C-11.0 WATER QUALITY MONITORING SUMMARY AND ANALYSES

C-11.1 Introduction

This chapter reviews results and findings from the 2004-05 monitoring year of water quality monitoring conducted by the Orange County Stormwater Program under the Third Term Permit, Order No. R9-2002-0001, from the San Diego Regional Water Quality Control Board. The wet and dry weather monitoring program designs are summarized below and described in much greater detail in two reports previously submitted to the Regional Board and available on the Program's website (http://www.ocwatersheds.com/StormWater/swp_documents_intro.asp). These are:

- Past Monitoring, Future Recommendations, and Receiving Waters Monitoring Program, which summarizes cumulative findings from the First and Second Term Permit monitoring programs, and presents the design of the Third Term Permit wet weather monitoring program; and

- San Diego Region Dry Weather Monitoring Program, which details a dry weather reconnaissance program targeted at identifying potential sources of pollution to the stormwater system.

This annual report moves beyond last year's report in that it includes new analysis approaches for estimating annual loads, evaluating recreational impacts in the coastal zone and prioritizing stormdrain outfalls, estimating the degree of unexplained toxicity, and displaying the results of bioassessment monitoring. In addition, the report takes advantage of the three years of bioassessment monitoring to conduct a cumulative analysis investigating the relationship between bioassessment IBI scores and measures of habitat quality (physical habitat, aquatic chemistry, toxicity).

The Third Term Permit monitoring program also represents an important evolution from previous monitoring in terms of its increased focus on ecological conditions in receiving waters, and on potential stormwater impacts in the nearshore coastal zone. Regional efforts are underway, through both the Stormwater Monitoring Coalition (SMC) and the Southern California Coastal Water Research Project (SCCWRP) to develop improved methods for the analysis and interpretation of such data. Future reports will incorporate these methods as they become available.

The following sections review the historical development of the water quality monitoring program (Section 11.2), describe the overall monitoring approach (Section 11.3), summarize monitoring procedures (Section 11.4) and methods of data analysis (Section 11.5), and present the monitoring findings (Section 11.6). The data presented in Section 11.6 are the result of the water quality monitoring conducted from July 1, 2004 to June 30, 2005. More detailed information specific to data from prior years can be found in each of the prior annual reports and the two prior Reports of Waste Discharge.

C-11.2 Program Development

Passage of an amendment to the Clean Water Act in 1987, the Water Quality Act, brought stormwater discharges into the NPDES Program and subsequent EPA regulations required municipal NPDES Permit applicants to develop a management program to effectively address the requirements of the Act.

In response to these regulations, the County of Orange (the Principal Permittee), the Orange County Flood Control District and incorporated cities (all three collectively referred to as Permittees) obtained NPDES Stormwater Permits No. CA 8000180 and No. CA 0108740 (subsequently referred to as the First Term Permits) from the Santa Ana and San Diego Regional Water Quality Control Boards. In 1996, the First Term Permits were replaced by Permits Nos. CAS0108740 and CAS618030 (subsequently referred to as the Second Term Permits). These have recently been replaced by the Third Term Permits.

The overall evolution of the Program's monitoring efforts during this period are illustrated in **Figure C-11.1**. Overall, the Program's evolution is characterized by:

- Continued development of a longer-term perspective for tracking trends in key pollutants and at high-priority locations
- A specific focus on problem areas and issues
- Attention to an expanding set of concerns related to stormwater, e.g., bioassessment, ambient coastal receiving waters.

C-11.2.1 Pre-NPDES water quality monitoring

From 1973 to 1990, the Principal Permittee conducted routine water quality monitoring on drainage facilities which are tributary to water bodies identified as waters of the state by the Regional Boards. The receiving waters were also monitored routinely to assess the chronic effects on established beneficial uses.

When the monitoring program was initiated in 1973, monthly nutrient and trace element sampling was performed at several locations. Sediment samples were collected semiannually to assess the impact of contaminant deposition and adsorption. Additional constituents such as mercury, selenium, DDT, PCBs and radioactivity were also evaluated on a semiannual basis to address public concerns regarding the pollution threat from these constituents.

C-11.2.2 First term permit monitoring under Order 90-38

In order to bring the pre-NPDES water quality monitoring program into conformance with the 1990 federal NPDES regulations and the First Term Permit objectives (Section 11.2), field screening to detect gross contamination was added to the program and the number of sampling sites in the channels and receiving waters were increased in order to better assess the amount and type of contamination in the storm drain system.

The First Term Permit water quality monitoring program consisted of field screening (channels only); dry-weather and storm sampling and a receiving water program.

C-11.2.3 Second term permit monitoring under Order 96-03

While the First Term Permit monitoring program produced useful information, the Permittees recognized (as has the rest of the nation) the high degree of uncertainty regarding the link between urban stormwater runoff and actual impairment of beneficial uses within the aquatic resources of Orange County.

Therefore, in response to the Second Term Permit objectives, the Permittees conducted a systematic re-evaluation of the water quality monitoring program which led to a re-statement of the monitoring program's primary goals. The primary and parallel goals of the monitoring program were re-stated as:

- To determine the role, if any, of urban stormwater discharges in the impairment of beneficial uses; and
- To provide technical information to support effective urban stormwater management program actions to reduce the beneficial use impairment determined to be associated with urban stormwater.

In order to organize the vast array of monitoring activities needed to carry out the objectives and goals, the Permittees identified three separate key elements within the Final Monitoring Program (May 1999).

These three key elements were:

- A focus on known sites (or Warm Spots) where constituents are substantially above system-wide averages;
- A parallel (and somewhat overlapping) focus on areas of critical aquatic concern (herein referred to as critical aquatic resources or CARs); and
- A countywide reconnaissance program to identify specific sources of contamination from sub-watershed areas as well as specific land use investigations in order to evaluate the effectiveness of a variety of BMPs

The monitoring program included an underlying rationale for each monitoring element, a discussion of how monitoring data will be used in decision-making, identification of potential links to other relevant monitoring programs being carried out by other agencies, a description of the basic monitoring design, identification of additional study design steps, and a description of anticipated monitoring activities.

These monitoring elements included many locations from the pre-NPDES and First Term Permit water quality monitoring programs that were of value because of the length of their historical record. Each key element of the Final Monitoring Program contains a description of the monitoring activities proposed to accomplish the objectives described above, as well as a description of the process for making decisions about how the monitoring program will respond to incoming data over time. This process was intended to be used at any time throughout the life of the monitoring program to reevaluate the direction of the program, or to reassess the appropriate allocation of resources within the program.

The second term monitoring program and subsequent elements utilized a five-year timeline (1998-99 through 2002-03) for addressing the goals/objectives associated with each task.

C-11.2.4 Third term permit monitoring under Order R9-2002-0001

In 2002 and 03, the Program completed development of the Third Term Permit monitoring programs for wet and dry weather, respectively. This program extends stormwater monitoring to a broader range of locations and to a wider array of methods for measuring impacts. For example, the Third Term monitoring plan will more completely examine storm drains that discharge directly to the coast and pose a potential health risk to swimmers and bathers. In addition, the new plan for the first time investigates the effects of stormwater plumes on the nearshore marine environment. Inland, the new monitoring plan has expanded to include bioassessment studies of creeks, along with the more consistent use of toxicity testing. Combined with the existing measurement of chemical parameters, this "triad" approach is intended to describe impacts more fully; more accurately identify their sources, and target follow-up studies and BMPs more effectively. Thus, the Third Term Permit monitoring program includes five key elements:

- Urban stream bioassessment monitoring
- Long-term mass loading monitoring
- Coastal storm drain outfall monitoring
- Ambient coastal receiving water monitoring
- Dry weather reconnaissance monitoring.

The overall monitoring approach and methods are summarized in the following sections.

C-11.3 Monitoring Approach

The objectives of the Receiving Waters Monitoring Program, as stated in Attachment B.1 of the Third Term Permit, are to:

- Assess compliance
- Measure the effectiveness of Urban Runoff Management Plans
- Assess the chemical, physical, and biological impacts to receiving waters resulting from urban runoff
- Assess the overall health and evaluate long-term trends in receiving water quality.

The monitoring program meets these objectives (with the proviso that measuring the effectiveness of Urban Runoff Management Plans also requires the implementation of focused evaluations of best management practices (BMPs)) by continuing and expanding the Second Term Permit monitoring emphasis on assessing impacts on aquatic resources, documenting long-term trends in water quality, targeting problematic discharge sites for more focused monitoring, and adding additional monitoring elements. The objectives for each program element are as follows:

Urban stream bioassessment:	Using a "triad" of indicators (bioassessment, chemistry, toxicity), describe impacts on stream communities and the relationship of any impacts to runoff, based on comparisons with reference locations on a year-to-year time frame.
Long-term mass loading:	Using measurements of key pollutants, loads shall decline over a time frame of years to decades, as compared with past and present levels.
Coastal storm drains:	Using a suite of bacterial indicators at high priority drain outfalls, track compliance with regulatory standards and any improvements due to BMP implementation.
Coastal receiving waters:	Using measure of runoff plume characteristics and extent, as well as measures of a suite of physical, chemical, and biological indicators, improve understanding of the impacts of runoff plumes on nearshore ecosystems.
Dry weather reconnaissance:	Using data from both random and targeted sites, define background dry weather conditions as a basis for identifying candidate sites for further focused source identification work.

The monitoring program will reflect the Program's continued evolution toward watershed management. As discussed in the following sections, monitoring sites in the various program elements have been located in specific watersheds, with the goal of improving the ability to understand stormwater processes and manage their impacts in a more functional manner.

C-11.4 Description of Monitoring Procedures

C-11.4.1 Urban stream bioassessment

The Permittees with assistance of Regional Board staff have selected twelve channels and three reference sites to conduct urban stream bioassessments using California Stream Bioassessment Procedure (CSBP) established by the California Department of Fish and Game (DF&G). A contract laboratory conducts the bioassessment sampling and taxonomic analyses on behalf of the Permittees. A description of the CSBP can be found at http://www.dfg.ca.gov/cabw/Field/csbpwforms.html.

In order to conduct the triad analysis, at the time of bioassessment sampling the Permittees collected grab samples for chemical and toxicity analysis. The suite of chemical constituents is the same as analyzed in the Mass Emissions Program. The aqueous toxicity is evaluated using three freshwater organisms, *Ceriodaphnia dubia*, *Selanastrum capricornutum*, and *Hyallela azteca*.

C-11.4.2 Long-term mass loading

The Permittees selected six channels in the San Diego Region to conduct mass emissions monitoring. The selection criteria included the following:

- Classification of the waterbody as a "Water of the State" in the Water Quality Control Plan for the San Diego Region;
- Suitability of the site drainage area to monitor area-wide contributions of storm water pollutant loading;
- Suitability of the site's hydrological characteristics to enable practical measurement of flow and collection of representative storm water samples;
- Maintenance of long-term data collection at appropriate existing monitoring stations (Laguna Canyon Wash, Aliso Creek, San Juan Creek, Trabuco Creek, Prima Deshecha Channel, and Segunda Deshecha Channel);
- Safety from traffic and other hazards;
- Suitable siting for sampling equipment; and
- Crew access for retrieving samples and maintaining equipment during storm conditions.

The Permittees used time-composite sampling as the primary method of monitoring the concentration and load of constituents at their Mass Emissions sites. This type of sampling is conducted with automatic samplers that consist of programmable pumps (peristaltic) which transport water from the channel to a collection reservoir in the autosampler base. The collection reservoir can be a single large composite bottle or a series of up to 24 bottles. The autosampler program can be modified to vary sample volumes and frequency of collection. In the San Diego Region two automatic samplers were used at each Mass Emissions site. One autosampler was used for monitoring water chemistry and the other was used for monitoring toxicity.

To collect samples for the analysis of water chemistry, 8, 1.8-liter glass bottles were used in the autosampler base. The water chemistry autosampler was programmed to collect three discrete samples per 1.8-liter bottle. To collect samples for toxicity testing, a single 5-gallon glass bottle was used in the autosampler base. The two samplers were programmed to collect at the same frequency to maintain the consistency between the composite samples produced by each.

Three storms were monitored at each Mass Emissions site. For each storm the water chemistry was monitored with a series of 3 to 5 composite samples collectively spanning approximately 96-hours. The sampling for toxicity testing was coincident with just one of these composite samples. The Permittees chose the following temporal segments of storms that would be monitored for toxicity.

- Storm 1 – first flush (first hour of storm).
- Storms 2 and 3 – 24-hour period beginning three hours after the initiation of the first flush sampling by the water chemistry autosampler.

During each storm the automatic sampling programs were initiated when the water level in the channel rose above a triggering device (level actuator or flowmeter) hardwired to the respective autosampler. When possible a single triggering device was used to trigger both samplers simultaneously. For the water chemistry sampler (and the toxicity sampler during the first storm) the frequency of collection during the first hour of a storm was set at 1 sample/12 minutes. After the sixth sample is collected at the one-hour mark, the collection frequency is decreased to once every 2 hours. Sampling of water chemistry spans approximately 96 hours to allow comparison of the data to 96-hour guidance criteria for chronic aquatic toxicity from the California Toxics Rule (CTR). The concentrations of dissolved heavy metals in the composite samples can be compared to acute toxicity criteria from the CTR. The concentrations of organophosphate pesticides can be compared to literature values of LC_{50}s for toxicity testing organisms.

Autosampler maintenance is performed periodically during the 96-hour period to change bottles, icepacks, and power supplies.

The first six samples collected during each storm were composited and represented the "first flush". The remaining bi-hourly storm samples were used to prepare composite samples that were representative of the subsequent parts of the storm. Unless a 24-hour composite sample was prepared for comparison to toxicity testing results, the samples beyond the first flush were composited using the stage hydrograph for the channel, or by evaluating the electrical conductivities of the samples in each bottle. Using hydrographs from the Principal Permittee's Automated Local Evaluation in Real Time (ALERT) system, samples collected beyond the first flush and representing the storm peak and recession were composited into a single sample. Storms spanning multiple days were broken up into two or more composite samples.

In the absence of a streamgauge hydrograph for the sampled channel, the conductivity of the samples from each bottle (in order of collection) was measured. Changes in conductivity usually denote the beginning or end of storm runoff. After the "first flush" of a storm, conductivities tend to immediately decrease during the rise of the storm hydrograph and slowly rise after the recession. Sample appearance (turbidity or fluvial sediment) can also be used in the compositing process. Storm samples tend to be more turbid and contain more fluvial sediment. Using these electroanalytical measurements and visual observations as a guide, composite samples were prepared to represent various parts of a storm.

Water chemistry samples were analyzed for pH, electrical conductivity, turbidity, nitrate, ammonia, total Kjehldahl Nitrogen (TKN), phosphate, orthophosphate, total suspended and settleable solids, volatile suspended solids, hardness, organophosphate pesticides, and total recoverable and dissolved copper, chromium, lead, cadmium, zinc, silver and nickel .

Samples for the analyses of dissolved metals were filtered with a 0.45 micron groundwater filtering capsule and then acidified with analytical grade nitric acid before submittal to the contract laboratory.

Toxicity of stormwater runoff samples were evaluated using three toxicity tests with marine organisms. The toxicity due to pesticides was measured using the mysid (*Mysidopsis bahia*) survival/growth test. The toxicity due to dissolved metals was measured using the sea urchin (*Stronglyocentrotus purpuratus*) fertilization and embryo development tests.

Time composite monitoring is supported by the Principal Permittee's precipitation and streamgaging network which consists of recording and/or transmitting ALERT gages. Mechanical recording raingages are weighing bucket type. Accumulated rainfall is recorded in analog format on drum charts. The ALERT precipitation gages are tipping bucket type with dataloggers. Data are recorded and transmitted in digital format; sensitivity is 1 mm (0.04 inches) of accumulated rainfall.

The Principal Permittee uses several types of streamgauges to monitor changes in water level. The oldest design is the stilling well with water level float; the newer types are manometer gages or pressure transducers. Data (water level versus time) are recorded on stripcharts. The ALERT interface to these gages consists of a connection from the recorder chart drive to an ALERT shaft encoder. ALERT information is recorded on a datalogger and transmitted to the Principal Permittee Katella yard base station in digital format. Sensitivity of the transmitted and recorded ALERT record is user-variable with the greatest sensitivity being a change in water level of 0.01 feet.

C-11.4.3 Coastal stormdrain outfall monitoring

The Permittees selected twenty-six coastal stormdrains to monitor the effects of urban runoff on the coastal zone. The following selection criteria were used:

- Outlet of the stormdrain is posted with a warning sign by the Orange County Health Care Agency;
- The stormdrain has an equivalent circular diameter greater than 39-inches or a daily dry-weather, discharge volume exceeding 100,000 gallons; and
- The stormdrain and the surfzone are accessible by monitoring staff.

Monitoring was conducted on both the discharge from the stormdrain and the surfzone 25 yards up-coast and 25 yards down-coast of the stormdrain-ocean interface. Grab samples were collected weekly for the analysis of total coliform, fecal coliform, and Enterococcus bacteria. An estimate of the flowrate from the stormdrain was made and the temperatures of the stormdrain discharge and the surfzone down-coast were measured.

The following criteria were established for monitoring:

- Samples were not collected on the day of rainfall;
- Samples were not collected from a stormdrain during the period when its discharge was diverted to a sanitation district; and
- During stormdrain diversion only a sample from the surfzone (down-coast of the stormdrain-ocean interface) was collected.

The following is a description of the methods used for grab-sample collection and flow estimation.

- Collecting the sample
  - o The sample containers (120-ml plastic bottles) were provided by the contract laboratory. Each bottle contained a small amount of sodium thiosulfate as a preservative.
  - o At each site, bacteriological sample bottles were filled using aseptic technique to avoid contaminating the sample. Samples were collected directly into the sample container to avoid cross-contamination from a transfer device. A fresh pair of powder-free disposable gloves was used at each site.

- The bottles were labeled with a sample ID number prior to collecting the sample. The date, time, and sampler initials were recorded on a logsheet. Sampling staff also recorded any observations that may have an influence on the quality of the sample including the presence of animal or human activity in the area, animal feces, stormwater runoff, etc.

- Samples from the stormdrain were collected a closely as possible to the center of the flow line. For wider channels a telescoping pole was used to collect the sample from the center. To avoid contamination by sediment at the bottom of the storm drain, samples were allowed to flow into the bottles rather than scooping the sample into the bottles. Surfzone samples were collected in ankle deep water. Sample bottles were filled to the bottle shoulder to allow space for mixing. After filling the bottles were carefully capped and placed in an ice-chest for transport to the laboratory.

- The time from sample collection to delivery to the laboratory was kept below six hours.

▪ Temperature measurement was conducted with a calibrated thermometer

▪ Estimating the flowrate was conducted using one of the following methods:

- Measuring the time required for a container of known volume to be filled by the discharge from the pipe or,

- Measuring the cross-sectional area of water in the pipe or drain. If the diameter of the pipe is known the cross-sectional area in ft^2 is

$$Area = R^2 \arccos \frac{R-h}{R} - (R-h)\sqrt{2Rh-h^2}$$

where R is the radius of the pipe, h is the depth of water (all in feet). This cross-sectional area was multiplied by the measured or estimated velocity (ft/sec) to determine the flowrate in ft^3/sec. The velocity was determined using one of the following methods.

  ▪ Using a Global Water Flow Probe, Marsh McBirney Flowmate, etc.

  ▪ Using the static stick method where the velocity of the water is calculated by $v = \sqrt{2gh}$ where v is the velocity in feet per second, h is the velocity head, and g is the acceleration due to gravity (32 ft/sec^2). Velocity head is the difference in the folding scale reading when measuring the depth with the wide edge perpendicular to the flow to that with the edge parallel to the flow. It is also known as the pile-up.

  ▪ Using the floating leaf method where the time required for a floating object to travel a known distance (e.g. 6 feet) is measured.

C-11.4.4 Ambient coastal receiving water monitoring

The monitoring of Ambient Coastal Receiving Waters will be used to evaluate the effect of urban runoff on the ecologically sensitive areas along the Southern Orange County coastline. The monitoring will be conducted in phases in order to establish a priority for future offshore monitoring projects. During the first three years the monitoring consisted of sampling the discharges to these coastal areas. Grab samples were collected using similar methods described in the Coastal Stormdrain Section above. These grab-samples were analyzed for water chemistry and aqueous toxicity. The suite of water quality constituents measured and the types of toxicity tests conducted were identical to those used in the Mass Emissions Program (see above). During the 2004-05 season aerial photography was used after one storm to assess the magnitude of the stormwater plumes from the coastal drains. The size of the plume in each area will be used in the matrix for prioritization.

Dana Point Harbor and Dana Cove are included in the Ambient Coastal Receiving Waters Program. During the second and subsequent years of the permit, monitoring in these areas will include assessments of sediment chemistry, sediment toxicity, and benthic infauna. On a semiannual schedule, benthic sediment will be collected to evaluate concentrations of copper, chromium, cadmium, lead, zinc, silver, nickel, chlorinated hydrocarbon and organophosphate pesticides, herbicides, PCBs, and Polynuclear Aromatic Hydrocarbons (PAHs). Sediment toxicity will be evaluated using the 10-day amphipod (*Eohaustorius estuarius*) survival test. Benthic infaunal analyses will be conducted using the methods developed by the Southern California Association of Marine Invertebrate Taxonomists (SCAMIT).

Benthic samples will be collected using a petite ponar dredge. Samples for benthic infaunal analyses will require five dredge samples per site to approximate the same sampling area used to establish the Regional Benthic Response Index (BRI).

C-11.4.5 Dry weather reconnaissance

The objectives of the Dry-Weather Monitoring Program are to determine the average condition of stormdrain discharges in the San Diego Region of the County, and to identify and eliminate illegal discharges and illicit connections (ID/ICs) to the stormdrain system.

To accomplish the first objective the Permittees established a set of 30 randomly selected stormdrains (random sites) in South Orange County. Each Permittee including the County of Orange has at least one random site within their respective jurisdiction. Each of these 30 sites will be sampled three times during the period from May 1 through September 30 of each year. The data from all of the samplings will be used to establish a database from which the average concentrations of each monitored constituent will be calculated. Monitoring at each site includes insitu measurements of turbidity, pH, temperature, specific conductance, and dissolved oxygen. Chemical measurements in the field include nitrate, ammonia, orthophosphate, total chlorine, phenol, MBAS (surfactants), and water hardness. Grab samples are collected for laboratory analyses of total suspended solids; total coliform, fecal coliform, and Enterococcus bacteria; oil and grease; dissolved metals; and organophosphate pesticides. Flowrate is estimated using the method described in the Coastal Stormdrain Outfall Program above.

In order to accomplish the second objective, the Permittees established a list of 26 "targeted" stormdrains in which ID/ICs were suspected. A statistical analysis of the data from the sampling of the random stormdrains will be used to establish the triggers for initiating reconnaissance for source identification in the watersheds of the targeted drains. The targeted drains will be sampled five times during the period between May 1 and September 30 of each year. Reconnaissance will be triggered if the results from two successive samplings at a random or targeted site exceed the upper bound of the tolerance interval of the random site data. For dissolved oxygen, two successive values below the lower bound of the tolerance interval would trigger a source investigation.

C-11.5 Methods of Data Analysis

C-11.5.1 Comparison to water quality guidance

Acute (CMC-Criteria Maximum Concentration) and chronic (CCC-Criteria Continuous Concentration) aquatic toxicity criteria from the CTR were used as guidance to evaluate dissolved metals data collected from storm channels and harbors. Water quality criteria from the CTR for both freshwater and saltwater are found in **Table C-11.1** and for sediment from other sources in **Table C-11.2**.

California Water Code Section 13170 authorizes the State Water Resources Control Board (SWRCB) to adopt water quality control plans for waters where standards are required by the Federal Clean Water Act (CWA) and its 1987 amendments, the Water Quality Act (WQA). According to Section 303(c)(2)(B) of the CWA, these plans must contain water quality objectives for priority pollutants that could be reasonably expected to affect the beneficial uses of the waters of the State.

On March 2, 2000, the State adopted the United States Environmental Protection Agency's (USEPA) Rules establishing numeric water quality criteria for priority toxic pollutants (commonly referred to as the CTR) for the State of California. The CTR sets criteria for dissolved heavy metals in freshwater that are based on water hardness and separate criteria for saltwater. The dissolved metals data were compared to the acute and chronic criteria for guidance purposes.

According to the CTR, for waters with a hardness of 400 mg/l or less as calcium carbonate, the actual ambient hardness of the surface water shall be used in those equations. For waters with a hardness of over 400 mg/l as calcium carbonate, a hardness of 400 mg/l as calcium carbonate shall be used with a default Water-Effect Ratio (WER) of 1, or the actual hardness of the ambient surface water shall be used with a WER. For this reporting period the former method was used.

In applying the CTR criteria to freshwater, if the time period to which the guidance applies is less than the length of the sampled period, a measured concentration greater than that guidance value will constitute an exceedance. For example, if the 1-hour guidance for lead (at a hardness of 100 mg/L as $CaCO_3$) is 65 µg/L, a concentration of 68 µg/L during a 24-hour period will be considered an exceedance of the guidance criterion.

In computing the mean concentration during a sampled period with multiple composite samples, values below the detection limit were assumed to be zero. This assumption allows for a more consistent evaluation from year to year as detection limits are lowered with alternative methods of analysis or new technology. The assumption also gives greater confidence to a designation of an exceedance of a guidance criterion as it reduces the likelihood that the exceedance was caused by an erroneous estimation of a non-detected value. During the latter part of this monitoring year, a new analytical services contract was established which required the laboratories to report lower detection limits for metals in freshwater and saltwater.

With respect to the saltwater guidance from the CTR, the average concentrations of dissolved metals in depth-integrated samplings from each 4-day storm monitoring of the Harbors and Bays were compared to the 4-day guidance criteria. The dissolved metals concentrations in each grab sample were compared to the 1-hr acute toxicity guidance criteria. There is no chronic guidance criterion for silver so only the acute criterion was used. Since total chromium was analyzed only the criteria for trivalent chromium (Chromium III) were used.

C-11.5.2 Toxicity testing

Toxicity tests span varying time periods depending on the type of organism function (survival, growth, reproduction, etc.) being evaluated. Endpoint data are used to compute statistics that can be compared against regulatory criteria. These statistics include Acute Toxicity Units (TUa) and Chronic Toxicity Units (TUc).

The concentration that causes 50% mortality of the organisms (the median lethal concentration, or LC_{50}) is calculated from the data for 96 hours (96-hour acute LC_{50}) and for day seven (seven-day chronic LC_{50}) using USEPA methods. The LC_{50} values are point-estimates expressed as "percent sample;" the lower the LC_{50} percentage the more toxic the sample. For acute regulatory standards, the LC_{50} acute value is used. For chronic regulatory standards, the seven-day chronic effects are estimated using the NOEC, or No Observed Effect Concentration, for both survival and reproduction. This is the highest concentration tested in which there was no statistically significant effect on the survival or reproduction compared to the control response. The lower the NOEC, the more toxic the sample.

For purposes of assessment between sites or between samplings, the endpoints described above are transformed into toxic units (TU). Toxic units are further divided into toxic units acute (TUa) and toxic units chronic (TUc) for acute and chronic endpoints, respectively. As toxicity increases, the toxic units increase.

TUa and TUc values are calculated very differently and are not interchangeable or related. The TUa equals 100/96-hr acute LC_{50}. If the LC_{50} is greater than 100%, then the TUa is calculated by the following formula:

TUa = log(100-S)/1.7 where S = percentage of survival in 100% sample. If S > 99%, the TUa is reported as zero, which is the lowest TUa value possible. The percent survival in the 100% concentration used in this formula is expressed as a percentage of the control survival. The TUc equals 100/NOEC. The lowest TUc possible, which indicates no toxicity, is 1. TUc values were calculated separately for survival and reproduction endpoints.

For some tests, if the test data meet acceptability criteria, inhibition concentrations, an IC_{25} and an IC_{50}, are calculated. These are the concentrations that cause a 25 percent or 50 percent inhibition of an organism's function such as growth, or cell density, in the Selanastrum test.

A reference toxicant test is also run to establish whether the test organisms used fall within the normal range of sensitivity. The reference toxicant test is conducted with known concentrations of a given toxicant (e.g., copper sulfate is used for *Ceriodaphnia*). The effect on the survival and reproduction of the animals is compared to historical laboratory data for the test species and reference toxicant. If the values are within two standard deviations of the historical average, the test organisms are considered to fall within the normal range of sensitivity.

Standard operating procedures for each of the specific tests conducted for both marine and freshwater organisms are detailed in **Attachment C-11-I**.

For toxicity tests conducted as part of the mass loads and ambient coastal program elements, available LC_{50} and EC_{50} data on key contaminants were used to compare the observed toxicity (measured as toxic units) to the expected toxicity. This analysis focused on the mass loads and ambient coastal program elements because toxicity was rarely observed in the bioassessment monitoring. The toxicity testing organisms used in this Program tend to be more sensitive to some categories of toxicants than others. For example, the *Mysidopsis* survival/growth (MSG) test tends to be very sensitive to OP pesticides and ammonia but less sensitive to metals. The Sea Urchin Fertilization (SUF) test is sensitive to dissolved metals and ammonia but not very sensitive to OP pesticides. The calculation of the predicted toxicity for each test reflects these sensitivities in that only the impact due to metals and ammonia is evaluated in the SUF test and only the impact due to OP pesticides and ammonia is evaluated in the MSG test.

LC_{50} data for the *Mysidopsis bahia* 96 hour survival test for ammonia, chlorpyrifos, diazinon, dimethoate and malathion were obtained from the PAN Exotoxicity database http://www.pesticideinfo.org/Search_Ecotoxicity.jsp which contains the results of over 220,000 toxicity tests. Results can be sorted by species, chemical or effect. Additional data were obtained from SCCWRP research studies. EC_{50} data for the sea urchin 40 minute fertilization test for ammonia, copper, and zinc were obtained from the same sources. The observed concentration of each chemical constituent (from the aquatic chemistry samples collected at the same time) was divided by the appropriate LC_{50} or EC_{50} value to produce an estimated TU_a from each constituent. These estimated TU_as were then summed and compared to the observed TU_a from the toxicity test, as in the following equations:

$$\frac{\text{Concentration of toxicant}}{\text{Average literature value of } LC_{50} \text{ or } IC_{50} \text{ of toxicant}}$$

The total predicted toxicity from n toxicants is $\sum_{i}^{n} \frac{[toxicant_i]}{[LC_{50} or IC_{50}]_i}$

The calculated TUa from the toxicity test can be compared to this predicted toxicity.

This approach to comparing observed and predicted toxicity has potential shortcomings, including:

- The lack of availability of relevant LC_{50} and EC_{50} data for the full range of chemical constituents of concern
- The implicit assumption of simple additivity of toxic effects. While probably not true, there is no clear guidance on how to accurately represent synergistic effects, which could very well vary from site to site and over time
- The fact that the predicted toxicity in several instances is larger than the observed toxicity, which serves to weaken confidence in the reliability of the LC_{50} and EC_{50} data.

Despite these shortcomings, this approach is useful for:

- Assessing the overall accuracy or reliability of the toxicity results
- Identifying specific chemicals that appear to contribute most to toxicity and that are therefore targets for further study and/or source identification and reduction efforts
- Identifying monitoring locations that may have consistently high levels of unexplained toxicity. In these cases, more sophisticated studies may be called for.

C-11.5.3 Bioassessment and Index of Biotic Integrity (IBI)

A complete description of methods for calculating the Index of Biotic Integrity for each site is contained in the annual report of the bioassessment monitoring, posted on the Program's website at http://www.ocwatersheds.com/StormWater/swp_documents_intro.asp. In brief, each site is evaluated in terms of a series of metrics (**Table C-11.3**), which are then scored (**Table C-11.4**) to provide a basis for determining the IBI scores themselves for each site. These scoring ranges are based on data from the southern California region, from southern Monterey County to the Mexican border. The refined southern California IBI is more sensitive than the preliminary IBI, particularly for sites in the Good and Very Good range. The new scoring ranges differ from those used in earlier years, which were based on data from San Diego County and reflect conditions only in streams in that region. The use of the more broadly applicable IBI follows the California Department of Fish and Game protocol, which continues to evolve. In addition, the Stormwater Monitoring Coalition is undertaking a project to develop a further improved IBI, or set of IBIs, representative of conditions throughout the entire southern California region. Thus, the IBI scores presented here may continue to shift somewhat in the future.

C-11.5.4 Evaluation of triad data

Evaluation of triad data (i.e., bioassessment, water chemistry, toxicity) was based on the framework developed by the Stormwater Monitoring Coalition's Model Stormwater Monitoring committee. This approach, which is described in detail in the SMC's report to the State Water Resources Control Board (ftp://ftp.sccwrp.org/pub/download/PDFs/419_smc_mm.pdf), is based on a weight of evidence approach that compares each of the three legs of the triad against each other. **Table C-11.5**, drawn from the SMC's report, summarizes the types of conclusions that can be drawn from various combinations of triad results. Thus, there is no routine or standard method for evaluating triad data. However, the triad data from the bioassessment stations for the most part led to relatively clear interpretations of causal factors for observed conditions.

Two additional analyses are included in this year's report to more thoroughly examine the relationships among the three legs of the triad. (In actuality, there are four legs if the physical habitat data collected as part of the bioassessment protocol are considered separately from the biological community data.)

For the first analysis, thresholds were established for each of the four data types (IBI, physical habitat, aquatic chemistry, and toxicity) in order to divide the range of values for each data type into four categories representing conditions from excellent to poor. IBI and physical habitat categories were based on the Fish and Game interpretation framework for these data types. Aquatic chemistry thresholds focused on dissolved metals. At each station, the total number of CTR exceedances at each sampling time was divided by the total number of constituents with relevant CTR criteria, resulting in a proportion for each station between 0 and 1.0. The exceedance proportion for each station was then indicated on a map of the sampling sites, according to the following color scheme:

- Green: 0 - < 0.14
- Blue: 0.14 - < 0.40
- Yellow: 0.40 - < 0.75
- Red: 0.75 – 1.0

Toxicity categories were based on the number of toxicity tests that showed toxicity above 25% mortality in the 100% dilution or, for Selenastrum, if the cell count in the 100% dilution was 2.5 times greater than the control. For each site, icons on a map of the monitoring sites representing the four data types were then colored green, blue, yellow, or red to summarize the overall range of conditions at each site.

For the second analysis, all data from the first three years of bioassessment sampling were analyzed for spatial and temporal patters in the benthic invertebrate community. These patterns were then compared to potential explanatory variables (physical habitat, aquatic chemistry, toxicity) to identify potentially causative relationships among the different data types. Two methods were used to describe spatial and temporal patterns in the benthic invertebrate community: cluster analysis and two-way coincidence tables.

Cluster analysis defines groups of stations with similar community composition. The results are displayed in a hierarchical tree-like structure called a dendrogram. On the dendrogram, two groups are first defined, and within these groups subgroups are defined. Subsequently, subgroups within the subgroups are defined. This process is continued until all stations are a separate subgroup. The hierarchical nature of the dendrogram allows the analyst to choose groups of stations that represent a scale of community differences relevant to the present project. Cluster analysis is also used to define groups of species that tend to have similar distributional patterns among the stations.

A two-way coincidence table is the station-species abundance data matrix displayed as a table of symbols indicating the relative abundances of the species at the stations. The rows and columns of the table are arranged to correspond to the order of stations and species along the respective station and species dendrograms. Since similar entities (stations or species) will tend to be closer together along a dendrogram, the row and column orders will efficiently show the pattern of species over the stations and station groups.

Since the rows and columns of the two-way coincidence table are ordered according to the dendrograms, the two-way coincidence table is also used to help delimit the station and species groups defined by the cluster analyses. At each potential separation of subgroups defined by the dendrogram, the two way coincidence table is examined to see the corresponding group differences in terms of species presences and abundances. This allows the analyst to choose groups with a level of community differences consistent with the goals of the project.

The methods discussed above are described only in very general terms. The specific steps included:

- Preliminary biotic data transformation, using a square root transformation and standardization by species mean of values >0 (Smith, 1976; Smith et al., 1988) [1]

[1] Smith, R.W. 1976. Numerical Analysis of Ecological Survey Data. PhD thesis, Univ. of S. Calif., Los Angeles. 401 pp.
Smith, R.W., B.B. Bernstein, and R.L. Cimberg. 1988. Community-Environmental Relationships in the Benthos: Applications of Multivariate Analytical Techniques. Chapter 11 In: Marine Organisms as Indicators. Springer-Verlag. New York: 247-326.

- Calculation of a Dissimilarity Index for cluster analysis of stations, using the Bray-Curtis Index, step-across procedure for dissimilarity >.8 (Bradfield and Kenkel, 1987; Clifford and Stephenson, 1975; Smith, 1984; Williamson, 1978)[2]
- Calculation of similarities for cluster analysis of species, using flexible clustering (β=-.25) (Clifford and Stephenson, 1975; Lance and Williams, 1967; Smith, 1982)[3]
- Creation of the two-way coincidence table (Kiddawa, 1968; Smith, 1976)[4].

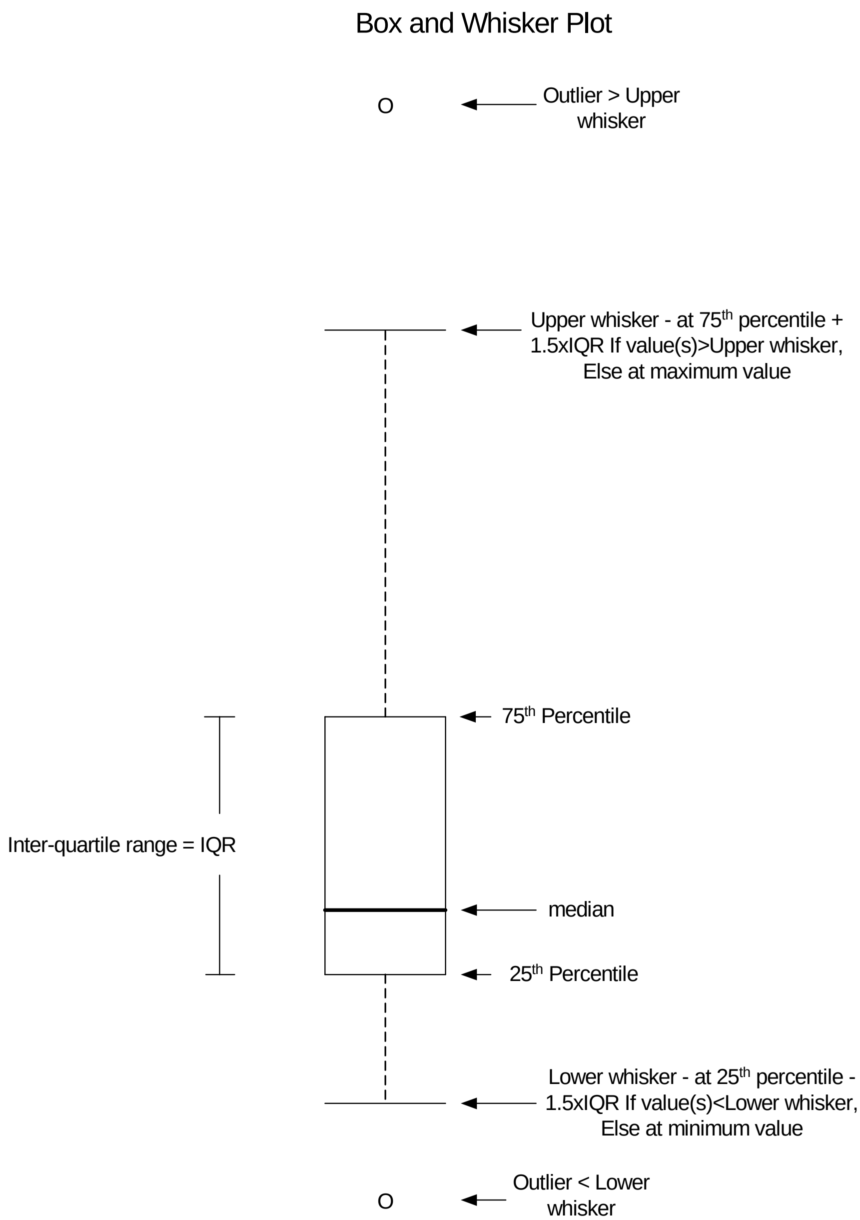
The distribution of the external parameters measured at each station/survey is described with box and whisker Plots (Tukey, 1977)[5], as illustrated in the example below:

[2] Bradfield, G.E. and N.C. Kenkel. 1987. Nonlinear ordination using shortest path adjustment of ecological distances. Ecology 68(3): 750-753.
Clifford, H.T. and W. Stephenson. 1975. An Introduction to Numerical Classification. Academic Press, New York: 229 pp.
Smith, R.W. 1984. The re-estimation of ecological distance values using the step-across procedure. EAP Technical Report No. 2. Email: rs@robertsmith.net.
Williamson, M.H. 1978. The ordination of incidence data. J. Ecol. 66: 911-920.

[3] Clifford, H.T. and W. Stephenson. 1975. An Introduction to Numerical Classification. Academic Press, New York: 229 pp.
Lance, G.N., and W.T. Williams. 1967. A general theory of classificatory sorting strategies. I. Hierarchical systems. Computer J. 9: 373-380.
Smith, R.W. 1982. Analysis of ecological survey data with SAS and EAP. Proc. 7th Annual SAS Users' Group International (SUGI). SAS Institute Inc. P.O. Box 8000, Cary NC 27511: 610-615.

[4] Kikkawa J. 1968. Ecological association of bird species and habitats in Eastern Australia; similarity analysis. J. Anim. Ecol. 37: 143-165.
Smith, R.W. 1976. Numerical Analysis of Ecological Survey Data. PhD thesis, Univ. of S. Calif., Los Angeles. 401 pp.

[5] Tukey, J.W. 1977. Exploratory Data Analysis. Addison-Wesley, Menlo Park, CA. 506 pp.



C-11.5.5 Mass load calculations

Mass loads were calculated using chemical and hydrographic data. Water level records from permanent streamgauging stations at or near the sampling site were processed using Hydstra and XstreamMeasures software. Water levels from the station's continuous stripchart recorder were digitized and converted to discharge rates using stage-discharge relationships (channel ratings). At sites which had ISCO water level recorders, the dataloggers were downloaded periodically and the information was stored in Hydstra. Using the respective rating tables for each site, the water level data were converted to flow rates. The total discharge in acre-feet during each sampled period was computed. By multiplying the total water discharge per sampled period by the pollutant concentration of the composite sample from the period and applying the proper conversion factors (acre-feet to lbs. of water), a mass load in pounds or tons of contaminant was calculated. For data reported as ND (non-detected), one-half of reported laboratory detection limits were used in the calculations.

Event mean pollutant concentrations were calculated to produce a site mean EMC that could be used in the estimation of the mass loads from unsampled storms. To calculate the EMC of a monitored storm the sum of the mass load from each composite sampling during a storm was divided by the total sampled volume of water during the same period. After applying the appropriate conversion factors, an event mean concentration in mg/L or μg/L was calculated. The site-mean EMCs were updated each year with the EMC data from that year.

Mean EMCs were used to estimate mass loads from un-sampled storms during the monitoring year in two distinct ways. The first estimates total annual loads on a site-by-site basis and the second on a watershed basis. In the first approach, an average site EMC for each stormwater contaminant was calculated by simply averaging the EMCs over the sampled storms. These site- mean EMCs were then used to estimate mass loads from un-sampled storms. To estimate these mass loads, the site mean EMC for a stormwater contaminant from a particular station was multiplied by the total annual volume of water discharged during un-sampled storms, and the appropriate unit conversion factors [2.718 liter · lbs/mg · ac-ft]. In the second approach, the watershed load was calculated by simply summing the total estimated annual loads from each monitoring site in the watershed. Only EMCs in which the 75-120% of the total storm runoff volume was sampled were used in these calculations.

C-11.5.6 Evaluation of coastal stormdrain water data

Coastal stormdrain data consist of temperature measurements and concentrations of bacterial indicators in the discharge and upstream and downstream of larger flowing stormdrains. Data analysis consisted of:

1. Comparing indicator levels at each drain to the state's AB411 standards
2. Ranking drains in terms of the proportion of total possible exceedances of the AB411 standards
3. Plotting indicator levels in the receiving water vs. those in the drain
4. Ranking drains in terms of the slope of the linear regression of receiving water indicator levels vs. those in the drain.

These analyses were performed for the entire year and for the AB411 season alone. In addition, analyses also focused on only those instances where field notes indicated that the outflow of a drain was flowing to the surfzone. However, these field notes did not consistently capture the status of each drain's flow. Analysis #4 should therefore be considered preliminary for this monitoring year. Field procedures have been adjusted to ensure that this information will now be systematically captured. The following paragraphs describe methods for analyses #'s 2 – 4.

For analysis #2, the actual number of receiving water samples collected at each drain throughout the year was summed. This did not always equal 312 (i.e., 52 weeks x 3 indicators x 2 locations) because it was not possible to collect the full suite of samples at each site throughout the entire year. The total number of AB411 exceedances was then divided by the total number of samples, resulting in a proportion for each drain between 0 and 1.0. The exceedance proportion for each site was then indicated on a map of the sampling sites, according to the following color scheme:

- Green: 0 - < 0.14
- Blue: 0.14 - < 0.40
- Yellow: 0.40 - < 0.75
- Red: 0.75 – 1.0

For analysis #3, the receiving water values for each indicator were plotted vs. the indicator values in the drain during the same sampling event, with receiving water values on the y-axis and drain values on the x-axis. Separate plots are presented for each indicator at each drain, with upstream and downstream data displayed with distinct symbols. The plots are divided into sectors suggesting the conclusions and possible management actions that would be appropriate when a preponderance of the data points fall into one sector or another.

For analysis #3, data were log transformed and then a standard least squares linear regression calculated for relationship between receiving water indicator values and drain values. A separate regression was calculated for each indicator / drain combination. Sites were then ranked in terms of the "p" value for the regression for each indicator. The "p" value reflects the strength of the drain – receiving water relationship. In combination with the other analyses, this can be used to help assess each drain's likely effect on receiving water conditions.

Analysis results were then evaluated to identify consistent spatial and temporal patterns. Drains with exceedance and/or regression ranks were evaluated more carefully to identify potential explanatory factors in their drainage areas.

C-11.5.7 Evaluation of ambient coastal receiving water data

The ambient coastal receiving water data were compared to marine CTR values and ranked in terms of their relative degrees of contamination. In addition, toxicity test results were compared to chemistry samples to identify potential explanations for any observed toxicity. Subsequent analysis will involve a qualitative assessment of the receiving water environment around each discharge in terms of its ability to assimilate runoff, the presence of other sources of contamination, and the presence of sensitive marine resources. This information will be used to arrive at relative rankings of the degree of runoff risk to each site, which will then provide a basis for prioritizing further studies of stormwater plume extent and impact.

C-11.5.8 Prioritization of dry weather sites for source identification

Only a single sampling of the random dry weather sites was conducted during this monitoring year. Because two consecutive "hits" of elevated levels are required to identify a site for more intensive source identification work by the relevant city, it was not possible to carry out this prioritization analysis. Data for this single sampling event are simply reported and the highest 10% of values identified.

C-11.6 Analysis of Data

The following sections present data summaries and interpretations for each of the major monitoring program components.

C-11.6.1 Urban stream bioassessment

Figure C-11.2 displays the bioassessment monitoring sites, which are sampled twice each year, in fall and spring. **Figures C-11.3** and **C-11.4** present the IBI scores for each bioassessment monitoring site (**Table C-11.6**).The urban affected sites in the study region had IBI ratings of Poor to Very Poor in the fall of 2004 and Fair to Very Poor in the spring of 2005. The reference sites had ratings that ranged from Poor to Very Good. As in previous years, reference site REF-CS, which was rated Poor in both surveys. The rating for this reference site has gradually declined over the three years of the program. It rated Fair in both the fall 2002 and spring 2003 surveys, and Fair in the fall 2003 survey and Poor in the spring 2004 survey. Despite these relatively low ratings, the community at this site also has some characteristics of undisturbed conditions, such as higher abundances of pollution intolerant taxa.

While the IBI rating of several sites remained consistent across the two surveys, the rating for others shifted somewhat. Overall, there was much more consistency among site scores in the fall 2004 survey, with two reference sites having very high scores and all other sites having scores in the Poor and Very Poor range. At one extreme, the IBI score for site SC-MB dropped from the high portion of the Very Poor range in fall 2004 to the very low portion of this range in spring 2005, when it was the lowest rated site. This was due to the loss of all but one taxon. Several other sites showed substantial reductions from the fall to the spring surveys. This may have been due to the extensive scouring and other habitat disturbance resulting from last winter's intense storms.

With only one exception (CC-CR in the spring survey), all of the non-reference sites were rated either Poor or Very Poor. CC-CR was also the highest rated non-reference site in last year's survey. In general, the IBI rankings primarily reflect the degree of habitat modification due to watershed urbanization. Thus, despite sometimes substantial variation from one survey to the next, stations are relatively consistent in terms of their overall IBI rank (**Figure C-11.5**). Overall, stations further downstream tend to have lower IBI scores than stations further upstream (**Figures C-11.5** and **C-11.6**), which reflects the pattern of development with denser development closer to the coast.

In addition to describing patterns and trends in benthic invertebrate, a further purpose of the bioassessment program element is to determine whether physical habitat, aquatic chemistry, and/or toxicity are correlated with IBI scores. If strong correlations exist, then this would suggest a causal relationship. Three approaches were used to search for such correlations and possible causal relationships.

First, broad patterns for each of the four types of indicator (i.e., IBI, physical habitat, aquatic chemistry, toxicity) were mapped. **Figures C-11.6a** and **C-11.6b** show that there are no clear relationships at this broad scale between IBI scores and any other type of variable. Thus, sites with poor overall IBI condition did not also have poor scores on either physical habitat, toxicity, or aquatic chemistry.

Second, and at a greater level of specificity, the detailed monitoring data for bioassessment, aquatic chemistry, and toxicity were examined to determine whether there are any clear relationships among these. Toxicity data (**Table C-11.7**), from both the fall 2004 and spring 2005 sampling periods, show that toxicity occurred in only two instances, both with the chronic *Cerioidaphnia* survival and reproduction test, and both at site SD-AP (Segunda Deschecha). The spring 2005 sampling at this site showed extreme toxicity in the survival test (100% mortality in the undiluted sample). The chemistry data for that date do not suggest a cause of the high toxicity. Quite possibly a carbamate pesticide such as Sevin may be the cause.

Overall, this is a lower level of toxicity than observed in the past two years, where at least two and as many as five sites showed elevated toxicity. SD-AP had the highest levels of conductivity (a surrogate for total dissolved solids-TDS) by far (**Table C-11.8**). The slight effect observed in the *Ceriodaphnia* reproduction test in the fall 2004 sampling may have been a function of the high TDS as these organisms show sensitivity to conductivities greater than 3000 µmhos. (Previous Program studies have shown that the high conductance in both Prima and Segunda Deshecha Channels is due to natural groundwater seeps in the channel walls.) Note that the Salt Creek site (SC-MB) had electrical conductivity measurements in the same range as Segunda Deshecha and the fall 2004 and spring 2005 *Ceriodaphnia* reproduction tests also showed also showed a slight toxic response. At this site however, Diazinon (125 ng/L) was detected in the fall 2004 sampling and *Ceriodaphnia* are sensitive to this organophosphate pesticide.

The absence of any consistent strong toxicity signal in this year's data suggests that toxicity is not a cause of lower IBI scores.

These two analyses, based on subjective comparison among the triad legs, suggest none of the other legs, physical habitat, aquatic chemistry, or toxicity are strongly correlated with the bioassessment leg, at least during this monitoring year. This hypothesis was tested in greater detail with a cluster analysis of the species data from all sites and all years. The values of the other triad legs within each site group were then examined for relationships with the biological patterns.

As a first step, the species data from all surveys was clustered to identify groupings of sites that were similar in terms of their community composition. **Figure C-11.7** shows the cluster analysis of all sites over the three years of surveys and **Figure C-11.8** the two-way coincidence table of the relative distribution of species in each site at each sampling time. Horizontal and vertical lines on the two-way coincidence table identify major groupings of species and sites, respectively. (Sites are identified by their site number, year of sampling, and month of sampling. Relative species abundances are shown as symbols. The abundance of each species was standardized in terms of its maximum at each site over all surveys. Smaller symbols represent a lower proportion of maximum abundance and larger symbols a larger proportion.)

These two figures clearly show three dominant patterns. First, reference sites are concentrated at the upper end of the dendrogram, which is equivalent to the left side of the two-way coincidence table. Second, fall and spring samples tend to group together within small subdivisions of the larger site pattern. This reflects a consistent difference in benthic communities between these two sampling times. Third, species with broader distributions across sites and times are concentrated in the lower half of the two-way coincidence table. Species with such broad distributions tend to be more pollution and/or disturbance tolerant. In contrast, species in the upper half of the two-way coincidence table have much more restricted distributions and in fact are found primarily at the reference sites. The two-way coincidence table clearly illustrates the nature of the seasonal difference. The first site group, which is made up of reference sites sampled during the fall, is dominated by species in the first species group. The second site group, in contrast, which is made up of reference sites sampled during the spring, is dominated by species in the second species group. Because such seasonal differences were so consistent, further analyses of the bioassessment results were separated by season.

Separate cluster analyses and two-way coincidence tables were created for spring and fall (**Figures C-11.9 – C-11.12**. These analyses identified station groupings within each season that were consistently different in terms of their species composition. For example, both show a set of sites populated primarily by tolerant species with wide distributions and another set of sites with much higher populations of species with more restricted distributions. This latter set of sites is made up almost exclusively of reference sites.

Variables were then grouped into biological parameters (e.g., numbers of taxa, magnitude of toxicity), physical habitat parameters (e.g., elevation, bank stability), water quality parameters (e.g., pH, dissolved oxygen), and potential pollutant parameters (e.g., copper, Diazinon). The values of each parameter were then plotted for each site group (**Figures C-11.13** and **C-11.14**), using box and whisker plots. “Group” on the y-axis of the box and whisker plots refers to the site groups from the dendrograms and two-way tables.

The box and whisker plots (**Figures C-11.13** and **C-11.14**) document that a handful of variables show consistent differences among the site groups and are therefore possible causes, or at least strong correlates, of the differences in community composition and IBI scores. In both spring (**Figures C-11.13a**) and fall (**Figure C-11.14a**), IBI score is strongly correlated with number of taxa. The two-way tables (**Figures C-11.10** and **C-11.12**) show that the reference sites near the left side of the tables have larger numbers of species. This is because they generally contain populations of both tolerant (widely distributed) and intolerant (narrowly distributed) species. However, toxicity did not differ in any consistent way across the site groups, in either spring or fall.

The pattern of physical parameters across site groups (**Figures C-11.13b** and **C-11.14b**) varies depending on the parameter. This is why the overall physical habitat score was not related to IBI score in **Figure C-11.6**. A subset of physical habitat parameters differs markedly across the site groups. In general, in both spring and fall, reference sites had much higher scores (with higher scores indicating more favorable conditions) for sediment deposition, elevation, riparian vegetation zone, and channel alteration. In addition, a further set of physical habitat parameters had higher values at reference sites in the spring but not in the fall. These included embeddedness, vegetation cover, and riffle frequency.

Only two water quality parameters differed consistently across the site groups (**Figures C-11.13c** and **C-11.14c**). Specific conductance and TKN (total Kjeldahl nitrogen) tended to be lower at reference sites. Chlorophyll a was higher at the sites with the lowest IBI scores, but not dramatically so. None of the potential pollutant parameters (**Figures C-11.13d** and **C-11.14d**) displayed any consistent differences across the site groups.

Viewed overall, the three years of bioassessment monitoring show that there are persistent differences in benthic invertebrate communities between seasons, irrespective of where the sites are located in the watershed. The data also show that there are persistent spatial differences between sites higher up in the watershed, which tend to be reference sites, and those lower in the watershed, which tend to be much more subject to urban influences. Toxicity and urban pollutants do not appear to be strongly related to either of these patterns, while some aspects of physical habitat and general water chemistry are so related. A preliminary conclusion evident from these results is that physical habitat disturbance is the primary explanation for low IBI scores in this area. However, it is also true that aquatic chemistry and toxicity samples taken at only two times in each year are essentially snapshots that may not reflect longer-term conditions affecting the benthic community. On the other hand, the program now has a series of six sampling events over three years that tend to support this conclusion.

C-11.6.2 Long-term mass loading

Mass loading monitoring is conducted for a wide range of constituents at the stations shown in **Figure C-11.15**. The intent is to monitor each station during three periods of stormwater runoff.

Water chemistry data from mass emissions stations were used to calculate loads and to assess water quality with respect to applicable acute and chronic toxicity criteria from the CTR. The calculation of loads also requires accurate stream discharge records. During the past year, one of the streamgauges malfunctioned, and because of channel reconstruction the record at another was limited to only a small portion of the year. The program does not yet have the ability to describe and track longer-term trends in loads at the six mass loading stations.

Table C-11.9 contains the measured stormwater mass loads of nutrients and dissolved metals at San Juan, Prima Deshecha, Segunda Deshecha, and Aliso Creek Channels. The corresponding flow-weighted event mean concentrations of these constituents were calculated and are presented in **Table C-11.10**. The concentrations of dissolved metals in each composite sample collected in the Mass Emissions program element were compared to the acute toxicity criteria from the CTR. **Table C-11.11** presents all of these data highlighting those which exceeded the criteria. **Table C-11.12** is a summary of the comparisons to the CTR criteria. Exceedances of the freshwater criteria were infrequent with only 4 of 56 samples exceeding the acute cadmium criteria. These exceedances however all occurred at Prima Deshecha Channel. The chronic toxicity criteria for cadmium in freshwater were exceeded in each of the two sampled storms at Prima Deshecha Channel (PDCM01) and at one of the three storms sampled at Segunda Deshecha Channel. The chronic toxicity criterion for nickel was exceeded during one storm at Prima Deshecha Channel. The acute toxicity criterion for dissolved copper in saltwater was exceeded most frequently with 33 of 567 samples showing concentrations higher than the criterion. As in the previous two years, Prima Deshecha Channel showed the greatest number of exceedances of CTR criteria for saltwater.

The toxicity results (**Table C-11.13**) show substantial toxicity (8 toxic units or above) at Prima Deschecha Channel on January 28, 2005. The Sea Urchin fertilization and Red Abalone larval development tests each measured 16 chronic toxicity units. The dissolved zinc concentration in that sample was very high (170 µg/L) and may have caused the effect. The Mysid survival and growth tests both measured over two acute toxic units. Although there was Diazinon (72.9 ng/L) in the sample, the amount would not account for such a high mortality (95%) in the undiluted sample. The toxicity most likely was caused by an unmeasured toxicant, possibly a pesticide. Samples collected from this channel on October 18, 2004 also showed strong toxicity in the Mysid survival and growth tests. High levels of Diazinon (245 ng/L) and Malathion (1280 ng/L) may have contributed to these results.

The first flush storm sample collected on October 18, 2004 from San Juan Creek showed high toxicity in the Red Abalone Larval Development test, and the Mysid survival/growth tests. The results of the Abalone test can possibly be explained by the high concentrations of total recoverable metals in the sample (copper-160 µg/L and zinc-600 µg/L). The results of the Mysid tests are most likely due to the presence of Diazinon (355 ng/L) and Malathion (821 ng/L) in the sample.

C-11.6.3 Coastal stormdrain outfall monitoring

Coastal stormdrain monitoring took place at the sites shown on **Figure C-11.17**. The results of the coastal stormdrain outfall monitoring are presented in **Attachment C-11-II** with exceedances of the AB411 standards in the surfzone highlighted in bold. The data do display substantial differences between stations in their relative frequency of exceedances of the AB411 single-sample standards, which are:

- Total coliforms: 10,000 cfu / 100 ml
- Fecal coliforms: 400 cfu / 100 ml

- Enterococcus: 104 cfu / 100 ml.

Table C-11.14 shows the proportion of all samples exceeding AB411 standards in the receiving water upstream and downstream of coastal drains, both for the entire year and for the AB411 season (May 1 through September 30). While the rankings change somewhat when only the AB411 season is focused on, the five stations ranked the highest, and which have exceedance rates of around 10% or higher, remain the same. Exceedances of AB411 standards were predominantly for Enterococcus (189 exceedances) and less so for fecal coliform (84) and for total coliform (83). **Figures C-11.18 and C-11.19** shows that these sites are concentrated along one particular portion of the coast. This provides a focus for further examination of the monitoring results and perhaps for additional special studies.

Exceedances were usually, but not always, associated with clearly elevated levels in the stormdrain itself (**Figure C-11.20**, **Attachment C-11-II**. **Figure C-11.20** provides a detailed graphic illustration of the relationship between indicator concentrations in the stormdrain itself and in the receiving water, for both the entire year and for the AB411 season. The figure is divided into segments that represent different likely conclusions about the extent to which the stormdrain discharge is causing elevated indicator levels in the receiving water. **Table C-11.15** provides the average discharge rate, over the entire year, for each drain and shows that, except for DSB5, drains with the highest number of exceedances had medium or high discharge rates (drain DSB4 was diverted and thus had no discharge).

Linear regressions of the receiving water indicator values against those from the drains (**Figure C-11.21**) provide additional insight into the relationship between the drains and the nearby receiving water, again for the entire year and for the AB411 season. **Table C-11.16** ranks the sites in terms of the strength of the relationship, as measured by the significance, or "p" value, of the regression slope, for both the entire year and for the AB411 season.

Taken together, these analyses identified several overall patterns, including:

- The proportion of exceedances is generally lower in the AB411 season than in the entire year, implying that exceedance rates are highest in the rainy season
- There are some exceptions to this pattern, as shown, for example, by sites DSB1 and CSBMP1 (**Table C-11.14**)
- Regressions are generally less strongly significant in the AB411 season than in the entire year (**Table C-11.16**), implying that the relationship between drains and nearby receiving waters is tighter in the rainy season.

In addition, some drains showed very similar patterns across all indicators while others (e.g., CSBMP1) demonstrated many more exceedances for one particular indicator. Some sites of particular interest (because of their high exceedance rate) included, from north to south:

- ACM1 (Aliso Creek mouth)

- SCM1 (Salt Creek mouth)
- DSB5 (Doheny Beach Creek)
- SJC1 (San Juan Creek mouth)
- POCHE (Prima Deshecha Channel mouth).

Table **C-11.17** summarizes conditions at these five drains. All except Aliso Creek mouth typically have stagnant sections or scour ponds at or very near their mouth that drain to the surfzone. Two (Salt Creek mouth and San Juan Creek mouth) also have large concentrations of birds that are almost always present. All except San Juan Creek mouth had highly significant regressions for at least some indicators for the entire year, suggesting a potentially strong effect of these drains on the nearby receiving water. However, only Doheny Beach and Poche Beach (for total coliforms) had statistically significant regressions during the AB411 season, suggesting that effects on the receiving water are more visible and persistent during the rainy season. It is puzzling that the regression analysis showed a significant relationship during the AB411 season at Doheny Beach, despite the fact that the drain is diverted during the summer months. This suggests that other processes may be occurring in this vicinity.

(DSB4, another site of concern because of its relatively high exceedance ranking, was diverted and thus did not have corresponding data from the drain itself, making it impossible to calculate regressions.)

These drains, with the exception of DSB5, have higher flow rates than other sites (**Table C-11.15**). Next year's analysis will thus include estimates of bacterial flux or loads in the regression analysis and will also systematically account for instances when drains are diverted and/or not flowing to the ocean because of sand berms on the beach. These sites all drain watersheds that are predominantly urbanized within the few miles of the coast where bacterial loads are most likely derived from. However, several of them (e.g., San Juan Creek and Aliso Creek) also contain substantial open area in their upper reaches.

These results show that a high exceedance rate in the receiving water is not necessarily associated with a strong statistical relationship with values in the drain. For example, the PICO site had strongly significant regressions but relatively low exceedance rates, while site SJC1 displayed the opposite pattern. SCCWRP's study of bacterial indicator levels at reference beaches (SCCWRP Tech. Rpt. #448) showed that exceedance levels at reference beaches were very low during dry weather but reached levels as high as 33% during wet weather. The exceedance levels documented in **Table C-11.14** are in some instances higher than this. The SCCWRP study will thus provide a basis in subsequent analyses for estimating the degree of anthropogenic contribution to these exceedance levels.

There was no consistent overall pattern in which exceedances occurred in both the upcoast and downcoast direction on the same date (**Attachment C-11-II**). Exceedances at these drains were sometimes, but not usually, associated with antecedent rainfall.

Some stakeholders have expressed concerns that bacterial contamination within Dana Point Harbor, and particularly at Baby Beach, might extend beyond the harbor to affect receiving waters along the coast. The Orange County Health Care Agency has 11 sampling locations within Dana Point Harbor that are monitored regularly. These data demonstrate that bacterial contamination is restricted to ankle-depth water at Baby Beach, with samples at deeper locations within the Harbor rarely exceeding the AB411 ocean standards. In addition, special studies conducted by the Agency to determine whether the larger number of boats moored in the Harbor during holiday periods might be a source of contamination found no exceedances in the deeper waters of the Harbor around the moorings.

C-11.6.4 Ambient coastal receiving water monitoring

The ambient coastal receiving water program component included both toxicity testing (with marine test organisms) and chemical sampling (**Figure C-11.22**). Both sets of analyses were performed on samples from the drain discharges. In addition, aerial photography was begun this year, although weather constraints limited the success of this effort. **Table C-11.18** presents the overall chemistry results, with exceedances of the acute saltwater CTR highlighted, and **Table C-11.19** a summary of the numbers of acute CTR exceedances at each sampling station. **Table C-11.20** presents the aqueous toxicity testing results. Toxicity tests were performed using the same marine test organisms as used for the long-term mass loading component.

Table C-11.19 shows that the CTR exceedances were primarily due to copper, with a smaller number due to cadmium, nickel, and zinc. One exception to this pattern was the extremely high concentration of Dimethoate, a pesticide, at station DSB5 on January 26, 2005.

Table C-11.20 shows several instances of substantial toxicity (8 toxic units or above) and elevated toxicity (4 toxic units or above), in both the sea urchin and mysid tests. The most toxic results were seen in stormwater runoff samples collected on January 26, 2005 and March 22, 2005 (**Figure 11.23**). The dry weather samplings on December 14, 2005 and February 16, 2005 also showed elevated toxicity at some sites. The samplings conducted on December 7, 2004, December 28, 2004 and April 28, 2005 showed the least amount of toxicity. From **Figure C-11.23**, it appears that samples collected near the onset of stormwater runoff or under dry weather conditions show the greatest toxicity. Samples collected after large amounts (~1.0 inches) of rainfall show very little toxicity. These observations would suggest that toxicity in the stormwater runoff from these stormdrains is more of an acute problem than a chronic problem.

On the dates toxicity was found at DSB1, DSB3 and DSB5, concentrations of dissolved metals were high enough to affect the sea urchin fertilization tests. At DSB1 and DSB3 the low OP pesticides concentrations could not account for the toxicity measured in the mysid survival tests. Perhaps an unmeasured pesticide was responsible. At DSB5, cadmium, copper, nickel, and zinc were typically above the acute CTR criteria and dimethoate was extremely high on January 26, 2005. This site exhibited higher levels of metals and more CTR exceedances than any other site (see **Figure C-11.24** for a summary of pollutant concentrations at DSB5). This may be due to the fact that DSB5 drains an area with heavy traffic and a very busy commercial area around the intersection of Del Obispo and Pacific Coast Highway. While elevated levels of metals could explain toxicity in the sea urchin test at this site, they are less likely to explain the toxicity seen in the mysid test, since mysids are thought to be more susceptible to ammonia and organics (such as pesticides). However, ammonia levels were low in all samples and OP pesticides were found in only one of three samples showing toxicity. This observation would suggest that an unmonitored pesticide may be causing the toxicity.

The potential relationship between toxicity and the levels of specific constituents in the water can be further investigated by comparing the observed toxicity with that predicted from the observed concentration of key constituents and the LC_{50} or EC_{50} for each toxicity test, keeping in mind the limitations described in section C-11.5.2. **Figure C-11.25** shows the comparison between observed and predicted toxicity for these four sites. In most cases, the predicted toxicity is roughly equivalent to, or less than, the observed toxicity. However, there are some instances where the predicted toxicity exceeds, sometimes substantially, the observed toxicity. In these instances, predicted toxicity was predominantly due to zinc, suggesting that dissolved zinc is less bioavailable than expected, perhaps because it is bound to organic ligands.

The first three years of monitoring data demonstrate a large degree of variability in conditions at the ambient coastal sites:

- The level of toxicity was lowest in the second year (2003-04) of the program, with only two stations showing elevated toxicity, compared to five stations in the first and third years. In addition, measured as toxic units, the degree of toxicity was much lower in the program's second year.
- The number of metals CTR exceedances was much higher in the program's third year (2004-05), with copper dominating in all three years.
- The mean concentration of copper in samples exceeding the chronic criterion was 19.5 μg/l in 2002 – 2003, 9.3 μg/l in 2003-04, and 19.5 μg/in the most recent monitoring year.
- The number of stations showing exceedances of the acute copper criterion varied, from 12 in the first and third years to five in the second year.

The higher level of toxicity in the most recent monitoring year (2004-05) may have been due to the relative high levels of rainfall during the 2004-05 rain year (**Figure C-11.16**).

C-11.6.5 Dry weather reconnaissance

The dry weather period (May 1 – September 30) does not precisely match the Program's reporting period (July 1 – June 30). For purposes of meeting the Program's annual reporting requirements, this report includes data from July 1, 2004 through June 30, 2005. Up to date monitoring results can be viewed on the Program's website at ftp://watershed-mgr:2alau54n@pfrdftp.ocgov.com/NPDESstormwater/DAMP/11.0%20Water%20Quality%20Monitoring/San%20Diego%20Region%20Dry%20Weather%20Monitoring%20Program/.

This report section summarizes basic monitoring results. Additional information on the permittees' activities to follow up on these data with source identification and other efforts are presented in Chapter 10.

The dry weather design includes both random and targeted sites (**Figure C-11.26**) that are sampled five times during each dry weather period. The purpose of the random sites is to define an average background condition in urban stormdrains. The purpose of the targeted sites is to focus specifically on stormdrains and/or locations known or thought to be sources of urban pollutants. A site (either random or targeted) was classified as problematic only when a pollutant was outside a tolerance interval (calculated from the entire set of random sites) or a control chart bound (calculated from the history of data at each site) on two consecutive sampling periods. Two targeted sites in Dana Point, DP@BR and DP@SA, were removed from the program this year. Further reconnaissance showed that these drains do not have any chronic flow and that sampling was taking place primarily from a stagnant pool of water that was not representative of dry weather urban runoff. In addition, both electrical conductivity and hardness were removed from the identification of problematic sites, since both are background constituents reflective of general conditions rather than deriving from urban runoff. Finally, the site descriptions were modified to remove the names of individual cities; these were replaced with broader watershed designations. In many cases, the pipe outfall is in one jurisdiction, while the drainage area is in one or more other cities. In addition, jurisdictions can change. For example, some drains such as COL11P01, are in an area that was recently incorporated by Rancho Santa Margarita and is no longer part of the County's jurisdiction.

In general, the random sites met this criterion to a much lesser extent than did the targeted sites, a confirmation that the targeted sites were successful in focusing on problem areas. There were 21 instances in which the value of a monitored constituent random site was outside the tolerance interval on consecutive sampling dates for the same pollutant, while this occurred 41 times at the targeted sites (**Table C-11.21**). There were no instances in which data points exceeded either the Shewart or CUSUM control chart bounds on consecutive sampling events. **Table C-11.21** shows that a wide range of constituents exceeded the tolerance interval bounds, including metals, pesticides, nutrients, and bacteria indicators. For reference, the dry weather program monitoring results are presented in **Attachment C-11-III**.

The random sites present a picture of the urban background conditions in the south County. These data have been extremely helpful in characterizing background variability in urban conditions and providing a basis for prioritizing focused source identification efforts. Similarly, plots of data from those stations where specific parameters exceeded the tolerance intervals on consecutive sampling events provide useful information about the nature and persistence of such elevated levels. This information is proving useful in helping to characterize the behavior of potential sources of contamination and to assess the effectiveness of upstream source control measures implemented by the respective permittees.

C-11-6.6 Dana Point Harbor Monitoring

Monitoring at Dana Point Harbor (**Figure C-11.27**) was based on the Triad approach, and included benthic infaunal, toxicity, and sediment chemistry analyses. **Table C-11.22** shows the sediment chemistry and sediment toxicity testing results and **Table C-11.23** the benthic infauna community analysis. **Table C-11.24** describes the BRI scoring ranges in terms of amount of deviation from reference condition.

Table C-11.22 shows that there were no data values that exceeded the NOAA Effects Range Median (ERM) concentration. However, copper and zinc were consistently anthropogenically enriched at virtually all stations and sampling times, while lead and cadmium were less so, and silver only rarely so. Sediment concentrations were consistently higher during the June sampling than the October sampling. This pattern was not pronounced last year. Sediment characteristics changed markedly between the two sampling events, with several stations showing large increases or decreases in the percent clay and percent silt/clay. These changes were not correlated with changes in chemistry concentrations or in toxicity.

During the past monitoring year, toxicity was consistently highest at DAPTEB, and this station had the highest concentrations of cadmium, copper, and zinc in the sediment. This site is adjacent to the largest stormdrain in the harbor, but detailed information about sources in this drainage are not yet available. In contrast, station DAPTDC had the lowest level of toxicity, the lowest values for copper and zinc, and the fewest number of anthropogenically enriched constituents. Between these extremes, however, there was no consistent relationship between levels of chemical constituents and the degree of toxicity. The Bight 03 study also documented a relationship between sediment toxicity and the level of fines in the sediment. However, this relationship is extremely "noisy" and there was no apparent correlation between toxicity and percent silt/clay in the small number of samples from Dana Point during the past monitoring year.

Figure C-11.28 and **Table C-11.25** provide a larger regional context for assessing the Dana Point sediment toxicity results. **Table C-11.22** shows that sediment toxicity values for individual sites in Dana Point Harbor, which, except for station DAPTEB, average less than 20% mortality, are less toxic than about half the stations in the Bight 03 study. It is important to note that the data from the Bight Program are not strictly comparable to the monitoring data from Dana Point Harbor because they were collected on the random Bight Program sampling grid, while the NPDES monitoring program deliberately sited stations in locations (i.e., at the mouths of stormdrains) more likely to be contaminated. Despite this, a subjective comparison shows that sediment toxicity at Dana Point Harbor is about average for bays and harbors in the Southern California Bight (**Figure C-11.28**, **Table C-11.25**), as documented in the Bight 03 report on sediment toxicity. Again, this is not unexpected, given that the Program's sites in Dana Point Harbor were deliberately sited at stormdrain discharges, with the exception of DAPTLR.

Sediment toxicity varies somewhat over time. **Figure C-11.29** shows sediment toxicity at each station over the past two years. While toxicity varies at each station across surveys, the overall relative levels of toxicity among stations remain the same., with DAPTEB having the lowest survival and DAPTDC the highest. The Bight 98 and 03 programs documented some increases and some decreases in sediment toxicity over the five year period between the two studies. However, there were no readily apparent explanations for these differences and there were no changes in the overall toxicity at Dana Point from Bight 98 to Bight 03. While temporal variability in sediment toxicity may result from changes in the physical and chemical characteristics of the sediment, the Bight 03 sediment toxicity report illustrates that these relationships are too variable to provide a basis for site-by-site explanations of shifts in toxicity levels.

The benthic infaunal analysis (**Table C-11.23**, **Figure C-11.30**) shows that station DAPTDC falls within reference condition (**Table C-11.24**) on both surveys. In contrast, station DAPTLR falls within Response Level 3, with greater than 50% of reference species lost, on both surveys. These two stations also had the highest and lowest abundance and diversity, respectively. The other three stations fell between these two extremes. BRI scores at each station were generally consistent across the two sampling periods, in contrast to the consistently higher sediment chemistry concentrations in the June sampling. This suggests that sediment chemistry alone is not predictive of benthic community conditions. While DAPTDC had the lowest sediment toxicity**, Figure C-11.31** shows that there was no overall consistent relationship between BRI scores and sediment toxicity. The figure shows only a weak (and statistically nonsignificant) relationship between toxicity and poorer benthic community conditions (i.e., higher BRI score). However, this relationship appears to be driven by the two data points at the lower left of the figure, which represent station DAPTDC. The bottom at this site consists mostly of rock and large gravel, with very little mud available for benthic macroinvertebrate habitat. This suggests that effects on the benthic infaunal community may not be driven by sediment toxicity, but by other factors such as physical disturbance. It also suggests that simple sediment chemistry values do not reliably predict potential toxicity, except perhaps at the extremes. These relationships are currently under investigation as part of the State Water Resources Control Board's Sediment Quality Objectives project. The findings and guidance from that effort will be applied by the Program as they become available.

C-11.6.6 Additional toxicity analyses

Past interpretations of toxicity testing results have depended in part on subjective comparisons of the observed toxicity to chemistry results. In some cases, more rigorous TIE (toxicity identification evaluation) studies can provide more detailed insight into the specific chemical compounds contributing to observed toxicity. TIE's can be problematic, however, because their cost and the logistics involved in performing them preclude carrying them out in all instances. For this reason, TIEs were targeted at those monitoring sites where substantial toxicity was observed during the past monitoring year. If toxicity was observed during the current monitoring year, then TIEs were immediately performed using the same sample water. All of these instances occurred at the ambient coastal receiving water stations. Unfortunately, the Phase I TIEs were unable to identify any specific sources of toxicity, leading the laboratory analysts to conclude that toxicity was due either to unknown toxicants or to volatile compounds that had evaporated from the sample between the time the toxicity tests were run and the time the TIEs were begun.

C-11.6.7 Quality Assurance

The quality of data produced by each of the three contractor laboratories was evaluated by submitting quality control samples with environmental samples. Most of the samples submitted were synthetic, comprised of aliquots of prepared standard solutions in nanopure water matrices. Quality Control sample conductivities were adjusted to levels similar to environmental samples with Ultrex grade sodium chloride. These synthetic samples were used to assess the accuracy of each laboratory. Replicate samples were also submitted to evaluate the precision of the laboratories.

The contractor laboratories conduct internal quality control programs utilizing certified reference materials (CRMs), spiked and replicate samples.

The results of the quality assurance program with the contract laboratory are summarized in **Attachment C-11-IV**. The allowable range of percent recovery for synthetic and samples is set at 70 - 130 for concentrations above 5 times the detection limit. For replicate samples in which the highest reported value exceeded 5 times the detection limit, the allowable range was set at 75-125 percent. For blank sample analyses the allowable range was the detection limit (dl) to 3(dl). Those results outside these ranges are boxed in the Attachment.

During the year the price agreements for laboratory analyses were renewed. Analyses under these new agreements began on April 1, 2005. Aqueous analyses for nutrients and OP pesticides are now conducted by Associated Laboratories. Trace metals in freshwater are conducted by Weck Laboratories and in seawater by CRG Laboratories. Sediment analyses are conducted by CRG laboratories. The bacteriological analyses for the Coastal Stormdrain Outfall Program are conducted by the Orange County Health Care Agency's Public Health Laboratory. CRG conducts the bacteriological analyses for the Dry-weather Reconnaissance Program.

Generally, the analyses of nutrients, metals, and bacteria were acceptable. There was a major problem with the analyses of OP pesticides after the change in laboratories. Several synthetic samples containing OP pesticides were submitted to the new laboratory but the results were all reported as undetected. A meeting will be held with the laboratory to attempt to resolve the problem.

C-11.6.8 Summary

The third year of monitoring under the Third Term Permit has expanded the information available for regional and watershed assessment of receiving water conditions and potential impacts on these from urban runoff. The expanded scope of the monitoring program encompasses not only inland creeks and streams but coastal receiving waters as well.

The monitoring data reviewed above begin to build a picture of year to year variability in conditions, as well as highlighting specific locations of potential concern. For example, a subset of coastal stormdrains with the highest frequency of AB411 exceedances in the receiving water was identified and these conditions were related in some instances to the configuration of the drain and its discharge area. For this program component, linear regression analysis was also used to identify those drains with the most consistent relationship between discharge and receiving water characteristics. The analysis of data from the bioassessment program component was improved by the addition of a multivariate pattern analysis that defined spatial and temporal patterns that persisted over the three years of monitoring data. A comparison of these patterns to the physical habitat, aquatic chemistry, and toxicity data showed that alterations to physical habitat are most strongly correlated with IBI scores. The analysis of the Dana Point Harbor sediment data produced a similar result, namely that benthic community conditions (i.e., BRI scores) were not strongly related to sediment toxicity.

Toxicity test results for the long-term mass loadings and the ambient coastal program components were further analyzed by comparing observed toxicity to a predicted toxicity level based on the observed aquatic chemistry values. In many cases where substantial toxicity was observed, this appeared to be due to zinc. In addition, this analysis identified monitoring sites where unexplained toxicity may warrant further study.

The dry weather reconnaissance program has been improved by creating monthly data summaries and posting these on the Program's website for use by the individual cities. These routine reports, as well as verbal communication with the city representatives, are providing a basis for targeted follow-up source control and education efforts.

For each of these program components, ongoing data analysis and evaluation, as well as consultation with the Permittees, will be used to refine data analysis methods and focus additional effort on areas of concern.