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Analysis of Alternative Watershed Management Strategies Addressing Aquatic Toxicity:

A case study of Organophosphate Pesticide Loading in Newport Bay, CA

A Group Project submitted in partial satisfaction of the requirements for the degree of Master's in Environmental Science and Management
for the
Donald Bren School of Environmental Science & Management

by

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June 2002

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The Group Project is required of all students in the Master's of Environmental Science and Management (MESM) Program. It is a three-quarter activity in which small groups of students conduct focused, interdisciplinary research on the scientific, management, and policy dimensions of a specific environmental issue. This Final Group Project Report is authored by MESM students and has been reviewed and approved by:

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ABSTRACT

Our study evaluated the efficacy of strategic management alternatives designed to reduce the level of toxicity in urban runoff being discharged within the Newport Bay Watershed, in Orange County, California. We first identified the key pollutants responsible for the aquatic toxicity in freshwater and saltwater environments, which were determined to be two organophosphate (OP) pesticides, Diazinon and Chlorpyrifos. We traced the major contributors of these pesticides to a corresponding landuse and developed export rate coefficients for each pesticide over coarse landuse classes. Model-based predictions were then used to evaluate the persistence of these pollutants over an array of scenarios, simulating base-case, policy-related usage restrictions and management practices aimed at improving water quality. Our watershed modeling analysis established that after the phase-out Diazinon would persist in all stormflow events in exceedance of the numeric criteria for aquatic toxicity while Chlorpyrifos concentrations appeared to be more moderate with respect to the criterion limits. Based on the results of our cost-effective analysis, we recommend the installation of several infiltration basins in conjunction with supplemental public education programs as a means to reduce the OP pesticide-related toxicity. In order to maintain and restore the ecological integrity of the Bay, we also recommend further usage restrictions on both pesticides throughout the watershed.

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EXECUTIVE SUMMARY

Following the inception of the Clean Water Act, water quality management strategies within the U.S. have traditionally focused on controlling point sources of pollution through regulation of pollution discharge to open aquatic systems. Regulations by state and federal agencies have forced dischargers to comply with water quality-based criteria under the National Pollutant Discharge Elimination System (NPDES). Although this approach has had success, it has become increasingly apparent that pollutants from non-point loading are significant sources of water body impairment.

The prevalence of non-point pollutants and their widespread distribution throughout many watersheds with mixed urban, residential and agricultural landuses has led to a stronger emphasis on managing non-point pollutants through a source-based approach (NRC, 2001). Within this context, the objective of the present report is to develop management strategies for Newport Bay, California, an area characterized by elevated aquatic toxicity derived from a suite of non-point pollutant sources.

To assess the relative contribution from different sources of toxicity, we have employed a linkage analysis by comparing environmental stressors, (such as landuse activities, channelization of natural rivers or an increase in impervious surfaces) to biological responses in the watershed. A significant body of research on aquatic life toxicity caused by non-point loading exists within Newport Bay (Lee and Taylor, 2001, Lee and Taylor 1999, SARWQCB, 2000, SARWQCB, 2001). Through this research two organophosphate (OP) pesticides have been identified as being responsible for over half of the observed aquatic life toxicity. Using the OP pesticides Diazinon and Chlorpyrifos as the primary environmental stressors responsible for aquatic toxicity in Newport Bay, we modeled the additive contribution from several general landuse classes in attempts to understand the primary sources responsible for the observed toxicity.

For our approach to modeling the environmental conditions in Newport Bay, we employed two watershed models, BASINS/HSPF and WARMF capable of representing key processes and simulating environmental stressors and the system response. The stressors were represented as an OP pesticide load per landuse and the response was a resulting water concentration. There was an inference based on toxicological data that the health of the local biological systems would show a direct response to the level of pollutant loading. Because they represent our scientific understanding of mechanistic processes within Newport Bay, models have played a central role in our analysis. The models quantitatively link watershed processes with effective management alternatives.

In light of the recent decision by the EPA to phase-out certain uses of both of the focus pesticides, there is uncertainty regarding future pesticide loading and related toxicity within Newport Bay. The uncertainty surrounding the phase-out highlights

the necessity of making management decisions in the absence of complete information. This relationship between science and policy is the crux of our project and the platform on which our analysis is based.

Water Quality Management requires the usage of models to relate watershed processes to a given management practice that might control pollution such as re-vegetating the riparian zone, building constructed wetlands or employing street sweeping techniques. The BASINS and WARMF models that we utilized in our study allowed us to integrate spatial data for a given watershed with pollutant loading values to predict how a system will respond to a set of management alternatives. These models provided a decision support framework to base our recommendations on.

Our watershed modeling analysis established that after the partial phase-out of these two pesticides, Diazinon would persist in stormflow events above both the state and federal criteria for aquatic toxicity. The results indicate that, on average, Diazinon will be found in toxic concentrations in all storm events in Newport Bay after the phase-out. In contrast, Chlorpyrifos concentrations appeared within the acute criterion limits between 1 and 4 days (and chronic limits between 0 and 3 days), even during storm events.

An important conclusion to be drawn from these model simulations is that the phase-out will be more effective at reducing the number of days above the criteria for Chlorpyrifos than Diazinon during storm flow events. Additionally, we discovered that a complete reduction in urban Diazinon uses is necessary to keep the concentrations below criteria levels. These findings provided the impetus to evaluate various BMPs as further toxicity-reduction will be necessary to protect the biological integrity of the Bay.

After evaluating various structural management practices, we concluded that infiltration ponds are the most cost-effective solution for reducing pesticide concentrations followed by infiltration basins and grass swales. It would require between \$16-\$34 million dollars to implement the necessary amount of infiltration basins (between 90-1,800 infiltration basins) to reduce the number of days in exceedence to below criteria levels. In conjunction with infiltration basins, we recommend tighter policy restrictions on Diazinon and Chlorpyrifos usage. The results are valuable as a means to assess the effectiveness of the adopted policy and speak to the need for predicting the degree of toxicity to be expected in the first few years of the phase-out. Inclusion of educational programs aimed at elevating the awareness level of the public will be key to the success of reducing toxicity in Newport Bay. Additionally, we recommend the development alternative strategies such as Integrated Pest Management (IPM) in urban, residential and agricultural settings.

The final results provide a valuable contribution to stakeholder groups interested in restoring and enhancing the beneficial uses in Newport Bay. Given the uncertainty surrounding the effectiveness of the pesticide phase-out and projected future concentrations, our project provides a means to make practical management decisions based on the best available scientific data.

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List of Commonly Used Abbreviations and Acronyms

ACOE- Army Corps of Engineers
BASINS- Better Assessment Science Integrating Point and Non-point Sources
BMP - Best Management Practice
CCC – Criterion Chronic Concentration
CDFG- California Department of Fish & Game
CMC – Criterion Maximum Concentration
DPR (CA)- Department of Pesticide Regulation
EPA – see USEPA
HQ- Hazard Quotient
IPM – Integrated Pest Management
NCCP/HCP- Natural Community Conservation Plan and Habitat Conversation Plan
NPDES- National Pollutant Discharge Elimination System
OP – Organophosphate
RIFA- Red Imported Fire Ant
RWQCB - California Regional Water Quality Control Board
SARWQCB - Santa Ana Regional Water Quality Control Board
SCAG - Southern California Association of Governments
SCCWRP- Southern California Coastal Watershed Restoration Project
SMW- State Mussel Watch
STP- Storm Treatment Practices
TIE- Toxic Identification Evaluation
TMDL - Total Maximum Daily Load
TSM- Toxic Substance Monitoring
USEPA – United States Environmental Protection Agency
USGS - United States Geologic Survey
V – Volume
WQ_v – Water Quality Volume
WARMF-Water Analysis Risk Management Framework

1.0 INTRODUCTION

The Newport Bay watershed has undergone many changes resulting from decades of urban growth. Open space and agricultural lands have been replaced by high-density residential and urban development. With urbanization, the landscape is now characterized by an altered drainage network, stormflow pattern and increased sediment and pollutant loading to stream channels and to Newport Bay.

The watershed has a large population (~800,000) and is located in a semi-arid region with a small amount of annual rainfall. The combination of impervious surfaces due to urbanization, and the short-duration high-intensity precipitation inputs common in Mediterranean climates, results in fairly flashy hydrographs. This translates into very low infiltration rates within the landscape and high levels of overland flow, which are discharged quickly to adjacent water bodies.

Dispersed in the overland stormflow is a suite of pollutants, including Diazinon and Chlorpyrifos, two organophosphate (OP) pesticides. These pesticides, while highly toxic to aquatic life, have a short half-life of less than six months. It follows that it is not the persistence of the pesticides (such as is the case with many legacy pollutants such as DDT), but rather the high loading and toxicity of OP pesticides, specifically Diazinon and Chlorpyrifos, at levels that are toxic to aquatic biota. Due to the flashy nature of the stormflow hydrograph, high inputs of pesticides from urban, residential and agricultural areas and the extent of impervious surfaces, the pesticides are frequently found in both soil and water samples at levels that exceed the stated water quality criteria. Given the current state of the stormflow hydrograph and the altered landscape it is very difficult to manage the loading of non-point pollutants such as OP pesticides.

In the last two years the U.S. Environmental Protection Agency (EPA) initiated a re-registration program for Diazinon and Chlorpyrifos that targets over the counter sales and other non-agricultural uses. This phase-out and registration program is designed to reduce residential loads from entering watershed/storm water systems and is predicted to reduce this source by between 50-75% for Chlorpyrifos and Diazinon respectively.

We approached our study of reducing the pesticide-related toxicity in Newport Bay by asking four questions:

- What are the primary sources and spatial distribution of OP pesticides within Newport Bay?
- What is the relative contribution of each of the sources, measured as a pesticide load per landuse?
- Will the proposed pesticide phase-out reduce concentrations within regulatory guidelines?

- What management practices will effectively reduce the OP pesticide-related toxicity, given current loading and future phase-out conditions?

We begin by describing the Newport Bay Watershed and discussing the meteorology, hydrology, geology, habitat classes and landuses found in this area. We present the Santa Ana Regional Water Quality Control Board (SARWQCB) Problem Statement for toxic substances as a basis for our research and as a guideline for our direction of study. Relevant water quality standards and legislation are presented to provide the policy context. The sources of OP pesticides and the fate and transport processes are explained in order to characterize their spatial distribution, mobility and expected persistence in the environment. Next, we examined the level of risk posed to human health and aquatic life from these pesticides, based on model-based risk assessment calculations.

We then present our analyses conducted with the BASINS and WARMF watershed models. We describe how the models were implemented, calibrated and applied to account for the representation of key fate and transport processes specific to the selected pesticides and the physical makeup of Newport Bay. These model scenarios form the basis of our predictions of future pesticide concentrations and the necessity to decrease the load entering surface waters in order to comply with water quality standards.

Finally, we present our evaluation of applicable Best Management Practices (BMPs) and their relative effectiveness at reducing pesticide-related toxicity in surface waters. BMPs are reviewed critically based on their effectiveness in removing OP pesticides and on the feasibility of implementation based on the estimated costs.

The results of our study should provide a valuable contribution to stakeholder groups interested in restoring and enhancing the beneficial uses in Newport Bay. Given the high level of uncertainty surrounding the effectiveness of specific OP pesticide phase-out and projected future concentrations, our project provides a means to make practical management decisions based on the best available scientific data.

2.0 BACKGROUND

2.1 Problem Statement

In the late 1980s the Santa Ana Regional Water Quality Control Board (SARWQCB) listed Newport Bay and its main tributary, San Diego Creek, as being impaired under section 303(d) of the Clean Water Act, in part due to failure in meeting the Newport Bay Basin Plan objectives for toxic substances. These listings were based on bioaccumulation of DDT, PCBs and other toxic substances found in aquatic organisms collected from Newport Bay and San Diego Creek. In 1993 Orange County completed the "Newport Bay Watershed Toxicity Study," commissioned by the Regional Water Quality Control Board (RWQCB), which found that heavy metals were not the main causes of toxicity as initially expected. The Toxicity Identification Evaluation (TIE) section of the study identified the organophosphate (OP) pesticides Diazinon and Chlorpyrifos to be responsible for approximately 50% of the toxicity in Newport Bay and San Diego Creek, (SARWQCB, 2000).

Diazinon and Chlorpyrifos pollution continues to threaten the human and ecological health of Newport Bay. In Orange County, which is drained by the Newport Bay Watershed, over 100,000 pounds of Diazinon and Chlorpyrifos are used annually. The principal use of these applications is residential structural pest control by both commercial and public users (Jones-Lee et al., 1999). Agricultural use of Diazinon and Chlorpyrifos also contributes to pesticide pollution in the Bay. Several nurseries located in the Newport Bay Watershed contribute runoff containing high concentrations of pesticides (specifically Diazinon and Chlorpyrifos) (SARWQCB, 2001). High pesticide loading into Newport Bay threatens the health of human residents and visitors to the area, as well as the natural environment of the Bay. Both Diazinon and Chlorpyrifos have been associated with bird and fish kills, as well as child poisoning (U.S.EPA, 2000).

2.2 EPA Phase-Out of Certain OP Pesticide Uses

2.2.1 Phase-out of Certain Diazinon Uses

In January 2001, the EPA released a revised risk assessment and an agreement with registered users to phase-out most of the Diazinon uses (U.S.EPA, 2001). Under the agreement, all indoor uses will be terminated, and all outdoor non-agricultural uses will be phased out over the next few years. Retail sales will be banned after December 31, 2002. The EPA expects that these actions will reduce the current residential Diazinon loading from residential users by approximately 75% of current conditions. Additionally 1/3 of agricultural uses will be removed. These estimates incorporate assumptions of residual uses.

2.2.2 Phase-out of Certain Chlorpyrifos Uses

The EPA has decided to restrict the residential use of Chlorpyrifos by the public because of its potential cumulative toxicity to humans, especially children. The usage of

Chlorpyrifos has been restricted as of June 2000, by an agreement between registered users and the EPA. While over-the-counter sales will be restricted, non-structural wood treatment and fire ant eradication will continue by professional users. The EPA has estimated that the phase-out will reduce current concentrations by approximately 50% (U.S.EPA, 2000).

The restricted uses of both pesticides under the phase-out may provide a suitable method for reducing the toxicity to aquatic life within our study area. If the restrictions on the sale and usage of these OP pesticides do not provide the intended reduction in pesticide concentrations, further regulation of the usage would be required. Considering that about two-thirds of the Chlorpyrifos and nearly one-half of the Diazinon uses will continue after the phase-out, it seems very likely that the toxicity problem will persist.

It is this uncertainty that has fueled our analysis and demanded the need to address the potential for future violations of the EPA or California Department of Fish and Game (CDFG) numeric criteria. Results from our model simulations of current and future predictions are discussed in the results section. A key component to this analysis is the evaluation of the anticipated improvement in water quality within Newport Bay resulting from the phase-out.

2.3 Water Quality Criteria

The current EPA and RWQCB approach to solving water quality problems is to control the constituent responsible for waterbody impairment. This is typically achieved through setting a numeric water quality objective for the specific pollutant in the RWQCB Basin Plan. As the SARWQCB Basin Plan does not contain numeric water quality objectives for toxicity or for pesticides, numeric targets are used for both OP pesticides. The lack of enforceable water quality criteria has contributed to the continued use of these organophosphates within the watershed. Furthermore, it highlights the need for better understanding of both the toxic effects to aquatic organisms as well as the physicochemical properties of these OP pesticides that determine where and how long they will persist once released to the environment.

The EPA and the California Department of Fish and Game have developed recommended water quality criteria for Diazinon and Chlorpyrifos. The numeric targets include acute and chronic exposure limits not to be exceeded at a certain concentration over a specified time duration. For the acute criteria, the target concentrations should not be exceeded for a one-hour period over a three-year duration. Chronic exposure limits should not be exceeded for a period of four days within a three-year period. Failure to meet these water quality targets results in failure to protect freshwater, saltwater and wildlife habitats, as listed under the watersheds beneficial uses (SARWQCB, 2000).

2.3.1 California Department of Fish and Game (CDFG) Numeric Criteria

The CDFG criteria recommend a freshwater Diazinon acute criterion (CMC) of 80 ng/L and a chronic criterion (CCC) of 50 ng/L (Table 2-1). No saltwater criterion has been developed for Diazinon. CDFG recommends a freshwater Chlorpyrifos CMC of 20 ng/L and a CCC of 14 ng/L (Table 2-2). The corresponding saltwater CMC is 20 ng/L and CCC of 9 ng/L. Studies by the EPA have indicated that the Diazinon and Chlorpyrifos toxicities are additive (CDFG, 2000a).

Table 2-1. Diazinon Numeric Criteria.

Freshwater	CDFG	EPA
	ng/L	
CCC	50	n/a
CMC	80	90

Table 2-2. Chlorpyrifos Numeric Criteria.

Freshwater	CDFG	EPA
	ng/L	
CCC	14	41
CMC	20	83
Saltwater		
CCC	9	5.6
CMC	20	11

2.3.2 EPA Numeric Criteria

The EPA's criteria are slightly less stringent, with recommendations for Diazinon with CMC of 90 ng/L for freshwater. For Chlorpyrifos the freshwater CCC is 41 ng/L and the CMC is 83 ng/L. Saltwater recommendations are 5.6 ng/L CCC and 11 ng/L CMC. For a more detailed discussion of the development criteria refer to the EPA reports on revised risk for Diazinon and Chlorpyrifos (U.S.EPA 2000, 2001).

2.4 Previous Studies/Investigations

This project has drawn upon a number of recent studies that have focused on identifying the extent of aquatic toxicity due to OP pesticides within Newport Bay. These include: The Toxic Substances Monitoring (TSM) and State Mussel Watch (SMW), California Department of Pesticide Regulation (DPR) Pesticide Use Reports, DPR Red Imported Fire Ant (RIFA) Monitoring, DPR Sales and Use Survey and the Aquatic Life Toxicity Investigations conducted under the 319(h) and 205(j) studies performed by Lee and Taylor (1999; 2001). For descriptions of each of these projects refer to the SARWQCB Problem Statement for the TMDL for Toxicity. The current report utilized the most recent data, primarily from the 205(j) and 319(h) Aquatic Life Toxicity Investigations and the DPR Pesticide Use Reports. This project sought to expand the research using models to help validate decisions regarding the phase-out and offer preliminary cost effectiveness information useful for planning projects to reduce Chlorpyrifos and Diazinon toxicity within the Bay.

3.0 NEWPORT BAY WATERSHED

3.1 Location

Newport Bay Watershed (USGS Cataloging Unit: 18070204) is situated in southern California 40 miles south of Los Angeles and 75 miles north of San Diego. The watershed, illustrated in Figure 3-1, drains about 154 square miles and contains some of the most populous cities within Orange County including Santa Ana, Tustin, Costa Mesa, Irvine, Lake Forest and Newport Beach with a total population in the watershed of approximately 800,000 people. The Newport Bay watershed contains three major topographic relief zones: mountains, coastal foothills, and central flats.

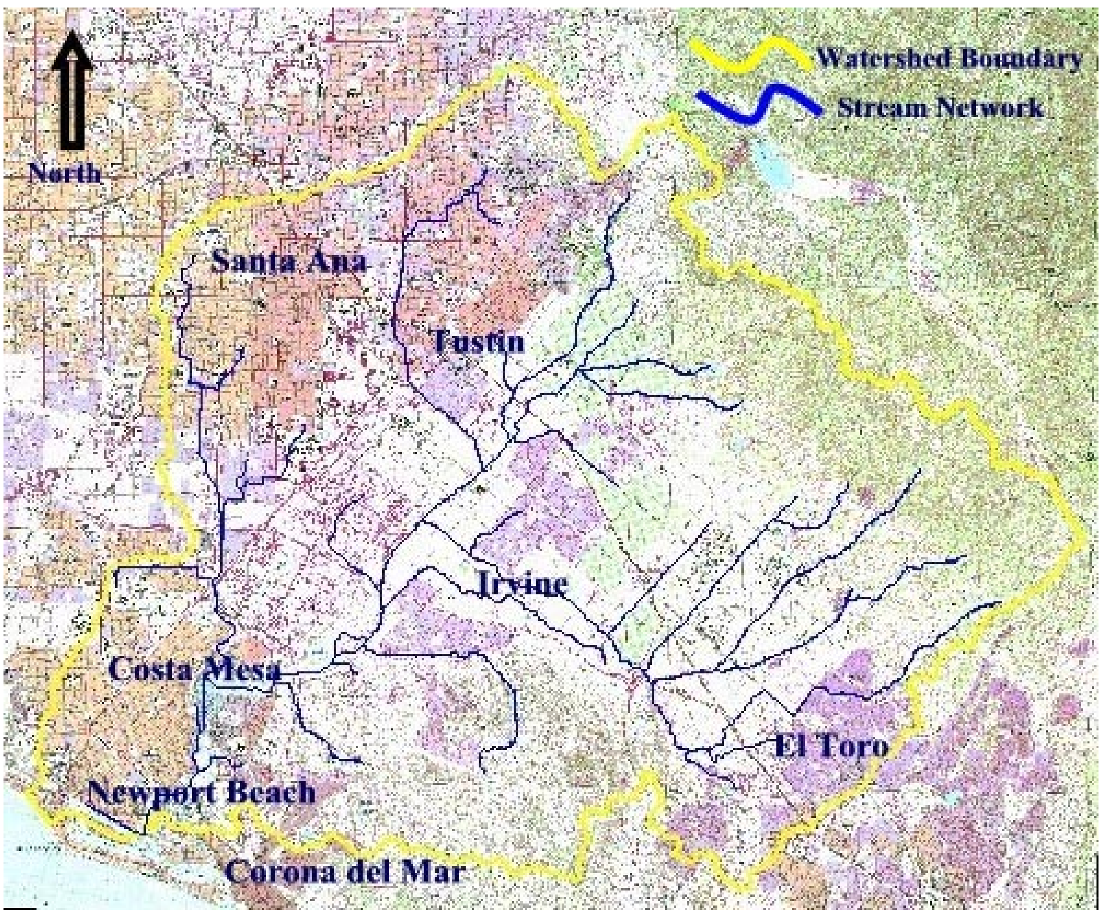


Figure 3-1. The Newport Bay Watershed and the Major Cities Located Within Its Boundaries.

3.2 Climate

Short mild winters and warm dry summers dominate Newport Bay Watershed, as is the general trend in southern California. The average summer high/low temperatures are 74 F (17 C)/45 F (7 C). The average winter high/low temperatures are 63 F (17 C)/45 F (7 C). The average rainfall is 12 inches per year, most of which occurs between November and April. During the winter months daily wind speed averages 4 mph from the west to southwest and lowers to about 2 mph in summer.

3.3 Hydrology

Newport Bay watershed contains two major tributaries, Peters Canyon Wash and San Diego Creek (see Figure 3-2). Peters Canyon Wash includes Peters Canyon, Rattlesnake Canyon, and Hicks Canyon, all of which are located in the Santa Ana Mountains. In total, the Peters Canyon Wash catchment drains 44 square miles (28,000 acres). Once out of the mountains, the creeks have been channelized to help control flooding and permit growth up to their borders.

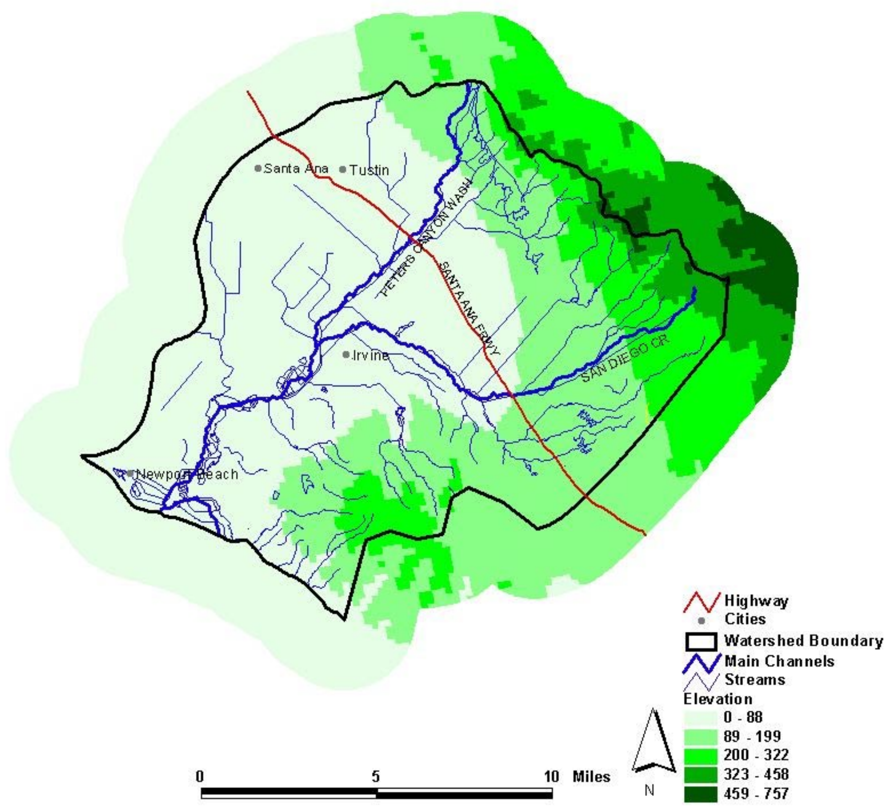


Figure 3-2. Tributaries and major roads within the Newport Bay Watershed.

Peters Canyon Wash combines with San Diego Creek above the University of California, Irvine. Upper San Diego Creek, before its confluence with Peters Canyon Wash, extends east to the Santiago Hills, which include Bee Canyon, Round Canyon, Aqua Chinon Wash, Borrego Canyon Wash, and Serrano Creek. Like Peters Canyon Wash, most of San Diego Creek's channels have been improved and channelized to drain roughly the same area as Peters Canyon Wash, 46 square miles (29,500 acres). Upper San Diego Creek has a base flow of 8 to 15 cubic feet per second (cfs) during dry weather and a mean range of 8 to 4500 cfs during storm events. Lower San Diego Creek, i.e. below the confluence of Peters Canyon Wash and San Diego Creek, enters Newport Bay averaging 30 cfs during the dry summer months and can exceed 30,000 cfs during storm events. Figure 3-2 presents a view of all the tributaries within the Newport Bay Watershed.

Peters Canyon Wash and San Diego Creek combined (75,520 acres) are the major contributors of freshwater and sediment to Newport Bay, and carry the vast majority of pollutants to the Bay. The Bay itself is generally split into two distinct bodies of water, "lower" and "upper" Newport Bay. The lower Bay (752 acres) historically was a large coastal lagoon that has been under development since the mid 1800's. The upper Bay (1,000 acres) is a narrow estuary about two miles long, which extends back to the mouth of lower San Diego Creek. The upper Bay also receives freshwater from tributaries other than San Diego Creek including Santa Ana – Delhi Channel and Big Canyon in conjunction with local springs and the surrounding area. Two boulder jetties, located at the rocky headland of Corona del Mar, border the Bay's entrance into the Pacific Ocean.

3.4 Geology and Geography

Newport Bay watershed is bordered by the San Joaquin Hills on the south and the Santiago Hills on the north. These hills force surface flow onto the central, flat Tustin Plain. The San Joaquin Hills and the Santiago Hills are steeply sloping (15 – 75 %) and composed of well-drained clays, and clay, sand, gravel, and cobble loams of the Alo-Bosanko and the Cienba-Anaheim-Soper associations. The flat (0 – 15 % slope) Tustin Plain is a collection of alluvial fans and flood plains. Historically, the Tustin Plain was the site of a large ephemeral lake and surrounding swampland. The soil associations of the Tustin Plain include Sorrento-Mocha and Metz-San Emigdio, well drained loams of various grain sizes; and Chino-Omni, poorly drained calcareous, silt loams and clays. The area around upper Newport Bay, a drowned river valley, is associated with Myford soils, moderately drained, sandy loams that developed on terraces.

3.5 Sediment

In the last 150 years the natural systems of the watershed have been severely altered by anthropogenic forces. Humans have altered the processes through which the Bay naturally replenished itself with fresh water and sand. Historically the Santa Ana River flowed into the Bay providing a major source of fresh water and the majority of the sediment entering the bay. This river also scoured sediment from the bay keeping its channels clear and deep during periods of heavy flow. In the 1920's when the Santa Ana River was diverted away from the bay to enter directly into the ocean, the sediment processes in the bay were permanently changed. No longer did the accumulated silt get washed out of the Bay by the rivers strong flow. Furthermore, landuse in the watershed was changing drastically as wetlands were drained and native areas were converted into farmland. Following World War II residential neighborhoods began to spring up around the area, particularly in the last two decades. Erosion increased as runoff and flow rates increased and the levels of sediment reaching the bay grew quickly (U.S. Army Corps of Engineers, 2001).

Historically sedimentation rates in Newport Bay were estimated to be approximately as slow as one inch of accumulation every 35-40 years. However, when the surrounding land uses were changed from ranching and agriculture to residential and commercial increasing erosion, while San Diego Creek was channelized, the sedimentation rate exploded to the current rates in which over 7 feet of sediment have been deposited in the last 12 years alone (USEPA, 1998). In the 1980's the scope of the sediment problem demanded study and action. The Southern California Association of Governments (SCAG), Orange County, local cities, the Irvine Company and California Department of Fish and Game formed a sediment committee to implement a sediment control plan (the 208 plan) (U.S. Army Corps of Engineers, 2001).

Presently it is estimated that approximately 250,000-275,000 tons of sediment are discharged into the watershed annually (USEPA, 1998). Of this, it is estimated that 50% is deposited in the bay itself and the other half within the network of tributaries. 94% of the sediment entering the bay comes from San Diego Creek. Currently the sediment within the upper Bay must be dredged to prevent choking and blockage with sediment nearly every year. The channels of San Diego creek also have to be cleared every few years to ensure the proper function as flood channels. While attempting to prevent habitat degradation by removing excess sediment this dredging is itself damaging to fragile ecosystems, not to mention its financial cost (U.S. Army Corps of Engineers, 2001).

Excessive sediment loading in Newport Bay adversely impacts the fragile habitats therein, and has been identified as the main threat to habitat health. It is this degradation, which violated water quality standards, that established the need for action to be undertaken. By reducing tidal flow the natural distribution of nutrients and oxygen are altered drastically; aquatic plants are covered by silt and sediment and die off and photosynthesis becomes impossible, benthic animals are unable to survive the lack of primary production, fish eggs are suffocated by collecting sediment, and habitats suffer as water depth changes. As habitats are degraded and become more homogenized by the accumulating sand, species diversity suffers (U.S. Army Corps of Engineers, 2001).

The OP pesticides are relatively insoluble in water, particularly Chlorpyrifos, which means that little will be found dissolved in storm water. Due to the affinity of these pesticides for sediment, both of them will bind to sediment particles. A more detailed discussion on the fate and transport of these pollutants is presented in Section 4. When these particles move through the watershed they transport the pesticide with them, delivering it to habitats where it can cause ecological damage. Once the pesticide dissolves into the water, it is very difficult and expensive to extract it using remediation technologies. Therefore, in order to control toxicity in Newport Bay, it is critical to control sediment loading from the land uses where pesticides are applied; by reducing sediment loads, pesticide loads will be similarly reduced.

In order to ensure proper management of the sediment problem in both the Bay itself and its tributaries, a total maximum daily load (TMDL) for sediment is being determined for both San Diego Creek and Newport Bay. The TMDL for sediment aims to bring about a 50% reduction in sediment load being discharged to the watershed. This goal will ensure that the habitats are protected from excess sediment and can return to their more historical sedimentation rates. Furthermore, dredging will be reduced from a nearly annual event to initially a once per decade occurrence to once every 20-30 years once the sediment target is met. This will ensure that the habitats are not disturbed as often and it will save large amounts of money from remediation costs eliminating sediment and pesticide bound to sediment from the Bay.

3.6 Habitat

Newport Bay Watershed includes ten major habitat types:

- scrub
- chaparral
- grassland
- vernal pools
- seeps and meadows
- marsh
- riparian
- woodland
- cliff and rock
- marine and coastal

The watershed's riparian communities are further classified into eight groups: floodplain sage scrub; coastal fresh water marsh; riparian herb; southern willow scrub; mulefat scrub; southern sycamore riparian woodland; southern arroyo willow forest; and southern black willow forest. Riparian and freshwater vegetation currently occurs along the Santa Ana-Delhi Channel and around San Diego Creek. Some riparian habitat is also found in Big Canyon, at North Star Beach, on the west side of the Bay near Irvine Avenue, and on Shellmaker Island (U.S. Army Corps of Engineers, 2000).

The cliffs, bluffs and mesas above the upper Bay are upland habitat, which support annual grasses and weeds introduced by past grazing practices. Dry slopes are characterized by coastal sage scrub, which includes bush sunflower, prickly pear, and black sage. North facing slopes support large shrubs such as lemonadeberry and toyon.

The lowlands of Newport Bay include seven major habitat types and are home to hundreds of plant and animal species. The upper Bay is further subdivided into three segments:

- Segment 1 is the furthest from the ocean and contains the mouth of San Diego Creek, as well as about a mile of upper Bay, including marshes and mudflats.
- Segment 2 is characterized by high bluffs on either side of the Bay, as well as mudflat and marsh islands within the water body.
- Segment 3 adjoins the lower bay has development more proximate to the water.

The Bay contains the highest fish diversity of the seven major coastal embayments between San Diego and Point Conception, including California Halibut, Topsmelt, Sand Bass, Anchovy and Gobies. The upper Bay, which is a State Ecological Reserve, contains a number of habitats, including salt marshes and a number of small islands both natural and man-made (U.S. Army Corps of Engineers, 2000).

3.7 Species

Newport Bay is home to numerous aquatic and terrestrial plant and animal species. The Bay supports many marine resources, including both invertebrates and fish populations, as well as land-dwelling flora and fauna, that continue to maintain existence in the increasingly urban landscape of southern California.

Several of the species inhabiting Newport Bay are listed as threatened and/or endangered at the state or federal level. The saltmarsh bird's beak (*Cordylanthus maritimus*) is a state and federally listed endangered plant that grows in the Bay. The arroyo southwestern toad (*Bufo californicus*) is an amphibian in the Bay that faces declining populations, and the Pacific pocket mouse (*Perognathus longimembris pacificus*) is a federally listed endangered mammal. Several insect species that inhabit the Newport Bay are considered sensitive insect species, although they are not yet listed.

High numbers of bird species occupy the Bay, making it an important habitat for both year-round residents and migrational species. Approximately one-hundred and eighty-two bird species regularly inhabit the Bay, of which thirty-three species are year-round inhabitants. Newport Bay provides resting habitat during migration and an over-wintering environment for many bird populations that are important throughout North America, South America and Hawaii.

Threatened or endangered bird populations in the Bay include the snowy plover (*Charadrius alexandrinus nivosus*), light-footed clapper rail (*Rallus longirostris levipes*), California black rail (*Laterallus jamaicensis coturniculus*), California least tern (*Sterna antillarum browni*), American peregrine falcon (*Falcon peregrinus*), Belding's savannah sparrow (*Passerculus sandwichensis rostratus*), California brown pelican (*Pelecanus occidentalis californica*) and the California gnatchatcher (*Polioptila californica*). The Newport Bay serves as vital habitat for many of these birds, such as the light-footed clapper rail, which is found only in coastal marshes of southern California and Baja. The Bay supports the highest numbers of light-footed clapper rails found in any southern California wetland, and is believed to be the only viable sub-population remaining in the United States. Newport Bay serves as an important nesting ground for several of the birds, such as the American peregrine falcon and California least tern, making the area an especially critical component of the future survival of these species (USACE, 2000).

The extreme sensitivity of birds to pesticide pollution makes the high number of threatened or endangered bird species in Newport Bay especially significant. Diazinon, in particular, is highly toxic to birds. For example, tests show that birds grazing on treated lawns for 15-80 minutes can receive a lethal dose. One major route of exposure for birds is the ingestion of insects and other invertebrates that contain diazinon residues. Reproduction studies suggest that diazinon exposure reduces both the clutch-size and number of surviving hatchlings when it is fed to birds at sub-acute doses. Further, diazinon indirectly affects the health of bird species by reducing diversity and numbers of aquatic prey items (ABC, 2001).

3.8 Landuse

Newport Bay Watershed is highly urbanized and this trend will continue as agricultural land is converted to residential, commercial, and light industrial uses due to population pressures. Almost the entire western portion of the watershed is developed and the eastern portion is quickly following suit. It is believed by Orange County Public Facilities & Resource Department that Newport Bay Watershed will reach full buildout within 20 to 50 years (see Figures 3-3 and 3-4). Buildout is associated with the development of an area according to a landuse plan. However, redevelopment and intensification will be continuous. The only areas protected from future development are mostly part of the Natural Community Conservation Plan and Habitat Conservation Plan (NCCP/HCP) Reserve system and are mainly located in the foothill regions of Santiago and San Joaquin Hills and around upper Newport Bay.

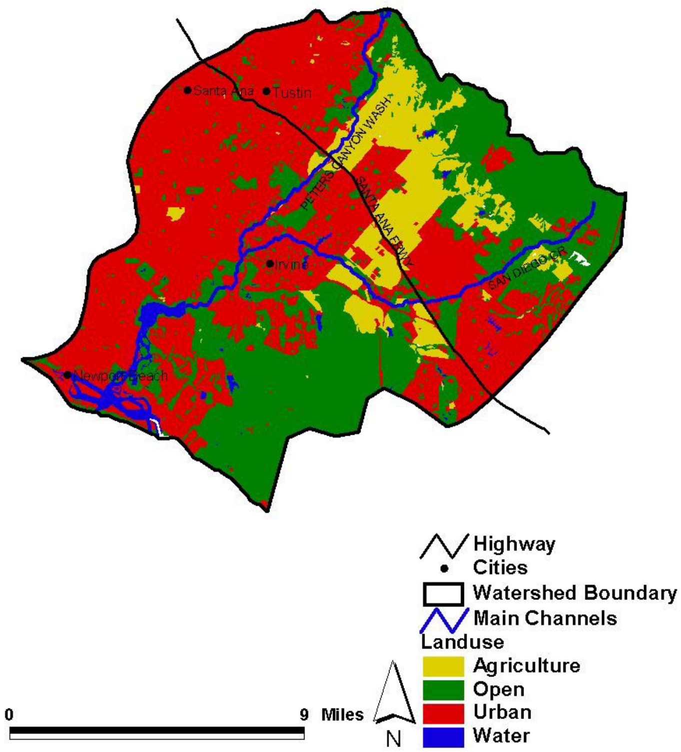


Figure 3-3. Major Landuse Designations Within Newport Bay Watershed. Open lands are considered mountains, parks or undeveloped.
Source: Adapted from SCAG (1993) data.

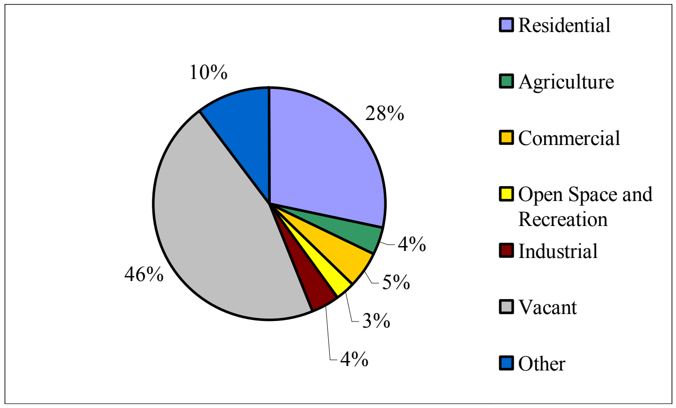


Figure 3-4. Percentage of the Major Landuses Within Newport Bay Watershed. Vacant lands are considered mountains or undeveloped.
Source: Adapted from SCAG (1993) data.

3.9 Beneficial Uses

The Newport Bay watershed provides residents and visitors to the area recreational opportunities that are largely unavailable in other portions of southern California's increasingly urban landscape. The Upper Bay's San Diego Creek sub-watershed is a popular recreational destination, offering hiking, walking, biking, mountain biking, equestrian riding and nature appreciation (U.S. Army Corps of Engineers, 2001). Lower Newport Bay provides about 600 acres of water for boating and tourism (Chang, 2000). In all, the recreational facilities of Orange County include approximately 27,000 acres of recreation harbors, beaches, parks and historical sites, the majority of which are within the Newport Bay Watershed (U.S. Army Corps of Engineers, 2001).

According to the SARWQCB, beneficial uses for waters within the Newport Bay Watershed include contact and non-contact water recreation, commercial or sportfishing and shellfish harvesting. Swimming, wading and skin or scuba diving involve body contact with water, and also make ingestion of water a reasonable possibility. Uses that would allow collection of organisms from the Bay for human consumption are also common in the Bay. The SARWQCB hopes to prevent the discharge of toxic substances at levels that will affect these beneficial uses, including the bioaccumulation of toxic substances in certain animals at levels that are harmful to humans who eat them (U.S. Army Corps of Engineers, 2001). Thus, the discharge of Diazinon or Chlorpyrifos at levels that would be unhealthy for humans when ingested, consumed through fish or absorbed through contact with water should be eliminated.

4.0 FATE AND TRANSPORT

4.1 Conceptual Model

Models play an important role in the development of water quality management plans because they represent the scientific understanding of physical, chemical, and biological processes. Available models range from complex, quantitative models to simpler, more conceptual models. A conceptual model can be used to illustrate the impacts of OP-pesticides on a hypothetical creek. For the purposes of our management plan, a conceptual model will first be used to illustrate the behavior of OP-pesticides in the environment (Figure 4-1). The model frames the following discussion of the physical and chemical properties of Diazinon and Chlorpyrifos, their transport to surface waters and their fate in the environment. We will then discuss in sections 7 and 8 the application of two quantitative models to study various loading scenarios and management alternatives.

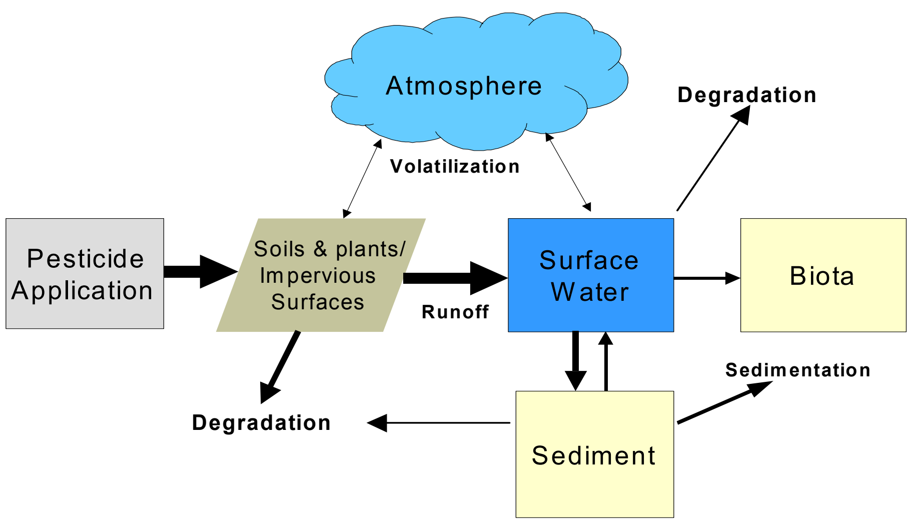


Figure 4-1. Simple Model That Represents the Essential Fate and Transport Processes for OP Pesticides in the Environment.

4.2 Environmental Properties - Diazinon

Diazinon ($C_{12}H_{21}N_2O_3PS$) is an organic compound with a molecular weight of 304 g/mol. Diazinon has a low Henry's Law Constant (Table 4-1), indicating that it is not very volatile and tends to stay in soil or water rather than escaping to the atmosphere. Diazinon has a fairly high octanol-water partition coefficient (K_{OW}) of 2000, and thus tends to adhere to sediment and organic matter (U.S.EPA, 2000). Diazinon's relatively high water solubility, compared to the levels associated with toxicity, leads to significant mobility of Diazinon in runoff from irrigation or rainfall. Therefore, despite its tendency to adhere to soil and sediment it is not contained locally at application sites. A relatively small fraction (about 2%) of the amount applied has been found to reach surface water (Scanlin and Feng, 1997). This small fraction results in aquatic toxicity observed in urban creeks. Table 4-1 summarizes the properties of Diazinon that contribute to its mobility in the environment.

Table 4-1. Chemical and Physical Properties of Diazinon at $25^{o}C$ That Contribute to its Mobility in the Environment.

Diazinon	
Molecular Formula	$C_{12}H_{21}N_2O_3PS$
Molecular Weight (g/mol)	304.36
Density (g/cm^3)	1.11
Water Solubility (mg/L)	40
Vapor Pressure (Pa)	0.0004
Henry's Constant, K_H (atm m^3/mol)	$1.09*10^{-7}$
Kow (log)	3.81
Koc (log)	2.28
Half-Life: Air	4.1 d
Water	43 d
Soil	50 d
Biota	32 d

Source: Watts (1998) and Lee (1998).

4.3 Environmental Fate - Diazinon

Diazinon's chemical properties and the physical features of the watershed determine its fate in the environment. Persistence of Diazinon at the application site and throughout a watershed depends on physical, chemical and biological factors, including temperature, humidity, light, soil and water pH, and microbial activity. Diazinon applied outdoors to soils, plants, and impervious surfaces reaches biota by way of surface water.

4.3.1 Degradation

Most of the Diazinon applied to soil and lawn surfaces breaks down in the soil before it reaches the storm drains and is not found in surface water. In soil, microbial degradation is the major route of Diazinon decomposition. Soil decomposition rates range from 2 to 4 weeks, though Diazinon 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. Photolysis is only an important degradation pathway in soils and on impervious surfaces when Diazinon is exposed to significant amounts of sunlight. 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 hydrolizes 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 photolysyis in water (U.S.EPA, 2000). The degradation rates for Diazinon are shown in Table 4-2.

Table 4-2. Degradation Constants for Diazinon.

Degradation Process	**Diazinon $t_{1/2}$ (d)**	**Diazinon k_r (day^{-1})**
Photolysis:	>150	<0.0046
Hydrolysis: (pH = 5)	12	0.0578
(pH = 7)	138	0.0050
(pH = 9)	77	0.0090
Biodeg.: (aerobic)	37	0.0187
(anaerobic)	34	0.0204

4.3.2 Volatilization

To some extent, Diazinon may evaporate from impervious surfaces or during spray applications. Because Diazinon is not especially volatile, air transport is relatively unimportant in urban areas. Atmospheric losses may occur, but studies show that most airborne Diazinon appears to redeposit locally (U.S.EPA, 2000), typically within the same watershed where it was applied. For this reason, Diazinon volatilization and deposition do not significantly affect Diazinon concentrations in urban creeks.

4.3.3 Sedimentation

Diazinon 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 Diazinon sink. Sediment leaching experiments performed by Alameda County found that Diazinon might also be re-suspended into the water column, making sediment a potential Diazinon 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 Diazinon in their sediment, or in creeks where water flows slowly over a long stretch of Diazinon-laden sediment (URS, 1999b). The leaching process for Diazinon from sediment to surface water has not been fully characterized.

4.3.4 Quantification of Environmental Fate

The fugacity based Environmental Equilibrium Partitioning Model (EQC) model version 1.01 (Canadian Environmental Modelling Centre, 1997), which was created by Trent University, was used to quantify the partitioning of Diazinon within a hypothetical environment to signify the behavior of the pesticide. The EQC model quantifies a chemical's behavior based upon its chemical-physical properties. The model includes the following media: air, water, soil, sediment, aerosols and suspended sediment. EQC 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 (Canadian Environmental Modelling Centre, 1997). According to the EQC model results, the majority of Diazinon partitions into the soil, but a significant amount of the pesticide also partitions into the water. The results of the EQC model are posted in Figure 4-2.

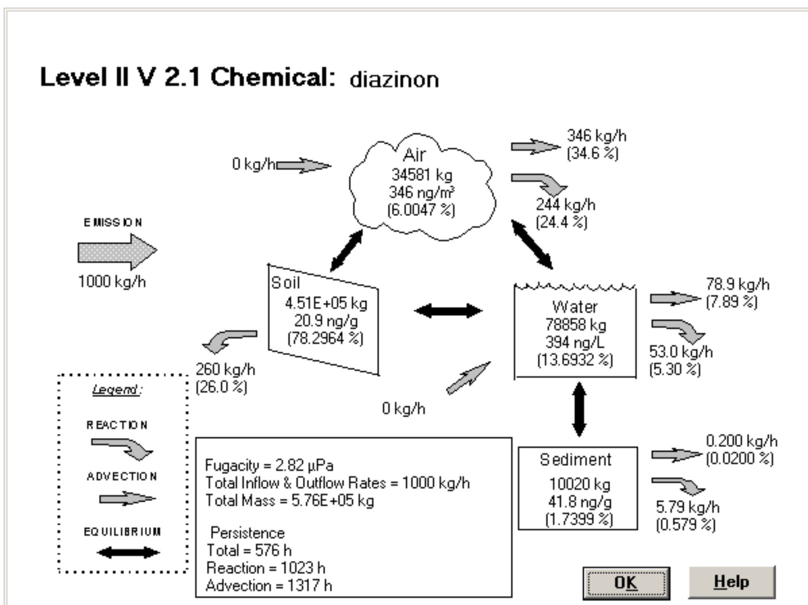


Figure 4-2. Diazinon Partitioning in the Environment, Modeled by EQC Level II.

Figures 4-1 and 4-2 illustrate the essential fate and transport processes and the partitioning behavior for Diazinon in the environment. The partitioning behavior is based on Diazinon's fugacity, or tendency to escape to the air, liquid or solid phases. As shown by the EQC diagram in Figure 4-2, Diazinon tends to partition strongly to soil and water and to a lesser degree, volatilizes to the atmosphere and binds to sediment. This behavior is strongly controlled by its' moderate solubility, low volatility (Henry's constant value) and low adsorption coefficient (K_{oc}) presented in Table 4-1. The simple EQC diagram and conceptual model indicate that Diazinon will be most commonly found in the soil and water. This general assumption is supported by sampling results presented in section 5.3.

4.4 Environmental Properties – Chlorpyrifos

Chlorpyrifos ($C_9H_{11}C_{13}NO_3PS$) is an organic compound with a molecular weight of 350.57 g/mol. This high molecular weight indicates that it would be likely to settle in the sediments due to the effects of gravity (Lee, 1998). Like Diazinon, Chlorpyrifos has a small Henry's Law Constant (Table 4-3), indicating that it is not highly volatile and is likely to remain trapped in soil or water rather than escaping to the atmosphere. Its tendency to adhere to soil is increased by its high octanol-water partition coefficient (K_{ow}). Chlorpyrifos has a lower water solubility than Diazinon and is less likely to partition into water. Thus, Chlorpyrifos loading in Newport Bay is likely to occur through sediment transport into the Bay. Table 4-3 summarizes the properties of Chlorpyrifos that contribute to its partitioning into different media in the environment.

Table 4-3. Chemical and Physical Properties of Chlorpyrifos at 25^o C That Contribute to its Mobility in the Environment.

Chlorpyrifos	
Molecular Formula	$C_9H_{11}C_{13}NO_3PS$
Molecular Weight (g/mol)	350.57
Density (g/cm^3)	1.4
Water Solubility (mg/L)	2
Vapor Pressure (Pa)	0.0027
Henry's Constant, K_H (atm m^3/mol)	$1.23*10^{-5}$
Kow (log)	5.11
Koc (log)	3.73
Half-Life: Air	N/A
Water	N/A
Soil	42 d (muck)
Biota	N/A

Source: Watts (1998) and Lee (1998).

4.5 Environmental Fate - Chlorpyrifos

As with Diazinon, the fate of Chlorpyrifos released into the urban environment is dependant upon both its chemical and physical properties. The persistence of Chlorpyrifos at the application site and throughout a watershed also depends on physical, chemical and biological factors of the watershed, including temperature, humidity, light, soil and water pH, and microbial activity. Figure 4-1, introduced in section 4.3 as a simple model that represents the essential fate and transport processes for Diazinon in the environment, also applies to Chlorpyrifos. Chlorpyrifos applied outdoors to soils, plants, and impervious surfaces reaches biota largely by way of sediments.

4.5.1 Degradation

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 have 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 degradation rates for Chlorpyrifos are given in Table 4-4.

Table 4-4. Degradation Rates for Chlorpyrifos.

Degradation Process	Chlorpyrifos $t_{1/2}$	Chlorpyrifos k_r (day^{-1})
Photolysis:	>150	<0.0046
Hydrolysis: (pH = 5)	73	0.0095
(pH = 7)	72	0.0096
(pH = 9)	16	0.0433
Biodeg.: (aerobic)	11-141	0.0630-0.0049
(anaerobic)	37	0.0187

4.5.2 Volatilization

The volatilization half-life of Chlorpyrifos in a flowing river has been estimated as 5.7 days. Once sorbed to the soil, Chlorpyrifos does volatilize; however, the significance of volatilization from the system is decreased greatly by aquatic sediment adsorption, making volatilization a relatively insignificant exposure pathway (Spectrum Laboratories, 2001).

4.5.3 Sedimentation

Once released into the water, Chlorpyrifos partitions significantly from the water column to sediments (Spectrum Laboratories, 2001). Because of its low water solubility and high soil binding capacity, it is common for Chlorpyrifos to sorb to soil and runoff into surface water via erosion. It is through this pathway that wildlife exposure to Chlorpyrifos occurs, creating the potential for bioaccumulation in fish and other aquatic organisms and thus entry into the food web. Once aquatic

Chlorpyrifos exposures cease, the pesticide rapidly depurates from fish (U.S.EPA, 2001).

4.5.4 Quantification of Environmental Fate

The fugacity based Environmental Equilibrium Partitioning Model (EQC) model version 1.01 (Canadian Environmental Modelling Centre, 1997), was also used to quantify the partitioning of Chlorpyrifos within a hypothetical environment. The model quantifies the chemical's behavior based upon its chemical-physical properties. The model includes the following media: air, water, soil, sediment, aerosols and suspended sediment. EQC simulates the major pathways of a substance released into the environment, including degradation, volatilization and sedimentation. EQC Level II assumes that thermodynamic equilibrium is achieved, and (unlike EQC Level I) includes advection and reaction processes (Canadian Environmental Modelling Centre, 1997). The results of the partitioning behavior of the EQC model are posted in Figure 4-3.

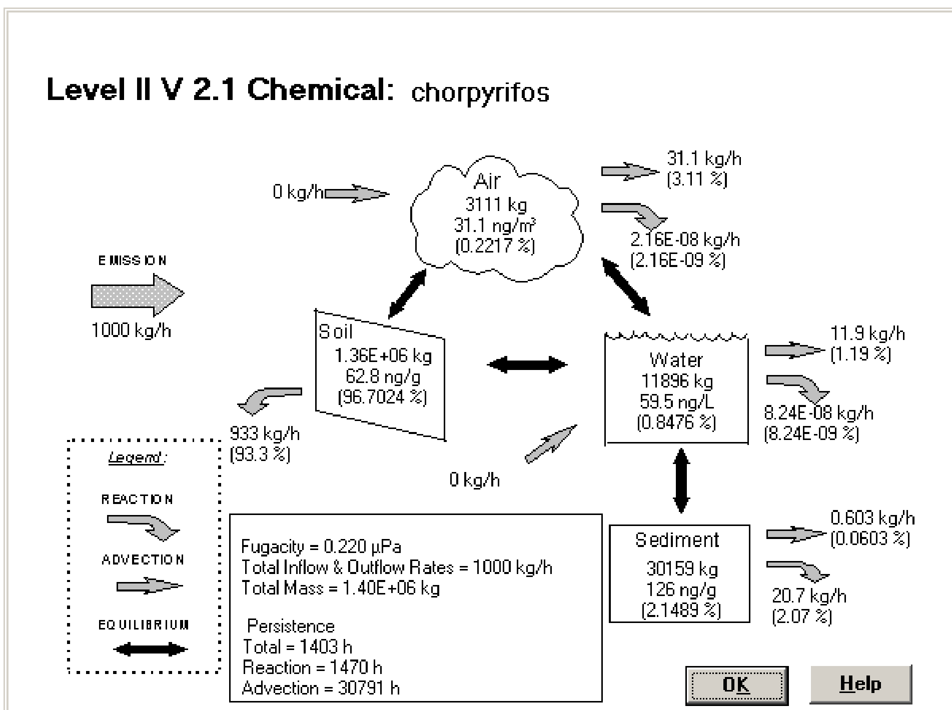


Figure 4-3. Chlorpyrifos Partitioning in the Environment, Modeled by EQC Level II.

Figure 4-3 illustrates the essential fate and transport processes and the partitioning behavior for Chlorpyrifos in the environment. As shown by the EQC diagram in Figure 4-3, Chlorpyrifos tends to partition strongly to soil and to a lesser degree, volatilizes to the atmosphere and binds to sediment. Very little Chlorpyrifos partitions into the water. This behavior is strongly controlled by its high octanol/water-partitioning coefficient, or Kow, presented in Table 4-3. The simple

EQC diagram and conceptual model indicate that Chlorpyrifos will be most commonly found in the soil upon release into the environment, in levels higher than Diazinon. Additionally, Chlorpyrifos has a lower tendency to partition to the aqueous or gaseous phases. This general partitioning behavior is supported by the sampling results presented in sections 5.3.

4.6 Transport to Stream Channels

In urban areas, small streams are greatly affected by the surface runoff collected in storm drains. The initial releases of OP pesticides occur during structural pest control, landscape maintenance, and other outdoor uses involving applications to soils and plants and paved areas (e.g., sidewalks, driveways and patios). Some pesticides may be released into the air, but like the application on plants, soils, and paved surfaces, the pollutant is then transported in surface runoff to storm drains during rain events or irrigation.

5.0 PESTICIDE SOURCES

One of the primary objectives of the current project is to gain insight into the significance of different landuses with respect to pesticide source loading. The sources of OP pesticides in the Newport Bay and San Diego Creek watersheds have been identified by sampling storm and base flows across different landuses and analyzing water and sediment concentrations (Lee and Taylor, 2001). In this section we offer a synthesis of what is known regarding pesticide usage, spatial distribution and loading per landuse within the basin. The goal of the pesticide source section is to characterize the principal sources of OP pesticides and their spatial distribution.

Most of the OP pesticides applied in Newport Bay adhere to surfaces, degrade in the environment, and are not found in surface water. However, a relatively small fraction does reach surface water. This fraction, estimated to be about 0.24% of the pesticides applied outdoors, is responsible for the aquatic toxicity observed in San Diego Creek and it's tributaries as well as in the Bay itself (SARWQCB, 2001).

5.1 Pesticide Usage

OP pesticide toxicity in Newport Bay is likely derived from residential and agricultural usage, where it is applied as an insecticide. High levels of Diazinon are also 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. 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). These results indicate that urban uses of OP pesticides account for over 90% of the load, while agricultural uses (including nurseries) accounted for the remainder. 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.

5.1.1 Seasonal Variation in Pesticide Concentrations

Pesticide concentrations in Newport Bay appear to vary seasonally, peaking in the fall, declining in the winter and rising again in the spring. This variation coincides with seasonal application rates. Data provided by the Department of Pesticide Regulation (DPR) indicate that pesticide application rates are higher during the drier, summer months corresponding to the pest life cycle (DPR, 1998). This data is shown in Figures 5-1 and 5-2. 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.

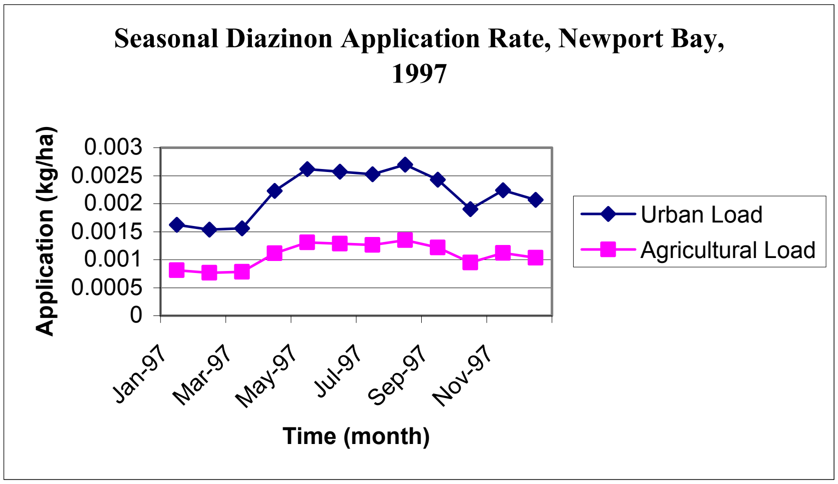


Figure 5-1. Seasonal Diazinon Application Rates.
Source: DPR (1998)

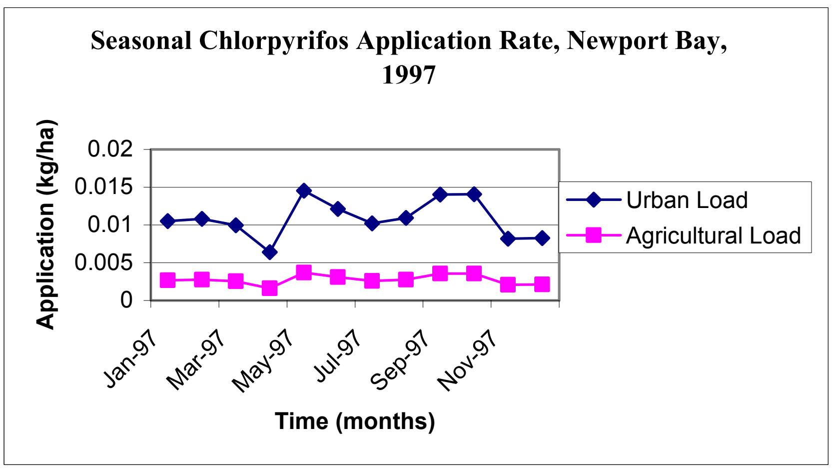


Figure 5-2. Seasonal Chlorpyrifos Application Rates.
Source: DPR (1998)

It should be noted that both pesticides occur in water samples during wet and dry cycles, which points to it's persistence in stormflow and baseflow conditions.

5.1.2 Diazinon Usage

Diazinon and Chlorpyrifos are commonly associated with urban and agricultural landuses. In Newport Bay, urban uses account for roughly 90% of the overall application, with agricultural uses (including nurseries) accounting for the remainder (SARWQCB, 2001).

Table 5-1 illustrates an increase in usage of Diazinon for urban applications and a decrease in agricultural usage over a five-year period. The landuse data also show a similar pattern, which is a general decline in agricultural area and in increase in urbanization over this time period (SARWQCB, 2001). Structural pest control uses accounts for an average of 60% of the total load annually, residential uses account for the approximately 35% of the usage within the watershed.

Table 5-1. Diazinon Use in Newport Bay (lbs 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.

5.1.3 Chlorpyrifos Usage

Table 5-2 shows the usage of Chlorpyrifos in Newport Bay from 1995-1999. Nursery and agricultural uses account for roughly 10% of total load; therefore residential and urban areas account for the majority of Chlorpyrifos loading. The increase in usage by nurseries in 1999 can likely be attributed to the requirements imposed by the California Department of Food and Agriculture (CDFA) Red Imported Fire Ant (RIFA) eradication program. The CDFA RIFA plan requires treatment of targeted areas with both Diazinon and Chlorpyrifos (CDPR, 1999).

Table 5-2. Chlorpyrifos Use in Newport Bay (lbs 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

As is the case with Diazinon, structural pest control, primarily in the treatment of wood protection from termites, is the largest application of Chlorpyrifos.

5.2 Spatial Distribution of Pesticides

5.2.1 Diazinon Distribution and Dominant Landuse

The results from the 319(h) grant indicate that the entire Upper Newport Bay watershed contributes to the Diazinon loading, with less contribution from certain specific areas (Lee and Taylor, 2001). The station at Campus drive (Figure 5-3, 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).

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 5-3 and the sampling station attributes are shown in Table 5-3.

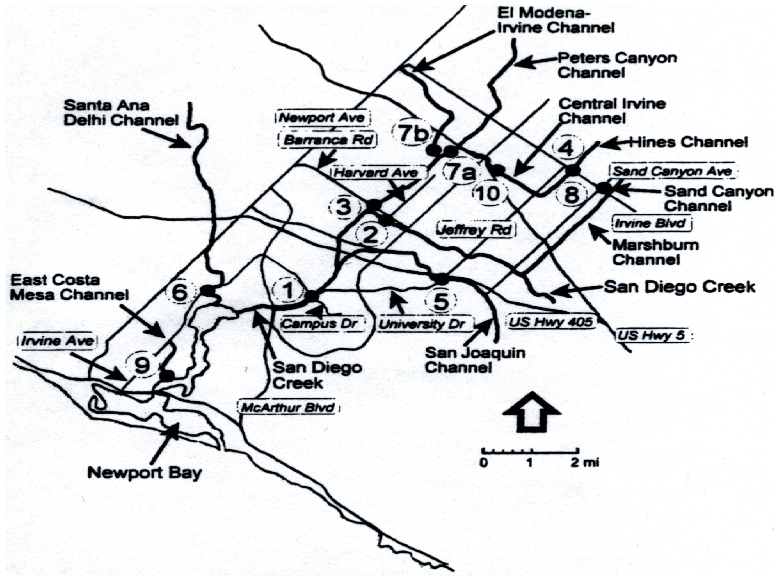


Figure 5-3. OP Pesticide Sampling Distribution.
Source: (Lee and Taylor, 2001)

5.2.2 Chlorpyrifos Distribution and Dominant Landuse

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 5-3. 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

Source: Lee and Taylor (2001)

5.3 Pesticide Sampling Results

5.3.1 Diazinon Sampling Results

Table 5-4 shows the sampling results by Lee and Taylor, conducted under the 319(h) study during the 1998 water year. 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 could be a result of Diazinon's tendency to partition into the aqueous phase more readily than to it adsorbs to sediments (Figure 4-3).

Table 5-4. Newport Bay Watershed Diazinon Sampling Results (1998).

Source	Detection Frequency	Min.	Max.	Avg.	Median
Water (ng/L)		(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 (ug/kg)					
Drainage Channels	2%	<10	49	---	---
Newport Bay	3%	<0.4	60	---	---

Freshwater	CDFG	EPA
CCC	50	n/a
CMC	80	90

Source: Results after SARWQCB (2001).

The results are not in agreement with studies done in Alameda County, which 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 (URS, 1999b), 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.

Toxicity levels in core sediment samples from the Bay are currently being determined by Steven Bay and other researchers at the Southern California Coastal Watershed Restoration Project (SCCWRP) and should be published later this year. This study will determine the degree to which sediments serve as a sink for Diazinon and/or Chlorpyrifos as well as a source of toxicity through remobilization of sediment to the water column.

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) discusses in Section 2.3. Diazinon was detected in stream channels 93% of the time and in Newport Bay in every sampling event conducted under the 319(h) study. In Section 6.0, we present a detailed discussion of the aquatic life toxicity in the Bay.

5.3.2 Chlorpyrifos Sampling Results

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 5-5). 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 5-5. Newport Bay Watershed Chlorpyrifos Sampling Results (1998).

Source	Detection Frequency	Min.	Max.	Avg.	Median
Water (ng/L)		(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

Source: Results after SARWQCB (2001).

The median concentrations of Chlorpyrifos in water and sediment obtained during baseflow and stormflow exceeded the CDFG'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.

5.4 Pesticide Export Load Per Landuse

Pesticide loading is largely a non-point 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 5-6 and 5-7 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).

5.4.1 Diazinon Export Loading

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, 1999b), 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 5-6. Diazinon Export Load per Landuse.

Condition	LandUse	Area		Load		Load
		(acres)	(%)	(lbs)	(%)	(lbs/acre)
Baseflow	urban	66,507	68%	2.4	88%	3.6E-05
	agriculture	9,286	10%	0.31	12%	3.4E-05
	Total	97,741	100%	**2.7**	100%	2.8E-05
Stormflow	urban	66,507	68%	24.1	96%	3.6E-04
	agriculture	9,286	10%	2.47	4%	2.7E-04
	Total	97,741	100%	**26.6**	100%	2.7E-04

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.

5.4.2 Chlorpyrifos Export Loading

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 5-7. Chlorpyrifos Export Load per Landuse.

Condition	LandUse	Area		Load		Load
		(acres)	(%)	(lbs)	(%)	(lbs/acre)
Baseflow	urban	66,507	68%	0.69	88%	1.03E-05
	agriculture	9,286	10%	0.1	12%	1.03E-05
	Total	97,741	100%	**0.78**	100%	8.01E-05
Stormflow	urban	66,507	68%	2.61	85%	3.92E-04
	agriculture	9,286	10%	0.46	15%	4.90E-04
	Total	97,741	100%	**3.06**	100%	3.13E-04

Source: Results after SARWQCB (2001).

5.5 Summary of Source Analysis Sampling Results

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 6 pounds of Chlorpyrifos and 35 pounds of Diazinon are annually discharged to Upper Newport Bay. 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 (Tables 5-1 and 5-2). Results from Lee and Taylor (2001) found that on average, about 1 to 2 lbs. of Diazinon and 1 to 1.5 lbs of Chlorpyrifos are discharged to the Upper Bay during a typical storm event of 1 to 2 inches.

As shown in Sections 5.4.1 and 5.4.2, 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, outlined in Section 2.2, will likely end a significant percentage 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.

6.0 ECOLOGICAL AND HUMAN RISK

6.1 Toxicology

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).

Toxic pollutants affect organisms through both chronic and acute toxicity. Chronic toxicity occurs when the death of organisms results from a prolonged exposure to the toxin, while acute toxicity occurs when death follows a short and often intense exposure (Newman, 1998). Due to the short half-lives of both Diazinon and Chlorpyrifos, Tables 4-1 and 4-3, and the general patterns of pesticide loading, in which flushes of pesticides are released in either stormflow or dry month nuisance flow (runoff created by humans, eg., runoff from residents washing cars), these OP pesticides do not generally persist in the watershed for longer than seven days (EXTOXNET, 1996). The half-life, or amount of time that it takes for half of a substance to dissipate, for Diazinon in animals is only 12 hours (Ladaa et al, 1998) and the total degradation time for Chlorpyrifos can be as low as 9 hours (US Fish and Wildlife Service, 1988). Thus, Diazinon and Chlorpyrifos exposure does not generally result in chronic toxicity for organisms in the natural environment. Further, based upon studies done on rats, these pesticides have not been found to pose a carcinogenic or mutagenic risk (EXTOXNET, 1996). Thus, the analysis of ecological and human risk posed by the pesticides Diazinon and Chlorpyrifos in this study involves only acute exposure.

6.1.1 Ecological Toxicity

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 1968 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. And a 1970 study by Hurlbert et al. found that 4 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).

6.1.2 Human Toxicity

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).

Recreationalists in Newport Bay face intake of Diazinon and Chlorpyrifos during recreational activities. The Newport Bay area offers opportunities for walking, boating, rowing, swimming and fishing, all of which create the potential for pesticide contact by recreationalists (The California Environmental Resources Evaluation System, 2001). 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). These activities create pesticide intake pathways for humans who either consume fish from the Bay or intake water orally during recreation. Dermal contact with water from the Bay also poses a risk for recreationalists. Dermal intake of Diazinon and Chlorpyrifos occurs through skin absorption as a result of contact with the sediments and waters of the Bay (EXTOXNET, 1996).

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 (Skinner, 2001). Diazinon and Chlorpyrifos bioaccumulate rapidly in aquatic organisms as a result of their chemical properties. Due to the high Log K_{ow}, or octanol-water partitioning coefficient, of Diazinon and Chlorpyrifos (3.3 and 4.7, respectively), the pesticides partition largely into soil or organic matter when released into the environment (EXTOXNET, 1996). Chlorpyrifos shows a particularly high affinity for biota, as can be seen by the bioconcentration factors (BCF) for Diazinon and Chlorpyrifos in wet fish, with a BCF of 540 liters of Diazinon in each kilogram of fish and 2700 liters of Chlorpyrifos in each kilogram of fish (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 contaminates 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 micrograms/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 from Newport Bay during periods of high pesticide loading could result in risks to human health.

Residents of Newport Bay 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 (0.012 Pa and 0.0027 Pa, respectively) (U.S. Department of Larbor, 2001).

6.2 Risk Assessment

In order to examine the risks posed to ecological and human health in Newport Bay by Diazinon and Chlorpyrifos pollution, our group investigated the loading of these pollutants during both stormflow and baseflow conditions. The analysis was carried out using the RivRisk 4.0 computer model developed by Tetra Tech, Inc for the Electric Power Research Institute (EPRI). RivRisk is a modeling tool that was created to evaluate the effects of a riverside power plant upon the river receiving emissions from the plant. The model also provides calculation of pollution in a water system based upon user-specific inputs, which proved useful in our analysis. RivRisk performs fate and transport calculations for the movement of pollutants through a water body, which lays the groundwork for human health and ecological risk assessments. The model assesses the acute risks to species in the environment and humans based upon the level of toxicity of Diazinon and Chlorpyrifos and the exposure of organisms to those pollutants, using information from the EPA (Tetra Tech, 2002).

The Newport Bay Watershed was simulated by looking at the river function within RivRisk (excluding groundwater seepage and atmospheric deposition), and with the following river parameters: width of 10 meters, depth of 3 meters, flow rate of 100 cubic meters per second, transverse eddy diffusion coefficient of 0.05 meters squared per second and suspended sediments concentration of 50 milligrams per liter. The parameters for Newport Bay are summarized in Table 6-1. Pesticide concentrations were based upon the export loading data displayed in Table 5-6 for Diazinon and Table 5-7 for Chlorpyrifos. Values for urban land use were used in order to simulate the most conservative scenario in terms of ecological and human health risk and because the portions of the Bay where highest usage occurs is highly urbanized.

Table 6-1. Parameters Used for Water Bodies in the Newport Bay Watershed Used in RivRisk Assessment of Ecological and Human Risk.

Width (m)	**Depth (m)**	**Flow rate (m^3/s)**	**Transverse eddy diffusion coefficient (m^2/s)**	**Suspended sediments concentration (mg/l)**
100	5	100	0.05	50

6.2.1 Ecological Risk Assessment

Ecological risk parameters in RivRisk are based upon the EPA's water quality criteria for pollutants, which are expressed as chronic water quality criterion, or Criteria Continuous Concentration (CCC), and acute water quality criterion, or Criteria Maximum Concentration (CMC). Because the pesticides Diazinon and Chlorpyrifos are not considered to be carcinogenic (EXTOXNET, 1996), only the CMCs were used in this analysis. The CMC used for Chlorpyrifos was 4.1×10^{-5} mg/L and for Diazinon was 9×10^{-8} mg/L. These values are based upon water quality criteria for *Ceriodaphnia dubia*, commonly used in the development of water quality criterion for Chlorpyrifos and Diazinon (US Environmental Protection Agency III). RivRisk compared the concentration of each pesticide in the river to the water quality criteria for *Ceriodaphnia* in order to generate ecological hazard quotients (HQs). An HQ of greater than 1 was considered to pose an unacceptable acute health risk.

The test organism *Ceriodaphnia dubia* was used as the indicator organism for the ecological health of the Newport Bay because it is representative of aquatic organisms that serve as larval food in fresh and marine waters (Lee, 2000). Because the deaths of sensitive organisms like *Ceriodaphnia,* which serve as the base of the food chain, are significant to the health of those species that prey on them, such organisms serve as good indicators for the health of the whole system. For example, the US Fish and Wildlife stated that agricultural runoff contains contaminants such as herbicides, pesticides and fertilizers that affect the arroyo toad, an endangered species found in Newport Bay, both directly and indirectly through effects on sensitive prey species (U.S. Fish and Wildlife Service, 1999). Additionally, the fact that *Ceriodaphnia* is an extremely sensitive aquatic species makes it is safe to assume that water quality standards set to protect this organism will be stringent enough to protect other aquatic species as well.

Our group did consider lethal doses of Diazinon and Chlorpyrifos for other species as well. Fish species are sensitive to Chlorpyrifos, as evidenced by the low LC50, or lethal concentration at which 50% of the organisms in the sample are killed by a single exposure. The LC50 of Chlorpyrifos for rainbow trout is 3 ppb. The LC50 of Diazinon for the same species is 2600 ppb (CDPR, 2000). Bird species are highly sensitive to Diazinon. For example, the LD50s for three bird species are quite low: Canadian goose LD50 is 6.16 mg/kg, mallard LD 50 is 1.44 mg/kg and house sparrow LD50 is 7.5 mg/kg (American Bird Conservancy, 2001). These values provided an understanding of the ecological risk posed by Diazinon and Chlorpyrifos for animals in the Newport Bay Watershed, although they were not the focus of the RivRisk analysis completed for this project.

6.2.2 Human Risk Assessment

Our group examined the human health risk in Newport Bay through RivRisk's assessment of the threat posed by Diazinon and Chlorpyrifos. The assessment was based upon pollutant toxicity and the acute exposure of humans to the pollutants. As in the assessment for ecological risk, the risks were measured using an HQ. An HQ of greater than 1 is considered to pose an unacceptable acute health risk.

The risks posed to human health by Diazinon and Chlorpyrifos were analyzed based upon intake of the pesticides in Newport Bay. Residential risks included oral intake through fish and shellfish consumption, dermal contact with soil and sediments, and inhalation. Recreational risks included oral intake through fish and shellfish consumption and through ingestion of water while swimming, dermal contact with water while swimming and dermal contact with soil and sediments, and inhalation.

The exposure frequency through ingestion of fish for residents of Newport Bay was given as an average of 100 days per year, and for recreationalists in the Bay was given as an average of 25 days per year. These exposure frequencies are based on the assumptions that fish make up a high percentage of the diet of residents who eat local fish and that recreationalists eat fish from the Bay a high number of days each year. These assumptions do not hold true for the majority of residents and recreationalists in the Bay. However, this assumption does allow the consideration of a worst-case scenario in which low-income residents, who may have little other means of feeding their families, or recreationalists who enjoy high levels of fishing and fish consumption from the Bay, experience high exposure. By taking this conservative estimate into consideration, our group was able to assume that we analyzed the worst-case scenarios for fish consumption in the Bay.

The parameters given with RivRisk for dermal contact with soil and sediments were accepted: surface area of 5800 centimeters squared, soil adherence factor of 1 and exposure frequency of 350 days per year. The given parameters for intake during swimming were also accepted: for ingestion of water, the ingestion rate was 0.05 liters per hour, the exposure time was 1 hour per day and only the exposure frequency was changed to 25 days per year; for dermal contact, a surface area of 23000 centimeters and an exposure time of 1 hour per day, and the exposure frequency was changed to an average of 25 days per year. The parameters are summarized in Table 6-2.

Table 6-2. Parameters Used for Human Exposure in RivRisk Assessment of Human Risk.

Ingestion rate of water during swimming (L/hr)	Exposure time during swimming (1 hr/day)	Exposure frequency during swimming (days/yr)	Surface area of soil in dermal contact (cm)	Exposure time for dermal contact with soil (hr/day)	Exposure frequency for dermal contact with soil (days/yr)
0.05	1	25	23000	1	25

6.3 Results

6.3.1 Ecological Risk Assessment Results

The results of the RivRisk analysis suggest that in both stormflow and baseflow conditions, Diazinon and Chlorpyrifos pose an excessive risk to the health of aquatic organisms. Stormflow conditions present a greater risk to ecological health than do baseflow conditions (Tables 6-3 and 6-4).

Table 6-3. Resulting HQ Values for Acute Toxicity During Baseflow.

	Adult Resident	Child Resident	Recreation	Ecological
Diazinon, Baseflow	0.5	2.5	0.5	2300
Chlorpyrifos, Baseflow	0.7	7.1	0.7	1.2

Table 6-4. Resulting HQ Values for Acute Toxicity During Stormflow.

	Adult Resident	Child Resident	Recreation	Ecological
Diazinon, Stormflow	2.2	10.0	2.2	8900
Chlorpyrifos, Stormflow	5.6	26.0	5.6	10

6.3.2 Human Risk Assessment Results

Results from the analysis of human health risks posed by Diazinon and Chlorpyrifos indicate that in baseflow conditions, an unacceptable risk to human health risk is created by both Chlorpyrifos and Diazinon for child residents of the Newport Bay Watershed. This is shown by the HQ value of greater than 1 that resulted from each of the analyses of children residents (Table 6-3). Adult residents and recreationalists do not face excessive risks from either Chlorpyrifos or Diazinon in baseflow conditions, as is seen by the resulting low HQ values (Table 6-3). In stormflow conditions, the risk to both human and ecological health is greater than during baseflow conditions. During stormflow, an unacceptable risk to human health is created by Chlorpyrifos and Diazinon not only for child residents of the Bay, but also for adult residents and recreationalists in the watershed (Table 6-4).

6.4 Discussion

Our analysis suggests that aquatic species face unacceptable risk during both stormflow and baseflow conditions. Improvements to the ecological health of Newport Bay would require reducing toxicity in the water and sediments entering Newport Bay to meet the water quality criteria set by the EPA for the test organism *Ceriodaphnia*. The validity of basing cleanup goals on meeting water quality criteria set for *Ceriodaphnia* has come under criticism from several sources. Studies indicate that small planktonic organisms can be thousands of times more sensitive to certain pesticide contaminants than fish, crayfish, snails or mammals. Regulations designed to protect water quality based upon use of such tiny indicator species may be over-protective (Capitolink, 2001). Further, *Ceriodaphnia* are not native to Southern Californian waters. Rather, this species is found naturally in New Zealand and Australia (Contamsites, 2002). However, the fact that this species represents the small zooplankton that serve as the base of the food chain in Newport Bay waters, our group decided that using *Ceriodaphnia* as an indicator organisms for the health of the ecosystem in Newport Bay was valid.

Based upon the analysis carried out with RivRisk in this study, stormflow in the Newport Bay Watershed results in Diazinon and Chlorpyrifos concentrations in the Bay that are unacceptably high for human health standards. During storm events high levels of runoff carry the pesticides off the surface of the watershed and into the Bay, polluting the waters and making them dangerous to humans who use the waters for beneficial uses such as fishing or swimming. During periods of baseflow, in which runoff from the watershed comes from human uses of water (such as car-washing), conditions continue to pose an unacceptable health risk to child residents of the Bay. The difference between the effects of the pesticides on human health for child residents and for adult residents or recreationalists during baseflow conditions, when pesticide concentrations decrease, lies in the smaller body size of children. Because they weigh less, children are more susceptible to damage from Diazinon and

Chlorpyrifos, resulting in an unacceptable risk to their health even during periods of baseflow when pesticide concentrations in Newport Bay are lower.

The risk posed to human and ecosystem health by Diazinon and Chlorpyrifos loading in the Newport Bay Watershed indicates that management plans for the area should include reductions in pesticide input into the water system. Plans for reducing pesticide concentrations in the Bay should consider the fact that humans draw on the Bay for many beneficial uses, including fishing and swimming, as well as the species (many of which are endangered) that rely upon Newport Bay for existence. Various options for reducing pesticide loading are proposed in Section 9.0.

7.0 BASINS

7.1 Methodology

We implemented the Better Assessment Science Integrating Point and Nonpoint Sources (BASINS) software, version 2.0, to simulate the fate and transport of Chlorpyrifos, Diazinon, and sediment, through the Newport Bay watershed. The BASINS software was developed by TetraTech, Inc., for the U.S. Environmental Protection Agency's Office of Water. 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. The purpose of using the BASINS model was to evaluate alternative BMPs for OP pesticides and determine which practices were most appropriate for a watershed management strategy to reduce the concentration of these pesticides in Newport Bay.

Our approach can be divided into several steps. We began by compiling spatial data such as topography, drainage and channel network, and land-use information. We then developed a framework or schematic of the watershed by delineating subcatchments. Hydrologic parameters were adjusted and surface flow and sediment transport were calibrated using U. S. Geological Survey (USGS) gaging station data. Chemical properties of the organophosphate pesticides were incorporated, and pesticide application rates were assigned. Once the model represented the physical characteristics of the watershed, simulations were run evaluating watershed responses to changes in land-use and management practices.

7.1.1 Spatial Data

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).

For our purposes, GIS data for the Newport Bay watershed was downloaded from the EPA website "http://www.epa.gov/OST/BASINS". This information included the following site-specific GIS data as well as meteorological data for the state of California:

- USGS Hydrologic Cataloging Unit #18070204 core data
    1. 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 (original BASINS data set included 1977 landuse)
- Meteorological data recorded at the Los Angeles International Airport

7.1.2 Watershed Framework

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 gaging stations. The Newport Bay watershed was delineated and divided into 4 catchments (Figure7-1):

- Peter's Canyon Subcatchment with base at USGS gaging station #11048540
- Upper San Diego Subcatchment with base at USGS gaging station #11048500
- Intermediate San Diego Subcatchment with base at USGS gaging station #11048550
- Lower San Diego Subcatchment including Newport Bay and coastal boundary

USGS gaging station #11048540 is located on Peter's Canyon Channel at Barranca Parkway, upstream of the confluence of Peter's Canyon Channel and San Diego Creek. USGS gaging station #11048500 is located on San Diego Creek at Culver Drive, also upstream of the confluence of Peter's Canyon Channel and San Diego Creek. USGS gaging station #11048550 is located at Campus Drive, below the confluence of Peter's Canyon Channel and San Diego Creek, just upstream of where San Diego Creek enters upper Newport Bay. The lower San Diego Creek subcatchment includes several creeks that drain directly to the ocean and was not incorporated into our model simulations. Stream flow data was obtained from the USGS National Water Information System (NWIS) website (http://waterdata.usgs.gov/nwis-w/CA/).

The original framework for our model was based on recorded historical events, allowing us to compare predictions from model simulations to actual values. Original BASINS core data included landuse data from 1977. The landuse categories and their relative areas are presented in Table 7-1. All categories of landuse types were assigned the default values of 100% perviousness with the exception of the category for Urban and Built-up landuse, which was assigned 50% perviousness.

Table 7-1: 1977 Landuse for Initial Model Framework

	Upper San Diego		Intermediate San Diego		Peter's Canyon	
Landuse	Area (acres)	% of total	Area (acres)	% of total	Area (acres)	% of total
Urban or Built-up/pervious	6,735	24%	11,791	48%	6,440	32%
Agriculture	8,953	31%	9,371	38%	8,608	43%
Rangeland	10,824	38%	3,053	12%	4,359	22%
Forest	318	1%	0	0%	179	1%
Barren	1,686	6%	516	2%	398	2%
Total acres per subcatchment	**28,518**	100%	**24,731**	100%	**19,984**	100%

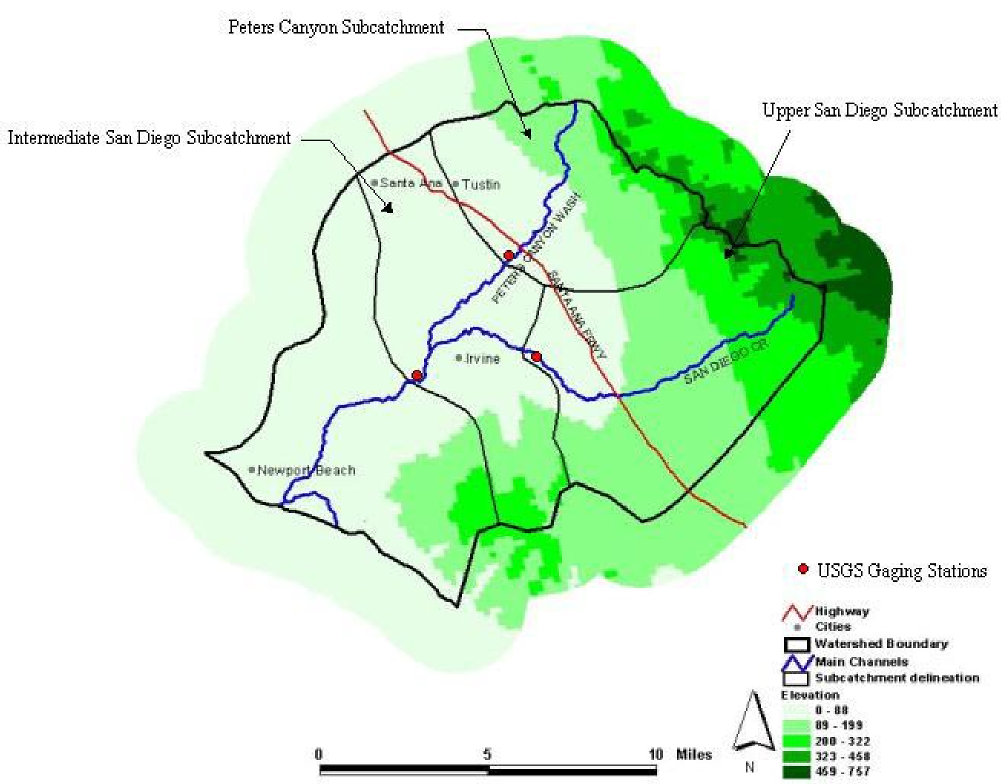


Figure 7-1. Newport Bay watershed delineation for BASINS simulations.

7.1.3 Model Calibration

The BASINS Nonpoint Source Model (NPSM) is an assessment tool that simulates nonpoint source runoff and pollutant loadings for the watershed and performs both flow and water quality analyses based on hydrologic parameters. We calibrated the model to simulate conditions experienced in the Newport Bay watershed over a period of 3 years, beginning October 1, 1982, and ending September 30, 1985. This period was chosen because it was the longest period of available, continuous, recorded stream flow for all subcatchments, and could be used concurrently with the 1977 landuse data. The Nonpoint Source Model (NPSM) was calibrated by adjusting various hydrologic and sediment parameters, and comparing simulated stream flow to actual stream flow recorded at the gaging stations. The hydrologic and sediment parameters pertained to vegetation, evapotranspiration, soil distribution, and characteristics that effect surface and subsurface flow. Calibrated parameters are presented in Appendix A; Table A-1.

The model fit was measured by comparing total flow over the period of calibration between the simulated and actual flow. The 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 gaging station #11048550)(Figure 7-2).

Sediment transport was calibrated by a combination of parameter value adjustments. The channel geometry for Peter's Canyon was made steep and narrow (Appendix A; Figure A-1), and the channel geometries for Upper San Diego Creek and Intermediate San Diego Creek were made flat and wide to replicate channelization (Appendix A; Figures A-2 and A-3). Variable definitions for channel geometries are given in Appendix A, Figure A-4. We selected a sediment distribution of 20% sand, 60% silt, and 20% clay. In spite of calibration efforts, the concentration of simulated suspended sediment remained lower than that found in the actual recorded data. 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.

7.1.4 Incorporation of Organophosphate Chemical Properties and Application Rates

Once the water parameters were calibrated, the 1977 landuse was replaced with 1993 landuse data (Table 7-2). There was a discrepancy in total area for each subcatchment between the 1977 and 1993 landuse data sets. The 1977 landuse indicated that there was a total of approximately 70,000 acres in the upper 3 subcatchments, and according to the 1993 landuse, area in the upper 3 subcatchments totaled approximately 100,000 acres. We compared the 2 landuse data sets by weighting each landuse as a percentage of the total area. Weighted percentages indicated that change in landuse from 1977 to 1993 resulted in a decrease in Agricultural land, an increase in Urban and Built-up land, and an increase in Recreational land (golf courses and parks) (Tables 7-1, and 7-2).

Pesticide application rates for the various landuse categories were obtained from the SARWCB in the TMDL Source Analysis for Diazinon and Chlorpyrifos and are presented in Appendix A, Table A-2 (SARWQCB, 2001). Chemical properties for the pesticides included decay rates for soil and water (Appendix A, Table A-3) and various parameters such as washoff and scour potency factors, initial concentration of the constituents in water and on land surfaces, and maximum storage capacity (Appendix A, Table A-4).

Table 7-2: 1993 Landuse

Landuse	Upper San Diego		Intermediate San Diego		Peter's Canyon	
	Area (acres)	% of total	Area (acres)	% of total	Area (acres)	% of total
Urban or Built-up/pervious	12,685	31%	19,028	49%	10,277	38%
Agriculture	5,793	14%	1,415	4%	6,158	23%
Open and Recreational	463	1%	1,422	4%	492	2%
Barren	22,000	54%	16,954	44%	9,888	37%
Total acres per subcatchment	**40,941**	100%	**38,820**	100%	**26,816**	100%

Source: SCAG (1993).

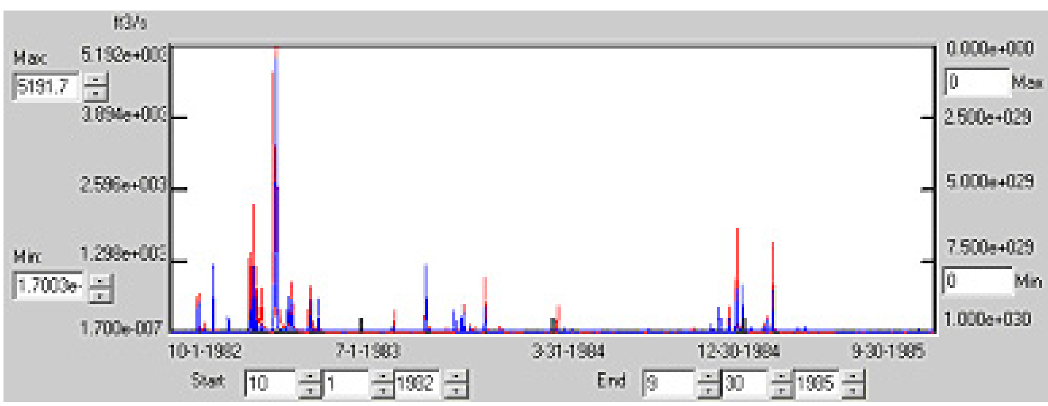


Figure 7-2. Stream Flow Hydrograph. Model fit was determined by comparing simulated stream flow (cfs) in red to actual recorded stream flow in blue. Stream flow from gaging station #11048550, located on San Diego Creek just above where San Diego Creek enters the upper Newport Bay. Max represents greatest flow rate (experienced during stormflow) and min represents lowest flow rate (found during dry baseflow).

7.1.5 Model Validation

Our approach was to develop and calibrate the model based on the 1977 landuse and meteorology, and then validate the model by running simulations with 1993 landuse and meteorology. Ideally, we had planned to test the model fit by comparing the 1993 simulated flow to actual recorded flow data. Unfortunately, the USGS has discontinued recording data at all gaging stations within the watershed. The most current recorded data was for 1985. Instead, we compared the simulated pesticide concentrations to recorded values presented by the SARWQCB (2001). We found that the maximum and minimum concentrations were within the appropriate range. 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 7-3. 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.

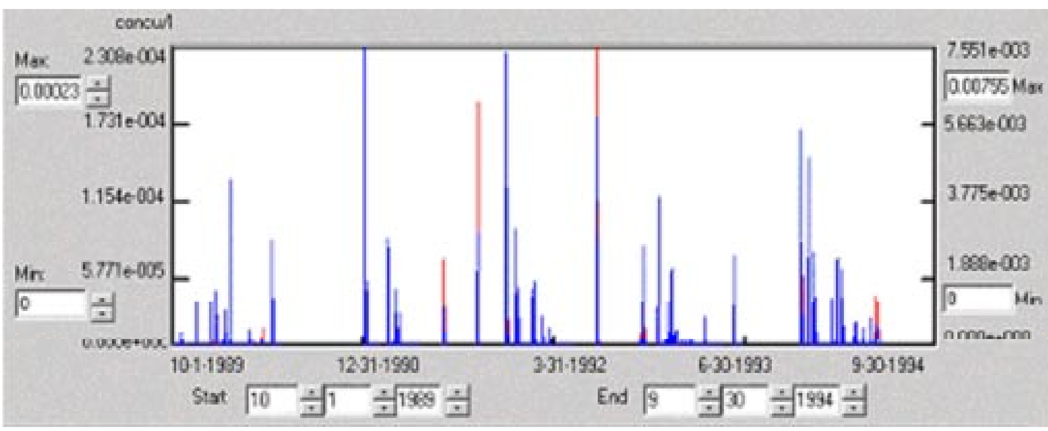


Figure 7-3. Constituent Concentration Hydrograph For The Baseline Scenario. Chlorpyrifos concentration (mg/L) is in red on the left axis, and Diazinon concentration (mg/L) is in blue on the right axis. Simulation period is from October 1, 1998 to September 30, 1994.

7.1.6 Management Strategy Evaluation

Various scenarios were simulated for a 5-year period beginning October 1, 1989 and ending September 30, 1994. The 5-year period was chosen because of the availability of meteorological data. Actual meteorological data was used for the duration of the simulation, and therefore, precipitation and flow regimes varied from year to year. All simulations are based on constant application rates (Appendix A, Table A-2); therefore, variability in constituent concentration can be attributed to variation in meteorology. Simulated flow at the base of the 3 subcatchments (USGS gaging

Station #11048550) for the period is presented in Figure 7-4. Following is a list of scenarios that were evaluated:

- Baseline - No Change
- Pesticide Phase-out
    - EPA projected phase-out
        1. Chlorpyrifos - 50% reduction of application rate in Urban and Built-up landuse, and 33% reduction in Agricultural landuse
        2. Diazinon – 75% reduction of application rate in Urban and Built-up landuse and a 33% reduction in Agricultural landuse
    - SQRWQCB projected phase-out
        1. 90% reduction of Urban and Built-up landuse applications for both pesticides
- Implementation of BMPs in conjunction with the EPA and SARWQCB projected phase-outs
    - Infiltration Trenches – 65% reduction in all landuses
    - Sand Filters – 72% reduction in all landuses
    - Vegetated swales – 47% reduction in all landuses
    - Infiltration Ponds – 65% reduction in all landuses
    - Education – 25% reduction in Urban and Built-up, and Agricultural landuses

For scenarios that incorporated management practices and phase-outs, the reductions were considered to be implemented at the beginning of the simulation (October 30, 1989), for comparison purposes. The following data was recorded and analyzed for four sequential years, beginning one year after implementation of the reduction:

- Maximum concentration of each pesticide (mg/L)
- Number of days during each rain period that the simulated concentration for each pesticide exceeded the Criterion Chronic Concentration (CCC) and Criterion Acute Concentration (CMC) levels recommended by the CDFG for freshwater (Tables 2-1, and 2-2).

Our analysis was based on storm events that occurred during 6-month rain periods beginning November 1st and ending April 30th of each year.

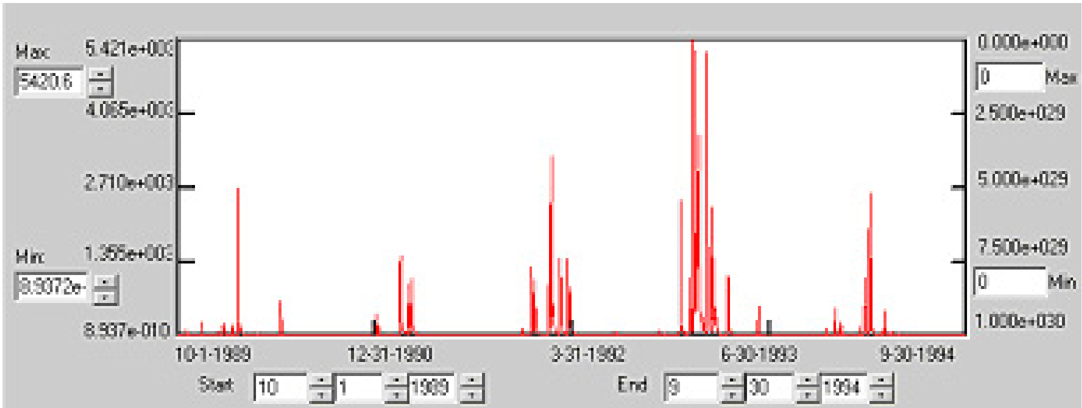


Figure 7-4. Simulated stream flow at the base of San Diego Creek (just before the San Diego Creek enters the upper Newport Bay). Simulation period began October 1, 1989 and ended September 30, 1994.

7.2 Results/Interpretation

7.2.1 Baseline and Phase-Out Scenarios

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 7-3). 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. In fact, the phase-out is likely to reduce Chlorpyrifos concentrations to acceptable levels.

We then compared the baseline scenario to the 2 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 landuses. In the EPA phase-out scenario, the predicted Chlorpyrifos concentrations were approximately 50% lower than concentrations simulated in the baseline scenario (Figure 7-5 A). Diazinon concentrations were predicted to be approximately 75% less in the EPA phase-out scenario than in the baseline scenario (Figure 7-4 B). 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.

A noteworthy difference between the 2 constituents is that 1 year after initiation of the phase-out, maximum Chlorpyrifos concentrations are approximately the same for all scenarios (baseline = 2.82×10^{-5} mg/L; EPA phase-out = 2.78×10^{-5} mg/L; and SARWQCB phase-out = 2.76×10^{-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 = 7.55×10^{-3} mg/L; EPA phase-out = 1.83×10^{-3} mg/L; SARWQCB phase-out = 1.11×10^{-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 in Section 5, 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 7-5 B). 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.

7.2.2 Phase-Out Scenarios in combination with Implementation of BMPs

Our final analysis looked at predicted maximum concentrations resulting from a combination of the projected phase-outs and various BMPs. Modeled BMPs are discussed in detail in Section 9. Most of the scenarios resulted in acceptable concentrations of Chlorpyrifos. Only Vegetated Swales and a 25% Education reduction resulted in one-day exceedence per rain period for Chlorpyrifos following the second simulated year. All of the simulations resulted in unacceptable concentration levels for Diazinon, even considering the phase out and BMPs. Figure 7-6 presents the number of days concentrations exceeded the Diazinon CCC, and Figure 7-7 presents the number of days that concentrations exceeded the Diazinon CMC. Our model predicts that after the phase-out, Diazinon concentrations will regularly exceed both the CCC and CMC levels. The amount of days per year that concentrations of Diazinon will be in excess of these criteria levels will vary with precipitation events. Based on meteorological data recorded between 1989 and 1994, and the phase-out reductions predicted by the EPA and the SARWQCB, Diazinon concentrations would have exceeded the CCC limits between 11 to 24 days per rain period, and the CMC limits between 10 and 23 days per rain period.

Table 7-3. Baseline Scenario Indicating Number of Days during Simulated Rain Period Concentrations Exceed the CCC and CMC Levels.

	Chlorpyrifos Days in exceedence		**Diazinon 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

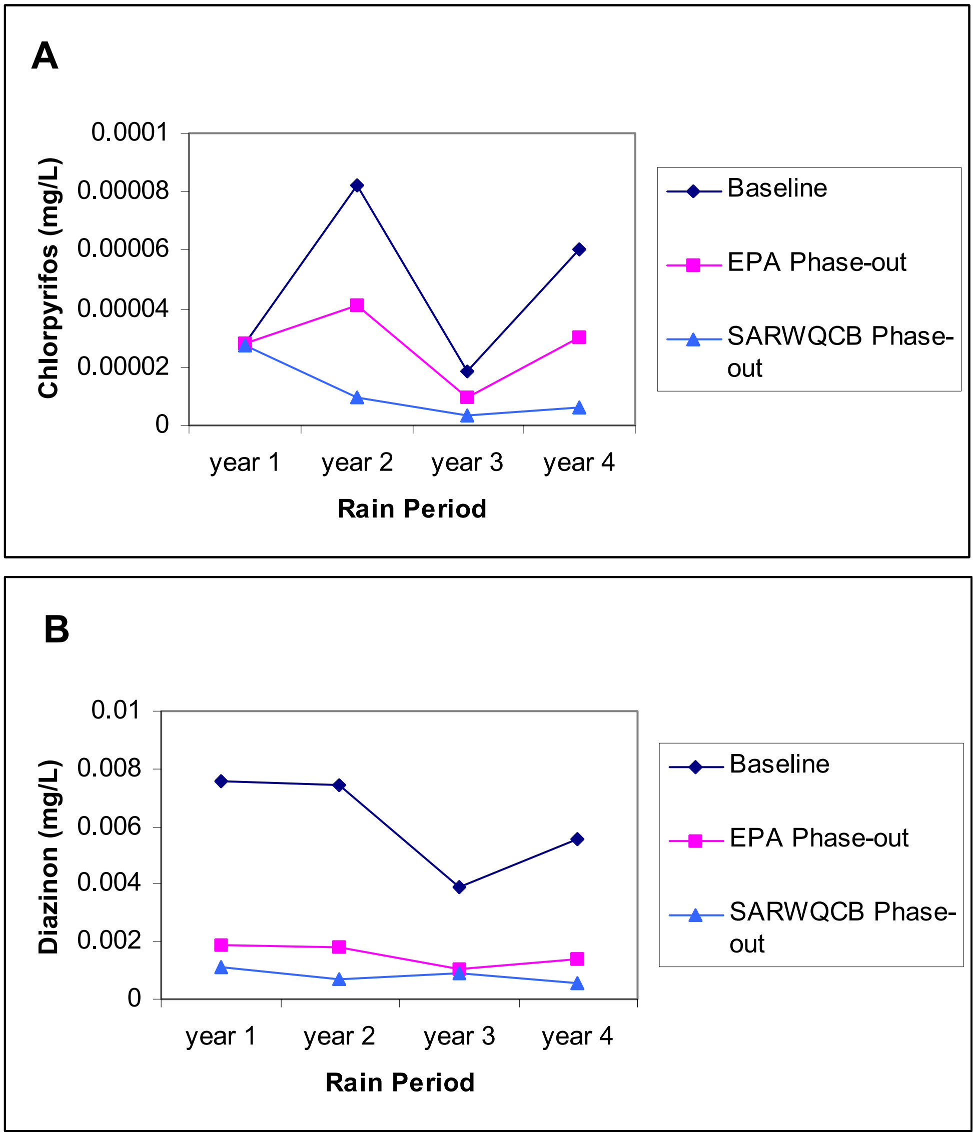


Figure 7-5 A and 7-5 B. Comparison of simulated maximum (stormflow) pesticide concentrations between the baseline scenario and the phase-out scenarios predicted by the EPA and the SARWQCB. Figure 7-5 A presents the maximum simulated Chlorpyrifos concentration during each rain period and Figure 7-5 B presents the maximum simulated Diazinon concentration during each rain year.

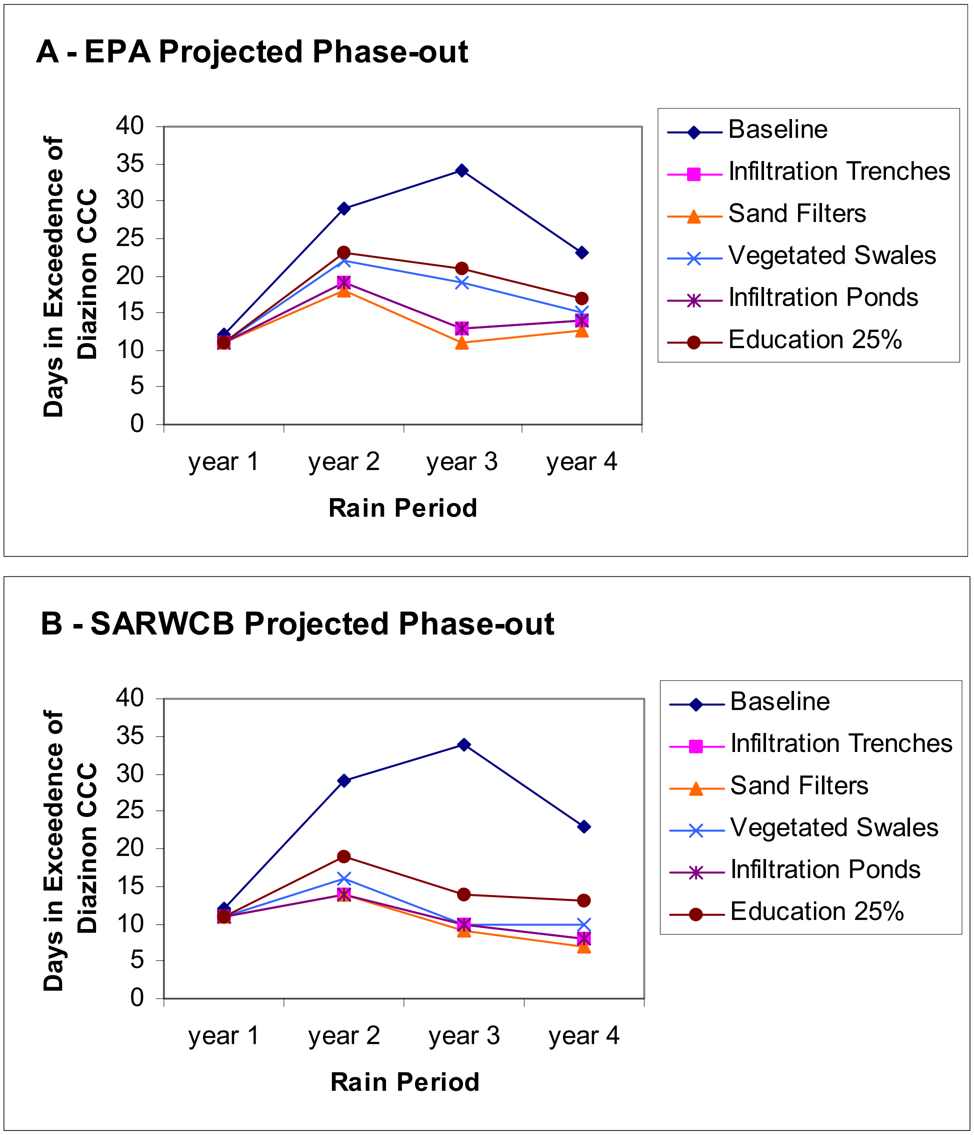
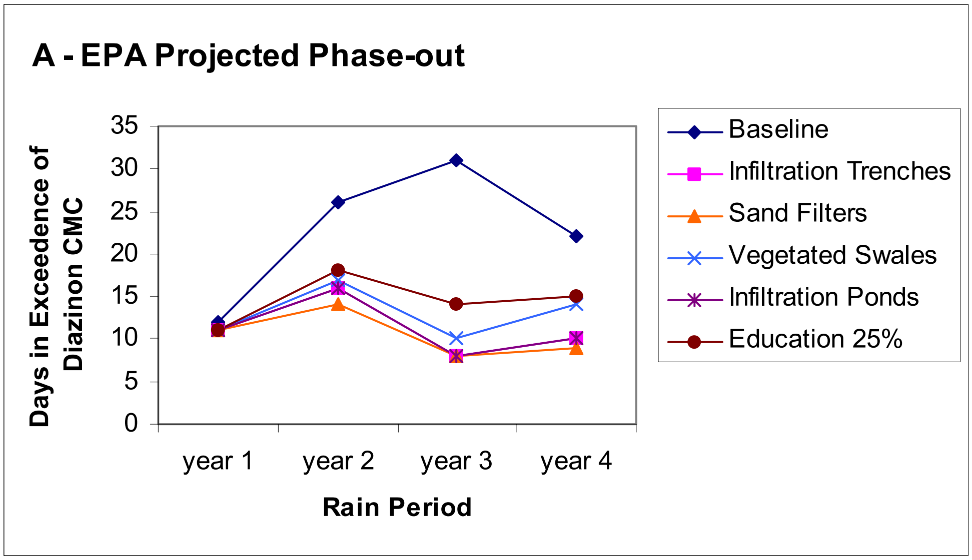


Figure 7-6 A and 7-6 B. Comparison of BMP scenarios in combination with phase-outs projected by the EPA and the SARWQCB, indicating the number of days that simulated Diazinon concentrations exceeded the CCC levels of $5.0*10^{-5}$ mg/L per rain period. Figure 7-6 A presents BMPs in combination with EPA projected phase-out, and Figure 7-6 B presents BMPs in combination with SQRWQCB projected phase-outs.



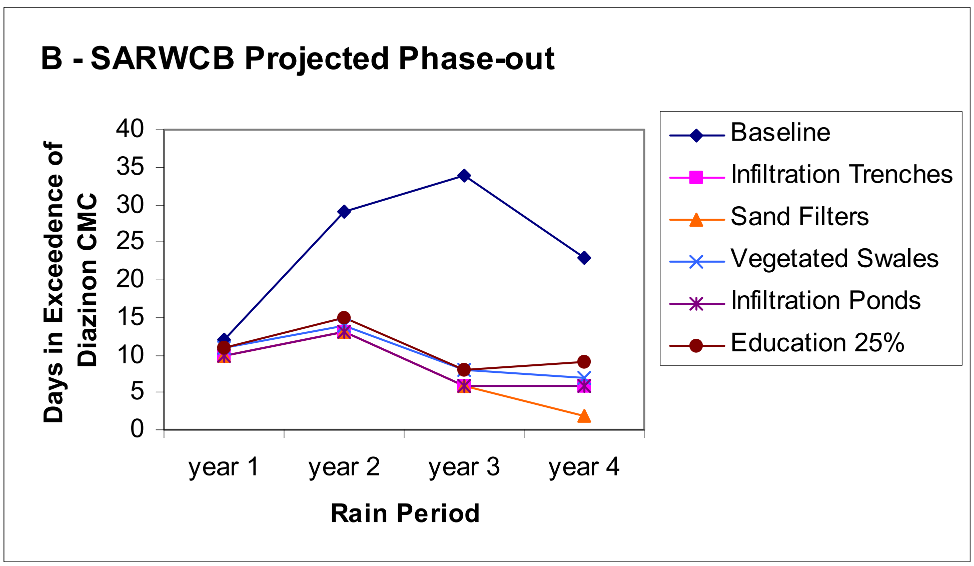


Figure 7-7 A and 7-7 B. Comparison of BMP scenarios in combination with the phase-outs projected by the EPA and the SARWQCB, indicating the number of days that simulated Diazinon concentrations exceeded the CMC levels of $8.0*10^{-5}$ mg/L per period. Figure 7-7 A presents BMPs in combination with EPA projected phase-out, and Figure 7-7 B presents BMPs in combination with SQRWQCB projected phase-outs.

8.0 WARMF

8.1 Methodology

The Watershed Analysis Risk Management Framework (WARMF) is a watershed decision program developed by Systech (EPRI, 2000). 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).

8.1.1 Watershed Framework

The purpose of this study was to implement a watershed model for Newport Bay Watershed and calibrate it using WARMF. Since a WARMF model for Newport Bay Watershed had not been implemented before, we modified an existing model, by updating observed information in the Data module and adjusting system, catchment, and river coefficients in the Engineering module. The three lowest catchments of the base watershed were used to represent Newport Bay Watershed; two representing the tributaries Peters Canyon Wash and upper San Diego Creek and a third below the confluence of the two (referred to as lower San Diego Creek) (Figure 8-1).

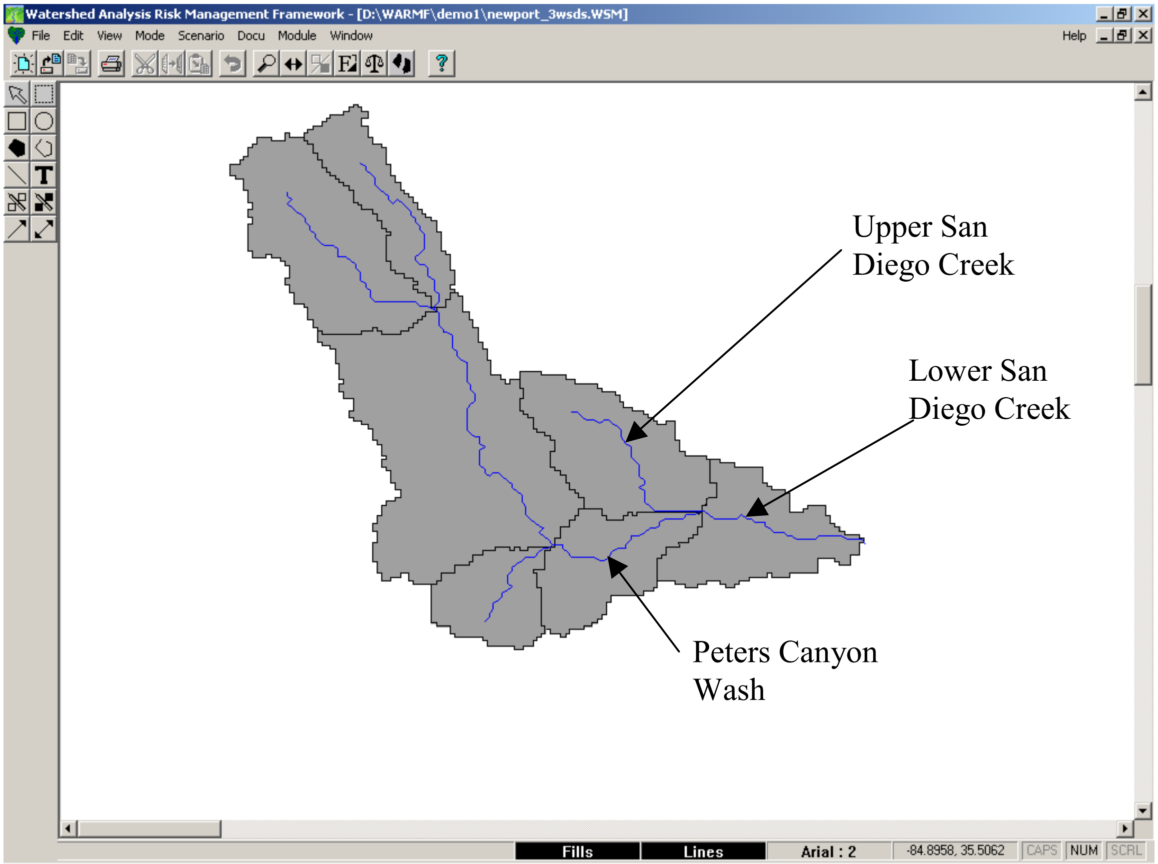


Figure 8-1. Illustration of the Locations of the Three Newport Bay Watershed Catchments Within the Base WARMF Watershed.

The four additional catchments, visible in the graphic in Figure 8-1, were turned “off” so as not to interfere in the processes of the three catchments representing Newport Bay watershed. The catchments were disabled by loading a meteorological file that indicated no precipitation feel to the land surface.

8.1.2 Spatial Data

The Data module of the base watershed was updated with observed data applicable to Newport Bay. Meteorological data for Newport Bay was collected from the National Climatic Data Center (NCDC) (http://lwf.ncdc.noaa.gov/oa/ncdc.html). Minimum temperature, maximum temperature, and precipitation were used from the Newport Beach Station (ID: 046175) for the entire watershed as this was the only weather recording station within the watershed. The Los Angeles International Airport Station (ID: 045114) was the only station within reasonable distance of the watershed, which recorded daily dewpoint and wind speed. However, these conditions were only available for 1984, so these values were used for all the years of data collected (Table 8-1). Cloud cover and pressure were not measured in any area useful to Newport Bay, so the values already present in the base watershed were used.

Table 8-1. NCDC Identification and Location of Meteorology Stations Used for the WARMF Newport Bay Watershed.

Location	Latitude/ Longitude	COOP ID	WBAN	Elevation (m)	Date Range Retrieved
Newport Beach, CA	33 36'N/ 117 53'W	046175	03107	3.0	01/01/1971 – 12/31/1995
Los Angeles International Airport, CA	33 56'N/ 118 24'W	045114	23174	30.5	01/01/1971 – 12/31/1995

Surface water data for the three catchments was acquired from the USGS NWIS site (http://water.usgs.gov/nwis). Several flow sites are available for Newport Bay Watershed. The three we selected for the WARMF model are close to the outflows of each delineated catchment and have the longest time periods of measurement (Table 8-2).

Table 8-2. USGS Identification and Description of the Flow Observation Stations Used for the Catchment Outflows.

Site Number	Location	Segment Dates
11048500	San Diego Creek at Culver Dr. near Irvine, CA	1972 – 1985
11048540	Peters Canyon Wash at Barranca Dr. near Irvine, CA	1982 – 1984
11048555	San Diego Creek at Campus Dr. near Irvine, CA	1977 – 1984

Numerous surface water quality observations (nitrogen, phosphorus, sediment, etc.) can be input into the WARMF watershed implementation. These variables, if measured within each catchment, 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 (Table 8-3). Total suspended sediment measurements were originally retrieved from the STORET legacy data center but more continuous data was measured at the USGS NWIS stations described in Table 8-2. 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.

Table 8-3. STRORET Legacy Data Center Water Quality Sites From Which a Number of Water Quality Observations Were Retrieved.

Site ID	Location	Latitude/Longitude
Barranca SED	Peters Canyon Wash at Barranca Dr. near Irvine, CA	33 41' 30"N/117 49' 23"W
SJQF14	San Diego Creek at Culver Dr. near Irvine, CA	35 40' 34"N/117 48' 55"W
SDMF05	San Diego Creek at Campus Dr. near Irvine, CA	33 39' 20"N/117 50' 41"W

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.

8.1.3 Model Calibration

Model fit 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, 1985. 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 8-4) 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 8-2 through 8-4). 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 (Figure 8-5 through 8-7). In general, all three segments matched the accumulated volume of the observed data. 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 base flow and does not peak as high in storm events.

Table 8-4. 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 (cms)	0.364	0.414	0.794	0.747	1.506	1.604
Maximum Sediment Concentration (mg/L)	5864	5880	12410	12500	9719	9960

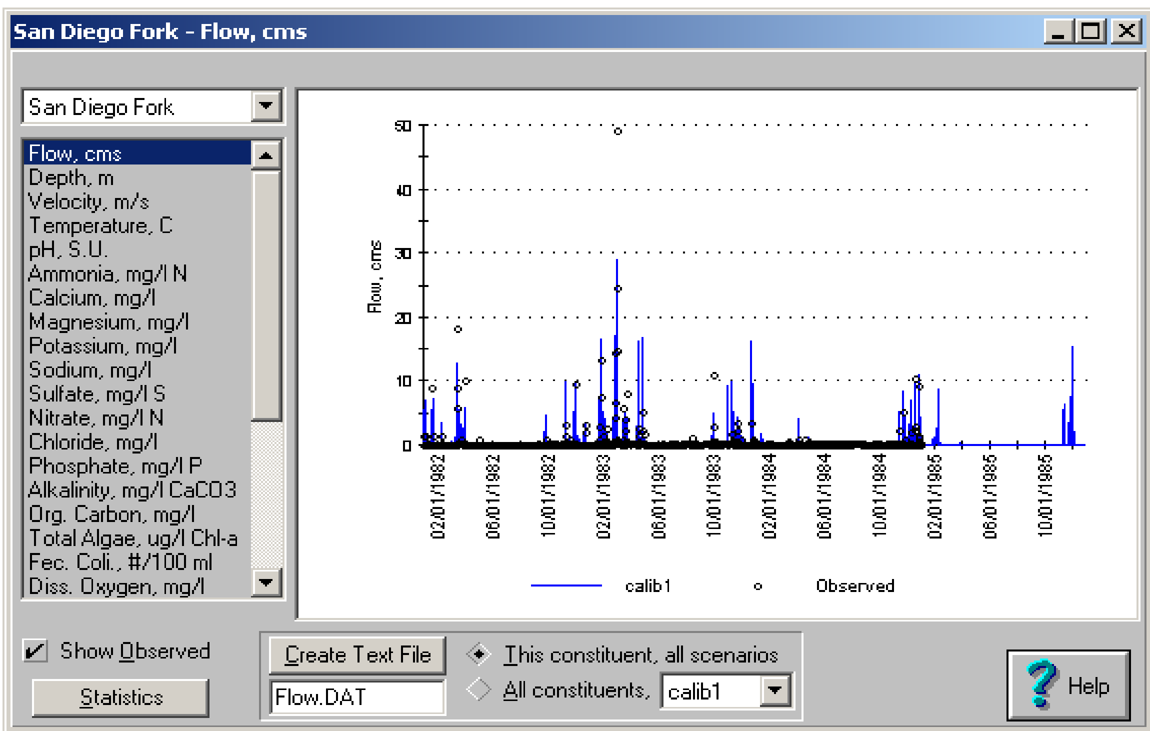


Figure 8-2. Hydrograph for Upper San Diego Creek From January 1, 1982 Through December 30, 1985. Simulated Flow (cms) Under Predicts Observed Stormflow.

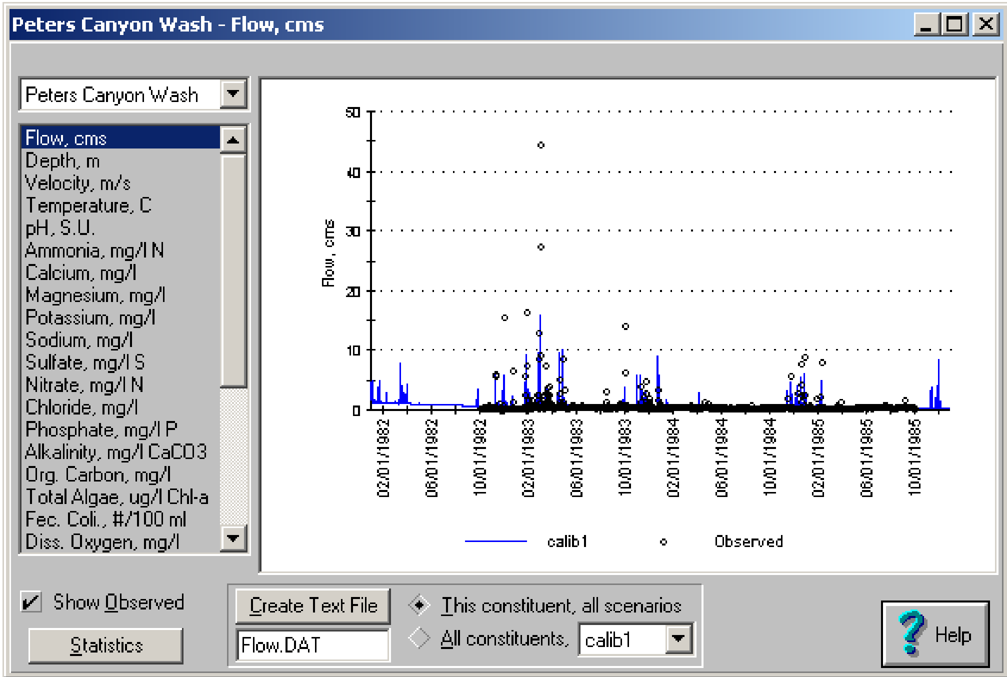


Figure 8-3. Hydrograph for Peter’s Canyon Wash From January 1, 1982 Through December 30, 1985.

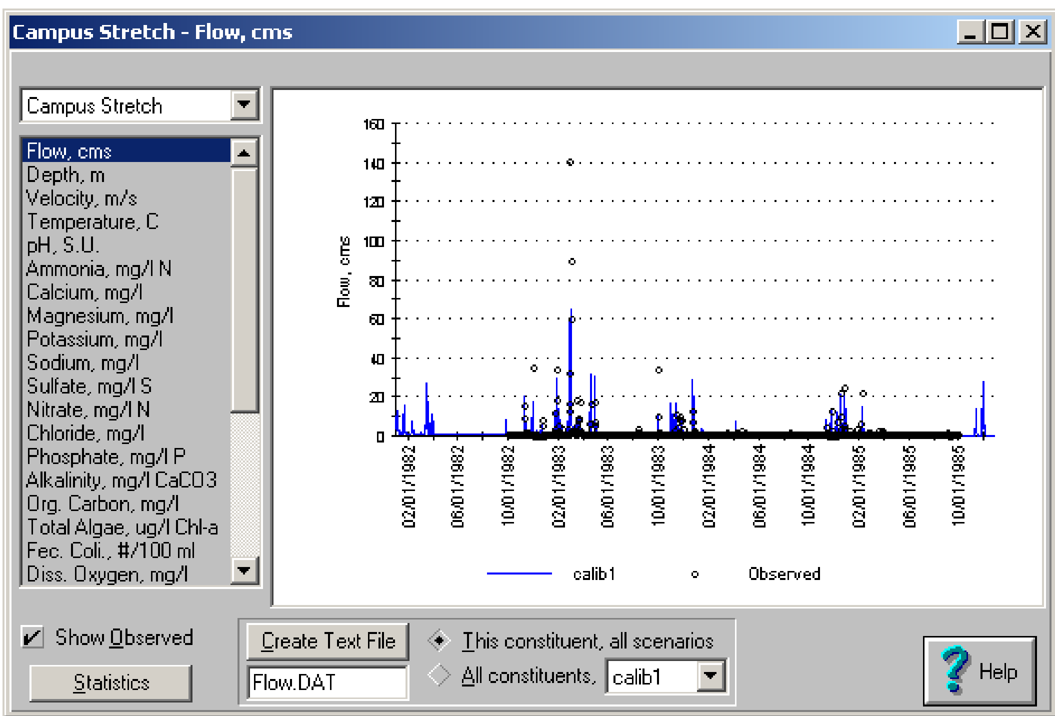


Figure 8-4. Hydrograph for Lower San Diego Creek From January 1, 1982 Through December 30, 1985.

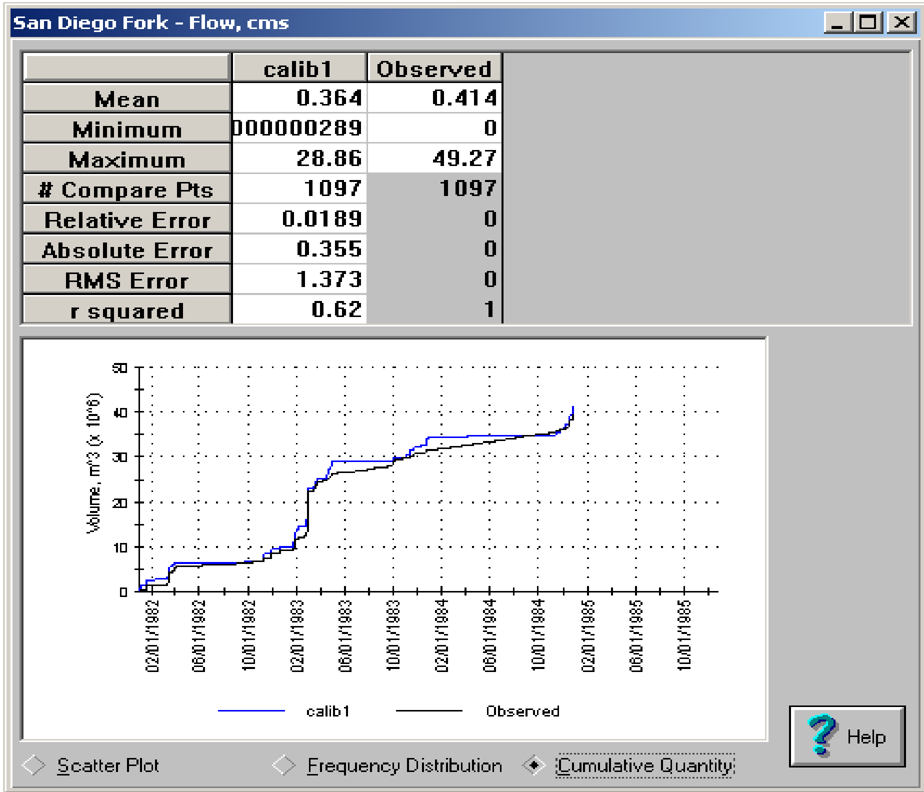
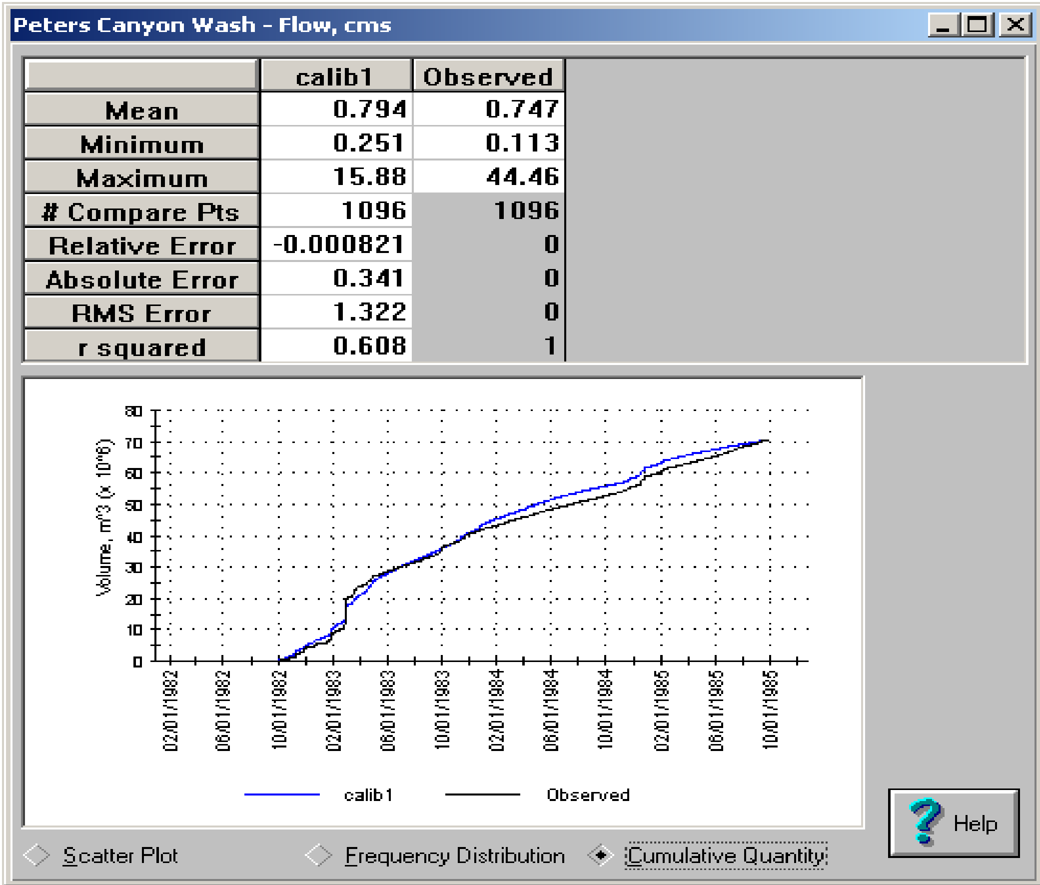
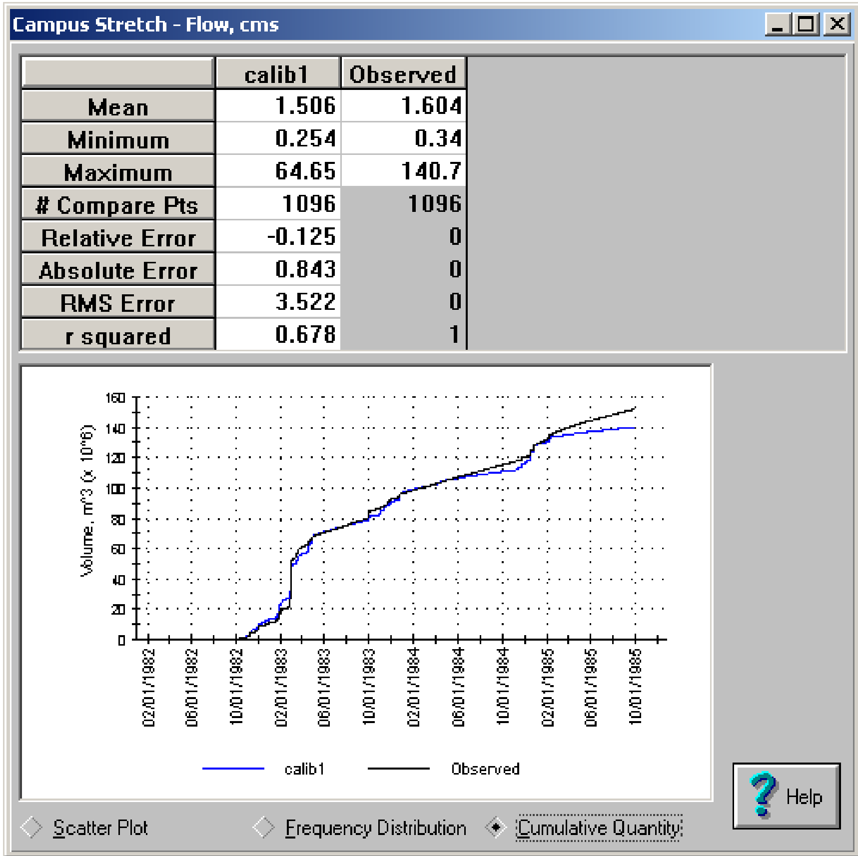


Figure 8-5. Simulated Flow (Blue) Cumulative Volume Versus Observed Flow (Black) Cumulative Volume For Upper San Diego Creek From January 1, 1982 Through December 30, 1985.



	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

Figure 8-6. Simulated Flow (Blue) Cumulative Volume Versus Observed Flow (Black) Cumulative Volume For Peter's Canyon Wash From January 1, 1982 Through December 30, 1985.



	calib1	Observed
Mean	1.506	1.604
Minimum	0.254	0.34
Maximum	64.65	140.7
# Compare Pts	1096	1096
Relative Error	-0.125	0
Absolute Error	0.843	0
RMS Error	3.522	0
r squared	0.678	1

Figure 8-7. 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 vital 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 8-4) 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 8-8 through 8-10 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.

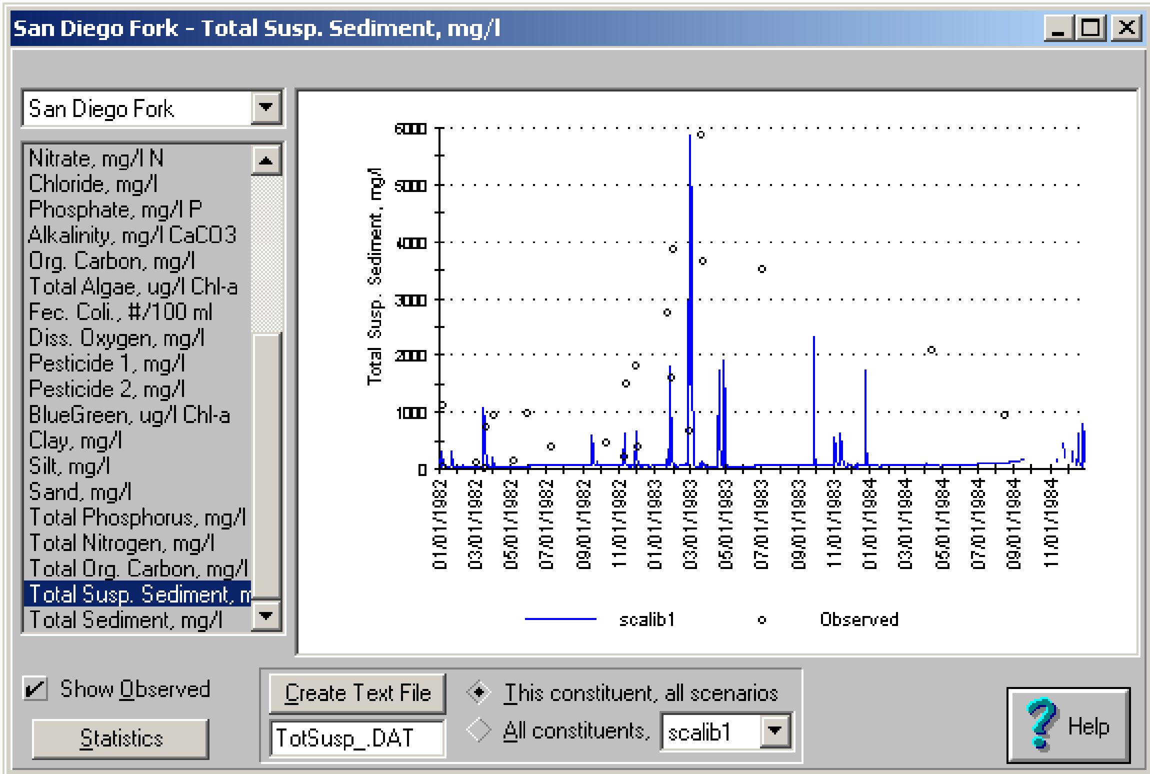


Figure 8-8. Simulated Total Suspended Sediment (Mg/L) for Upper San Diego Creek From January 1, 1982 Through January 1, 1985.

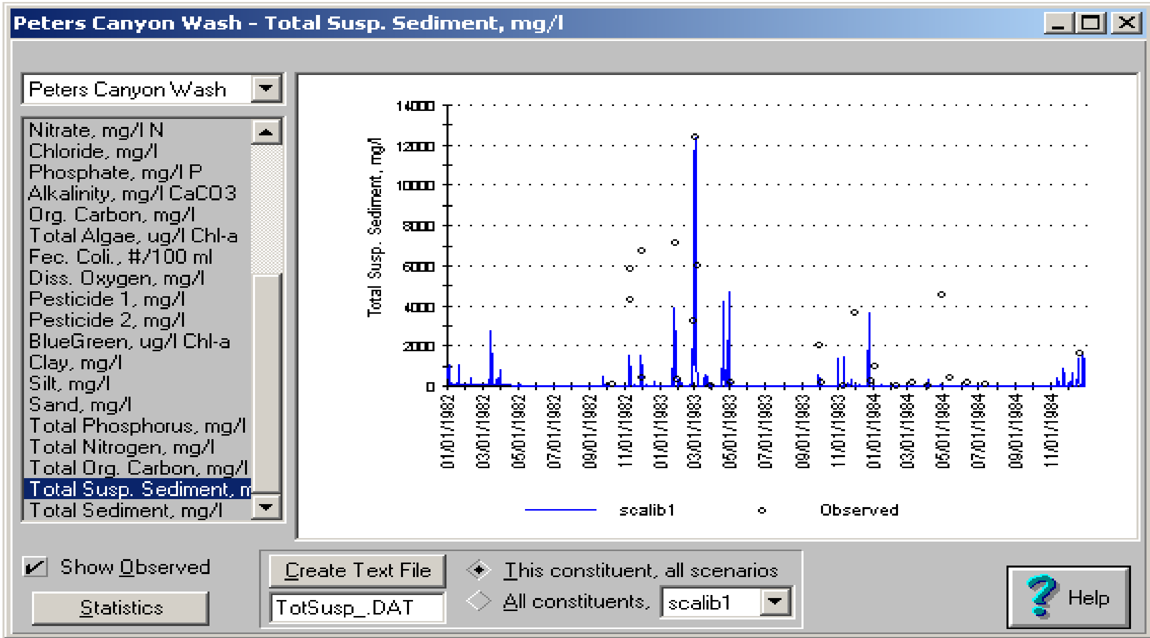


Figure 8-9. Simulated total suspended sediment (mg/L) for Peter's Canyon Wash from January 1, 1982 through January 1, 1985. Model captures the height of some observed sediment peaks but not all. Peter's Canyon Wash produces the most sediment within the watershed.

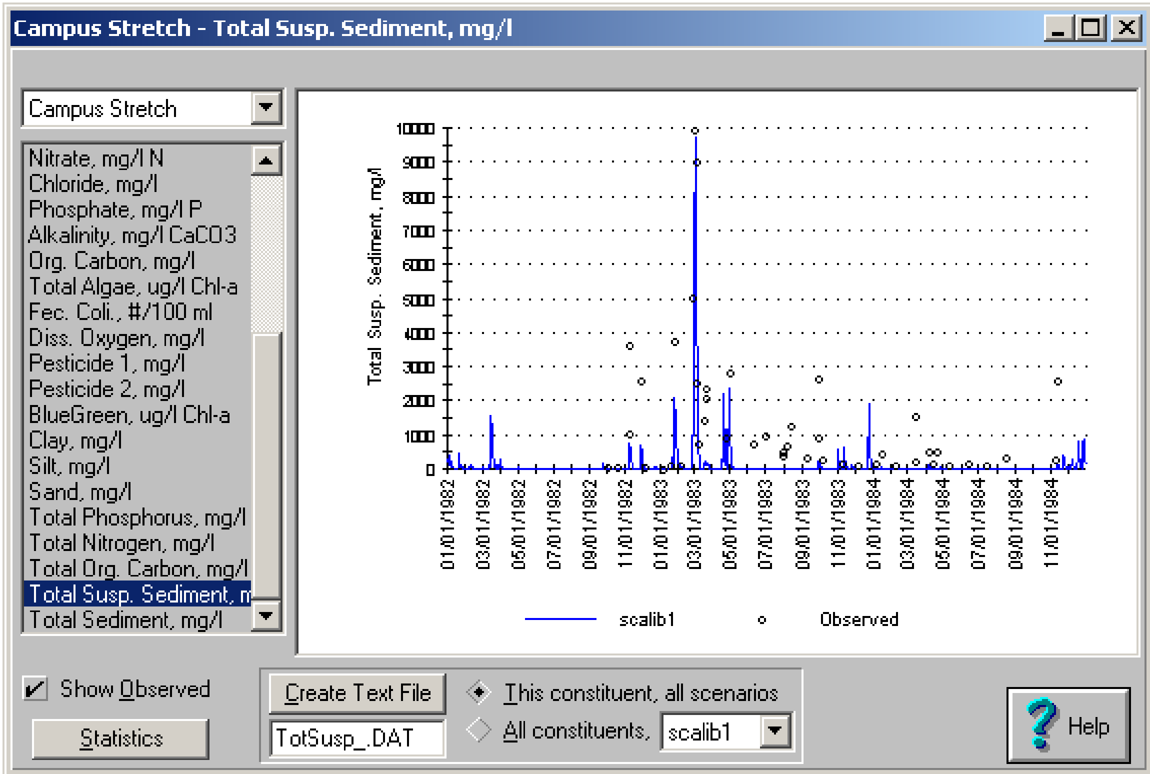


Figure 8-10. Simulated Total Suspended Sediment (mg/L) for Lower San Diego Creek from January 1, 1982 Through January 1, 1985. Model Captures the Height of Some Observed Sediment Peaks but not all.

8.1.4 Pesticide Modeling

The satisfactory calibration of the flow and total suspended sediment prepared the model for accurately simulating the fate and transport of Diazinon and Chlorpyrifos. In order to simulate the concentrations of the two pesticides in the stream segments the land application rates of each pesticide had to be put into the system. Chlorpyrifos and Diazinon are mainly associated with agricultural and urban landuses, therefore using 1999 application rates for Orange County obtained from the SARWQCB (2001), we calculated the application amount being applied to each landuse per month per catchment as seen in Table 8-5.

Table 8-5. Land Application Pesticide Amounts (Kg/ha) per Catchment (SARWQCB, 2001).

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

At a system level, WARMF required the decay rates of each pesticide in water and soil. To complement the BASINS model the decay rates for the OP-pesticides in Table 8-6 were input into the WARMF system coefficients. The decay rates were obtained from the Santa Ana Regional Water Control Board in the TMDL Source Analysis for Diazinon and Chlorpyrifos (SARWQCB, 2001).

Table 8-6. Pesticide Decay Rates (1/day) Used in WARMF for Diazinon and Chlorpyrifos (SARWQCB, 2001).

Pesticide	Land	Water
Diazinon	0.025	0.0039
Chlorpyrifos	0.0058	0.0092

Diazinon and Chlorpyrifos were then 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 8-11 illustrates the simulated concentration (mg/L) of Diazinon in lower San Diego Creek and Figure 8-12 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.

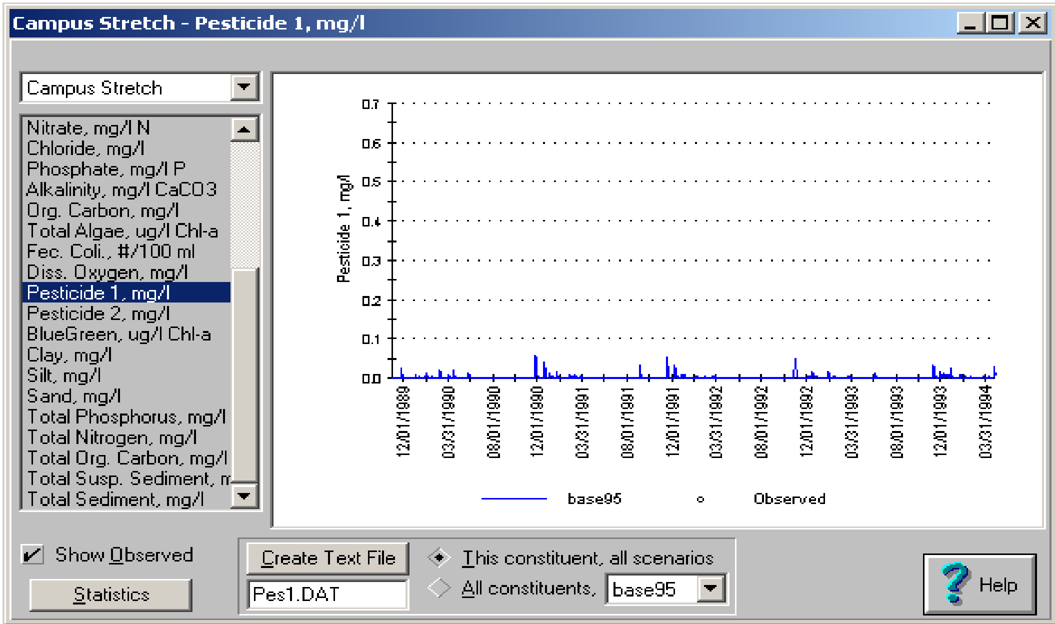


Figure 8-11. Simulated Diazinon Concentrations (mg/L) for Lower San Diego Creek from November 1, 1989 Through April 30, 1995.

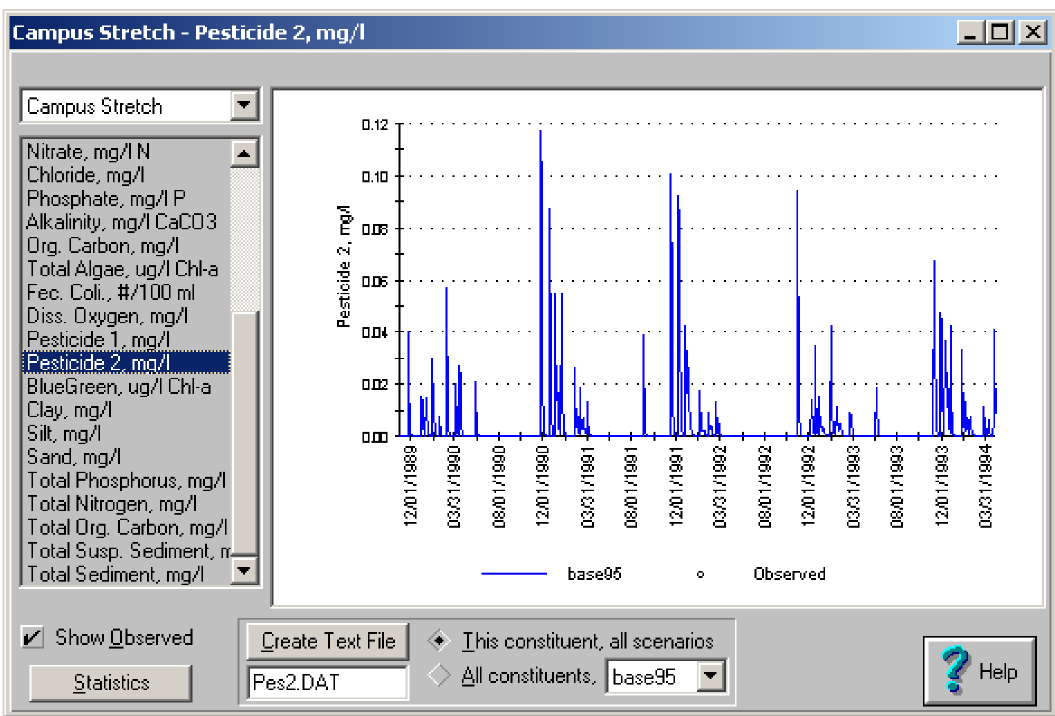


Figure 8-12. Simulated Chlorpyrifos Concentrations (mg/L) for Lower San Diego Creek from November 1, 1989 Through April 30, 1995.

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 median and concentration ranges for each pesticide are in Table 5-4 and Table 5-5. The observed data is discussed in detail in Section 5.3). Since the watershed was not divided into catchments comparable to those set up in WARMF only the values found at the outflow of the watershed as a whole were used to judge the accuracy of the lower San Diego Creek. The simulated median concentration of Diazinon are 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 landuses. (The median and maximum concentrations of Diazinon and Chlorpyrifos for each catchment can be found in Appendix B, Tables B-1 through B-3.)

8.2 Management Strategy Evaluation

Once hydrologic and sediment calibration was completed and pesticide properties and application rates had been entered into the model, various phase-out and BMP scenarios were run from November 1, 1989 to April 30, 1995 in order to evaluate their effect on pesticide concentrations. The following scenarios were evaluated by adjusting the 1999 pesticide application amounts:

- Baseline - No Change
- Phase-out of pesticide use
  - EPA projected phase-out
    - Chlorpyrifos - 50% reduction in urban landuse, and 33% reduction in agricultural landuse
    - Diazinon – 75% reduction in urban landuse and a 33% reduction in agricultural landuse
  - 75% pesticide reduction in urban landuse
  - 50% pesticide reduction in urban landuse
  - 25% pesticide reduction in urban landuse
- Implementation of BMPs in conjunction with the EPA projected phase-out
  - Street sweeping every 150 days at 50% efficiency in conjunction with EPA phase-out
  - Street sweeping every 50 days at 50% efficiency in conjunction with EPA phase-out

Each scenario's results were analyzed by the following criteria:

- Maximum concentration of each pesticide (mg/L)
- Number of days during each rain year that the simulated concentration for each pesticide exceeded the Criterion Chronic Concentration (CCC) and Criterion Acute Concentration (CMC) levels (Table 7-2).

8.3 Results

While the model was run continuously from November 1, 1989 to April 30, 1995, the pesticide results were analyzed for four annual sessions (1990 - 1994) from November 1 to April 30 for each catchment as this captures the rainy season in Newport Bay and therefore the period of greatest pesticide mobilization (Figure 8-13).

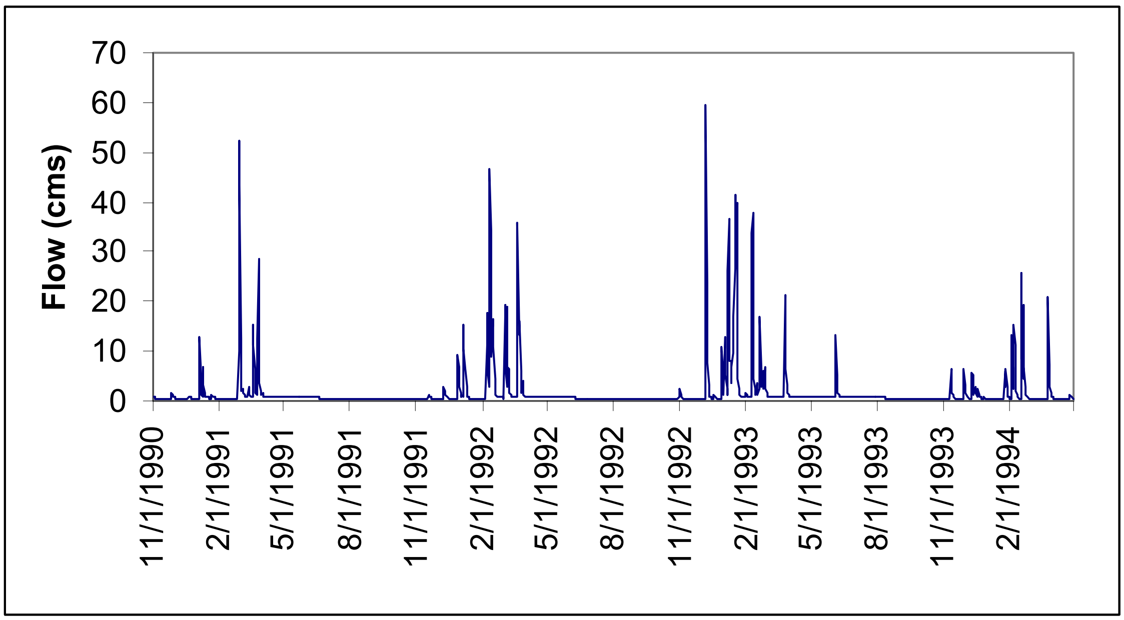


Figure 8-13. Hydrograph for Lower San Diego Creek Showing That the Majority of Rainfall Occurs Between Early November and Late April.

The phase-out scenarios decreased maximum pesticide concentration (mg/L) at regular intervals, as expected, with a slight decrease over time (Figures 8-14 and 8-15). Pesticide concentrations were low in 1993 because that session had the highest amount of rain and therefore diluted the pesticide concentrations. Conversely, 1994 had low intensity storm events reducing the amount of pesticide lifted from the surface. Days in exceedance for both pesticides also decreased with each pesticide load reduction. However, despite a reduction in maximum concentration over the years, 1993 and 1994 Diazinon exceedance days increased in all scenarios (Figures 8-16 and 8-17). Chlorpyrifos exceedance days were not significantly different from year to year (Figures 8-18 and 8-19).

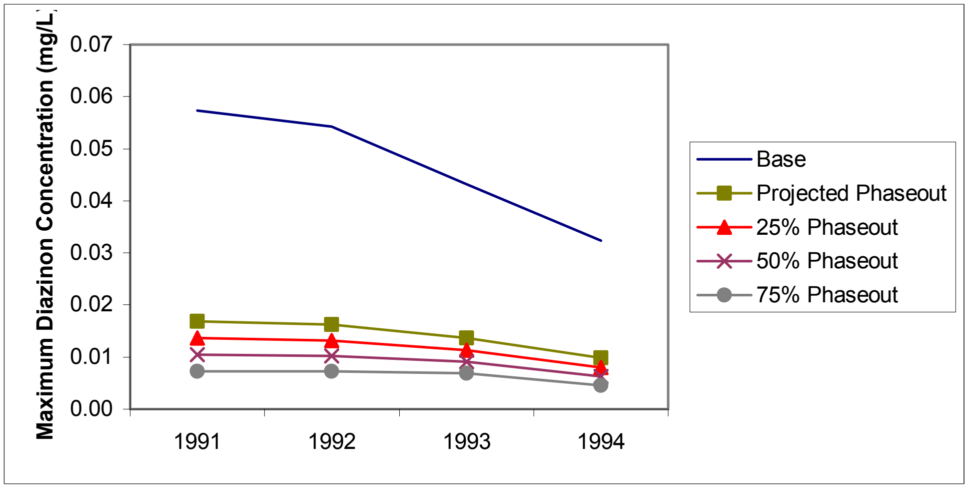


Figure 8-14. Lower San Diego Creek Maximum Diazinon Concentrations (mg/L) for the Phase-out Scenarios.

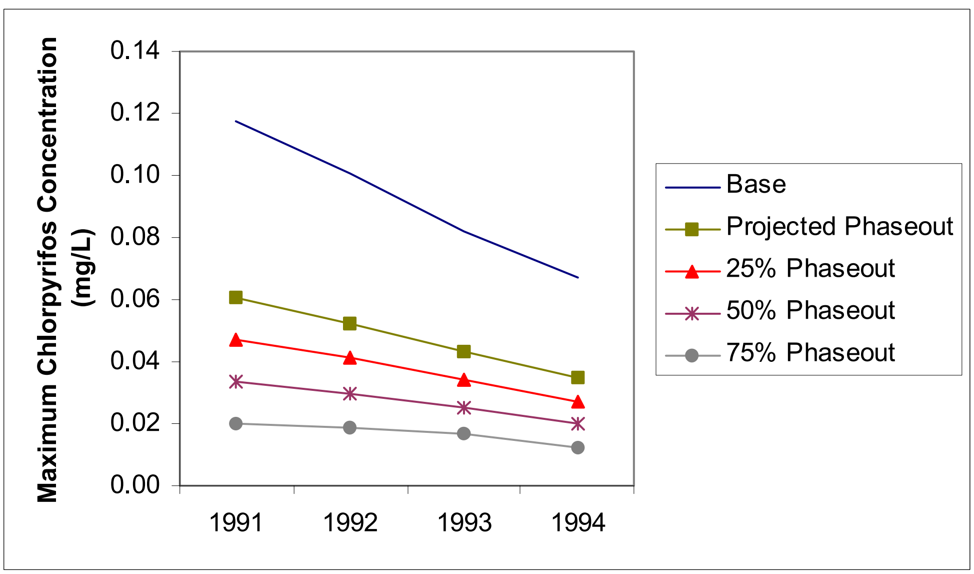


Figure 8-15. Lower San Diego Creek Maximum Chlorpyrifos Concentrations (mg/L) for the Phase-out Scenarios.

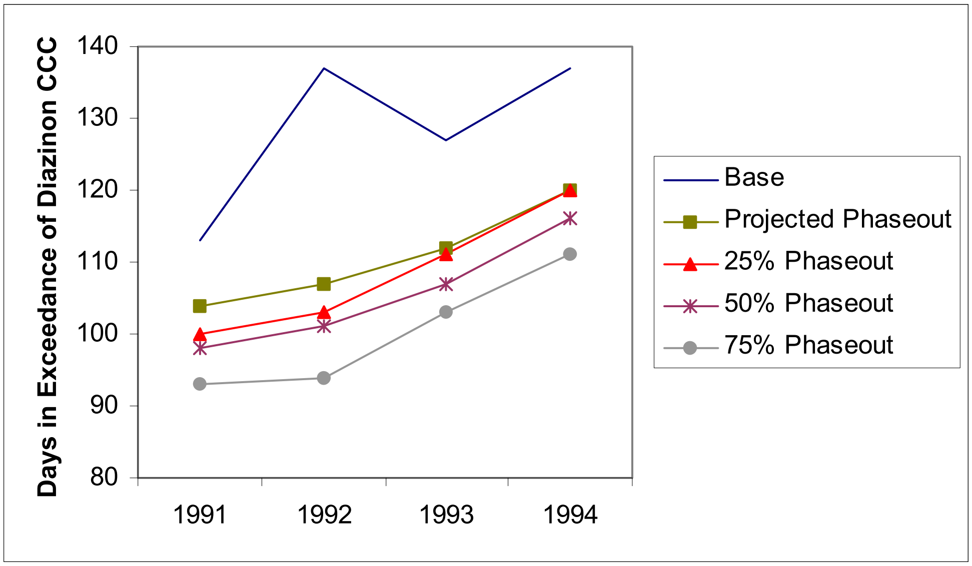


Figure 8-16. Number of days Diazinon was in Exceedance of the CCC Criteria Level for Each Scenario in Lower San Diego Creek.

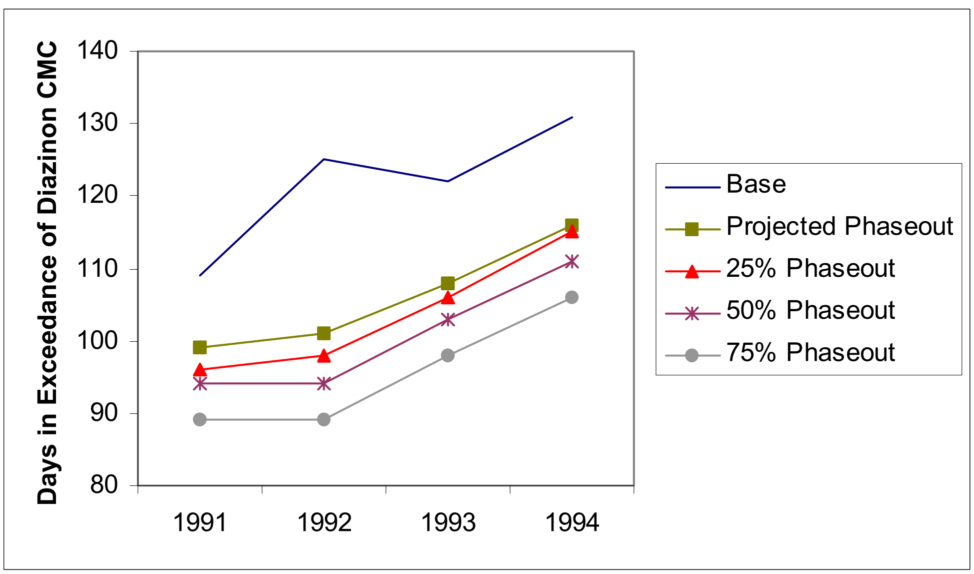


Figure 8-17. Number of Days Diazinon was in Exceedence of the CMC Criteria Level for Each Scenario in Lower San Diego Creek.

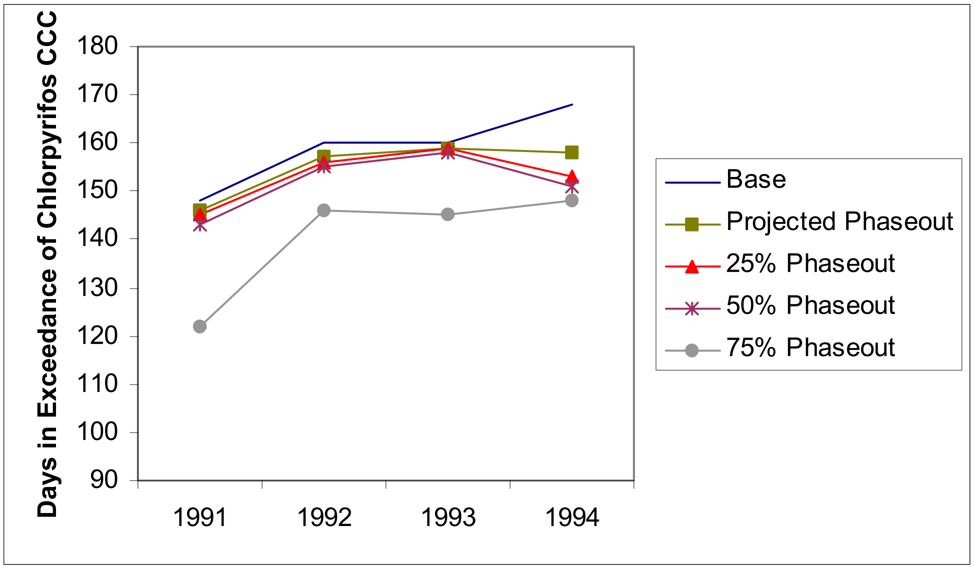


Figure 8-18. Number of Days Chlorpyrifos was in Exceedence of the CCC Criteria Level for Each Scenario in Lower San Diego Creek.

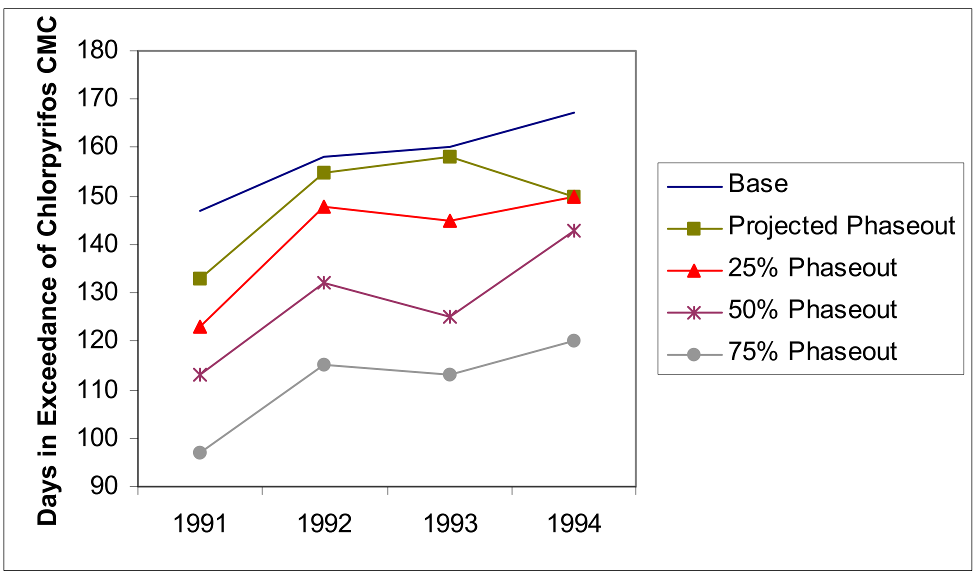


Figure 8-19. Number of Days Chlorpyrifos was in Exceedence of the CMC Criteria Level for Each Scenario in Lower San Diego Creek.

The street sweeping scenarios were implemented in conjunction with the projected EPA phase-out application amounts. WARMF implements a BMP module through which the user can designate the frequency at which streets within the watershed are swept and the efficiency level of the street sweeping operation. Street sweeping every 150 days had little effect on the pesticide concentrations. Street sweeping every 50 days had an effect very similar to that seen when an additional 25%

pesticide load reduction occurs on urban landuse. (Figures 8-20 and 8-21) Days in exceedence for both scenarios were the same for each pesticide and similar to the numbers seen when the 25% phase-out is applied.

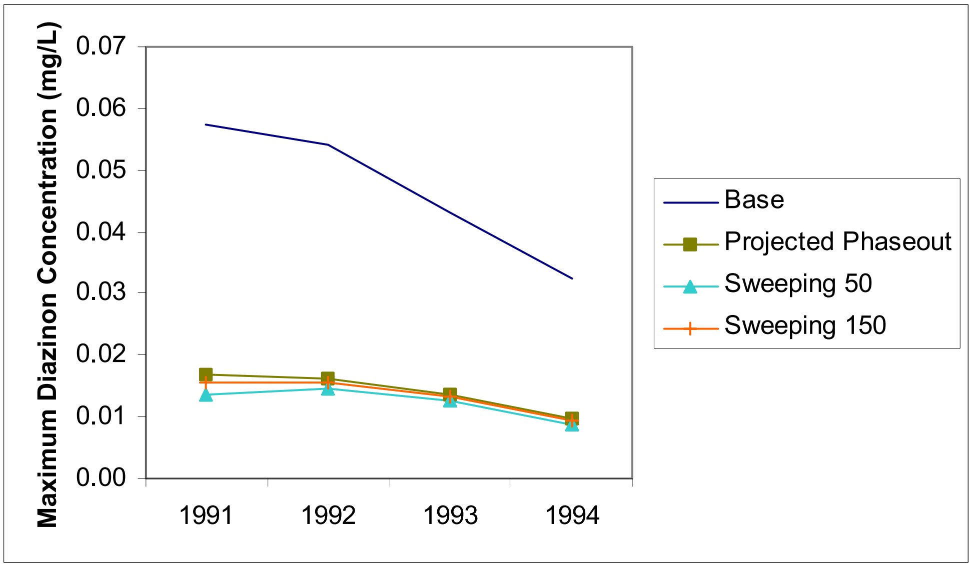


Figure 8-20. Maximum Diazinon Concentrations (mg/L) for the Street Sweeping BMP Scenarios in Lower San Diego Creek.

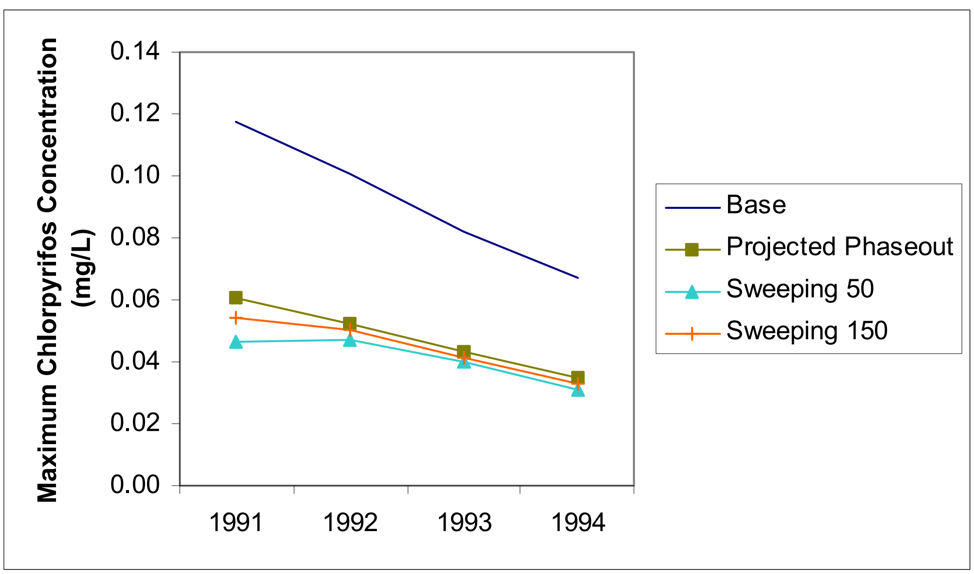


Figure 8-21. Maximum Chlorpyrifos Concentrations (mg/L) for the Street Sweeping BMP Scenarios in Lower San Diego Creek.

Spatially, upper San Diego Creek had the highest simulated Diazinon concentrations each year, followed by the watershed as a whole. The median concentration was usually three to four times higher than that of the entire watershed and the maximum concentrations were at least an order of magnitude higher. This difference is probably due to the fact that upper San Diego Creek contains the highest percentage of urban landuse. Peter's Canyon Wash had the lowest simulated Diazinon concentrations each year; a median concentration an order of magnitude less than the median concentration of upper San Diego Creek and a maximum concentration and order of magnitude less than that of upper San Diego Creek because Peter's Canyon Wash is dominated by agriculture.

Upper San Diego Creek simulated median Chlorpyrifos concentrations were an order of magnitude higher than that of Peter's Canyon Wash and the entire watershed each year. The maximum concentrations of Chlorpyrifos were an order of magnitude higher than Peter's Canyon Wash and the watershed as a whole. This can be explain by the fact that while Peter's Canyon Wash is dominated by agriculture, upper San Diego Creek still contains the most of this landuse. (See Appendix B for actual median and maximum pesticide concentrations for each catchment.)

8.4 Discussion

The Newport Bay watershed model implemented in WARMF simulated median Diazinon concentrations within the range of the median observed concentrations found by the Santa Ana Regional Water Control Board in the TMDL Source Analysis for Diazinon and Chlorpyrifos (SARWQCB, 2001). It simulated median Chlorpyrifos concentrations an order of magnitude greater than the median concentration observed. Maximum concentrations for both pesticides were always at least two orders of magnitude higher than measured maximum concentrations. This is the result of an imperfect simulation of flow storm events. The reduction of the volume of water flowing through the system during storm events decreases the dilution of a finite amount of pesticide applied to the land surface causing stream concentrations to be higher than those observed. This lack of dilution causes pesticide concentrations to be in exceedence of CCC and CMC levels. The sensitivity of the pesticide concentrations to the storm events simulated by WARMF caused us to base our BMP recommendations on the CCC and CMC exceedence days predicted by BASINS.

9.0 BEST MANAGEMENT PRACTICES

9.1 Introduction

Given the results of Sections 7 and 8, further management solutions are needed. BMPs are techniques or structural controls that manage the quantity and improve the quality of storm water runoff in the most cost-effective manner. BMPs can either be engineered structural devices or institutional education programs designed to reduce runoff and pollutant loads. The practices differ as to costs, effectiveness of pollutant removal, site feasibility and financial suitability for a location. In addition, BMP goals can vary with location and project depending on desired flow control or pollutant removal. Solutions to the toxicity problem in Newport Bay involve incorporating BMPs, either independently or in tandem, to reduce the load of pesticides into Newport Bay. This section of our analysis explores various management practice solutions, identifies alternatives best suited for the physical environment of the watershed, isolates the most effective practices for removing sediment, and evaluates the costs associated with each practice to identify the primary management alternative that solves our stated objectives.

9.2 Background

Reductions in the amount of pervious surfaces within the watershed, due to recent changes in landuse from farmland to residential and comercial areas, have greatly increased the volume of runoff that occurs during storm events. As the water can no longer soak into the soil and enter the groundwater system, it pools and runs off the asphalt and concrete before entering the channelized storm drain system, thus reducing the natural infiltration and runoff regime which provided groundwater recharge. This runoff mobilizes pollutants that accumulate on impervious surfaces in neighborhoods and industrial areas between storms and quickly transports the concentrated flow into the tributaries of Newport Bay. Due to the nature of the pollutants considered in this report, reducing the OP pesticide concentrations in the storm runoff before it reaches the Bay will have a beneficial effect on the receiving water body and its inhabitants. By implementing structural BMPs, pollutants trapped on soil particles and dissolved in the water will be subject to treatment by gravitational settling and biological degradation.

Best management practices can be divided into two types: structural and non-structural. Structural practices involve the construction or implementation of a physical component that will help reduce pollution. Non-structural practices may involve education programs, which will convey a pollution reduction message that relies on individuals to take appropriate actions to reduce pollutant use. In general all BMPs will be evaluated based on their individual:

- Toxicity Reduction Effectiveness
- Site Characteristics
- Cost Effectiveness

9.3 Structural Management Plans

Structural BMPs are physical measures taken to alter the transport mechanism of a target pollutant. These measures could include: storm drain filters, porous pavement, bioswales, retention ponds and other physical barriers or filters. These practices have a wide variation in cost and efficiency due to the general scope of targeted pollutants. Limited research has been done on the effectiveness of BMPs to reduce pesticide concentrations in runoff, particularly in western states, thus using the most applicable research we compiled a list of relevant BMPs for analysis including:

- Retention & Detention Basins (Figure 9-1)
- Infiltration Practices (Basins & Trenches) (Figure 9-2)
- Vegetated Swales & Filter Strips (Figure 9-3)
- Sand Filters (Figure 9-4)

9.3.1 Retention and Detention Basins

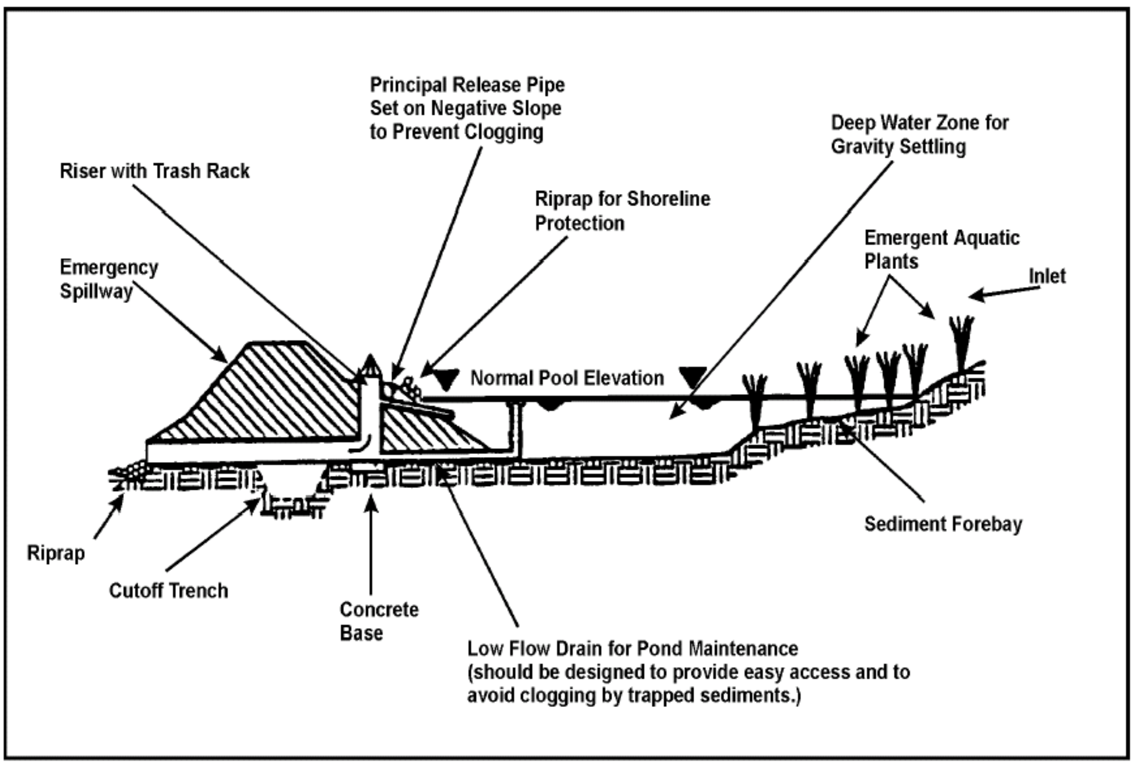


Figure 9-1. Retention/Detention Basin - Potential BMP.
Source: EPA (http://www.epa.gov/owm/mtb/wetdtnpn.pdf)

Retention and detention basins are designed to capture a volume of runoff. A detention basin temporarily retains runoff water for subsequent release, and a retention basin will retain runoff water until it is displaced, in part or in total, by the next runoff event (U.S. EPA, 1999). Benefits of both include removal of sediment (and adsorbed pesticides) by gravitational settling, and reduction of downstream runoff. In addition, retention basins can be designed to allow water to be held for residence times that correspond with degradation rates of targeted chemicals, resulting in a reduction of OP pesticide concentrations before the water is released. Due to the adsorption characteristics of Chlorpyrifos, this type of system would be effective in reducing Chlorpyrifos concentrations and thus loading to the creeks and Bay. Since Diazinon is more likely to be in the water column wet ponds are probably less effective in removing dissolved Diazinon and other mobile compounds (Watershed Protection Techniques, 2001).

As large amounts of standing water are present in most retention and detention basins, they are a perfect breeding location for mosquitoes. Recently, this problem has been focused on by public health specialists as a potential public health threat and nuisance. The potential for disease transmission in the form of viruses, protozoa, and bacteria from mosquitoes does exist in the US (Stormwater, 2002). Yet many other types of insects and pests can find suitable living conditions in the standing water of such BMPs. Recent awareness of this problem has led to simple modifications of many storm water BMPs to reduce the possibility of standing water. What this attention has shown is the importance of maintenance in the operation of any BMP to maintain function.

Another drawback of retention/detention basins in general is the amount of area that is required to store large volumes of water. Retention basins in particular, as they are designed to hold very large volumes of water for long periods of time, have to be constructed large enough to capture the predetermined volume of storm runoff from a storm event which requires treatment. In areas with high land prices, such as in the Newport Bay watershed, this raises the cost of implementation to a point at which it may not be a cost effective BMP. Currently there are nine detention basins in the Newport Bay watershed near the base of the foothills, primarily for flood control. Due to their locations, they receive little storm water runoff from highly urbanized areas and in general are not strategically located to be effective in the removal of the OP pesticides.

9.3.2 Infiltration Basins/Trenches

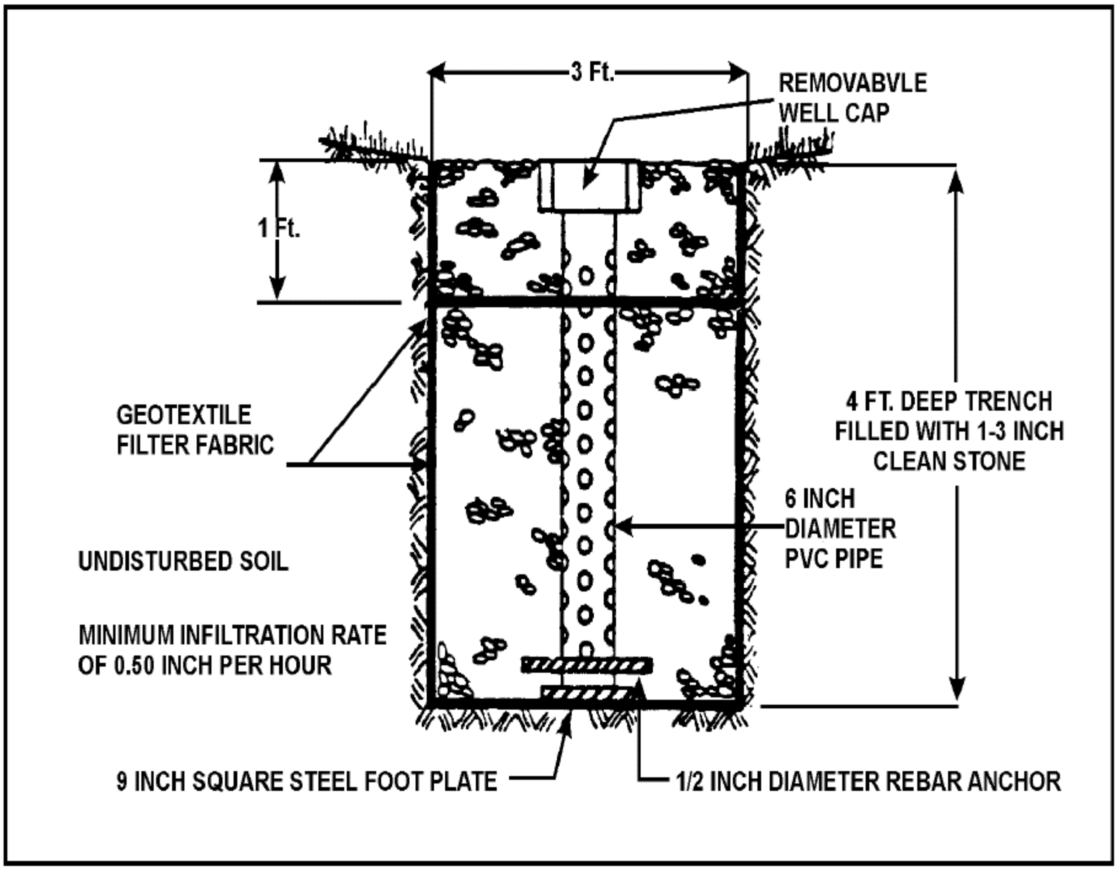


Figure 9-2. Infiltration Basin - Potential BMP.
Source: EPA http://www.epa.gov/owm/mtb/infltrenc.pdf

Infiltration basins capture stormflow and allow the water to infiltrate to the subsurface over a period of hours or days. Generally the water in an infiltration basin is designed to allow percolation in about 3 days so as to prevent mosquitoes from breeding and odors from developing (U.S.EPA, 1999). The benefits of infiltration basins include a reduction of downstream peak flow, sediment removal, and degradation of contaminants as water percolates through the various soil layers. Other types of BMPs such as sand filters are often placed at the entrance of infiltration systems to provide additional pollutant and sediment removal before the water enters the soil (U.S.EPA, 1999). Generally, infiltration trenches have relatively low cost because they can be simple in design and require a small design footprint (U.S.EPA, 1999). This makes them particularly suitable for highly urbanized areas. Furthermore, these basins help to recharge groundwater in areas with large percentages of impervious surfaces. In general, ground water has a lower incidence of OP pesticide contamination than streams because water infiltrating the land surface moves slowly through soil and rock formations on its way to ground water and through the aquifer. This contact with soil and consolidated materials and the slow rate of flow allow greater opportunity for sorption and degradation of pesticides (USGS, 1999). The short half-lives of both these pesticides reduces the potential for long-term groundwater contamination. Thus, a system that promotes the ponding of water and concentrating of OP pesticides so that they can rapidly infiltrate the subsurface is an effective and viable option.

Thoughtful landscape design can incorporate these types of systems with sports fields that offer large amounts of pervious surfaces while maintaining multiple use values. However, large dry infiltration basins are often considered to be an eyesore by the public and are not always well received in residential areas. As a majority of the pesticide-laden runoff is coming from residential areas in the Newport Bay watershed, careful design of the infiltration basins would have to be undertaken to appease the public.

9.3.3 Vegetated Swale and Filter Strip

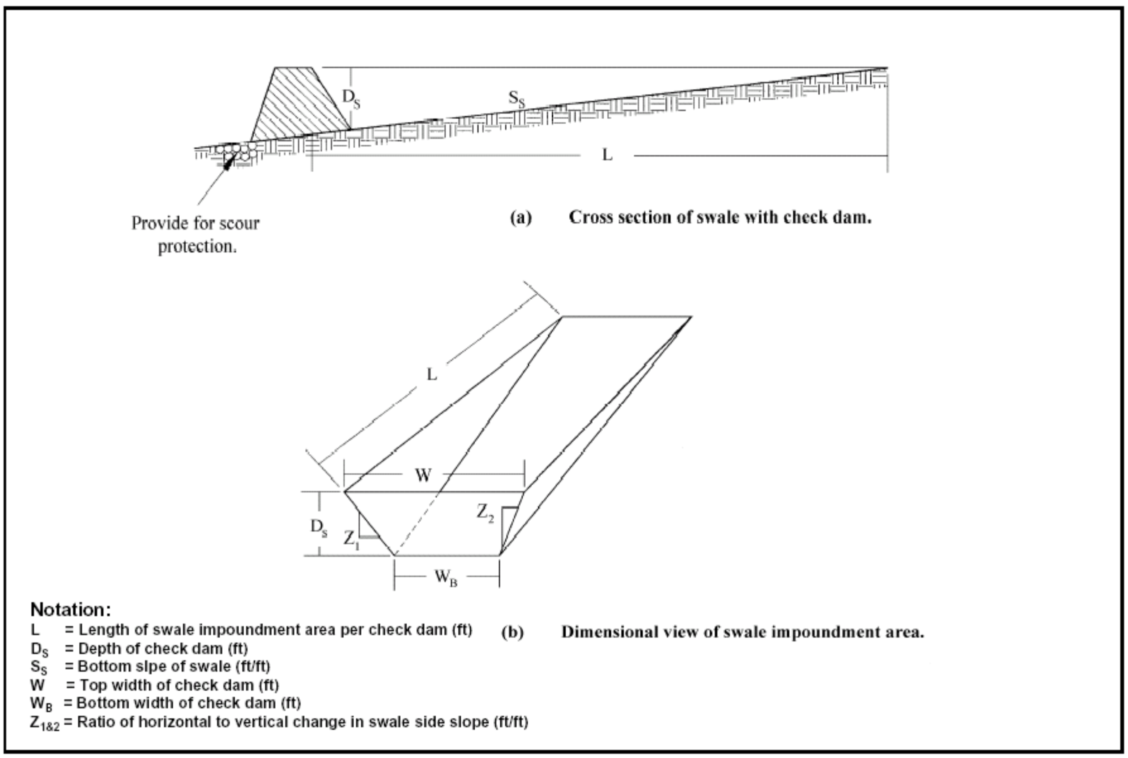


Figure 9-3. Vegetative Swale - Potential BMP.
Source: EPA http://www.epa.gov/owm/mtb/vegswale.pdf

Vegetative swales are broad, shallow channels with a dense stand of vegetation covering the side slopes and bottom channel. Vegetative swales can decrease downstream runoff, trap sediment and pollutants, and promote infiltration.

Vegetative swales can be dry or wet. Dry swales can be implemented in areas where standing water is not desired, such as in residential areas. Wet swales can be implemented in areas where standing water is acceptable, and where groundwater level is close enough to the surface to maintain the conditions between storm events (U.S.EPA 1999). Because Newport Bay has a semiarid climate, dry swales are a practical choice as far as maintenance and costs are concerned.

Vegetated filter strips are densely planted, and uniformly graded areas that intercept sheet runoff from impervious surfaces such as parking lots, highways, and rooftops. Grass filter strips can be planted with turf grass or other vegetation. Filter strips are designed to trap sediments and allow for partial infiltration resulting in decreased downstream peak flow. Stormflow can enter the filter strip by sheet runoff, or be distributed along the width of the strip using a gravel trench or other level spreader. Vegetated filter strips are often used in combination with other BMPs. They can be implemented for pretreatment of storm water prior to its entering other filters or retention systems, or in combination with riparian buffers to stabilize drainage channels and stream banks (U.S.EPA, 1999).

The effectiveness of vegetated buffers depends on the physical and chemical characteristics of the pesticides. Pesticides with high adsorption ability will be more easily trapped in a vegetated buffer strip. The importance of water infiltration as a mechanism of trapping moderately adsorbed pesticides (i.e. relatively soluble such as Diazinon) is illustrated by studies that have shown that riparian buffers do little to reduce concentrations of moderately adsorbed pesticide in runoff, although concentrations of the strongly adsorbed Chlorpyrifos are reduced (USDA, 2000). The efficiency of buffers to trap runoff and remove pesticides was studied in Oklahoma. Sixteen-foot wide Bermuda grass (*Coynodon dactylon*) strips were found to yield approximately 62 to 99 percent removal of Chlorpyrifos in runoff (USDA, 2000).

These systems would remove a larger fraction of Chlorpyrifos than Diazinon due to adsorption characteristics and sediment removal efficiencies of such practices. Filter strips require maintenance to maintain their removal efficiency, which will increase their cost. Furthermore, in regions with a high water table, standing water can pool in these systems causing maintenance problems as well as a source of odors and pests. In more arid climates vegetated filter strips can prove very effective. Vegetated swales could be installed throughout the watershed more easily than some other infiltrative BMPs as large-scale excavation is not required.

9.3.4 Sand Filter

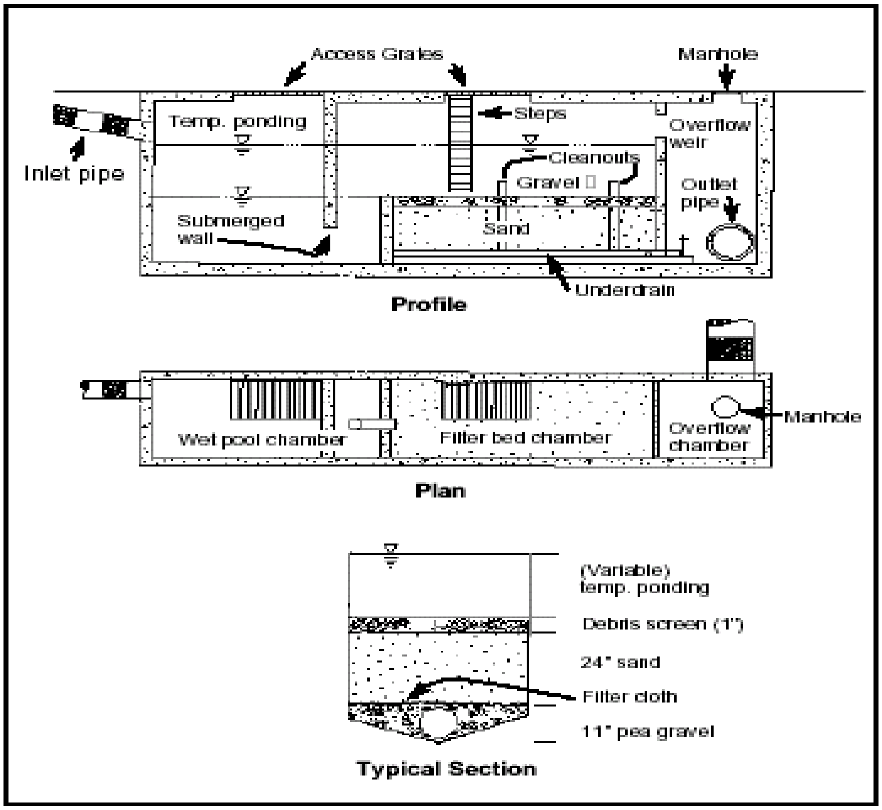


Figure 9-4. Sand Filter – Potential BMP.
Source: U.S.EPA http://www.epa.gov/ost/stormwater/usw_c.pdf

The sand filter in its most simple form has been used for decades as a basic water quality treatment technology. Sand provides both filtration and a home for beneficial bacteria, which further degrade pollutants in the water. In the basic form for treating storm water, the flow enters a concrete collection basin where it slows and pools allowing sediment to settle out of water. The stormflow then drains through a sand filled chamber, which cleans the water further. From here the water exits to a treatment center or storm drain. At present there are many different designs of sand filters, due to differences in treatment volume and target pollutant. Recently, large scale, multiple chamber sand filters have been used as a management technique to improve storm water quality. Sand filters can be very effective at removing certain pollutants from water including suspended and dissolved constituents. When the sediment load is very high, the filter can fill with sediment and clog, and as the through flow rate is reduced, water can flow over bank and flood. Due to this problem, regular maintenance of sand filters is required, particularly after large storms, if they are to maintain their optimum removal efficiencies. Sand filters could assist in the Newport Bay watershed by removing sediment, which may carry OP pesticide molecules sorbed to it. Sand filters work well in dry climates, so they would fit well in the watershed, yet their high cost can negate large-scale applications of this technology.

9.3.5 Street Sweeping

Street Sweeping programs exist in many municipalities and are designed to circulate machinery capable of cleaning streets and roads to prevent trash, sediments and other debris from entering stormwater. Recent estimates are that new vacuum assisted dry sweepers might achieve a 50-88% overall reduction in the annual sediment loading for a residential street, depending on sweeping frequency (www.stormwater.net, 2002). WARMF model scenarios that simulate street sweeping every fifty days predict an efficiency of 25% pesticide reduction can be achieved. This management practice involves up front capital costs for equipment and annual maintenance and operation expenses based on frequency and distance of use, thus larger municipalities with larger budgets have the luxury of choosing street sweeping as an alternative.

9.4 Nonstructural Management Plans

Reducing pollutant levels through methods other than structural BMPs can also have beneficial impacts on the Bay. Educating people within the watershed about the correct use of pesticides can have the greatest benefit by reducing the amount of pesticides used by homeowners and commercial landscape services. Furthermore, this education will promote correct use of all pesticides, including those that will replace Diazinon and Chlorpyrifos once they are phased out. Non-structural practices studied in this report include:

- Education Programs
    - Media Campaigns
    - Intensive Training
    - Integrated Pest Management
- Household Hazardous Material Collection Programs
- Public Policy/Development Standards

9.4.1 Education

Education as a BMP implies a variety of plans targeted at raising people's awareness so they make behavior decisions that positively affect the watershed. In the scope of alternative management plans, education is a combination of techniques used to reduce the amount of pesticides applied in the watershed and promote proper handling and disposal and alternative solutions to pest problems. Education on the proper methods of application, application rates and alternatives to pesticides can help to reduce the amount of pesticides that are carried by urban runoff. Alternatives to pesticides, such as in integrated pest management program and pesticide alternatives such as insecticidal soap or natural bacteria can also reduce the need for pesticides

(U.S.EPA, 1999). It is difficult to quantify the effectiveness that education has on directly reducing the loading of these two particular pesticides. Our analysis investigated various forms of education and drew from the literature to adopt pesticide reductions based on various education programs such as media campaigns, intensive training and integrated pest management.

9.4.2 Media Campaigns

Media campaigns include radio, television, direct mail, signs, advertisements, billboards, and newspapers to broadcast a general watershed message or specific information on pesticide use. In Alameda County, California, municipalities banded together to educate the people about storm water problems in general. Surveys indicate that nearly 70 percent of those aware of the campaign changed their behavior, which represented 48 percent of the total surveyed sample (NRDC, 2001).

9.4.3 Intensive Training

This type of training involves using workshops, consultation and guidebooks to send a much more complex message to a smaller and more interested audience. From market surveys each of these techniques can produce up to 10-20% improvement in selected watershed behaviors among their respective target audiences (Pollution Prevention Fact Sheet, 2001). This involves using workshops, consultation, and guidebooks to send a much more complex message to a smaller and directly involved audience (www.stormwatercenter.net, 2001). Intensive training may be especially advantageous to educate users such as structural pest control operators who will continue to use both pesticides.

9.4.4 Integrated Pest Management (IPM)

In an effort to reduce the loading of pesticides into the environment and to develop a system of pest control which is less damaging to the environment, scientists developed a system known as, "Integrated Pest Management (IPM)." IPM is considered a long-term solution to pest problems, which may include measures such as habitat manipulation, biological control or the use of natural pest controls, such as insects. IPM seeks to change cultural practices with regards to pesticide use by making the public more aware of ecosystem based thinking and the damage of pesticides (http://www.ipm.ucdavis.edu, 2001). While far from being a quick solution to the toxicity problem in California, IPM seeks to bring about gradual awareness and change at a pace that will maximize benefits and minimize economic impacts as IPM becomes more widely accepted.

9.4.5 Improved Labeling

This education method involves attaching a tag to the packaging of every pesticide container sold in the watershed. The tag could be attached directly at the point of sale and could be drawn up with information specific to that watershed and pesticide type. Information printed on the tag could include basic toxicity information, correct application and storage methods, and disposal contact information for local collection sites that handle the contained type of hazardous waste. This type of educational BMP would directly reach every buyer of pesticides in the watershed, which would be unsurpassed in efficiency in comparison to other BMPs. Furthermore, it is a very low cost option as the major expenditure would be the printing and handling costs associated with the tags.

9.4.6 Household Hazardous Waste Collection Program

Many counties around the country are realizing the benefits of creating a program through which private citizens can properly dispose of hazardous household wastes. By preventing the wastes from reaching landfills, ground and surface water supplies are protected, preventing risk to the environment and public health, and avoiding future cleanup costs. Orange County has a household hazardous waste collection program which currently contracts the actual collection and disposal out to a waste disposal company. This company operates four collection centers around the county where the public can bring their waste for free disposal. Annually approximately 50,000 households drop off wastes for disposal. In 1999-2000 2.9 million pounds of wastes were collected of which 105,000 pounds were poisons (pesticides and herbicides). In 2000-2001 2.8 million pounds of waste were collected, of which 106,500 pounds were poisons. Orange County's program does not currently advertise its programs and services. However, waste collection contractors who collect general refuse for Orange County often include fliers on the program in their monthly bills in order to reduce the amount of hazardous waste they have to handle.

9.4.7 Public Policy/Development Guidelines

The use of planning, policy and development guidelines can be composed to reduce the amount of impervious surface in a watershed, reducing the amount of storm water runoff and pollution reaching the Newport Bay. Additionally, setting planning boundaries will also lower the cost of services to rural areas so the monies saved could be used in areas of higher development. This study recognizes the potential for landuse planning as a tool comparable to other non-structural management practices, however the effectiveness is difficult to quantify as are the costs associated with implementation, and benefits of such policies; making an analysis beyond the scope of this project.

9.5 Analysis

9.5.1 Suitability of Various Best Management Practices

From the range of the potential BMPs that could be applied to provide pollution control within the Newport Bay Watershed only certain practices will adequately provide effective functions due to the specifics of the watershed. The above BMPs were analyzed with respect to the climate, land requirements and construction cost. The arid climate of the area poses restrictions on the suitability of certain BMPs that require wet vegetation and ponded water year round. Furthermore, due to the level of urbanization within the watershed it will be difficult to find areas of open land to implement large scale BMPs able to handle the amount of runoff from large residential sites. The size of land area required by each BMP will have an impact on its cost of implementation. We selected BMPs that promote the infiltration and short-term ponding of storm water and evaluated them on their efficiency of removing pollution and the cost to construct such a practice and ranked them from highest to lowest in order to choose the best available alternative.

9.5.2 Pollution Removal Efficiency Analysis

In order to reduce toxicity in the Bay, OP pesticides must be prevented from reaching surface waters. The pesticide load can be sorbed to sediment, thus controlling sediment offers a useful approach to controlling Chlorpyrifos and Diazinon related toxicity.

The transport of Chlorpyrifos is assumed to behave similarly to total suspended sediment (TSS), thus practices that are efficient at removing TSS will be efficient at removing Chlorpyrifos. Evaluation of the effectiveness of Diazinon reduction, however, is dependent upon the residence time between application and the eventual introduction of the pesticide to streams or the Bay. Greater residence times allow for more degradation to occur before the pesticides reach surface waters that are inhabited by aquatic organisms. BMPs that promote sedimentation and infiltration processes will result in the greatest residence times for pesticide transport.

We assumed that structures that reduce the velocity of flow and allow sedimentation to occur would remove Chlorpyrifos bound to sediment. These ponding structures will increase biotic process promoting higher bacterial process and higher degradation rates of Diazinon. Thus, the most appropriate structural BMPs for Newport Bay include: retention basins, detention basins, infiltration trenches, sand filters and grassed swales (Refer to Table 9.1). The National Pollutant Removal Database developed by the Center for Watershed Protection (2000) compiled data from field sampling upstream and downstream of various BMPs. Estimates of storm treatment practices efficiency should not be regarded as a fixed or constant value, but rather as a general estimate of long-term performance (Winer, 2000). Due to the variability in design specifications and variations in calculating efficiencies the performances are best classified with a qualitative ranking. Table 9-1 is a summary of the results for BMP efficiencies published in the EPA's Storm Water Management Practices Guidelines (1999) and the Center for Watershed Protection's Pollutant Removal Database (2000) with a qualitative ranking created for this report used for further analysis.

Table 9-1. Sediment Removal Efficiency Of Various BMPs.

BMP Type	Range (Mean) of TSS Removal[1]	Median Removal (%) Efficiency (TSS)[2]	Qualitative Ranking	Rank
Infiltration Practices (Including basins and trenches)	50-96% (89)	NA	Good	1
Retention Basins	70% (70)	65	Good	1
Detention Basins	50-80%	61	Good	1
Street Sweeping	50-80%[3]	N/A	Good	1
Grass Swales	30-65% (66)	68	Moderate	3

1. EPA, Center for Watershed Protection.
2. National Pollutant Removal Performance Database.
3. Street Sweeping Efficiencies taken from www.stormwatercenter.net, 2002.

Infiltration systems in general are very effective at removing pollutants from storm water (U.S.EPA, 1999). Infiltration systems can be considered 100 percent effective at removing pollutants that partition to water, and therefore infiltrate through the system, since the pollutants found in this volume are not discharged directly to surface waters (U.S.EPA, 1999). Based on the above results, these BMPs were then evaluated with respect to the costs associated with the construction, operation and maintenance of each BMP, in order to calculate the most cost-effective alternative available.

9.5.3 Cost Analysis

In evaluating the proposed management practices, we considered the cost to construct and maintain structural and non-structural BMPs as an integral portion of our recommendation to reduce pesticides from reaching Newport Bay. Publications by the Center for Storm Water Management, journal articles and summaries published in recent EPA Storm Water Management Practice documents were used to compare construction and maintenance costs across management options. Brown and Schueler have stated that typical costs of constructing BMPs can be calculated with equations based on the size, or volume of water to be treated (U.S.EPA, 1999). Equations published by Wiegand et al. in the late 1980's (Wiegand, 1986) have been cited in

EPA documents as recently as 1999 (U.S.EPA, 1999) and are thought to be appropriate for the purposes of this project. We applied these equations and calculated the estimated cost of constructing and maintaining various BMPs, leading to our recommendation as to the most cost-effective BMPs available for OP pesticides.

Structural BMPs costs can be highly variable due to design variations and criteria. Variables in construction and maintenance costs include: Storm water treatment practice geometry, site characteristics, and influent pollutant concentration. Site characteristics that can also influence pollutant removal capability include soil type, rainfall, latitude, catchment size, watershed landuse, and percent impervious (Winer, 2000). Non-structural educational BMPs have costs associated with programs such as media campaigns or training. However the effectiveness of these solutions is difficult to quantify in the same scaleable measures as structural BMPs, in a dollar per pollutant removed figure.

9.6 Cost Equations

9.6.1 Structural BMP Construction Costs

Construction costs of structural BMPs were calculated using equations drawn from field applications, and reflect the construction cost based on the volume of water treated. Storm water treatment practices have certain spatial requirements such as surface area and volume necessary to treat a quantity of storm water from a catchment and this variability is incorporated into the equations (U.S.EPA, 1999). Our efforts to account for site variability throughout the watershed started with a range of sizes for Retention, Detention and Infiltration practices. We assumed that we would need enough unit BMPs to satisfy the relationship between BMP size and the treatable volume of water produced in a runoff event in order to compare the cost to construct one type of BMP with other BMPs. This relationship is estimated from the fraction of the watershed that is impervious and the surface area requirement for a BMP as a percentage of the impervious surface (Refer to Table 9-2). Using this information we used the corresponding surface area outlined in our cost equation to calculate the number of BMPs necessary to satisfy the minimum land requirements to achieve effective treatment. Effective treatment is undefined in the reported text (U.S.EPA, 1999 from Claytor and Schueler, 1996) and is assumed to be the level reported by Center for Watershed Protection Best Management Practice Database (1999) and summarized in Table 9-1.

These are conceptual level costs and are based on estimates and assumptions sufficient for the level of computations necessary within our analysis. We have attempted to be consistent in the approach for all BMP costing, so that the construction costs can be compared across various BMPs. All costs have been adjusted using the EPA recommended rainfall adjustment factor that modifies the cost based on rainfall regions (southern California rainfall adjustment factor equals 1.24, based on U.S.EPA, 1999).

Table 9-2. Relative Land Consumption of Storm Water BMPs.

BMP Type	Land Consumption (% of Impervious Area)[1]	% of Impervious Area Used in This Study	Amount of Land Required in Newport Bay (Acres)
Retention Basin	2-3%	2.5%	450
Sand Filter	0-3%[2]	2%	360
Infiltration Trench	2-3%	2.5%	450
Infiltration Basin	2-3%	2.5%	450
Swales	10-20%	15%	2,704

1. EPA Storm Water Management Practices.
2. Sand Filters have the ability to be constructed underground thus not requiring land surface.

9.6.2 Retention Basins

Cost equations published in EPA's Storm Water Management Guidelines (1999) from Brown & Schuler (1997) consider the volume of the basin a strong predictor of construction cost. Retention Basins are evaluated using the equation (U.S.EPA, 1999):

$$\text{Individual Cost} = 18.5V^{0.70},$$

Where V is the total basin volume in cubic feet

The number of retention basins required in the Watershed was calculated using the land requirements for this BMP to be effective (2.5% of impervious surface from Table 9.2) and the amount of impervious land in the watershed (assumed 18,000 acres), which requires 450 acres of retention basins. In order to evaluate a range of sizes of retention basins a matrix of sizes and costs was created (Refer to Table 9.3). The surface area to volume ratio was adapted from infiltration basin studies within the San Fernando Valley (Chralowicz et al. 2001).

Table 9-3. Retention Basin Cost Summary.

Size surface area (Acres)	Volume (Cubic Feet)	Number Required in Watershed	Base Capital Cost (Millions)
0.25	23,300	1800	$ 47
0.5	74,000	900	$ 53
1	148,000	450	$ 43
3	547,000	150	$ 36
5	952,000	90	$ 32

9.6.3 Detention Basins

Detention Basins are evaluated using a similar equation (U.S.EPA, 1999):

Individual Cost = $7.47V^{0.78}$,

Where V refers to the total basin volume in cubic feet. The number of retention basins required in the Watershed was calculated using the land requirements for this BMP to be effective (2.5% of impervious surfaces from Table 9.2) and the amount of impervious land in the watershed (assumed 18,000 acres) results in 450 acres of detention basins required in the watershed. In order to evaluate a range of sizes of detention basins a matrix of sizes and costs was created (Refer to Table 9-4). The surface area to volume ratio was adapted from infiltration basin studies within the San Fernando Valley (Chralowicz et al. 2001).

Table 9-4. Detention Basin Cost Summary.

Size surface area (Acres)	Volume (Cubic Feet)	Number Required in Watershed	Base Capital Cost (Millions)
0.25	23,300	1,800	$ 68
0.5	74,000	900	$ 52
1	148,000	450	$ 58
3	547,000	150	$ 54
5	952,000	90	$ 50

9.6.4 Infiltration Practices

Generally, costs are highly variable for infiltration practices from site to site, and difficult to estimate due to the lack of recent cost data (U.S.EPA, 1999). Acknowledging the scarcity of information, the EPA has summarized cost equations developed by Brown and Schuler (1997), which attempt to capture the range of costs.

9.6.5 Infiltration Trenches

Information regarding infiltration trenches was created using a cost equation developed by Brown & Schuler (1997) with various trench sizes adopted from SWRPC and thought most appropriate to occur in the watershed.

Individual Cost = 2.5V

Where V refers to the treatment volume (cubic feet) within the trench, assuming a porosity of 32% (U.S.EPA, 1999). The cost equation for infiltration trenches was published as a range between 2V to 4V, where Brown and Shuler believed the average cost to be 2.5V (U.S.EPA, 1999). In order to create a matrix of alternatives of reasonable size we adopted the sizes used in studies reported by SWRPC in the Storm Water Treatment Practices (1999). The surface area was then converted to acres to calculate the number necessary within the Watershed. The reported sizes of these practices are smaller than other practices and are assumed to be appropriate for the dense residential and commercial development within Newport Bay Watershed (Refer to Table 9-5).

Table 9-5. Infiltration Trenches Cost Summary.

Size (Acres)	Dimensions (Feet)	Volume (Cubic Feet)	Number Required in Watershed	Base Capital Cost (Millions)
0.01	3 x 4 x 100	1,200	40,541	$ 150
0.023	6 x 10 x 100	6,000	16,199	$ 301
0.003	3 x 1 x 100	300	161,987	$ 150

9.6.6 Infiltration Basins

For Infiltration Basins we used a different equation developed by Schueler:

$$\text{Cost} = 13.2V^{0.69},$$

Where V refers to the water volume (defined below) the basin is designed to treat. The number of infiltration basins required in the watershed was calculated using the land requirements for this BMP to be effective (2.5% of impervious surfaces from Table 9-2) and the amount of impervious land in the watershed (assumed 18,000 acres) results in 450 acres of infiltration basins required in the watershed. In order to evaluate a range of sizes of detention basins a matrix of sizes and costs was created (Refer to Table 9-6). The surface area to volume ratio was adapted from infiltration basin studies within the San Fernando Valley (Chralowicz et al. 2001).

Table 9-6. Infiltration Basins Cost Summary.

Size surface area (acres)	Volume (Cubic Feet)	Number Required in Watershed	Base Capital Cost (Millions)
0.25 acre	23,300	1,800	$ 30
0.5 acre	74,000	900	$ 34
1 acre	148,000	450	$ 21
3 acre	547,000	150	$ 18
5 acre	952,000	90	$ 16

9.6.7 Sand Filter

Sand filters have not been used as long as other BMPs for storm water treatment (U.S.EPA, 1999), thus recent cost equations are based on a limited application of these as management practices. Our estimates utilize cost information from the installation of a sand filter system for the City of Denver, CO and cost equations. The Denver example was chosen to capture the range of costs between an actual application and the theoretical cost equations. The sand filter system installed in Denver cost between \$30,000 and 50,000 (U.S.EPA, 1999). The EPA (1999) reports there is a wide range in costs (between \$2-\$6 per cubic foot of water quality volume) with a mean of \$2.50 per water quality volume. Using this number results in the cost equation:

$$Cost = 2.5WQ_v$$

Water Quality Volume is an abstraction of volume of water from a small storm event (less than 2") used for planning discharge volumes and BMP size specifications where the Soil Conservation Science method fails. The method for computing peak discharge for water quality storm is adapted from Schueler and Claytor (1996) and summarized in Storm Water Treatment Practices (U.S.EPA, 1999). WQ_v methodology relies on the impervious fraction of a watershed, the precipitation and the resulting runoff within the catchment. The equation below was used to calculate WQ_v for this practice.

$$WQ_v = [0.05+0.009(I)] \times P \times A/12 \times 43,560 \text{ ft}^2/\text{acre}$$

Where
I = impervious fraction of watershed
P= precipitation (inches)
A= watershed area (acres)

This WQ_v can then be used to calculate a corresponding Curve Number (CN), which relates the storm to a flow rate thus engineering techniques can be applied to design a properly sized catchment or diversion. For the purposes of sand filters it was assumed that the impervious fraction was 50%, of the developed land within the watershed (36,000 acres) and for a 1-inch storm. The resulting volume of water equaled 150×10^6 ft^3 for the cost equation. For the purposes of this study, we calculated the cost of constructing sand filters as the range of the resulting costs developed from the two methods described above (Refer to Table 9-7).

Table 9-7. Cost of Sand Filter Construction in Newport Bay.

Method	Cost per Impervious Acre	Impervious Acres in Watershed[1]	WQ_v[2]	Construction Cost (Millions)
Extrapolation from similar Project in Denver	$40,000	18,000	N/A	$ 721
$2.5WQ_v$	N/A	N/A	150×10^6, ft^3	$ 464

1. Impervious land in the Watershed was calculated as 50% of the developed land (36,000 acres).
2. See Appendix for Calculation of Water Quality Volume.

9.6.8 Grass Swales

Vegetative management practices evaluated here consist of grass swales. Published estimates vary depending on the method for establishing vegetation. Our analysis focused on starting vegetation from seed at an estimated cost of $0.25 per square foot (U.S.EPA, 1999). The number of acres of swales required in the watershed was calculated using the land requirements for this BMP to be effective (from Table 9-2) and the amount of impervious land in the watershed (assumed 18,000 acres) (Refer to Table 9-8).

Table 9-8. Grass Swale BMP Cost Summary.

Method	Cost (Seed)	Amount of Land Required (Acres)	Base Capital Cost (Millions)
Grass Swales	$ 0.25 sq. ft.	2,704	$ 36

9.6.9 Street Sweeping

Street Sweeping possess direct operation and maintenance costs. Innovations in sweeper technology have improved the performance of these machines at removing finer sediment particles, especially for machines that use vacuum assisted dry sweeping to remove particulate matter (www.stormwatercenter.net, 2002). Considering the results of the source analysis, it is assumed the most appropriate method of a street sweeping program would begin with a vacuum assisted type. Cost data is calculated from 1997 dollars (U.S.EPA, 1999) and based on a dollar per curb mile scale factor. Tables 9-9 and 9-10 estimate the cost of running a street sweeping program within the watershed based on results from our modeling sections (Section 8).

Table 9-9. Street Sweeper Cost Data.

Sweeper Type	Life (years)	Purchase Price ($)	Operation and Maintenance Costs ($/curb mile)
Vacuum-Assisted	8	150,000	15

Source: EPA (1999).

Assumptions in the analysis used to create Table 9-10: Annualized Sweeper Costs:

- One sweeper serves approximately 8,000 curb miles during a year (U.S.EPA, 1999).
- The annual interest rate is 8 percent.
- Dollars in 1997 values.

Table 9-10. Annualized Sweeper Costs ($/curb mile year) .

Sweeper Type	Sweeping Frequency					
	Weekly	Bi-weekly	Monthly	Four times per year	Twice per year	Annual
Vacuum-Assisted	946	473	218	73	36	18

Source: EPA (1999).

A street sweeping program that coincides with the application of pesticides and seasonally dry conditions (i.e. summer/fall) before runoff begins would capture the bulk of the pollutants while minimizing the frequency that the streets need to be swept. This option is viable for the region, as model scenarios predict that street sweeping the entire watershed every fifty days would reduce the amount of pesticides in the runoff by up to 25%, while sweeping every one hundred and fifty days has little effect on the amount reaching the Bay. Covering every curb mile in the watershed may be extreme, however it is a viable option to the agencies within the area. Assuming the active agencies pursued this alternative it would cost between $73-$218 a curb mile (in addition to the purchase and O&M costs) for one machine based on the number of sweepings required to be effective.

9.7 Structural BMP Cost Discussion

Best management practices are designed to treat an appropriate volume of storm water that is a function of the type of practice and the size of the catchments intercepting the storm water. Costs were calculated assuming sufficient BMP units would be installed in the watershed to meet the effectiveness threshold based on the amount of impervious land within each catchment. The effectiveness of pesticide removal was extrapolated from the efficiencies of suspended solid removal and the results from the fate and transport analysis. This served our project in two ways:

1) It allowed us to model various BMPs within the entire watershed using a non-point source model by assuming the BMP would reduce the load by the amount over the entire watershed, and
2) Allowed us to compare various structural BMP costs to each other using the same methodology and rank them in magnitude of costs.

These equations reflect the economies of scale and provide relationships concerning the cost and the size of practice. Infiltration and detention practices decreased in cost when the surface area increased, indicating that larger constructed ponds have lower base construction costs. Not included in the construction figures is the cost of land. The cost of land may be the highest variable in determining the overall cost of installing a BMP (U.S.EPA, 1999). Portions of Newport Bay Watershed are highly urbanized and the cost to purchase land for any one of these BMPs may far outweigh the cost to construct such a structure. In this sense, BMPs that have lower land requirements, such as infiltration trenches and infiltration ponds are considered more likely to be implemented and succeed within the Watershed.

In addition to the construction costs we calculated the operating and maintenance costs as a percentage of the base construction costs. Base construction costs can be used as a strong predictor of operation and maintenance costs (U.S.EPA, 1999). As the second phase of our cost analysis, BMPs were compared on their construction costs and ranked in order from least to most expensive and are summarized below in Table 9-11.

Table 9-11. Cost Results Summary.

BMP Type	Construction Cost Range (Millions)	Operation & Contingency (30% of Construction Costs, in millions)	Annual Maintenance Costs (7.5% of Avg. Construction, in millions)	Cost Ranking (Lowest to highest)
Infiltration Basins	$ 16-34	$ 4.8-11	$ 1.5	1
Retention Basins	$ 25.5-38	$ 7.65-11.4	$ 2.54	2
Grass Swales	$ 29	$ 8.7	$ 1.7	3
Detention Basins	$ 49.5-55	$ 14.8-16.5	$ 0.5	4
Infiltration Trenches	$ 121-243	$ 36.3-72.9	$ 13.7	5
Sand Filters	$ 460-720	$ 112.5-216	$ 65.7	6
Street Sweeping[1]	$ 0.15	$ 73-218/curb mile	$0.580-1.5[2]	1

1. Street sweeping is a method to remove pollutants directly from the pavement prior to a storm event, where the other practices are designed to treat storm water.
2. It is assumed the street sweeper covers 8,000 curb miles a year.

General statements regarding construction costs must be considered with care. Comparison of BMPs strictly on costs may misrepresent the benefits associated with each BMP that go beyond the designed function. Additionally, the benefits of a constructed retention pond, or swale go beyond obtaining pesticide reduction goals. Residents and developers may realize increased property values from aesthetically landscaped controls that control urban runoff (U.S.EPA, 1995). This kind of cost benefit analysis was not performed in our analysis, however we feel it should be performed during the planning process, design and siting of BMPs within the watershed. For the purposes of this study, costs are estimated using a cost function, quantified for specific conditions, and then compared relative to other BMPs.

9.7.1 Non-structural BMP Costs

The administration of non-structural BMPs is distributed over a population for a given area and is fundamentally different than physical structures to treat storm water. Engineers can collect samples from site visits and quantify the loads before and after a structural BMP is installed. This is not as straightforward for nonstructural programs. Estimated costs to implement such programs are listed in Table 9-12 is compiled from EPA documents and interviews with County of Orange staff (Grogen, 1997). The cost information is used as a benchmark to compare various alternatives and should be recognized as annual budgets not the start up costs (Table 9-13).

Table 9-12. Non-Structural Costs Summary.

BMP Type	Annual Budget
Household Hazardous Materials Collection[1]	**$ 30,000**[3]
Public Outreach/Education[2]	**$ 270,000**

1. This is a county wide program that operates four collection centers and distributes fliers through mail programs.
2. This is based on a project by the City of Seattle (U.S.EPA, 1999); Cost breakdown in Appendix C.
3. 2.5 people, flier distribution $5,000 year, 2-4 times a year (Orange County, 2002).

9.7.2 Non-Structural BMP Cost Discussion

Non-structural BMPs represent a range of management alternatives used to minimize pollutant loading into urban creeks and streams. Direct marketing methods geared toward homeowners and residents are considered the primary alternative for the problem identified in Newport Bay. An education program that targets the users of pesticides and poses alternative strategies such as Integrated Pest Management may have the highest benefit within the watershed. However, due to the difficulty in assessing the cost-effectiveness of such non-structural plans we realize the need for additional information such as market surveys that can quantify a change in behavior based on various practices before we can make isolated conclusions.

9.8 Summary

In selecting the proper management strategy to reduce OP-pesticide-based toxicity in Newport Bay, we considered the nature of the pollutants and the physical environment of Newport Bay. Based on the fate and transport mechanisms of the pesticides, practices that retain storm water for an amount of time sufficient to allow infiltration and degradation to occur are considered valuable alternatives. Additionally, based on the climatic requirements of the BMPs (e.g. year round water to sustain biological systems) we determined which are less feasible. Considering the level of development within the watershed, structural BMPs that have smaller space requirements were also given greater consideration. This, along with the suspended sediment removal efficiency and the cost to construct such practices are the foundations that allowed us to rank management plans and make a clear recommendation.

Management Plans were ranked according to their effectiveness to remove suspended solids as reported in sampling surveys by the Watershed Protection Center, (1997) and the EPA and reported above in Table 9-1. The top performing plans were further investigated for the cost to construct them and ranked in order from cheapest to most expensive and summarized in Table 9-11. Together this information was summarized in Table 9-13 in order to select the most cost-effective solution. From the prior analysis it became clear which plan is highest in both rankings and was the basis for our overall ranking and recommendation. The highest-ranking practices include infiltration basins and street sweeping programs.

Due to the difficulty in quantifying the costs of non-structural management plans and comparing them in similar context, this level of analysis was not achieved for them. Summarizing the discussion concerning the benefits and costs associated (short of a cost benefit analysis) with each non-structural management plan the most cost effective solution will be educating the residents of the watershed.

Table 9-13. Structural BMP Cost Effective Summary Table.

BMP Type	Effectiveness Ranking	Cost Ranking	Overall Ranking
Infiltration Basins	1	1	1
Street Sweeping	1	1	1
Retention/Detention Basins	1	3	3
Vegetated Swales	3	3	4

Concerning Infiltration Basins, the agencies and residents within Newport Bay watershed would need to spend between \$16-\$67 million dollars to implement the necessary amount of infiltration basins. This reflects the range of units necessary to meet the calculated design efficiency based on suspended solid removal and impervious surfaces. This level is assumed to be sufficient to lower the concentrations below the proposed criteria. Regarding a street sweeping program, a program that sweeps frequently during the dry season can achieve sediment removal efficiencies of up to 80% for capital costs close to \$200,000 and maintenance costs between \$0.5-1.5 million per year.

Municipalities within the watershed could purchase 100 vacuum sweepers for the same price as installing the required infiltration basins, however the operation and maintenance costs for this many machines may outweigh the benefits. A more detailed analysis is needed to evaluate the costs and benefits of both programs to find the optimal combination of practices.

Based on the analysis presented in this section a comprehensive management program that eliminates sources, educates the public and trade people and physically treats storm water would be the most effective solution to reduce toxicity in urban runoff.

10.0 CONCLUSIONS AND RECOMMENDATIONS

Our study explored the sources of toxicity in Newport Bay related to the OP pesticides Diazinon and Chlorpyrifos and the effectiveness of the EPA and SARWQCB phase-out and re-registration policies. We also evaluated a range of BMPs designed to reduce the observed toxicity. Based on a source-analysis approach we analyzed the current levels of toxicity (with respect to CCC and CMC numeric targets), the physical and chemical properties of Diazinon and Chlorpyrifos and the human and ecological risk. We evaluated the risk to ecological and human health resulting from Diazinon and pesticide loading in Newport Bay with the RivRisk 4.0 computer model. Our results suggested that current levels of toxicity in the Bay are higher than is acceptable, creating ecological and human health risks. These findings indicate that a management plan that addresses unsafe concentrations of pesticides in the Bay is necessary. Given the expected phase-out of certain Diazinon and Chlorpyrifos uses, such a management plan should consider both current and future pesticide loading concentrations and implement BMPs accordingly. In order to better understand the implications of the phase-out, we employed watershed assessment to analyze the phase-out of these pesticides and to offer a decision support framework for assessing the predicted effectiveness of the EPA phase-out. Finally, we evaluated the costs associated with various best management practices in order to make a final recommendation addressing the aquatic toxicity derived from these compounds.

Results from our source analysis found that surface runoff is the source of virtually all of the loadings, while atmospheric deposition, sediment remobilization and groundwater sources are insignificant. About 6 pounds of Chlorpyrifos and 35 pounds of Diazinon are annually discharged to Upper Newport Bay. 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 (SARWQCB, 2001). On average; about 1 to 2 lbs. of Diazinon and 1 to 1.5 lbs of Chlorpyrifos are discharged to the Upper Bay during a typical storm event.

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. Due to the high inputs from urban landuses it is likely that with increased urbanization pesticide-related toxicity will continue to be an issue.

The OP pesticide usage restrictions, outlined in Section 2.2, will likely end a significant percentage 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.

Results from our fate and transport analysis found that both pesticides are highly toxic at low concentrations. Their short half-lives of six months or less in the environment illustrate that it is not the lengthy persistence of these pesticides in the environment but rather a high level of toxicity occurring over a short duration. Due to their physicochemical properties, Diazinon is more likely to be found in aqueous systems and occasionally in sediments while Chlorpyrifos is consistently found in both water and sediments throughout the watershed. As both soil and water must be considered in effective management alternatives, we evaluated options that addressed toxicity removal in both media.

Our watershed modeling analysis established that after the partial phase-out of these two pesticides, Diazinon would persist in stormflow events above both the CCC and CMC criteria for aquatic toxicity. The results indicate that, on average, Diazinon will be found in toxic concentrations in all storm events in Newport Bay after the phase-out. In contrast, Chlorpyrifos concentrations appeared to be within the criterion limits more often, even during storm events. Chlorpyrifos concentrations exceed the CCC levels between 1 and 4 days per rain year, and the CMC levels between 0 and 3 days per rain year. These violations highlight the importance of storm water in the transport processes.

Important to note were the differences between the two models we employed. WARMF expressed sensitivity to hydrological events that raised concern to its ability to predict base flows well. Conversely, BASINS predicted stormflow concentrations well, while base flow concentrations were moderate leading us to focus on stormflow. WARMF predicts violations of numeric criteria in almost every precipitation event, by a magnitude larger than sampling observations. BASINS could predict concentrations within reasonable range with a dampened sensitivity compared to WARMF. Although sensitive, the WARMF model did provide us with the benefit of modeling street sweeping efforts and allowed us to come up with an efficiency response to this management practice necessary for cost efficiency analysis.

An important conclusion to be drawn from these model simulations is that the phase-out will be more effective at reducing the number of days above the criteria for Chlorpyrifos than Diazinon. Additionally, we discovered that, short of a stricter command and control policy revision, a complete reduction in urban Diazinon uses is necessary to keep the concentrations below criteria levels. These findings provided the impetus to evaluate various BMPs as further toxicity-reduction will be necessary to protect the biological integrity of the Bay.

After evaluating various structural management practices, we concluded that infiltration basins and street sweeping are the most cost-effective solution followed by retention/detention basins and grass swales. This determination was based on a number of criteria. Crucial to the initial selection of our management strategies were the fate and transport characteristics of the pesticides, the physical components of the watershed, the physical requirements of the landscape, and land consumption requirements of each BMP. Secondary analysis focused on the efficiency at which sediment and pollutants were removed by such practices and the cost associated with constructing them. In order to reduce the aquatic toxicity using infiltration basins it would require between 90 and 1,800 infiltration basins depending on size at a total cost between $16-$34 million dollars to construct. Operation and contingency costs for infiltration basins amount to $7 million, while annual maintenance costs amount to roughly $2 million per year.

In order to reduce toxicity using street sweeping practices costs would range from roughly $200,000 capital cost to purchase a street sweeper and between $0.5 to 1.5 million a year in operation and maintenance. If infiltration basins are implemented to control the predicted pesticide-related toxicity, we recommend monitoring for the presence of Diazinon, Chlorpyrifos and the accumulation of additional contaminants in the stored water and sediments, as a means to assess their overall removal function.

Street sweeping programs implemented throughout areas of high pesticide loading can have the highest cost effectiveness especially if coordinated with seasonal applications and proceeding precipitation events. Despite the high costs, infiltration basins offer an effective management strategy for achieving our goal of reducing pesticide-related toxicity. Additionally, the installation of these basins will provide supplementary water quality functions to Newport Bay as they effectively capture and store sediment as well as other non-point pollutants such as fertilizers, metals and pesticides common in urban watersheds. Unfortunately, this analysis was beyond the scope of this project.

In conjunction with these strategies, we recommend tighter policy restrictions on Diazinon and Chlorpyrifos usage. The predicted concentrations of Chlorpyrifos will still violate the suggested criteria, however less of a problem than Diazinon. A hypothetical shift in pesticide use to Chlorpyrifos would strengthen our recommendation of using infiltration basins based on the fate and transport results and Chlorpyrifos tendency to adsorb to sediment. This recommendation is warranted based on our model results, though is beyond the scope of this project with respect to implementation. The results are valuable as a means to assess the effectiveness of the adopted policy and speak to the need for predicting the degree of toxicity to be expected in the first few years of the phase-out.

Inclusion of educational programs aimed at elevating the awareness level of the public will be key to the success of reducing toxicity in Newport Bay. Educational programs can be implemented at a fraction of the cost of most structural BMPs. While our study did not quantify the effectiveness of education programs, targeting management options at the source rather than fixing a problem after toxicity occurs is a desirable approach in getting at the heart of the issue. Effective educational programs may be in the form of fliers, enhanced labeling on pesticide products as well as the initiation of collection programs organized to collect unused pesticides after the phase-out is completed. Additionally, we recommend the development alternative strategies such as Integrated Pest Management (IPM) in urban, residential and agricultural settings.

Our analysis would be enhanced if there were better information available on measuring the effectiveness of our targeted BMPs at reducing pesticide-related aquatic toxicity. It follows that an important area of future research should focus on the effectiveness of BMPs, such as infiltration basins, grass swales or educational programs at reducing pesticide concentrations. Proposals for this research could be addressed through the development of pilot projects for determining specific loads per landuse within the watershed and use this quantity to measure the effectiveness of storm water treatment practices. This would have enabled us in the current study to identify specific neighborhoods within Newport Bay that have high loading rates, which could then be used for the highest priority for new and innovative management practices and make site specific recommendations with respect to construction costs.

In addition, we recommend that storm water runoff be sampled and monitored more thoroughly for continued presence of OP pesticides as well as pyrethroids and other classes of pesticides that are likely replacements after the phase-out of Diazinon and Chlorpyrifos is complete. With appropriate physicochemical pesticide data, the models we implemented (BASINS, WARMF and RivRisk) can be easily adapted to simulate the role of future pesticides as sources of aquatic toxicity. The fate and transport and toxicity characteristics of other organic pollutants can be entered into the models and simulated before they become water quality issues.

We believe the final results provide a valuable contribution to stakeholder groups interested in restoring and enhancing the beneficial uses in Newport Bay. Given the uncertainty surrounding the effectiveness of the pesticide phase-out and projected future concentrations, our project provides a means to make practical management decisions based on the best available scientific data.

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APPENDIX A: SUPPLEMENTARY INFORMATION FOR THE BASINS MODEL

A.1 Calibration Parameters

Table A-1. Hydrologic parameters.

Parameter	Peter's Canyon	Upper San Diego	Intermediate San Diego
PWAT-PARM2			
FOREST	0	0	0
LZSN	3	8	8
INFILT	0.01	0.05	0.05
LSUR	300	300	300
SLSUR	0.14	0.035	0.035
KVARY	1	1	1
AGWRC	0.92	0.92	0.92
PWAT-PARM3			
PETMAX	40	40	40
PETMIN	35	35	35
INFEXP	2	2	2
INFILD	2	2	2
DEEPFR	0.1	0.2	0.2
BASETP	0	0.05	0.05
AGWETP	0	0	0
PWAT-PARM4			
CEPSC	0.1	0.1	0.1
UZSN	0.1	0.4	0.4
NSUR	0.15	0.2	0.2
INTFW	2.5	1	1
IRC	0.8	0.5	0.5
LZETP	0.2	0.2	0.2
predicted max	1597	1638	5192
observed max	1570	1740	5125

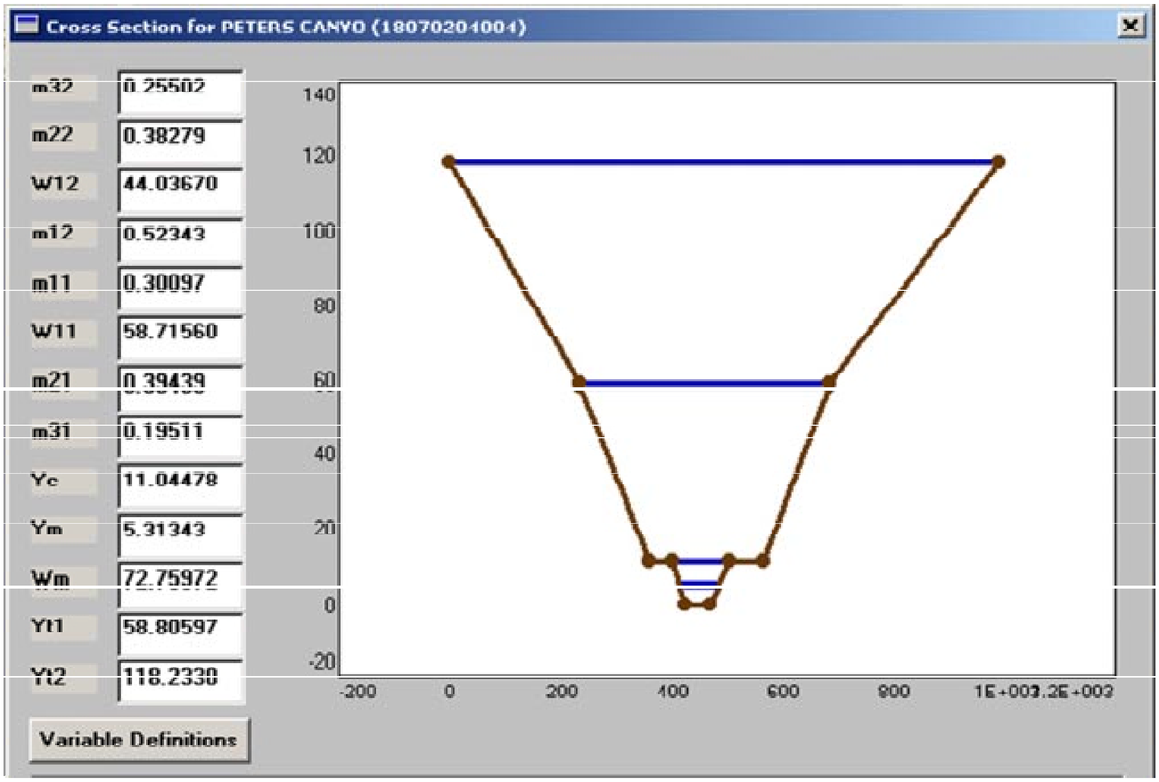

Figure A-1. Channel Geometry for Peter's Canyon Subcatchment.

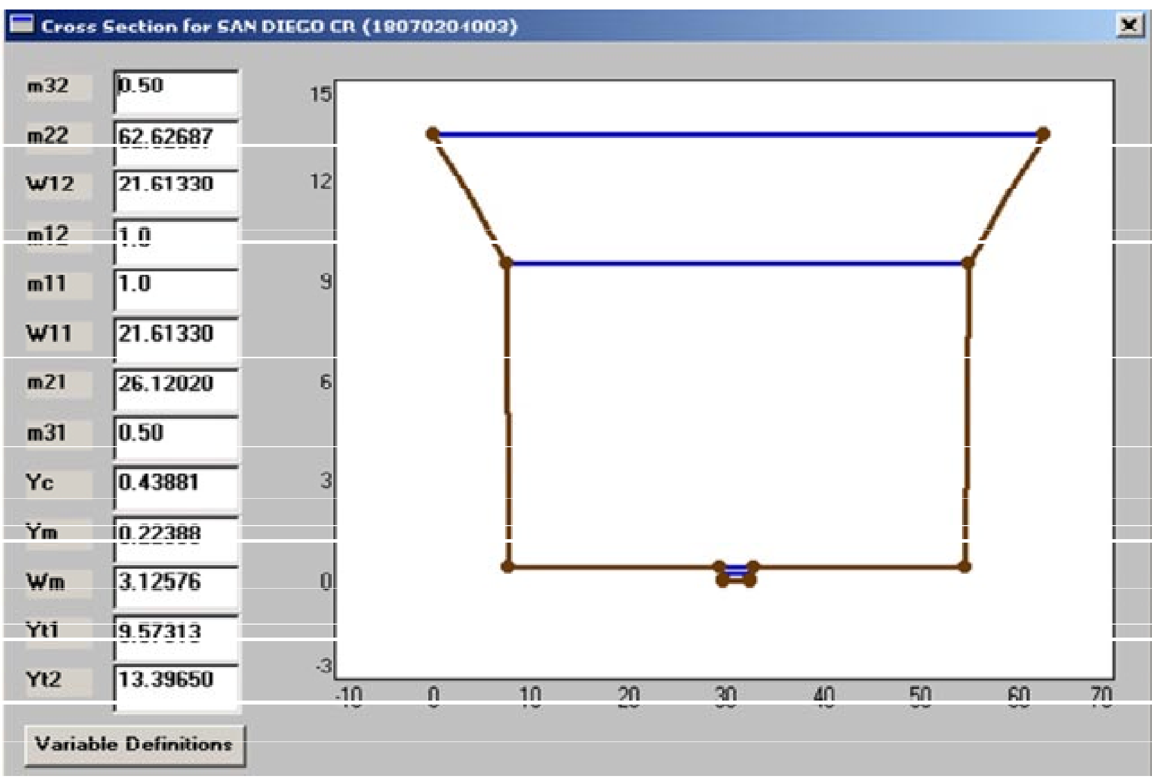

Figure A-2. Channel Geometry for Upper San Diego Subcatchment.

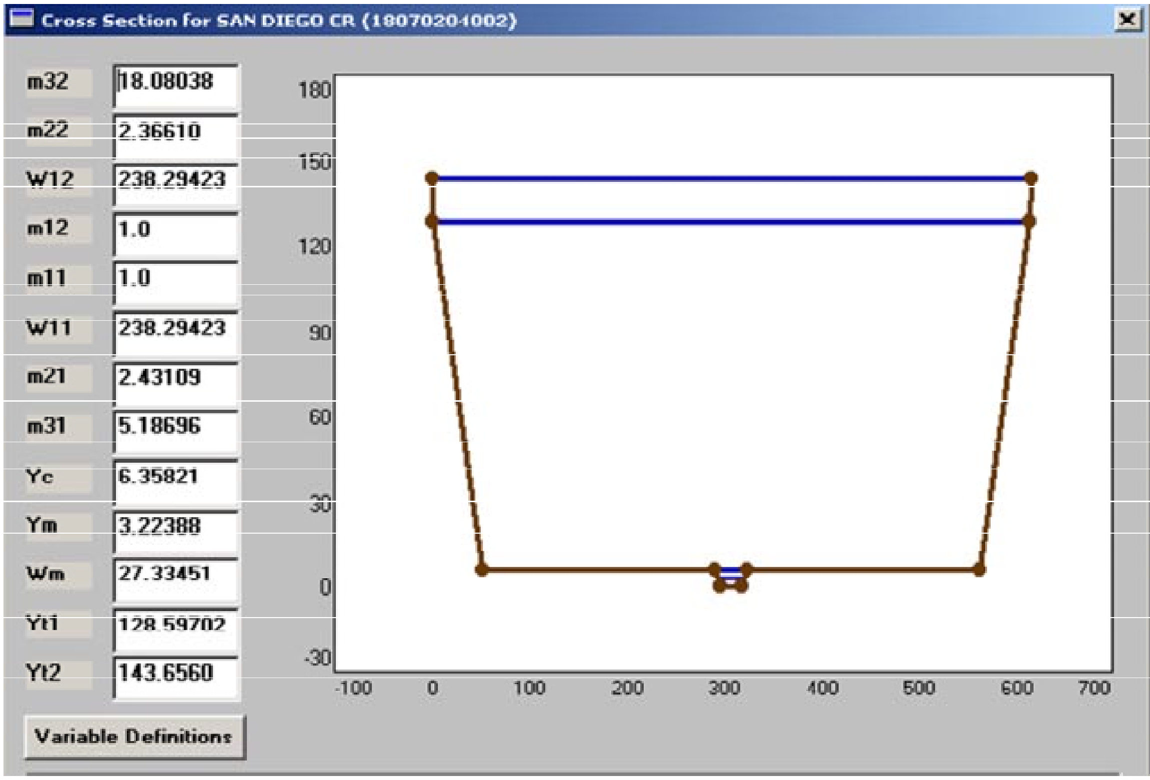

Figure A-3. Channel Geometry for Intermediate San Diego Subcatchment.

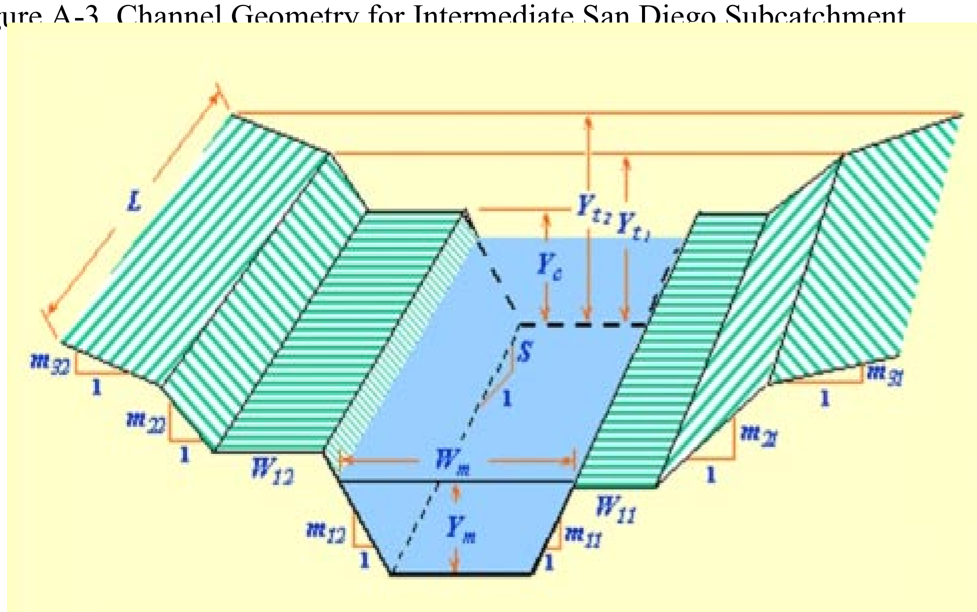

Figure A-4. Variable Definitions for BASINS Channel Geometry.
Source: US EPA (1998).

Table A-2. Baseline Pesticide Application Rates.

Chlorpyrifos	Application Rate **kg/acre/day**
Urban and Built-up	0.000877
Agriculture	7.52E-05
Open Space and Recreation	0.000105
Diazinon	
Urban and Built-up	0.000395
Agriculture	0.0000291
Open Space and Recreation	0

Table A-3. General Decay Parameters.

Parameter	**Chlorpyrifos**	**Diazinon**
GQ-GDECAY		
FSTDEC	0.0092	0.0039
THFST	1.08	1.08
GQ-SEDDECAY		
KSUSP	0.0058	0.025
THSUSP	1.08	1.08
KBED	0.0058	0.025
THBED	1.08	1.08

Table A-4. Water Quality Parameters.

Parameter	Chlorpyrifos Urban	Agriculture	Open/Rec	Diazinon Urban	Agriculture	Open/Rec
PQUAL-QUAL-INPUT						
SQO	10	10	10	0.05	0.05	0.05
POTFW	0	0	0	0	0	0
POTFS	0	0	0	0	0	0
SQOLIM	500	500	500	250	250	250
WSQOP	1.64	1.64	1.64	1.64	1.64	1.64
IOQC	0	0	0	0	0	0
AOQC	0	0	0	0	0	0
IQUAL-QUAL-INPUT						
SQO	10	n/a	n/a	0.05	n/a	n/a
POTFW	0	n/a	n/a	0	n/a	n/a
POTFS	0	n/a	n/a	0	n/a	n/a
SQOLIM	500	n/a	n/a	250	n/a	n/a
WSQOP	1.64	n/a	n/a	1.64	n/a	n/a
IOQC	0	n/a	n/a	0	n/a	n/a
AOQC	0	n/a	n/a	0	n/a	n/a

APPENDIX B: SUPPLEMENTARY INFORMATION FOR THE WARMF MODEL

Table B-1. Maximum and median concentrations of diazinon and chlorpyrifos simulated by WARMF for lower San Diego Creek. Table B-1 also lists the number of days each scenario exceeded CCC and CMC limits each rain session (November 11 – April 30).

Base - 1999 Application rates								
	Diazinon				Chlorpyrifos			
	Max Conc. (mg/L)	**Median Conc. (mg/L)**	**CCC**	**CMC**	**Max Conc. (mg/L)**	**Median Conc. (mg/L)**	**CCC**	**CMC**
1991	0.06	0.000162	113	109	0.12	0.000498	148	147
1992	0.05	0.000140	137	125	0.10	0.000557	160	158
1993	0.04	0.000262	127	122	0.08	0.000854	160	160
1994	0.03	0.000474	137	131	0.07	0.001378	168	167
Projected Phaseout								
	Diazinon				Chlorpyrifos			
	Max Conc. (mg/L)	**Median Conc. (mg/L)**	**CCC**	**CMC**	**Max Conc. (mg/L)**	**Median Conc. (mg/L)**	**CCC**	**CMC**
1991	0.02	0.000045	104	99	0.06	0.000263	146	133
1992	0.02	0.000040	107	101	0.05	0.000290	157	155
1993	0.01	0.000079	112	108	0.04	0.000433	159	158
1994	0.01	0.000135	120	116	0.03	0.000715	158	150
Additional 25% Phaseout - Urban								
	Diazinon				Chlorpyrifos			
	Max Conc. (mg/L)	**Median Conc. (mg/L)**	**CCC**	**CMC**	**Max Conc. (mg/L)**	**Median Conc. (mg/L)**	**CCC**	**CMC**
1991	0.01	0.000037	100	96	0.05	0.000211	145	123
1992	0.01	0.000033	103	98	0.04	0.000230	156	148
1993	0.01	0.000063	111	106	0.03	0.000341	159	145
1994	0.01	0.000108	120	115	0.03	0.000563	153	150
Additional 50% Phaseout - Urban								
	Diazinon				Chlorpyrifos			
	Max Conc. (mg/L)	**Median Conc. (mg/L)**	**CCC**	**CMC**	**Max Conc. (mg/L)**	**Median Conc. (mg/L)**	**CCC**	**CMC**
1991	0.01	0.000029	98	94	0.03	0.000159	143	113
1992	0.01	0.000026	101	94	0.03	0.000169	155	132
1993	0.01	0.000049	107	103	0.03	0.000251	158	125
1994	0.01	0.000081	116	111	0.02	0.000406	151	143
Additional 75% Phaseout - Urban								

	Diazinon				Chlorpyrifos			
	Max Conc. (mg/L)	**Median Conc. (mg/L)**	**CCC**	**CMC**	**Max Conc. (mg/L)**	**Median Conc. (mg/L)**	**CCC**	**CMC**
1991	0.01	0.000019	93	89	0.02	0.000102	122	97
1992	0.01	0.000018	94	89	0.02	0.000112	146	115
1993	0.01	0.000033	103	98	0.02	0.000162	145	113
1994	0.00	0.000055	111	106	0.01	0.000253	148	120
Street Sweeping Frequency - 50 Efficiency - 50								
	Diazinon				Chlorpyrifos			
	Max Conc. (mg/L)	**Median Conc. (mg/L)**	**CCC**	**CMC**	**Max Conc. (mg/L)**	**Median Conc. (mg/L)**	**CCC**	**CMC**
1991	0.0137	0.000041	100	97	0.05	0.000213	145	132
1992	0.0145	0.000038	106	101	0.05	0.000266	157	155
1993	0.0127	0.000069	111	108	0.04	0.000412	159	158
1994	0.0089	0.000118	118	116	0.03	0.000639	158	150
Street Sweeping Frequency - 150 Efficiency - 50								
	Diazinon				Chlorpyrifos			
	Max Conc. (mg/L)	**Median Conc. (mg/L)**	**CCC**	**CMC**	**Max Conc. (mg/L)**	**Median Conc. (mg/L)**	**CCC**	**CMC**
1991	0.0154	0.000044	102	99	0.05	0.000251	145	133
1992	0.0154	0.000039	106	101	0.05	0.000280	157	155
1993	0.0132	0.000074	112	108	0.04	0.000426	159	158
1994	0.0094	0.000131	120	116	0.03	0.000679	158	150

Table B-2**.** Maximum and median concentrations of diazinon and chlorpyrifos simulated by WARMF for Peters Canyon Wash. Table B-2 also lists the number of days each scenario exceeded CCC and CMC limits each rain session (November 11 – April 30).

Base - 1999 Application rates								
	Diazinon				Chlorpyrifos			
	Max Conc. (mg/L)	**Median Conc. (mg/L)**	**CCC**	**CMC**	**Max Conc. (mg/L)**	**Median Conc. (mg/L)**	**CCC**	**CMC**
1991	0.025	0.000009	74	66	0.049	0.000202	128	127
1992	0.026	0.000016	98	79	0.060	0.000287	137	137
1993	0.007	0.000016	96	83	0.013	0.000259	149	149
1994	0.015	0.000013	88	74	0.031	0.000239	151	150
Projected Phaseout								
	Diazinon				Chlorpyrifos			
	Max Conc. (mg/L)	**Median Conc. (mg/L)**	**CCC**	**CMC**	**Max Conc. (mg/L)**	**Median Conc. (mg/L)**	**CCC**	**CMC**
1991	0.009	0.000004	48	42	0.027	0.000112	126	119

1992	0.008	0.000006	59	53	0.032	0.000157	137	136
1993	0.003	0.000007	71	65	0.007	0.000140	149	148
1994	0.006	0.000005	58	53	0.017	0.000133	144	119
Additional 25% Phaseout - Urban								
	Diazinon				Chlorpyrifos			
	Max Conc. (mg/L)	**Median Conc. (mg/L)**	**CCC**	**CMC**	**Max Conc. (mg/L)**	**Median Conc. (mg/L)**	**CCC**	**CMC**
1991	0.008	0.000003	45	41	0.022	0.000096	126	104
1992	0.007	0.000006	57	51	0.025	0.000131	137	131
1993	0.002	0.000006	69	61	0.006	0.000117	149	135
1994	0.005	0.000005	54	51	0.014	0.000113	130	117
Additional 50% Phaseout - Urban								
	Diazinon				Chlorpyrifos			
	Max Conc. (mg/L)	**Median Conc. (mg/L)**	**CCC**	**CMC**	**Max Conc. (mg/L)**	**Median Conc. (mg/L)**	**CCC**	**CMC**
1991	0.007	0.000003	42	39	0.017	0.000079	126	87
1992	0.005	0.000005	52	50	0.019	0.000105	136	113
1993	0.002	0.000005	66	57	0.005	0.000096	148	107
1994	0.004	0.000004	53	50	0.011	0.000093	121	106
Additional 75% Phaseout - Urban								
	Diazinon				Chlorpyrifos			
	Max Conc. (mg/L)	**Median Conc. (mg/L)**	**CCC**	**CMC**	**Max Conc. (mg/L)**	**Median Conc. (mg/L)**	**CCC**	**CMC**
1991	0.005	0.000003	40	35	0.012	0.000061	107	58
1992	0.004	0.000004	51	46	0.013	0.000078	131	88
1993	0.002	0.000005	61	49	0.004	0.000073	137	87
1994	0.004	0.000004	51	50	0.008	0.000073	116	76
Street Sweeping Frequency - 50 Efficiency - 50								
	Diazinon				Chlorpyrifos			
	Max Conc. (mg/L)	**Median Conc. (mg/L)**	**CCC**	**CMC**	**Max Conc. (mg/L)**	**Median Conc. (mg/L)**	**CCC**	**CMC**
1991	0.008	0.000004	47	42	0.022	0.000112	126	119
1992	0.007	0.000006	59	52	0.027	0.000157	137	136
1993	0.003	0.000007	71	65	0.007	0.000140	149	148
1994	0.005	0.000005	56	52	0.015	0.000133	144	119
Street Sweeping Frequency - 150 Efficiency - 50								
	Diazinon				Chlorpyrifos			
	Max Conc. (mg/L)	**Median Conc. (mg/L)**	**CCC**	**CMC**	**Max Conc. (mg/L)**	**Median Conc. (mg/L)**	**CCC**	**CMC**
1991	0.008	0.000004	48	42	0.025	0.000112	126	119
1992	0.008	0.000006	59	52	0.030	0.000157	137	136

1993	0.003	0.000007	71	65	0.007	0.000140	149	148
1994	0.006	0.000005	58	53	0.016	0.000133	144	119

Table B-3. Maximum and median concentrations of diazinon and chlorpyrifos simulated by WARMF for upper San Diego Creek. Table B-3 also lists the number of days each scenario exceeded CCC and CMC limits each rain session (November 11 – April 30).

Base - 1999 Application rates								
	Diazinon				Chlorpyrifos			
	Max Conc. (mg/L)	**Median Conc. (mg/L)**	**CCC**	**CMC**	**Max Conc. (mg/L)**	**Median Conc. (mg/L)**	**CCC**	**CMC**
1991	0.17	0.000759	135	134	0.38	0.002220	134	132
1992	0.21	0.000376	149	140	0.37	0.001852	182	180
1993	0.12	0.000450	164	154	0.23	0.001660	181	166
1994	0.29	0.000718	149	143	0.53	0.002259	173	161
Projected Phaseout								
	Diazinon				Chlorpyrifos			
	Max Conc. (mg/L)	**Median Conc. (mg/L)**	**CCC**	**CMC**	**Max Conc. (mg/L)**	**Median Conc. (mg/L)**	**CCC**	**CMC**
1991	0.05	0.000212	118	113	0.20	0.001148	131	129
1992	0.06	0.000109	126	121	0.19	0.000970	166	163
1993	0.04	0.000133	132	125	0.12	0.000870	165	163
1994	0.07	0.000217	125	119	0.27	0.001172	159	157
Additional 25% Phaseout - Urban								
	Diazinon				Chlorpyrifos			
	Max Conc. (mg/L)	**Median Conc. (mg/L)**	**CCC**	**CMC**	**Max Conc. (mg/L)**	**Median Conc. (mg/L)**	**CCC**	**CMC**
1991	0.04	0.000163	115	111	0.15	0.000888	130	128
1992	0.05	0.000088	124	117	0.15	0.000767	164	157
1993	0.03	0.000106	129	122	0.10	0.000693	164	162
1994	0.05	0.000178	122	116	0.20	0.000922	158	156
Additional 50% Phaseout - Urban								
	Diazinon				Chlorpyrifos			
	Max Conc. (mg/L)	**Median Conc. (mg/L)**	**CCC**	**CMC**	**Max Conc. (mg/L)**	**Median Conc. (mg/L)**	**CCC**	**CMC**
1991	0.03	0.000118	112	110	0.11	0.000644	129	127
1992	0.04	0.000067	119	112	0.11	0.000568	163	146
1993	0.03	0.000083	124	116	0.07	0.000516	163	161
1994	0.04	0.000139	118	113	0.14	0.000672	157	155
Additional 75% Phaseout - Urban								
	Diazinon				Chlorpyrifos			

	Max Conc. (mg/L)	Median Conc. (mg/L)	CCC	CMC	Max Conc. (mg/L)	Median Conc. (mg/L)	CCC	CMC
1991	0.02	0.000074	109	107	0.06	0.000401	127	125
1992	0.03	0.000046	115	105	0.07	0.000359	147	142
1993	0.02	0.000059	117	109	0.05	0.000338	161	160
1994	0.02	0.000099	113	110	0.07	0.000425	155	153
Street Sweeping Frequency - 50 Efficiency - 50								
	Diazinon				Chlorpyrifos			
	Max Conc. (mg/L)	**Median Conc. (mg/L)**	**CCC**	**CMC**	**Max Conc. (mg/L)**	**Median Conc. (mg/L)**	**CCC**	**CMC**
1991	0.04	0.000149	117	112	0.15	0.001057	130	128
1992	0.06	0.000109	125	121	0.17	0.000958	166	163
1993	0.04	0.000131	132	125	0.12	0.000870	165	163
1994	0.05	0.000215	124	119	0.19	0.001164	159	157
Street Sweeping Frequency - 150 Efficiency -50								
	Diazinon				Chlorpyrifos			
	Max Conc. (mg/L)	**Median Conc. (mg/L)**	**CCC**	**CMC**	**Max Conc. (mg/L)**	**Median Conc. (mg/L)**	**CCC**	**CMC**
1991	0.05	0.000188	117	113	0.17	0.001095	130	129
1992	0.06	0.000109	126	121	0.18	0.000969	166	163
1993	0.04	0.000132	132	125	0.12	0.000870	165	163
1994	0.06	0.000217	125	119	0.24	0.001169	159	157

APPENDIX C: SUPPLMENTARY COST ANALYSIS TABLES

C.1 Introduction

Various management practices are evaluated with respect to overall costs. Cost equations compiled and published by the EPA are used for estimating base construction costs of structural BMPs. These include Retention Basins, Detention Basins, Constructed Wetlands, Infiltration Practices, Filters and Biofilters (swales and filter strips). Using these capital construction costs one could then calculate the contingency and permitting costs and the annual maintenance costs using the tables described below.

C.2 Design, Contingency and Permitting

Design, Contingency and Permitting costs are included in all our management alternatives. A published range of 25% to 32% includes design, contingencies and permitting fees (EPA, 1999). For the purposes of this study O&M costs were 30% of the construction costs.

C.3 Operation and Maintenance Costs

The EPA has summarized studies (Brown & Schuler, 1997) that operation and maintenance (O&M) costs can be estimated as a percentage of construction costs. It is recognized by this study that these costs can vary widely across BMPs and location and are used only as general costs and are not representative of the actual costs that might be incurred by responsible agencies in the Newport Bay Watershed. Table C-1 taken from the EPA Storm water Guidelines Document summarizes maintenance costs calculated from various sources.

Table C-1. Annual Maintenance Costs.

BMP	Annual Maintenance Cost (% of Construction Costs)	% Assumed in This Study	Source(s)
Retention Basins and Constructed Wetlands	3-6%	4.5%	Wiegand et al, 1986; Schueler, 1987; SWRPC, 1991
Detention Basins[1]	<1%	1%	Livingston et al, 1997
Constructed Wetlands[1]	2%	2%	Livingston et al, 1997; Brown and Schuler, 1997
Infiltration Trench	5-20%	12.5%	Schueler, 1987; SWRPC, 1991
Infiltration Basin[1]	1-3%	2%	Livingston et al, 1997; SWRPC, 1991
	5-10%	7.5%	Wiegand et al, 1986; Schueler, 1987; SWRPC, 1991
Swales	11-13%	12%	Livingston et al, 1997; Brown and Schueler, 1997
Filter Strips	$320/acre (Maintained)	$320/acre (maintained)	SWRPC, 1991

Livingston et al (1997) reported maintenance costs from the maintenance budgets of several cities, and percentages were derived from costs in other studies.

Source: EPA (1999)

These are general costs used as a way to compare various management practices for the most cost effective solution to the water quality problem associated with Newport Bay.

C.4 Non-Structural Costs: Public Education and Outreach

Eliminating the source of pollution is the most effective way to increase the health of any ecosystem. Public education and outreach programs rely on this notion to improve water quality in disturbed systems. The City of Seattle actively runs education programs directed at watershed health and pollution prevention. Although the effectiveness of this program is not yet known cost figures are used to establish general guidelines for comparison with other BMPs.

Public education programs can be implemented in a variety of ways. Watershed awareness programs could target school-aged children, residents in neighborhood groups, or perhaps staff at nurseries, golf courses, or local hardware stores. In general, the goal of public education is to make the public more aware of the potential or existing concern, thereby gaining their support for the program, and fostering a change in behavior. Examples of education costs are varied and dependent on the target population and selected method. The Washington Department of Energy (DOE) institutes an aggressive education program, incorporating classroom and field involvement programs in Seattle. The City of Seattle has a population of approximately 537,000 (U.S. Census web page). The 1997 budget for public education costs is presented below in Table C-2. The costs are given as examples, and do not necessarily reflect typical effort or expenditures (EPA, 1999).

Table C-2. Public Education Costs in Seattle, Washington.

Item	Description	1997 Budget
Supplies for volunteers	Covers supplies for the Stewardship Through Environmental Partnership Program	$17,500
Communications	Communications strategy highlighting a newly formed program within the city	$18,000
Environmental Education	Transportation costs from schools to field visits (105 schools with four trips each)	$46,500
Education Services / Field Trips	Fees for students visits to various sites	$55,000
Teacher Training	Covers the cost of training classroom teachers for the environmental education program	$3,400
Equipment	Equipment for classroom education, including displays, handouts, etc.	$38,800
Water Interpretive Specialist: Staff	Staff to provide public information at two creeks	$79,300
Water Interpretive Specialist: Equipment	Materials and equipment to support interpretive specialist program	$12,100

Source: U.S. EPA (1999).