Chapter 5 Water Quality and Watershed Implications

5.1 Overview

This chapter describes the water quality evaluations conducted as a part of the R3 Study and outlines the potential implications of these evaluations on the San Diego Creek Watershed. Specific information includes:

- A discussion of two approaches to the evaluation of water quality
- A summary of the study methods relating to water quality
- Development of "before and after" assessments of water quality to evaluate the effectiveness of ET technology and public education
- Detailed discussions of the evaluation approaches and findings based on these approaches
- A discussion of the implications of the findings for water quality in the San Diego Creek Watershed, focusing on TMDL constituents

More detailed information is provided in Appendices E1 and E2.

5.2 Introduction

Two independent reviews of water quality measurements were conducted as a part of this study. The initial review was conducted by SCCWRP as a part of its participation in the R3 Study and is included in its entirety as Appendix E1. This review used parametric statistical techniques (t-test; ANOVA), which provide a good descriptive review of the study data, but are generally considered to have less statistical power for detecting differences in data than other statistical tests. In general, because of the variability of the data and limitations in sample quantities, this review concluded that there was virtually no difference in either the concentration or "flux" (concentration times flow) of pollutants over time or between study treatments.

A subsequent statistical overview by Geosyntec Consultants was commissioned by IRWD to review alternative and possibly more "robust" data analysis techniques that might identify differences in study data not uncovered during the initial review. This work, which included the review of only a portion of the data set, focused on additional descriptive techniques (time series plots; box plots; probability distributions) and the use of non-parametric statistical techniques (rank-sum test; K-W). For some of the parameters reviewed, these techniques suggest that differences in measured water quality did occur across time and between study treatments. The entire Geosyntec report is provided in Appendix E2.

As noted above, both of the completed statistical reviews of the study data are included in the Appendices of this report. The remainder of this chapter of the report discusses the key findings of each review.

5.3 SCCWRP Water Quality Review

This section describes the SCCWRP evaluation approach, sampling and laboratory analysis, data analysis, and interpretations of the results. Watershed implications are also discussed.

5.3.1 Evaluation Approach

A before-after, control-impact (BACI) design was used to evaluate the effectiveness of both the sprinkler technology and public education. Each neighborhood was sampled every other week between December 2000 and June 2001. In June 2001, homes in one of the neighborhoods were outfitted with the ET controllers. Since homeowners with the retrofitted ET controllers were simultaneously being educated, a well-defined public education campaign was also begun with these homeowners. To ascertain the difference between education and ET technology, homeowners in a second neighborhood were targeted with an identical public education campaign, but without effect of the ET retrofit technology. There was no education or technology intervention in the remaining three neighborhoods, which served as control neighborhoods to document the effect of no treatment. Sampling at the five neighborhoods continued every other week from June 2001 to June 2002.

5.3.2 Sampling and Laboratory Analysis

Each neighborhood was hydrologically self-contained and drained to a single underground pipe. At each of these five locations, samples were collected for flow and water quality. Stage (water depth) and velocity were recorded at 5-minute intervals using an ultrasonic height sensor mounted at the pipe invert and a velocity sensor mounted on the floor of the pipe. Flow was calculated as the product of velocity and wetted cross-sectional area as defined by the stage and pipe circumference. Despite the relatively continuous measurement of flow, many of the flow measurements were excluded due to faulty readings. Synoptic flow and water quality measurements were only available for two sites over the course of the entire study (i.e. before and after intervention), including the ET controller + education and education only sites. Flow measurements at the time of water quality sampling for the three control sites were considered faulty and discarded.

Grab samples for water quality were collected just downstream of the flow sensors in the early morning using peristaltic pumps and pre-cleaned Teflon tubing. Samples were placed in individual pre-cleaned jars, placed on ice, and transported to the laboratory within one hour. Each sample was analyzed for 19 target analytes, five microbiological parameters, and four toxicity endpoints (Table 5-1). Target analytes included trace metals, nutrients, and organophosphorus (OP) pesticides. Microbiological parameters included fecal indicator bacteria and bacteriophage. Toxicity was evaluated using two marine species, the purple sea urchin *Strongylocentrotus purpuratus* and the mysid *Americamysis bahia*. All of the laboratory methodologies followed standard protocols developed by the USEPA or Standard Methods.

5.3.3 Data Analysis

Data analysis consisted of five steps: 1) comparison of water quality among the five neighborhoods prior to intervention; 2) comparison of water quality concentrations over time by neighborhood; 3) comparison of water quality concentrations before and after intervention by treatment type; 4) comparison of pollutant flux before and after intervention by treatment type; and 5) correlation of toxicity measures with potential toxicants in dry weather runoff.

Comparison of water quality concentrations among the five neighborhoods prior to intervention was conducted to assess if there were inherent differences among treatment sites for each

Table 5-1
Reporting Level and Method for Target Parameters

	Reporting Level	Method
Metals (ug/L)		
Antimony	0.2	EPA 200.8
Arsenic	1.5	EPA 200.8
Barium	0.2	EPA 200.8
Cadmium	0.2	EPA 200.8
Chromium	0.3	EPA 200.8
Cobalt	0.1	EPA 200.8
Copper	1.5	EPA 200.8
Lead	0.3	EPA 200.8
Nickel	0.2	EPA 200.8
Selenium	5.0	EPA 200.8
Silver	0.4	EPA 200.8
Zinc	5.0	EPA 200.8
Nutrients (mg/L)		
Ammonia as N	5.0	EPA 350.1
Nitrate/Nitrite as N	5.0	EPA 353.2
Total Kjeldahl Nitrogen	10.0	EPA 351.2
Ortho-Phosphate as P	0.5	EPA 365.1
Total Phosphorus	1.0	EPA 365.4
OP Pesticides (ng/L)		
Chlorpyrifos	20.0	IonTrap GCMS
Diazinon	20.0	IonTrap GCMS
Microbiology		
Enterococcus (MPN/100 mL)	2	SM9230B
Fecal Coliform (MPN/100 mL)	2	SM9221B
Total Coliform (MPN/100 mL)	2	SM9221B
MS2 Phage (PFU/100 mL)	2	EPA 1602
Somatic Phage (PFU/100 mL)	2	EPA 1602
Toxicity (% effluent)		
Sea Urching Fertilization EC50	NA	EPA 1995
Sea Urching Fertilization NOEC	NA	EPA 1995
Mysid EC50	NA	EPA 1993
Mysid NOEC	NA	EPA 1993

Note: ug/L = micrograms per liter; MPN/100 mL=most probable number per 100 milliliters; PFU/100mL=plaque forming units per 100 milliliters; mg/L=milligrams per liter; ng/L=nanograms per liter.

constituent. This analysis was conducted using ANOVA using Tukey's post hoc test for identifying the significantly different neighborhoods. All data was tested for normality and homogeneous variance prior to testing. Only the microbiological data was determined to be non-normally distributed, so these results were log transformed prior to data analysis.

Comparison of water quality concentrations over time was accomplished by creating temporal plots of monthly mean concentration. Comparisons of water quality concentration before and after intervention by treatment type were accomplished using a standard t-test of the mean concentration before versus mean concentration after intervention. The mean concentrations for ET controller + education, education only, and ET controller + education – education only for each sampling event were normalized by the grand mean of the control sites for the same sampling event.

Pollutant flux estimates were calculated by the product of the concentration and volume at the time of sampling and then normalized to the area of the sampled neighborhood. Pollutant flux before and after treatment was compared somewhat differently since the lack of flow data at the control sites did not permit an estimate of flux for these neighborhoods. Mean pollutant flux before and after intervention was compared using standard t-tests at the ET controller + education and education only neighborhoods without normalization to control values.

Correlation of toxicity with toxicant concentrations was accomplished using a Pearson product moment correlation. These correlations are inferential only and do not presume resulting correlations automatically identify the responsible toxicants. In order to help identify potential causative toxic agents, concentrations of the correlated constituents were compared to concentrations known to induce toxicity in the respective test organisms.

5.3.4 Evaluation Results

There were significant differences in water quality among sites prior to intervention (Appendix E1, Table WQ3). Site 1004, the control site, had the greatest mean concentrations for 15 of the 24 constituents evaluated prior to the ET controller intervention. In particular, all of the mean nutrient concentrations were greater at Site 1004 than the other sites. On the other hand, Sites 1001 and 1002 generally had the lowest average concentrations prior to the ET controller intervention. Cumulatively, these sites had the lowest mean concentrations for 17 of the 24 constituents evaluated. Site 1002 also had the least toxicity, on average, of all five sites. Finally, Site 1003 had an intermediate status. Mean concentrations of enterococcus and fecal coliforms at this site were greater than any other site (fecal coliforms significantly greater than Sites 1001 and 1002), but the mean concentrations of five trace metals (chromium, copper, cobalt, nickel, selenium) were lowest at this site.

Water quality concentrations and toxicity were highly variable over time during the study period. Temporal plots of concentrations and toxicity for each site demonstrated that there was no seasonal trend and no overall trend with time. There were, however, occasional spikes in concentrations for many constituents that appeared to fall into one of two categories. The first category was recurring spikes in concentration that were unpredictable in timing and location. The second category of concentration spike was single or infrequent peaks. Occasionally these spikes would occur across multiple sites, without commensurate changes in concentration at the treatment sites (1001 or 1005). More often, infrequent spikes were isolated to a single site. For example, concentrations of chlorpyrifos climbed to over 10,000 ng/L in July 2001, but averaged near 50 ng/L the remainder of the year at site 1005. Similarly, concentrations of ammonia and total phosphorus spiked 10 and 25-fold prior to June 2001 at the control site (1004) with less variability and overall lower concentrations the remainder of the study.

There were few significant differences that resulted from the intervention of education, ET controller + education, or ET controller + education – education only, relative to control sites (Table 5-2). Only six of the 24 constituents evaluated showed a significant difference between pre and post-intervention concentrations after normalizing to mean control values. These significant differences were a net increase in concentrations of ammonia, nitrate/nitrite, total phosphorus, chlorpyrifos, diazinon, and fecal coliforms. These statistical analyses were the result of one of two circumstances. In the first circumstance, there were individual large spikes in concentration at treatment sites, but not at control sites following intervention. Therefore, the net difference in concentrations between controls and treatments increased following the intervention. In these cases, removal of the outlier samples resulted in no significant difference among treatment effects relative to controls before intervention compared to after intervention. In the second circumstance, there were large spikes in concentrations at control site(s) prior to the intervention that later subsided, while treatment site concentrations and variability remained steady. Therefore, the difference between treatments and controls changed following interventions, although it was not a result of the education or technology.

Although there were no significant differences in pollutant flux as a result of the intervention, significant differences were noted in pollutant flux among sites prior to intervention. Site 1001, the ET controller + education site, had the greatest mean flux for 22 of the 24 constituents evaluated prior to the ET controller intervention. The mean flux for 20 of these 22 constituents was significantly greater at Site 1001 than the mean flux at Site 1005 (t-test, $p<0.05$). Site 1005 had greater mean fluxes only for MS2 phage and ammonia. The differences among the fluxes prior to (and after) intervention were the result of two factors: greater flow and, at times, greater concentrations at Site 1001 compared to Site 1005. Mean dry weather flow at the time of water quality sampling was nearly three times greater at Site 1001 than Site 1005.

Toxicity was inconsistently found at all five of the sampling sites, and there was no change in toxicity as a result of the intervention (Table 5-3). The two species tested did not respond similarly either among sites, among treatments, or over time. Correlation of toxicity with constituent concentrations yielded few significant relationships for either species (Table 5-3). Mysid toxicity was correlated with diazinon and several trace metals, but the strongest relationship was with diazinon concentration. Moreover, the concentrations of diazinon were well above the levels known to cause adverse effects in mysid, while trace metals were not. Sea urchin fertilization toxicity was only correlated with concentrations of zinc. The concentrations of zinc were well above the level known to induce adverse effects in this species.

Table 5-2
Significance of ANOVA Results for the Effect of ET Controller + Education, Education Alone, and the Difference Between ET Controller + Education and Education Alone Relative to Control Concentrations. (No data indicates p > 0.05)

	Effect of ET Controller + Education	Effect of Education Alone	Difference Between ET Controller + Education and Education Alone
Metals			
Antimony			
Arsenic			
Barium			
Cadmium			
Chromium			
Cobalt			
Copper			
Lead			
Nickel			
Selenium			
Silver			
Zinc			
Nutrients			
Ammonia as N	0.03	0.02	
Nitrate/Nitrite as N	0.02		
Total Kjeldahl Nitrogen			
Ortho-Phosphate as P			
Total Phosphorus		0.03	
OP Pesticides			
Chlorpyrifos	<0.01	<0.01	<0.01
Diazinon		<0.01	
Microbiology			
Enterococcus			
Fecal Coliform	0.04		
Total Coliform			
MS2 Phage			
Somatic Phage			
Toxicity			
Fertilization EC50			
Fertilization NOEC			
Mysid EC50			
Mysid NOEC			

Table 5-3
Correlation Coefficients (and p value) of Constituent Concentrations with Toxicity Endpoints (No Observed Effect Concentration, NOEC and Median Effect Concentration, EC50) in Dry Weather Discharges from Residential Neighborhoods in Orange County, CA. (No data indicates p > 0.05)

	Sea Urchin Fertilization NOEC	Mysid Survival NOEC	Sea Urchin Fertilization EC50	Mysid Survival EC50
Antimony		-0.273 (0.009)		
Arsenic		-0.3396 (0.001)		
Barium				
Cadmium				
Chromium		-0.244 (0.021)		-0.219 (0.044)
Cobalt		-0.330 (0.002)		-0.279 (0.010)
Copper				
Lead		-0.215 (0.042)		
Nickel				
Silver		-0.260 (0.013)		-0.229 (0.035)
Zinc	-0.277 (0.005)		-0.274 (0.006)	
Chlorpyrifos				
Diazinon		-0.426 (0.001)		-0.468 (0.001)
Ammonia				

5.3.5 Interpretation of Results

The evaluation was unable to find large, significant reductions in concentration or pollutant flux as a result of education and/or ET controller retrofit technology. This may indicate that the technology and/or education are inefficient for improvements in water quality. Equally as important, however, was the absence of meaningful increases in concentrations. Of the small number of concentrations that showed significant increases, most could be explained by highly variable spikes in concentrations reminiscent of isolated entries to the storm drain system, as opposed to ongoing chronic inputs or the effects of best management practices evaluated in this study.

If significant changes did occur, the evaluation design may not have detected these changes due to two factors. First, the variability in concentrations within and between sites is naturally high and the evaluation simply collected too few samples. After taking into account the variability and relative differences in mean concentrations, zinc was used as an example constituent to determine what sample sizes would be required to detect meaningful differences. Assuming that the sampling yielded the true mean and variance structure that actually existed at the five sites, power analysis indicated that a minimum sample size of no less than five-fold would have been required to detect the differences observed in zinc concentrations during this study.

The second factor that could have hindered the ability to detect meaningful differences in water quality is that the technology and education treatments were applied at the spatial scale of individual homes, while the evaluation design sampled at the neighborhood scale. This problem was exacerbated because only a fraction (approximately one-third) of the homes within the neighborhoods sampled had the technological or educational treatments. Therefore, the treatments were effectively diluted, decreasing the ability to detect differences in water quality.

5.3.6 Watershed Implications

It appears that residential dry weather flows measured in the R3 Study may contribute significant proportions of some constituents to overall watershed discharges. The study sites were located within the San Diego Creek watershed, the largest tributary to Newport Bay. The Orange County Public Facilities and Resources Department (OCPFRD) publishes monitoring data on San Diego Creek to provide environmental managers the information they need to properly manage the Bay (OCPFRD 2002). The dry weather monitoring data was compiled at the mouth of San Diego Creek from OCPFRD during 2001-2002 and compared the concentrations to our results from residential neighborhoods (Table 5-4). Mean concentrations of chlorpyrifos, diazinon, copper and zinc were much higher in upstream residential neighborhoods than concentrations measured at the mouth of San Diego Creek. These residential dry weather contributions were amplified by the fact that the San Diego Creek watershed is primarily composed of residential land uses. In contrast, concentrations of selenium, arsenic, and total phosphorus in the residential dry weather discharges were much lower than the cumulative dry weather discharges from San Diego Creek, indicating that residential areas may not be the primary source of these constituents.

Table 5-4
Comparison of Mean Concentrations (95% Confidence Intervals) in Residential Dry Weather Discharges from this Study Compared to Concentrations in Dry Weather Discharges from San Diego Creek at Campus Drive During 2001-2002. (Data from OCPFRD)

	San Diego Creek	Residential
Parameter	**Mean (95% CI)**	**Mean (95% CI)**
Nitrate	5.16 (0.72)	4.76 (1.96)
Phosphate	1.98 (0.07)	1.16 (0.20)
Diazinon	0.13 (0.07)	1.52 (0.52)
Chlorpyrifos	0.05 (0.01)	0.35 (0.44)
Copper	11.59 (2.83)	23.59 (5.65)
Arsenic	6.58 (0.40)	2.68 (0.26)
Selenium	21.22 (2.65)	2.46 (0.03)
Zinc	22.08 (2.75)	60.09 (8.26)

5.4 Geosyntec Water Quality Review

This section presents examples of alternative approaches to data analysis, data analysis methods, example results, and watershed implications.

5.4.1 Examples of Alternative Approaches to Data Analysis

These example analyses focus on TMDL constituents: nutrients (total nitrogen [TN] and total phosphorus [TP]), metals (copper, lead, zinc, cadmium), pesticides, and pathogens (fecal coliform). The analyses also focus on dry weather flows, as reduction of these flows was a major objective of the R3 Study.

5.4.2 Data Analysis Methods

Exploratory Data Analysis

Visual inspection of data and exploration of factors that could potentially influence data (e.g. seasonal trends, rain events)

1. Divide data into pre and post- intervention groups.
2. Construct time series plots to visually inspect data and visually examine for seasonal trends. Overlay storm event markers to identify any relation to rainfall volume or antecedent dry period (ADP).
3. Investigate normality or log normality of data sets. Select appropriate statistical tests.
4. Construct probability plots for pre-intervention and post-intervention periods.
5. Prepare quantile plots.
6. Prepare side-by-side box plots.
7. Calculate descriptive statistics

Hypothesis Testing

Test data for skewness, normality, and statistically significant differences. Skewness and normality tests are only needed if parametric approaches are conducted. Use of non-parametric approaches is recommended for consistency because normality will not be met in all cases. Nonetheless, examples are provided to show that several of the data sets do not come from a normal distribution.

1. Skewness hypothesis test for symmetry.
2. Shipiro-Wilkes normality test.
3. Mann-Whitney rank-sum test.
4. For the data sets that have greater than 50 percent censored data (i.e., data only known to be less than the detection limit), hypothesis tests for differences in proportions.

5.4.3 Example Results

The first step in the data analysis was to construct individual time-series plots for each site to identify seasonal periodicity, step-trends, and monotonic trends. Plotting each site individually reveals more information than plotting all sites together. Also, by overlaying storm events, the role of rainfall volumes and the ADP may be more apparent and may indicate whether additional analyses are warranted (e.g., correlating ADP with concentration). Figure 5-1 is an example time-series plot with storm event markers overlain for TP for Site 1001. As shown on the figure, the pre-intervention period had much more rainfall, which likely added to the variability in runoff concentrations and fluxes. However, it is apparent that the winter and spring concentrations appear to be lower and less variable during the post-intervention period. The irrigation controllers may have had an effect on the runoff concentrations by reducing the amount of irrigation during moister weather conditions (i.e., high soil moisture). A similar effect for TN is shown on Figure 5-2. Additional time-series plots are provided in Appendix E2.

Figure 5-1
Example Time-series Plot of Total Phosphorus with Storm Event Markers.

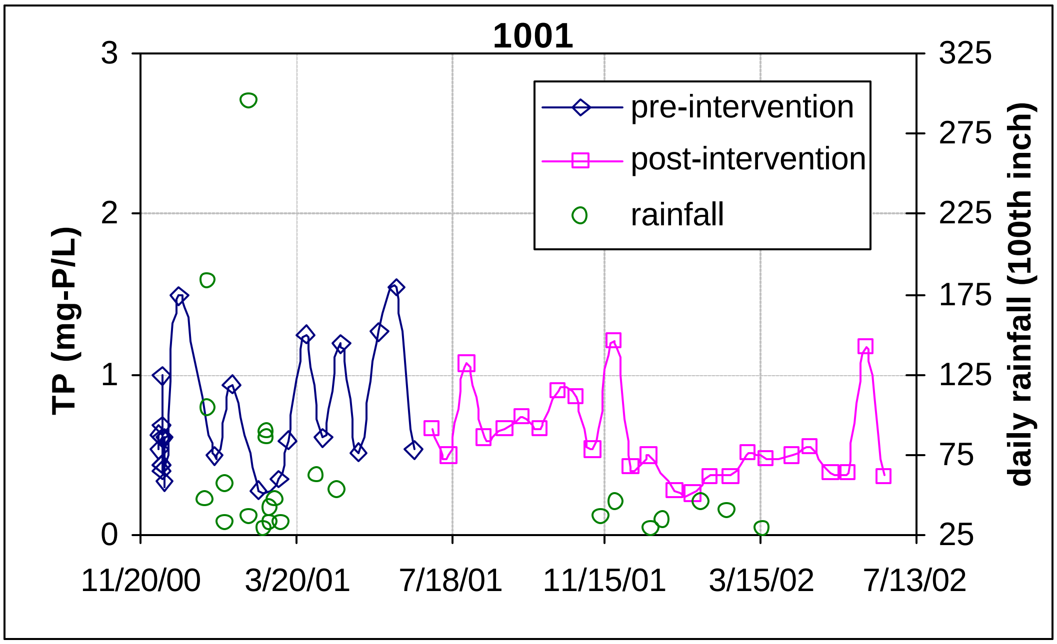
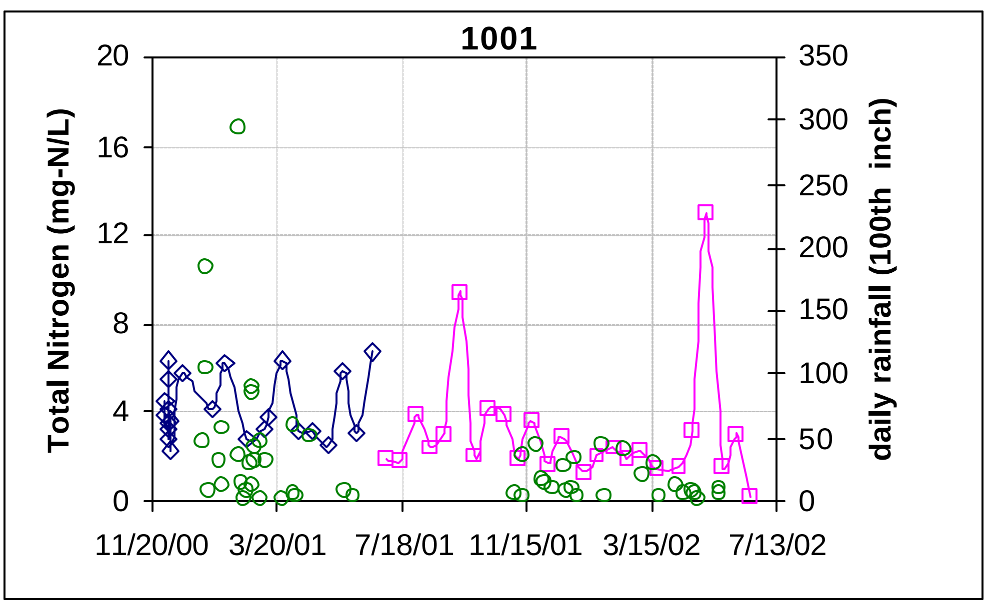


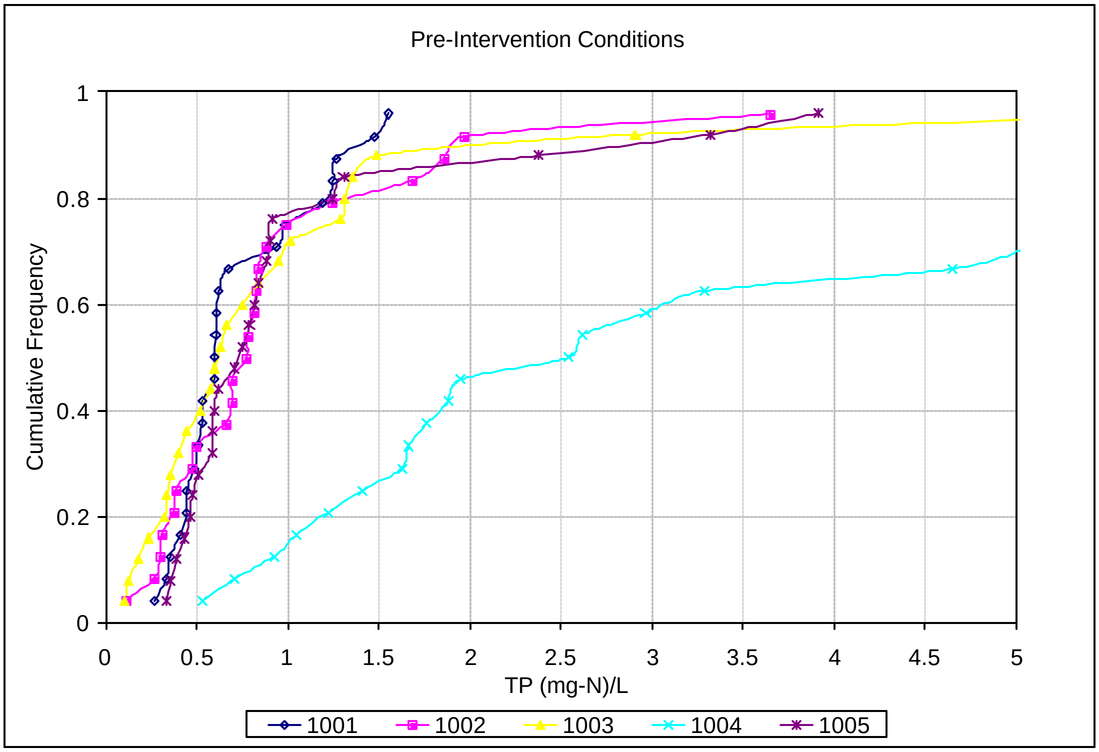
Figure 5-2
Example Time-series Plot of Total Nitrogen with Storm Event Markers.



5.4.3.1 Comparison of Water Quality Data Prior to Intervention

To visually investigate whether the test sites have similar runoff characteristics, probability plots were constructed. Figure 5-3 is an example of a probability plot for TP for all of the test sites. The figure shows that all of the sites have a similar distribution except for Site 1004.

Figure 5-3
Example Probability Plot of Total Phosphorus for All Sites Prior to Intervention.



The next step in the data analysis was to calculate parametric and non-parametric descriptive statistics. Table 5-5 is an example table of descriptive statistics for TN for all sites for both the pre- and post-intervention periods. (Additional descriptive statistics are included in Appendix E2). Table 5-5 includes the number of data points (n), the detection percent (%>MDL/RL), the mean, median, 25 percent trimmed mean, min, max, 25^{th} percentile, 75^{th} percentile, standard deviation, interquartile range (IQR), and the coefficient of skewness (g_s). Also included in the table are critical skewness coefficients (g_{cr}), which are readily available in statistics texts. If the coefficients of skewness are less than these critical values, then the data is symmetric. It should be noted that the measures of central tendency (mean and median) and variability (standard deviation) of the sites during the pre-intervention period are quite different, indicating the data arises from different distributions. The median values are consistently smaller than the mean (in some cases substantially smaller), demonstrating the influence of the outliers on the measure of central tendency. Only three pre-intervention data sets are symmetric, and none of the post-intervention data sets are. Failure to pass the symmetry test indicates the data is not normal. However, passing the symmetry test does not indicate the data is normal; this requires a normality test. The symmetry test, which is easier to conduct than normality tests, serves as an initial screen for normality to reduce the number of data sets needing further investigation.

Table 5-5
Example Table of Descriptive Statistics for Total Nitrogen for Each Site for Pre- and Post-intervention.

		1001		1002		1003		1004		1005	
Parameter	**Statistic**	**Pre**	**Post**	**Pre**	**Post**	**Pre**	**Post**	**Pre**	**Post**	**Pre**	**Post**
TN (calculated)	**n**	23	25	23	25	23	25	23	25	23	25
(mg-N/L)	**% > MDL/RL**	100%	80%	98%	90%	98%	96%	98%	96%	100%	98%
	Mean	4.24	3.09	5.31	3.44	3.66	4.42	48.00	10.18	6.89	7.74
	Median	3.84	2.27	3.95	2.55	2.66	2.50	19.01	5.57	5.06	4.36
	Trimmed mean	3.94	2.40	4.53	2.76	2.93	3.01	33.11	6.47	5.08	4.42
	min	2.30	0.30	1.50	0.78	1.46	0.45	3.28	0.74	2.48	1.07
	max	6.76	12.99	13.83	11.40	12.12	19.91	141.06	40.80	20.41	67.12
	25th percentile	3.20	1.79	2.27	2.10	2.11	2.04	9.05	2.71	3.52	3.47
	75th percentile	5.68	3.13	8.02	4.36	4.81	5.17	94.79	19.18	7.07	5.62
	St Dev	1.41	2.67	3.56	2.51	2.48	4.39	49.17	10.73	5.29	12.85
	IQR	2.48	1.34	5.75	2.26	2.70	3.13	85.74	16.47	3.55	2.15
	Skewness, g_s	0.55	2.82	0.84	1.87	2.13	2.27	0.74	1.37	1.88	4.46
	g_{cr}	0.96	0.92	0.96	0.92	0.94	0.92	0.96	0.92	0.94	0.92
	Symmetric ($g_s < g_{cr}$)?	Y	N	Y	N	N	N	Y	N	N	N

The non-parametric equivalent to the ANOVA test is the K-W test, which tests for a difference between the medians of independent data groups. The K-W test will also test whether the datasets are derived from the same distribution.

Comparison of the mean ranks in Table 5-6 provides an indication of whether the data groups are derived from the same distribution. A p values < 0.05 indicates that two or more of the data groups have different distributions. Examination of the mean ranks in Table 5-6 shows that Sites 1001, 1002, and 1005 have somewhat similar mean ranks, and Sites 1003 and 1004 have somewhat different mean ranks. This suggests that Sites 1003 and 1004 have a different distribution than the other sites. Thus, the K-W test was performed on just Sites 1001, 1002, and 1005. These results are shown in Table 5-7. The p-value is now greater than 0.05, so the distributions of the TN data are not significantly different. Based on this analysis, Site 1002 was determined to be the only control site for comparison of TN data. Furthermore, it is clear that Site 1004 should not be considered as a control site for TN, and Site 1003 should be used with caution.

Table 5-6
Example of Kruskal-Wallis Test Results for Total Nitrogen at the Test Sites Prior to Intervention.

Test:	**Kruskal-Wallis ANOVA**		
Comparison:	**Total Nitrogen: 1001, 1002, 1003, 1004, 1005**		
Performed by:	**GeoSyntec Consultants**		
n	115		
Total Nitrogen	**n**	**Rank sum**	**Mean rank**
1001	23	1128.0	49.04
1002	23	1162.0	50.52
1003	23	774.0	33.65
1004	23	2150.0	93.48
1005	23	1456.0	63.30
Kruskal-Wallis statistic	41.71		
p	<0.0001	(chisqr approximation)	

Table 5-7
Example of Kruskal-Wallis Test Results for Total Nitrogen at Sites 1001, 1002, and 1005 Prior to Intervention.

Test:	**Kruskal-Wallis ANOVA**		
Comparison:	**Total Nitrogen: 1001, 1002, 1005**		
Performed by:	**GeoSyntec Consultants**		
n	69		
Total Nitrogen	**n**	**Rank sum**	**Mean rank**
1001	23	710.0	30.87
1002	23	761.0	33.09
1005	23	944.0	41.04
Kruskal-Wallis statistic	3.27		
p	0.1948	(chisqr approximation)	

5.4.3.2 Comparison of Water Quality Data Before and After Intervention

Side-by-side box plots and probability plot comparisons of pre-intervention and post-intervention were constructed to identify any apparent differences in the central tendency and concentration distributions between the two data sets. Figure 5-4 shows side-by-side box plots of total nitrogen at all of the test sites. Site 1004 was omitted due to its high variability. The figure shows that Site 1001 has a distinct decrease in TN while the other sites do not. However, other sites do show a decreasing trend in median concentration and inter-quartile ranges.

Figure 5-4
Side-by-side Box Plots of Pre- versus Post-Intervention for Total Nitrogen at All Sites.

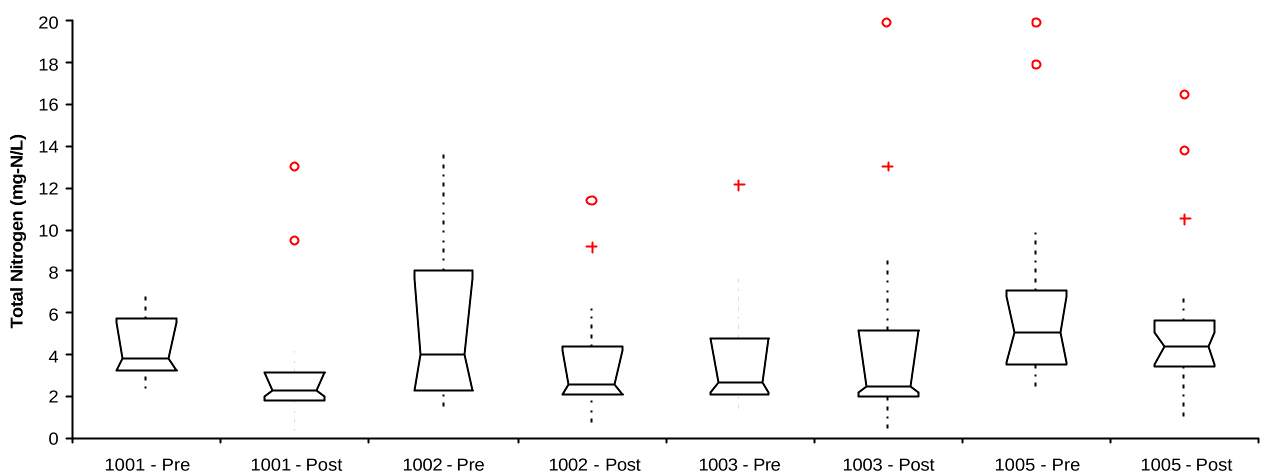
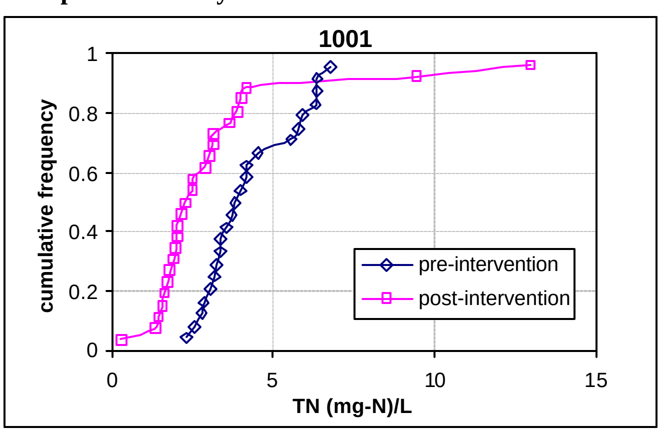


Figure 5-5 is a probability plot of TN for Site 1001 before and after intervention. (Additional probability plot comparisons are included in Appendix E2.) This figure shows a distinct reduction in TN at the site. However, since the data is from different time-periods, this difference could be related to temporal variability.

Figure 5-5
Example Probability Plot of Pre- versus Post-intervention for Total Nitrogen at Site 1001.



To evaluate if temporal variability caused by the different monitoring periods has anything to do with the difference in TN concentrations, the probability plots of the pre- and post-intervention period for Site 1001 were plotted with those for Site 1002 and Site 1005 (as these were determined to be the only valid control sites). These comparison plots are shown on Figure 5-6 and Figure 5-7. For pre-intervention, the distribution of Site 1001 more closely follows the distribution of Site 1005 than that of Site 1002, and for post-intervention the opposite is true. This indicates that the year-to-year variability alone cannot explain the reduction in TN at Site 1001.

Figure 5-6
Example Probability Plot for Total Nitrogen of Site 1001 versus Site 1002 for the Pre- and Post-Intervention Periods.

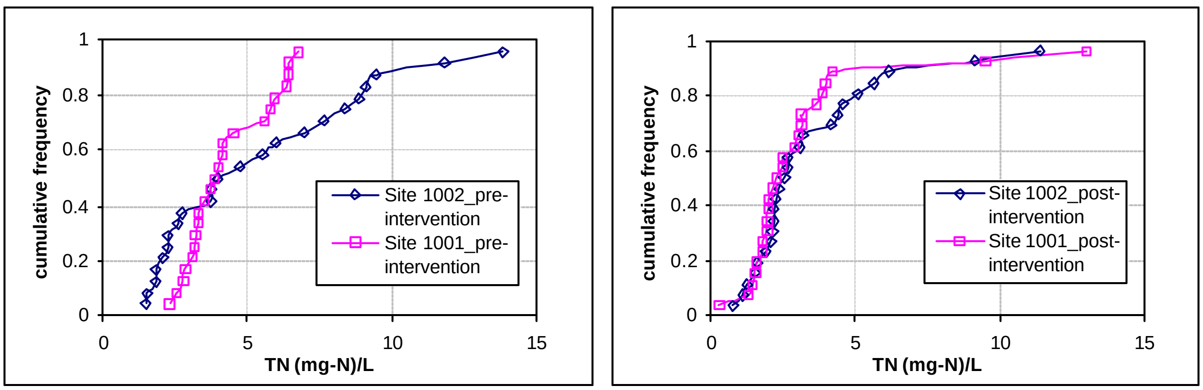
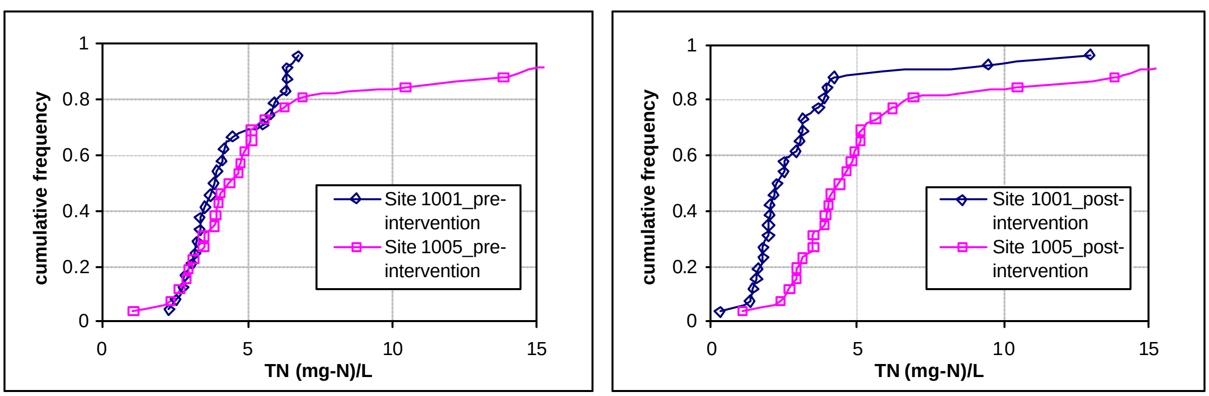


Figure 5-7
Example Probability Plot for Total Nitrogen of Site 1001 versus Site 1005 for the Pre- and Post-Intervention Periods.



The Mann-Whitney test (rank-sum) was used to determine if there is a statistical difference in the median values of two independent data sets (by rejecting the hypothesis that they are the same). Tables 5-8 through 5-10 show the output of the Mann-Whitney tests on Sites 1001, 1002, and 1005, respectively. The tables show a statistically significant difference ($p<0.05$) in the medians between the pre- versus post-intervention TN data at both Sites 1001 and 1002, but not at Site 1005. Furthermore, the difference in the medians at Site 1001 is at a higher level of confidence (more statistically significant) than the difference at Site 1002 (i.e., greater than 99 percent significant compared to about 96 percent significant). The magnitudes of these differences (Hodges-Lehmann estimator) are about 1.5 and 1.3 milligrams of nitrogen per liter (mg-N/L) for Sites 1001 and 1002, respectively. These tests indicate that the difference in the TN medians at Site 1001 from pre-intervention to post-intervention cannot be explained by the year-to-year variation alone (e.g., the intervention appears to have had an effect). It also indicates that the public education applied to Site 1005 did not appear to make a significant difference.

Table 5-8
Example Mann-Whitney Test for Difference in Medians for Total Nitrogen at Site 1001 from Pre- Versus Post-intervention.

Test :	**Mann-Whitney test**			
Alternative hypothesis	**1001: Pre versus Post**			
Performed by:	**GeoSyntec Consultants**			
n	48			
1001	**n**	**Rank sum**	**Mean rank**	**U**
Pre	23	736.0	32.00	115.0
Post	25	440.0	17.60	460.0
Difference between medians	1.497			
95.2% CI	0.883	to +?	(normal approximation)	
Mann-Whitney U statistic	115			
1-tailed p	0.0002	(normal approximation)		

Table 5-9
Example Mann-Whitney Test for Difference in Medians for Total Nitrogen at Site 1002 from Pre- Versus Post-Intervention.

Test:	**Mann-Whitney test**			
Alternative hypothesis:	**1002: Pre versus Post**			
Performed by:	**GeoSyntec Consultants**			
n	48			
1002	**n**	**Rank sum**	**Mean rank**	**U**
Pre	23	651.0	28.30	200.0
Post	25	525.0	21.00	375.0
Difference between medians	1.289			
95.2% CI	0.065	to +?	(normal approximation)	
Mann-Whitney U statistic	200			
1-tailed p	0.0355	(normal approximation)		

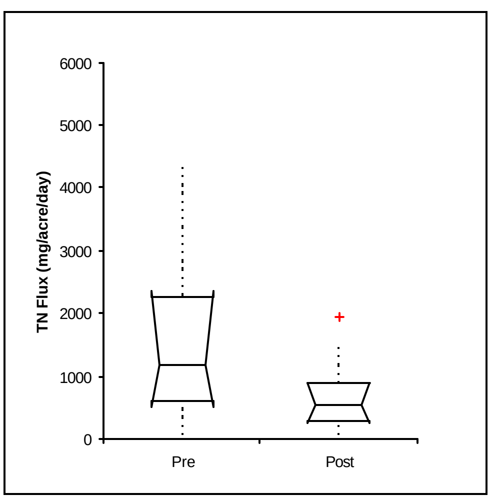
Table 5-10
Example Mann-Whitney Test for Difference in Medians for Total Nitrogen at Site 1005 from Pre- Versus Post-intervention.

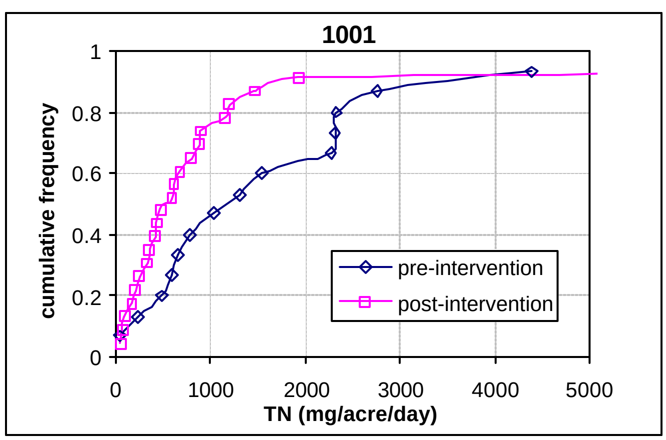
Test:	**Mann-Whitney test**			
Alternative hypothesis:	**1005: Pre versus Post**			
Performed by:	**GeoSyntec Consultants**			
n	48			
1005	**n**	**Rank sum**	**Mean rank**	**U**
Pre	23	610.0	26.52	241.0
Post	25	566.0	22.64	334.0
Difference between medians	0.530			
95.2% CI	-0.446	to +?	(normal approximation)	
Mann-Whitney U statistic	241			
1-tailed p	0.1686	(normal approximation, corrected for ties)		

5.4.3.3 Comparison of Constituent Fluxes Before and After Intervention

The statistical procedures applied to the concentrations examples above were also applied to the constituent fluxes (mass loadings). For completeness, an abridged example analysis is provided here. Figure 5-8 includes side-by-side box plots and probability plots of total nitrogen flux data milligrams per acre per day (mg/acre/day) for Site 1001 at pre- and post-intervention. There appears to be a significant decrease in the median, as well as an overall reduction in the distribution of values.

Figure 5-8
Side-by-side Box Plot and Probability Plots of Pre- Versus Post-Intervention for Total Nitrogen Flues at Site 1001.





Table

Table 5-11 shows the results of the Mann-Whitney test (rank-sum) for the total nitrogen flux at Site 1001. The medians from pre- to post-intervention are statistically significantly different at the 95 percent confidence level ($p<0.05$). The magnitude of the difference (the Hodges-Lehmann estimator) is approximately 530 mg/acre/day, indicating a relatively large reduction in total nitrogen loads from the neighborhood. However, as discussed below, the extent to which the ET controllers contributed to this reduction is unclear.

The nitrogen fluxes used in this analysis were computed as the product of the measured concentration and the average daily flow. Therefore, the reduction in TN flux could be due to a reduction in flow, a reduction in concentration, or a combination of both. Analyses presented earlier showed a statistically significant reduction in median TN concentration at Site 1001 between the pre- and post-intervention periods. Similarly, analyses discussed elsewhere in this report indicate that there was a statistically significant reduction in flow at Site 1001 between the pre- to post-intervention periods; however, it was cautioned that the pre- and post-intervention periods are not comparable due to seasonal differences in the data collection period. Thus, observed reductions in flow in 1001 could be influenced by seasonal factors. Therefore, the extent to which the ET controllers contributed to a reduction in flow is unknown. Consequently, reductions in TN flux could be attributed to a combination of TN reduction, flow reduction, and/or seasonal factors.

Table 5-11
Example Mann-Whitney Test for Difference in Medians for Total Nitrogen Flux at Site 1001 from Pre-Versus Post-intervention.

Test :	**Mann-Whitney test**			
Alternative hypothesis	**1001 flux (mg/acre/day): Pre vs. Post**			
Performed by:	**GeoSyntec Consultants**			
n	36			
1001_flux (mg/acre/day)	**n**	**Rank sum**	**Mean rank**	**U**
Pre	14	320.0	22.86	93.0
Post	22	346.0	15.73	215.0
Difference between medians	529.389			
95.1% CI	115.985	to +?	(normal approximation)	
Mann-Whitney U statistic	93			
1-tailed p	0.0239	(normal approximation)		

The above results suggest that it would be valuable to complete a more robust statistical evaluation of the data because some significant management implications could be determined.

5.4.4 Watershed Implications

The water quality evaluation results were examined in the context of existing TMDLs in the San Diego Watershed. Most of the existing TMDLs are reviewed below, and possible inferences and implications of the R3 Study data for TMDL compliance are discussed. The sediment and organophosphorus pesticide TMDLs were not reviewed because sediment data was not collected (the vast majority of sediments are transported by storm flows) and because Schiff and Tiefenthaler (SCCWRP, 2003) have previously conducted an extensive analysis of the OP pesticide data.

5.4.4.1 Comparisons with Regulatory Requirements

Mean dry-season concentrations for nutrients, toxics, metals, and pathogens at the R3 Study Sites were compared with regulatory objectives including TMDL's, California Toxics Rule (CTR) criteria, and Basin Plan objectives in Tables 5-12 and 5-13. These comparisons are strictly descriptive and provide a rough sense of dry-season residential water quality in comparison to regional water quality objectives. This comparison shows substantial variability between neighborhoods and among constituents.

Table 5-12
Comparison of Dry Season Concentrations of Nutrients and Toxics at R3 Study Sites with Regulatory Objectives

Parameter/Location	Objective	Site 1001	Site 1002	Site 1003	Site 1004	Site 1005
TIN (San Diego Creek Reach 1 / Reach 2)	13 mg/L / 5 mg/L (RWQCB-TMDL)	4.079 mg/L	0.464 mg/L	2.18 mg/L	18.16 mg/L	4 mg/L
		Percent of Samples above Toxics TMDL				
		Site 1001	**Site 1002**	**Site 1003**	**Site 1004**	**Site 1005**
Chlorpyriphos -Acute (San Diego Creek Reach 1)	18 ug/L (RWQCB-TMDL)	36.59	N/A	N/A	22.76	43.9
Chlorpyriphos - Chronic- (San Diego Creek Reach 1)	12.6 ug/L (RWQCB-TMDL)	46.34	N/A	N/A	26.02	49.59
Diazinon - Acute- (San Diego Creek Reach 1)	72 ug/L (RWQCB-TMDL)	70.73	N/A	N/A	69.11	73.17
Diazinon - Chronic- (San Diego Creek Reach 1)	45 ug/L (RWQCB-TMDL)	74.80	N/A	N/A	75.61	77.24

Table 5-13
Comparison of Dry Season Concentrations of Metals and Pathogens at R3 Study Sites with Regulatory Objectives

		Percent of Samples above CTR Criteria				
Parameter	**Objective**	**Site 1001**	**Site 1002**	**Site 1003**	**Site 1004**	**Site 1005**
Copper -Acute	13 ug/L (CTR Criteria for Metal Toxicity*)	43.59	43.59	46.14	46.15	71.79
Copper - Chronic	9 ug/L (CTR Criteria for Metal Toxicity*)	74.36	56.41	76.92	74.36	87.18
Lead -Acute	65 ug/L (CTR Criteria for Metal Toxicity*)	0	0	0	0	0
Lead -Chronic	2.5 ug/L (CTR Criteria for Metal Toxicity*)	10.26	28.21	10.26	12.82	28.21
Zinc -Acute	120 ug/L (CTR Criteria for Metal Toxicity*)	0	7.69	5.13	7.69	15.38
Zinc -Chronic	120 ug/L (CTR Criteria for Metal Toxicity*)	0	7.69	5.13	7.69	15.38
		Median Dry Season Fecal Coliform				
Parameter	**Objective**	**Site 1001**	**Site 1002**	**Site 1003**	**Site 1004**	**Site 1005**
Fecal Coliform	200 MPN/100 mL (RWQCB Basin Plan)	1400 MPN/100 mL	3000 MPN/100 mL	5000 MPN/100 mL	13000 MPN/100 mL	65000 MPN/100 mL

5.4.4.2 Nitrogen

***Nitrogen Water Quality Objectives and TMDLs* –** The Basin Plan water quality objectives for nitrogen in San Diego Creek are 13 milligrams per liter (mg/L) Total Inorganic Nitrogen (TIN) in Reach 1, and 5 mg/L TIN in Reach 2 (RWQCB, 1995). Reach 1 extends from Newport Bay to Jeffrey Road, and Reach 2 extends from Jeffrey Road to the headwaters. There is no numeric standard for nitrogen in Upper Newport Bay in the Basin Plan.

The nitrogen TMDL for Upper Newport Bay is based on the general goal of reducing nutrient loads to Newport Bay by 50 percent, to levels observed in the early 1970s (USEPA, 1998b). The nitrogen TMDL sets phase-in limits on TN loads to Newport Bay (see Table 5-14). Separate loads are established for the dry and wet seasons (dry season is from April 1 to September 30). In addition, the winter load is exclusive of storm flows with an average daily flow greater than 50 cubic feet per second (cfs) in San Diego Creek at Campus Drive.

There is no TMDL for nitrogen loads in San Diego Creek, Reach 1 because it was reasoned that attainment of the 50 percent reduction in nitrogen loads to Newport Bay would result in compliance with the Basin Plan in-stream water quality standard for Reach 1 (13 mg/L TIN). However, for Reach 2, it was determined that the average in-stream nitrogen concentrations would likely remain close to or above the Basin Plan in-stream water quality standard (5 mg/L TIN), even with attainment of the Newport Bay TMDLs. Therefore a TMDL of 14 lbs/day TN was established for Reach 2 (see Table 5-14) and is applicable for all flows exclusive of storm flows greater than an average daily flow of 25 cfs in San Diego Creek at Culver Drive.

Table 5-14
Summary of Nutrient TMDLs for Upper Newport Bay and San Diego Creek

TMDL	Dec 31, 2002	Dec 31, 2007	Dec 31, 2012
Newport Bay Watershed, TN – Summer load (4/1 to 9/30)	200,097 lbs	153,861 lbs	
Newport Bay Watershed, TN – Winter load (10/1 to 3/31; non-storm)			144,364 lbs
Newport Bay Watershed, Total Phosphorus – Annual Load	86,912 lbs	62,080 lbs	
San Diego Creek, Reach 2, daily load			14 lbs/day
Urban Runoff Allocation for the Newport Bay Watershed Summer load Winter load	 22,963	 11,481	 38,283

***Study Data Comparison with Nitrogen Water Quality Objective* –** The Basin Plan water quality objectives are expressed in terms of TIN, which is comprised of nitrate/nitrite nitrogen and ammonia. By far the majority of the TIN in San Diego Creek is comprised of nitrate/nitrite nitrogen, as measured ammonia concentrations were typically quite low with a majority below the detection limit. For this reason, only the nitrate/nitrate concentration data is compared to the Basin Plan objectives in this report.

Table 5-15 shows the mean and median nitrate/nitrite concentrations measured in the five study sites. The mean and median nitrate/nitrite concentration of all sites except 1004 was below the Reach 2 Basin Plan objective of 5 mg/L TIN. As discussed previously, Site 1004 may not be a representative control site because the underlying distribution of pre-intervention nitrogen data appears to be different from the other sites. Similar arguments may also be true for Site 1003. With the exception of Site 1004, mean nitrate/nitrite concentrations suggest that, <u>on average</u>, residential runoff from these sites does not contribute to the exceedance of Basin Plan standards for TIN in receiving waters in San Diego Creek, Reach 1 and 2. The Reach 2 water quality objective was occasionally exceeded in all sites, except for the post intervention conditions in 1001 and 1002.

Table 5-15
Mean and Median Nitrate/Nitrite Concentration (mg/l) by Site (all data).

	1001		1002		1003		1004		1005	
	Pre	**Post**	**Pre**	**Post**	**Pre**	**Post**	**Pre**	**Post**	**Pre**	**Post**
n	23	25	23	25	24	25	23	25	24	25
Mean	2.56	1.47	2.57	1.07	2.13	1.71	36.50	6.61	2.61	4.13
Median	2.32	1.38	1.56	0.93	1.68	0.94	16.88	2.29	2.45	1.48
n>5 mg/L	1	0	4	0	1	2	18	8	2	1
n>13 mgL	0	0	0	0	0	0	12	4	0	1

The mean and median nitrate/nitrate concentrations in Sites 1004 and 1005 exhibit exceedances of the 5 mg/L standard during pre- and/or post intervention conditions. Site 1004, in particular, had high levels of measured nitrate/nitrite concentrations, especially during the pre-intervention period. A number of these high readings exceed the Reach 1 water quality objective of 13 mg/L TIN. The results from Site 1004 are not consistent with those from the other four study sites, and the source of the high readings is unknown. Localized conditions involving excessive fertilizer usage by a few users could possibly be a factor in these elevated readings. In particular, the R3 Study mentions an unknown connection to a neighboring watershed, which could explain the source of elevated nutrient levels.

The Mann-Whitney (rank-sum) test was performed to compare the statistical difference between median concentrations during pre- and post-intervention periods. The median nitrate/nitrite in the post-intervention period was lower at all sites, and the difference was statistically significant at the 0.05 confidence level. As the control stations exhibited this trend, the data (i.e. entire data sets with unequal seasonal coverage) cannot be used to ascertain if the structural and educational BMPs were effective in reducing the runoff concentrations of nitrate/nitrite.

Clearly another factor is contributing to reduced concentrations in the post-intervention period. One possibility that was investigated is differences in seasons, year-to-year variability, and sampling times of the pre- and post-intervention data. Table 5-16 presents mean and median concentrations for comparable seasons and sampling times. The table shows that there are still noticeable reductions in all of the median concentrations, except Site 1005. Applying the Mann-Whitney (rank-sum) test to the data, it was found that statistically significant differences between median nitrate/nitrite concentrations in the pre- and post-intervention periods occurred only at Sites 1001 and 1004, as compared to all sites when all data is considered. These results indicate that seasonal effects are present in the data and should be considered in the study evaluation. It may be inferred from these results that there were significant reductions in the nitrate/nitrite concentration in the intervention site during the wet season that may, in part, be attributable to the structural BMPs. It is unknown whether similar reductions would occur in dry weather runoff during the dry season because such data was not collected during the pre-intervention period.

Table 5-16
Mean and Median Nitrate/Nitrite Concentration (mg/l) by Site for Comparable Seasons and Sampling Times[1]

	1001		**1002**		**1003**		**1004**		**1005**	
	Pre	**Post**	**Pre**	**Post**	**Pre**	**Post**	**Pre**	**Post**	**Pre**	**Post**
n	18	14	18	14	19	14	18	14	19	14
Mean	2.38	1.43	1.95	0.95	2.17	1.66	26.24	6.57	2.24	6.27
Median	2.22	1.48	1.16	0.96	1.50	1.02	8.94	2.06	2.03	1.96
n>5 mg/L	0	0	2	0	1	1	13	4	1	1
n>13 mg/L	0	0	0	0	0	0	7	3	0	1

1 – evening samples were deleted from the pre-intervention data. The post-intervention data include only those data collected in months identical to the pre-intervention period.

Study Data Comparison with Nitrogen TMDLs - The nitrogen TMDL is expressed in terms of total nitrogen TN loads. TN concentrations were calculated from the monitoring data as the sum of the nitrate/nitrite nitrogen and total Kjeldahl nitrogen (TKN) nitrogen. Table 5-17 shows the mean and median TN concentrations measured in the five study sites. The mean and median TN concentration in dry weather runoff are generally in the range of 2 to 5 mg/L, with the exception of Site 1004 where substantially higher concentrations were measured. The rank sum tests indicated that median TN concentrations were significantly lower (in a statistical sense) in the post-intervention period in Site 1001 (structural BMPs, see Table 5-8), and at Site 1002 (control, see Table 5-9). Based on the probability plots in Appendix E2, Site 1004 is expected to as well. However, Sites 1003 and 1005 did not show statistically significant reductions. These results did not change when only subsets of the data were used to consider possible effects stemming from the sampling time and sampling months.

Table 5-17
Mean and Median TN Concentration (mg/l) by Site

	1001		**1002**		**1003**		**1004**		**1005**	
	Pre	**Post**	**Pre**	**Post**	**Pre**	**Post**	**Pre**	**Post**	**Pre**	**Post**
All Data										
n	23	25	23	25	23	25	23	25	23	25
Mean	4.24	3.09	5.31	3.44	3.66	4.42	48.00	10.18	6.89	7.74
Median	3.84	2.27	3.95	2.55	2.66	2.50	19.01	5.57	5.06	4.36
Subsets[1]										
n	18	14	18	14	18	14	18	14	18	14
Mean	4.18	2.78	4.51	2.63	3.71	3.71	33.99	8.91	6.98	9.91
Median	3.62	2.02	3.22	2.21	2.51	2.47	12.14	3.74	4.17	3.96

1 – Data subsets with comparable sampling time and seasons. Evening samples were deleted from the pre-intervention data. The post-intervention data include only those data collected in months identical to the pre-intervention period.

TN flux estimates were calculated for Sites 1001 and 1005 (Table 5-18). The flow measurements at Sites 1002 to1004 are not reliable. Therefore, flux estimates were not calculated for these sites. Flux estimates were calculated as the product of the constituent concentration and the average daily flow occurring on the day of the sample collection. The flux estimates were found to be quite variable as they depend on both flow and concentration measurements. Table 5-18 shows that median TN flux estimates decreased from the pre- to post-intervention periods for both sites. Mann-Whitney (rank sum) tests show the reductions to be statistically significant (Table 5-11). Because comparable data is not available for the control sites, it is not possible to infer whether these reductions are influenced by the ET controllers in the intervention site (1001). Also, as previously discussed, the reduction in TN flux may be attributable to a reduction in flow, a reduction in concentration, seasonal factors, or a combination of these.

Table 5-18
Mean and Median TN Flux (mg-N/acre/day) by Site

	1001		**1005**	
	Pre	**Post**	**Pre**	**Post**
All data				
n	14	22	10	21
Mean	1476	1667	2104	6537
Median	1164	530	1568	1177
Subset[1]				
n	12	14	10*	8
Mean	1384	587	2104	1716
Median	902	497	1568	960

1 – Data subsets with comparable sampling time and seasons. Evening samples were deleted from the pre-intervention data. The post-intervention data include only those data collected in months identical to the pre-intervention period.
* – Same as the all data case

Although the flux estimates in Table 5-18 are limited in number, duration, and location, they can be used to speculate about the magnitude of the urban area contribution of TN loads to Newport Bay and the potential reduction in loads from structural and nonstructural BMPs. Based on the limited flux data, the annual TN load to Newport Bay in dry weather runoff from urban areas in the San Diego Creek Watershed is estimated to range between 37,000 to 50,000 lbs per year under existing land-use conditions (see Table 5-19). This is for the most part below the 2012 urban runoff allocation of 49,764 lbs. The annual TN load is estimated to increase to 50,000-67,000 lbs per year under build-out conditions.

According to the 2001 report on the nutrient TMDL (OCPFRD, 2001), the average daily TN load in San Diego Creek at Campus Drive was 540 lbs/day between July 2000 and June 2001. This converts to an annual load of about 197,000 lbs, which is below the 2007 TMDL (note: San Diego Creek is the majority but not sole contributor of TN loads to Newport Bay). Estimates in Table 5-19 suggest that dry weather runoff from urban areas account for about 20 to 25 percent of the annual TN in the San Diego Creek Watershed. If it is assumed that flux reductions observed in the post intervention period are attributable to the structural and nonstructural BMPs, and if similar interventions could hypothetically be implemented on a watershed-wide basis, then the potential reduction in annual dry weather TN loads is estimated to range between 12,500-20,000 lbs. This would represent a reduction of about 6-10 percent of the current TN loads and about 30-40 percent of the estimated current dry weather urban loads. These estimates are based on few data collected in a limited area and should therefore be considered preliminary in nature.

Table 5-19
Estimated Annual TN Loads in Dry Weather Runoff from Urban Areas in the San Diego Creek Watershed

	TN flux (mg-N/acre/d)	**Annual TN Load to Newport Bay (lbs) Existing land-use[1]**	**Annual TN Load to Newport Bay (lbs) Built-out land-use[2]**
Pre-intervention conditions	1160 – 1560	37,300 – 50,500	50,000 – 67,000
Post-intervention conditions	530 – 1180	17,000 – 38,000	23,000 – 51,000
Potential reduction		~12,500 – 20,000	~16,000 – 27,000

1 –Used 40000 acres or about 53% of the San Diego Creek Watershed area (IRWD, 2003). For comparison, urban land use in 1999 use was estimated at 35,500 acres of the watershed area at Campus Drive (Tetra -Tech, 2000).
2 – Used 53500 acres or about 71% of the San Diego Creek Watershed area (IRWD, 2003).

The following conclusion can be made based on the analyses above:

- Average and median nitrate/nitrite concentrations in dry weather runoff are below the Reach 2 water quality objective (5 mg/L), for most but not all study sites.
- Occasional exceedance of the Reach 2 water quality objective occurred in all study sites.
- The majority of measured nitrate/nitrite concentrations at Site 1004 during the pre-intervention period were greater than the Reach 2 water quality objective of 5 mg/L. The data is not consistent with those from the other sites. The cause is unknown, but could possibly be related to the unknown connection to the neighboring nursery discussed in the R3 report.
- Sampling periods (months) and sampling time (morning versus evening) were found to affect the statistical significance of differences between pre- and post- intervention median nitrate/nitrate concentration in some of the sites. The sampling period and sampling time did not affect the statistical significance of differences between pre- and post-intervention median TN concentrations.
- Median TN fluxes at Sites 1001 and 1005 were statistically smaller in the post-intervention period. The extent to which the structural and nonstructural BMPs contributed to these reductions cannot be determined due to the lack of reliable flow data in the control sites.
- Preliminary estimates of annual TN loads to Newport Bay in dry weather runoff from urban sources range between 37,000 to 50,000 lbs per year, or about 20 to 25 percent of the current TN loads.
- The potential reductions in annual dry weather TN loads due to implementation of BMPs on a watershed basis is estimated to range between 12,500-20,000 pounds per year. This would represent a reduction of about 6-10 percent of the current TN loads and 30-40 percent of the urban loads.

5.4.4.3 Phosphorus

The majority of the annual TP load in the San Diego Creek Watershed occurs in the wet season, and has been correlated with sediment loads generated by storm events (USEPA, 1998b). This correlation suggests that a majority of phosphorus occurs in particulate form attached to sediments. The main sources of the TP are in Peters Canyon Wash and San Diego Creek above Culver Drive (USEPA, 1998b).

Phosphorus TMDL **–** There is no numeric objective for phosphorus for San Diego Creek in the Basin Plan. Because measured TP and sediment loads are correlated, it was determined in the TMDL that a 50 percent reduction in TP loads would be achieved through compliance with the sediment TMDL (USEPA, 1998a). Accordingly, the TMDL for TP was based on a 50 percent reduction of average annual load estimated at 124,160 lbs (USEPA, 1998b). The TMDLs are applicable for all flow conditions. The target compliance date was set for December 31, 2007.

The annual TP load allocation for urban areas is 4102 lbs by 2002, reducing to 2960 lbs by 2007. According to the USEPA (1998b), the TP is allocated in the same proportion as sediments. The annual urban area (stabilized vs. construction) sediment allocation for the Newport Bay Watershed is 50 tons distributed over 95.3 square miles (see Table 5 in USEPA, 1998a). This is a very small allocation over a large area. By contrast, the annual construction allocation is 6500 tons distributed over the assumed 3.0 square miles under construction in any one year. Using the same proportions of sediment load allocations, the TP load rate based on the 2007 urban allocation is 2960 lbs/95.3 square miles = 0.0485 lbs/acre/yr. If the construction and urban allocations are combined, the TP load rate based on the combined 2007 urban and construction allocations is (2960+12810) lbs/(95.3+3.0) square miles = 0.251 lbs/acre/yr.

Study Data Comparison with TMDLs **–** Similar to the nitrogen TMDL, the phosphorus TMDL is expressed in terms of total annual TP loads. Table 5-20 shows the mean and median TP concentrations measured in the five study sites. The mean and median TP concentrations in dry weather runoff are below 1.2 mg/L in all sites, with the exception of Site 1004, where substantially higher concentrations were measured. Comparison of the pre- and post-intervention median TP concentrations in all data (Table 5-20) reveals an increase in the median TP concentration during the post-intervention period for all sites except the intervention Site 1001 and Site 1004. In contrast, when subsets of the data with similar seasons and sampling times are considered (Table 5-20), there is a decrease in the median TP concentration at all sites except 1005. This indicates that there are seasonal influences in the data, which presumably are related to rainfall. Unfortunately, no data is available to permit comparison of pre- and post-intervention concentrations for dry weather flows during the dry season.

Table 5-20 Mean and Median TP Concentration (mg/l) by Site

	1001		**1002**		**1003**		**1004**		**1005**	
	Pre	**Post**	**Pre**	**Post**	**Pre**	**Post**	**Pre**	**Post**	**Pre**	**Post**
All Data										
n	23	25	23	25	24	25	23	24	24	25
Mean	0.73	0.60	0.92	0.84	0.98	1.21	3.33	1.50	1.01	1.19
Median	0.60	0.51	0.77	0.82	0.62	0.67	2.54	1.05	0.73	0.85
Subsets[1]										
n	18	14	18	14	19	14	18	13	19	14
Mean	0.78	0.47	0.91	0.67	1.13	0.57	2.62	1.33	0.93	1.24
Median	0.61	0.41	0.73	0.56	0.75	0.58	1.82	1.07	0.75	0.83

1 – Data subsets with comparable sampling time and seasons. Evening samples were deleted from the pre-intervention data. The post-intervention data include only those data collected in months identical to the pre-intervention period.

TP flux estimates were calculated for Sites 1001 and 1005 using the approach discussed in the nitrogen section above. Table 5-21 shows that median TP flux estimates decrease from the pre- to post-intervention periods at the intervention site (1001), but not in the education only site (1005). Mean fluxes increased at both sites. However, as discussed earlier, the mean values are strongly influenced by outliers and do not provide a good measure of central tendency for the data. Application of the Mann-Whitney (rank sum) test shows the reduction in median TP flux at Site 1001 is statistically significant. This suggests that the structural BMPs had a positive influence in reducing the TP fluxes. However, because comparable data is not available for the control sites, it is not possible to ascertain the extent to which the ET controllers contributed to these reductions. Also, as discussed previously, reductions in flux could be influenced by several factors: reduction in concentration, reduction in flow, and/or seasonal variability.

Table 5-21
Mean and Median TP Flux (mg-P/acre/day) by Site (all data)

	1001		**1005**	
	Pre	**Post**	**Pre**	**Post**
All data				
n	14	22	10	21
Mean	265	370	473	1327
Median	164	109	219	219

Similar to the previous analyses of TN loads, the TP flux estimates in Table 5-21 can be used to speculate about the magnitude of the urban area contribution of TP loads to Newport Bay and the potential reduction in loads from structural BMPs. Based on the limited flux data, the annual TP load to Newport Bay in dry weather runoff from urban areas in the Newport Bay Watershed is estimated to range between about 5,000 to 11,000 lbs per year (see Table 5-22), assuming a total urban area of 95.3 square miles obtained from Table 5 of the sediment TMDL (USEPA, 1998a). These estimated annual TP loads are greater than the urban allocation (for both dry and wet weather) and are less than the combined urban and construction allocations (Table 5-22). However, these estimates are based on dry weather data only, and it is expected that a major portion of the TP loads will occur in runoff from winter storms. Therefore, actual annual TP loads would be expected to be greater. If it is hypothesized that flux reductions observed at the intervention site (1001) could be realized over the entire watershed, then the potential reduction in annual dry weather TP loads from urban areas is estimated at 2700 lbs. As stated previously, these estimates are based on few data collected in a limited area and should therefore be considered preliminary in nature.

Table 5-22
Estimated Annual TP Loads in Dry Weather Runoff from Urban Areas in the San Diego Creek Watershed

	TP flux (mg-P/acre/d)	Annual TP Load Rate to Newport Bay (lbs/acre/year)[1]	Annual TP Load to Newport Bay (lbs/year)
2007 Urban Area Allocation for Newport Bay		0.0485	2960
2007 Combined Urban and Construction Area Allocation for Newport Bay		0.251	15770
Pre-intervention conditions (median fluxes)	164 – 219	0.132 – 0.176	8049 – 10748
Post-intervention conditions (median fluxes)	109 – 219	0.088 – 0.176	5350 – 10748
Potential reduction			2700

1 - urban area is 95.3 square miles and the construction area is 3.0 square miles based on Table 5 in USEPA,1998a

5.4.4.4 Metals

Metals TMDLs – The USEPA (June 2002) determined that TMDLs are required for dissolved copper, lead, and zinc in San Diego Creek, Upper Newport Bay, and Lower Newport Bay, and that TMDLs are required for cadmium in San Diego Creek and the Upper Newport Bay. The TMDLs for San Diego Creek are expressed as concentration limits, based on the California Toxic Rule (CTR) criteria at various hardness values that are associated with different flow regimes (Table 5-23). The flow regimes are based on 19 years of flow measurements in San Diego Creek at Campus Drive. The concentration-based TMDLs apply to all freshwater discharges to San Diego Creek, including discharges from agricultural, urban, and residential lands, and storm flow discharges. The applicable flow regime at any location in the entire watershed is determined on the basis of discharge at Campus Drive.

Table 5-23
Summary of Dissolved Metal TMDLs for San Diego Creek

Dissolved Metal (?g/l)	**Base flow (0–20 cfs) hardness @ 400 mg/L**		**Small flows (21-181 cfs) hardness @ 322 mg/L**		**Medium flows (182-814 cfs) hardness @ 236 mg/L**		**Large flows (>814 cfs) hardness @ 197 mg/L**
	Acute	**Chronic**	**Acute**	**Chronic**	**Acute**	**Chronic**	**Acute**
Cadmium	19.1	6.2	15.1	5.3	10.8	4.2	8.9
Copper	50	29.3	40	24.3	30.2	18.7	25.5
Lead	281	10.9	224	8.8	162	6.3	134
Zinc	379	382	316	318	243	244	208

Metals Sources **–** The USEPA (June 2002) conducted a source analysis as part of the TMDL preparation. Surface runoff is the largest contributor of metals loads in the San Diego Creek watershed, which includes natural and man made sources (USEPA, June 2002). Much of the metals loads are from natural sources. The estimated anthropogenic contributions are metal specific and range from about 33 percent for zinc to 63 percent for cadmium (USEPA, June 2002). A primary anthropogenic source of heavy metals is runoff from urban roads, which contributes to sources of cadmium (tire wear), copper (brakes, tires), lead (brakes, tires, fuels and oils), and zinc (tires, brakes, galvanized metals). Use of copper sulfate by nurseries may also be a minor source of copper loads. Other copper and zinc uses in building materials (roofing and roof drains) may be another source.

The USEPA found that metal inputs were heavily influenced by rainfall and stream flow rates. Monitoring results were reported to be highly variable due to different rainfall amounts and flows during each water year. The USEPA estimated that base flows account for 25 percent of the total metal loadings, with the remainder from low, medium and large flows caused by storms.

The USEPA's preliminary analyses suggest that: 1) a primary source of metals in dry weather runoff in the study watershed is from roads (i.e. wash off of metals in driveways, parking lots, streets, gutters, etc.); 2) the runoff concentrations will be influenced by rainfall which result in wash off of accumulated metals; and 3) the concentrations can be variable depending on the amount of rainfall.

Study Data Comparison with Base Flow TMDLs **–** The metals TMDLs for base flow conditions are based on meeting the CTR criteria at a total hardness of 400 mg/L. The CTR criteria express maximum allowable concentrations in receiving waters for acute (short term) and chronic (4-day) exposure periods. The acute and chronic criteria are expressed as values that cannot be exceeded more that once in three years. Although the criteria are applicable in the receiving waters and not in the urban runoff per se (i.e. the measured dry weather discharge), exceedance of the CTR in the urban discharge would suggest a potential for the discharge to contribute to an exceedance in the receiving waters.

Table 5-24 shows the mean and median heavy metal concentrations in the five study sites. With the exception of mean copper concentrations in some of the sites, all mean and median concentrations were below the chronic and acute CTR criteria. Copper, lead, and zinc concentrations occasionally exceeded the chronic CTR criteria, and copper and zinc concentrations occasionally exceeded the acute criteria. These exceedances suggest that the dry weather runoff can potentially contribute to an exceedance in the receiving waters. However, if intervention is determined to be effective in reducing runoff flows, then the BMPs would help to reduce impacts of these potential exceedances by allowing for greater dilution with the in-stream flows.

Table 5-24
Mean and Median Metal Concentrations (mg/L) by Site (all data)

	1001		1002		1003		1004		1005	
	Pre	**Post**	**Pre**	**Post**	**Pre**	**Post**	**Pre**	**Post**	**Pre**	**Post**
Cadmium										
n	23	25	23	25	24	25	23	25	24	25
Mean	0.26	0.14	0.47	0.44	0.27	0.17	0.64	0.22	0.21	0.29
Median	0.27	0.10	0.24	0.10	0.10	0.10	0.36	0.10	0.10	0.10
n>6.2 ? g/l	0	0	0	0	0	0	0	0	0	0
n>19.1 ? g/l	0	0	0	0	0	0	0	0	0	0
Copper										
n	23	25	23	25	24	25	23	25	24	25
Mean	13.5	16.9	27.3	30.3	11.5	26.6	21.8	17.7	32.1	30.8
Median	11.5	11.4	10.9	14.0	11.1	14.3	12.7	11.4	12.3	20.4
n>29.3 ? g/l	2	2	3	7	0	2	5	4	3	5
n>50 ? g/l	0	1	3	3	0	2	2	3	3	2
Lead										
n	23	25	23	25	24	25	23	25	24	25
Mean	0.8	1.6	5.9	4.7	0.8	1.6	3.5	1.5	1.0	3.2
Median	0.6	0.6	0.9	1.2	0.6	0.8	0.7	0.7	0.7	1.3
n>10.9 ? g/l	2	1	2	3	0	0	2	0	0	1
n>281 ? g/l	0	0	0	0	0	0	0	0	0	0
Zinc										
n	23	25	23	25	24	25	23	25	24	25
Mean	58.7	37.2	115.2	86.3	56.3	56.8	83.6	40.9	74.0	75.0
Median	56.0	50.2	53.4	57.2	50.7	53.9	50.8	43.8	52.4	54.5
n>382 ? g/l	0	0	1	2	0	0	1	0	0	0
n>379 ? g/l	0	0	1	2	0	0	1	0	0	0

Dry weather metals monitoring information in the Central Irvine Channel, the immediate receiving water of the study watersheds, was unavailable. OCPFRD dry weather monitoring data is available in San Diego Creek at Campus Drive, which is quite a way downstream from the study sites. Data collected between December 2001 and June 2002 (Table 5-25) shows that average dry weather concentrations at Campus Drive are well below mean and median concentrations measured in dry weather runoff from the study watershed. Similar comparisons cannot be made for lead and cadmium because the method detection limits in the OCPFRD data are greater than those in the R3 data. None of the OCPFRD dry weather data exceeded the chronic or acute criteria.

Table 5-25
Summary of OCPFRD Dry Weather Monitoring Data of San Diego Creek at Campus Drive (12/01 to 6/02)

	Cadmium	**Copper**	**Lead**	**Zinc**
Sample number	24	24	24	24
Range	All < 1 ?g/l	<2 – 16 ?g/l	<2-2.4 ?g/l	<10-16
Mean		7.4 ?g/l	most <2 ?g/l	most <10
Median-		6.8?g/l		

These comparisons suggest that metal loads in dry weather runoff from the study (urban) watersheds could be a contributing factor to dry weather copper and zinc loads measured at Campus Drive. These dry weather discharges do not result in non-compliance of the base flow metal TMDL at Campus (based on the reviewed data only). It is unknown if the elevated concentrations measured in the dry weather urban runoff result in exceedance of the CTR criteria in the immediate receiving waters. If flow reductions observed in the intervention watershed are attributable to the ET controllers, then these controllers would help to reduce impacts from any potential exceedances of the TMDL because the discharges would be subject to greater dilution by the in-stream flows.

5.4.4.5 Pathogens

Pathogens are agents or organisms that can cause diseases or illnesses, such as bacteria and viruses. Fecal coliform bacteria are typically used as an indicator organism because direct monitoring of human pathogens is generally not practical. Fecal coliform are a group of bacteria that are present in large numbers in the feces and intestinal tracts of humans and animals, and can enter water bodies from human and animal waste. The presence of fecal coliform bacteria implies the water body is potentially contaminated with human and/or animal waste, suggesting the potential presence of associated pathogenic organisms.

***Fecal Coliform TMDL* –** The RWQCB has adopted phased TMDL criteria for pathogens, with the initial focus on additional monitoring and assessment to address areas of uncertainty. The goal of the Newport Bay TMDL is compliance with water contact recreational standards by 2014:

- Fecal coliform concentration of not less than five samples per 30 days shall have a geometric mean less than 200 MPN/100 ml, and not more than 10 percent of the samples shall exceed 400 MPN/100ml for any 30-day period.

A second goal is to achieve the shellfish harvesting standards by 2020:

- The monthly median fecal coliform concentration shall be less than 14 MPN/100 ml, and not more than 10 percent of the samples shall exceed 43 MPN/100 ml.

The TMDLs are applicable for all flow regimes.

***Study Data Comparison with Fecal Coliform TMDLs* –** Table 5-26 shows the mean and median fecal coliform concentrations measured in the five study watersheds. From 70 percent to 100 percent of all fecal coliform measurements were greater than 400 MPN/ml in all study watersheds. This level of exceedance is substantially greater than the allowable 10 percent. The mean and median fecal coliform concentrations also exceed the 400 MPN/100ml criterion in all study watersheds. There was insufficient data to calculate the 30-day geometric mean (a minimum of 5 samples per 30 days needed). However, the TMDL criterion (30-day geometric < 200 MPN/100 ml) would likely be exceeded, assuming that any additional data would be of the same magnitude as those collected. Exceedance of the TMDL criteria in all study watersheds suggests that urban dry weather runoff is likely a contributing factor to any dry weather exceedance of the TMDL in the receiving waters.

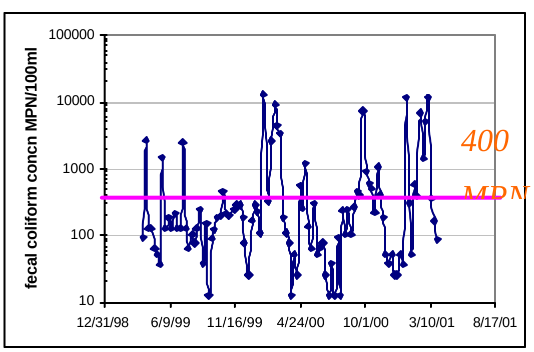
Table 5-26
Mean and Median Fecal Coliform Concentration (MPN/100ml) by Site

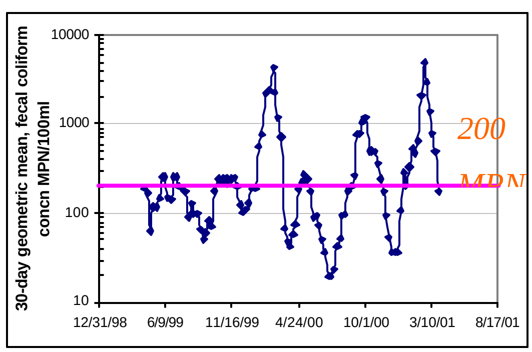
	1001		1002		1003		1004		1005	
	Pre	**Post**	**Pre**	**Post**	**Pre**	**Post**	**Pre**	**Post**	**Pre**	**Post**
All Data										
n	22	24	21	24	23	24	21	24	23	24
Mean	4921	3003	5582	128193	34526	28980	28205	34185	17976	10326
Median	2300	1400	1700	3000	13000	4000	13000	13000	8000	8000
% > 400 MPN/100ml	82%	67%	86%	79%	100%	88%	95%	83%	92%	93%
Subsets[1]										
n	17	14	17	14	18	14	17	14	18	14
Mean	2545	3054	3090	5074	13783	37479	23312	20166	8524	6109
Median	2200	950	1400	1400	8000	2650	8000	6500	4000	2900
% > 400 MPN/100ml	100%	71%	82%	79%	100%	86%	94%	79%	100%	93%

1 – Data subsets with comparable sampling time and seasons. Evening samples were deleted from the pre-intervention data. The post-intervention data include only those data collected in months identical to the pre-intervention period.

Dry weather coliform monitoring information in the Central Irvine Channel was not available. Therefore, it is unknown if elevated fecal coliform concentrations measured in the study watershed contribute to an exceedance of the TMDL in the immediate receiving waters. The OCPFRD has collected dry and wet weather *E. coli* monitoring information in San Diego Creek at Campus Drive (OCPFRD, September 2001), which is considerably downstream from the study watersheds. A plot of the equivalent fecal coliform concentration (assuming an 80 percent *E. coli* content) shows exceedance of the TMDL occurs primarily during the wet season, although dry season exceedances are also evident (see Figure 5-9). This suggests that dry weather urban runoff is potentially a contributing factor to exceedance of the TMDL in dry weather flows at Campus Drive. The ET controllers would reduce the impacts from these potential exceedances if they were determined to be effective in reducing the dry weather runoff volumes.

Figure 5-9
Time Series of Fecal Coliform Levels of San Diego Creek at Campus Drive (converted from measured E. *coli* concentrations)





Median fecal coliform concentrations presented in Table 5-26 may be used to evaluate the influence of the structural and non-structural BMPs. When all monitoring data sets are considered, the median fecal coliform concentrations are equivalent or increase from pre- to post- intervention conditions in all sites except the 1001 (intervention site) and 1003 (a control site). Based on the Mann-Whitney (rank-sum) test, the reduction in median concentrations at Site 1001 and 1003 is significant at the 95 percent confidence level. Thus the site with the irrigation controllers corresponded to a significant reduction in median fecal coliform concentrations, in comparison to two of the three control sites, while the education only watershed exhibited no discernable reduction in median concentrations.

When subsets of the data with similar seasons and sampling times are considered (Table 5-26), there is a decrease in the median fecal coliform concentration at all sites except 1002. However, because of the smaller sample sizes, the decrease is median concentration is statistically significant only at Site 1003. This suggests that there could be seasonal influences in the monitoring data, but the data is not sufficient to determine if there are statistically significant differences in the median concentrations.

5.5 Conclusions

The initial review of water quality data from the study found virtually no difference in concentrations or pollutant flux over time. The technological and education treatments provided essentially no detectable increase or decrease in water quality following the intervention.

The follow-up review utilizing more robust statistical methods on a sample of study data suggests that the interventions did result in changes in water quality. TN levels in the retrofit neighborhood following intervention were found to be significantly lower than levels before intervention, whereas no detectable differences were noted before and after intervention in the education neighborhood. Relatively large observed reductions in TN flux in the retrofit neighborhood could be influenced by seasonal factors, and the extent to which the ET controller contributed to the reduction is unknown. Similarly, although reductions in TP flux were observed in the retrofit neighborhood, the effect of the ET controllers cannot be determined.