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Model Monitoring Program for Large Ocean Discharges in Southern California

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Foreword

The goal of this document is to build the basic design for the ocean monitoring programs of the four largest POTW dischargers in southern California. Although the focus of this report is toward large POTWs, many of the principles, framework, and design apply equally well to other ocean dischargers such as small POTWs, industrial dischargers, power generating stations, and stormwater dischargers. The monitoring design is intended to develop consistency among programs, improve the effectiveness of each program in meeting the needs of management, and the increase the efficiency with which monitoring is conducted.

Recommendations for individual monitoring program elements are provided as a means for building the basic ocean monitoring design. The recommendations were built upon lengthy discussions by both the regulatory and regulated communities throughout southern California between 1998 and 2001. As such, the references to state regulatory mandates pertain to the guidance available at the time of the discussions provided under the 1997 California Ocean Plan.

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I. INTRODUCTION

Ocean monitoring programs conducted by publicly owned treatment works (POTWs) as part of National Pollutant Discharge Elimination System (NPDES) permit requirements have existed in the Southern California Bight (SCB) for nearly thirty years. In the 1970s, monitoring programs focused upon characterizing the marine environment since our understanding of the ocean environment was still growing. At that time, NPDES discharger impacts on the marine environment were not well understood; many impacts were masked by the state of science and many assessments were confounded by natural variability inherent in the ecosystem. In these early days, the design of ocean monitoring programs were often based upon an analysis of variance (ANOVA) model that compared sites near a single point source to a site, or sites, distant from that source (Tetra Tech 1982, U.S. EPA 1991). The design of ocean monitoring programs has changed very little since 1970, despite the changing needs of environmental managers. In the 1990s, environmental managers have an increased understanding of the marine environment and the result of three decades of monitoring has shown us that management activities at POTWs can improve the health of the environment. Currently, the potential effects that NPDES permittee discharges have on the beneficial uses in receiving waters are more subtle than in the 1970s, hence, the questions that environmental managers now ask are different.

Ocean monitoring in the SCB is costly. Approximately $17 million is spent annually on marine monitoring programs by NPDES permittees in the SCB (Schiff et al. 2000). There is no unified approach to implementing these programs. Since the various facilities lie in separate jurisdictional boundaries governed by different regulatory agencies, most monitoring programs have been designed independently and vary in many respects, including the effort expended. As a result, the data from these different programs are often not comparable due to differences in sampling methodology, analytical procedures, and quality assurance. Even when the data are comparable, they are stored in a series of independent, incompatible electronic storage media that make the data difficult to access, retrieve and summarize. This lack of a unified approach not only limits technical comparability among programs, but it has resulted in inequitable levels of effort and resource expenditures among the various facilities monitoring the SCB.

The needs of ocean monitoring programs have changed over the last 30 years. While the ANOVA-based monitoring design is adequate for addressing some regulatory issues, this model has proven to be insufficient for providing important information required by resource managers, including regulators and permitted dischargers, to enable better decision-making regarding the protection of beneficial uses. These types of information include a more accurate and complete characterization of reference conditions and natural variability, quantification of the spatial extent as well as magnitude of impact, establishment of rates of improvement (or degradation), determination of cumulative impacts from multiple sources that commingle, and establishment of cause/effect mechanisms for identifying sources of problems.

Regional monitoring efforts have been one response to the changing needs of ocean monitoring programs over the last 30 years (NRC 1990b, Cross and Weisberg 1996, SCBPP 1998). Large-scale assessments provide context to resource managers by describing the range of impacts and placing human impacts into the context of variability from natural oceanographic events such as El Niño. Regional monitoring provides a description of regional reference conditions, in part replacing the limited number and different reference sites used by facility-specific programs. Regional monitoring also leads to methods standardization and improved quality control through intercalibration exercises. However, maximizing these benefits requires integration between regional monitoring and facility-specific monitoring. Regional monitoring in the SCB is relatively new and it is unclear exactly how, when, and where the integration with local effects monitoring should be accomplished.

Goal of This Document

The goal of this document is to build the basic design for the ocean monitoring programs of the four largest POTW dischargers in southern California. Although the focus of this report is toward large POTWs, many of the principles, framework, and design apply equally well to other ocean dischargers such as small POTWs, industrial dischargers, power generating stations, and stormwater dischargers. The monitoring design is intended to develop consistency among programs, improve the effectiveness of each program in meeting the needs of management, and increase the efficiency with which monitoring is conducted. Specifically, this document addresses the following questions:

- What are the specific management questions of concern?
- What is the basic approach to monitoring design?
- What frequency do samples need to be collected?
- What locations or distances do samples need to be collected?
- What specific measurements (indicators) need to be measured?

This document is the third in a series of steps to develop a model monitoring program. The first step was to derive the most important management questions that regulators and dischargers ask in order to make resource management decisions (SCCWRP 1999). We derived these questions by conducting interviews with regulators and permittees throughout southern California. The second step was to conduct an independent review of existing monitoring programs to assess their effectiveness and efficiency (SCCWRP 1999). Our review demonstrated that most monitoring programs were effective at answering most of the management questions posed by regulators and dischargers, but they were inefficient and could be improved through modification of their monitoring designs. In other circumstances, the questions that the monitoring programs had been designed to answer were no longer important and new questions have evolved as our understanding of the environment has improved.

This document is the result of a consensus-based approach that builds upon the first two documents by designing a recommended program that is both effective and efficient. The document was built by interactions among regulators and permittees at collaborative meetings over a 32 month period. The document is built to serve as a blueprint for developing a monitoring program and, as such, is not site-specific. It provides the approach and rationale for designing the monitoring program and often describes recommended strategies for ensuring effectiveness, efficiency, and comparability. It should serve as the starting point for creating or refining a monitoring program and provide the guidelines for regulators and permittees to discuss site-specific needs and designs.

Where this document focuses on sampling design, subsequent documents will address logistical aspects of implementation. Method manuals for field collection, laboratory processing, and information management are presently being prepared. These manuals will help to ensure data quality, consistency, and comparability among monitoring programs in the SCB.

II. PRINCIPLES AND FRAMEWORK FOR A MODEL MONITORING PROGRAM

Developing consensus about a model program among disparate groups required first identifying some guiding principles on which to base decisions regarding alterations and refinements to existing programs. Moreover, we needed a framework on which to build the program. This chapter describes those principles and framework.

Principles of Model Monitoring

Four fundamental principles guided our ideas for each monitoring element. The first principle focused on the need to monitor. While the collection, treatment and disposal of wastewater is a societal necessity for the protection of the public's health and is required of communities by federal and state statutes, California state law regulates disposal of treated wastewater to the ocean as a privilege (Chapter 4, Article 4, Sec 13263 (g) of the California Water Code). NPDES permits are issued to grant the privilege for discharging to public waters predicated on demonstration that the discharge does not result in environmental degradation or impacts to beneficial uses. Monitoring is necessary to develop this demonstration and is part of exercising the privilege.

Our second principle is that while discharger permittees have monitoring responsibilities, monitoring should be focused on activities that directly relate to management questions that need to be answered, rather than gathering data for data's sake. The answer to these monitoring questions should have decision value, with managers being prepared to take one action if the answer is yes and a different action if the answer is no. In some cases, the action can be as simple as conducting more sampling to better understand the problem (or less sampling if there appears not to be a problem), but the link between data collection and potential actions should be explicit.

The third principle is that monitoring programs need to address questions posed at different spatial scales by a variety of different audiences. Discharge monitoring has traditionally focused on the impact in the immediate vicinity of the discharge to address regulatory issues. Monitoring also needs to address public concerns about the health of the environment, which are often regional in scale. An example might include the public's perception about the health of fish. While the public might be concerned about the health of the fish community in the immediate vicinity of an outfall, they often take a more holistic view by asking "are the fish communities in the Santa Monica Bay healthy", or "are fish communities in southern California healthy"? It is the cumulative responsibility of all NPDES dischargers to answer both the site-specific questions regarding the impact of their discharge as well as the more regional questions to address public concerns.

A fourth principle is that the level of monitoring should be proportional to the level of concern about the question to be addressed. The greater the potential for environmental impact, the more monitoring that is necessary to address regulatory and public concerns. Similarly, the less the potential impact, the less monitoring that is necessary. As a corollary to this principle, the level of monitoring should adapt to the findings. One of our greatest criticisms of existing monitoring programs is that they are inflexible; monitoring continues regardless of what is learned, needed, or relevant. Throughout this document, references are made to "adaptive monitoring". These references indicate events or thresholds that can serve as triggers for additional (or lesser) monitoring effort based on findings within the monitoring programs.

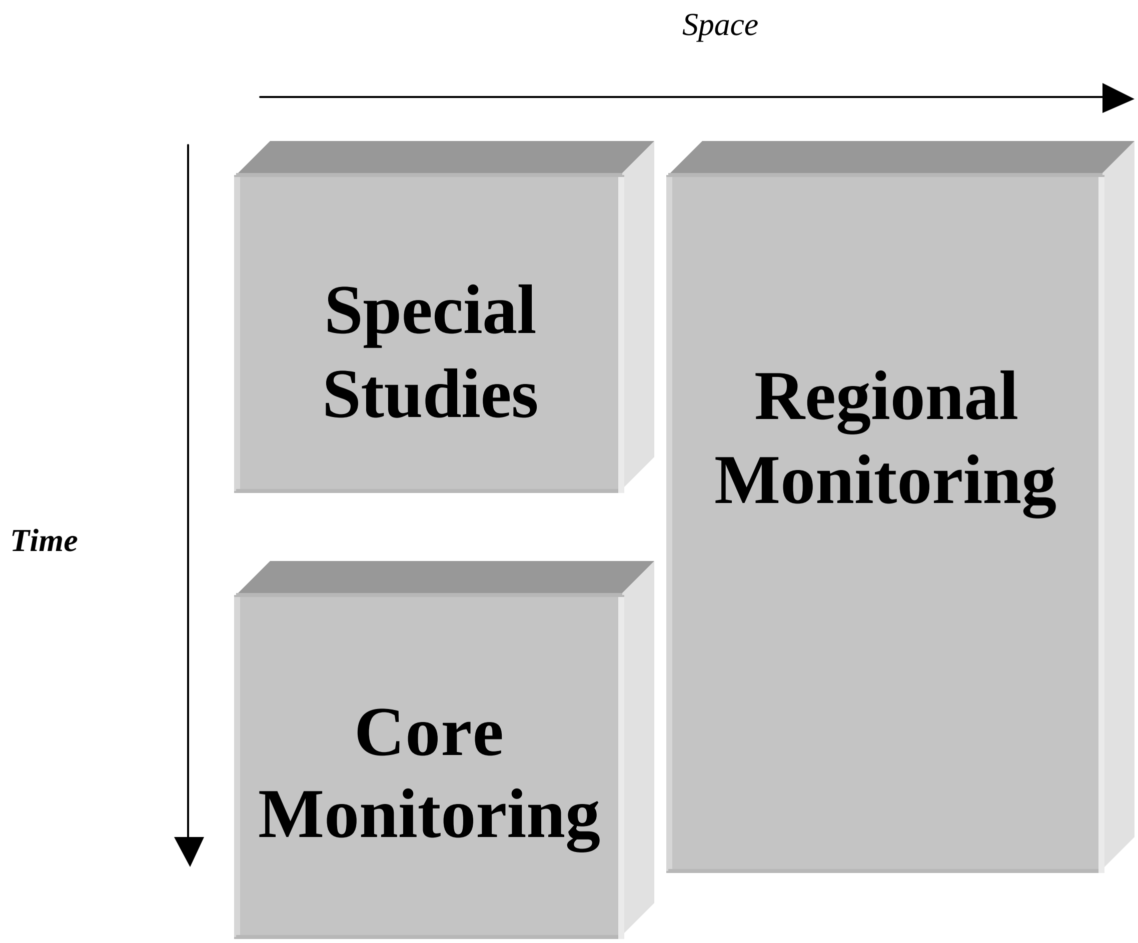

FIGURE II-1. Model monitoring framework

Organization of this Document

Ten monitoring elements are addressed in this document: effluents, shoreline water quality, water column, sediment, fish, marine birds, marine mammals, intertidal zones, wetlands, and kelp beds. These elements were selected because they are each studied as part of present discharge monitoring programs or because they have been suggested as elements to be included in NPDES monitoring programs by the Santa Monica Bay Restoration Program. The recommended monitoring design for each of these elements is described in a separate chapter.

Each chapter contains three sections. The first section compares and contrasts effort expended by existing monitoring programs. The second section provides an evaluation of the existing monitoring programs. The third section describes the technical design issues that were addressed in molding the model monitoring program. Under each design issue we present the preferred approach for the model program and the rationale for the approach.

III. EFFLUENT MONITORING

There are three management questions for effluent monitoring:

Q1: ***Is the effluent concentration of selected constituents below levels that will protect human health and aquatic life?***

Q2: ***What is the mass of selected constituents that are discharged annually?***

Q3: ***Is the effluent concentration or mass changing over time?***

The primary reason for monitoring effluent concentrations prior to discharge is to evaluate discharge characteristics and to assess compliance with effluent limitations, thereby ensuring that water quality standards are achieved in the receiving water. Water quality standards are contained in the California Ocean Plan (Ocean Plan) (SWRCB 1997) and include beneficial uses of the ocean, numeric and narrative water quality objectives necessary to protect those uses, and a policy to prevent the degradation of water quality. Depending on which beneficial use requires the lowest concentration to be protective, Ocean Plan numeric water quality objectives are either public health or aquatic life based. Public health derived Ocean Plan objectives are based on estimated fish and shellfish consumption, using a California-specific risk assessment approach. In contrast, aquatic life protection water quality objectives are derived from laboratory tests on sensitive life stages of marine organisms. Effluent limitations are derived using water quality objectives, background seawater concentrations, and the discharger-specific seawater-to-effluent dilution ratio ("initial dilution" or mixing zone). Regulatory agencies allow dischargers to measure pollutant concentrations in their final effluent, which contains higher, easier to measure, concentrations. These concentrations are then compared to effluent limitations. These monitoring data are then used to trigger source tracking or initiate receiving water monitoring for potential effects, among other actions.

A second method used for assessing risk to aquatic life is the use of toxicity tests. These tests expose sensitive life stages of marine organisms to effluent to assess their acute or chronic impact (U.S. EPA 1995). The advantages of these tests are two-fold. First, the risk is directly measured. Second, the tests can capture toxicity that occurs from unmeasured constituents or from synergistic effects of multiple constituents below their individual water quality objectives. Resource managers can use toxicity monitoring to assess if their discharge is toxic, trigger toxicity identification evaluations (TIE) and track sources or modify the treatment process to reduce environmental risk.

The second question regarding mass emissions is an important component of an effluent monitoring program because it provides resource managers with the tool to compare contributions of constituents from different facilities or groups of facilities (e.g., one POTW versus another POTW or all POTWs versus urban runoff). Identifying which facilities contribute the greatest mass emissions helps managers utilize their resources for reducing the most appropriate inputs. Finally, as mass-based regulations, such as total maximum daily loads (TMDLs), become more important, mass emission monitoring will become critical. Of course, all discharges to a receiving water body, regardless of source, need to be assessed in order for managers to use this information effectively.

The third question enables resource managers to track discharges from a single facility over time. If effluent concentrations or mass emissions from a facility are increasing over time, then resource managers can use this information to carefully consider if management actions are necessary. On the other hand, if management action is taken to reduce emissions, monitoring of trends in mass emissions or effluent concentrations will enable resource managers to assess the effectiveness of these actions.

Compare and Contrast Among Agencies

The annual number of effluent constituent measurements differed substantially among the four largest POTWs (Table III-1). A 5-fold difference was found in the number of organic constituent measurements and a 7-fold difference was found in the number of metal constituent measurements.

One reason for the differences in the number of measurements among the four dischargers was that effluent constituents were measured at different frequencies. None of the metal or organic constituent analysis frequencies were common to all dischargers (Table III-2). For example, the City of San Diego's Point Loma Wastewater Treatment Plant (PLWTP) analyzes its effluent weekly for trace metals, compared to monthly analyses by the Orange County Sanitation Districts (OCSD), and monthly or quarterly analyses by the Los Angeles County Sanitation District's Joint Water Pollution Control Plant (JWPCP) and the City of Los Angeles' Hyperion Treatment Plant (HTP). Similarly, PLWTP analyzes its effluent weekly for DDTs and PCBs, while OCSD and JWPCP analyze monthly, and HTP monitors quarterly (Table III-2).

Differences among the facilities were also apparent in effluent constituent reporting levels (Table III-3). The range of reporting levels for any single constituent varied by more than a factor of two among the four facilities. Most constituent reporting levels were within a factor of 20, but antimony and dioxin showed a 77-fold difference in reporting levels (0.3-23 µg/L) and a 29-million-fold difference in reporting levels (0.0017-50,000 ng/L), respectively.

Evaluation of Existing Effort

The management question "*Is the effluent concentration of selected constituents below levels that will protect human health and aquatic life?*" is effectively being answered by all four large POTWs for most effluent constituents. The majority of effluent constituent concentrations and toxicity test results are consistently below 1997 California Ocean Plan objective-based effluent limits. A few constituents have analytical reporting levels above 1997 California Ocean Plan objective-based effluent limits; therefore, the management question for these chemicals cannot be answered (Table III-4). However, these problems are the result of a technical inability to reach extremely low levels (e.g., dioxins) rather than a specific facility's ineffectiveness.

While dischargers are answering the management question effectively for the majority of constituents, they are not answering the question in the most cost-efficient manner. There has already been acceptance that daily data are not required to ensure compliance with water quality thresholds. However, little or no justification is evident in the current sampling designs to validate the required frequencies for most of the analytes. Most of the frequencies were set at subjectively pre-determined intervals without considering the actual risk of effluent concentrations exceeding the prescribed threshold.

The next management question for effluent monitoring pertains to mass emissions. Each of the large POTW agencies has effectively addressed this management question within their facility. It has been this data, in conjunction with mass emission estimates from special studies of other sources, which has demonstrated the relative importance of urban runoff as a major pollutant source to the SCB. Unlike the early 1970's, stormwater mass emissions currently rival traditional point sources for suspended solids, nutrients, and most trace metals (Schiff et al.. 2001). However, assessing regional mass emissions on an ongoing basis has been ineffective. This is primarily due to a number of constituents that are routinely below reporting levels; hence, mass emissions cannot be accurately evaluated from the existing data. The varying reporting levels among facilities further compound the problem. When estimating mass emissions for truncated data, constituents below the reporting level can either be assigned a zero, one-half the reporting level, or set equal to the reporting level. The load estimates using each of these methods varies greatly, particularly for large POTWs who discharge tremendous volumes, and there is significant bias associated with each technique (Raco-Rands and Steinberger 2001).

The third management question for effluent monitoring pertains to trends. Large POTW monitoring programs have been effective at tracking trends in effluent quality, particularly for tracking changes in mass emissions. Agencies have demonstrated dramatic reductions in mass emissions over the last 30 years including a 70% reduction in suspended solids, a 95% reduction in trace metals, and a > 99% reduction in chlorinated hydrocarbons. This has been some of the most cited data by local agencies and has drawn recognition worldwide as a shining example of effective environmental management (Schiff et al.. 2000). However, the frequency of sampling currently used in the SCB to detect trends varies among agencies and no systematic or objective rationale has been agreed to by local resource managers for the amount of trend to be detected over a given amount of time.

Recommended Sampling Design

Q1: Is the effluent concentration of selected constituents below levels that will protect human health and aquatic life?

The primary design element for this management question is sampling frequency. Sampling frequency should be proportional to the potential risk of exceeding the water quality threshold. The more likely a facility is going to exceed a threshold, the more frequently that facility should be testing. Likewise, sampling frequency should be decreased if the risk of exceeding a water quality threshold is low.

We recommend two monitoring designs that can be used to address the appropriate risk-based frequency. The first is a variance-based approach, which uses statistical modeling of variance for a particular constituent to assess optimum sampling frequency. The second is a quality assurance-based (QA) approach. The decision for which approach to use is based upon the historical effluent monitoring data. If adequate estimates of variance can be derived, for example routinely detected concentrations from the last permit cycle, then the variance-based approach is preferred. If estimates of variance cannot be derived, which would be the case for constituents that are routinely below detectable concentrations or bioassay results that are consistently non-toxic, then the QA approach is recommended.

Variance-based approach

The variance-based approach uses the distribution of historical monitoring data to determine the probability, or risk, of the next sample exceeding the water quality threshold (Figure III-1). For example, a greater number of samples would be required when the data are highly variable, or when effluent concentrations are close to exceeding their prescribed limit. Conversely, when there is less risk of exceeding a threshold, such as when data are not variable or are distant from the threshold, frequencies may be decreased. The variance-based approach is contingent upon statistical predictions about the likelihood of exceedance, which can be evaluated using power analysis. By examining the historical effluent data, power analysis can determine the optimal number of analyses required, with a specified level of confidence that an exceedance will not occur. This method is essentially the reverse of predicting a confidence interval, where estimated confidence is determined from a known number of analyses and the associated variability of the data.

In the case of normally distributed and independent samples, the sample size necessary for this approach is given by:

(Equation 1) $$n \approx \frac{z_{1-\alpha_1}^2 \sigma^2}{(T-\theta_0)^2}$$

where
n = number of samples per year,
T = the threshold value,
θ_0 = the estimated current concentration,
α = the desired confidence level,
σ^2 = the sample variance (s^2) (Note: To estimate 6 month median use $\sigma^2 = {s^2\pi}/{2}$), and
z = probability estimate from normal distribution table (z score).

The ability of the variance-based approach to yield sample size appropriate to achieve desired level of confidence depends on the accuracy of estimates, θ_0 and σ^2.

Historical effluent concentration data from 1989-96 for each large POTW were collated to demonstrate the use of power analysis. Power curves for each of the POTWs demonstrated that a relatively small number of samples per year were needed to determine that lead concentrations would not exceed their respective six-month median threshold with a relatively high (99%) level of confidence (Figure III-2). The JWPCP and HTP would need to analyze one sample per year to be 99% confident they have not exceeded the 1997 California Ocean Plan objective-based effluent limit, whereas OCSD and PLWTP would need to analyze two samples per year to achieve the same result (Figure III-2). This difference among facilities reflects higher variability or values closer to the threshold for effluent lead concentrations for OCSD and PLWTP compared to JWPCP and HTP. The fact that not all POTWs required the same number of samples to assess sampling frequency for 1997 Ocean Plan objective-based thresholds also holds true for different constituents within the same facility (Table III-5).

Inherent in this approach is an understanding of the variance for each constituent. We recommend that the variance be re-evaluated at least once every five years (i.e. once each permit cycle). Additionally, the mean and variance should be re-evaluated if a significant plant adjustment is made, such as improved treatment operations. The mean and variance should also be re-established if samples begin to fall outside of the expected range of predicted concentrations. This adaptive monitoring trigger would increase sampling frequency when a threshold defining past operations is crossed, even if the 1997 Ocean Plan threshold has not yet been exceeded, with the rationale that a sample falling outside of the historical range could indicate a plant upset. If a single sample exceeds the prediction interval, then a second confirmatory sample should be taken. If the second sample falls outside the prediction interval, a new mean and estimate of variance needs to be established. Assessing the threshold for these prediction intervals should follow the algorithm described by Zar (1984):

(Equation 2) $$\theta > q_0 \pm s\left(1+\frac{1}{N}\right)^{1/2} z_{1-a_2/2}$$

where:
N = total number of samples since last reassessment,
θ = current sample concentration,
θ_0 = the estimated current concentration (mean of concentrations since last reassessment),
á = the desired confidence level,
σ^2 = the sample variance (s^2), and
z = one-tailed probability estimate from normal distribution table (z scores).
We recommend using a 99% prediction interval where $z = 2.576$

The variance-based approach may reduce the sampling frequency that is currently specified in some NPDES permits. To alleviate the fear that a reduction in frequency will result in an inability to track short-term alterations in effluent quality, we recommend using other mechanisms that are already in place at most facilities. Plant operation monitoring, pre-treatment programs, and biosolids monitoring often take daily measurements of general constituents. While the results from these other programs may not be useful for compliance monitoring, they can be used to confirm that the effluent hasn't changed demonstrably over shorter time periods.

QA-based approach

The variance-based approach does not work when the variability of constituent concentrations is unknown. This occurs for constituents such as DDT, PCB and dioxin which are frequently reported as non-detects because their detection levels using conventional laboratory methods are so close to the 1997 Ocean Plan standard. In the case of toxicology, the variability is unknown because effluents are rarely toxic at the dilutions tested and dischargers do not test effluent concentrations greater than their dilution credits allow, for chronic toxicity testing.

Unlike the variance-based approach, the QA-based approach does not incorporate proximity to the threshold or variability. Originally developed for the manufacturing industry, the QA approach is based on a tiered pass/fail system, in which sampling effort increases with the number of exceedances. It is based on a binomial probability distribution to assess the likelihood of predicting exceedances. If enough samples meet pre-specified quality assurance guidelines (water quality thresholds), then the frequency of QA checks can be reduced and still keep managers confident that the process control is working properly. However, if a QA failure occurs, then the frequency of sampling needs to be increased to the original frequency to restore management confidence in QA. If repetitive QA failures occur, then sampling frequency is further increased until, ultimately, mangers are convinced that some management action is necessary to improve performance.

The number of samples in the initial sampling tier, and the number of exceedances leading to a management action depends on several variables, including 1) P_0, the acceptable probability of exceedance, 2) P_1, the unacceptable probability of exceedance, 3) $á_1$, the probability that increased sampling is mandated when the probability of exceedance is actually below P_1, and 4) â, the probability that reduced sampling is mandated when the probability of exceedance is actually above P_0. A fifth parameter for QA approach to chemistry is $á_2$, the probability that management action is mandated when the probability of exceedance is actually below P_1. The guidelines for using these parameters follow the sequential probability ratio developed by Wald (1947):

1. Each sample is evaluated whether or not it was a "hit":
   If the number of hits are greater than B* and less than A*, then continue sampling at same rate;
   If the number of hits are greater than or equal to A*, then increase sampling frequency (move to higher tier); or
   If the number of hits are less than or equal to B*, then decrease sampling frequency (move to lower tier).

2. If the evaluation leads to an increased sampling rate, frequencies do not resume the initial sampling rate until the condition $B^* < k < A^*$ again holds.

3. If sampling frequencies are reduced, then the sampling rate remains at a low rate until the first exceedance, upon which sampling returns to initial rate and the process is started all over again. Exceedances in toxicity testing will be identified by the

magnitude or persistency of the toxicity, and will not necessarily be triggered by a single "hit".

Where:

(Equation 3) $$B^* = \frac{\ln\left(\dfrac{\boldsymbol{b}}{1-\boldsymbol{a}}\right)}{\ln\left(\dfrac{p_1(1-p_0)}{p_0(1-p_1)}\right)} - n\frac{\ln\left(\dfrac{1-p_1}{1-p_0}\right)}{\ln\left(\dfrac{p_1(1-p_0)}{p_0(1-p_1)}\right)}$$

(Equation 4) $$A^* = \frac{\ln\left(\dfrac{1-\boldsymbol{b}}{\boldsymbol{a}}\right)}{\ln\left(\dfrac{p_1(1-p_0)}{p_0(1-p_1)}\right)} - n\frac{\ln\left(\dfrac{1-p_1}{1-p_0}\right)}{\ln\left(\dfrac{p_1(1-p_0)}{p_0(1-p_1)}\right)}$$

and:
n = # of samples
k = number of exceedances

We have provided an example of this approach for both chemical constituents and toxicity testing (Figures III-3 and III-4). Sampling is initially conducted at a set frequency (monthly) to establish that the effluent is meeting water quality thresholds. After the sufficient number of non-detects or non-toxic samples occurs in this initial tier, the sampling frequency is reduced (twice per year) and remains low, until an exceedance occurs. Following an exceedance, the initial sampling frequency is re-initiated (monthly) to either confirm that the problem is real, or to re-establish that the effluent is meeting water quality thresholds. Additional exceedances within the initial sampling tier make this phase longer. If excessive exceedances occur then sampling frequency is further increased (weekly). However, if chronic exceedances occur, management actions such as a Pollution Minimization Program (PMP) or Toxicity Identification Evaluation (TIE) are necessary.

Indicators

Regardless of which approach is used, there is a predetermined list of indicators that need to be measured. This list of constituents includes those named in the most recent 1997 Ocean Plan (i.e. SWRCB 1997) (Table III-6). These are the constituents for which regulatory thresholds are defined and for which management actions are required if the thresholds are exceeded.

For measures of effluent toxicity, dischargers should primarily focus on chronic toxicity. Acute tests are less sensitive than the sublethal toxicity tests, and do not provide any additional data for decision-making. Therefore, we suggest adopting the proposed SWRCB plan of eliminating acute toxicity testing requirement for dischargers with dilutions less than 320:1. However, as constrained by the 1997 Ocean Plan, we include acute toxicity testing in the list of constituents to measure, with a recommendation for improvement. The acute toxicity limit in the 1997 Ocean Plan still retains some of its technology based origin, where acute tests were first used as part of plant operations evaluation, not environmental protection. In order to be more environmentally relevant, the 1997 Ocean Plan Table A acute toxicity test Effluent Limitation should be replaced with a Table B Water Quality Objective that includes a dilution factor.

Q2: What is the mass of selected constituents that are discharged annually?

As with the first question, the primary design consideration for the mass emission question is sampling frequency. Ideally, managers would provide a level of precision with which to estimate mass emissions when faced with decision-making. However, estimates of precision for mass emissions have not been developed in the current regulatory framework and when quizzed, local managers were unable to cite a requisite precision for management decision-making when making comparisons among sources. Therefore, we recommend that sampling frequency be driven by the other two questions addressed in this chapter.

A second design element for this question is the level of detection, as present estimates of mass emissions are significantly hampered for selected constituents by data sets that are truncated by non-detectable analytical results. Therefore, we recommend that a special study be conducted using ultra-low detection limits once every five years to better estimate the actual concentration for constituents that are frequently below the reporting level. These concentrations would be used to estimate mass emission rates for the succeeding five years. In order to compare emission rates among discharges, we encourage all agencies to participate in a regional effort that uses the same analytical lab to analyze all the samples. We expect these methods to be other than the standard or EPA approved methods currently used in most labs. The goal of these special studies is not regulatory compliance, but to identify a concentration that can be used for estimating mass emission. If a particularly facility wants greater precision for this mass emission estimator, they may wish to conduct more than one special study every five years. If the constituent is not detectable during the special study(ies), then the ultra-low reporting level would be used as the estimator for calculating mass emissions.

Indicators

The list of indicators for this question is not the same list as the 1997 California Ocean Plan. Instead, we recommend selecting only those constituents that either 1) accumulate in the environment; 2) are on the §303(d) list of impaired waterbodies; or 3) exceed levels of concern in more than 15% of the sediments in the SCB (Table III-6). In this case, we recommend that levels of concern in the sediment equate to the Effects Range – Low as defined by Long et al. (1995). The rationale for using these three criteria is that it focuses only on those constituents that have the potential to cumulatively cause concern to environmental managers. This adaptive monitoring strategy enables regulators and dischargers to add or remove constituents and allows them the flexibility to deal with new and/or historical chemicals.

Q3: Is the effluent concentration or mass changing over time?

Once again, the major design element for estimating trends in concentrations or mass emissions is sampling frequency. Sample design parameters for trend analysis include quantifying the variability in existing effluent concentrations, identifying the amount (percent) of change managers wish to detect, the amount of time over which the change should occur, and the level of confidence managers need for assessing that change. In general, the larger the variance, the smaller the increment of change, the shorter the time period, or the more confidence a manager needs will translate into greater sampling frequency.

In order to estimate the number of samples necessary to detect trends with specified levels of confidence, we recommend using the regression-based model described by Gerrodette (1987), which serves as a useful approximation for a host of trends including both linear and exponential:

(Equation 5) $$n \geq \sqrt[3]{\frac{12cv^2\left(z_{\alpha/2} + z_b\right)^2}{r^2}}$$

where:
n = number of samples,
$cv = \dfrac{s/\sqrt{n}}{\overline{x}}$ is coefficient of variation
á = the desired confidence level,
r = the proportional change per year, and
z = probability estimate from normal distribution table (z score).

Optimally, we would recommend a sampling frequency consistent with the level of trend detection and with the desired confidence needed by mangers for decision-making. There is no regulatory mandate for assessing trends in effluent concentrations or mass. Although trend information has been widely used to assess environmental stewardship, managers in southern California were quizzed and were unable to identify a desired trend detection capability. Therefore, we recommend that sampling frequency for trend detection be dealt with on a site-specific basis as local management needs dictate. In absence of a predefined level of trend detection locally, we recommend choosing a sampling frequency that maximizes cost efficiency. In this case, power analysis can be used to identify the most efficient monitoring frequency that will detect the greatest amount of change for the fewest number of samples.

As an example, the most cost-efficient sampling frequency was identified using power analysis by examining effluent data from the four large POTWS between 1989 and 1996. The inflection point in these curves indicates the point of diminishing returns. For lead, the optimal number was approximately 6 samples annually for each of the four large POTWs. Although the number of samples at each POTW was similar, the amount of trend detection was not. In this example, the amount of trend detection at 6 samples per year varied from about 15% over five years at Point Loma to 150% over five years at OCSD. The difference in trend detection is due to the variability in effluent concentrations among facilities. In the end, the cost efficiency approach may yield similar frequencies among facilities, but will likely yield different capabilities to detect trends.

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TABLE III-1. Number of effluent constituent measurements per year. HTP = L.A. City Hyperion Treatment Plant; JWPCP = LACSD Joint Water Pollution Control Plant; OCSD = Orange County Sanitation Districts; PLWTP = City of San Diego Point Loma Wastewater Treatment Plant.

Constituent	HTP	JWPCP	OCSD	PLWTP
General	4487	4227	1239	3270
Metals	172	132	148	988
Organics	244	268	528	1212
Toxicity				
Acute	12	12	4	12
Chronic	12	12	12	12

TABLE III-2. Effluent constituent analysis frequency.

Constituent	Daily	Weekly	Monthly	Quarterly
General				
Suspended solids, Total BOD	HTP JWPCP OCSD PLWTP	-	-	-
Turbidity	HTP JWPCP PLWTP	-	OCSD	-
Floating particulates	PLWTP	-	HTP	-
Oil and Grease	JWPCP PLWTP	HTP	OCSD	-
Total dissolved solids	PLWTP	HTP	-	-
Volatile susp. solids	PLWTP HTP	-	-	JWPCP
TOC	-	HTP JWPCP	-	-
Residual Cl^-	HTP JWPCP	-	-	-
Ammonia-N	-	OCSD PLWTP	HTP JWPCP	-
Nitrate-N	-	PLWTP	HTP JWPCP	-
Nitrite-N	-	-	JWPCP	-
Phosphate	-	PLWTP	JWPCP	-
Total phosphorus	-	-	HTP	-
Cyanide	-	PLWTP	HTP JWPCP OCSD	-
Total coliforms	JWPCP	HTP	PLWTP	-
Enterococcus	JWPCP	HTP	-	-
Fecal coliforms	-	HTP JWPCP	-	-

TABLE III-2 (Continued)

Constituent	Daily	Weekly	Monthly	Quarterly
Metals/Metalloids				
Al, Ba, Co, Fe, Mn, V	-	PLWTP	-	HTP
Sb, Be	-	PLWTP	OCSD	HTP JWPCP
As, Hg, Cd, Cu, Pb Ni, Se, Ag, Zn	-	PLWTP	HTP JWPCP OCSD	-
Total Cr	-	PLWTP	HTP JWPCP	-
Hexavalent Cr	-	-	HTP OCSD	-
Th	-	PLWTP	-	HTP JWPCP OCSD
Organics				
DDTs, PCBs, Chlor. phenols	-	PLWTP	JWPCP OCSD	HTP
Nonchlor. phenols, Other Cl^- pesticides	-	PLWTP	OCSD	HTP JWPCP
Organotins	-	-	PLWTP	HTP JWPCP
PAHs, Benzidines, Acrolein, Dioxin, Acrylonitrile, Other VOCs, Purg. aromatics, Other base/neutral extractables	-	-	OCSD JWPCP PLWTP	HTP
Toxicity				
Acute	-	-	HTP JWPCP PLWTP	OCSD
Chronic	-	-	HTP JWPCP OCSD PLWTP	-

TABLE III-3. Effluent constituent reporting levels.

Constituent	HTP	JWPCP	OCSD	PLWTP
Metals/Metalloids (µg/L)				
Aluminum	100	-	-	50
Antimony	5	0.3	4	23
Arsenic	1	0.4	2	0.18
Barium	10	-	-	10
Beryllium	0.3	0.5	0.6	0.39
Cadmium	2	0.8	0.1	1
Hexavalent chromium	10	-	-	-
Total chromium	4	20	1	5
Cobalt	2	-	-	4
Copper	10	4	1	4
Iron	20	0.4	-	30
Lead	3	8	1	18
Manganese	10	-	-	4
Mercury	0.3	0.04	0.2	0.27
Molybdenum	10	-	2	-
Nickel	5	10	2	14
Selenium	1	0.1	2	0.4
Silver	0.4	4	2	6.6
Thallium	5	30	4	40
Vanadium	5	-	-	7
Zinc	10	15	2	4
Organics				
Organotins (µg/L)	0.005	0.098	-	0.1
Phenols (µg/L)	-	2-19	5	-
Chlorinated phenols (µg/L)	1-7	2-16	3.3-6.9	1.6-6.1
Nonchlorinated phenols (µg/L)	1-34	2-19	2.6-11	1.8-6.1
DDT (µg/L)	0.002-0.01	0.01-0.03	0.02	0.02-0.04
PCB (µg/L)	0.025-0.065	0.08-0.9	0.3	0.07-0.6
Purgeable aromatics (µg/L)	0.04-0.08	0.3-1.0	0.18-0.58	1-2.9
Benzidines (µg/L)	2, 14	0.101	20	40, 170
PAHs (µg/L)	1	0.015-0.42	1-10	0.8-7.8
Dioxin (ng/L)	0.0008-0.0017	3,000	50,000	0.0008-0.008

TABLE III-4. 1997 Ocean Plan objective based effluent limitations considering dilution credit, and discharger effluent reporting levels. Underlined values indicate reporting levels greater than the 1997 Ocean Plan objective based effluent limit.

	1997 Ocean Plan Objective based effluent limit (µgL)				Reporting level (µg/L)			
Constituent	HTP	JWPCP	OCSD	PLWTP	HTP	JWPCP	OCSD	PLWTP
endrin	0.168	0.332	0.296	0.408	0.004	0.02	0.007	30
aldrin	0.002	0.004	0.003	0.004	0.008	0.01	0.004	0.02
benzidine	0.006	0.011	0.010	0.014	47	0.1	20	40-170
chlordane	0.002	0.004	0.003	0.005	0.005	0.01-0.04	0.27-0.06	0.048
3,3-dichlorobenzidine	0.680	1.345	1.199	1.652	2	0.14	20	40
dieldrin	0.003	0.007	0.006	0.008	0.006	0.02	0.005	0.04
hexachlorobenzene	0.018	0.035	0.031	0.043	1	1	4	1.4
PAHs	0.739	1.461	1.302	1.795	1-2	0.015-0.42	1-10	0.8-7.8
PCBs	0.002	0.003	0.003	0.004	0.025-0.065	0.08-0.9	0.3	0.07-0.6
TCDD equivalents	$3x10^{-7}$	$6x10^{-7}$	$6x10^{-7}$	$8x10^{-7}$	0.0003-0.001	1	50	0.000093
toxaphene	0.018	0.035	0.031	0.043	0.113	0.3	0.23	0.24
DDT	0.014	0.028	0.025	0.035	0.002-0.010	0.01-0.03	0.007-0.039	0.02-0.04

TABLE III-5. Annual samples necessary to achieve 99% confidence that effluent is within 1997 Ocean Plan effluent limitations or water quality objectives.

Constituent	HTP	JWPCP	OCSD	PLWTP
Silver	2	<1	2	<1
Arsenic	<1	<1	<1	<1
Cadmium	<1	<1	<1	<1
Cyanide	52	2	<1	<1
Chromium	<1	2	<1	<1
Copper	<1	<1	2	2
Mercury	<1	2	52	2
Ammonia - N	<1	<1	<1	<1
Nickel	<1	<1	<1	2
Lead	<1	<1	2	2
Selenium	<1	<1	<1	<1
Zinc	<1	<1	<1	<1
Acute Toxicity	84	180	12	180
Grease & Oil	36	24	12	360
Total DDT	Huge #	Huge #	Huge#	Huge#

Table III-6. Chemical constituents to be analyzed in the effluent monitoring program, listed under the category they were selected.

Constituent	Partial list of 1997 California Ocean Plan Effluent Constituents	15% of SCB sediments greater than ERL	Local 303(d) list	Bioaccumulative
Antimony	X			
Arsenic	X	X		
Cadmium	X	X		
Chromium	X	X		
Copper	X	X	X	
Lead	X	X	X	
Mercury	X	X	X	X
Nickel	X	X		
Selenium	X			X
Silver	X	X	X	
Zinc	X	X	X	
Thallium	X			
Cyanide	X			
Total chlorine residual	X			
Ammonia	X			
Phenolic compounds	X			
Organotins	X			
Other chlorinated pesticides (e.g., chlordane)	X		X	X
Dioxins	X			
PAHs	X	X		
DDTs	X	X	X	X
PCBs	X	X	X	X
Purgeable aromatics	X			

ERL = Effects Range Low (Long et al.. 1995)
303(d) list = Inventory of impaired waterbodies that California reports to USEPA

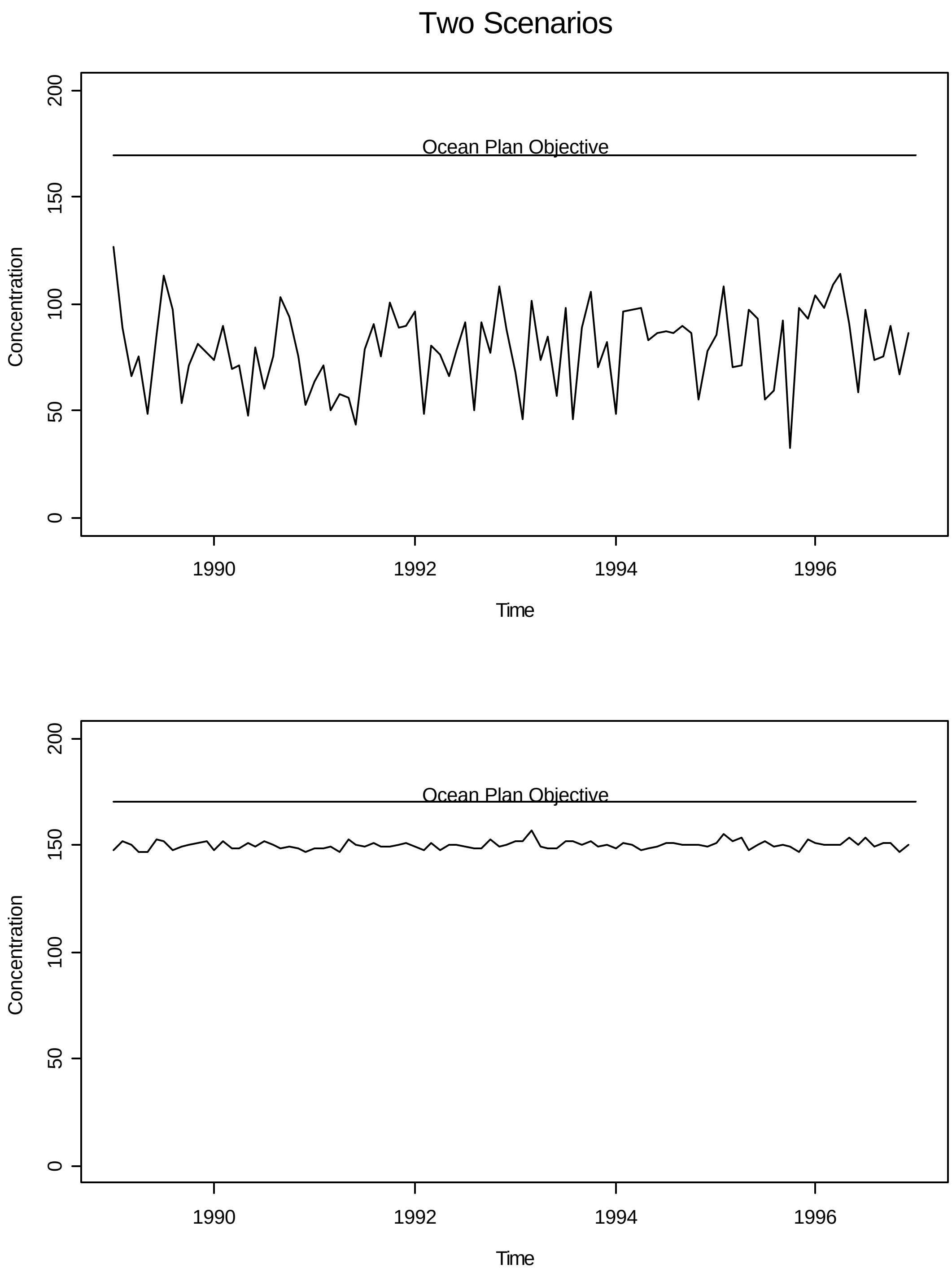

Figure III-1. Effluent constituent variability relative to California Ocean Plan objectives. Increases in variability are more tolerable with distance from the objective.

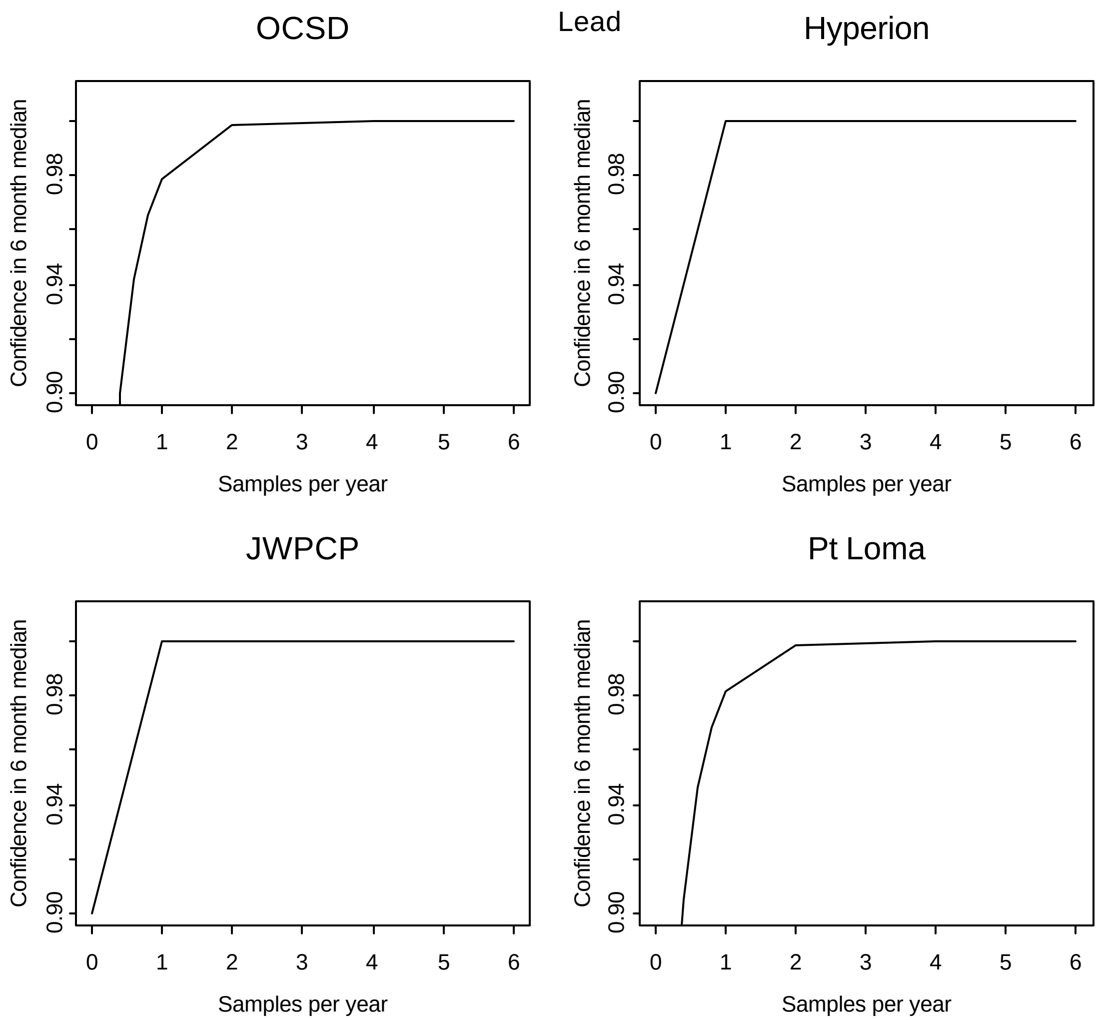

Figure III-2. Sampling effort required to achieve an acceptable level of confidence for lead effluent concentrations. Power analysis was used with the historical discharge data from 1989-1996 for each of the four dischargers.

P_0 = 0.1, P_1 = 0.5, á$_1$ = 0.1, á$_2$ = 0.001, â = 0.05

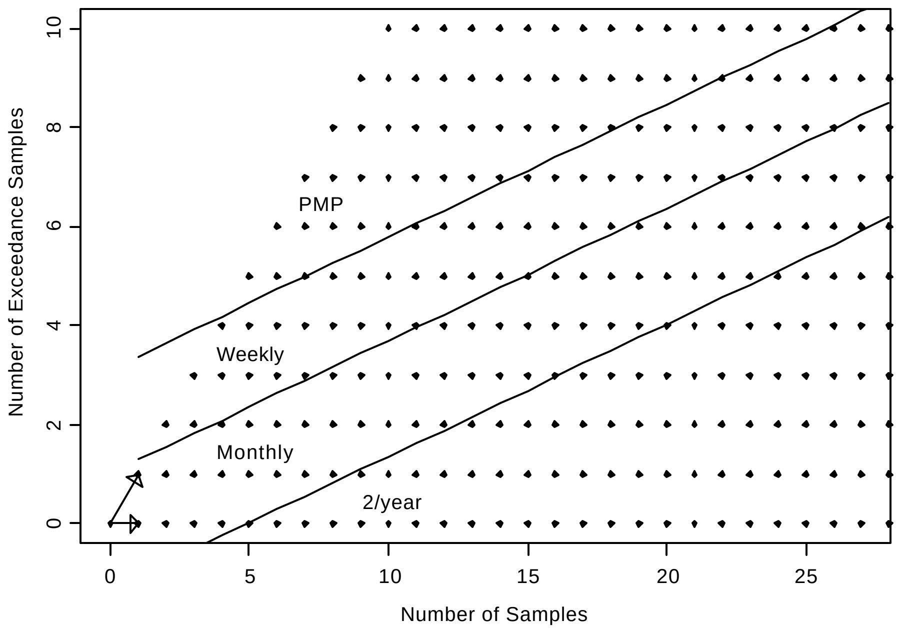

Figure III-3. Sampling frequency approach for chemicals with reporting levels near 1997 Ocean Plan limits. P_0 = acceptable probability of exceedance (used for lowest line). P_1 = unacceptable probability of exceedance (used for upper line(s)). á$_1$ = probability that increased sampling is mandated when the probability of exceedance is actually below P_1. á$_2$ = probability that management action (Pollution Minimization Program, PMP) is mandated when the probability of exceedance is actually below P_1. â = probability that reduced sampling is mandated when the probability of exceedance is actually above P_0.

$P_0 = 0.075$, $P_1 = 0.2$, á = 0.1, â = 0.2

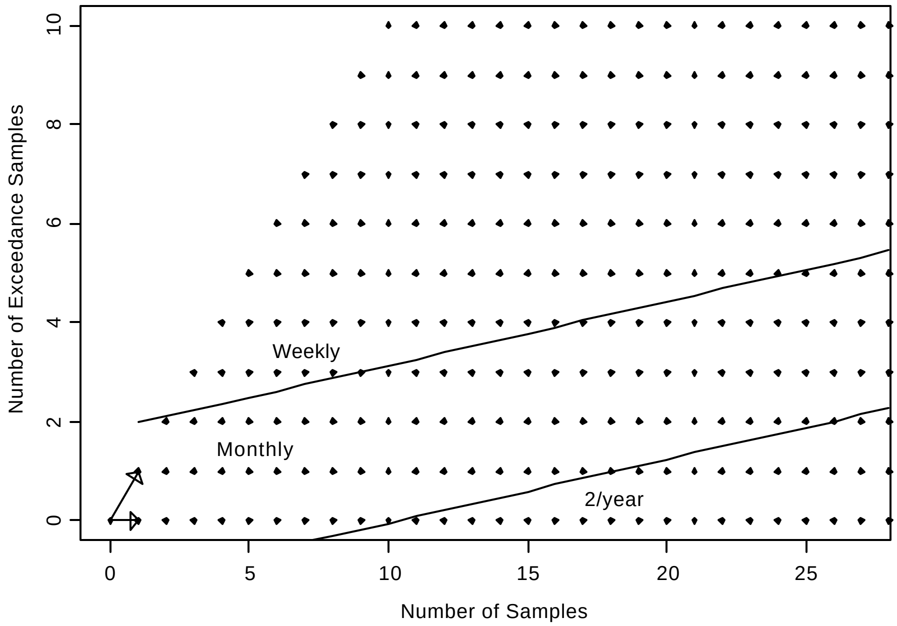

Figure III-4. Sampling frequency approach for chronic toxicity. P_0 = acceptable probability of exceedance (used for lower line). P_1 = unacceptable probability of exceedance (used for upper line). á = probability that management action (Toxicity Identification Evaluation, TIE) is mandated when the probability of exceedance is actually below P_1. â = probability that reduced sampling is mandated when the probability of exceedance is actually above P_0.

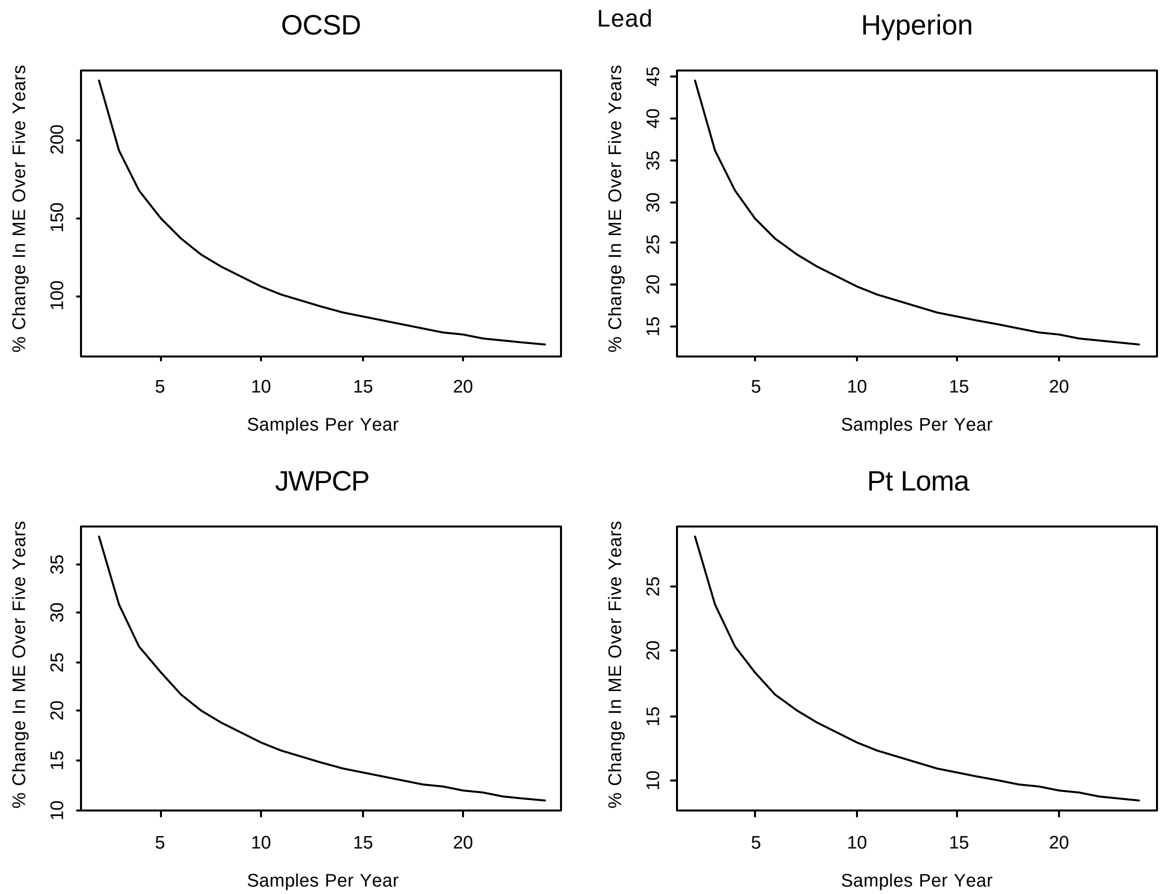

Figure III-5 Percent change in mass over 5 yr vs. # samples/yr

IV. MICROBIOLOGICAL MONITORING

There are two management questions for microbiological monitoring:

Q1: Does sewage effluent reach water contact zones?

Q2: Are densities of bacteria in water contact zones below levels that will ensure public safety?

Each microbiological monitoring program consists of an offshore, a nearshore, and a shoreline monitoring component. Offshore monitoring is used to track the effluent plume. Bacteria are a sensitive tracer of the effluent plume in offshore areas because there are no other sources of these bacteria in the offshore marine environment. In contrast, shoreline monitoring, which was originally designed to help survey for waste plumes encroaching on the beach and for tracking spills into the storm drain system, is now relied upon in part to assess public water-contact safety. While compliance issues along the shoreline are still important to ocean dischargers, this question has been refocused on public health and safety. This is because the county health departments, which have the responsibility to close or post beaches in response to high bacterial counts, have grown dependent upon the POTW shoreline monitoring data. Nearshore monitoring is conducted for both plume tracking and water-contact safety purposes. For one agency, nearshore monitoring also serves to address shellfish safety purposes.

Comparison and Contrast Among Agencies

The level of microbiological monitoring is disproportionate among POTW programs. A five-fold difference was found in the number of analyses conducted per year (Table IV-1), which reflects differences in sampling frequency, number of stations sampled, and number of indicators measured. Sampling frequency at shoreline stations varies from daily (5-7 times per week) to biweekly sampling. The number of shoreline stations sampled is somewhat comparable among agencies, differing by a factor of only two. The number of nearshore and offshore stations are not comparable; cumulatively, they differ by a factor of eight. Indicators and methods are consistent for shoreline stations, all POTW programs measure fecal coliform, total coliform and enterococcus; however the nearshore and offshore analyses are less consistent. The PLWTP and JWPCP analyze for total coliform, fecal coliform and enterococcus at nearshore stations, while HTP analyzes exclusively for total coliform, and OCSD targets fecal coliforms.

Sampling is also disproportionate between POTWs and county health programs. The number of annual analyses the four largest POTWs conduct along the shoreline is almost twice the number of analyses as the county health departments (Table IV-2). Part of this difference is that POTWs analyze more indicators than the health departments (Table IV-3). Most POTWs measure all three indicators while, historically, most health departments have rarely measured more than two. While POTWs process more samples, they tend to sample fewer stations than their health department counterparts. This reflects a difference in sampling frequency between the two groups. County health departments sample each site weekly to monthly, whereas POTWs sample most sites multiple times per week (Figure VI-1). The POTWs and health departments also differ in the methodologies they use for processing samples. The POTWs have continued to rely on membrane filtration, whereas health departments have transitioned to the use of chromogenic substrate tests, which cost 75% less than membrane filtration.

Evaluation of Existing Effort

Monthly surveys have repeatedly demonstrated that plumes from large POTWs typically stay far from shore and are usually submerged, particularly under average conditions such as when the water column is stratified (e.g., strong thermocline). However, the present monitoring strategy of collecting samples at set intervals is not designed to catch the rare events, when the plume is most likely to surface or move towards shore, such as during storm events or Santa Ana conditions. Therefore, offshore microbiological monitoring should become an adaptive component of a water quality monitoring program that provides real-time information about plume location. Thus, offshore monitoring would not be conducted on a continual basis, but would be focused only on those rare events when bacterial encroachment on areas of human water contact is likely. Individual agencies, however, have indicated that the current approach to offshore plume tracking is not a large burden, and that monthly offshore monitoring is something the public deems worthy.

The current offshore monitoring strategy is inefficient because multiple indicators are used to track the plume. Multiple indicators should be measured at times or locations when body-contact issues are of concern, such as kelp beds or other swimming areas. However, microbiological data analysis of offshore monitoring does not focus on comparisons to AB411 or 1997 California Ocean Plan standards; rather it is compared to background levels to identify the presence and concentration of the plume. Therefore, a single indicator could be used for plume tracking. Total coliform is the most sensible of the three indicators since it is not found naturally in the marine environment and is the most concentrated of the three indicators presently measured. Of course, using bacteria would be inappropriate indicators for effluents that are disinfected.

Nearshore monitoring is ineffective. As with offshore monitoring, the rare events that may drive the plume toward shore are unlikely to be captured using the current set sampling schedule. Except for those sites in kelp beds, nearshore monitoring provides little new information to human health protection that shoreline monitoring does not already provide. Because the data produced from nearshore monitoring does not add any information to plume tracking or public water-contact safety (except where nearshore water contact or shellfish harvesting areas), it is not used to make management decisions, and should be discontinued.

Shoreline monitoring to detect sewer intrusion into storm drains is effective, but inefficient. Sanitary sewer incursions are not the only, and in many cases not even the primary, source of bacteria to storm drains. Although the stormwater National Pollutant Discharge Elimination System (NPDES) permittees in the Southern California Bight have the responsibility to maintain storm drain systems and check for illicit connection and illegal discharges, none conduct shoreline monitoring. Not only does this represent an inappropriate allocation of costs, but without participation by storm drain managers in the monitoring program, no formal mechanism exists to identify and resolve problems that are discovered. For example, sanitary surveys may be an appropriate adaptive monitoring strategy when chronic bacterial exceedances occur near a storm drain outlet. However, without a management framework that includes both POTWs and storm drain managers, there is no coherent system to perform these surveys and efficiently identify or resolve water quality problems.

Shoreline monitoring for public health is inefficient. The public health portion of the shoreline monitoring effort is not well integrated with the county health department monitoring and, as a result, is inefficient in most cases. The POTWs sample more frequently, measure more indicators and use more expensive methods than the health departments, even though the data are being placed into a common data set for a common purpose. If the purpose of monitoring sites away from storm drains is primarily to provide data to the county health department to make decisions, then monitoring design should be integrated and comparable among POTWs, county health departments, and stormwater dischargers.

Recommended Sampling Design

Shoreline Monitoring Design Issues

There are a variety of design-related issues for building a shoreline model monitoring program that integrates the needs of POTWs, regulators, public health departments and stormwater agencies. These include five temporal sampling design issues: 1) number of days per week to sample, 2) days of the week to sample, 3) allocation of effort among seasons, 4) allocation of effort for storm events, and 5) time of day or tidal stage to sample. There are also three spatial design issues: 1) appropriate distance between sampling sites, 2) proximity of sampling sites to a storm drain, and 3) depth and distance from shore. Two additional design issues include which bacterial indicators to measure and which methods to be used. Finally, a model program should address allocation of effort among agencies in a cooperative shoreline monitoring program.

Temporal

Number of days per week to sample. There are several types of coastal areas in southern California, including wetlands, rocky shorelines, ports and sandy beaches. Each of these areas differ in the amount and type of public recreation available; for example the number of visitors and the type of water-contact activity at a wetland is likely to be quite different from that at a popular surfing or swimming beach. In contrast, port and military-controlled areas usually are inaccessible to the public; consequently no recreational activities are expected. Current shoreline monitoring strategies, however, are not optimized by use; existing strategies do not prioritize sampling frequency among sites based on the amount of water-contact expected at a given location, yet shoreline microbiological monitoring is conducted to assess public water-contact safety. Sampling frequency should be more closely correlated with the type and amount of usage at each site; high use swimming beaches should be sampled more frequently than low use areas not expected to have water-contact activities.

Likewise, the current strategies are not optimized in terms of risk of bacterial exposure. The risk of exposure will be different among sites, depending if there is a potential source nearby, and if the source has a history of contamination problems. Specifically, beaches near chronic sources of contamination, such as certain storm drains, should be sampled more frequently than beaches near potential sources without histories of problems. Similarly, beaches without known sources nearby (e.g., no sewage lines or storm drains), should be sampled the least.

Therefore, shoreline sampling frequency should be correlated with both the amount of beach usage and the potential risk of exposure for each site (Table IV-4). Beaches with lifeguard stations are assumed to be high use areas, and should be given a higher priority over accessible sandy beaches without lifeguards. Similarly, high risk areas should be given a higher priority over areas with lower risk. Following the recommendations in Table IV-4, monitoring will range from no sampling at inaccessible shoreline areas, to daily sampling at high use, lifeguarded areas near chronic sources of contamination.

The actions a manager is likely to take following an exceedance will depend on the magnitude or persistence of the problem. Managers will want to know whether the problem is a random hit, in which case there may be no action taken, or if there is a spill that demands additional sampling and immediate attention. Therefore the sampling frequency should be adaptive, with increased monitoring enacted based on the magnitude of the exceedance (e.g., all three bacteria indicators fail at twice the standard), or if the problem is persistent (e.g., two consecutive failed samples).

Days of the week to sample. For sites with less than a daily sampling frequency, the most appropriate day of the week to sample should be decided by the individual monitoring agency. Daily monitoring by the City of L.A. for 20 stations over a 5 year period has indicated that no single day of the week had a higher proportion of exceedances (Table IV-5). When no disproportionate number of beach failures occur at any point during the week, the sampling day should not make a difference. Therefore, selection of which day to sample should be left to the monitoring agency.

Allocation among seasons. Because shoreline microbiology monitoring is conducted to assess public water-contact safety, the most appropriate sampling strategy should be based on the potential risk of bacterial exposure and the amount of beach usage at each site. Since risk patterns and beach usage do not strictly follow the calendar, monitoring frequency should be adjusted according to each site throughout the year (Table IV-4) and not necessarily by season.

While usage is down at some areas during the winter, reflecting amount of tourism and school schedules as well as reductions in air and water temperatures, surfing areas have consistently high usage throughout the year. Therefore sampling frequency should not be reduced at all areas during the winter.

In terms of risk, there is a common misconception that winter months have a higher proportion of failures, coinciding with storm events. While sewage collection systems do fail most often during rainstorms, five years of daily monitoring data at Santa Monica Bay stations indicate the total number of shoreline bacteria exceedances are comparable between summer and winter months (Schiff et al.. 2001).

Allocation to storm events. Sewage collection systems tend to fail most often during large storm events. However, the most appropriate time to sample in relation to storm events is not clear. Sampling could be conducted during a storm to catch the highest bacterial counts, or conducted after the storm since there is a standard three day warning period for the public to avoid water-contact following a storm event. Alternatively, sampling could be performed at the normally scheduled interval, regardless of when the storm occurs.

Because of the standard three day warning period, we believe that sampling during a storm event is unnecessary, and is potentially hazardous to the person collecting the sample. Therefore, we decided it was most appropriate to sample on the second day following a storm event. This would be safest, and would enable data to be available to extend the standard three day warning period, if necessary.

Time of day/tidal stage. The limited number of studies that have examined the effects of tide or time of day on bacterial concentrations are inconsistent. In some areas, tidal phase does appear to have an affect on bacterial concentrations; for example bird feces can be flushed from marshes during outgoing tides, or can be mobilized into the water from the shoreline during incoming tides. In other areas, highest bacterial concentrations are found in the early morning.

The alternatives considered for sampling time were 1) sample at a fixed time of day, eventually catching all phases of the tide, 2) select a particular tide phase to sample, and 3) allow the individual labs to identify the most appropriate time to monitor. Because this appears to be a local issue, it was decided to allow each agency to determine the most appropriate time of day or tidal stage to sample.

Spatial

Distance between sampling sites. The current sampling distance between stations is different among POTWs, and there appears to be little justification for these distances. Two options were identified for shoreline monitoring: 1) use a rotating panel sample design, going to a different set of sites each week, and 2) sample at set stations, at a set distance. While the rotating panel design would give better spatial coverage, we believe the time between samplings at a given site would not allow adequate temporal characterization of a station. Therefore, sampling should be at set stations, located at set distances along a beach. A one mile increment between stations was selected because we believe this represents an adequate balance between spatial coverage and sufficient resolution to identify problems. Because this distance is not based on empirical data, further work should be done to assess the adequacy of this distance. This set distance between sites can be shortened if local issues, such as beach usage and potential risk of exposure, so dictate. Local geographical characteristics that make an area inaccessible (e.g. cliffs) may also alter the recommended one mile monitoring distance.

In order to locate the source of an exceedance and better define the extent of the problem, additional samples should be collected 50 yards on either side of a station that fails. This extra effort would occur following a trigger of either the magnitude of exceedance (e.g., all three indicators are greater than twice the standard), or its persistency (e.g., two consecutive failed samples).

Storm drain sampling locations. Over half of the shoreline near freshwater outlets fail bacterial standards (Figure IV-2), and about 10% of the shoreline within 100 yards of an outlet fail. Clearly, storm drains can be major conveyances of bacteria to the marine environment. However, current monitoring near storm drains is inconsistent among agencies, with no justification for the sampling distances. Current sampling near storm drains ranges from within the wavewash of the outlet to 83 yards from the drain. Because sampling is usually conducted on only one side of the drain, and the direction of the flow from the outlet is not consistent and usually not known, the freshwater plume could be missed entirely if the sample is collected at the wrong place. Similarly, sampling within the wavewash does not give an accurate assessment of bacteria concentrations away from the outlet, where most people are likely to swim.

Three alternatives were proposed for sampling near storm drains: 1) sample at a set distance from the drain, 2) develop models that allow samples collected within the outlet to be used to extrapolate bacterial concentrations at a given distance from the drain, and 3) develop models to sample in the wavewash and extrapolate with distance. Permanent signs warning people to avoid water contact near drains should be posted on both sides of the outlet regardless of which of the three options is selected. Twenty-five yards was selected as the distance from drains to post signs because the current warning signs that are used are readable from this distance.

Because models are not yet available to implement the second or third options, we decided that sampling should be conducted at a set distance from each drain. Therefore at storm drain locations, sampling should be conducted 50 yards along the shore from the drain (25 yards from the posted signs), in the direction of the flow.

It was recognized that sampling at a set distance from storm drains is not based on empirical data. The models that are being developed will allow a more accurate idea of where the plume is most likely to be, and what bacterial concentrations can be expected as a function of distance from the outlet. Once developed, these models should replace or augment the method of sampling at a set distance.

Appropriate depth. Two options for the most appropriate depth for collecting shoreline samples were either knee or ankle deep water. While infection is caused most often by ingestion of contaminated water, and people are more likely to submerse their head in knee-deep water, ankle deep water was used in the Santa Monica Bay Restoration Project epidemiological study used to derive California State Assembly Bill 411 (AB411) bacterial standards. In addition, ankle deep water is where most small children (the most susceptible age group) play, and the safest depth to collect the samples. Therefore, we decided that samples should be collected from ankle deep water. Furthermore, samples should be collected on the incoming wave, to reduce the possibility of contamination from the person taking the sample.

Indicators

Appropriate indicators. Three indicators of bacteria are used by POTWs and health departments: total coliform, fecal coliform, and enterococcus. Each of these indicators is now required for shoreline monitoring by emergency regulations adopted under AB411. Therefore, all three indicators should be included in a shoreline microbiology sampling program.
Two alternative options considered were 1) use the indicator that has the greatest overlap among tests, as suggested by the limited data available, and 2) measure one indicator nominally, and augment with the other two tests when high values are encountered. However, since AB411 initiates mandatory analysis of all three indicators, these two options are not appropriate.

Appropriate methods. Three methods of detecting bacterial indicators are currently used in southern California: membrane filtration, multiple tube fermentation, and chromogenic substrate kits. The various methods give comparable results, but there are differences in the amount of time and effort. The chromogenic substrate IDEXX kits are the fastest methods available and are 75% cheaper than the membrane filtration methods currently used. Since these methods are still relatively new, however, some concerns exist regarding the accuracy of the IDEXX kits.

Two approaches were considered 1) allow each lab to use any approved method, and 2) use the most rapid analytical method available, as long as it has been approved as a valid technique. We decided to recommend the approach of using the fastest method. The reason being that a faster turn around time of the data means quicker posting of beaches, if necessary. As of January 2001, IDEXX kits have been officially approved by State Health Service, and informally approved by USEPA. However, if the monitoring question reverts back to a compliance issue, we recommend that dischargers continue the use of traditional membrane filtration or multiple tube fermentation methods until the accuracy of IDEXX kits are further validated.

Other Issue

Who pays?/resource allocation. The current shoreline monitoring places a disproportionate burden on POTWs, with little or no monitoring from other dischargers of bacteria (e.g., stormwater agencies). However, since sewer and stormwater agencies are separate in some counties, and combined in others, this is a county-specific issue. Therefore, allocation of resources for shoreline microbiological monitoring should be determined within each county.

References

Schiff, K.C., J. Morton and S.B. Weisberg. 2001. Retrospective evaluation of shoreline water quality along Santa Monica Bay beaches. Annual Report 1999-2000. S.B. Weisberg and D. Elmore (eds). Southern California Coastal Water Research Project. Westminster, CA.

TABLE IV-1. Number of microbiological analyses per year.

	Shoreline	Nearshore and Offshore	Total
HTP	14,220	9,000	23,220
JWPCP	2,916	3,020	5,936
OCSD	3,840	624	4,464
PLWTP	1,872	4,320	6,192
Total	22,848	16,964	39,812

TABLE IV-2. Comparison of annual effort between POTWs and County Health Departments.

	Shoreline		Near/Offshore	
	No. of Sites	No. of Analyses per Year	No. of Sites	No. of Analyses per Year
4 Large POTWs	59	22,848	53	16,964
Other NPDES	212	27,423	81	7,116
Health Depts.	171	12,656	-	-
Total	442	62,927	134	24,080

TABLE IV-3. Comparison of methods used for shoreline monitoring between POTWs and Public Health Agencies.

		SCB Public Health Agencies	Large POTWs
Total coliform			
	Mult tube ferm	7,090	3,840
	Membrane filt	468	9,228
	Colilert	728	-
Fecal coliform			
	Mult tube ferm	4,282	-
	Membrane filt	-	7,332
	Colilert	728	-
Enterococcus			
	Mult tube ferm	1,932	-
	Membrane filt	-	7,326
	Enterolert	728	-
	Total	15,956	27,726

TABLE IV-4. Sampling frequency based beach usage and potential risk of exposure.

Usage	High Risk*	Medium Risk*	Low Risk*	No Known Source*
Lifeguarded, or high use dive or surf	daily or 5/week	5/week	weekly or 5/month	weekly or 5/month
Accessible sandy beach, or low use dive or surf	2-3/week	weekly or 5/month	weekly or 5/month	none
Other accessible shoreline; low use dive or surf	weekly or 5/month	weekly or 5/month	monthly	none
Inaccessible shoreline	none	none	none	none

*High risk - a source of contamination flows continually and is a known problem
*Medium risk - a source flows intermittently or flow is low but continuous, and there is an occasional contamination spike
*Low risk - a potential source exists, e.g., a public restroom or near a POTW, but is not usually a problem
*No known sources - no sewage lines are known

TABLE IV-5. Proportion of shoreline microbiological exceedances based on day of the week.

Indicator	Percent of Exceedances
Enterococcus	
Monday	12.5%
Tuesday	13.9%
Wednesday	14.4%
Thursday	14.2%
Friday	14.0%
Saturday	15.3%
Sunday	15.6%
Fecal Coliform	
Monday	13.0%
Tuesday	15.4%
Wednesday	16.0%
Thursday	14.8%
Friday	13.4%
Saturday	13.8%
Sunday	13.6%
Total Coliform	
Monday	12.4%
Tuesday	16.7%
Wednesday	18.3%
Thursday	15.4%
Friday	13.4%
Saturday	12.3%
Sunday	11.5%

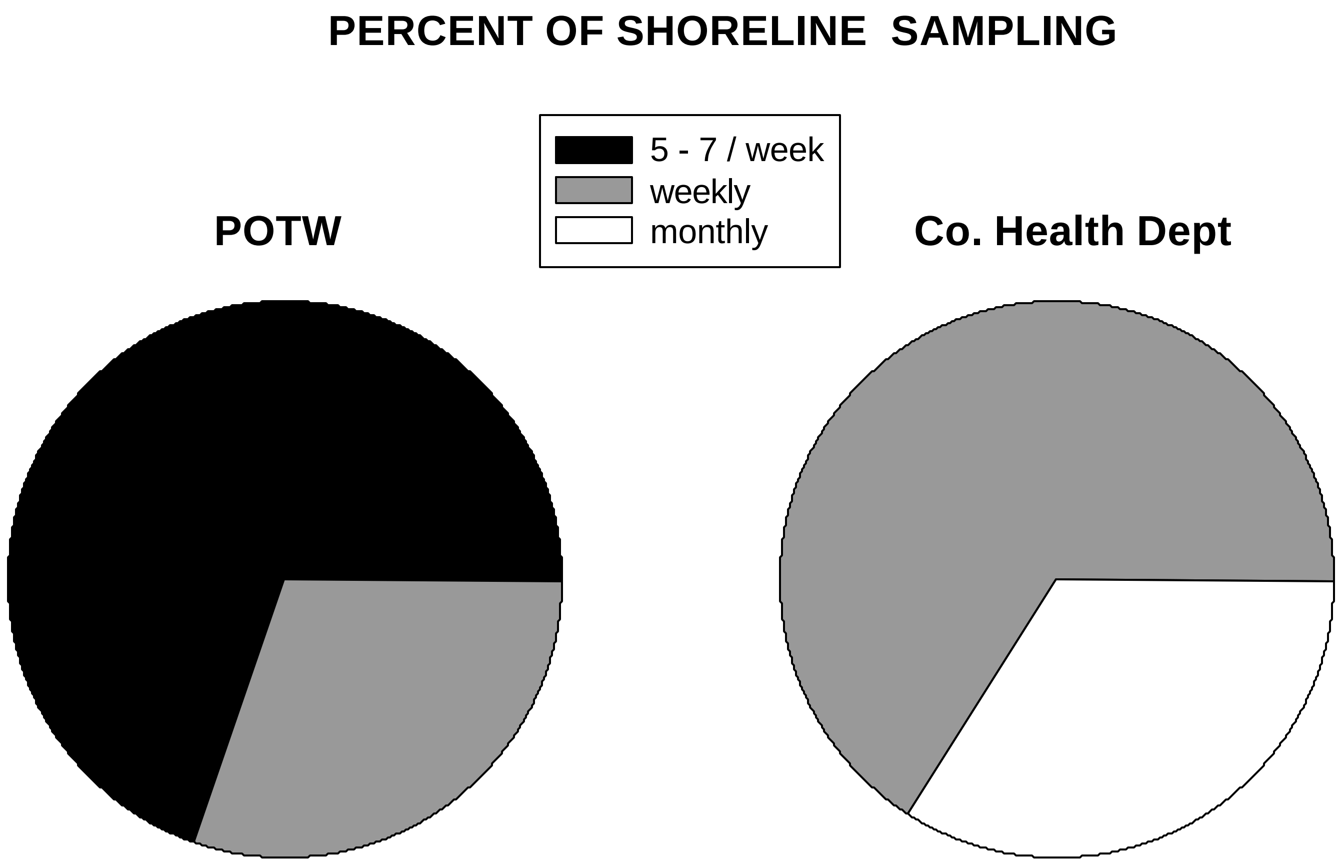

FIGURE IV-1. Frequency of shoreline sampling by Publicly Owned Treatment Works (POTWs) and County Health Departments.

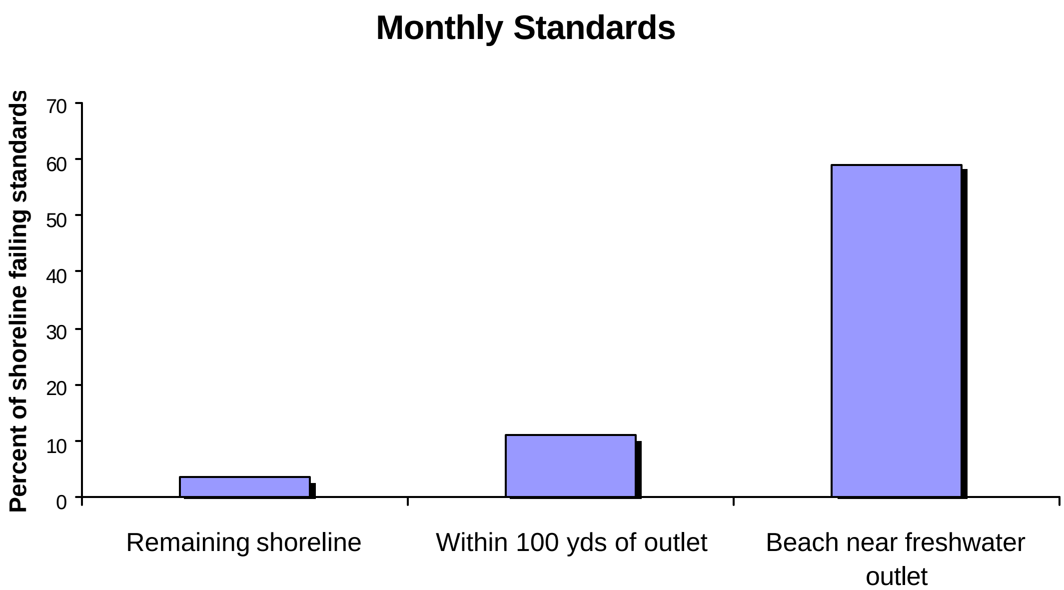

FIGURE IV-2. Percentage of shoreline failing bacterial standards near freshwater outlets, compared to the remaining shoreline.

V. WATER QUALITY MONITORING

There are two management questions for ambient water quality monitoring:

Q1: ***Are water column physical and chemical parameters within ranges that ensure protection of the ecosystem?***

Q2: ***What is the fate of the discharge plume?***

Most POTWs in the SCB have designed their outfalls to quickly mix and diffuse effluents with receiving waters. Water quality monitoring is conducted to assess if their plume has been sufficiently mixed to maintain protection of the ecosystem in receiving waters. This first question is a site-specific question and focuses on the local environmental impacts around an outfall. For instance, water column ecosystems are susceptible to a reduction in light, an alteration in pH, deprivation of dissolved oxygen (D.O.), or an increase in nutrients. Light reduction can contribute to a decrease in primary production that will have a ripple effect through the ecosystem, eventually leading to reductions in fish abundance and assemblage parameters. Alterations in pH and D.O. can have acutely toxic effects on fish and invertebrates; while not an observed problem in southern California, D.O. reductions have been responsible for fish kills in other affected ecosystems around the nation. Elevated nutrient levels can cause aesthetically objectionable algal blooms that may eventually lead to depressed D.O. levels when oxygen demand gets too high.

An equally important, but distinctly different question that managers need to know is where their plume is going. First, most managers need to know if their plume is moving towards shore or the surface where it may encroach upon water contact zones. In this case, human health concerns are of interest and water quality thresholds exist for bacteria (*see Microbiological Monitoring*). Second, plume direction and mixing has a direct effect on sediment loading. Although, light transmittance may be within acceptable levels for water column assessments, the direction of the plume determines where the discharged particles will eventually settle. Years of accumulations may affect sediments in locations where the plume direction is most consistent. In this case, ecosystem health issues are primary concerns in terms of habitat quality and impairments of benthic communities (*see Sediment Monitoring*). As one can see, this question has both a facility-specific component for local impacts as well as a regional component for when multiple plumes commingle.

Compare and Contrast Among Agencies

The level of effort expended on water quality monitoring has differed both among agencies and over time. Prior to July 1998, each of the four large POTWs had separate water quality monitoring programs and sampling effort varied significantly among agencies. This was due to inconsistencies in the number of stations sampled and the number of water quality parameters analyzed (Table V-1). Sampling frequency did not contribute to the differences in effort since each agency sampled at monthly intervals, but differences in receiving water environments (i.e., width of shelf) may have contributed to the differences in number stations. Conductivity-temperature-depth (CTD) casts were the only measurements common to all four agencies while 7 other parameters were measured by three or fewer dischargers, most notably nutrients and chlorophyll fluorescence.

In July 1998, three of the dischargers (HTP, LACSD, and OCSD) altered their water quality monitoring programs to initiate the Central Bight Cooperative Program (CBCP), which also included the City of Oxnard. The design of the CBCP altered what was once a more temporally intensive, but spatially limited facility-specific monitoring program to a more spatially intensive, but temporally limited regionally-oriented program (Figure V-1). For example, water quality monitoring used to be conducted monthly at 76 sites and is currently conducted quarterly at 145 sites (Table V-1). The PLWTP, however, still maintains its original facility-specific monitoring program from 1995. As a result, the effort in water quality sample collection among the four largest dischargers now varies by a factor of five. However, the level of similarity among programs bightwide has increased because the three participating agencies use comparable sampling methods, measurement parameters, and have developed an information management system to share data.

Evaluation of Existing Effort

The existing monitoring programs have effectively addressed the management question about ensuring protection of the [water column] ecosystem. All four of the large POTW monitoring programs have demonstrated for more than 15 years that they consistently meet 1997 Ocean Plan objectives for pH, dissolved oxygen (D.O.), and transmissivity. When local alterations in these parameters have been noted, they have been attributable to natural phenomena unrelated to outfall discharge (e.g., storms, upwelling), or are identified to be within the range of natural variability (Conversi and McGowan 1994). With some facilities increasing their levels of treatment, and thereby reducing their discharge of BOD and suspended solids, there is little likelihood of future declines in D.O. or transmissivity, unless there is a large increase in the population without commensurable improvements in POTW infrastructure.

While the existing monitoring designs have effectively determined that D.O., pH, and transmissivity consistently do not exceed water quality objectives, they are not designed to address nutrient impacts as a potential stimulator of phytoplankton growth. Several studies during the 1970's suggested that upwelling was a larger source of nutrient enrichment than POTWs (Eppley 1986), but with the exception of weekly phytoplankton sampling by OCSD, little routine nutrient or phytoplankton monitoring has been conducted since that time. Concentrations of ammonia are analyzed in effluents by each of the dischargers, but the CBCP agencies have only recently begun measuring ammonia and chlorophyll fluorescence in the environment. The lack of ambient nutrient monitoring in southern California is in direct contrast to nationwide initiatives of assessing eutrophication and outbreaks of harmful algal blooms (e.g., *Pfisteria*). Harmful algal blooms, however, have not been a problem in southern California.

The existing water quality monitoring programs have effectively demonstrated that for most of the year and under typical oceanographic conditions, POTW plumes remain submerged and far from shore (Conversi and McGowan 1992). However, the historical monitoring programs have not been effective at assessing where the plume is located in the offshore environment, or under what conditions the plume is likely to move towards shore. The primary reason that managers are unable to answer questions about where the plume is located under typical oceanographic conditions is because the existing data are underanalyzed. Tremendous effort has been expended to collect spatial information over the last 20 years, but most analysis has focused on a spatial description of single events. Little data analysis has been attempted to integrate the time series data or to link correlative variables (i.e., wind, waves, tide, temperature, barometric pressure, etc.) enabling predictions of plume location.

Managers should also want to predict where there plume may go during atypical oceanographic conditions (i.e. storm events). The primary reason that managers cannot make these predictions is because these atypical oceanographic conditions have not been well sampled. Episodic events are not well-characterized by a monitoring strategy that samples at infrequent, preset intervals (i.e. monthly to quarterly).

Recommended Sampling Designs

Q1: Are water column physical and chemical parameters within ranges that ensure protection of the ecosystem?

The primary design element for this question is sampling frequency. Since Ocean Plan water quality objectives have been consistently met for over 15 years, there are two design options for future monitoring needs. The first option is a maintenance strategy whereby infrequent sampling is conducted as a check on existing conditions. Quarterly monitoring, such as the CBCP utilizes, is sufficient to meet this need. The second option is a more temporally intensive monitoring program to assess trends in water quality during all types of oceanographic conditions including episodic events. This option becomes particularly attractive if the facility wishes to make real-time predictions of plume locations (*See next water quality question*). In this case, we recommend moving to a continuous real-time water quality sensor moored near the outfall. This technology is only now becoming commercially available. If this option is selected, we strongly suggest that a special study be conducted to assess the viability of this technique on a routine basis. Several outside agencies including the US Geological Survey, local universities, and SCCWRP have indicated an interest in collaborating on these projects.

Spatial designs constitute a secondary design element for this question. Like the temporal designs above, there are two options for managers. The first option is a strategy whereby only sampling sites nearest the outfall and appropriate reference condition site(s) are sampled. Several facilities may wish to share reference condition site(s). This option will provide only the most basic of monitoring data necessary for managers to ensure compliance with the 1997 Ocean Plan, but will lack any information about other potentially confounding factors such as naturally-occurring perturbations or the influence of other anthropogenic sources. The second design option includes more spatial design elements that can capture the natural variability in water quality within the SCB and identify anthropogenic anomalies near discharges. An appropriate spatial design includes the CBCP design of cross-shore transects at predetermined depths.

Indicators. Regardless of which design option is selected there is a minimum list of indicators that need to be measured. These indicators are those specified in the 1997 California Ocean Plan for water column analysis including D.O., pH, and light transmittance. We also recommend monitoring the major nutrient discharged by large POTWs, ammonia, and the response indicator, phytoplankton (by chlorophyll fluorescence). These two indicators are currently being used by the CBCP. While concentrations of ammonia are analyzed in discharger effluents, we recommend measuring ammonia in the environment in order to identify relationships with phytoplankton abundance or composition. The utility of measuring ammonia and phytoplankton should be evaluated over time.

Visual observations, such as floating particulates or oil and grease have not been useful indicators for many years, but are listed in the 1997 Ocean Plan. Therefore, these indicators should be noted, but only on those occasions where personnel are on site such as during outfall inspections or quarterly CTD surveys.

Q2: What is the fate of the discharge plume?

Managers have two approaches for answering this question. The first approach is hindcasting. Hindcasting uses historical data to determine where the plume has been. Hindcasting is most applicable for those facilities not disinfecting the effluent and that cannot ensure, with a reasonable amount of confidence, that their plume will remain far offshore and water contact or shellfishing zones during all types of oceanographic conditions. The second approach is real-time forecasting. Real-time forecasting uses remotely sensed measurements to predict where the plume currently is or will be. Real-time forecasting is most applicable for those facilities not disinfecting the effluent and that cannot ensure their plume will remain far offshore and distant from water contact or shellfishing zones. Designing a sampling program for hindcasting or real-time forecasting is largely a function of temporal scale. We outline each of these sampling designs below.

Hindcasting. Each of the large POTWs have years of water quality data that have been collected and will provide the information necessary to conduct hindcasting. We recommend that each facility use this historical data to develop a map that delineates isoclines of plume occurrence (Figure V-2). In addition, we recommend that facilities use correlative variables (i.e. wind, waves, tide, temperature, barometric pressure, etc.) to assess the probability of plume location under differing environmental conditions.

Real-time forecasting. The design elements for real-time forecasting are more complicated. The approach uses an adaptive monitoring design that initially focuses on temporally intensive measurements near the outfall. When these measurements identify a trigger that the plume may be encroaching on shellfishing or water contact areas, then a spatial design element is incorporated to identify the locations of impact.

We recommend that the temporally intensive element should consist of continuous measures of oceanographic conditions that drive plume dispersion (i.e. winds, waves, tides, temperatures, current direction, etc.). This should be linked to the continuous measures of water quality (*See previous question*). This sampling design would consist of a moored, telemetered system. This type of system will enable managers to identify the atypical episodic events when plumes are most likely to move towards shore such as when the thermocline breaks down and currents are moving towards shore. When these conditions occur, field crews could be deployed to investigate if the plume has reached those water contact areas and, if so, where. Because of the logistics involved, this last recommendation should be evaluated first as part of a special study.

Moored systems with remote sensing telemetry are only recently becoming commercially available and have not been rigorously tested for routine monitoring applications. Therefore, we recommend a special study be conducted to assess the viability of this technique on a routine basis. Several outside agencies including the US Geological Survey, local universities, and SCCWRP have indicated an interest in collaborating on these projects.

Indicators. The moored system will require measurements of conductivity, temperature, and depth to assess the strength of the thermocline, and detect plume surfacing. The system will also require a current meter to detect plume direction.

The spatial sampling conducted in response to episodic events will be performed by field crews taking CTD casts and offshore microbiological samples; bacteria are often the most sensitive indicators of the plume. The multiple indicators measured in the shoreline microbiological monitoring (total coliform, fecal coliform, enterococcus) are not necessary for offshore monitoring, because the data are not compared to AB411 or 1997 Ocean Plan standards. Rather, offshore bacteria data are compared to background levels in order to identify the presence and concentration of the plume. Therefore, a single indicator could be used for offshore plume tracking. Total coliform is the most sensible of the three indicators since it is not found naturally in the marine environment and is the most concentrated of the three indicators presently measured in effluents. However, when the plume reaches water contact zones and public health is a concern, all three microbial indicators should be tested (See *Microbiological Monitoring*). Dischargers that disinfect their effluent do not necessarily need real-time plume forecasting.

References

Conversi, A. and J. McGowan. 1992. Variability of water column transparency, volume flow and suspended solids near San Diego sewage outfall (California): 15 years of data. *Chemistry and Ecology* 6:133-147.

Conversi, A. and J. McGowan. 1994. Natural versus human-caused variability of water clarity in the Southern California Bight. *Limnology and Oceanography* 39:632-648.

Eppley, R. 1986. Plankton dynamics of the Southern California Bight. Springer-Verlag Press, New York. 373 pp.

TABLE V-1 Water quality sampling effort among facilities before and after the start of the Central Bight Cooperative Program in July 1998.

	HTP	JWPCP	CSDOC	PLWTP
--------- Before July 1998 ---------				
# Samples/year*	1,776	856	1,764	2,016
# Sites	32	28	17	46
Frequency	monthly	monthly	monthly	Monthly
--------- After July 1998 ---------				
# Samples*/year	216	192	308	552
# Sites	54	48	43	46
Frequency	quarterly	quarterly	quarterly	monthly

* = equivalent to # CTD casts

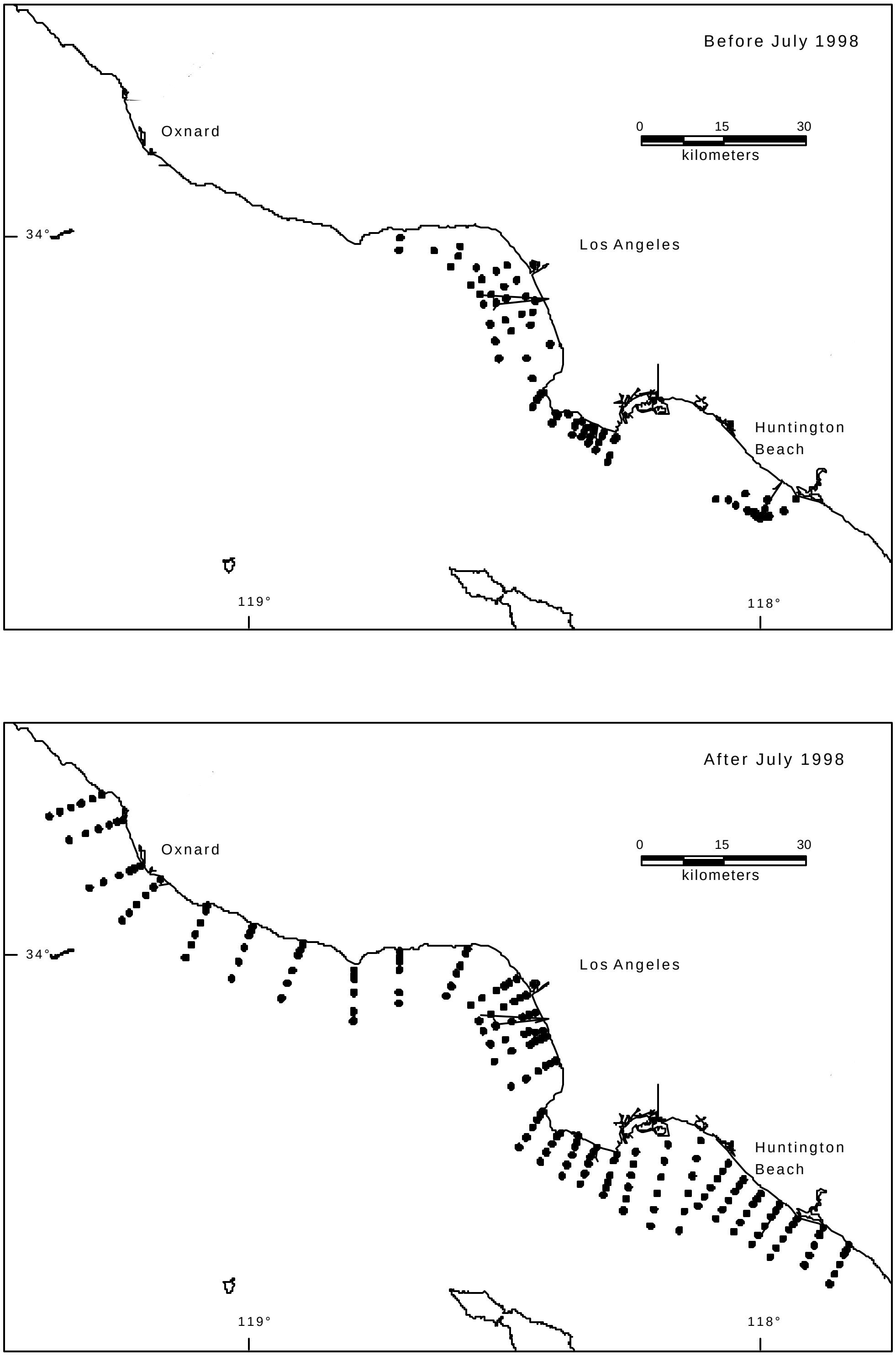

FIGURE V-1. Location of water quality stations before 1998, and after 1998 as part of the Central Bight Cooperative Program.

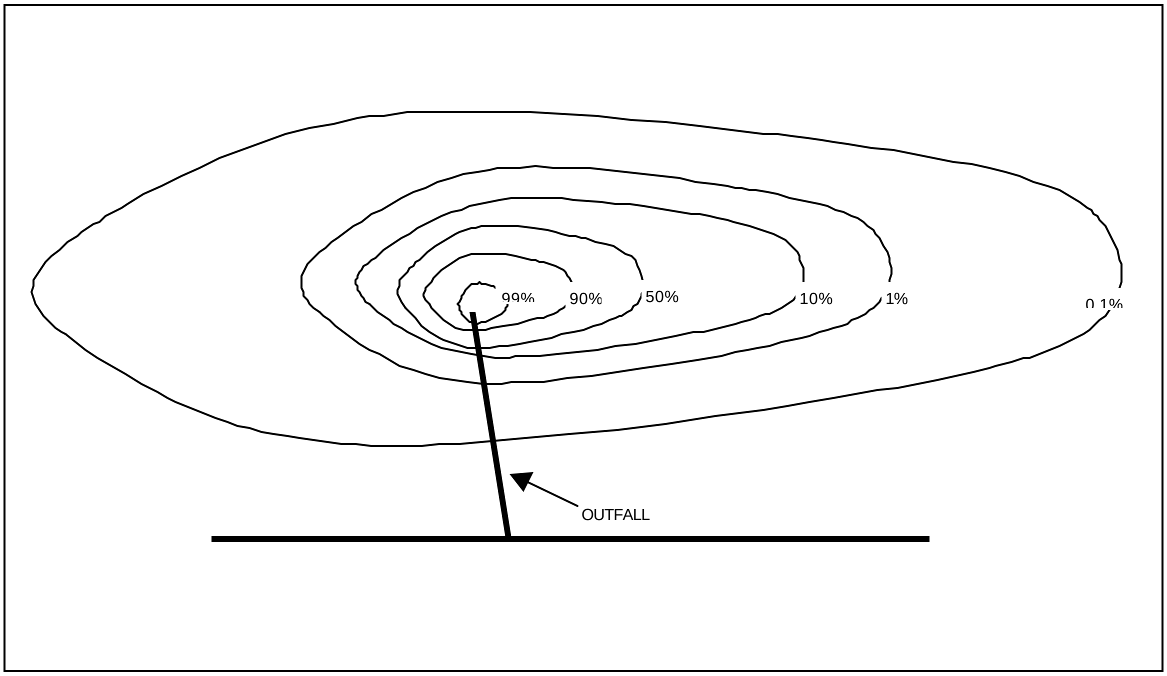

FIGURE V-2. Hypothetical isocline map of plume occurrence. Each isocline represents the proportion of time that the plume may occur at that location. Separate maps could be constructed for varying oceanographic conditions.

VI. SEDIMENT MONITORING

The most important management questions for sediment monitoring are:

> ***Q1:*** ***Are sediments in the vicinity of the discharge impaired? If so, what is the spatial extent of impairment?***
>
> ***Q2:*** ***Are sediment conditions changing over time?***

Sediments integrate constituents that are discharged to the ocean. The particles that come from POTW discharges, and any associated contaminants, will eventually settle to the seafloor where they are incorporated into the existing sediments. Sediments accumulate these particles over the years and may reach the point where sediment quality has degraded and beneficial uses are impaired. The beneficial uses most often associated with sediment quality are aquatic life and public safety (seafood bioaccumulation). Public safety is addressed in the chapter on fish monitoring. Impairment of sediment quality that can affect aquatic life is monitored by assessing habitat quality such as grain size and organic carbon content, sediment contamination such as anthropogenic constituents, biological communities such as healthy benthic communities, and interactions among all three components such as sediment toxicity.

Environmental managers can use sediment monitoring as a means to evaluate if effluent concentrations or mass emissions are accumulating in receiving water environments, particularly if their effluent is exceeding water quality thresholds. An assessment of magnitude and/or spatial extent of impairment enable resource managers to rank sites and evaluate which locations are most critical for immediate action. Finally, sediment monitoring can be used for beneficial use assessments in other program elements, particularly assessments of impairment to fish.

Sediment monitoring has both a local and a regional component. Environmental managers will want to look at local sediments to assess the effect of their facility-specific discharge. However, managers also need a regional assessment of sediment conditions. They need a regional assessment because the public wants to know the overall health of the SCB ecosystem (NRC 1990). They also need regional reference conditions to assess the spatial extent and magnitude of sediment impairment found locally. Finally, they need a regional assessment so they can place context on the impairment they do find (e.g., is my site the worst site?).

Environmental managers can utilize trends in sediment condition to make decisions regarding the need for additional actions. If the trend in sediment condition is improving, then the manager can utilize this information to demonstrate that the actions already undertaken have been effective at reducing risks to beneficial uses. If the trend in sediment condition is getting worse, then the need to take action may increase. If there is no trend, particularly if the local environment is not threatened, then little or no action may be required.

Compare and Contrast Among Agencies

All four large POTWs have similar sediment monitoring designs, but they implement their designs very differently. The basic design consists of a grid with transects located along isobath contours. However, considerable differences were found among the facilities in the level of effort expended on sediment monitoring. For example, the number of sediment chemistry analyses conducted in a one-year period differ 7-fold, although the number of benthic infaunal analyses conducted during the same period differ only 2-fold.

The biggest differences are in sampling frequency (Tables VI-1 and VI-2). The OCSD samples approximately 80% of its sediment chemistry and benthic infaunal stations on an annual basis, and approximately 20% on a quarterly basis. The JWPCP samples approximately 30% of its benthic infaunal stations semi-annually, the remaining 70% of its benthic infaunal stations annually, and all of its sediment chemistry stations biennially. The HTP samples all of its benthic infauna and sediment chemistry stations annually, while PLWTP samples all of its stations for these parameters quarterly.

The degree of station replication for sediment chemistry among agencies varies from 1 to 3 samples (Table VI-2). This same variability also exists for within-agency replicate differences at two facilities. The JWPCP and OCSD analyze three replicates for sediment chemistry at selected sites along the 60-m contour and one sample at all other sites. In contrast, HTP and PLWTP collect one sample at all sediment chemistry sites.

The number of replicates collected for benthic infauna varies from 1 to 5 samples among agencies (Table VI-1). The JWPCP and HTP collect one sample at each site during the winter and five replicate samples along their 60-m contours during the summer. The OCSD collects three replicates for benthic infauna at their 60-m stations on a quarterly basis and one replicate at all other sites on an annual basis. The PLWTP collects 2 replicates at all of its benthic infauna sites.

The number of sediment chemistry and benthic infaunal sampling stations is relatively similar among the agencies (within a factor of two). However, the amount of area over which these stations are distributed varies by a factor of eight, and is unrelated to annual flow or mass loading. Instead, the number and distribution of stations usually appear to be related to the characteristics of the receiving environment. For example, HTP discharges 15,000 metric tons (MT) of suspended solids annually and maintains 40 sediment chemistry stations distributed over approximately 316 km^2, while JWPCP discharges 30,000 MT of suspended solids annually and maintains 44 sediment chemistry stations over approximately 80 km^2. However, the discharge site characteristics for these two agencies are different; HTP discharges into a bay, whereas JWPCP discharges onto a narrow shelf with generally faster ocean currents.

The sediment chemistry constituents analyzed among the agencies differ considerably for organic compounds (Table VI-3). Each discharger analyzes sediment for PCBs and PAHs, but the types of each compound analyzed are different. For example, JWPCP analyzes sediment for 3 PCB Aroclors and 13 PAH derivatives, whereas OCSD analyzes for 44 PCB congeners and 45 PAH derivatives. Entire analyte classes are analyzed by only a subset of the agencies. Volatile organic compounds (VOCs) in sediment, for example, are only analyzed by JWPCP. The OCSD dropped VOC analyses in their 1998 permit because these compounds were consistently not detectable in sediments. The types of sediment metal constituents analyzed are more consistent among facilities. Of the 15 metals, 9 are analyzed by all four agencies, and 11 are analyzed by three or more agencies (Table VI-4).

Evaluation of Existing Effort

Sediment monitoring has been a part of each agency's monitoring program since its inception and has proven to be highly effective. Each of the agencies has been able to demonstrate temporal declines in magnitude of discharge effects; most have also been able to demonstrate declines in the spatial extent of discharge effects. Combined with demonstrated declines in mass emissions, the sediment monitoring data have demonstrated the effectiveness of effluent control programs through improvements in the benthic communities and decreases in sediment chemical concentrations. Sediment data have also provided an important foundation for 301(h) waiver decisions.

While sediment sampling programs have been effective for addressing several management questions, they are inefficient for addressing the two questions that managers have indicated during interviews should be addressed: (1) *what is the spatial extent of sediment impairment?* and (2) *is the sediment condition (i.e., contaminant concentration and bioeffects) changing over time?* Present sampling designs fail to distinguish these objectives, which have different design needs, resulting in inefficient allocation of effort.

Describing a spatial pattern requires gathering data from as many sites as possible. To describe a spatial pattern efficiently, the number of replicates collected at a site and the number of repeated visits to the site (e.g., quarterly or annual sampling) should be minimized in favor of sampling more sites. In contrast, trend assessments are more efficiently accomplished through numerous repeated visits to a site.

At present, most programs commingle these two questions in a common sampling design. Large grids of sampling sites are visited repeatedly over time, often many times per year, and often with replicates. Revisiting each site during every survey favors the trend question, but doing so at all sites appears to provide more trend information than is required to address present management questions. As discharge rates have declined and the affected area around discharge pipes has decreased, the need for trend monitoring at all of the historically monitored sites has declined.

The practice of measuring replicates at every site appears to be an artifact of the historical approach of using an ANOVA model for spatial assessment. In an ANOVA design, the condition at each site is evaluated relative to a reference site(s) and replication is necessary to determine whether sites differ statistically. More recently, though, regional reference conditions and indices that quantify condition of an individual sample relative to regional reference condition (e.g., the Benthic Response Index for benthic infauna, iron normalization curves for metals) have been developed through a cooperative regional monitoring program. This has reduced the need for replication to characterize the condition of individual sites, allowing more efficient allocation of effort toward description of spatial patterns at sites where replication is not needed for trend analysis. Some programs have already started to adopt such a strategy by identifying their most important trend sites and sampling these at higher frequencies, while surveying the entire grid on a less frequent basis. In some cases, the sites are located along the 60 m isobath and are attempting to identify trends in linear gradients.

The design issues above presuppose that the boundaries for the maps of exposure and effects are well known, which is not the case. The area over which monitoring is conducted varies considerably among dischargers, without apparent rationale for these differences. In most cases, the area sampled is the same as when the programs were initiated 30 years ago. Sampling boundaries would be more appropriately established by comparing sediment conditions to chemical and biological thresholds to determine which areas are impaired.

Recommended Sample Designs

There are three independent, but integrated, sampling designs in our recommended sediment monitoring program. Two of the designs focus on the management question "what is the spatial extent of impairment?", but concentrate on different spatial scales. The first sampling design is a local program that focuses on mapping gradients near an outfall, while the second is a regional program that assesses cumulative impacts beyond the boundaries of the local outfall. The third sampling design focuses on the trends question "are sediment conditions changing over time?", and assesses local changes in conditions near the outfall. Each of these three designs are discussed in separate sections below.

Q1: Are sediments in the vicinity of the discharge impaired? If so, what is the spatial extent of impairment?

Answering this management question is a two-step process. Resource managers will first want to establish that there is an impact near their discharge before extending their monitoring to greater distances. Alternatively, if there is no impact near the discharge, then additional sampling is unwarranted. This example of adaptive monitoring, whereby resource managers can use the monitoring to establish further need, is an efficient mechanism for minimizing costs and increasing effectiveness of a program. While large POTWs have already identified an impact near their discharge, other types of discharges, such as small POTWs, may not.

Local Spatial Design

The goal of a local sediment monitoring program is to produce maps of sediment conditions around their discharge. Maps are one of the most effective means for communicating spatial extent. They have the capability to add context to interpreting results that long tables of data cannot convey. Maps are easily understood by non-technical audiences and can be especially useful for transmitting magnitude and spatial extent information by the addition of contours. Contours of increasing sediment concentration, contours of numbers(s) of indicators that exceed thresholds, and contours of previous year(s) extent are all insightful tools to relay detailed information in a meaningful format that will provide the appropriate context to decision-makers.

Producing quality maps with known estimates of confidence relies on spatial statistics, which uses a high sampling density and low sample replication. Although multiple techniques are available for creating maps, kriging or co-kriging are the preferred options (Leecaster 2001). The major design elements for kriging techniques focus on distribution and density of sampling sites, size of area to be mapped, replication and frequency, and indicators.

Distribution and density of sampling sites (Local Spatial Design). There are several options for how and where to place sampling sites for constructing a map. We recommend using a fixed grid design, not unlike what is already in use for most monitoring programs. The main differences is the distance between sites may decrease depending on the optimal spacing among sites. An accurate depiction of spatial patterns requires samples that are close enough together to allow meaningful interpolation, but not so close together as to yield duplicative information. To determine optimal sampling density, we recommend using spatial statistical models, such as the semi-variogram, with historical data in order to summarize the relationship between indicator variability and distance between stations (Cressie, 1993). Optimal density is identified by the inflection point of the semi-variogram where correlation between indicator and sampling distance begins decrease. At this point in the variogram, higher sampling densities do not result in proportionally greater confidence of spatial estimates (Figure VI-1).

Preliminary analysis of grain size data in Santa Monica Bay suggests that spatial covariance is optimal at a sampling distance of 1 km for Hyperion (Leecaster 2001). However, the relationship in spatial variability is not known for other agencies and data from their current sampling designs are insufficient to model the spatial relationship. Therefore, we recommend that each agency should initiate a special study to obtain the variability-distance relationships necessary to determine the optimal distance between sites for all indicators that are to be mapped.

Size of area (Local Spatial Design). The size of the area monitored should be optimized to reflect the area impacted by the local discharge. We recommend using both chemical and biological thresholds to evaluate sediment conditions in order to determine the boundaries of local mapping. For sediment chemistry, numerical sediment quality guidelines (e.g., ERL, ERM) exist to compare local concentrations to national thresholds of potential biological significance (Long et al.. 1995). To assess biological effects, local indices (e.g., BRI) have been developed (Bergen et al.. 2000).

Using the concepts of numerical thresholds and indices, we have identified three categories of sediment impairment for use in the local monitoring program (Table VI-5). They are “affected”, “altered”, and “degraded” in increasing order of impairment. We recommend mapping all areas of local sediment alteration. For sediment chemistry, alteration is characterized as an exceedance of an ERL, while biological alteration is characterized by a slightly reduced biodiversity (i.e., BRI 34-44). An exceedance of either the biological or chemical threshold would cause an area to be categorized as altered. The USEPA has indicated there may be other tools or thresholds to help evaluate sediments, in addition to the ERL and BRI thresholds suggested.

We recognize that certain factors may make it difficult to define the area of alteration. For example, the area monitored may be limited by the local bathymetry (e.g., canyons, slopes), or the impairment resulting from wastewater discharge may be obscured by other sources of contaminants (e.g., urban runoff). Therefore, we recommend that the edge of alteration be used as a guideline for assessing mapping boundaries. Final decisions regarding the mapping boundaries ultimately become a site-specific issue that needs to be agreed on by the local managers.

Replication and Frequency (Local Spatial Design). Since the mapping design is entirely focused on characterizing more area rather than focusing on characterizing any single location, we recommend that only one replicate (n=1) be collected at each site. Agencies that have a need or desire to characterize differences between sites or group of sites, or perform quality assurance procedures may want to collect replicate samples, at the option of the local manager.

We recommend creating a map at a frequency of every five years. This recommendation is based on the fact that sediment conditions do not change rapidly. If more frequent assessments of trends are a desired product, then see the sampling design recommended for Question Number 2. Instead, creating a map every five years is an efficient opportunity to integrate local programs with regional monitoring surveys that occur on five year intervals. Moreover, the mapping data can then be used for decision-making on renewal of facility NPDES permits that also occur on five-year cycles.

Indicators (Local Spatial Design). We recommend three categories of indicators that should be measured in common by all dischargers in a local sediment monitoring program; chemistry, habitat variables, and benthic infauna. Sediment toxicity should also be included when it can be demonstrated that these tests have adequate precision, and that the measurements can be used in management decision-making.

Sediment Chemistry (Local Spatial Design). We recommend sampling constituents that have the capability to accumulate in the environment including: 1) chemicals that bioaccumulate; 2) chemicals that exceed the ERL in more than 15% of the SCB; and 3) constituents on local 303(d) lists (Table VI-6).

For those constituents that are measured in sediments we recommend that reporting levels should quantify constituents for threshold evaluation. To accomplish this, reporting levels for metals should be $^{1}/_{5}$ the ERL and that organics should be measured to $^{1}/_{2}$ the ERL. This requirement is consistent with existing regional monitoring program requirements and are readily attainable by most qualified laboratories. If a constituent does not have an ERL threshold, then we recommend using the regional monitoring reporting level as a default value.

Habitat Variables (Local Spatial Design). Noncontaminant factors such as grain size and organic matter, while not toxic to benthos, affect biological responses. Measurements of habitat variables are used in interpretation of benthic community impairments. Therefore, we recommend that total organic carbon (TOC) and sediment grain size be measured, at a minimum. Other habitat variables (i.e. sulfide, redox, total volatile sulfides, AVS/SEM, biological oxygen demand, chemical oxygen demand) can be measured at the option of site-specific management needs.

Benthic Infauna (Local Spatial Design). We recommend that benthic organisms need to be identified to the lowest possible taxa (e.g. species level) in order to be able to derive community parameters and develop BRI scores. Estimation of phylum-level biomass upon wet weight measurements has been a common element of POTW infaunal monitoring programs. However, biomass measurements, particularly taken as wet weight, are less useful for decision-making, and should not be required in local monitoring.

Sediment Toxicity (Local Spatial Design). Currently, no programs in the SCB measure sediment toxicity as part of their routine monitoring programs although OCSD has begun to incorporate sediment toxicity tests as part of their strategic process studies. Sediment toxicity would be a useful addition because of its value in interpretation of sediment quality. Sediment quality criteria for assessing chemistry and biocriteria for assessing benthic infauna data are not yet available. To make these comparisons, each of the agencies currently relies upon sediment quality guidelines and locally derived indices to interpret chemical and biological data. However, we recognize the limitations these guidelines have, especially when the chemistry and biology data disagree. In these instances, a weight of evidence approach may help in assessing sediment conditions. The weight of evidence approach is enhanced by more evidence, which we believe sediment toxicity measurements may be able to provide.

Sediment toxicity measurement also provides assurance that unmeasured chemicals are not causing a problem, reducing the need to measure a larger array of contaminants in the sediment. Much as water column toxicity measures are used to screen for unmeasured chemicals in effluent, sediment toxicity screens for unmeasured chemicals accumulated in sediment. Sediment toxicity will become even more valuable when sediment toxicity identification evaluations (TIEs) are further developed because TIEs provide a mechanism for identifying the causative toxic agents, if toxicity is encountered. Sediments near wastewater discharges contain a variety of chemical constituents, many of which exceed sediment quality guidelines. The advantage of the sediment TIE is that it narrows the list of chemicals to only those which are responsible for toxicity, enabling resource managers to focus their actions on effective remedies.

Sediment toxicity testing is a standard method used by several national and state programs, providing context for interpreting local trends. Amphipod toxicity tests are used routinely in the California Toxic Hot Spot and Clean-up Program, EPA's Environmental Monitoring and Assessment Program, NOAA's National Status and Trends Program, and in the SCB regional monitoring studies. The relative toxicity measured locally can then be compared to measurements made nationally.

We recommend that local programs incorporate sediment toxicity testing when a greater history of their use is acquired in SCB regional monitoring programs. The regional monitoring steering committee has employed sediment toxicity testing in both the 1994 and 1998 regional monitoring programs. There is a need, however, to demonstrate that toxicity testing has sufficient precision and repeatability to ensure that the measurements are meaningful to local regulators and dischargers for decision-making.

Regional Design

Unlike local spatial monitoring designs, regional monitoring designs use inferential statistics to produce estimates of the area impaired. Instead of maps, bar charts or pie charts are produced that identify km^2 or % area impaired. The major sampling design elements for the inferential approach include the distribution and density of sampling sites, size of area to be mapped, replication and frequency, and indicators.

Distribution and density of sampling sites (Regional Design). Inferential statistics use a stratified-random sampling design for distribution of sampling sites. We recommend readers consult Stevens (1997) for a thorough description of this approach. Ultimately, the regional monitoring steering committee will define the final number of sampling sites and their location.

We recommend using power analysis to determine the optimum number of sites within each strata (Figure VI-2). The optimum number corresponds to the inflection point between number of samples and amount of confidence in areal estimates. Approximately

30 sites per strata provides the most efficient allocation of sites; a greater number of sampling sites would not provide a proportional increase in confidence.

Replication and frequency (Regional Design). Since the regional monitoring is entirely focused on characterizing more area rather than focusing on characterizing any single location, we recommend that only one replicate (n=1) be collected at each site.

We recommend conducting regional monitoring at a frequency of every five years. This recommendation is based on the fact that sediment conditions do not change rapidly. If a more frequent assessment of trends is a desired product, then see the sampling design recommended for Question Number 2. Instead, assessing percent area impaired every five years is an efficient opportunity to integrate with regional monitoring surveys with local programs that create maps on five year intervals. Moreover, the percent area estimates can then be used for decision-making on renewal of facility NPDES permits that also occur on five-year cycles.

Indicators (Regional Design). We recommend four categories of indicators for use in the regional monitoring design. These include chemistry, benthic infauna, habitat variables, and sediment toxicity. There is almost complete overlap of indicators and methods between the local spatial monitoring and regional spatial monitoring designs. This enables integration among the two monitoring programs. Minor differences do occur for chemistry and toxicity. For chemistry, the regional monitoring steering committee has consistently selected constituents with thresholds for comparison, even if they are not abundant in SCB sediments. The regional monitoring steering committee has consistently required sediment toxicity testing in regional monitoring programs, which was an optional component in our recommended local monitoring design.

Q2: Are sediment conditions changing over time?

Local Trends Design

There are several design elements for a local trends monitoring program that include locations of sites, replication, frequency, and indicators.

Location of sites (Local Trends Design). Unlike the spatial design location elements that favored lots of sampling sites to get estimates of areal extent, trend monitoring should focus on specific locations to assess sediment conditions at sites of particular management importance. As such, this will be a site specific issue with three options. The three options are another example of adaptive monitoring because each option represents an selection of effort commensurate with amount of environmental impact.

There are some POTWs whose discharge do not appear to be impacting the local sediments. Some small POTWs fit into this category. For these facilities, the design could be as simple as a single site near the zone of initial dilution (ZID) and a reference site. In fact, reference sites could be shared among facilities. In this case, monitoring is used to ensure that conditions are not degrading over time.

The second design option is for POTWs that are having an effect on sediment conditions, or discharge to more than one depth zone or habitat type. For these discharges, multiple sites should be required that encompass the range of impairment, depths, or habitat types, and should include appropriate reference sites in similar depths and habitats.

The third design option is for dischargers that have large impacts or have a demonstrated need (i.e. EPA 301(h) waiver facilities) to assess trends in gradients. The recommended design for these facilities is based on a repeated measures ANOVA analysis of sites along an isobath at varying distances from the outfall (SAIC and MEC 1997). In order to characterize possible outfall effects, the sampling locations should include ZID, near-field, and far field sites.

Regardless of which of the three designs is selected, we recommend that dischargers select the most appropriate sampling sites based on information from either their historic trend monitoring or the local spatial design. Selection of historical sites has some merit since they will continue any record that may already exist. However, additional sites may be selected based on the results of the local spatial monitoring since it will identify the most impacted locations.

Replication (Local Trends Design). To investigate the issue of replication, historical data from OCSD was examined to determine the amount of variability that is associated with sample replication. It was reasoned that replication should be proportional to the amount of variability detected, with a greater number of replicates required when variability is high. A sum of the squares linear model was used to examine variability due to four factors: replication, station location, year, and random variability. Sediment chemistry and benthic infauna data from 13 stations between 1985-1997 were used in the calculations. During this time, three replicates (a total of three independent samples) were taken for sediment chemistry at each site, and five replicates (five independent samples) were collected for benthic infauna. The model estimated that replication accounts for $\leq$2% of the variance for most metals and $\leq$1% of the variance for organic constituents, benthic community parameters or abundance of selected indicator taxa (Tables VI-7 and VI-8). Most of the observed variance was due to random variability and differences between stations indicating that replication is not necessary. Therefore, we recommend a single sample be collected during each visit. Some managers may be uncomfortable with only a single sample because of a potential data outlier or in case of accidental lose, so a second sample can be collected at the option of the local manager.

Frequency (Local Trends Design). Frequency (sample size) is a function of the amount of trend desired, the length of time necessary to detect the trend, and the desired level of confidence in detecting the trend. When asked what amount of trend would be required for decision-making, or the desired confidence in detecting that trend, SCB managers were unable to provide a unified answer. Therefore, we recommend a default frequency of once per year. Greater sampling frequencies could be used, but seasonal variability will compound one's ability to detect biological trends. OCSD has used statistical techniques to detrend seasonal data, but this level of effort seems unwarranted unless specific trend goals are needed as they are at OCSD. Provided that specific trend goals are defined, power curves should be used to define the optimum frequency required to detect that trend.

If sampling is to occur annually, we recommend that the sampling event take place during the summer season. We recommend the summer season for two reasons. First, the summer season is the most stable and consistent time period of the year thus minimizing variance and enhancing ones ability to detect trends. Second, summer is when regional monitoring surveys occur thus providing another opportunity for integration among local and regional programs.

A second type of trend that some dischargers may want to track is the extent of affected area, in order to assess if their area of impact is increasing or decreasing. In this case, dischargers can use data from the local spatial design (see above). Trends monitoring is not the main focus of the local spatial design, but the data are appropriate for this type of trend-in-space analysis. Local spatial monitoring design recommendations include sampling every five years, which limits managers to longer time frames for observing trends in spatial extent.

Indicators (Local Trends Design). The list of indicators will be the same as for the local spatial design. This includes measurements of sediment chemistry, habitat variables and benthic infauna (*see Local Spatial Design Indicators above*). The same indicators will be measured in the two designs because constituents of concern near an outfall need to be monitored on both temporal and spatial scales.

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TABLE VI-1. Current infaunal assemblage sampling effort.

Agency	# Stations	# Replicates	Frequency
HTP	33	1	semiannually
	7	1 winter, 5 summer	semiannually
JWPCP	15	1	semiannually
	3	1 winter, 5 summer	semiannually
	26	1	annually
OCSD	10	3	quarterly
	39	1	annually
PLWTP	21	2	quarterly

TABLE VI-2. Current sediment chemistry sampling effort.

Agency	# Stations	# Replicates	Frequency
HTP	40	1	annually
JWPCP	41	1	annually *
	3	5	annually *
	18 (subset of above)	1	semiannually *
	21	1	biennially
	3	3	biennially
OCSD	10	3	quarterly
	39	1	annually
PLWTP	23	1	quarterly

* general constituents only

TABLE VI-3. Number of sediment constituent analyses per year.

Constituent	HTP	JWPCP	OCSD	PLWTP
General (TOC, AVS)	120	248	297	368
Metals	360	135*	1287	1380
Organics				
DDTs	240	90*	594	552
PCBs	280	45*	4356	644
PAHs	520	195*	4455	2208
phenolics	40	15*	-	-
halogenates	40	15*	-	-
others	280	105*	990	1012

* = half of biennial value

TABLE VI-4. Sediment constituent reporting limits. Also included are sediment quality guidelines (Long et al.**, 1995). NA = not available. Dash = not analyzed.**

Constituent	HTP	JWPCP	OCSD	PLWTP	Effects Range Low	Effects Range Median
Metals/Metalloids (mg/dry Kg)						
Aluminum	-	-	-	5	-	
Antimony	-	0.18-0.35	-	5	-	
Arsenic	0.2	2	0.01	0.08	8.2	70
Beryllium	-	0.1	0.05	0.2	-	-
Cadmium	0.1	0.7-1.0	0.01	0.5	1.2	9.6
Chromium	2	10	0.5	3	81	370
Copper	4	2	0.5	2	34	270
Iron	-	-	0.6	3	-	-
Lead	0.3	2	0.1	5	46.7	218
Mercury	0.03	0.05	0.02	0.047	0.15	0.71
Nickel	1.2	6	0.5	3	20.9	51.6
Selenium	-	0.73-1.2	0.1	0.11	-	-
Silver	0.03	0.2	0.01	3	1.0	3.7
Thallium	-	0.44-0.85	NA	10	-	-
Zinc	4	11	0.5	4	150	410
Organics (µg/dry Kg)						
DDT	0.5-2.0	1-5	0.1-0.4	0.26-0.94	1.58	46.1
PCB	10-20	10-50	2	NA	22.7	180

TABLE VI-5. Sediment mapping thresholds.

Degree of Impairment	Mapping thresholds	
	Chemical	Biological
Degraded	Exceedance of concentration threshold (e.g., ERM)	Loss of functional groups (e.g., BRI >34)
Altered	Exceedance of concentration threshold (e.g., ERL)	Slightly reduced biodiversity (e.g., BRI 25-34)
Affected	Enriched (e.g., metal:iron replacement)	Same organisms as reference, but different proportions (e.g., BRI <25)

TABLE VI-6. Chemical constituents to be analyzed in sediments for the local spatial and local trend programs, listed under the category they were selected. The partial list of 1997 California Ocean Plan effluent constituents is provided for comparison.

Constituent	Partial list of 1997 California Ocean Plan effluent constituents	Bioaccumulative	Exceeds ERL in more than 15% of SCB	Appears on local 303(d) list
Antimony	X			
Arsenic	X	X		
Cadmium	X			
Chromium	X			
Copper	X			
Lead	X			X
Mercury	X	X		X
Nickel	X			
Selenium	X	X		
Silver	X			X
Zinc	X			X
Thallium	X			
Cyanide	X			
Total chlorine residual	X			
Ammonia	X			
Phenolic compounds	X			
Organotins	X			
DDTs	X	X	X	X
Other chlorinated pesticides (e.g., chlordane)	X	X		X
PCBs	X	X	X	X
PAHs	X			X
Dioxins	X			
Purgeable aromatics	X			
Other constituents of local concern	?			

ERL = Effects Range Low (Long et al.. 1995).
SCB = Southern California Bight.
303(d) list = Inventory of impaired waterbodies that California reports to USEPA.
Other constituents of local concern represents contaminants of interest that are found in a discharger's effluent or in sediments near the discharger's outfall, but do not necessarily exceed the ERL in >15% of the SCB.

TABLE VI-7. Percentages of variance in chemical measures accounted for by replicates, stations, and years. Data are for samples collected during the summer between 1985-1997 at 13 Orange County Sanitation District monitoring stations. Three samples were collected for each sampling event, with one event per year.

Constituent	Replicate	Station	Year	Random Variation
General				
Chemical Oxygen Demand (mg/kg)	0.01	9.66	80.21	10.11
Cyanide (mg/kg)	0.26	5.34	52.72	41.68
Sulfides (mg/kg)	0.24	22.08	19.16	58.51
TOC (%)	0.06	87.86	1.73	10.34
Metals/Metalloids (mg/kg)				
Antimony	6.14	10.23	6.52	77.11
Arsenic	0.04	17.90	67.74	14.32
Beryllium	0.02	7.17	81.50	11.30
Cadmium	0.04	55.68	6.64	37.64
Chromium	0.03	37.84	46.13	16.01
Copper	0.02	67.79	6.23	25.95
Lead	0.02	60.30	20.75	18.94
Mercury	0.15	9.23	6.93	83.68
Nickel	0.01	63.71	23.83	12.46
Selenium	2.39	20.24	19.44	57.94
Silver	0.08	56.78	19.09	24.05
Thallium	0.07	1.13	71.29	27.51
Zinc	0.01	72.01	12.45	15.54
Organics (µg/kg)				
DDTs	0.17-0.92	4.38-26.69	12.72-17.83	55.32-82.77
PCBs	0.00-0.21	1.81-6.32	35.69-86.18	11.99-59.73
PAHs	0.00-0.84	2.27-10.97	1.88-62.80	27.14-94.74
Other chlorinated pesticides	0.01-0.85	1.23-5.37	11.25-89.02	9.74-82.77

TABLE VI-8. Percentages of variance in biological measures accounted for by replicates, stations, and years. Data are for samples collected during the summer between 1985-1997 at 13 Orange County Sanitation District monitoring stations. Five samples were collected for each sampling event, with one event per year.

Measurement	Replicate	Station	Year	Random Variation
Community				
Total abundance (per sample)	0.08	44.63	6.60	48.69
Wet wt. biomass (g)	0.35	4.37	2.30	92.97
Benthic Response Index	0.03	85.66	5.18	9.13
Evenness	0.14	28.14	15.66	56.06
Number of species (per sample)	0.07	54.29	5.85	39.80
Shannon-Weiner Index (base 2)	0.14	45.24	11.28	43.34
Species				
Amphiodia spp.	0.11	75.18	1.00	23.71
Capitella capitata complex	0.08	22.35	7.31	70.27
Euphilomedes carcharodonta	0.01	52.28	13.11	34.59
Euphilomedes producta	0.28	26.27	19.89	53.57
Parvilucina tenuisculpta	0.03	20.70	38.10	41.18

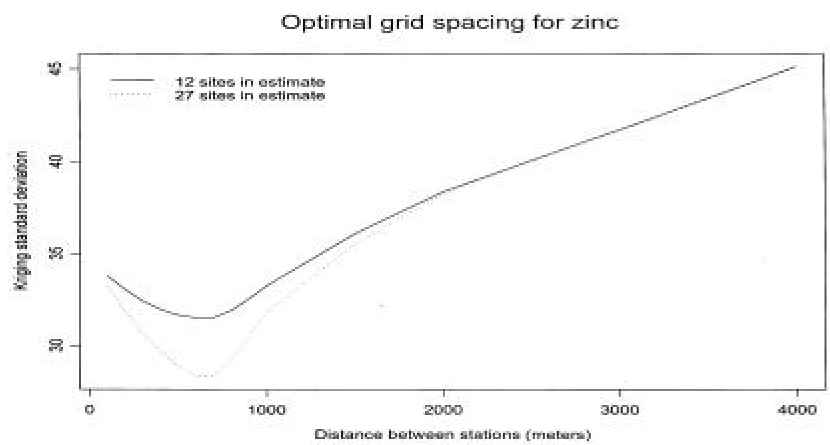

Figure VI-1. Variability in zinc measurements in relation to sampling density.

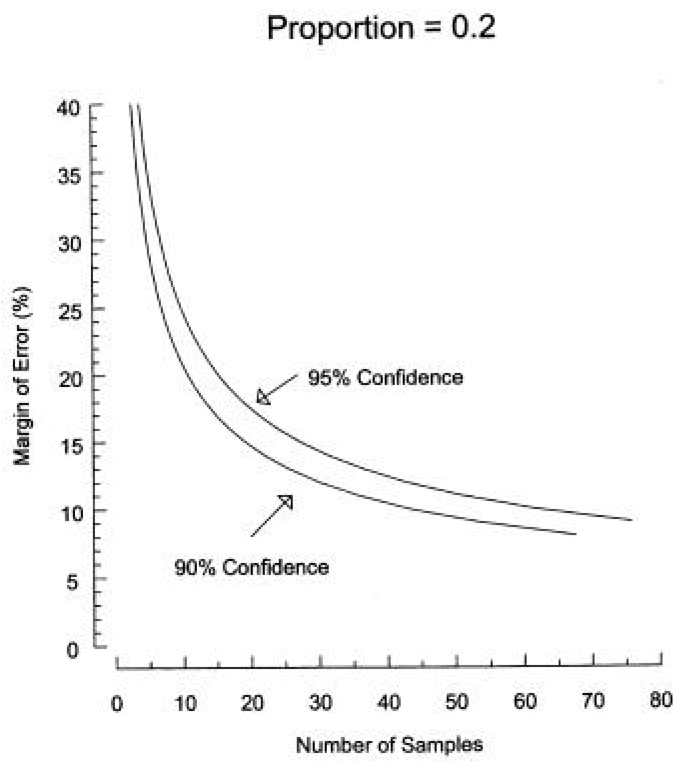

FIGURE VI-2. 90% confidence intervals about an estimate of percent of area changed as a function of sample size.

VII. FISH AND EPIBENTHIC INVERTEBRATE MONITORING

Three management questions address fish and epibenthic invertebrate (termed “fish”) related beneficial uses in the SCB:

Q1: Is the health of fish populations and communities impaired?
Q2: Are fish populations and communities changing over time?
Q3: Is fish tissue contamination changing over time?

In contrast to the benthic infauna, which serves scientists well as an indicator of environmental stress, the public has a much clearer image of fish and a better understanding of their importance (both as a part of an ecological community and as a source of food). Questions regarding the health of fish communities and populations are important because of this public interest. Moreover, there is a direct relationship between fish abundance and recreational value including such angling activities as catch per unit effort or species-specific catch. This question is typically asked by the public in a general, or regional, sense rather than at a specific location.

Managers investigate the effects on fish by assessing community assemblages and contaminant concentrations in fish populations. Community parameters and tissue concentrations are typically compared between outfall site(s) and reference site(s); many times they are also compared to historical levels within a site. Research has shown that alterations in fish community assemblages near POTW outfalls is a result of the composition of benthic invertebrate communities that serve as food to fish. For example, as crustacean species are replaced by polychaete species in response to POTW impacts to infauna, the fish communities shift towards more polychaete-feeding species such as English sole (*Parophrys vetulas*) and Dover sole (*Microstomus pacificus*) (Allen 1977, Cross et al.. 1985, Allen et al.. 2001). In addition, dramatic changes in fish abundance and community assemblages are often the result of large, regional-scale shifts in the natural environment, such as alterations in ocean temperature during El Niño.

There are three main reasons managers choose to assess contaminant concentrations in fish tissues. The first is seafood safety and the protection of human health (See Chapter on *Seafood safety monitoring*). The second reason managers choose to assess tissue concentrations is the risk to environmental health, such as bioaccumulation in higher order predators that consume fish. This question has been addressed, in large part, due to the prevalence of total DDT in the SCB (Young and Mearns 1979, Schiff and Allen 1997). The third reason managers choose to examine tissue concentrations is to assess the health of the fish itself, since it may contract illness or become more susceptible to predation as a result of increased tissue burden. Unlike quantitative risk assessments for human health concerns, there are no quantitative risk assessments for fish.

Compare and Contrast Among Agencies

The total effort for fish assemblage monitoring is somewhat comparable among agencies. The number of trawls per year for fish assemblage monitoring varies from 38 to 56 (Table VII-1). The number of fish assemblage trawl stations monitored per year ranges from 9 to 12.

Although the total effort is relatively similar among facilities, inconsistencies were found in trawl replication, frequency and spatial extent of assemblage monitoring. The number of replicate trawls varies from 1 to 3 among agencies. The JWPCP and PLWTP maintain single trawls at all sites, while both HTP and OCSD conduct multiple trawls at their 60 m contour sites. The HTP conducts 2 trawls at these stations and 1 trawl at all others sites, while OCSD conducts 3 trawls at most 60 m stations and 2 trawls at all other sites.

The area monitored by trawls differs by a factor of six among facilities (Table VII-1). As with sediment chemistry sampling, the difference in area sampled appears to be more affected by the characteristics of the discharge area than by the annual volume of flow or mass loading. For example, both HTP and JWPCP discharged approximately 340 mgd in 1996. However, HTP maintains 9 fish assemblage trawl stations distributed over approximately 186 km^2, while JWPCP maintains 12 stations over approximately 80 km^2.

The total effort for assessing bioaccumulation in local fish is not comparable among agencies. The number of tissue samples analyzed per year varies by a factor of three among facilities, ranging from 39 to 120. The number of stations ranges from 2 to 8 among facilities for fish tissue chemistry, and the sampling frequency varies from semi-annual to annual.

Although eight species are targeted for tissue analysis to address bioaccumulation in POTW fish monitoring programs, no single species is measured by all agencies (Table VII-2). Five different species are targeted by only a single agency and only one species is measured by three agencies (hornyhead turbot, *Pleuronichthys verticalis*). White croaker (*Genyonemus lineatus*) and bigmouth sole (*Hippoglossina stomata*) are targeted by only two agencies.

There is little consistency in the approach for selecting tissue types to be analyzed for assessing trends in bioaccumulation. The JWPCP analyzes muscle tissue for chemical analysis, whereas HTP, OCSD and PLWTP analyze both liver and muscle tissues. Sample replication is also inconsistent among facilities. The HTP and PLWTP analyze 3 composite samples for both tissue types in each species, OCSD analyzes tissues in 10 individuals, and JWPCP analyzes 3 composites for one species (Dover sole) and 10 individual samples for another species (white croaker).

A large discrepancy was found in the types of constituents analyzed in fish tissues (Table VII-3). The number of metals analyzed by each agency ranges from 0 to 17. Only two organic analytes, DDTs and PCBs, are common to all agencies. However, some agencies report PCB Aroclors, while others report congeners. Organic constituents that are not analyzed by all four agencies include additional chlorinated pesticides, PAHs, and phenolic compounds.

Evaluation of Existing Effort

Large POTWs have been conducting relatively effective monitoring programs to assess impacts to fish populations and community assemblages over the last 30 years. Although effects on fish communities were conspicuous at some outfalls in the 1970s, little or no effect has been identified at these local scales for the last 10 to 15 years (Stull 1995) other than what can be accounted for by shifts in physical factors (e.g., El Niño). Instead, it appears that assessment of fish communities is currently a regional question asked by the public. Only two large-scale surveys have been conducted in the last 10 years, such as the 1994 and 1998 Regional Monitoring Surveys, both of which have led to similar observations. There is little effect on fish communities and populations near POTWs other than increased biomass and abundance; the occurrence of lesions and gross pathologies remained low (Allen et al.. 1998).

While there is little observable impact currently, managers still want to answer questions about trends to satisfy the public's curiosity about the health of fish communities. Therefore, trend monitoring will still be an important and effective tool for answering management questions. However, most current monitoring programs commingle a spatial grid-based design with a trend design that consists of multiple sampling events. Like the sediment monitoring program that suffers the same problem, this commingled design is inefficient. The problem is compounded in fish monitoring where haul-to-haul and site-to-site variability is naturally large, thus making subtle differences difficult to detect.

Sublethal impacts are largely ignored by most POTW monitoring programs. Sublethal impacts, however, are more sensitive than population and community condition and can indicate exposure to pollutant inputs. The OCSD is the only agency that routinely measures histopathology during its fish surveys; it chronically finds differences among fish caught from impacted areas compared to reference sites. Other investigators, including SCCWRP, have also observed increases in other biomarkers.

The POTW monitoring for bioaccumulation in local fish has been effective at addressing management questions that assess trends within each agency. Every agency has a historical record for its respective species and tissue types, some dating back more than 20 years. These data sets have shown decreases in tissue concentrations, at times more than an order of magnitude, since the 1970s (Stull 1995). These data sets are extremely useful management tools, particularly when combined with reductions in mass emissions and improvements in sediment chemistry and biota.

Although current tissue concentration monitoring provides managers with the ability to assess trends locally, it has been ineffective at assessing bioaccumulation regionally. This can be attributed, in part, to differences among programs. For example, no single species and tissue type was monitored by all agencies. However, the spatial extent of bioaccumulation is not necessarily a local issue and should encompass the cumulative contributions from all sources that discharge to the ocean as well as assessing the range of tissue concentrations from reference areas. Moreover, large-scale regional estimates of bioaccumulation provide useful data to managers for informing the public or for local decision-making.

Recommended Sampling Designs

There are three questions associated with fish monitoring. The first management question addresses the impacts to fish populations and community integrity. It requires a spatial design that is regional in scope. The second management question addresses the temporal trends of potential outfall effects on community integrity and populations. It requires a local temporal design. The third management question addresses the uptake of contaminants in fish. It requires both a regional and a local scale component because the management endpoints associated with each scale are different. In the case of local bioaccumulation, managers are trying to assess if improvements to their facilities are reflected in ecologically higher order consumers near their outfall. In the case of regional bioaccumulation, managers are trying to assess if tissue concentrations have the potential to effect predators that consume contaminant-laden fish. The third question can be accomplished most efficiently by integrating each of its monitoring designs within questions one and two.

Q1: Is the health of fish populations and communities impaired?

We recommend using a regionally-based monitoring design for this spatial extent question. Large-scale regional monitoring designs use inferential statistics to produce estimates of the area impaired. The goal is to produce bar or pie charts that identify km^2 or percent area of impaired. The major sampling design elements for the inferential approach include the distribution and density of sampling sites, replication and frequency, and indicators.

Distribution and density of sampling sites (Regional Design)

Inferential statistics use a stratified-random sampling design for distribution of sampling sites. We recommend readers consult Stevens (1997) for a thorough description of this approach. Ultimately, the regional monitoring steering committee will define the final number of sampling sites and their location.

Approximately 30 sites per strata provides the most efficient allocation of sites based upon power analysis (Figure VI-2). The most efficient number of sites corresponds to the inflection point between sample size and confidence in areal estimates; a greater number of sampling sites would not provide a proportional increase in confidence.

Replication and frequency (Regional Design)

Since the regional monitoring is entirely focused on characterizing more area rather than focusing on characterizing any single location, we recommend that only one replicate (n=1) be collected at each site.

We recommend conducting regional monitoring at a frequency of every five years. This recommendation is based on the fact that existing fish populations and communities are relatively healthy and frequent measurements are not mandated. If a more frequent assessment of trends is a desired product, then see the sampling design recommended for Question Number 2.

Indicators (Regional Design)

We recommend sampling demersal fish using otter trawls, as has been done for many years (Mearns and Stubbs 1974). This will increase the integration among regional and local monitoring programs. Each haul should identify every fish to the lowest taxon possible. Demersal, soft-bottom fish are the preferred indicator for two reasons. First, soft-bottom habitat is the most common benthic habitat in the SCB. Second, sediments can accumulate contaminants from potential pollutant sources, thus demersal fish have the greatest potential for exposure.

To incorporate an assessment of sublethal effects, we recommend continuing measurements of external anomalies, including tumors, fin erosion and lesions, and external parasites. Sublethal indicators such as external anomalies can be more sensitive indicators of contaminant effect than community integrity, and as such, can be an early warning of higher level impacts. Assessment of external anomalies does not require additional sampling effort, since these measurements can be made on fish collected for assessment of community integrity. Other sublethal indicators exist that may provide managers answers to questions regarding fish exposure and the potential for outfalls to act as an epicenter for disease. These indicators include a measure of DNA damage in blood cells (Comet assay) (Tice et al.. 1990) and liver histopathology (CSDOC 1998). While each of these indicated differences among fish in the SCB, none of these indicators has been tracked to a cause and effect relationship either at the individual, population, or ecosystem level. Therefore, we recommend that a special study of sublethal indicators be used in a regional monitoring context to ensure they perform by producing information useful to environmental decision-making. If they do perform, then they should be considered for use in routine local monitoring of exposure.

Q2: Are fish populations and communities changing over time?

We recommend that local fish monitoring designs focus their effort on assessing trends in populations and community assemblages. The major design elements for focusing a local trends monitoring program include location of sites, replication and frequency, and indicators.

Location of sites (Local Trends Design)

Unlike the regional spatial design that favors many sampling sites to provide estimates of areal extent, trend monitoring should focus on specific locations to assess fish conditions at sites of particular management importance. As such, this will be a site specific issue with three options. The three options are another example of adaptive monitoring because each option represents an increase in effort commensurate with amount of environmental impact.

There are some POTWs that discharge to a constrained habitat and do not appear to be impacting the local fish populations or communities. Some small POTWs fit into this category. For these facilities, the design could be as simple as a single site near the zone of initial dilution (ZID) and a reference site. In fact, reference sites could be shared among facilities. In this case, monitoring is used to ensure that conditions are not degrading over time.

The second design option is for POTWs that are having an effect on sediment conditions, or discharge to more than one depth zone or habitat type. For these discharges, multiple sites should be required that encompass the range of impairment, depths, or habitat types, and should include appropriate reference sites in similar depths and habitats.

The third design option is for dischargers that have large impacts or have a demonstrated need (i.e. EPA 301(h) waiver facilities) to assess trends in gradients. The recommended design for these facilities is based on a repeated measures ANOVA analysis of sites along an isobath at varying distances from the outfall (SAIC and MEC 1997). In order to characterize possible outfall effects, the sampling locations should include ZID, near-field, and far field sites.

Regardless of which of the three designs is selected, we recommend that dischargers select the most appropriate sampling sites based on information from either their historic trend monitoring. Selection of historical sites is preferred since this will continue any record that may already exist.

Frequency (Local Trends Program)

The frequency (sample size) for trend monitoring is a function of the degree of natural variability, the amount of observable change over a fixed amount of time, and specified levels of confidence. We recommend using power analysis to determine the optimum frequency for detecting these quantifiable trends. SCB managers, however, could not provide these quantifiable trends. This may be due to the regulatory framework; balanced indigenous populations is a narrative standard and no numerical thresholds or criteria exist. In absence of power analysis, we recommend using annual sampling for monitoring local trends. We further recommend that sampling occur during the least variable time of the year to enhance a manager’s ability to detect trends. The most consistent time of year for fish is summer. Sampling during this time period will also integrate with regional sampling, which has an index period of August and September.

Indicators (Local Trends Program)

We recommend that managers use the same indicators as in the regional design (see Question No. 1). The only exception would be the sublethal indicators that require special studies.

Q3: Is fish tissue contamination changing over time?

There are two approaches for answering this management question. The first approach is at the local scale. In this approach, contaminants in fish tissues provide managers a link between facility discharges and uptake in biota. This approach is strictly a local question; the biota simply serve as an integrator of discharges over time. In the second approach, contaminants in fish tissues provide managers a link between predator and prey and the capability of contaminants moving up the food chain. This is a regional question that integrates many sources of pollutants and addresses the wide feeding ranges of most higher level predators such as larger fish, birds, or marine mammals. We recommend using distinctly different designs for these two approaches in order to optimize efficiency for answering each question.

Local Tissue Monitoring

Local trend monitoring has three sampling design elements including number and location of sampling sites, frequency, and indicators.

Number and location of sites (Local trends program). We recommend using the three options for site selection outlined in the *Q2: Are fish populations and communities changing over time.* The number of sites and their location is dependent upon the amount of potential impact observed near the outfall; the larger the impact the greater the number of sites that need to be monitored. This adaptive monitoring strategy is the most efficient for different types of discharges.

Frequency (Local Trends Program). We recommend using the frequency outlined in the *Q2: Are fish populations and communities changing over time.* Without quantifiable trends required for managers, we recommend a default frequency of sampling annually during the summer.

Indicators (Local Trends Program). Although our comparison of programs identified a large discrepancy in the species and tissues that are measured among current monitoring programs, we recommend that existing designs be used for assessing local trends in bioaccumulation. In some cases, 25 years of tissue concentrations have been measured and maintaining that history is the best approach. If new programs are begun, or minor additions are made, we recommend monitoring the species held most in common among the large POTWs (Hornyhead turbot). Similarly, we recommend that liver tissues be analyzed if new programs are begun since livers typically have concentrations 10-fold higher than muscle tissues and detectable values are necessary for managers to assess changes in concentrations. Finally, we recommend only measuring substances that bioaccumulate as part of a local fish tissues monitoring program. These compounds include chlorinated hydrocarbons (DDTs, PCBs) and certain elements (Hg, Se, As).

Regional Tissue Monitoring

Regional trend monitoring has three sampling design elements including number and location of sampling sites, frequency, and indicators.

Number and location of sites (Regional trends program). We recommend using the number and location of sites outlined in the *Q1: Are fish populations and communities impaired*. The number of sites and their location is dependent upon a stratified-random sampling design. While the final number and location of sites is ultimately dependent upon the regional monitoring steering committee, sample sizes of approximately 30 sites per strata is the most efficient.

Frequency (Regional Trends Program). We recommend using the frequency outlined in the *Q1: Are fish populations and communities impaired*. We recommended a sampling frequency of every five years to occur during the summer.

Indicators (Regional Trends Program). Since no single species occurs at all depths across the entire SCB region, we recommend using feeding guilds that will achieve regional coverage. Feeding guilds represent ecologically similar species that occur in different habitats. Although the regional monitoring steering committee will make the ultimate decision, the sanddab guild is the preferred demersal fish species guild in the SCB (Table VII-4) (Allen et al. in press). These species are closely related to sediment, easily captured, abundant, relatively sedentary, and are consumed by larger fish.

We recommend that whole fish be sampled rather than individual tissue types. This is because predators will consume the entire individual. The constituents measured should include those with wildlife-risk thresholds (DDT, PCB, toxaphene, methylmercury) (Table VII-5).

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TABLE VII-1. Fish assemblage trawling effort.

	HTP	JWPCP	OCSD	PLWTP
# Trawls/year	56	48	44	38
Sampling frequency	Quarterly	Quarterly	Semiannually	Quarterly/ Semiannually
Area sampled (km^2)	186	31	36	75
Trawl depths	18 m	23 m	18 m	60 m (semiannual)
	60 m	61 m	36 m	88 m (quarterly)
	150 m	137 m	55 m	104 m (quarterly)
			60 m	
			137 m	

TABLE VII-2. Target species for trawl-caught fish bioaccumulation that can be used for wildlife protection assessments.

Species	HTP	JWPCP	OCSD	PLWTP
white croaker	–	M	M,L	–
hornyhead turbot	M,L	–	M,L	M,L
bigmouth sole	–	–	M,L	M,L
Dover sole	–	M	–	–
barred sand bass	–	–	M,L	–
longfin sanddab	–	–	–	M,L
Pacific sanddab	–	–	–	M,L
California scorpionfish	–	–	–	M,L
speckled sanddab	–	–	–	M,L

M = Muscle
L = Liver

TABLE VII-3. Constituent analysis reporting levels for fish and invertebrate tissue samples. Dash = not analyzed.

Constituent	HTP muscle	HTP liver	JWPCP muscle & liver	OCSD muscle & liver	PLWTP muscle & liver
Metals/Metalloids (mg/wet Kg)					
Aluminum	-	-	-	-	2.6
Antimony	0.07	0.25	-	-	3.7
Arsenic	0.1	0.5	-	-	1.4
Beryllium	0.01	0.05	-	-	0.035
Cadmium	0.04	0.04	-	-	0.34
Chromium	0.1	0.08	-	-	0.33
Copper	0.26	0.28	-	-	0.76
Iron	-	-	-	-	1.3
Lead	0.2	0.6	-	-	2.5
Manganese	-	-	-	-	0.2
Mercury	0.02	0.17	-	0.02	0.012
Nickel	0.15	0.5	-	-	0.79
Selenium	0.12	0.6	-	-	0.13
Silver	0.01	0.05	-	-	0.62
Thallium	0.1	0.5	-	-	5.7
Tin	-	-	-	-	4.6
Zinc	0.7	1.2	-	-	0.58
Organics					
DDT (µg/wet Kg)	0.5-2	0.5-2	5	0.1-0.6	8.8-48.4
PCB (µg/wet Kg)	10-20	10-20	20	4.9	4-7
Remaining organochlorine pesticides (µg/wet Kg)	0.5-15	0.5-15	-	0.1-8	0.6-1.9
Total organic halides (mg/wet Kg)	7	-	-	-	-
Base/neutral/acid extractables (mg/wet Kg)	0.16-325	0.16-325		7-25	0.012-0.48

TABLE VII-4. Fish species in the sanddab guild that were targeted for bioaccumulation assessment in the 1998 Regional Monitoring program (Bight'98) by all four dischargers.

Species
Speckled sanddab (*Citharichthys stigmaeus*)
Longfin sanddab (*Citharichthys xanthostigma*)
Gulf sanddab (*Citharichthys fragilis*)
Pacific sanddab (*Citharichthys sordidus*)
Petrale sole (juvenile) (*Eopsetta jordani*)
Slender sole (*Eopsetta exilis*)
California halibut (juvenile) (*Paralichthys californicus*)

TABLE VII-5. Environment Canada predator-risk tissue residue guidelines

Constituent	Guideline concentration
DDTs	14.0 µg/kg
Toxaphene	6.3 µg/kg
Total PCBs	0.79 ng TEQ/kg
Methylmercury	33.0 µg/kg

VIII. SEAFOOD SAFETY MONITORING

The management question that addresses seafood safety in the SCB is:

Q1: Are seafood tissue concentrations below levels that will ensure public safety?

"*Is the seafood fish safe to eat*?" is perhaps one of the most frequently asked questions by the public (NRC 1990). In the SCB, the California EPA Office of Environmental Health and Hazard Assessment (OEHHA) is the agency with jurisdiction over seafood safety that answers this question (Bernstein et al.. 1999). Because it is so difficult to determine the actual number of people contracting illness (i.e. cancer) from eating contaminated seafood, OEHHA uses quantitative risk assessments that estimate the increase in the incidence of illness (i.e. one in 100,000) based upon specified consumption rates (i.e. one meal a week). From these relationships, OEHHA derives advisory tissue concentrations (ATCs) for each chemical of concern. If edible seafood tissues exceed the ATC thresholds, then OEHHA may post seafood advisories.

Compare and Contrast Among Agencies

Not all large POTWs conduct seafood tissue monitoring (Table VI-1). Three large POTWs conduct rig fishing and two of these agencies also conduct invertebrate tissue chemistry analyses. Of the large POTWs that do conduct seafood monitoring, the annual number of sportfish tissue samples analyzed differs by a factor of five and the number of invertebrate tissue samples analyzed differs by a factor of three.

Among the large POTWs that conduct seafood monitoring, the disparity in sampling effort is a result of differences in sampling frequency and number of species targeted per site (Table VI-1 and VI-2). The sampling frequency among agencies for sportfish ranged from three times per year to once every two years. The sampling frequency for invertebrate seafood monitoring ranged from twice per year to once every two years.

No single species was targeted by all three large POTWs that analyze sportfish, or by both of the two agencies that analyzed invertebrates (Table VI-2). The number of species targeted at each site also differed among facilities. The HTP collected three fish species at each site, PLWTP collected one species of rockfish at each site, and JWPCP targets two species at each site.

The existing large POTW monitoring programs measure different tissues and different chemical parameters. The HTP analyzes muscle tissue from fish and invertebrate species, the PLWTP analyzes both muscle and liver tissue from fish, and the JWPCP analyzes muscle and liver tissues from fish and gonad from red sea urchins (*Strongylocentrotus franciscanus*). Of the three large POTWs that do conduct seafood monitoring, only 2 of 22 parameters are measured in common (DDTs and PCBs).

Evaluation of Existing Effort

The seafood monitoring programs have been ineffective at addressing management questions regarding human health. Only three of the four largest POTWs conduct seafood sampling and analysis as part of their routine monitoring. Moreover, no common approach or design has been adopted for making assessments of seafood safety among these three agencies. For example, the programs sample and analyze a variety of species, at dissimilar frequencies, and with different target analytes. This has begun to change, with both HTP and JWPCP working together to jointly design a program for Santa Monica Bay.

The lack of monitoring by some agencies, coupled with inconsistencies among agencies that do monitor, prevent finding an answer to what should be a regional question. *Is the seafood safe to eat?* is a question that needs to be addressed not just near POTW outfalls, but at all locations where fish are caught for consumption. For example, no routine monitoring program has been established for fish that are caught by sport fishermen from commercial passenger fishing vessels, piers or beaches. Not only is seafood monitoring a regional question, but the sources of seafood contaminants need to be more broadly defined and costs appropriated. Although POTWs are not the only contributor of pollutants that can bioaccumulate in seafood, it is the only group of dischargers that conducts any routine seafood monitoring.

Perhaps the greatest inefficiency in the seafood monitoring program, however, is that the POTWs are not the managers who make decisions about seafood for human consumption. It is OEHHA, not POTW managers, that posts fish advisories or closures; the primary decision-makers are not integrated into the monitoring design. Once again, this has begun to change in Santa Monica Bay, where OEHHA assisted in the development of the new HTP and JWPCP seafood monitoring design.

Recommended Sampling Designs

Since POTWs are not the users of the monitoring data, we recommend that large POTWs integrate their seafood monitoring programs with OEHHA's monitoring designs to address the management needs for seafood advisories. Moreover, POTWs are not the only source of inputs to the ocean of contaminants that can accumulate in seafood. Therefore, other sources for these contaminants of concern should participate in this monitoring program. In absence of OEHHA taking the lead in guiding a regional seafood monitoring program, we outline some of the major design elements below.

Q1: Are seafood tissue concentrations below levels that will ensure public safety?

Sampling locations

Since this program will be designed to protect recreational anglers, we recommend that the sampling sites be located where anglers fish. This should include not only POTW outfalls, but also include piers, party boats, and the shoreline. We recommend using the sampling location paradigm established by Bernstein et al. (1999) for Santa Monica Bay. While repeated visits to a single location can provide good assessments about that site, recreational anglers often visit more than one site. Therefore, creating zones or clusters of sampling sites may provide a better assessment of risk than just a single location. This is particularly true for anglers who fish from (small) boats or commercial passenger fishing vessels.

Frequency

The frequency of sampling is a function of distance from an ATC and the rate of change. We recommend using the frequency paradigm established by Bernstein et al. (1999) for Santa Monica Bay. In general, tissue concentrations do not change very rapidly, so adequate sampling for regional assessments could be conducted every five years. If tissue concentrations are near the ATC and/or they are changing rapidly, annual sampling will provide a better assessment of trends. The general rule is if the concentration is expected to cross the ATC within 10 years, then a shift to annual sampling is recommended.

The most appropriate season to sample will depend on when recreational seafood are caught, which may vary by species. For example, collection of some species is limited by migration patterns (e.g., warmer-water finfish extend into the SCB during summer months), while collection of other species is limited by legal sport license seasons (e.g., lobster season is October through March).

Indicators (Human Health)

The species of seafood to collect for tissue analysis should be fish and invertebrates that are caught and consumed by recreational anglers. Likewise, the tissues to be analyzed for bioaccumulation should be those consumed by humans, which is primarily muscle in finfish, but includes gonad tissue in sea urchins.

We recommend measuring only those contaminants that bioaccumulate in seafood tissues and that OEHHA uses for managing seafood advisories (Table VI-3). This list includes total DDTs, PCBs, arsenic, mercury, and selenium.

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TABLE VIII-1. Sportfish sampling effort. Dash = not sampled.

	HTP	JWPCP	OCSD	PLWTP
# Stations	3 zones	3 zones	-	2 zones
# Samples / year	90	18*	-	24
Sampling frequency	triannually	biennially	-	semiannually

* = half of biennial value

TABLE VIII-2. Current seafood target species. Dash = not sampled

HTP	JWPCP	OCSD	PLWTP
white croaker California scorpionfish ocean whitefish squarespot rockfish barred sand bass cabezon *Sebastes* yellow rock crab	white croaker kelp bass red sea urchin	–	*Sebastes*

TABLE VIII-3. Seafood consumption fish tissue concentration action levels

Constituent	OEHHA limit (mg/kg)	USFDA tolerances, action levels, and guidance levels (mg/kg)	Median international standard (mg/kg)
Metals/Metalloids			
Arsenic	1.0	86	1.4
Mercury	0.3	1	0.50
Selenium	20		0.3
Organics			
Chlordane	0.3	0.3	
DDT	0.1	5.0	5
PCB	0.02	2.0	

IX. KELP MONITORING

There are two management questions for kelp monitoring:

Q1: ***Are kelp beds changing in areal extent over time, if so are some locations changing at different rates than others?***

Q2: ***Are kelp bed communities healthy?***

Giant kelp (*Macrocystis pyrifera*) plays an important ecological role in the coastal marine environment of the SCB (North and Hubbs 1968). Kelp beds are amongst the most productive marine ecosystems in the world. Giant kelp is a dominant food source for many organisms in the SCB. It also forms large structures that provide refuge and habitat for many rocky subtidal invertebrates and fish. Finally, kelp beds are a nursery ground for several marine species. Because of its unique ecology and beauty, kelp beds are also a valuable resource to the public as recreational diving and fishing areas, and for commercial harvesting of kelp and other kelp forest organisms.

Several factors can negatively effect kelp recruitment, growth, and survival (North and Hubbs 1968) eventually devastating kelp beds: 1) winter storms; 2) warm, nutrient poor water that dominates the SCB during El Niño conditions; 3) grazing such as sea urchin predation; and 4) sedimentation. Anthropogenic-related factors can also contribute to kelp declines including: 1) reduced water clarity; 2) silt deposition; and 3) contamination. In fact, kelp has been used as a standard toxicity testing organism to assess the potential impacts of effluent discharges (Chapman et al.. 1995).

The first management question, "*are kelp beds changing in areal extent over time?*", examines changes in the size of kelp beds as an indicator of kelp health. Often, kelp bed extent is operationally defined as the spatial coverage of the kelp canopy during low tide. The extent of kelp canopy can change dramatically because of natural conditions, often shrinking to a fraction of its spring extent during the fall months. The large degree of natural variability makes it difficult to assess anthropogenic impacts. As a result, managers must use data from other kelp beds, distant from anthropogenic sources, to compare to kelp beds near anthropogenic sources.

The second question "*are kelp bed communities healthy?*" examines the condition of not just kelp, but the entire kelp bed ecosystem including other algae, fish and invertebrate assemblages. For managers, this is a much more integrated and intensive effort to assess. While areal extent of canopy coverage can be examined from remote sensing such as aerial photography, kelp bed community assessments require underwater divers and equipment; a very laborious and expensive proposition. Moreover, managers will see tremendous natural variability at this level as well, often confounding the potential impacts from anthropogenic sources. However, examining conditions at the community level may be the only method to verify if a kelp bed is being impacted or to explain what is causing the disturbances observed in kelp canopy areal extent.

Compare and Contrast Among Agencies

Only two of the four large POTWs in southern California have kelp monitoring programs. Moreover, these two agencies use very different approaches to monitor kelp canopy and kelp bed communities. The PLWTP uses quarterly aerial photography surveys to examine changes in the canopy size of 19 different kelp beds from Corona del Mar to Point Loma (North and Jones 1991). This program is conducted in collaboration with the six other NPDES ocean dischargers within the San Diego RWQCB jurisdiction. In contrast, the LACSD conducts diver surveys of kelp bed communities. The LACSD maintains 12 diver stations located in three depth zones in a single kelp bed off Palos Verdes. Annual sampling at these sites includes fish transects and rocky subtidal quadrats.

PLWTP and LACSD also contribute to, or use data from, special studies on kelp bed dynamics. The PLWTP supports ongoing research on kelp bed communities in the Point Loma Kelp bed conducted by investigators at the Scripps Institute of Oceanography. The LACSD uses aerial overflight information supplied by the California Department of Fish and Game that conducts aerial photography of the Palos Verdes kelp bed every two years.

Evaluation of Existing Effort

Aerial surveys are an effective and efficient approach for addressing the management question *"are kelp beds changing in areal extent over time?"*. This type of monitoring is efficient because they can gather information from many kelp beds in the SCB during a very short time interval ($\leq$ 1 day) at relatively low cost. Only by comparing kelp beds throughout the SCB, can managers distinguish between site-specific effects and those changes caused by oceanographic events occurring on a regional scale. Moreover, many surveys can occur per year using this monitoring approach enabling a better assessment of trends.

Although aerial overflight photography is efficient at answering the first management question, it is completely ineffective at answering the second management question "*are kelp bed communities healthy?*". The monitoring approach used in the Palos Verdes kelp bed is the more effective design for assessing this management need. However, the LACSD is also inefficient because there has not been a demonstrated need for this type of assessment. In fact, there has not been an observable effect in the Palos Verdes kelp bed since the 1970's (MBC 1988). Impacts from POTWs are expected to be even less likely when LACSD shifts to full secondary treatment.

Recommended Sampling Designs

We recommend an adaptive monitoring strategy for kelp bed assessments. This adaptive monitoring strategy is built upon two fundamental concepts. First, kelp bed aerial extent is a regional issue. An understanding of large-scale responses to oceanographic conditions is required before an understanding of outfall effects at local scales can be assessed. The second concept is that while aerial photography can be an efficient and effective screening tool to observe changes in local kelp beds, managers need to investigate further before an assessment of cause and effect can be ascertained; proximity to potential anthropogenic sources cannot be used to infer degradation. This adaptive monitoring strategy is woven into a stepwise implementation of the two management questions. Ongoing monitoring for Q1 should occur, but Q2 should be undertaken when Q1 indicates a need for further investigation.

Q1: Are kelp beds changing in areal extent over time and, if so, are some locations changing at different rates than others?

We recommend using aerial photography or other remote sensing technology to address this management question. Moreover, this approach should be a regional design whereby multiple kelp beds, preferably from the entire SCB, are simultaneously monitored. Not only should all kelp beds be monitored, but we suggest that all ocean dischargers should participate in this program, even if kelp does not exist near their discharge. Kelp beds are important ecological areas subject to cumulative impacts as well as an important public resource. The most cost-efficient regional monitoring design is a collaborative approach whereby all agencies participate in a common program.

We do not recommend specific sampling design elements for this management question for two reasons. First, this should be a regionally designed program and it is inappropriate to recommend specific monitoring design elements without having the collaborative agencies together to identify their specific management needs. Second, there are many agencies tasked with stewardship of kelp bed resources including the California Department of Fish and Game, RWQCB, National Park Service, and California Coastal Commission. It is also inappropriate to recommend specific monitoring design elements without having all of the data users involved in identifying what data they require for decision-making.

Q2: Are kelp bed communities healthy?

This question should be considered only when there is a demonstrated need, such as when kelp bed extent is changing abnormally compared to other kelp beds in the SCB (see question one). In a sense, this question becomes similar to a special study, looking to assess if kelp bed communities are also being degraded or perhaps to determine the potential cause(s) of the impairment.

There are a variety of techniques available for making assessments of kelp bed community health. These include assessments of physical habitat (i.e. quality and quantity of rocky substrate), giant kelp abundance and density using either vessel mounted (e.g. side-scan sonar) or underwater (i.e. diver transects) methods, plus assessments of invertebrate and fish population or community assemblage parameters. The use of each of these techniques is completely site-specific. Therefore, it is inappropriate to recommend specific monitoring design elements to answer this question.

References

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X. SEABIRD MONITORING

There is one management question for seabird monitoring that is important relative to large POTWs:

Q1: Are contaminants bioaccumulating in seabirds?

Many seabirds are near the top of the food chain in the near-coastal ecosystem of the SCB. As such, they are susceptible to food chain biomagnification of contaminants discharged from large POTWs. The best example is the California brown pelican (Anderson and Hickey 1970). Total DDT, discharged from the Montrose Corporation through the LACSD ocean outfall, entered the food chain and bioaccumulated in eggs of this once threatened species. The result was a thinning of the eggshell, which eventually cracked during nesting. The reproductive failures led to large population declines during the 1970's. When the discharge of total DDT was ceased, the levels in the eggs of brown pelicans slowly decreased. Brown pelicans now enjoy a robust breeding population (Fry et al.. 1987).

Compare and Contrast Among Agencies

None of the large POTWs currently have a seabird monitoring program.

Evaluation of Existing Effort

There are two main reasons why seabird monitoring is not conducted by local POTWs. First, the spatial scale for seabird monitoring is not limited to a single facility. Most seabirds have extended ranges; some ranges exceed the entire SCB. Therefore, a local monitoring program, and perhaps even a regional monitoring program, would be ineffective at addressing management questions about seabird bioaccumulation. Second, large POTWs are not the agencies that make decisions regarding managing the health of seabird populations.

Recommended Sampling Design

We recommend that large POTWs become minor contributors to a larger-scale monitoring program, should such a program be established by those agencies responsible for seabird management. This recommendation is based upon two reasons. First, bioaccumulation in seabirds is not a local monitoring design. It is the cumulative impact from all discharges that leads to bioaccumulation in higher trophic level organisms that have extended ranges. Therefore, other agencies and types of discharges should contribute to such a monitoring program. Second, POTWs are not the only threat to seabird populations. Seabird managers also perceive oil spills, commercial fishing, and loss of breeding habitat as major threats to seabird populations (Baird 1993). Only the agencies responsible for seabird management can unite the various components of an integrated monitoring program and, therefore, should lead the effort to assess seabird health.

References

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XI. MARINE MAMMAL MONITORING

There is one management question for marine mammal monitoring that is important relative to large POTWs:

Q1: Are contaminants bioaccumulating in marine mammals?

Many marine mammals are near the top of the food chain in the near-coastal ecosystem of the SCB. As such, they are susceptible to food chain biomagnification of contaminants discharged from large POTWs. The best example is the California sea lion (DeLong et al.. 1973). Total DDT, discharged from the Montrose Corporation through the LACSD ocean outfall, entered the food chain and bioaccumulated in mothers of this once threatened species. The result was premature pupping, which eventually led to death of the newborn. The reproductive failures led to large population declines during the 1970's. When the discharge of total DDT was ceased, the levels of total DDT in the pups slowly decreased. California sea lions now enjoy a robust breeding population.

Compare and Contrast Among Agencies

None of the large POTWs currently have a marine mammal monitoring program.

Evaluation of Existing Effort

There are two main reasons why marine mammal monitoring is not conducted by local POTWs. First, the spatial scale for marine mammal monitoring is not limited to a single facility. Most marine mammals have extended ranges; some ranges exceed the entire SCB. Therefore, a local monitoring program, and perhaps even a regional monitoring program, would be ineffective at addressing management questions about marine mammal bioaccumulation. Second, large POTWs are not the agencies that make decisions regarding managing the health of marine mammal populations.

Recommended Sampling Design

We recommend that large POTWs become minor contributors to a larger-scale monitoring program, should such a program be established by those agencies responsible for marine mammal management. This recommendation is based upon two rationale. First, bioaccumulation in marine mammals is not a local monitoring design. It is the cumulative impact from all discharges that leads to bioaccumulation in higher trophic level organisms that have extended ranges. Therefore, other agencies and types of discharges should contribute to such a monitoring program. Second, POTWs are not the only threat to marine mammal populations. Managers also perceive oil spills and commercial fishing as major threats to marine mammal populations (Bonell and Dailey 1993). Only the agencies responsible for marine mammal management can unite the various components of an integrated monitoring program and, therefore, should lead the effort to assess marine mammal health.

References

Bonnel, M. and M. Dailey. 1993. Marine mammals pp 604-681, *In* (Dailey, M., D. Reish, and J. Anderson eds) Ecology of the southern California Bight: a synthesis and interpretation. UC Press London, England.

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XII. WETLANDS MONITORING

Coastal wetlands in the SCB are becoming rare habitat. In Santa Monica Bay alone, more than 95% of the historic wetlands have been lost due to development, dredge and fill operations, or construction of marinas (SMBRP 1988). Not only is habitat loss a matter of management concern, but several environmental stressors exist for wetlands including urban runoff and discharges from inland treatment plants and industrial facilities.

Compare and Contrast Among Agencies

None of the large POTWs currently have a wetland monitoring program.

Evaluation and Recommended Sampling Design

There is no likely mode of impact to wetland areas from large POTWs. The four facilities evaluated in this report do not discharge into or upstream of wetlands. Large POTWs in southern California discharge far offshore and are unlikely to impact wetland areas. Moreover, the agencies responsible for the large POTWs do not have any jurisdiction over wetland management decision-making. Consequently, we recommend that POTWs with ocean outfalls should not be required to participate in wetlands monitoring. Therefore, we do not recommend monitoring questions or sampling designs.

References

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XIII. INTERTIDAL HABITAT MONITORING

Intertidal habitats in the SCB are highly variable, ranging from rocky intertidal habitats to sandy shoreline areas and mudflats. A common feature among intertidal communities is their ability to withstand natural environmental stressors; organisms in these habitats must deal with desiccation and wave action on a daily basis. A second common feature is their risk from anthropogenic stressors. Habitat loss and trampling are amongst the most common anthropogenic stressors for intertidal communities, but risk from pollutant sources such as urban runoff and discharges from inland treatment plants and industrial facilities also exist.

Compare and Contrast

None of the large POTWs currently have an intertidal monitoring program.

Evaluation and Recommended Sampling Design

There is no likely mode of impact to intertidal areas from large POTWs. The four facilities evaluated in this report do not discharge into or upstream of intertidal habitats. Large POTWs in southern California discharge far offshore and currently have no intertidal monitoring program. Moreover, the agencies responsible for the large POTWs do not have any jurisdiction over intertidal management decision-making. Consequently, we recommend that POTWs with ocean outfalls should not be required to participate in intertidal monitoring. Therefore, we do not recommend monitoring questions or sampling designs.