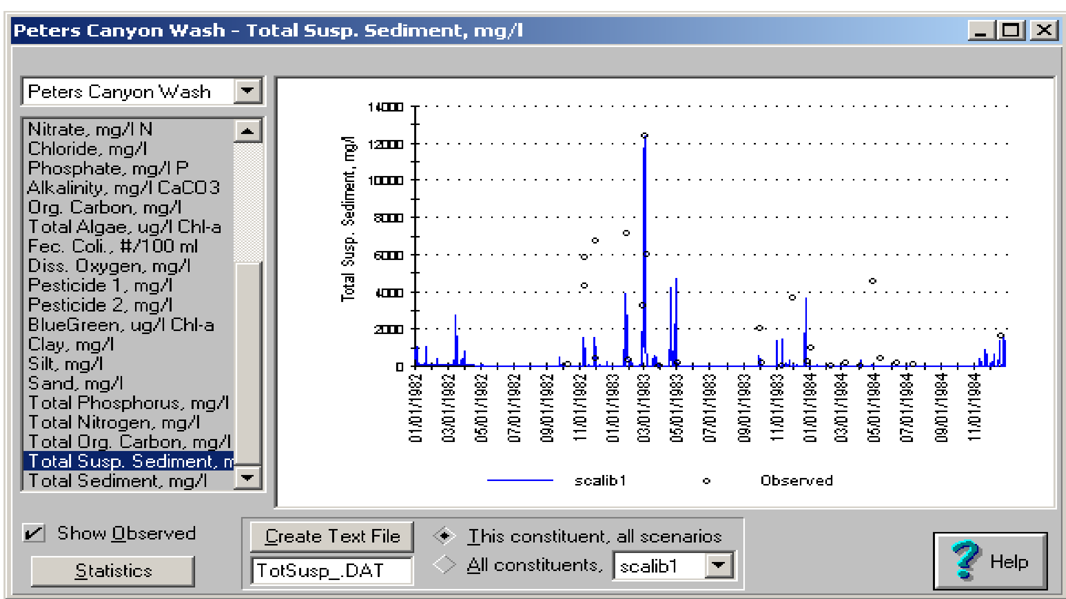
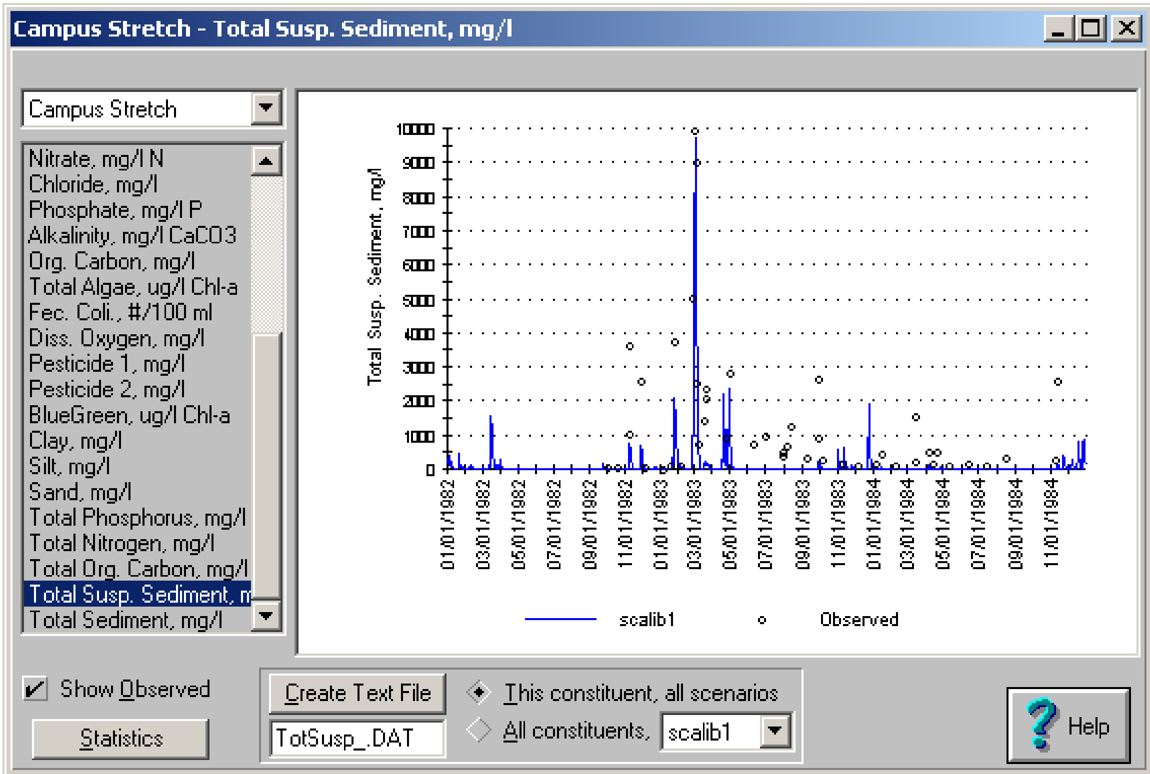


Figure 15. Simulated Total Suspended Sediment (mg/L) for Lower San Diego Creek from January 1, 1982 Through January 1, 1985.



## 3. Land Use Loading Rates

Key to any water quality simulation that considers non-point sources are the assumptions used for land-use loading rates. Thus, we spent a considerable amount of time obtaining this information for the project, including not only the loading rates per land-use, but the spatial and temporal variability.

### 3.1 OP Pesticides Loading Rates

One of the primary objectives of this project was to gain insight into the significance of different land uses with respect to pesticide source loading. The primary categories for reported OP pesticide usage in Newport Bay are: structural pest control, nurseries, agricultural, landscape and other non-residential uses (SARWQCB, 2001). A study by Scanlin and Feng (1998) found results suggesting that residential users applying the pesticides in accordance with label directions may still be contributing significantly to aquatic toxicity.

Using information from the California Department of Pesticide Regulation (CDPR), one observes an increase in usage of Diazinon in the Newport Bay area for urban applications and a decrease in agricultural usage, over the 1995-1996 five-year period (Table 5). Structural pest control uses accounts for approximately 40% of the total load, while residential uses account for about 55% of the usage within this watershed.

Table 5. Diazinon Use in Newport Bay (kg Active Ingredient)

| Use | 1995 | 1996 | 1997 | 1998 | 1999 |
|---|---|---|---|---|---|
| Structural | 3,493 | 2,809 | 3,778 | 4,615 | 4,417 |
| Nursery | 207 | 167.8 | 160.6 | 242 | 229 |
| Agriculture | 401 | 149.2 | 273 | 173 | 85.8 |
| Landscape | 206 | 152.4 | 119 | 122.4 | 157.8 |
| Non-residential | 1.96 | 9.24 | 0.32 | 0.34 | 1.06 |
| Reported subtotal | 4,309 | 3,288 | 4,331 | 5,153 | 4,890 |
| Estimated Residential Use | 4,787 | 3,843 | 5,042 | 6,129 | 5,919 |
| Total | 9,096 | 7,131 | 9,373 | 11,282 | 10,810 |

*Source:* Based on DPR (1999) database.

Table 6 shows the usage of Chlorpyrifos in Newport Bay from 1995-1999. Nursery and agricultural uses account for roughly 3% of total load; therefore residential and urban areas account for the majority of Chlorpyrifos loading. Structural insect control represents about 62% of the load, primarily in the treatment of wood protection from termites, with unpermitted residential uses representing about 35%. The large increase in usage by nurseries in 1999 can likely be attributed to the requirements imposed by Red Imported Fire Ant (RIFA) eradication program of the California Department of Food and Agriculture (CDFA). The CDFA RIFA program requires treatment of targeted areas with both Diazinon and Chlorpyrifos (CDPR, 1999).

Table 6. Chlorpyrifos Use in Newport Bay (kg Active Ingredient)

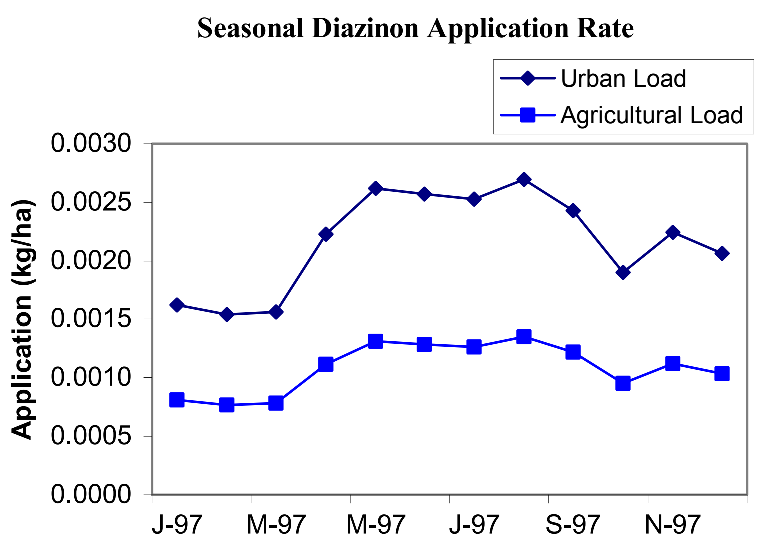
| Use | 1995 | 1996 | 1997 | 1998 | 1999 |
|---|---|---|---|---|---|
| Structural | 7,653 | 14,435 | 13,973 | 17,797 | 14,981 |
| Nursery | 130.4 | 154.4 | 194.2 | 198.8 | 583 |
| Agriculture | 283 | 190.4 | 290 | 129 | 226 |
| Landscape | 289 | 246 | 275 | 216 | 201 |
| Non-residential | 1.4 | 53.7 | 0.32 | 0.32 | 7.06 |
| Reported subtotal | 8,356 | 15,079 | 14,732 | 18,341 | 15,998 |
| Estimated Residential Use | 4,333 | 8,037 | 7,772 | 9,826 | 8,285 |
| Total | 12,689 | 23,116 | 22,504 | 28,167 | 24,283 |

*Source*: Based on DPR (1999) database

To obtain a loading per unit area, we considered the land use information available for Newport Bay in 1993. We assumed that structural pest control would apply mostly to residential and commercial buildings. Since the industrial land use is small, there is probably a small error associated with some additional use in these facilities. Using the GIS land use information, the probable spatial distribution of the loading can be predicted. Naturally, loading is usually focused on a few structures within a particular residential area, so this spatial distribution is intended to provide a first approximation of the actual distribution.

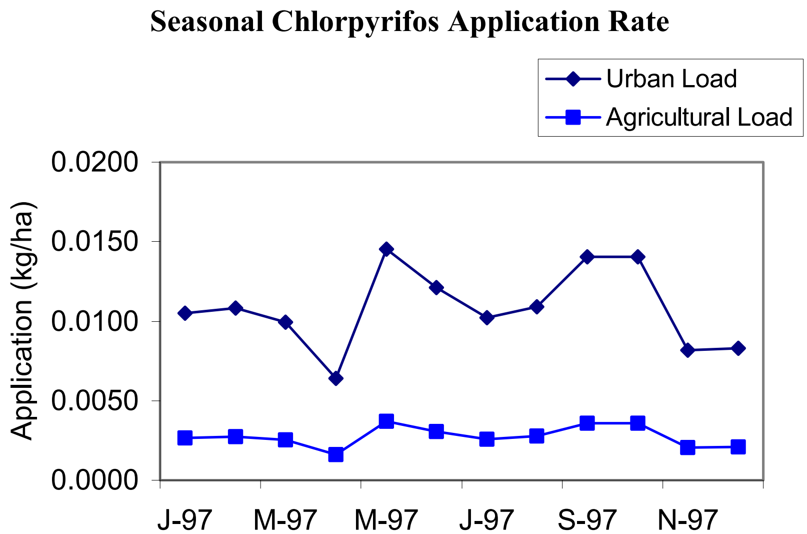
Data provided by CDPR indicate that pesticide application rates are higher during the drier, summer months corresponding to the pest life cycle (CDPR, 1998), as presented in Figures 16 and 17. Some of the variation is also likely due to hydrologic factors. Runoff concentrations are generally higher during rainy season stormflow conditions, as the accumulated pesticide is transported to urban creeks.

Figure 16. Monthly average Diazinon application rates, data from 1997.



Source: CDPR (1998)

Figure 17. Monthly average Chlorpyrifos application rates, data from 1997.




Source: CDPR (1998)

As discussed in Section 1.2, these two OP pesticides are expected to sorb strongly to soils and be degraded mostly near the point of application. Only a small fraction is expected to reach the waterbodies under normal management practices. It should be noted that both pesticides occur in water samples during wet and dry cycles, which points to their persistence in stormflow and baseflow conditions.

## 3.2 OP Pesticide Export Load per Landuse

Pesticide loading is largely a non-point source problem, with uncertainty about the precise contribution from specific locations and landuse types within the study area. For this reason, it is useful to estimate an export rate coefficient that corresponds to a given landuse. The amount of pesticide exported from a landuse not the amount of pesticide applied to a specific landuse. From the application data compiled by DPR (1999), export rates of both Diazinon and Chlorpyrifos can be calculated for urban and agricultural areas. Load calculations were completed by dividing the cumulative application for either urban or agricultural areas by its representative area in the watershed. The export rates represent a general load per landuse and are used to calculate the relative contribution from urban and agriculture pesticide inputs.

Tables 7 and 8 show approximate exports per landuse coming from urban and agricultural landuses (SARWQCB, 2001). Contributions from open spaces are assumed to be negligible. The load for agriculture and urban uses were on the same order of magnitude for baseflow and stormflow conditions. The export rates shown in the tables formed the basis for our modeling approach, which simulates the spatial distribution of the two pollutants within Newport Bay. For model input parameters, we interpolated export rates over the areal extent of the watershed, for each landuse. The output from the model can be viewed as a pollutant load/area/time or as a concentration (mass/volume).

Diazinon was often detected throughout the year and not just during storm events. These results are consistent with a study on Diazinon and dry weather flows done in Alameda County, California (URS, 1999), indicating that there are high peak concentration values during baseflow despite the smaller overall contribution. The load contribution from agricultural lands during baseflow highlight that pesticide loading is not restricted to stormflow.

Table 7. Estimated Annual Diazinon Export Load per Landuse.

| Condition | Land Use | Area (acres) | Area (%) | Load (lbs) | Load (%) | Load (lbs/acre) |
|---|---|---|---|---|---|---|
| **Baseflow** | urban | 66,507 | 68% | 2.4 | 88% | $3.6 \times 10^{-5}$ |
| | agriculture | 9,286 | 10% | 0.31 | 12% | $3.4 \times 10^{-5}$ |
| | Total | 97,741 | 100% | 2.7 | 100% | $2.8 \times 10^{-5}$ |
| **Stormflow** | urban | 66,507 | 68% | 24.1 | 96% | $3.6 \times 10^{-4}$ |
| | agriculture | 9,286 | 10% | 2.47 | 4% | $2.7 \times 10^{-4}$ |
| | Total | 97,741 | 100% | 26.6 | 100% | $2.7 \times 10^{-4}$ |

*Source*: Results after SARWQCB (2001).

Stormflow runoff carries a larger load of Diazinon, especially from urban landscapes. This results from the combination of a larger export rate and a greater contributing urban area. Additionally, Diazinon is detected in surface waters 98% of the time during stormflow events and 89% of the time during baseflow, stressing its ubiquity in residential, urban or agricultural environments.

Chlorpyrifos is associated with agricultural uses in both baseflow and stormflow. As with Diazinon, high agricultural loads during dry weather flow may be the result of excessive irrigation during the summer months. The loading rate from urban and agricultural landuse is identical during baseflow conditions and only slightly higher for agriculture during stormflow. Chlorpyrifos is not as mobile as Diazinon in the environment and despite a larger annual application, less Chlorpyrifos is transported to San Diego Creek and Newport Bay.

Table 8. Estimated Annual Chlorpyrifos Export Load per Landuse.

| Condition | Land Use | Area (acres) | Area (%) | Load (lbs) | Load (%) | Load (lbs/acre) |
|---|---|---|---|---|---|---|
| **Baseflow** | urban | 66,507 | 68% | 0.69 | 88% | $1.03 \times 10^{-5}$ |
| | agriculture | 9,286 | 10% | 0.1 | 12% | $1.03 \times 10^{-5}$ |
| | Total | 97,741 | 100% | 0.78 | 100% | $8.01 \times 10^{-5}$ |
| **Stormflow** | urban | 66,507 | 68% | 2.61 | 85% | $3.92 \times 10^{-4}$ |
| | agriculture | 9,286 | 10% | 0.46 | 15% | $4.90 \times 10^{-4}$ |
| | Total | 97,741 | 100% | 3.06 | 100% | $3.13 \times 10^{-4}$ |

*Source*: Results after SARWQCB (2001).

Using these various sources of information, we estimated a monthly OP pesticide application rate that varied according to season and landuse (Table 9). This information was then input to the models.

Table 9. Land Application Pesticide Amounts (kg/ha) per Catchment

| Month | Residential | | Cultivated | |
|---|---|---|---|---|
| | **Diazinon** | **Chlorpyrifos** | **Diazinon** | **Chlorpyrifos** |
| January | 0.001621 | 0.010513 | 0.00811 | 0.002669 |
| February | 0.001536 | 0.010821 | 0.000768 | 0.002747 |
| March | 0.001561 | 0.009949 | 0.000781 | 0.002526 |
| April | 0.002228 | 0.006406 | 0.001114 | 0.001626 |
| May | 0.002619 | 0.014518 | 0.00131 | 0.003685 |
| June | 0.002568 | 0.012125 | 0.001284 | 0.003078 |
| July | 0.002526 | 0.010221 | 0.001263 | 0.002594 |
| August | 0.002695 | 0.010915 | 0.001348 | 0.002771 |
| September | 0.002429 | 0.014043 | 0.001215 | 0.003565 |
| October | 0.001901 | 0.014049 | 0.000951 | 0.003566 |
| November | 0.002242 | 0.008158 | 0.001121 | 0.002071 |
| December | 0.002067 | 0.008283 | 0.001034 | 0.002103 |

## 3.2 Nutrients Loading Rate

The nutrient source identification and characterization analysis focused on the pollutants of primary concern in the Santa Ana River watershed: nitrogen (ammonia, nitrite, nitrate) and phosphorus. Both nitrogen and phosphorus are also present in organic matter. Some data is available for organic nitrogen, but since measurement of ammonia is more common than measurement of organic nitrogen, loading sources will be presented here in terms of ammonia. Since the natural reaction to convert nitrite to nitrate is faster than the reaction producing nitrite from ammonia, very little nitrite is normally present in nature. Therefore, the sources of nitrite are assumed to be exclusively surface and subsurface point source discharges. We divide the sources into point and non-point sources following common classification.

### Direct Sources

Direct sources are those which discharge directly to the surface waters in the affected watershed sub regions. Loading from these sources is only attenuated through in-stream processes including sediment adsorption and uptake by periphyton. Loading from these sources, as well as accompanying assimilative capacity, may also be removed by diversions.

#### *Reservoir Releases*

The releases from Prado Dam are treated in a manner similar to point sources, and no attempt is made to ascertain the ultimate source of pollutants. Flow for these sources is known from USGS gaging stations downstream of the dam, water quality is estimated from measured values or extrapolated as necessary. In the absence of specific data, nitrite loading is assumed to be zero.

### *Direct Point Sources*

Direct point sources are those which discharge directly to surface waters such as Santa Clara River and its tributaries. Each of these has a permit from the National Pollution Discharge Elimination System (NPDES). The flow and loading data for each was also compiled from Discharge Monitoring Reports (DMRs) and information from the Sanitation District when available.

## Subsurface Discharges

There are many groundwater waste discharges in the Santa Ana River watershed. In each case, there is a mechanism which allows the waste to percolate into the soil. Once into the soil, the water and its associated pollutants disperse, although they may be assimilated through soil adsorption or uptake by vegetation. These sources are not associated with any particular land use, but are assumed to be dispersed proportionately over all land uses.

### *Groundwater Discharges*

The State of California issues permits for groundwater discharges. Whenever available, such data was used to estimate the flow and nutrient concentrations of these sources.

### *Septic Systems*

Septic system loading is estimated by multiplying the number of septic systems, the number of people served by each septic system (assumed to be 2.3), and flow and loading per capita. It is assumed that there is no loading of nitrite or nitrate from septic systems, although nitrification of ammonia will indirectly produce these species. It is assumed that the loading is uniform every month of the year and throughout the analysis time period. The per capita loading is assumed to be 75 gallons/capita/day (Wagener 2002) at a concentration of 32 mg/l of ammonia as nitrogen and 6 mg/l of phosphate as P (Maizel et al 1997).

## Land Application Sources

These sources represent pollutants loaded to the land surface. Some portion is assimilated by soil and vegetation. The remainder may be transported through the soil to surface waters based on natural and irrigation hydrology. These sources are associated with specific land uses.

### *Diversions for Groundwater Recharge / Irrigation*

There are a number of locations within the watershed where water is diverted from the main stem or streams and applied to the land for agriculture. When available, this data was considered in the model. The concentration is assumed to be that of the river or tributary. The loading was calculated from the flow and average monthly concentrations from water quality monitoring near each diversion.

### *Well Pumping Irrigation*

Irrigation water pumped from the aquifer contains nitrogen and phosphorus. Agricultural pumping flow was compiled for each region of the watershed. Well water quality was also compiled. Water quality was averaged to estimate loading. For orchards and row crops, approximately 0.75 m of irrigation water is applied each year (Daugovich, 2002). The total irrigation water needed is calculated by multiplying 0.75 m by the area of orchard and cropland land uses in each region.

### *Atmospheric Deposition*

There are two forms of atmospheric deposition: wet and dry. Wet deposition is from pollutants present in rain. Dry deposition is from gradual accumulation on the ground and leaf surfaces during dry weather. Dry deposition includes particulate matter and uptake of gases including $NO_x$. $NO_x$ is converted to nitrate upon uptake by vegetation. Atmospheric deposition may be assimilated in the soil and in vegetative uptake, with some portion reaching surface waters through the natural hydrologic cycle. The following equations govern the collection of pollutants through atmospheric deposition (Chen, 2001).

Wet deposition to land use j $D_{jw}$ (kg/d) is a function of the amount of precipitation, the concentration of the precipitation, and the land area, as shown in equation 1.

$$D_{jw} = \frac{PC_p A_j}{10^9} \qquad (1)$$

where P is the precipitation rate (cm/d), $C_p$ is the precipitation concentration (mg/l), and $A_j$ is the area of land use j ($cm^2$). The dry deposition to land use j $D_{jd}$ (kg/d) is the sum of the particulate deposition to leaf surfaces $D_{jdl}$, the particulate deposition to the ground $D_{jdg}$, and the gaseous uptake by leaves $U_{jdl}$ as shown in equations 2-5.

$$D_{jd} = D_{jdl} + D_{jdg} + U_{jdl} \qquad (2)$$

$$D_{jdl} = \frac{e_d V_d C_a L_j A_j}{10^{15}} \qquad (3)$$

$$D_{jdg} = \frac{V_d C_a A_j}{10^{15}} \qquad (4)$$

$$U_{jdl} = \frac{e_d U_d C_a L_j A_j}{10^{15}} \qquad \text{(eq. 5)}$$

where $e_d$ is the dry collection efficiency (assumed 0.6 from Chen 1983), $V_d$ is the particulate deposition velocity (cm/d), $U_d$ is the gaseous uptake velocity (cm/d) and $C_a$ is the atmospheric concentration ($\mu g/m^3$). Since gaseous uptake means the nitrogen is absorbed to meet the nutrient demand of vegetation, this is not available for watershed loading, and it is omitted from the atmospheric loading in this analysis ($U_d = 0$). Linkage analysis will take this effect into account in determining the uptake needed by vegetation beyond $NO_x$ uptake.

Table 10 shows the monthly particulate deposition rate from Joshua Tree National Park (CASTNET 2001). The Joshua Tree site is the nearest of a national network of monitoring stations. A study prepared for the California Air Resources Board indicates that particulate deposition velocity of nitrate is approximately 0.182 times the gaseous deposition velocity of $HNO_3$ (Russell 1990). Russell cites two other studies (Finlayson-Pitts and Pitts 1986; McRae and Russell 1984) which give land use adjusted summer $HNO_3$ deposition rates ranging from 1.0 to 4.7. WARMF performs its own land use adjustments from a neutral deposition velocity. Adjusting the cited HNO3 deposition rates so that WARMF would approximate the same deposition flux gives a land use neutral $HNO_3$ deposition rate of approximately 1.2. Using the 0.182 relationship between particulate nitrate and gaseous $HNO_3$ from Russell, the estimated summer particulate deposition velocity of 0.22 cm/s is within 5% of that from Joshua Tree. Table 11 shows the estimated monthly leaf area index for each land use (Nikolov, 1999).

Table 10: Monthly Particulate Deposition Rate, cm/s

| | **Jan** | **Feb** | **Mar** | **Apr** | **May** | **Jun** | **Jul** | **Aug** | **Sep** | **Oct** | **Nov** | **Dec** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Particulate Deposition | 0.11 | 0.14 | 0.17 | 0.21 | 0.24 | 0.24 | 0.22 | 0.22 | 0.19 | 0.15 | 0.12 | 0.11 |

Table 11: Monthly Leaf Area Index for each Land Use

| **Land Use** | **Jan** | **Feb** | **Mar** | **Apr** | **May** | **Jun** | **Jul** | **Aug** | **Sep** | **Oct** | **Nov** | **Dec** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Deciduous | 0 | 0 | 0 | 0.5 | 1 | 2.5 | 4 | 4.5 | 4.5 | 1 | 0 | 0 |
| Mixed Forest | 1 | 1 | 1 | 2 | 2 | 3 | 4 | 4 | 3 | 2 | 1 | 1 |
| Orchard | 4 | 4 | 4 | 4 | 4 | 4 | 4 | 4 | 4 | 4 | 4 | 4 |
| Coniferous | 2 | 2 | 2 | 3 | 3 | 4 | 4 | 4 | 3 | 3 | 2 | 2 |
| Shrub / Scrub | 0.5 | 0.5 | 0.5 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 0.5 | 0.5 |
| Grassland | 0.5 | 0.5 | 0.5 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 0.5 | 0.5 |
| Park | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| Golf Course | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| Pasture | 0.5 | 0.5 | 0.5 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 0.5 | 0.5 |
| Cropland | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| Marsh | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 1 |
| Barren | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Water | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Residential | 0 | 0 | 0 | 0.2 | 0.4 | 1 | 1.6 | 1.8 | 1.8 | 0.4 | 0 | 0 |
| High Density Residential | 0 | 0 | 0 | 0.2 | 0.4 | 1 | 1.6 | 1.8 | 1.8 | 0.4 | 0 | 0 |
| Comm./Industrial | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

With precipitation data from various meteorological stations (NCDC, 2002), air quality and rain chemistry from nearby stations (NADP, 2002; CARB, 2002), and data for land uses within each subregion, one can calculate the total atmospheric deposition. We assume that only ammonia and nitrate are deposited from the atmosphere and that loading of phosphorus is insignificant.

### *Fertilization*

Fertilization is applied to the land surface for the purpose of being taken up by orchards and row crops. What is not taken up may be assimilated in the soil or may be transported to surface waters. Since fertilization occurs on land which is irrigated, it has a greater opportunity for transport than atmospheric deposition. Tables 12-14 show fertilization rates per unit area for agricultural land uses (UC Ag. Extension, 2002) and estimated unit rates for other land uses from animal waste, debris, and other sources. Nitrogen in fertilizer is assumed to be 50% ammonia and 50% nitrate.

When these application rates are applied based on the land use area in each subregion, the result is the net loading rate to each subregion.

Although we have done a detailed accounting of the nutrient loading to each landuse in the watershed, we continue to collect data in this regard, with further discretization into the types of agricultural activities (e.g. citrus trees, avocado, other orchards, row crops), since they represent an important fraction of the loading.

Table 12: Monthly Unit Land Application Rate of Ammonia Nitrogen, kg/ha/d

| **Land Use** | **Jan** | **Feb** | **Mar** | **Apr** | **May** | **Jun** | **Jul** | **Aug** | **Sep** | **Oct** | **Nov** | **Dec** | Mean |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Deciduous | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Mixed Forest | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Orchard | 0.000 | 0.000 | 0.000 | 0.123 | 0.245 | 0.245 | 0.245 | 0.245 | 0.245 | 0.123 | 0.000 | 0.000 | 0.123 |
| Coniferous | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Shrub / Scrub | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Grassland | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Park | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Golf Course | 0.000 | 0.000 | 0.000 | 0.960 | 0.960 | 0.960 | 0.960 | 0.960 | 0.960 | 0.000 | 0.000 | 0.000 | 0.480 |
| Pasture | 0.068 | 0.068 | 0.068 | 0.068 | 0.068 | 0.068 | 0.068 | 0.068 | 0.068 | 0.068 | 0.068 | 0.068 | 0.068 |
| Farm 1[1] | 0.000 | 0.000 | 0.000 | 0.577 | 0.577 | 0.577 | 0.577 | 0.577 | 0.577 | 0.000 | 0.000 | 0.000 | 0.288 |
| Farm 2[2] | 0.288 | 0.288 | 0.288 | 0.577 | 0.577 | 0.577 | 0.577 | 0.577 | 0.577 | 0.288 | 0.288 | 0.288 | 0.433 |
| Farm 3[3] | 0.412 | 0.412 | 0.824 | 0.824 | 0.824 | 0.824 | 0.824 | 0.824 | 0.824 | 0.824 | 0.824 | 0.412 | 0.721 |
| Marsh | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Barren | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Water | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Residential | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 |
| High Density Residential | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 |
| Comm./Industrial | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 |

1. Farms at elevations above 600 m
2. Farms at elevations above 300 m but below 600 m
3. Farms in the floodplain at low elevations

Table 13: Monthly Unit Land Application Rate of Nitrate Nitrogen, kg/ha/d

| **Land Use** | **Jan** | **Feb** | **Mar** | **Apr** | **May** | **Jun** | **Jul** | **Aug** | **Sep** | **Oct** | **Nov** | **Dec** | Mean |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Deciduous | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Mixed Forest | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Orchard | 0.000 | 0.000 | 0.000 | 0.123 | 0.245 | 0.245 | 0.245 | 0.245 | 0.245 | 0.123 | 0.000 | 0.000 | 0.123 |
| Coniferous | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Shrub / Scrub | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Grassland | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Park | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Golf Course | 0.000 | 0.000 | 0.000 | 0.960 | 0.960 | 0.960 | 0.960 | 0.960 | 0.960 | 0.000 | 0.000 | 0.000 | 0.480 |
| Pasture | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Farm 1[1] | 0.000 | 0.000 | 0.000 | 0.577 | 0.577 | 0.577 | 0.577 | 0.577 | 0.577 | 0.000 | 0.000 | 0.000 | 0.288 |
| Farm 2[2] | 0.288 | 0.288 | 0.288 | 0.577 | 0.577 | 0.577 | 0.577 | 0.577 | 0.577 | 0.288 | 0.288 | 0.288 | 0.433 |
| Farm 3[3] | 0.412 | 0.412 | 0.824 | 0.824 | 0.824 | 0.824 | 0.824 | 0.824 | 0.824 | 0.824 | 0.824 | 0.412 | 0.721 |
| Marsh | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Barren | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Water | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Residential | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 |
| High Density Residential | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 |
| Comm./Industrial | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 |

1. Farms at elevations above 600 m
2. Farms at elevations above 300 m but below 600
3. Farms in the floodplain at low elevations

Table 14: Monthly Unit Land Application Rate of Phosphorus, kg/ha/d

| **Land Use** | **Jan** | **Feb** | **Mar** | **Apr** | **May** | **Jun** | **Jul** | **Aug** | **Sep** | **Oct** | **Nov** | **Dec** | Mean |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Deciduous | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Mixed Forest | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Orchard | 0.000 | 0.000 | 0.000 | 0.061 | 0.123 | 0.123 | 0.123 | 0.123 | 0.123 | 0.061 | 0.000 | 0.000 | 0.061 |
| Coniferous | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Shrub / Scrub | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Grassland | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Park | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Golf Course | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Pasture | 0.033 | 0.033 | 0.033 | 0.033 | 0.033 | 0.033 | 0.033 | 0.033 | 0.033 | 0.033 | 0.033 | 0.033 | 0.033 |
| Farm 1[1] | 0.000 | 0.000 | 0.000 | 0.399 | 0.399 | 0.399 | 0.399 | 0.399 | 0.399 | 0.000 | 0.000 | 0.000 | 0.200 |
| Farm 2[2] | 0.200 | 0.200 | 0.200 | 0.399 | 0.399 | 0.399 | 0.399 | 0.399 | 0.399 | 0.200 | 0.200 | 0.200 | 0.299 |
| Farm 3[3] | 0.254 | 0.254 | 0.509 | 0.509 | 0.509 | 0.509 | 0.509 | 0.509 | 0.509 | 0.509 | 0.509 | 0.254 | 0.445 |
| Marsh | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Barren | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Water | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Residential | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 |
| High Density Residential | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 |
| Comm./Industrial | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 | 0.001 |

1 Farms at elevations above 600 m
2 Farms at elevations above 300 m but below 600
3 Farms in the floodplain at low elevations

## 4. Observed Water Quality

Surface water quality observations (nitrogen, phosphorus, sediment, pesticides, etc.) were retrieved from the STORET legacy data center (http://www.epa.gov/storpubl/legacy/gateway.htm) for the period January 1, 1971 through December 31, 1995. A station near the outflow of each catchment was selected, and the data set retrieved. Total suspended sediment measurements were originally retrieved from the STORET legacy data center but more continuous data was measured at the USGS NWIS stations. Using a text editor, we joined each catchment's STORET dataset with its corresponding USGS NWIS total suspended sediment values and imported the tables into WARMF and BASINS, as needed for the calibration of each model.

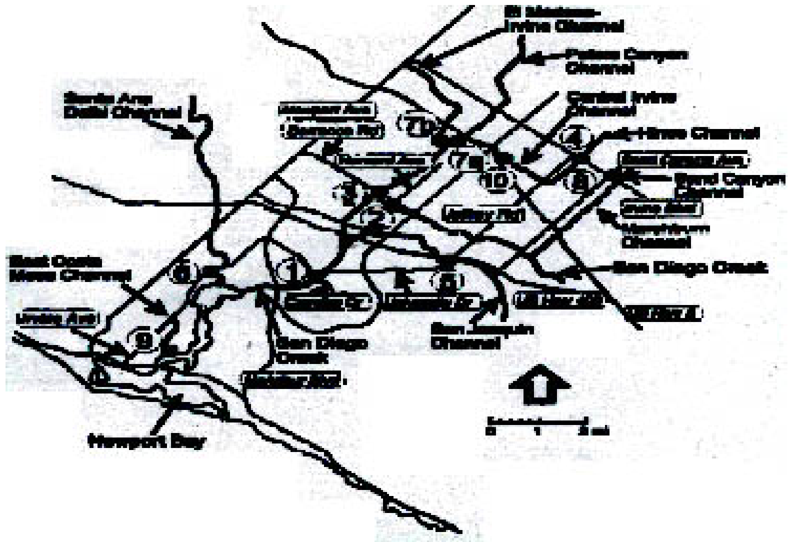
In addition, the recent USGS NAWQA study results have been providing us with useful observations to calibrate the models. In particular, the detailed study by Izbicki et al. (2000) which characterized the stormflow chemistry below Prado Dam from 1995 to 1998 has been of considerable value.

Observed pesticide concentrations vary seasonally, peaking in the fall, declining in the winter and rising again in the spring. High levels of Diazinon and Chlorpyrifos are discharged from commercial nurseries in the upper portions of the watershed during both stormflow and baseflow conditions. Stormflow occurs during precipitation events and is the overland flow that reaches the streams and flows into the Bay. Base flow is the dry season flow in the streams resulting from such sources as groundwater discharge, irrigation and urban uses.

A study by Lee and Taylor (2001) indicates that the entire Upper Newport Bay watershed contributes to the Diazinon loading, with less contribution from certain specific areas. The station at Campus drive (Figure 18, station 1 below) shows the highest loading rate, followed by agricultural areas such as the Sand Canyon Channel (station 8) and Central Irvine channel residential areas (station 10).

Figure 18. OP Pesticide Sampling Distribution.

Source: (Lee and Taylor, 2001)




The sampling stations were distributed throughout the watershed at a range of areas designed to characterize relationships between landuse and pesticide loading. These areas incorporated Peter's Canyon wash, Upper and Lower San Diego Creek but not the Bay itself. The sampling scheme is represented in Figure 8 and the sampling station attributes are shown in Table 15.

Table 15. Sampling Stations and Dominant Landuse.

| Station | Location | Dominant Landuse |
|---|---|---|
| 1 | San Diego Creek at Campus Drive | Mixed, Residential Ag., Nursery |
| 2 | San Diego Creek at Harvard Avenue | Mixed, Residential Ag., Nursery |
| 3 | Peters Canyon Channel at Barranca Pkwy | Mixed, Residential Ag., Nursery |
| 4 | Hines Channel at Irvine Blvd | Nursery, Ag. |
| 5 | San Joaquin Channel University Drive | Ag., Open Space |
| 6 | Santa Ana-Delhi Channel at Mesa Drive | Residential, Commercial |
| 7a | Peters Canyon Channel at Walnut Avenue | Residential, Ag., Nursery |
| 7b | El Modena Irvine Channel upstream of Peters Canyon Channel | Residential, Commercial |
| 8 | Sand Canyon Avenue-NE corner of Irvine Blvd | Ag. |
| 9 | East Costa Mesa Channel at Highlands Drive | Residential, Commercial |
| 10 | Central Irvine Channel at Monroe | Residential, Ag, Nursery |

Detection frequencies of Chlorpyrifos are higher along Peters Canyon Channel, with maximum concentrations found at Hines Channel (Station 4) where there is a large nursery (Lee and Taylor, 2001). Reach 1 of San Diego Creek had high concentrations as well corresponding to mixed landuse types at Campus, Coronado and Harvard streets. Chlorpyrifos was detected in all samples collected in Upper Newport Bay, though overall the concentrations were lower here than in the freshwater reaches of San Diego Creek and Peters Canyon Channel (Lee and Taylor, 2001).

Table 16 summarizes the sampling results by Lee and Taylor (2001) for Diazinon during the 1998 water year, conducted as part of a 391(h) study. The goal of the study was to characterize the extent of aquatic life toxicity derived from OP pesticides in Newport Bay. Samples were collected in the upper portions of the watershed and in Newport Bay to provide a composite representation of the detection frequency. The results show frequent detection of Diazinon in the surface water but infrequent detection in sediment samples. This is explained by Diazinon's tendency to partition into the aqueous phase more readily than to it adsorbs to sediments. Note that for Diazinon the CMC is 80 ng/L and the CCC is 50 ng/L per the California Dept. of Fish and Game, indicating a high risk to aquatic life. Note the extremely high peak concentrations. Based on results from the 319(h) Aquatic Life Toxicity study, both the average and median concentrations of Diazinon exceed the Criterion Chronic Concentration (CCC) and the Criterion Maximum Concentration (CMC). Diazinon was detected in stream channels 93% of the time and in Newport Bay in every sampling event conducted under the 319(h) study.

A study done in Alameda County (URS, 1999) found that Diazinon was frequently detected in fine-grained sediment samples. The Alameda study found that there was an inverse relationship between sampled grain sizes and the adsorption of Diazinon to sediments, with highest detections in clay soils. The discrepancy in results may be due to a lower fraction of fine-grained organic material in Newport Bay soils with respect to those in Alameda Creek watershed. Another possible explanation for the low detection of Diazinon in sediments is the heavy stream channelization of the Upper Newport Bay watershed. Channelization of streams decreases the channel roughness and transports water molecules rapidly through the extent of the channel before discharging them to the Bay. Turbulence caused during high velocity stormflows may cause Diazinon to be leached from sediments as it travels through the extent of the creek, increasing the likelihood of finding Diazinon in water samples but not in sediments.

Table 16. Newport Bay Watershed Diazinon Sampling Results (1998).
*Source*: Results after SARWQCB (2001).

| **Source** | **Detection Frequency** | **Min.** | **Max.** | **Avg.** | **Median** |
|---|---|---|---|---|---|
| **Water samples** | | (ng/L) | (ng/L) | (ng/L) | (ng/L) |
| Drainage Channels | 93% | <40 | 10,000 | 471 | 220 |
| Baseflow | 89% | <40 | 10,000 | 473 | 160 |
| Stormflow | 98% | <50 | 7,990 | 451 | 357 |
| Upper Newport Bay | 100% | 197 | 720 | 386 | 357 |
| **Sediment samples** | | (ug/kg) | (ug/kg) | (ug/kg) | (ug/kg) |
| Drainage Channels | 2% | <10 | 49 | --- | --- |
| Newport Bay | 3% | <0.4 | 60 | --- | --- |

In contrast to Diazinon, Chlorpyrifos is consistently found in the sediments and detected in less than half of the 200 drainage channel samples taken under the 319(h) study (Table 17). This is likely a result of its' high adsorption coefficient. Chlorpyrifos is also less mobile in the environment, which indicates that it is more likely than Diazinon to remain on site after application.

Table 17. Newport Bay Watershed Chlorpyrifos Sampling Results (1998).
*Source*: Results after SARWQCB (2001).

| Source | Detection Frequency | Min. | Max. | Avg. | Median |
|---|---|---|---|---|---|
| **Water samples** | | (ng/L) | (ng/L) | (ng/L) | (ng/L) |
| Drainage Channels | 45% | ND | 770 | 139 | <50 |
| Baseflow | 35% | ND | 670 | 162 | <40 |
| Stormflow | 56% | ND | 770 | 123 | 50 |
| Upper Newport Bay | 100% | 2 | 132 | 43.3 | 41.5 |
| **Sediment** (ug/kg) | | | | | |
| Drainage Channels | 100% | 17 | 29 | 23 | 23 |

| Freshwater | CDFG | EPA |
|---|---|---|
| CCC | 14 | 41 |
| CMC | 20 | 83 |
| Saltwater | | |
| CCC | 9 | 5.6 |
| CMC | 20 | 11 |

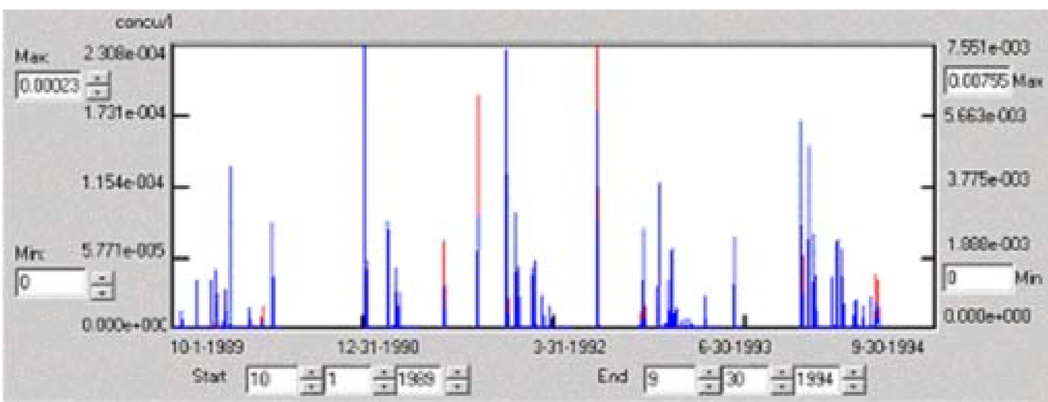
The median concentrations of Chlorpyrifos in water and sediment obtained during baseflow and stormflow exceeded the CDFG's and EPA's CCC and CMC values for freshwater and saltwater. The Chlorpyrifos samples indicate that the pesticide is less mobile in the environment than Diazinon, and reinforce the role of sediment transport in pesticide-related toxicity.

## 5. Simulation Results

To date we have calibrated the BASINS and WARMF models for OP pesticide loading and we are in the process of fine tuning the nutrient loading to match the observed water quality in these regions, and thus be able to discern the influence of various land uses on observed water quality, and then be able to make predictions about the potential changes of water quality over time as land use continues to be modified over the next decades.

Using the BASINS model, we found that the maximum and minimum concentrations were within the range of values observed by Lee and Taylor (2001) as well as Izbicki et al. (2000). The greatest simulated Chlorpyrifos concentration was 0.00075 mg/L and the Diazinon concentration was 0.0122 mg/L. Baseline Chlorpyrifos concentrations ranged between 0 and 0.00003 mg/L, and baseline Diazinon concentrations ranged between 0 and 0.002 mg/L. A simulation of constituent concentration is presented in Figure 19. Fluctuations in constituent concentration are shown to be strongly correlated to fluctuation in stream flow. Peak concentrations occur at the beginning of each storm event, and concentrations often decrease to zero between storm events.

Figure 19. Simulated (BASINS) constituent concentration for Baseline Scenario.
Chlorpyrifos concentration (mg/L) is in red on the left axis
Diazinon concentration (mg/L) is in blue on the right axis.




For the WARMF model, Diazinon and Chlorpyrifos were continuously simulated from November 1, 1989 through April 30, 1995. This date range was chosen due to the availability of meteorological data and because the date range overlapped that of the landuse information. Figure 20 illustrates the simulated concentration (mg/L) of Diazinon in lower San Diego Creek and Figure 21 shows the simulated concentration of Chlorpyrifos (mg/L) in lower San Diego Creek. Neither pesticide reaches a concentration above 0.117 mg/L. Both pesticides peak in time with stormflow events, though to a lesser degree.

The simulated data was compared against observed ranges and median concentrations found in the literature in order to determine the accuracy with which the model was predicting the natural watershed. The simulated median concentration of Diazinon is within the same order of magnitude as the observed median baseflow. However, the maximum simulated Diazinon concentration was at least two orders of magnitude higher than the observed median stormflow concentration and one order of magnitude greater than maximum concentrations measured in agriculture and urban runoff. The median simulated concentration of Chlorpyrifos in lower San Diego Creek was at least one order of magnitude greater than the observed median baseflow concentration. The maximum Chlorpyrifos concentration output by the model was four orders of magnitude greater than the median observed stormflow concentration and three orders of magnitude greater than surface flow maximum measurements taken off of urban and agriculture land uses.

Figure 20. Simulated Diazinon Concentrations (mg/L) for Lower San Diego Creek from November 1, 1989 Through April 30, 1995.

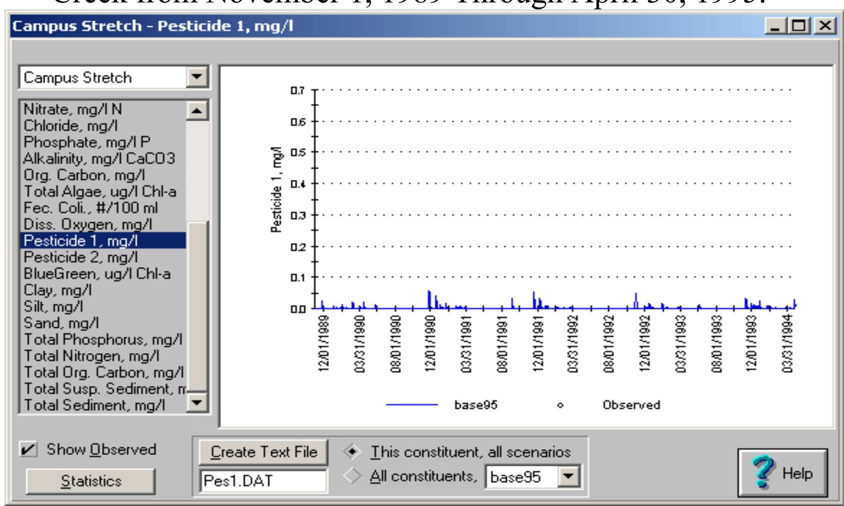
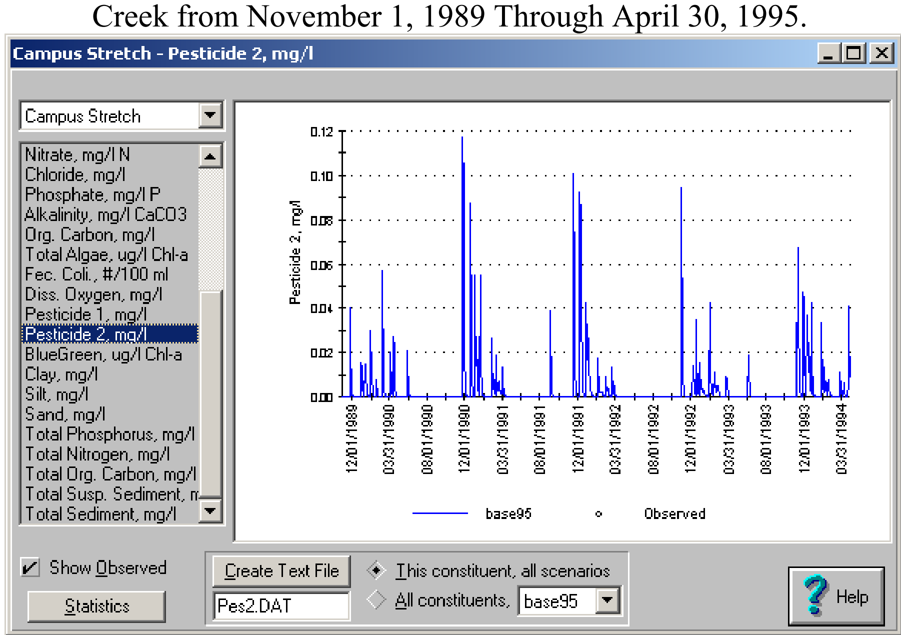



Figure 21. Simulated Chlorpyrifos Concentrations (mg/L) for Lower San Diego Creek from November 1, 1989 Through April 30, 1995.



The first step in analyzing our simulations was to examine the baseline scenario. Results from the baseline, or no change scenario, indicate that critical concentration levels for Diazinon would consistently be exceeded. Over the 5-year simulation, the CCC was exceeded on 12 to 34 days per rain period, and the CMC was exceeded on 12 to 31 days per rain period (Table 18). In contrast, Chlorpyrifos concentrations appeared to be within the criterion limits more often. Chlorpyrifos concentrations exceeded the CCC levels between 1 and 4 days per rain period, and the CMC levels between 0 and 3 days per rain period. This analysis confirms that further management strategies are required to sufficiently reduce pesticide concentrations to meet criterion regulations. Chlorpyrifos does not pose as serious a problem as Diazinon.

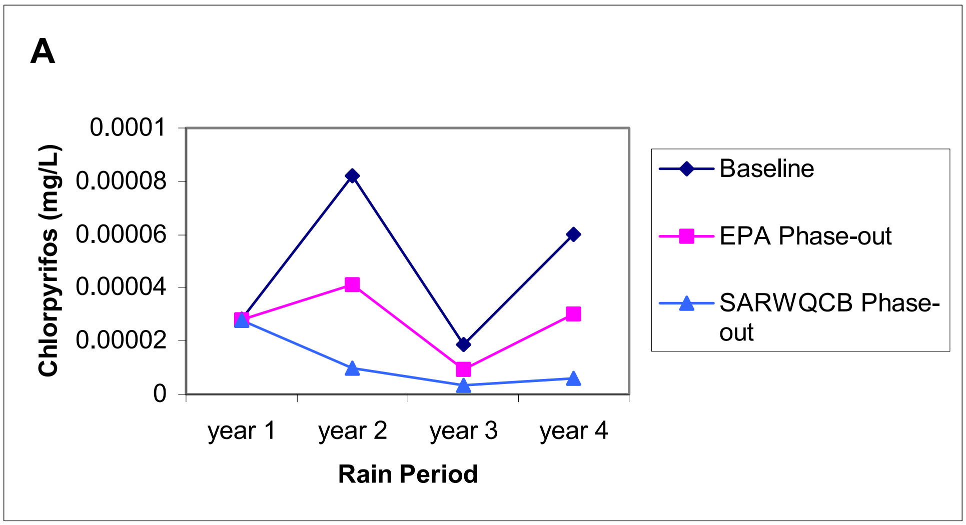
Table 18. Baseline Scenario Indicating Number of Days during Simulated Rain Period Concentrations Exceed the CCC and CMC Levels.

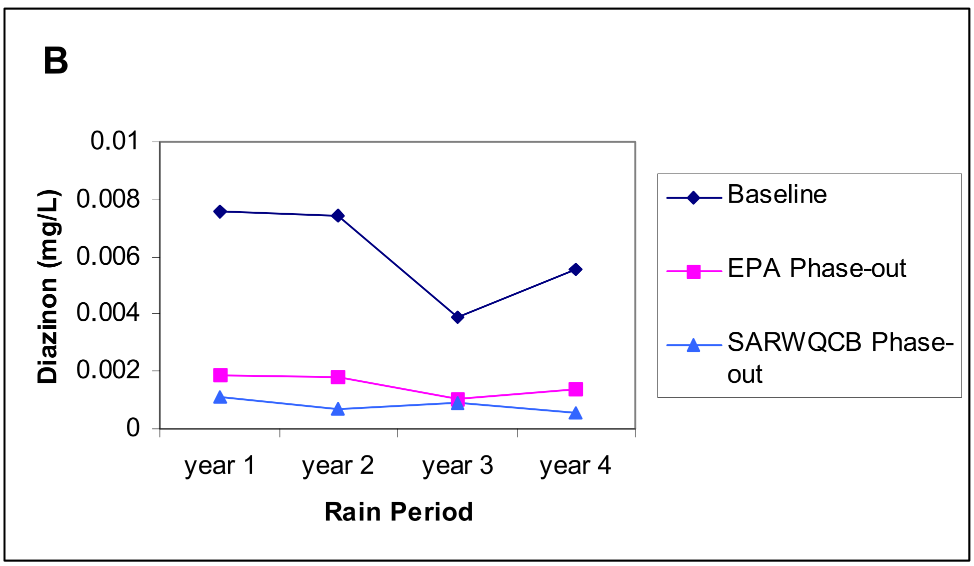
| | | **Chlorpyrifos** | | **Diazinon** | |
|---|---|---|---|---|---|
| | | **Days in exceedence** | | **Days in exceedence** | |
| | | **CCC** | **CMC** | **CCC** | **CMC** |
| Baseline | | | | | |
| | year 1 | 4 | 3 | 12 | 12 |
| | year 2 | 2 | 2 | 29 | 26 |
| | year 3 | 1 | 0 | 34 | 31 |
| | year 4 | 2 | 1 | 23 | 22 |
| EPA Projected Phase-out | | | | | |
| | year 1 | 4 | 3 | 11 | 11 |
| | year 2 | 1 | 1 | 24 | 23 |
| | year 3 | 0 | 0 | 24 | 19 |
| | year 4 | 1 | 1 | 20 | 15 |
| SARWQCB Projected Phase-out | | | | | |
| | year 1 | 4 | 3 | 11 | 11 |
| | year 2 | 0 | 0 | 21 | 17 |
| | year 3 | 0 | 0 | 16 | 10 |
| | year 4 | 0 | 0 | 14 | 11 |

We then compared the baseline scenario to phase-out scenarios predicted by the EPA and the SARWQCB. The parameter that was compared was the maximum constituent concentration resulting from each simulated period (4 rain years). The simulated concentrations were found to be closely correlated to the reduction in application rates for pesticide applied to urban and Built-up land uses. In the EPA phase-out scenario, the predicted Chlorpyrifos concentrations were approximately 50% lower than concentrations simulated in the baseline scenario (Figure 22A). Diazinon concentrations were predicted to be approximately 75% less in the EPA phase-out scenario than in the baseline scenario (Figure 22B). For both pesticides, these reductions were correlated to the reduction in pesticide application rates to urban and Built-up landuse.

A comparison of the baseline scenario to the phase-out scenario predicted by the SARWQCB indicated similar trends. During the simulated period, predicted pesticide concentrations in the phase-out scenario were approximately 90% less than in the baseline scenario. By the end of the five-year simulation, pesticide concentrations were reduced in the phase-out scenarios proportionate to their prescribed reduction in application rates to urban and Built-up landuse. Although the two agencies predict different reductions, each simulation resulted in reduced pesticide concentrations in the streams entering the Newport Bay that were approximately the same as the reduction in application rate to the urban and Built-up landuse.

Figure 22A and 22B. Comparison of simulated maximum (stormflow) pesticide concentrations between the baseline scenario and the phase-out scenarios predicted by the EPA and the SARWQCB. Figure 22A presents the maximum simulated Chlorpyrifos concentration during each rain period and Figure 22B presents the maximum simulated Diazinon concentration during each rain year



A noteworthy difference between the two OP pesticides is that one year after initiation of the phase-out, maximum Chlorpyrifos concentrations are approximately the same for all scenarios (baseline = 2.82 x $10^{-5}$ mg/L; EPA phase-out = 2.78 x $10^{-5}$ mg/L; and SARWQCB phase-out = 2.76 x $10^{-5}$ mg/L), whereas predicted Diazinon concentrations had already decreased by 75% in the EPA phase-out scenario, and 85% in the SARWQCB phase-out scenario (baseline = 7.55 x $10^{-3}$ mg/L; EPA phase-out = 1.83 x $10^{-3}$ mg/L; SARWQCB phase-out = 1.11 x $10^{-3}$ mg/L). This comparison of the baseline scenario to the phase-out scenarios provides evidence of the differences in fate and transport between Chlorpyrifos and Diazinon. As discussed above, Diazinon is more soluble than Chlorpyrifos, and Chlorpyrifos has a greater tendency to adsorb to sediment than Diazinon. Results from the model simulations reflect these tendencies. It takes longer for Chlorpyrifos to be removed from the system. There is a lag time of approximately one year before reductions in the application of Chlorpyrifos results in decreased concentrations. On the contrary, Diazinon concentrations reflect the phase-out application reductions in a shorter period of time (Figure 22B). One-year after implementation of the phase-out, Diazinon concentrations were already reduced by 75%. Because Diazinon has a relatively high solubility, it will dissolve in water and is flushed through the system more quickly than Chlorpyrifos.

One of the primary objectives of this analysis is to determine the significance of different source contributions corresponding to specific landuse classes throughout the watershed. Our analysis used the data from the 208(j) and 319(h) studies, which found that surface runoff is the source of virtually all of the export loadings, while atmospheric deposition, sediment remobilization and groundwater sources are insignificant. About 3 kg of Chlorpyrifos and 16 kg of Diazinon are annually discharged to Upper Newport Bay via the creeks. This amounts to less than 0.025 percent of the applied Chlorpyrifos mass, and about 0.3 percent of the applied Diazinon mass in the Newport Bay Watershed. Results from Lee and Taylor (2001) found that on average, about 0.5 to 1 kg of Diazinon and 0.4 to 0.8 kg of Chlorpyrifos are discharged to the Upper Bay during a typical storm event of 25 to 50 mm.

Runoff derived from urban landuses accounted for 88% of the Diazinon baseflow and 96% of the stormflow load, with agricultural sources accounting for the rest of the load. For Chlorpyrifos, runoff derived from urban landuses accounts for about 85% of the baseflow and stormflow loads, while agriculture (including nurseries) accounts for about 15% of the load. Diazinon concentrations in San Diego Creek exceed the CCC both during baseflow and during stormflow. The stormflow Chlorpyrifos concentrations in San Diego Creek exceed the CCC, however the baseflow is non-detectable. Chlorpyrifos samples collected from Newport Bay exceed the CDFG saltwater CCC. These concentrations occur in a freshwater lens that persists for several days during storm events (SARWQCB, 2001).

The OP pesticide usage restrictions currently being implemented will likely eliminate a significant fraction of current Diazinon use in the Newport Bay watershed. If runoff concentrations show a corresponding decline, OP pesticide concentrations in San Diego Creek may drop below the EPA and CDFG CMC and CCC values for freshwater and saltwater. However, it is uncertain whether the partial phase-out will be fully effective, or even whether a successful partial phase–out will result in acceptable concentrations.

## 6. Products to date from this project

As part of this project, we have produced the following deliverables:

Master's Thesis: Lee Harrison, Meighan Jackson, Giles Pettifor, Linda Purpus, Jot Splenda, Sarah White. *Analysis of Alternative Watershed Management Strategies Addressing Aquatic Toxicity: A case study of Organophosphate Pesticide Loading in Newport Bay, CA*. Bren School of Environmental Science & Management. Committee in charge: Professors James Frew and Arturo Keller. June 2002.

Robinson, Timothy H., Al Leydecker, John M. Melack and Arturo A. Keller. 2002. *Nutrient Concentrations in Southern California Streams related to landuse*. Coastal Water Resources, AWRA 2002 Spring Specialty Conference Proceedings, Lesnick, John R. (Editor). American Water Resources Association, Middleburg, Virginia, TPS-02-1, pp 339-343.

Presentation at California and the World Ocean '02:

- *Evaluation of the Impact of and Management Strategies for Diazinon and Chlorpyrifos in Newport Bay***,** by Arturo Keller, Lee Harrison, Meighan Jackson, Giles Pettifor, Linda Purpus, Jot Splenda, Sarah White

Presentations at the American Water Resources Association's 2003 International Water Congress on Watershed Management for Water Supply Systems, to be held in New York:

- *Effect of the temporal scale of precipitation on water quality simulation*, by Yi Zheng and Arturo Keller
- *Understanding the uncertainties in a TMDL calculation*, by Arturo Keller

We are currently preparing manuscripts for submission to peer-reviewed journals for the OP pesticide study in Newport Bay, and will produce similar manuscripts for the nutrient and pesticide analysis of the Lower Santa Ana River Basin.

The project has resulted in the training (partial funding) of two Ph. D. students (Yi Zheng and Tim Robinson), who continue the work with the WARMF and BASINS models, as well as six Master's students (Lee Harrison, Meighan Jackson, Giles Pettifor, Linda Purpus, Jot Splenda, Sarah White) who used the Newport Bay area and modeling tools as a focus of their study. Other Master's students (Ian Adam, Emily Taylor, Ben Pitterle) have also contributed to parts of the study.

The project also has generated a voluminous amount of information, which has been incorporated into a graduate level course, namely ESM 595D Watershed Quality Modeling & Management, which trains about 10 graduate students per year on the implementation and use of watershed models.

We also had fruitful discussions with John Izbicki from the USGS NAWQA study in the Santa Ana River Basin, to understand in more detail the stormflow chemistry of this system. John Izbicki presented a related seminar at the Bren School of Environmental Science & Management in 2002.

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# Bioavailability of Particle-Associated Pesticides in Northern San Francisco Bay

## Basic Information

| | |
|---|---|
| **Title:** | Bioavailability of Particle-Associated Pesticides in Northern San Francisco Bay |
| **Project Number:** | 1999CA0009G |
| **Start Date:** | 9/1/1999 |
| **End Date:** | 12/30/2002 |
| **Funding Source:** | 104B |
| **Congressional District:** | 43 |
| **Research Category:** | Water Quality |
| **Focus Category:** | Agriculture, Solute Transport, |
| **Descriptors:** | agriculture, contaminant transport, herbicides, insecticides, invertebrates, pesticides, suspended sediments |
| **Principal Investigators:** | Donald Weston |

## Publication

1. none

**Bioavailability of particle-associated pesticides in the northern San Francisco Bay**

Principal investigator: Donald Weston, University of California, Berkeley, CA
USGS Collaborator: Kathryn Kuivila, USGS, Sacramento, CA

Progress report: January 30, 2003

Progress:
Recent studies have shown that pulses of pesticides from agricultural runoff have been detected in the major rivers of the San Francisco Estuary after heavy rains, and that a significant part of the pesticides are found associated with suspended particles in concentrations that exceed predictions based on equilibrium partitioning. The aim of this project was to determine if particle-bound pesticides were desorbable and bioavailable in concentrations high enough to be toxic to sensitive aquatic species.

Three sampling sites were chosen: one each in the Sacramento River, the San Joaquin River, and the Napa River. Sampling was done after rainstorms in February 2001 and 2002, and during the summer of 2002. Sediment samples were tested for toxicity using Chironomus tentans. The sediments were also extracted with water, and the water tested for toxicity with Ceriodaphnia dubia and Raphidocelis subcapitata (formerly Selenastrum capriconutum). Generally speaking, instances of toxicity were rare in the summer sampling period and the winter of 2002 when rainfall events tended to be fairly small and localized. The most widespread toxicity was seen after the February 2001 storm when suspended sediment collected at all three sites caused elevated mortality in the midge C. tentans. When these same suspended sediments were extracted with water, and the water tested with C. dubia, chronic toxicity (reproductive impairment), but no acute mortality, was seen at all three sites. The water was also tested with R. subcapitata, an algae that would be sensitive to herbicide toxicity, but no effect on the algae was seen. Unlike the suspended sediment tests, bedded sediments, in general, showed only infrequent toxicity at most sites and in most sampling events.

Several insecticides and herbicides were present in measurable concentrations on the suspended and/or bedded sediment, including eptam, molinate, diazinon, disoulfoton, butylate, dimethoate, azinphos methyl, oxyflurofen, piperonyl butoxide, bifenthrin, permethrin, and DDD. However, concentrations were generally low (<100 ppm), and there were no clear patterns that implicated the measured analytes as responsible for the observed toxicity.

Attempt to use the digestive fluid of a polychaete as a biologically realistic extractant in order to quantify the bioavailable fraction were generally unsuccessful. While a few compounds were detected in the gut fluid after extracting sediments (carbaryl, chlorpyrifos and diazinon), concentrations were consistently near detection limits. While the digestive fluid approach has shown considerable promise with other contaminant types (e.g. PAH, PCB) in other studies, the relatively low pesticide concentrations on these sediments and the small amounts of sediment and fluid used in the extractions created considerable difficulties with analytical detection limits.

In summary, our results suggest that suspended sediment particles present in the San Joaquin, Napa and Sacramento rivers may account for a significant part of the ambient toxicity reported in the San Francisco Estuary after rain events, but the specific contaminants responsible for this toxicity remain unidentified. Suspended sediment present in the rivers after heavy winter rains appear to be of greater concern than that present during the dry season.

Student supported:

The majority of the research conducted at UC Berkeley was performed by a Post-doctoral Research Associate, Jonas Gunnarsson. Several UC Berkeley undergraduate students participated at various times throughout project duration.

Publications:

A publication, jointly authored by the UC Berkeley investigators and USGS collaborators, in now in preparation.

Presentation:

Presentations resulting from this project were given at:

Society for Environmental Toxicology and Chemistry Annual Meeting in 2001 and 2002.

Northern California Chapter of the Society for Environmental Toxicology and Chemistry in 2001.

The CALFED Science Conference, Sacramento, CA in 2003.

# California Water Resources Center WRIP

## Basic Information

| | |
|---|---|
| **Title:** | California Water Resources Center WRIP |
| **Project Number:** | 2001CA3541B |
| **Start Date:** | 3/1/2001 |
| **End Date:** | 2/28/2002 |
| **Funding Source:** | |
| **Congressional District:** | 43 |
| **Research Category:** | |
| **Focus Category:** | Water Quality, Water Use, Solute Transport |
| **Descriptors:** | watershed, riparian, hydrology, viruses, water management,Water quality |
| **Principal Investigators:** | John Letey, William Yeh, Thomas Meixner |

## Publication

# Error in uploaded file

Unable to read fy2001_annual_report.pdf for 2001CA3541B as well-formed .PDF

# Is Urban Runoff a Source of Human Pathogenic Viruses to Recreational Beach Waters?

## Basic Information

| | |
|---|---|
| **Title:** | Is Urban Runoff a Source of Human Pathogenic Viruses to Recreational Beach Waters? |
| **Project Number:** | 2001CA3841B |
| **Start Date:** | 3/1/2001 |
| **End Date:** | 2/28/2002 |
| **Funding Source:** | 104B |
| **Congressional District:** | 43 |
| **Research Category:** | Biological Sciences |
| **Focus Category:** | Water Quantity, Non Point Pollution, Surface Water |
| **Descriptors:** | RT-PCR, PCR, Hepatitis A virus, enterovirus, adenovirus,viruses, urban runoff, bacteria |
| **Principal Investigators:** | Sunny Jiang |

## Publication


1. Sunny C. Jiang and Weiping Chu, Urban Runoff as a Source of Human Viral Contamination to Southern California Recreational Beach Waters, Water Research, submitted

# Is Urban Runoff a Source of Human Pathogenic Viruses to Recreational Beach Waters?

**Sunny C. Jiang**
**Environmental Analysis and Design**
**UC Irvine**

## ***Project Summary***

To assess the impact of urban runoff on the coastal water quality, water samples were collected at 21 sites in 11 rivers and creeks along the Southern California coast for bacteriological and virological examinations during summer 2000. The water samples were tested for the presence of three types of human viruses (adeno, entero and hepatitis A viruses) using nested- and RT-PCR methods. In addition, they were tested for three types of fecal indicator bacteria (total coliform, fecal coliform and enterococcus) as well as the somatic and F-specific coliphage.

Human viruses were detected in sample volume as low as 105 ml of water. Hepatitis A viruses were the most frequently found, and were detected in 81% of water samples we collected at the urban rivers of Southern California. All of sites that tested positive for adenovirus (52%) were also positive for entero and hepatitis A viruses. The fecal indicator bacteria were found in all of the water samples examined. There was, however, not a clear relationship between fecal indicator concentrations and the presence of human viruses in the water. At the site where the quality of its water ranked second best in terms of the bacterial indicators, all three types of human viruses were detected.

To understand the seasonal dynamics of pollutant loads from the urban runoff, water samples were collected at the mouths of the Los Angeles River, the San Gabriel River, and the Santa Ana River during both the wet and dry seasons. Human viruses were most frequently found at the mouth of the Los Angeles River. In general, both fecal indicator bacteria and human viral densities of the water at the river mouths were associated with storm events. The first storm of the wet season most likely carried a heavy pathogen load and were expected to impact the coastal water quality more than subsequent storm events. Urban runoff is a major contributor to coastal water pollution.

## ***Publications***

Sunny C. Jiang and Weiping Chu, Urban Runoff as a Source of Human Viral Contamination to Southern California Recreational Beach Waters, *Water Research, submitted*

## ***Professional Presentations***

Sunny Jiang, Pacific Rim Shellfish Conference. San Diego, California. April 4-6, 2001

Sunny Jiang, Ballona Wetland Foundation Conference. Los Angeles, California. May 17-18, 2001

Sunny Jiang, 101st Annual meeting of American Society for Microbiology. Orlando, Florida. May 20-24, 2001

Sunny Jiang, 82nd Annual meeting of American Association for the advancement of Science, Pacific Division, Irvine, California, June 17-20, 2001

Sunny Jiang and Weiping Chu, American Society of Limnology and Oceanography Summer Meeting, Victoria, British Columbia, Canada, June 10-14, 2002

## ***Student Training***

Sam Choi, Graduate; Environmental Analysis and Design, UC Irvine.
Miyuki Fujita, Undergraduate, Applied Ecology, UC Irvine
Clifford Tse, Undergraduate, Applied Ecology, UC Irvine
Elaine Jacinto, Undergraduate, Applied Ecology, UC Irvine
Joanne Choe, Undergraduate, Applied Ecology, UC Irvine
Jennifer Cheng, Undergraduate, Applied Ecology, UC Irvine
Desiree Eakin, Undergraduate, Applied Ecology, UC Irvine
Amana Rafigue, Undergraduate, Applied Ecology, UC Irvine
Dalisa Tran, Undergraduate, Applied Ecology, UC Irvine
Kevan Savage, Undergraduate; Applied Ecology, UC Irvine.

## ***Additional Funding***

$400,000. “Real time PCR detection of Human viruses and indicators in water”. Water Environment Research Foundation. March 1, 2002 - February 30, 2004.
$36,179. “Determining the relationship between fecal indicators and human pathogenic viruses in Newport Bay Watershed”. August 15, 2001 - August 14, 2002. Contract with City of Newport Beach

## ***Collaborative Efforts***

County of Orange, California. Investigation of pollution source at a small watershed of Southern California.

# Landscape Level Controls on Nitrate-Nitrogen in Forested and Chaparral Catchments of Southern California

## Basic Information

| | |
|---|---|
| **Title:** | Landscape Level Controls on Nitrate-Nitrogen in Forested and Chaparral Catchments of Southern California |
| **Project Number:** | 2001CA3901B |
| **Start Date:** | 3/1/2001 |
| **End Date:** | 2/28/2002 |
| **Funding Source:** | |
| **Congressional District:** | 43 |
| **Research Category:** | |
| **Focus Category:** | Water Quality, Nitrate Contamination, Management and Planning |
| **Descriptors:** | landscape hydrology, nonpoint source pollution, atmospheric deposition, biogeochemistry, drinking water, nitrogen retention, watershed management,water quality |
| **Principal Investigators:** | Thomas Meixner |

## Publication


1. Meixner, T, M., E. Fenn, and M. A. Poth, Nitrate in Polluted Mountainous Catchments With Mediterranean Climates In Optimizing Nitrogen Management in Food and Energy Production and Environmental Protection, Proceedings of the 2nd International Nitrogen Conference on Science and Policy. The Scientific World, 1, DOI 10.1100/tsw.2001.324.
2. Meixner T., E. B. Allen, K Tonnessen, M. Fenn, and M. Poth, Atmospheric Nitrogen Deposition: Implications for Park Managers of Western U.S. Parks, Park Science, 21(2), p. 30-33, 2002.
3. Meixner, T., and M. E. Fenn, Riparian Areas as Biogeochemical Filters, Mediterranean Climates and the Telescoping Ecosystem, A Cross-Cultural Perspective on Current Problems in Ecosystem and Natural Resources Management: An International Course on Issues Related to Ecosystem Management. Ed. J. Tenhunen, University of Bayreuth, Bayreuth, Germany, 2002.

# Landscape Level Controls on Nitrate-Nitrogen in Forested an Catchments of Southern California

**Dr. Thomas Meixner**
**Environmental Sciences**
**UC Riverside**

**Mark Fenn**
**US Forest Service Forest Fire Laboratory**
**Riverside**

## ***Project Summary***

The mountains of Southern California receive amongst the highest rates of anthropogenic N deposition in the world (~40 kg $ha^{-1}$ $yr^{-1}$) and as a result stream water $NO_3^-$ concentrations in smog-impacted summer-dry montane ecosystems in the Los Angeles air basin are the highest for natural catchments in North America. The localized nutrient enrichment in the mountains surrounding the Los Angeles metropolitan area may be the precedent for the future of forests and other ecosystem types near urbanizing areas in the western United States, as emissions of $NO_x$ and $NH_3$ increase with urban expansion.

Terrestrial ecosystems with semiarid climates have limited capacity to process and retain chronic inputs of N. Available data indicates that stream flow from watersheds under heavier influences of the smog generated in Los Angeles have higher $NO_3^-$ concentrations than those that are farther away. However, the $NO_3^-$ concentrations of stream flows in a watershed are extremely variable. The stream flow through the Devil Canyon catchment in the western San Bernardino Mountains, 100 km east of Los Angeles is a case in point. Although aerial N deposition should be similar throughout the Devil Canyon watershed, $NO_3^-$ concentrations vary by several orders of magnitude among the sampling sites. The spatially varied distribution of $NO_3^-$ in stream flow provides a unique opportunity to investigate the landscape scale dynamics of biogeochemical and hydrologic processes that exert the greatest control on $NO_3^-$ export from semi-arid forested catchments with elevated N deposition. We conducted detailed water quality sampling at eight streams in the Devil Canyon watershed. So far we have been able to reach several conclusions from our observations.

1. The $NO_3^-$ and dissolved organic carbon (DOC) concentrations of water increase as the stream flow increases. In a couple of the smaller streams, there is a noticeable first “flush” of $NO_3^-$ at the onset of the winter rainy season and then it is followed by a drop in $NO_3^-$ concentrations as stream flow continues at a level higher than the base flows of the summer and fall. The increase in $NO_3^-$ with stream flow in the larger streams at the commencement of the rainy season may also indicate a flushing process. However, we do not observe a decrease in $NO_3^-$ concentrations as the rainy season progresses indicating that a flushing process is not so apparently responsible for the increases of $NO_3^-$ in the larger streams.

2. The strong correlation of DOC and $NO_3^-$ concentrations may indicate a denitrification control on $NO_3^-$ input to streams. The concept of a denitrification control on stream nitrate and DOC concentrations is further bolstered by results of longitudinal surveys and mass balance analyses that indicating plant uptake and denitrification in the riparian zone, rather than a mass dilution process, are responsible for the decline in $NO_3^-$ concentrations.

3. Perennial streams have high $NO_3^-$ concentrations while ephemeral streams do not. This difference points to groundwater as the source of the high levels of $NO_3^-$ we observe in the perennial streams. Geochemical mixture modeling for the watershed indicates that the perennial streams of the watershed are dominated by groundwater seeping to the surface in all seasons. The mixture modeling also indicate a disconnect between the streams of the watershed and the surrounding landscape since stream composition bears little resemblance to soil water from zero-tension lysimeters in the watershed. Furthermore, the evidence indicates a decoupling of the impact of N deposition on terrestrial and aquatic systems in Mediterranean climates. The primary reason for the decoupling involves the asynchrony between when atmospheric deposition occurs (summer), the time period of maximum soil $NO_3^-$ availability and leaching (winter), and the time of maximum plant N demand (spring).

Our results have important implications for wildlife and water resources management agencies as they respond to the adverse impacts of atmospheric N deposition on water quality. For wildlife managers the findings indicate that the streams with the best habitat, those with large and consistent flows, are those most likely to be impacted by the effects of N deposition. For water resource managers the results indicate that the times when they are most likely to get water for recharge or for filling reservoirs, periods of high flow, are also the periods which are expected to have the highest nitrate concentrations indicating less of a chance to use waters draining deposition impacted watersheds to dilute groundwater impacted by historic agricultural groundwater contamination.

## ***Publications***

Meixner, T, M., E. Fenn, and M. A. Poth, Nitrate in Polluted Mountainous Catchments With Mediterranean Climates In Optimizing Nitrogen Management in Food and Energy Production and Environmental Protection, *Proceedings of the 2$^{nd}$ International Nitrogen Conference on Science and Policy. The Scientific World,* 1, DOI 10.1100/tsw.2001.324.
Meixner T., E. B. Allen, K Tonnessen, M. Fenn, and M. Poth, Atmospheric Nitrogen Deposition: Implications for Park Managers of Western U.S. Parks, *Park Science*, 21(2), p. 30-33, 2002.

Meixner, T., and M. E. Fenn, Riparian Areas as Biogeochemical Filters, Mediterranean Climates and the Telescoping Ecosystem, A Cross-Cultural Perspective on Current Problems in Ecosystem and Natural Resources Management: An International Course on Issues Related to Ecosystem Management. Ed. J. Tenhunen, University of Bayreuth, Bayreuth, Germany, 2002.

## ***Professional Presentations***

J. McGovern, The Role of Hyporheic and Riparian Processes on Controlling Nitrogen Flux in a Mediterranean-Type Montane Environment, Chemical and Environmental Engineering Annual Graduate Student Conference, Riverside, California, August 28, 2001.

T. Meixner, Nitrate in Polluted Mountainous Catchments With Mediterranean Climates Abbreviated title- Mediterranean Climate Nitrate, In Optimizing Nitrogen Management in Food and Energy Production and Environmental Protection: Proceedings of the 2$^{nd}$ International Nitrogen Conference on Science and Policy, Potomac, Maryland, October 17, 2001.

T. Meixner, Aquatic Impacts of Atmospheric Deposition in a Mediterranean Climate: The Asynchrony Hypothesis, CEA-CREST Annual Conference, Pasadena, California, May 8, 2002.

T. Meixner, Riparian Areas as Biogeochemical Filters, Mediterranean Climates and the Telescoping Ecosystem, A Cross-Cultural Perspective on Current Problems in Ecosystem and Natural Resources Management: An International Course on Issues Related to Ecosystem Management, Berchtesgaden, Germany, May 23, 2002.

## ***Student Training***

Jeff McGovern, MS student, Chemical and Environmental Engineering, UC Riverside.
Bridgette Valeron, Undergraduate, Environmental Sciences, UC Riverside.
Julie Quinn, Undergraduate, Environmental Sciences, UC Riverside.
Mathias Schmuck-Wakefield, Undergraduate, Environmental Sciences, UC Riverside.

## ***Additional Funding***

National Science Foundation, August 2001-July 2004, UC Riverside portion $85,000.
Environmental Protection Agency, July 2001-June2004, $450,000

UC-Multi-campus Research Incentive Fund, July 2001-July 2002, UC Riverside portion $3,300, UC San Diego portion - $11,700

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# Hydrodynamics of Shallow Water Habitats in the Sacramento-San Joaquin Delta

## Basic Information

| | |
|---|---|
| **Title:** | Hydrodynamics of Shallow Water Habitats in the Sacramento-San Joaquin Delta |
| **Project Number:** | 2001CA3961B |
| **Start Date:** | 3/1/2001 |
| **End Date:** | 2/28/2002 |
| **Funding Source:** | 104B |
| **Congressional District:** | 43 |
| **Research Category:** | Engineering |
| **Focus Category:** | Hydrology, Wetlands, Solute Transport |
| **Descriptors:** | biota, sediments, hydrodynamics,ecosystems |
| **Principal Investigators:** | Mark Stacey, Laurence R. Costello |

## Publication


1. Baek, S. and Stacey, M., Steady-state salinity of shallow water habitats adjacent to a tidal channel, Journal of Hydraulic Engineering. In Press.

# Hydrodynamics of Shallow Water Habitats in the Sacramento-San Joaquin Delta

**Mark Stacey**
**Civil and Environmental Engineering**
**UC Berkeley**

## ***Project Summary***

The exchange dynamics between shallow water habitats in the Sacramento-San Joaquin Delta (the Delta) and the adjoining channels have been analyzed using numerical simulations. These modeling activities focused on: (1) determining the salinity of shallow-water habitats adjacent to tidal channels and how the salinity depends on the geometry of the opening; and (2) exploring how temperature variations in the Delta alter these channel-shallow exchanges.

### ***Shallow-water Habitat Salinities***

The development of a shallow water habitat adjacent to northern San Francisco Bay (the North Bay) usually begins with the creation of an opening in a levee adjoining a tidal channel in the bay. For ecological considerations, the resulting salinity of the shallow water habitat is of critical concern. One difficulty in designing these habitats lies in the fact that the salinity in the channels of the North Bay can vary by 5-10 parts per thousand on the tidal timescale through the advection of the salinity field by tidal currents. As a result, the timing of the exchange of waters between the channel and the shallows relative to the tidal currents in the channel will largely determine the mean salinity of these “off-channel” habitats.

To address this question, we have focused on an analytic solution, which will be most useful for management considerations. The approach is to link two one-dimensional solutions to predict the timing of the average flow between the channel and the shoal. The first is a simple tidal model of the flows and salinities in the channel, which includes as a prescribed parameter the phasing between the tidal stage and the tidal currents. The second model is a one-dimensional parameterization of the exchange flows between the channel and the shallows, which are driven by the difference in stage between the channel and the shallows, but is resisted by a frictional force that will depend on the geometry of the opening. Solving the cross-sectionally averaged momentum equation for the exchange flow, we determine the timing and salinity of the waters flowing into the shallow water habitats as a function of the mean channel salinity, the salinity gradient along the channel, the phasing of currents and stage in the channel, and a drag coefficient for the channel-shallow transition.

The solution indicates that the salinity in these shallow off-channel water bodies will exceed the mean channel salinity by as much as several parts per thousand, due to the timing of the exchange flows, but depends on the geometry of the transition through the integrated drag coefficient. To determine appropriate values of the drag coefficient, we used the three-dimensional hydrodynamic model, TRIM3d. The model was run for an idealized channel-shallow geometry for four different geometric cases . The results of the three-dimensional simulation were integrated to reproduce the one-dimensional parameterization used in the analytic solution, and each term in the one-dimensional equation was evaluated to determine the average drag coefficient. The salinity in the shallow water habitat may change by as much as 5 parts per thousand depending on the geometry of the transition. From a management perspective, this may provide a mechanism for controlling the salinity in the shallow-water habitat. For a given geometry, these results would determine the optimal location for the shallow-water habitat, in order to achieve desired salinities.

### *Temperature Effects on Exchanges*

In September 2001, using direct hydrodynamic observations, it was seen that the exchange between Mildred Island and the adjoining channel was generally a typical jet-structure during the flooding tide . However, on warm days, the jet structure was modified during the afternoon due to the effects of temperature stratification and wind. As a result, we have begun development of a temperature module for use in TRIM3d. This activity is still underway and will continue under funding received from CALFED.

## ***Publications***

Baek, S. and Stacey, M., Steady-state salinity of shallow water habitats adjacent to a tidal channel, *Journal of Hydraulic Engineering.*

## ***Professional Presentations***

Stacey, Mark, Mildred Island, Lake, Lagoon or Estuary. What is it?, . PI Organizational meeting for CALFED funded activity, USGS, Menlo Park, California, February 6, 2002

## ***Student Training***

Seungjin Baek, M.S./Ph.D., Civil and Environmental Engineering, UC Berkeley.

## ***Additional Funding***

The three-dimensional modeling work begun under the WRC funding has led to a longer-term collaborative study with the USGS and funded by CALFED for $113,000.

## ***Collaborative Efforts***

The Mildred Island study was a collaboration with the USGS which occurred in advance of the CALFED funding described in the previous section.

# Modeling and Optimization of Water Quality in a Large-Scale Regional Water Supply System

## Basic Information

| | |
|---|---|
| **Title:** | Modeling and Optimization of Water Quality in a Large-Scale Regional Water Supply System |
| **Project Number:** | 2001CA3981B |
| **Start Date:** | 3/1/2001 |
| **End Date:** | 2/28/2002 |
| **Funding Source:** | 104B |
| **Congressional District:** | 23 |
| **Research Category:** | Water Quality |
| **Focus Category:** | Management and Planning, Models, Water Supply |
| **Descriptors:** | reservoirs, network flow model, multiobjective analysis, systems analysis, nonlinar programming, regional water supply systems, optimization of water resources systems,water resources planning and management |
| **Principal Investigators:** | William Yeh |

## Publication


1. Tu, M-Y., Frank T-C. Tsai, and William W-G. Yeh, Optimization of water distribution and water quality by hybrid genetic algorithm, Submitted to Operations Research, August 2002.

# Modeling and Optimization of Water Quality in a Large-Scale Regional Water Supply System

**William W-G. Yeh**
**Civil and Environmental Engineering**
**UC Los Angeles**

## ***Project Summary***

In a regional water distribution system involving multiple source water of varying quality, water agencies often find that it is necessary to employ blending at certain control points in the system to ensure the quality of water they deliver. We are developing a mathematical model that simulates the operation of regional-scale water distribution system and optimizes the quality of the distributed water.

In general, the water distribution system is represented by a network, in which supply sources, reservoirs, ground-water basins, junctions and demands are represented by different types of nodes; pumping stations, hydroelectric power plants, and pipes are arcs linking the nodes. In this network, water available at various nodes may be delivered to any designated location through the arcs which may be directional or undirectional. An undirectional arc allows water to flow in either direction, but not in both directions at the same time. Waters from different sources with different water quality are considered as distinct commodities, which concurrently share a single water distribution system. The objective function optimizes the volumes and quality of water at the delivery points. Blending requirements are treated as constraints and specified for each control points in the water distribution system. The mixing is assumed to be that incoming waters of different quality are instantaneously mixed at the merging junction and that the outgoing water from the junction has the same blend.

The operation of the multicommodity flow model is optimized by employment of a hybrid genetic algorithm (GA) and a generalized reduced gradient algorithm (GRG). First, the GA is used to globally search for the directions of all two-way flow arcs in the planning horizon. With the directions of all two-way flow arcs determined, GRG algorithm optimizes the objective function of the multicommondity model for fitness evaluation and chromosome evolution. The proposed approach is an iteration procedure between the GA and GRG. This approach has the following advantages: (1) it converts an undirected network to a directed network that is amiable to standard optimization, (2) it separates the highly nonlinear two-way flow constraints from the gradient-based algorithm, and (3) GA with multiple starting points increases the likelihood of reaching a global optimum.

The proposed model was tested and verified on a simplified, but realistic water distribution system. It was then applied to the water distribution system of the Metropolitan Water District of Southern California (MWD). MWD supplies water to a population of approximately 17 million people in Southern California with a service area of 5,200 square miles. Additionally, sensitivity analyses were performed to analyze the impact of blending requirements. The results demonstrate the applicability of the proposed model to a real-world, large-scale regional water distribution system.

## ***Publications***

Tu, M-Y., Frank T-C. Tsai, and William W-G. Yeh, Optimization of water distribution and water quality by hybrid genetic algorithm, Submitted to *Operations Research*, August 2002.

## ***Professional Presentations***

Tu, M-Y., F. T-C. Tsai, and W.W-G. Yeh Optimization of water distribution and water quality by genetic lgorithm and nonlinear programming. Fall Annual Meeting, American Geophysical Union, San Francisco, California, December 10-14, 2001.

## ***Student Training***

Ming-Yen Tu, Ph.D. student, program in water resources, UC Los Angeles.
Frank T-C. Tsai, Ph.D. student, program in water resources, UC Los Angeles.

# Examining the relative influence of riparian and upland landcover and landuse on instream habitat: Improved methods for the Russian River basin.

## Basic Information

| | |
|---|---|
| **Title:** | Examining the relative influence of riparian and upland landcover and landuse on instream habitat: Improved methods for the Russian River basin. |
| **Project Number:** | 2001CA4001B |
| **Start Date:** | 3/1/2001 |
| **End Date:** | 2/28/2002 |
| **Funding Source:** | 104B |
| **Congressional District:** | 43 |
| **Research Category:** | Ground-water Flow and Transport |
| **Focus Category:** | Wetlands, Models, Agriculture |
| **Descriptors:** | buffer strips, riparian, watershed, scale, landuse, salmon habitat, restoration, GIS, remote sensing |
| **Principal Investigators:** | Nina Maggi Kelly |

## Publication


1. Opperman, J. and A. Merenlender. in press. Factors influencing the success of riparian restoration in the Russian River basin: deer, sheep, and hydrology. Proceedings of the California Riparian Habitat and Floodplains Conference, Sacramento, California, March 2001

# Examining the Relative Influence of Riparian and Upland Land Cover and Land Use on Instream Habitats: Improved Methods for the Russian River Basin

**Nina Maggi Kelly**
**Environmental Sciences,**
**Policy and Management**
**UC Berkeley**

## ***Project Summary***

Numerous studies have established that large woody debris (LWD) plays a critical role forming and maintaining habitat for anadromous fish. However, nearly all of these studies have been conducted in the conifer-dominated forests of the Pacific Northwest (PNW). In California, anadromous fish also utilize watersheds dominated by hardwoods. Although managers are currently trying to restore habitat for anadromous fish in these watersheds, very little is known about the role of LWD in shaping this habitat. In this study, we are examining the relationships between riparian vegetation, instream LWD, and fish habitat in hardwood watersheds. We are studying riparian and landscape-scale influences on aquatic habitat in the Russian River basin and other hardwood-dominated watersheds in Northern California. Progress has been made in three primary areas: Riparian vegetation, large woody debris, and in-stream habitat.

We conducted field work examining the above described interactions in 25 hardwood-dominated streams in Northern California. Although the loading of woody debris (volume of LWD per unit area of channel) in hardwood streams was considerably lower than values reported from streams in conifer-dominated forests in the PNW, we found many of the same relationships between instream wood and fish habitat. For example, debris-formed pools had significantly greater cover (i.e. shelter) values than any other pool type. In hardwood streams with relatively high LWD loading, the majority of pools were formed by woody debris and, across streams, the occurrence of pools was positively correlated to LWD loading. Because hardwood debris is considerably smaller than that derived from mature conifers, single pieces of hardwood debris rarely formed a pool. Instead, pool formation was generally influenced by complex debris jams that spanned or partially spanned the channel. These jams were often stabilized by the presence of living trees within the debris jam. Many riparian hardwoods have resilient growth forms such that they can be pushed over into the stream, become part of a debris jam, and then continue living and growing as long as their root system maintains contact with the bank. This "living LWD" results in more stable debris jams because key pieces of the debris jam are stabilized by their living root masses and do not decay.

### ***Improved methods for characterizing hardwood riparian corridors through remote sensing***

The structure and extent of the riparian forest strongly influences the amount and size of LWD that enters a channel. Because of the important relationship between riparian vegetation, LWD, and fish habitat, land managers seek methods to monitor and characterize the quality of riparian vegetation within a basin or across a region. We are developing techniques to characterize riparian corridors in hardwood-dominated regions using high-resolution remotely sensed imagery. We are collaborating with a scientist at California State University, Sonoma, to analyze riparian corridors in the Sonoma Valley using ADAR - high-resolution (4 m), multi-spectral imagery. The techniques that will allow us to: 1) identify riparian corridors on the landscape; 2) quantify their width and extent (i.e. presence of gaps); and 3) characterize their species composition and structure (e.g. size, maturity).

***Landscape-scale influences on aquatic habitat***
We also studied how land use and land cover (LULC) across the landscape influences habitat within streams. In particular, we are examining the relationship between LULC at various scales and the embeddedness (the amount of fine sediments) of spawning gravels utilized by anadromous fish. We are utilizing 10 m Digital Elevation Models (DEMs) to designate unique watersheds above each of 380 reaches in the Russian River basin that have been surveyed for fish habitat, including embeddedness values, by the California Department of Fish and Game. Within these watersheds we are investigating the relationship between gravel embeddedness and LULC at several scales: 1) a 30 m buffer surrounding the reach; 2) the same buffer but also extended varying distances above the reach; and 3) the entire watershed above the reach. Initial results indicate that the LULC of the entire watershed explains the most variability in the embeddedness values; the level of embeddedness is positively correlated with the amount of agriculture and development in the watershed, while it is negatively correlated with the amount of forest and chaparral in the watershed. We are currently using the same data sources to develop a Hydrological Proximity Model (HPM) that creates an index to describe the way that LULC interacts with runoff patterns over the entire watershed. We will then examine whether the outcomes of HPM provide additional explanatory power.

## ***Publications***

Opperman, J. and A. Merenlender. *in press*. Factors influencing the success of riparian restoration in the Russian River basin: deer, sheep, and hydrology. *Proceedings of the California Riparian Habitat and Floodplains Conference*, Sacramento, California, March 2001.

## ***Professional Presentations***

Opperman, J. and A. Merenlender. Restoration of riparian corridors and large woody debris in California's hardwood-dominated watersheds. Joint Annual Meeting of the Ecological Society of America and the Society for Ecological Restoration, Tucson, Arizona, August, 2002.
Opperman, J. and A. Merenlender. Evaluating riparian restoration projects to guide anadromous fish habitat restoration in California's hardwood-dominated watersheds, American Fisheries Society Annual Meeting (California-Nevada Chapter), Lake Tahoe, April 2002.

Opperman, J., C. Brooks, A. Merenlender, and Z. Young. Large woody debris and fish habitat in hardwood-dominated streams of the Russian River Basin, Poster presented at the Salmonid Restoration Federation Annual Conference, Ukiah, California, March 2002.
Opperman, J. and A. Merenlender. Large woody debris and habitat for endangered fish in California's Mediterranean-climate watersheds, San Francisco Bay Area Conservation Biology Symposium, San Francisco State University, California, January 2002.

Opperman, J. Riparian restoration in the Russian River basin, California Riparian Habitat and Floodplains Conference, Sacramento, California, March 2001.

## ***Student Training***

Michael Gerstein, Undergraduate, Environmental Science, UC Berkeley
Sanaz Mamarsedegghi, Undergraduate, Conservation and Resource Studies, UC Berkeley
Susan Mahler, graduate, ESPM, UC Berkeley
Jeff Opperman, graduate, ESPM, UC Berkeley

## ***Additional Funding***

This research project is also being supported by the following grant: "Influences of riparian vegetation and landscape-scale factors on salmonid habitat in Mediterranean-climate hardwood watersheds," from California Department of Fish and Game. 2001-2003. $93,000.

## ***Collaborative Efforts***

We are collaborating on imagery analysis with Professor Ross Meentemeyer, Department of Geography, Sonoma State University, California.

A channel-spanning debris jam creating a pool with high shelter values. The piece of wood in the foreground (in the shape of a Y rotated counter-clockwise) is a living red willow (Salix laevigata) which has fallen into the stream but is still rooted and living. This living key piece traps much of the other wood in the debris jam.

# Information Transfer Program

The Water Resources Center, through the main office, has filled approximately 120 requests for publications and mailed out annual publications to aproximately 550 recipients. All the publications are available on the website at http://waterresources.ucr.edu. The Water Resources Center archives has filled approximately 12,112 tranactions. The archival collection can be found at http://www.oac.cdlib.org/dynaweb/ead/berkeley/wrca/.

The Center for Water Resources continues its outreach efforts through the newsletter (Currents) and website. The newsletter is mailed to over 3,500 recipients and is abailable on-line. The website has received hundreds of visitors and continues to be used as a communication tool.

# Student Support

| Student Support | | | | | |
|---|---|---|---|---|---|
| **Category** | **Section 104 Base Grant** | **Section 104 RCGP Award** | **NIWR-USGS Internship** | **Supplemental Awards** | **Total** |
| **Undergraduate** | 14 | 10 | 0 | 4 | 28 |
| **Masters** | 5 | 10 | 0 | 0 | 15 |
| **Ph.D.** | 3 | 11 | 0 | 0 | 14 |
| **Post-Doc.** | 0 | 1 | 0 | 0 | 0 |
| **Total** | 22 | 32 | 0 | 4 | 57 |

# Notable Awards and Achievements

Additional funding generated by the section 104 funded projects amounted to just over $1.1 million.

# Publications from Prior Projects


1. MacKay, D.M., R. Wilson, G. Durrant, K. Scow, A. Smith, M. Einarson and B. Flower. 2000. "In-Situ Treatment of MTBE by Biostimulation of Native Aerobic Microorganisms". In: the Environmental Protection Agency/American Petroleum Institute MTBE Biodegradation Workshop, Cincinnati, Feb. 1-3, 2000. MacKay, D.M., R. Wilson, G. Durrant, K. Scow, A. Smith, M. Einarson and B. Flower. 2000. "Field tests of enhanced intrinsic remediation of an MTBE plume". In: Abstracts of Papers 219th Meeting of the American Chemical Society, San Francisco, CA March 26-30th, 2000. 210 (1-2): ENVR 230