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ANALYSES OF AMBIENT MONITORING DATA FOR THE SOUTHERN CALIFORNIA BIGHT

Prepared for:

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FINAL

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TABLE OF CONTENTS

TABLES

FIGURES

EXECUTIVE SUMMARY

This report represents a first step toward bringing together the monitoring data on benthic communities and sediment parameters from the Southern California Bight into a regional data set. The main body of the report is divided into three sections in which we:

- identify the data sets available and assess their inter-compatibilities,
- utilize the available data on benthic communities to describe the regional patterns in community structure, and
- propose an analytical procedure that allows deviations from the regionally derived reference conditions to be assessed.

In Section 1, seven data sets are compared in terms of their frequency and methods of collection, parameters measured, analytical methods used, and data storage format. Problems were noted in taxonomic standardization due to revisions of taxonomy and differences between taxonomist ability and identification procedures. Physical and chemical parameters present greater standardization problems due to methodological differences and choices of parameters to be measured.

In Section 2, the data on benthic community structure, sediment grain size and depth were compiled from all available sources and analyzed to detect patterns in community structure. This section discusses:

- the results of ordination analysis of benthic community structure which show the effects of outfalls on community structure,
- measures of community structure, community organization, population measures and indicator species, and
- the population responses of two commonly used indicator species, the ophiuroid *Amphiodia urtica* and the bivalve *Parvilucina tenuisculpta*, to gradients of organic enrichment and pollution.

In Section 3 a method of analysis is proposed for the assessment of deviations from regional reference conditions due to anthropogenic influences. The definition of reference communities is discussed and illustrated using a subset of the data presented in Section 2. Requirements for indicator variables are discussed, and a statistical test for assessing deviations from reference conditions is outlined. Examples of the technique's application are presented. The statistical procedure is discussed in Appendix 3A.

The report concludes with a set of recommendations for future work including:

- expand regional monitoring efforts based on consistent management objectives for the entire Southern California Bight,
- further expand and assess the technique proposed in Section 3 for additional data sources,
- develop needs and priorities for standardizing historical data sets,
- develop guidelines for standardization of data and monitoring efforts on a regional basis, and
- develop an overall monitoring program incorporating both point source and regional monitoring requirements.

PREFACE

Annually over $17 million is spent on point source monitoring programs within the boundaries of the Southern California Bight (NRC 1990a). In recent years, the present approach of point source monitoring has drawn criticism from several sources (ex. NRC 1990a), especially regarding its ability to make region-wide assessments. The data obtained or analyses performed are often incompatible between monitoring programs and are difficult to obtain, with the result that regional trends cannot be determined from a synthesis of the available information. Furthermore, problems associated with the lack of clear monitoring objectives often result in poorly designed monitoring programs. Several recently published critiques of current monitoring programs in the Southern California Bight have pointed to the need to coordinate monitoring programs on a regional basis (Thomas 1988; Ford and Conway 1988, 1989; NRC 1990b).

The United States Environmental Protection Agency, Region IX, wishes to develop regional approaches toward monitoring programs. Such an approach should ideally allow for discrimination between natural disturbances (ex. long-term environmental changes or episodic events) that affect the region as a whole, and anthropogenic disturbances such as point and non-point source discharges.

The ultimate goal of a regional monitoring strategy would be to determine the extent of natural variation (both spatial and temporal) in chemical, physical and biological components of the ecosystem that occurs in natural (i.e. non-impacted) reference conditions and to be able to use these results to assess deviations from reference conditions. A truly regional approach requires the cooperation of regulatory agencies, dischargers and independent monitoring agencies. It would be necessary to obtain periodic information on reference conditions, including biological community structure, sedimentary parameters and chemistry, and oceanographic data. Additionally, discharge compliance monitoring must be standardized in terms of the variables measured, methodology used, and the manner in which the data are reported.

For reasons that will be discussed in Section 1, a complete regional assessment is not feasible at this time given the scope of this project. This report, therefore deals primarily with benthic community data from the Southern California Bight along with limited sediment data.

In this report we achieve three goals which are described as individual tasks. Task 1 was to assess the inter-compatibility of the various data sets obtainable from Southern California Coastal Water Research Project

(SCCWRP) reference surveys, other regional surveys and available discharger data. In Task 2 the benthic community data from the surveys were analyzed to characterize regional reference conditions. And in Task 3 our understanding of the reference conditions was used to develop an approach for assessing deviations from reference conditions. This approach was designed to allow monitoring for compliance with State and Federal regulations. We conclude the report with a series of recommendations for further development of the regional monitoring data base.

SECTION 1 - FEASIBILITY OF A REGIONAL DATA SET

1.1 *Introduction*

The study of reference conditions for the Southern California benthos will require an examination of available data throughout the bight. It is not sufficient to utilize data only from areas considered to be unaffected by human activities or other unusual environmental conditions. A broad overview of data from representative benthic habitats, including altered habitats, is necessary as a first step. Such an overview will enable one to see and contrast the characteristics of benthic communities in a range of different habitats. It is not always perfectly clear which areas truly represent ambient conditions. The benthos in some areas thought to be relatively pristine may have characteristics of altered habitats, or areas suspected to have been altered may resemble other areas known to be unaffected. In the end, this process will enable one to understand the full range of community responses and thereby sort out the relatively unaffected areas from areas moderately or seriously altered. Only after the reference areas are accurately delimited can the study of the reference characteristics in these areas proceed.

Another advantage of having data that includes reference and altered habitats is that it becomes possible to observe how various aspects of the benthic community change as one goes from an unaltered to an altered habitat. Such information will be useful in developing and choosing indicator variables that could be used to assess compliance with sediment and water quality standards.

It will also be beneficial to obtain data that include temporal coverage. Long term data sets can be also used to study the effects of natural environmental conditions (e.g., winter storms, El Niño, gradual oceanographic changes over time) on the benthos in both reference and altered areas.

The larger sewage outfalls discharging at about 60 meters depth are one of the major human activities potentially affecting the subtidal benthos in the Southern California Bight. The monitoring programs associated with these outfalls are the sources of much of the available benthic data from the area. Additional benthic data are available from SCCWRP (Southern California Coastal Water Research Project), the Bureau of Land Management (BLM), and the State Water Quality Control Board. All the discharger and SCCWRP data were obtained at water depths between 30 and 300 meters. In this report, data taken within this depth range are emphasized. Much of the other available data are from spatially limited surveys in harbors or around power plants associated with shallower water depths. Such data would represent localized, atypical conditions when compared to the bulk of the benthic data.

The present study will also need to consider data measuring habitat characteristics (depth, physical and chemical sediment parameters) at the benthic sampling locations. This information is of direct interest, since it would be incomplete and unsatisfying to study the patterns in the benthos without reference to the corresponding habitat patterns. In addition, information on the habitat will be helpful in separating representative reference locations from other altered or atypical locations.

In some cases, it would also be of interest to obtain information on water quality, water currents, and other oceanographic parameters. Such parameters are usually quite variable in the short term, making snapshot measurements taken during the benthic sampling poor indicators of general conditions in the area. The less variable sediment parameters will often be better indicators of general oceanographic conditions. For example, sediment grain size can reflect current speed, and the general bottom current direction can usually be seen from the distribution of discharged materials in the sediments.

The goal of this section is to examine available benthic and sediment data and determine if these data can utilized in the study of reference conditions. To accomplish this goal, we evaluate the data in terms of spatial and temporal coverage, and describe the present incompatibilities that exist within and between the different data sources. Finally, we suggest approaches that could address the data gaps and incompatibilities that would prevent a unified and more complete analysis of ambient conditions.

1.2 *Data Sources*

The data from the four largest outfall monitoring programs in the region are examined. The outfalls are managed by the City of Los Angeles (Hyperion Treatment Plant, Playa del Rey), the County Sanitation Districts of Los Angeles (CSDLA, Whites Point on the Palos Verdes Peninsula), the County Sanitation Districts of Orange County (Huntington Beach), and the City of San Diego (Point Loma). Data from SCCWRP include three "control" or "reference" surveys that most importantly include many potential ambient locations that are apparently beyond the direct influence of the outfalls (Word and Mearns 1979, Thompson *et al.* 1985). The sampling locations for these programs are shown in Figures 1-1 to 1-6. Descriptions of the monitoring programs are found in NRC (1990) and SCCWRP (1988).

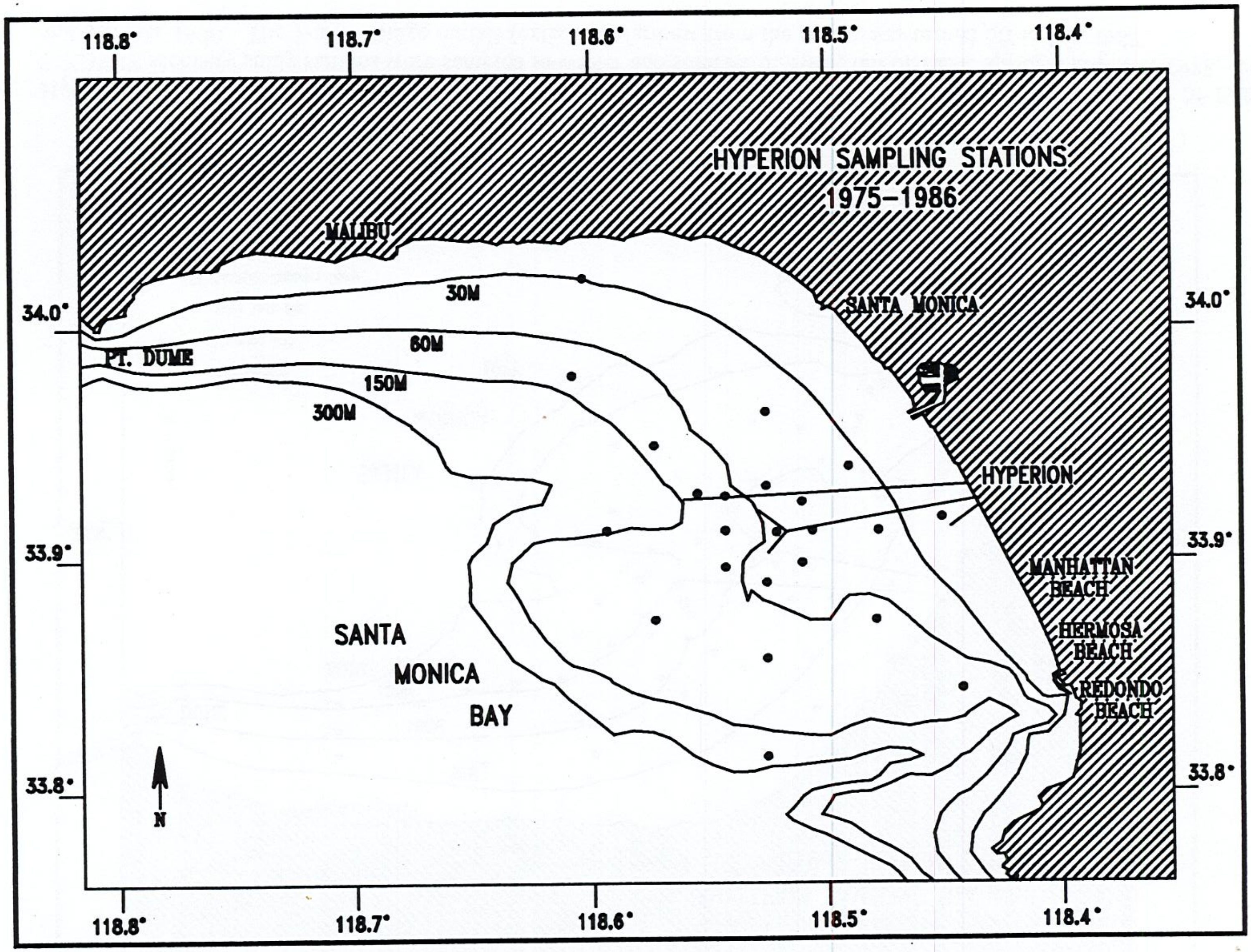


Figure 1-1. Hyperion sampling stations from 1975 to 1986. Samples were taken in summer and winter of each year.

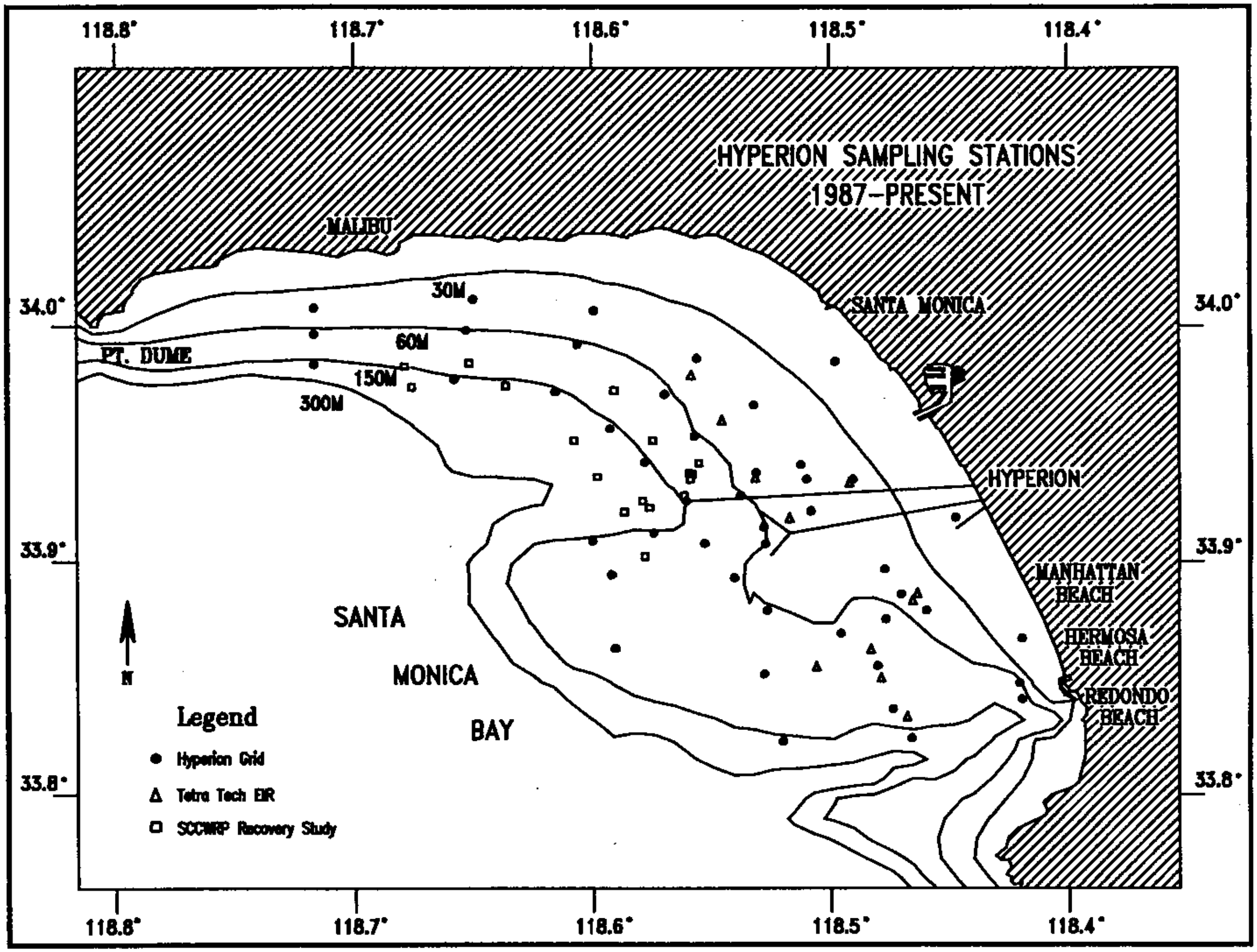


Figure 1-2. Hyperion sampling stations since 1987. The ten Tetra Tech EIR stations were sampled in winter and summer of 1989. The 17 SCCWRP recovery study stations were sampled in winter and summer of 1986, fall of 1987, spring and fall of 1988, winter 1989, and summer 1990. The 7-mile sludge outfall (extending farthest from the coast) was turned off in late 1987.

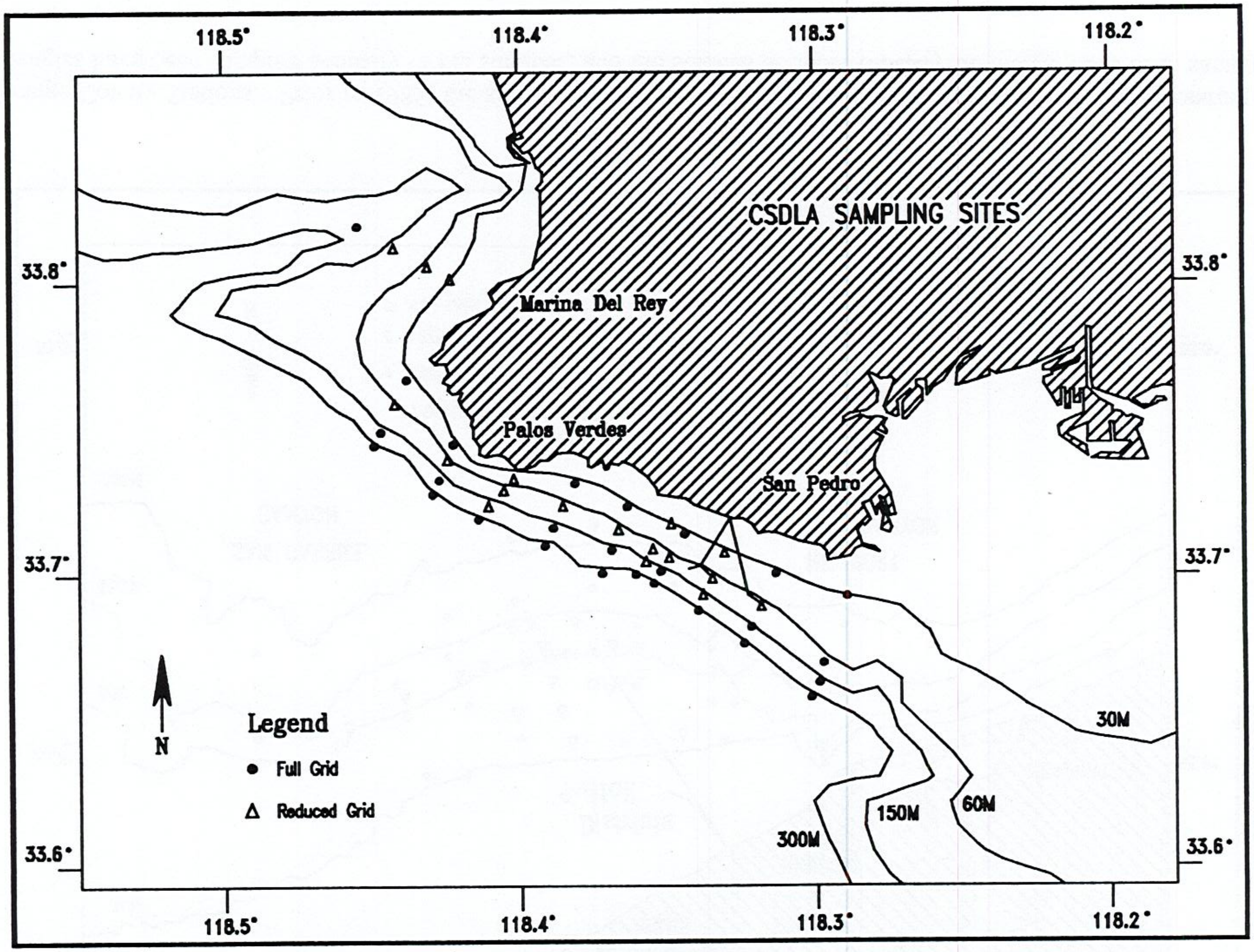


Figure 1-3. County Sanitation Districts of Los Angeles sampling stations since 1972. County Sanitation Districts of Los Angeles (CSDLA) sampling stations. The full grid includes both symbols. From 1972 to 1976, the full grid was sampled, with the exception of four most northern stations in 1972 and 1973. From 1977 to the present, summer and winter samples were taken on either the full or reduced grid. Since 1988, the full grid was sampled in the summer and the reduced grid was sampled in the winter.

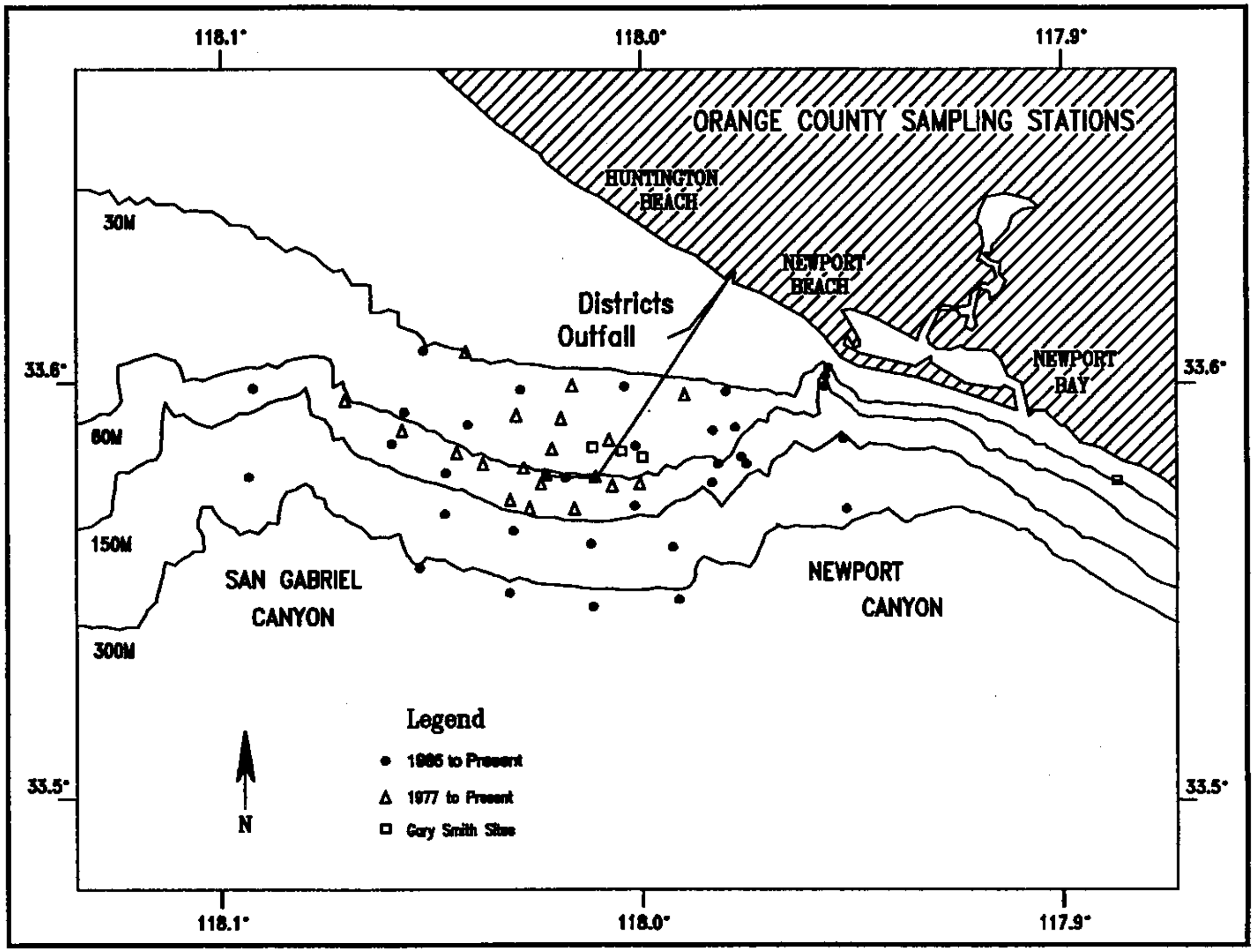


Figure 1-4. Orange County Stations. Prior to 1985, the samples were taken annually. Since 1985, the stations corresponding to the dots and triangles have been sampled annually in the summer, and the stations at approximately 60 meters have been sampled quarterly.

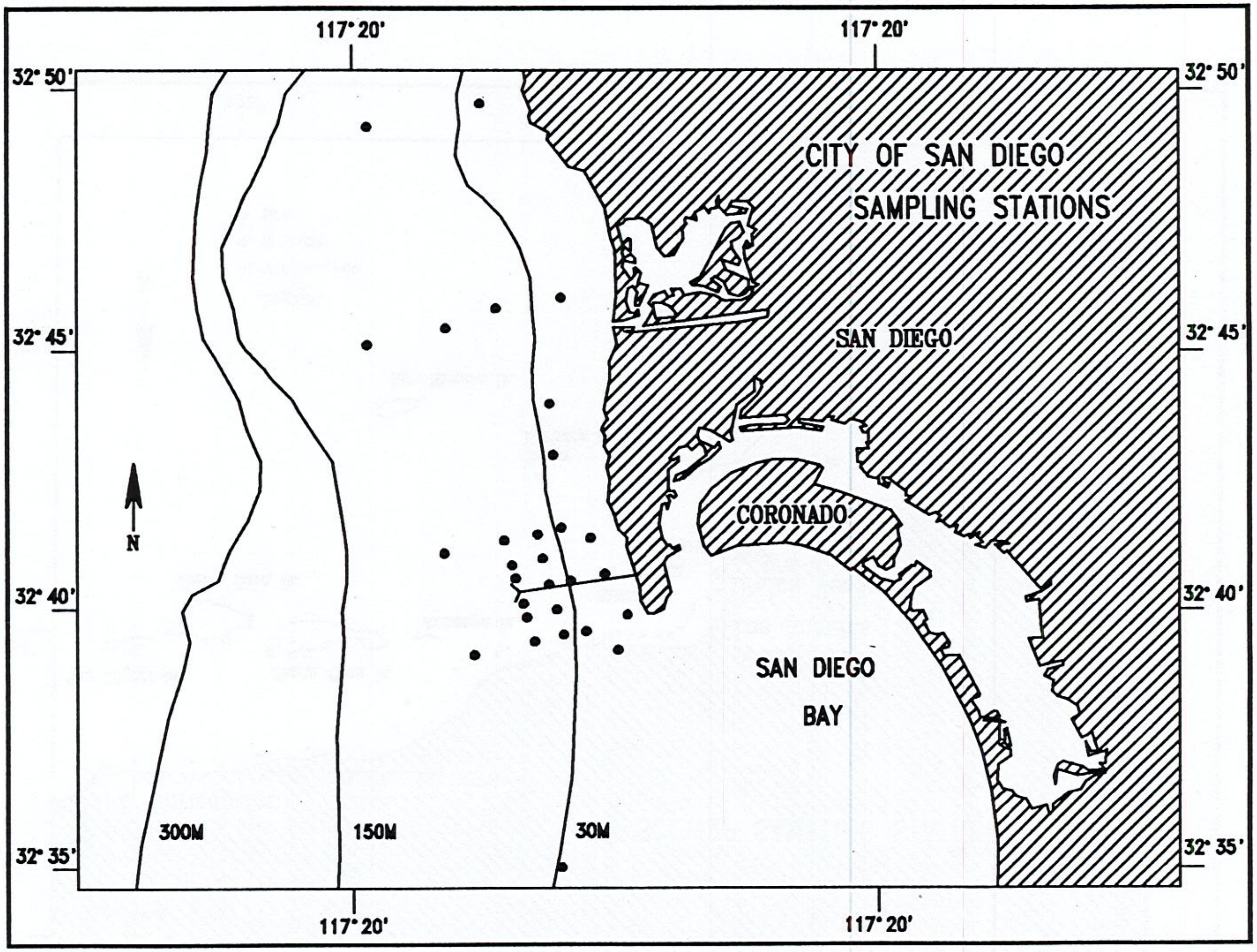


Figure 1-5. Point Loma sampling stations. Samples were taken quarterly.

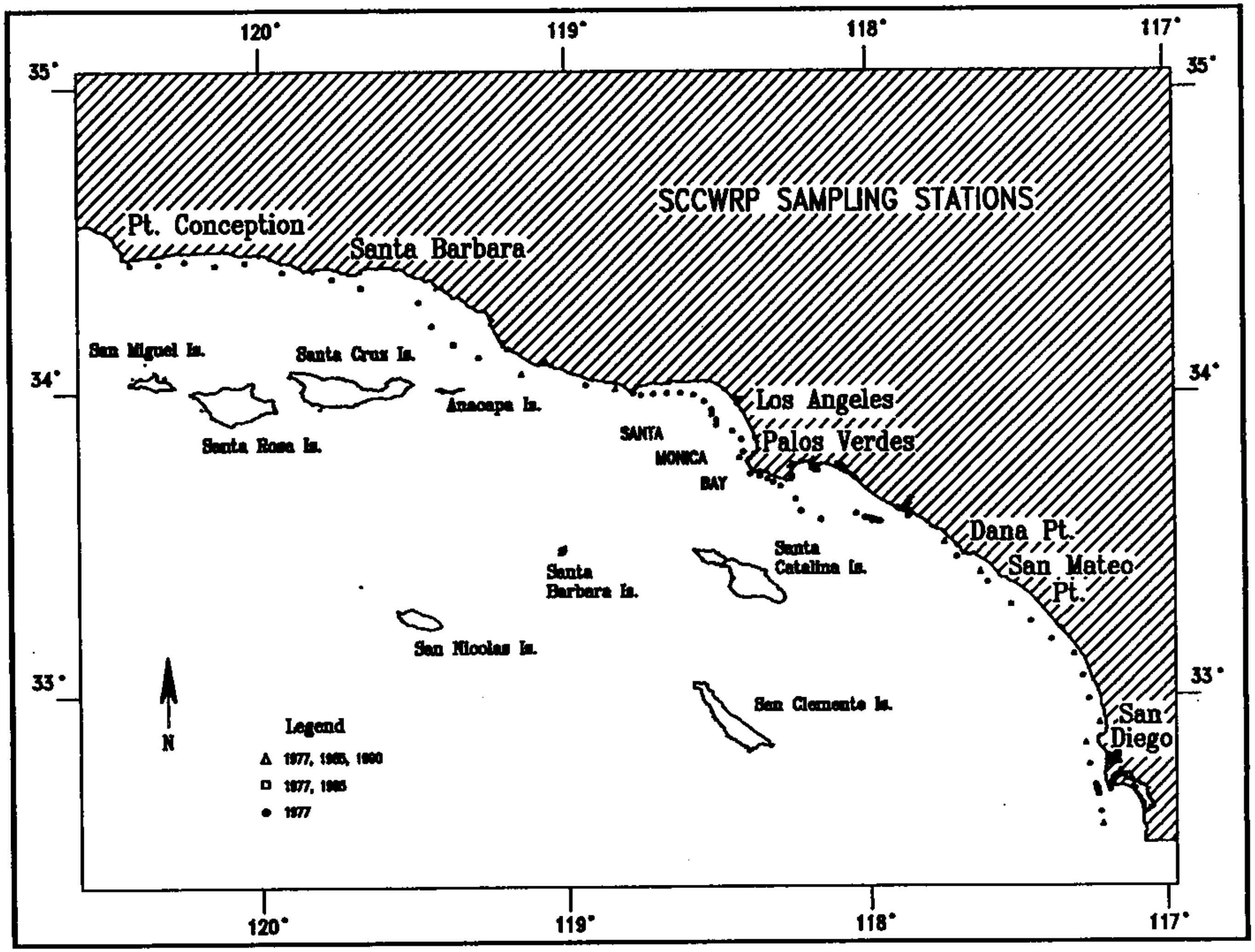


Figure 1-6. SCCWRP reference survey sampling stations. Only 60 meter stations shown. In 1985 and 1990, samples were also taken at 30 and 150 meters depth at a lesser number of stations along transects perpendicular to the coastline.

Some relevant data from separate studies are included with the dischargers' data. Since 1986, SCCWRP has been studying the effects on the benthos of the termination of discharge (in 1987) from the 7-mile sludge outfall of the Hyperion Treatment Plant (Thompson and Dorsey 1989). In 1989, two benthic surveys were undertaken by Tetra Tech Inc. to support an Environmental Impact Report for a proposed new Hyperion outfall in Santa Monica Bay. Data from these two projects have been taxonomically standardized to match the Hyperion benthic data and are presently stored in the same data base management system with the Hyperion monitoring data. Thus, data from these two projects are considered as part of the Hyperion data.

Smith (1974) observed the benthos before and after the present Orange County discharge was initiated in 1971. Benthic infaunal specimens from this project have recently been reidentified to be taxonomically compatible with the Orange County monitoring data, and will be considered as part of the Orange County surveys.

Finally, data from two large-scale benthic sampling projects in Southern California are evaluated. The first, known as the State survey, was sponsored by the State Water Quality Control Board and covered the time period from 1956 to 1959 (Allan Hancock Foundation 1965, Jones 1969). The other project, to be called the BLM survey, was sponsored by the Bureau of Land Management (BLM, now the Minerals Management Service or MMS) from 1975 to 1978 (Fauchald and Jones, 1977, 1978a, 1978b). The pertinent sampling locations for these two projects are shown in Figures 1-7 and 1-8, respectively.

1.3 *Spatial and Temporal Coverage*

Benthic infauna

Figure 1-9 summarizes the station locations as shown in Figures 1-1 to 1-8. When assessing spatial variability, it is important to hold time relatively constant; otherwise spatial and temporal variability will become confounded. Thus, it is important to evaluate both the spatial and temporal coverage of the data. Table 1-1 shows the temporal overlap of the different programs.

The spatial coverage appears extensive when all times are pooled (Figure 1-9), but at any one time, good coverage of reference locations existed only when samples from a large-scale project such as the State, BLM,

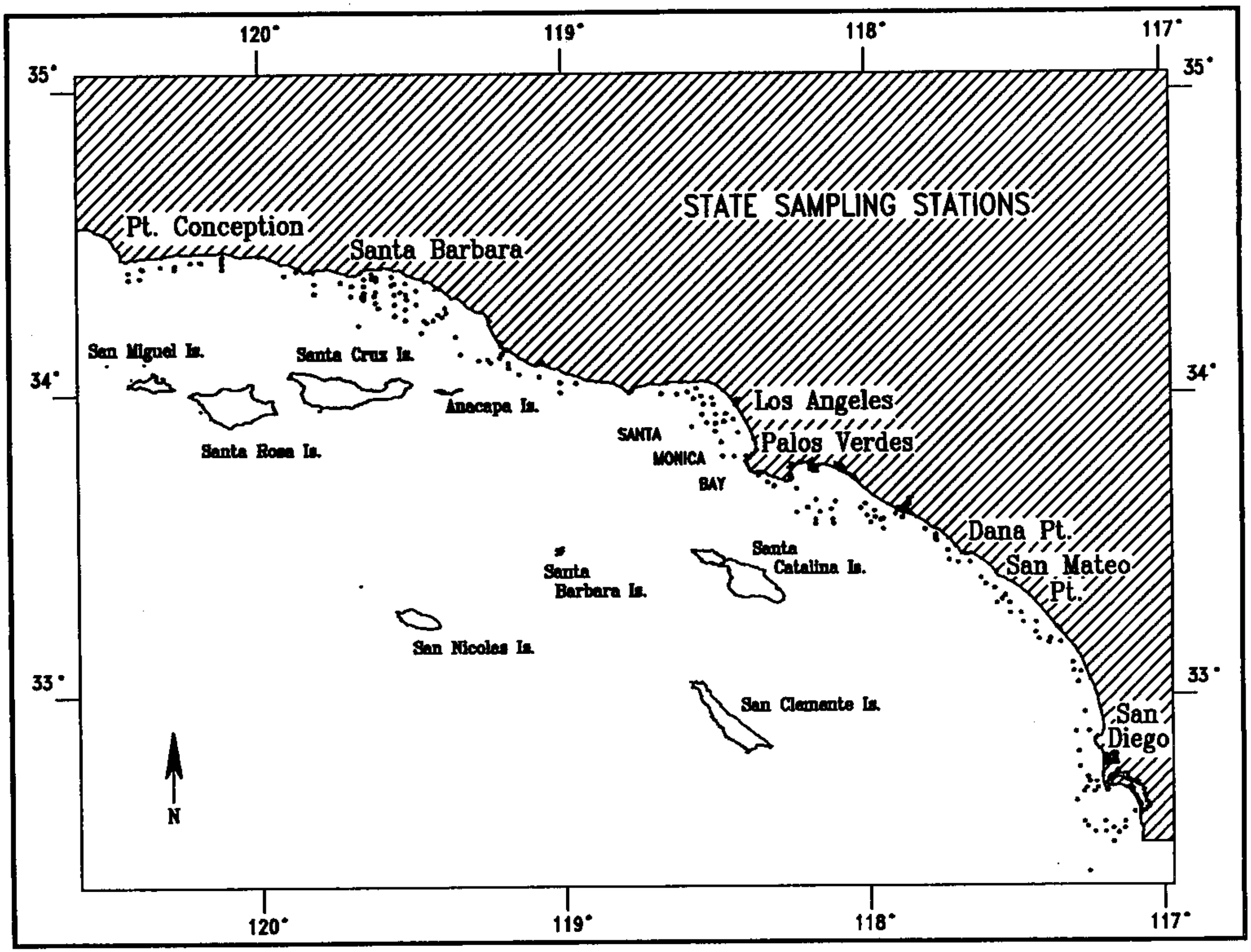


Figure 1-7. State survey sampling stations at depths from 30 to 300 meters.

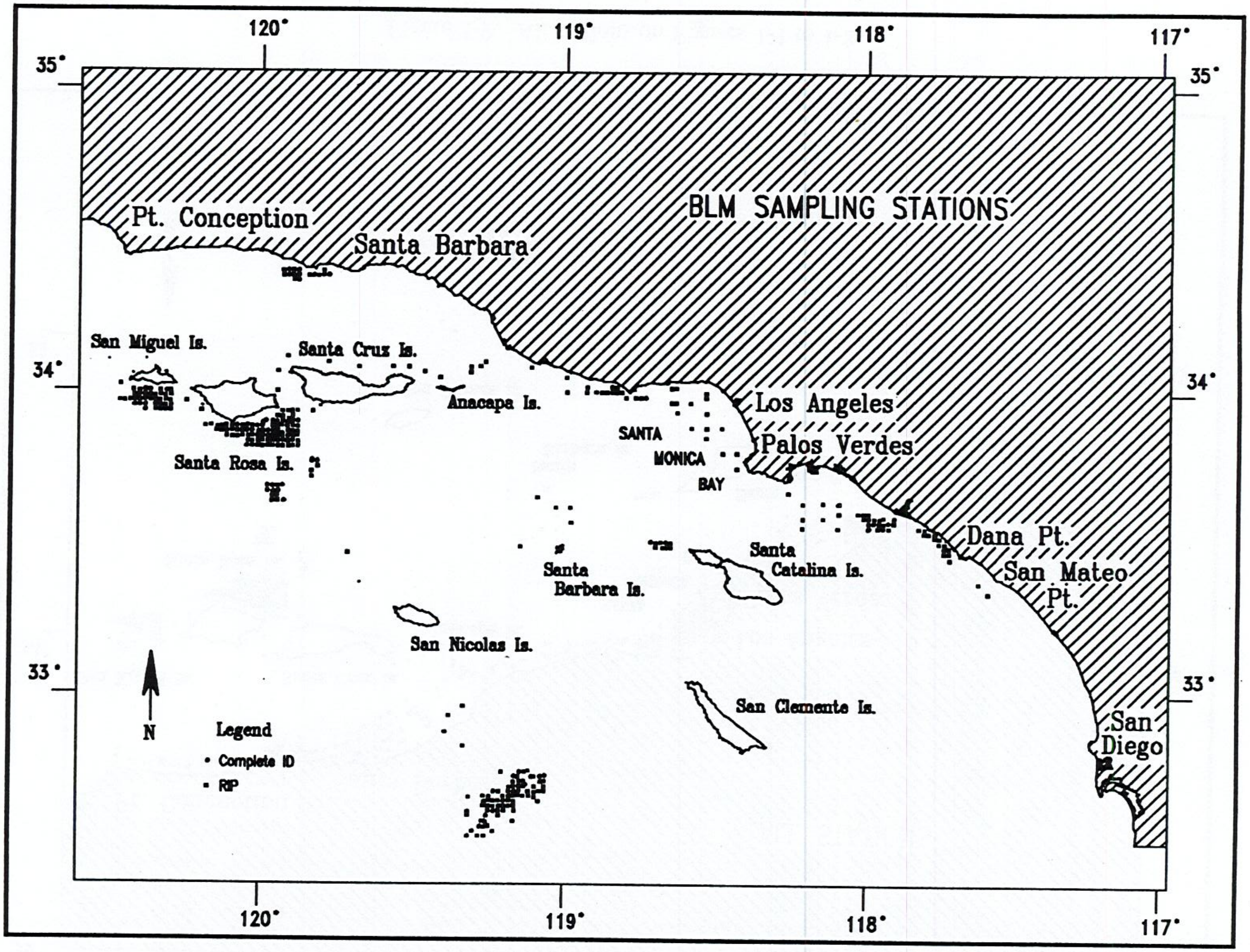


Figure 1-8. BLM sampling stations at depths from 30 to 300 meters. Station symbols indicate where the rapid identification procedure (RIP) or complete taxonomic identifications were used.

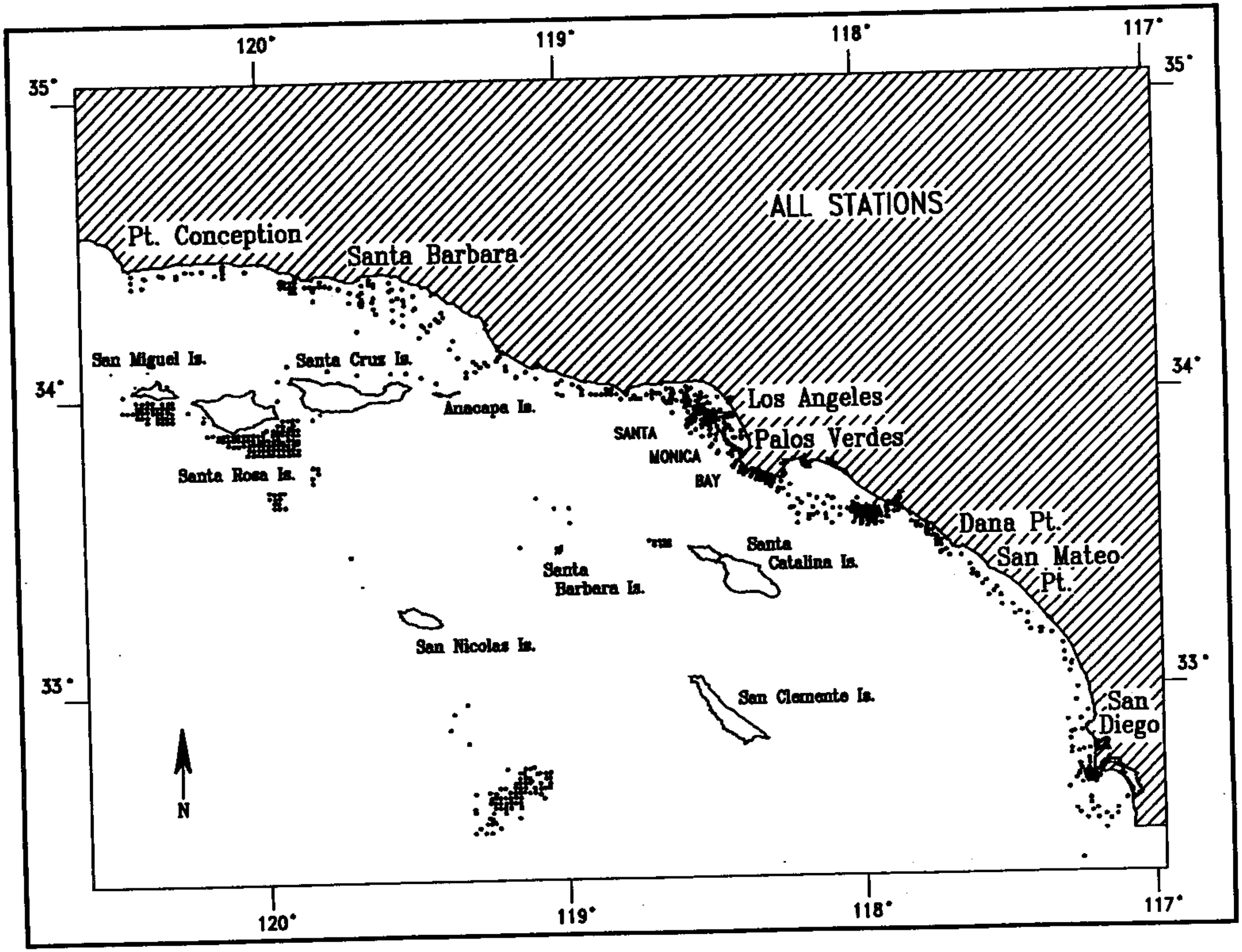
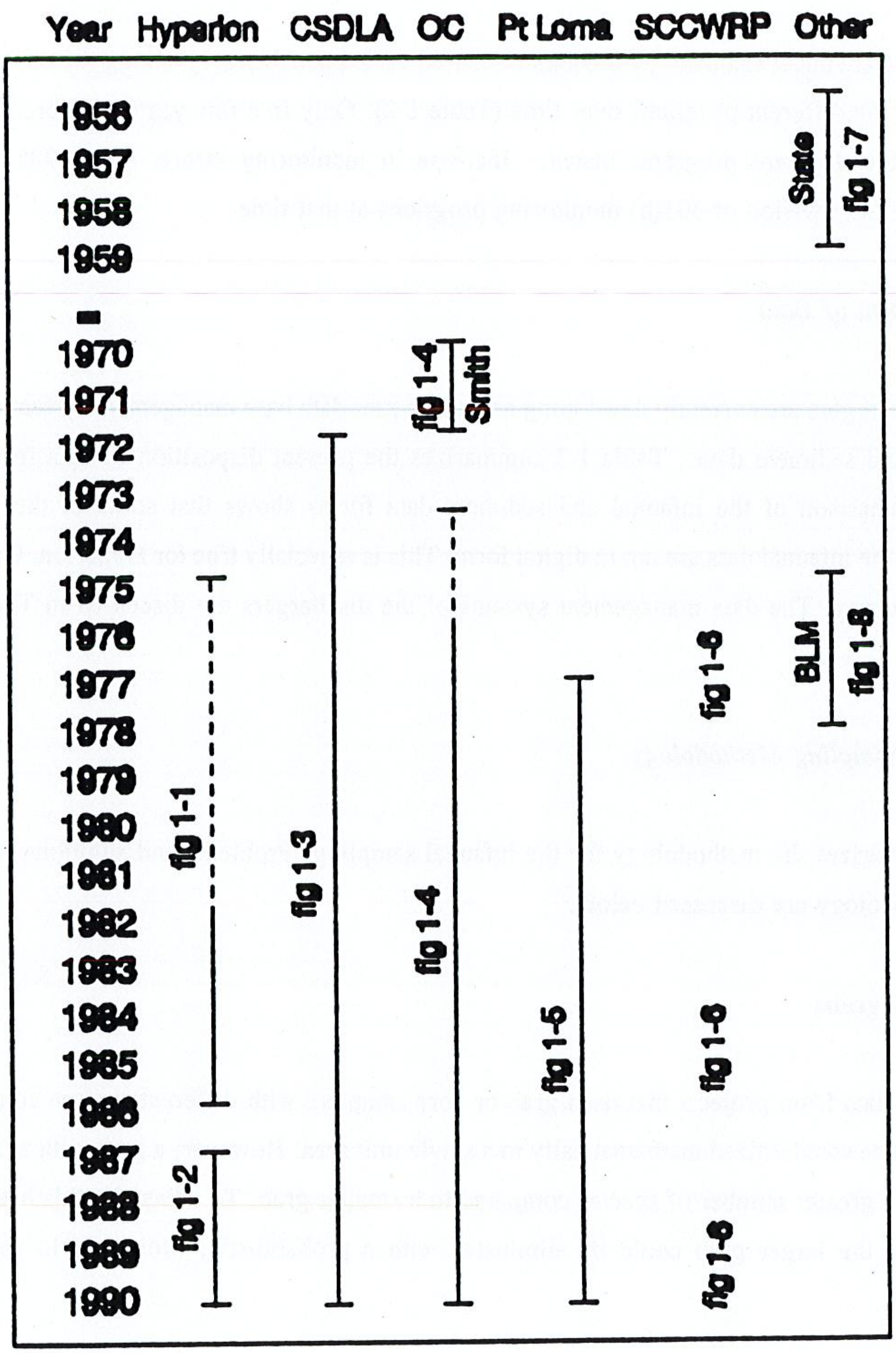


Figure 1-9. All stations on Figures 1-1 to 1-8.

Table 1-1

TEMPORAL COVERAGE OF BENTHIC INFAUNAL SAMPLING. FIGURES INDICATED SHOW STATION LOCATIONS AND SAMPLING TIMES WITHIN THE YEAR



or SCCWRP survey was available. This is due to the limited number of potential reference stations available from the discharger monitoring programs.

Sediment parameters.

The physical and chemical sediment parameters measured varied considerably among the different programs and also within the different programs over time (Table 1-2). Only in a few years do more than just a few parameters in the different programs match. Increase in monitoring efforts after 1985 is due to the implementation and revision of 301(h) monitoring programs at that time.

1.4 *Present Form of Data*

Most of the dischargers are currently developing new and better data base management systems for managing their infaunal and sediment data. Table 1-3 summarizes the present disposition of data from the various sources. A comparison of the infaunal and sediment data forms shows that some of the sediment data associated with the infaunal data are not in digital form. This is especially true for Hyperion, Orange County, and the State survey. The data management systems of the dischargers are discussed in Thompson *et al.* (1991).

1.5 *Infaunal Sampling Methodology*

Table 1-4 summarizes the methodology for the infaunal sampling. Problems and solutions associated with different methodology are discussed below.

Comparison of grabs

When merging data from projects that used grab or core samplers with different surface areas, the species abundances can be standardized mathematically to a single unit area. However, a grab with a larger area will tend to sample a greater number of species compared to a smaller grab. To adjust for this bias, some of the rarer species in the larger grab could be eliminated with a probabilistic, Monte Carlo approach (to be developed).

Table 1-2

TEMPORAL COVERAGE OF SEDIMENT MEASUREMENTS

	Pre 1970	1970	1971	1972	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990
SedimentGrain Size Parameters																						
%Sand	W			C	C		B		SP	P	P	P	P	CP	P	CP	SOP	OP	HOP	HCOP	HCOP	SHCOP
%Silt	W			C	C		B		SP	P	P	P	P	CP	P	CP	SOP	OP	HOP	HCOP	HCOP	SHCOP
%Clay	W			C	C		B		SP	P	P	P	P	CP	P	CP	SOP	OP	HOP	HCOP	HCOP	SHCOP
QualitativeSedimentComposition		O	O	O											H	H	H	H				
SedimentPhi Size Intervals					C	O	O	O	O	O	O	O	O	O	O	CO	O	O	O	O	O	CO
Metals																						
Silver						O			S								SCOP	OP	HOP	HOP	HOP	SHCOP
Cadmium					C	O			S				H	H	H	H	SHCOP	HOP	HOP	HOP	HOP	SHCOP
Chromium					C	O			S						H	H	SHCOP	HOP	HOP	HOP	HCOP	SHCOP
Copper					C	O			S				H	H	H	H	SHCOP	HOP	HOP	HOP	HOP	SHCOP
Nickel					C	O			S						H	H	SCOP	OP	OP	OP	OP	SHCOP
Lead					C	O			S				H	H	H	H	SHCOP	HOP	HOP	HOP	HOP	SHCOP
Zinc					C	O			S				H	H	H	H	SHCOP	HOP	HOP	HOP	HOP	SHCOP
Arsenic																	O	O	HO	HO	HO	HCO
Berylium																	O	O	HO	HO	HO	HCO
Mercury																	O	O	HO	HO	HO	HCO
Selenium																	O	O	HO	HO	HO	HO
Thallium																	O	O	HO	HO	HO	HO
UnspecifiedMetals	P	CP	CP	P	P	OP	OP	OP	SOP	OP	OP	OP	OP	OP	OP	COP						
CaCO3	W						B	B	B	B												
VolatileSolids(percentor total)									S						H	H	SHO	HO	O	O	O	CO
OrganicCarbon(percentor total)		O	O	O			B	B	SB	B							S	O	HO	HO	HO	HCO
ChemicalOxygenDemand						P	P	P	P	P	P	P	P	P	HP	HP	HOP	HOP	OP	COP	COP	COP
OrganicNitrogen	W			C	C	CP	CP	CP	CP	CP	CP	CP	CP	CP	HCP	HCP	HC	HC	C	C	C	C
BiologicalOxygenDemand	P					P	P	P	P	P	P	P	P	P	HP	HP	HP	HP	P	P	P	P
Sulfides		O	O	O					P	P	P	CP	CHP	CHP	CP	CP	COP	COP	HCOP	HCOP	HCOP	HCOP
QualitativeSulfideAssessment		C	C	C	C	C	C	C	C	C	C	C	C	C	HC	HC	HC	HC	C	C	C	C
Oil & Grease						P	P	P	P	P	P	P	HP	HP	P	P	O	O	HO	HO	HO	HO
Pesticides& PriorityPollutants																						
DDTs		C	C		C	P	P	P	SP	P	P	P	P	P	P	P	SHCOP	O	HP	HP	HO	SHCO
PCBs		C	C			P	P	P	SP	P	P	P	P	HP	HP	HP	SHCOP	OP	OP	OP	HOP	SHCOP
PAHs									S								SOP	OP	HOP	HOP	HOP	SHOP
VOAs																	OP	OP	HOP	HOP	HOP	HOP
Other or UnspecifiedPesticides	P	P	P	P	P	P	P	P	P	P	P	P	P	P	HP	HP	SHOP	HOP	HOP	HOP	HOP	SHOP
UnspecifiedPriorityPollutants and/or ChlorinatedHydrocarbons					C				S						H	CH	SHOP	HOP	HOP	HCOP	HCOP	SHCOP
Cyanide									P	P	P	P	P	P	P	P	OP	OP	OP	OP	COP	COP
Phenols/PhenoliCompounds									P	P	P	P	P	P	P	P	OP	OP	HOP	HOP	HOP	HOP
SedimentTemperature					C	P	P	P	P	P	P	P	P	P	P	P	CP	CP	CP	CP	CP	CP
SedimentEh													C	C	C		C	C	C	C	C	C
SedimentpH	W												C	C	C		C	C	C	C	C	C
SedimentEs												C	C	C	C	C	C	C	C	C	C	C
CSDLA Qual SedimentEvaluation		C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C	C
RadioactiveCompounds									P	P	P	P	P	P	P	P	P	P	P	P	P	P
Dioxins																	P	P	P	P	P	P

Legend: C = CSDLA, O = OrangeCounty, S = SCCWRP, H = Hyperion, P = Pt. Loma, W = StateSurvey, B = BLM

Table 1-3

PRESENT STORAGE FORM OF INFAUNAL SAMPLING AND SEDIMENT SAMPLING. ANY DATA OUTSIDE THE DATE RANGES IN PARENTHESES DATA ARE AVAILABLE IN WRITTEN FORM ONLY

DATA SOURCE	INFAUNA	SEDIMENT
Hyperion	SAS data sets (1983-present)	SAS data sets (1987-present)
CSDLA	Data files	Data files
Orange Co.	Paradox data base (1977-1984) PRODAS data base (1985-present)	PRODAS data base (1985-present)
Pt. Loma	SAS data sets (1988-present)	SAS data sets
SCCWRP	SAS data sets	SAS data sets & ASCII files on PC
State	Data tape	Report only
BLM	FOCUS data base	FOCUS data base

Table 1-4

SUMMARY OF BENTHIC INFAUNAL SAMPLING PROCEDURES

PROJECT	GRAB	GRAB AREA	SCREEN SIZE	REPLICATION
Hyperion	Van Veen	.1 m^2	1 mm	5 reps at 60 m (1987-1991 summer only) rest 1 rep
CSDLA	Shipek 1972-1979	.04 m^2	1 mm	4 reps
	Van Veen 1980-1991	.1 m^2		5 reps at 5 60 m stations (1989-1990 summer only) rest 1 rep
Orange Co.	Van Veen 1970-1972	.1 m^2	.5 & 1 mm	4 reps
	Peterson 1977-1984	.1 m^2	1 mm	1 rep
	Van Veen 1985-1991	.1 m^2	1 mm	5 reps at 60 m (1985-1991) rest 1 rep
Pt. Loma	Van Veen	.1 m^2	1 mm	5 reps
SCCWRP	Van Veen	.1 m^2	1 mm	1 rep
State	Orange Peel	.25 m^2	.71 mm	1 rep
BLM	Box Core	.063 m^2	.5 & 1 mm	mostly 1 rep

The efficiency of the different grabs may vary. The Shipek grab tends to have a bow wave that can prevent the capture of lightweight surface organisms. In addition, the surface water tends to drain out of the grab as it is raised to the surface, potentially causing further loss of lightweight or floating organisms. When merging data from more efficient sampling devices with Shipek data, care should be taken to deemphasize or eliminate from consideration smaller, lightweight organisms that may be lost with the Shipek grab. The Orange Peel grab has been criticized as being inefficient by Hedgpeth (1957), but defended by Jones (1969).

Comparison of screen sizes.

Data from a 1 mm screen are available from all projects with the exception of the State survey, which used a .71mm screen size. This would present a problem if the State survey data were to be merged with any of the other data. In this case, analyses would have to be confined to larger organisms that would not be differentially sampled by the two screen sizes.

Comparison of replication

The bulk of the data consists of a single replicate at a station. For the last few years, the dischargers have replicated some stations at outfall depth (around 60 m).

When comparing data from stations with different numbers of replicates, some caution must be exercised. For example, if the data from the replicates at the stations are averaged to compensate for the differences in numbers of replicates, the stations with greater numbers of replicates will still tend to contain a greater number of species. With multivariate methods such as ordination, stations with many replicates can be overemphasized at the expense of stations with fewer replicates. Analysis of variance procedures can become unbalanced and potentially problematical when the replication varies greatly from treatment to treatment.

To avoid such problems arising from variable replication at the stations, one may want to use the same number of replicates or a single replicate for each station. Often this will involve elimination of some of the replicates at stations where the larger number of replicates are taken. For community analyses not requiring statistical tests, the best way to determine which replicates to retain at a station when some must be eliminated is to compute dissimilarity index values comparing the replicates at a station. The replicate(s) with the lowest average dissimilarity to the other replicates should be retained. When this is done, the most typical replicate(s) are retained and outliers will tend to be eliminated. When statistical tests are to be applied to the data, the replicates to be used should be randomly selected.

1.6 *Infaunal Taxonomy*

The present lack of taxonomic standardization makes it very difficult to merge data from the different projects. We know from direct experience with all the pertinent data that the same infaunal organisms are not always given the same taxonomic designation in different projects. In the absence of an aggressive between-project taxonomic standardization program, and in the presence of differing levels of taxonomic expertise and specialization among the different projects, this situation is inevitable. It would be much worse if it were not for the work of the SCAMIT (Southern California Association of Marine Invertebrate Taxonomists), which sets taxonomic standards for the Southern California infauna. The taxonomists of the dischargers and SCCWRP are all members of this organization. Unfortunately, this standardization is only partially reflected in the data bases from the different projects. Many more detailed procedures and much more effort will be required before the data from the different projects will be entirely compatible. SCCWRP and EcoAnalysis have proposed to the Santa Monica Bay Restoration Project (SMBRP, part of the EPA NEP program) to develop a detailed plan for taxonomic standardization of historical and future benthic infaunal data from the Southern California Bight. This plan will be developed with the participation of the taxonomists of all the dischargers and SCCWRP, and should be complete within a year.

In the past, we have merged data from SCCWRP with subsets of discharger data. The necessary taxonomic adjustments involved a labor-intensive process requiring the cooperation of SCCWRP and discharger taxonomists. Unfortunately, the resulting data base quickly became out of date since the original data bases were updated with corrections and changes, but the merged data base was not updated to reflect these changes. Thus, the most efficient approach in the future will involve keeping all the separate data bases taxonomically standardized so that they can be merged at any time. Otherwise, multiple (original and merged), redundant data bases will have to be maintained (probably unsuccessfully).

There can also be problems with a lack of taxonomic standardization over time within the same project. Turnover of taxonomists, along with learning and research advances by taxonomists over time, can lead to uneven taxonomic resolution within the same data base, unless each taxonomic change is applied to the entire data base (present and historical data) when it occurs. Presently, the data from Hyperion, CSDLA, and

SCCWRP are taxonomically consistent over time. The data from Orange County are taxonomically consistent from 1985 to 1989. It is assumed that the data from the State survey and BLM are taxonomically consistent over time, since they both occurred over a shorter period of time (about 4 years).

Over the past thirty years, the taxonomy of benthic invertebrates has advanced considerably. The oldest project, the State survey, may require more work than others to make it compatible with the later data from other sources.

The data from the BLM project will present some special taxonomic problems. The infauna in 69% of the stations (between 30 and 300 meters depth) were identified with a Rapid Identification Procedure (RIP), which involves identifying taxa for a group within a set time limit (about 10 minutes). This necessarily allows for identification of many species only at higher taxonomic levels (genus and above), and produced estimates instead of direct counts of abundant organisms. As part of another study (SAIC 1986), BLM RIP data was merged with other data (generated by the SAIC 1986 study) to examine changes over time. To accomplish this, the original BLM specimens had to be reidentified by taxonomists. There was no other way to match the RIP taxa only identified to higher taxonomic levels without degrading all data to the coarse level of the RIP data.

The data standardization procedure for historical data involves resolving taxon names for organisms that are the same but are named differently in the different data bases. In some cases, more experienced taxonomists can resolve many of these conflicts without examining the original specimens. Another way to avoid examination of specimens is to degrade all the data to a higher taxonomic level where there is agreement. If one needs to examine the original specimens, they are generally available. The specimens for the State and BLM surveys are at the Los Angeles County Museum. The dischargers and SCCWRP store their own specimens.

Finally, it should be noted that if the goal of an analysis is to examine spatial or temporal patterns of some of the most common and easily-identified infaunal organisms, then the data bases from the different sources could be useful immediately. Such taxa will probably be correctly identified in almost all cases. Also, community parameters (e.g., numbers of species, species diversity, total abundance) that do not necessarily require precise taxonomic nomenclature could probably be computed and utilized with the present data bases.

However, this must be done with caution, since varying degrees of taxonomic lumping and splitting among the different data sources may still make number of species and diversity measures incomparable.

1.7 *Comparability of Sediment Parameters*

The main problem with comparability of sediment parameter measurement is the variation in methodology that can cause variation in results. We have not examined in detail the methodology used for all the sediment parameters for all projects. Interviews with scientists from the dischargers and SCCWRP, however, indicate that there are some known incompatibilities.

The dischargers now use EPA protocols to measure chemicals in the sediments, but SCCWRP does not. The detection limits associated with the EPA methods are higher than those associated with the SCCWRP methods. Even within EPA protocols, the laboratories are allowed to improve on these methods, leading to further lack of standardization.

Some examples of methodological differences leading to known incompatibilities include:

1) The sulfides measurements taken in 1985 around the Orange County outfall are not compatible with the later sulfide data. Similarly, early CSDLA sulfide measurements are not compatible with later measurements.

2) PCB's, DDT's and PAH's are measured differently among the dischargers.

3) Results from CSDLA, SCCWRP, and Orange County for sediment metals are not comparable; CSDLA fully digests the sediments, while SCCWRP and Orange County do not, giving higher values for CSDLA with the same sediment.

4) The amount of organic matter has often been measured in different, somewhat incompatible ways. For example, total volatile solids (TVS), percent total organic carbon, BOD, or percent total organic nitrogen have all been measured to quantify this one sediment component.

5) Even less exotic measurements such as sediment size distribution are not always consistent. The sediment size data from Point Loma are not compatible over time due to inconsistencies in the amount of sediment digestion to remove organics. For many surveys, CSDLA only has qualitative measures of sediment size that are not compatible with the quantitative measures.

6) The detection limits from the Orange County data are unique among the dischargers (and SCCWRP) in that they are adjusted for the amount of organic matter in the sediments. This can result in the detection limit for the same chemical to vary widely from sample to sample.

Differing detection limits among the methods may or may not be a problem, depending on what one wants to do with the merged data. If the main interest is to contrast reference locations with outfall locations, then the differences can be large and varying detection limits may not matter. However, if examining variation within or among ambient locations, many if not all of the data values will often be near or below detection limits, and the analyses will be greatly affected.

The examples of inconsistencies given above are no doubt only the tip of the iceberg. Before sediment data from different projects are merged for analyses, the data should be reviewed for compatibility by the respective chemists and geologists. In many cases, the incompatibilities may not be an issue, since the same parameters are often not even measured at the same time by the different projects (see Table 1-2).

1.8 *Conclusions*

It should be feasible to assemble a regional benthic infaunal data set. At a minimum, the SCCWRP reference surveys data and the dischargers' data can be taxonomically standardized and merged for analyses (to be discussed in Sections 2 and 3). It will probably be much more difficult and expensive to taxonomically standardize the State survey and BLM data with the rest of the data. The level of taxonomic standardization required will depend on the indicator variables chosen for monitoring. It will be much more difficult to standardize if indicators based on community composition are to be used, as compared to indicators such as community structure parameters (e.g., species diversity) and abundances of individual species or phyla. In another report to the SMBRP, a detailed taxonomic standardization plan will be proposed.

On the other hand, a regional data set of the physical and chemical sediment parameters will be somewhat limited in breadth and usefulness. The parameters measured within and among the different projects varied widely, and even when the same variables were measured, there are many potential methodological differences that may make much of the data incompatible. The best regional data set that can be assembled will probably contain variables indicating the depth, sediment size, and organic matter found at a location. Even this will be difficult, since sediment size and organic matter were not always measured in the same manner.

SECTION 2 - BENTHIC MACROFAUNAL ASSEMBLAGES FROM REFERENCE AREAS

2.1 *Introduction*

A knowledge of reference, or natural background, conditions is essential for evaluating the impacts of human activities, because such impacts can only be measured by comparison to unaffected conditions. Such comparisons help ensure that presumed impacts are actually the result of human activities and are not in fact natural changes.

While this sounds straightforward in theory, appropriately describing reference conditions can be complex in practice. Firstly, natural spatial and temporal variability can make it difficult to measure reference conditions precisely enough to conclude that observed changes are in fact different from natural conditions. Thus, natural background "noise" can obscure the "signal" generated by an impact. Secondly, natural variability occurs on an extremely wide range of spatial and temporal scales and capturing all of these in a monitoring program is virtually impossible. For example, relatively infrequent and unexpected events such as large storms and El Niños may disturb the typical patterns of seasonal and interannual variability. Third, long-term trends can result in large-scale shifts in background conditions over time, even though the shift from any one year to the next is relatively inconspicuous.

Even when such problems are surmounted, making actual comparisons between possibly impacted and reference conditions can be complicated by time lags and complex ecological interactions. Thus, it can be difficult to decide exactly which aspect of reference conditions to use when determining if an impact has occurred. It is therefore necessary to evaluate several different measures that describe reference conditions. The benthic survey data used in this chapter have been described in Section 1. They come from several independent studies carried out over a long period of time. Some of them are synoptic, spatially broad, one-time studies (e.g., the State Survey) and others are more locally focused ongoing monitoring studies. The only long-term (>2 years) region-wide studies are the SCCWRP Reference Surveys, repeated in 1977, 1985, and 1990.

The synoptic, bight-wide studies essentially provide a snapshot at one point in time. They were designed as "baseline" studies whose goal was to describe existing conditions. Because of the difficulties involved in combining their data with those from more recent studies, they contribute only a limited amount to understanding temporal variability. In addition, because of their strictly descriptive nature, they do not contribute much to an understanding of the processes, mechanisms, and dynamics that structure the benthos.

Existing monitoring programs collect large amounts of data. However, since they focus on separate areas around specific discharges, the sampling designs, and the resultant data, are not well suited to characterizing regional conditions. Thus, the broad regional surveys were simply descriptive and occurred at isolated times, while existing ongoing monitoring programs repeat sampling through time but focus on isolated areas. To achieve comparability, monitoring program design and methodology should be standardized on a regional basis.

As a result, many regionally important questions have not been addressed. These include questions such as: What is the fate of the bulk of the sediment discharged from major outfalls? Are there farfield impacts that occur throughout the bight, rather than just around discharges? How does the character and magnitude of known impacts compare to natural changes in bight-wide natural background conditions? These and other questions can only be partially answered by combining data from existing studies. In-depth understanding of region-wide conditions and processes will ultimately result from studies that are specifically designed to address regional issues, as opposed to site-specific ones.

The following summary and synthesis of existing knowledge makes several key points. First, there are identifiable reference assemblages and sub-assemblages in benthic communities on the mainland shelf. Second, while these assemblages vary naturally over time, there are consistent and readily identifiable differences between these reference assemblages and those in contaminated or disturbed areas. Finally, there is a suite of measures and indicators that can be used to demonstrate these differences.

Because of the shortcomings and gaps in the available data, this summary should be viewed as a first step in the development of a regional picture of reference conditions in the benthos. A more complete understanding will only come from true regional studies that encompass the region as a whole, are repeated through time, and include the functional and dynamic processes that affect benthic communities. Finally, we leave for Section 3 a more detailed discussion of how measures of various aspects of benthic assemblages can be used quantitatively in regional monitoring programs.

2.2 *Factors Affecting Benthic Communities*

The oceanographic environment off Southern California is complex, and several features of this environment are strongly correlated with patterns in benthic communities on the mainland shelf. The mainland shelf itself includes all subtidal areas between the surf zone and approximately 150 m depth. The width of the shelf varies, ranging from 1 km off Palos Verdes to 12 km off Santa Barbara and Imperial Beach. As a result of this relatively narrow shelf, deep water is close to shore and slopes are steep, ranging from 5 to 15 percent.

The mainland shelf is a component of the 4 major nearshore basin systems: the Santa Barbara, Santa Monica, and San Pedro Basins, and the San Diego Trough. The shelf is dissected by 12 large submarine canyons which cut across the shelf and entrain sediment down-slope into the basins. Thus, the shelf can be described as a system of sedimentation cells, separated by submarine canyons, that function independently (Emery 1960, Inman and Chamberlain 1960).

The shelf break, an important feature marked by a rapid increase in depth, occurs between 100 and 150 m depth. Sediments on the inner shelf (<30 m) are usually coarse sand with a large biogenic carbonate fraction. On the outer shelf, sediments are usually silty-clay, with localized intrusions of coarse sand such as off Imperial Beach and Newport Beach and in Santa Monica Bay. Rock outcrops and cobble substrates are occasionally interspersed with the soft sediments on the shelf, providing a habitat for other organisms. In particular, kelp beds may exist on rocky substrate less than 40 m deep. Deeper rocky areas contribute only a small portion of the total area of the mainland shelf (less than 2%).

Current patterns and circulation over the mainland shelf are similarly complex and vary on both seasonal and longer time scales (Jones 1971, Hickey 1979, Lentz and Winant 1979, Hendricks 1977, Jackson 1986, Zedler and Norby 1986). These intricate patterns influence larva dispersal of benthic organisms and the character and distribution of nutrients.

In addition to these bathymetric, sedimentary, and hydrographic features, there are other long-term, low-frequency, interannual events that may be connected with significant changes in benthic communities. These include El Niño events that alter water temperatures, nutrients, and community composition, as well as storm-generated, deep-penetrating, southerly swells that mix and resuspend shelf sediments. Periods of drought and/or heavy rainfall affect mainland runoff and therefore the supply of sediment to the shelf.

These spatial and temporal differences create dissimilar habitats, each with somewhat different kinds of benthic organisms. However, understanding of the direct mechanisms through which such differences affect benthic communities is limited. For example, Spies (1984) has pointed out that little is known about how currents, temperature, primary productivity, and other oceanographic factors affect benthic assemblages.

Much more information exists about animal-sediment relationships, but our knowledge is largely correlative rather than mechanistic or functional. Multivariate analyses indicate that water depth, sediment grain-size, and organic content are the sediment parameters most strongly corresponding to patterns in benthic species composition and abundance (e.g., Smith and Green 1976, Fauchald and Jones 1979a, Thompson *et al.* 1987). However, dynamic studies of the effects of factors such as sedimentation or deposition rate on benthic assemblages in the region have not been conducted.

The influence of biological mechanisms such as competition, predation, commensalism and life history phenomena on benthic communities is also very poorly understood. Studies performed in other areas have shown that activities of benthic animals such as construction of tubes or mucous lined burrows, feeding, respiration, and locomotion can alter sediment quality, transport and surface microcirculation (e.g., Aller and Yingst 1978, Ekman *et al.* 1981). These changes, in turn, affect other organisms and thus community composition and structure (e.g., Stull *et al.* 1986a and 1986b). However, understanding the relationship of such mechanisms to specific features of mainland shelf communities would require an experimental approach such as that used in the deep basins of the region (Smith 1986).

The overall effects of anthropogenic contaminants on benthic assemblages have been well documented (e.g., Smith and Green 1976, Stull *et al.* 1986b, Swartz *et al.* 1986). However, it is not known which contaminants actually cause these effects, or how to separate contaminant effects from those due to organic enrichment or to the accompanying increase in sedimentation rate. Because animals' sensitivities to different contaminants differ, the task of understanding effects species by species and contaminant by contaminant is daunting. Field studies have not succeeded in determining the causes of contaminant effects because all contaminants along outfall gradients co-vary, confounding attempts to identify meaningful correlations. Laboratory exposures to contaminated sediments have shown that significant reductions in growth and gonad production may occur (Anderson *et al.* 1988, Nipper *et al.* 1989, Thompson *et al.* 1989), with attendant potential effects on community composition. Again contaminant concentrations in whole sediments co-vary, making it impossible to determine which contaminants cause the observed effects. Laboratory studies using individual species in tests with single contaminants are just beginning to occur. For example, sulfides at representative field concentrations have been shown to cause reduced growth and reproductive output in the urchin *Lytechinus pictus* (Thompson *et al.* 1991). More of this type of experimentation must be done, but questions about comparability of lab results to field situations remain.

2.3 *Measurements of Community Structure*

Since most wastewater discharges occur at depths greater than 30 m, the following summary focuses on the outer mainland shelf (30 - 150 m). The concept of an assemblage is useful as a basis for organizing large amounts of data about the spatial and temporal distributions of hundreds of benthic species. Defining assemblages as groups of species that consistently appear together in particular habitats, provides a shorthand for description and discussion. More importantly, it provides a conceptual structure for determining whether and how conditions change over space and time. In our discussion below, classification into nominal benthic assemblages reflects general patterns of benthic species composition but does not necessarily imply that distinct zonation occurs. Rather, species composition and abundances change over gradients of space, time, and substrate types.

There are many different ways of defining benthic assemblages, each with its strengths and weaknesses. The following sections present several measures that have proved useful in past research and monitoring. Most of these, such as use of indicator species or species diversity, should be familiar to most readers. However, we also utilize variables derived from ordination analysis as measures of change in benthic assemblage species composition. Ordination analysis is described in Appendix 2A. The following sections provide evidence that: 1) identifiable reference benthic assemblages and sub-assemblages exist on the shelf; 2) while these change somewhat over time, there remain consistent and recognizable differences between reference assemblages and contaminated assemblages; and 3) there are a variety of measures and indicators that are helpful in documenting such differences.

Species Composition and Abundance

Benthic macrofaunal communities in relatively uncontaminated, or reference, areas of the Southern California mainland shelf may be divided into 3 large-scale assemblages:

a. Inner shelf (10 - 30 m)

b. Fine sediment outer shelf (30 - 150 m)

c. Coarse sediment outer shelf (30 - 150 m)

Macrofaunal assemblages on the outer mainland shelf (30 - 150 m) have been studied extensively and in many cases classified into sub-assemblages (Hartman 1955, 1966; Barnard and Hartman 1959; Barnard and Ziesenhenne 1961; Allan Hancock Foundation 1965; Jones 1969; Fauchald and Jones 1979a, 1979b, 1983; Word and Mearns 1979, Thompson *et al.* 1987). The earliest of these studies determined that most of the muddy outer shelf areas are characterized by a large-scale "*Amphiodia*" assemblage (Barnard and Ziesenhenne 1961, Jones 1969), so named because the small, red ophiuroid *Amphiodia urtica* is usually numerically dominant. However, this assemblage also refers to the other numerically subordinate species that typically occur with *Amphiodia* (Table 2-1). As shown in Table 2-1, the assemblage is distinguished by a group of species that appear in both surveys, taken more than 15 years apart. Abundances of these species may shift over time and additional species may become temporarily abundant. Thus, the large-scale, outer shelf assemblage is not static, but changes over time, while maintaining recognizable features.

Even though *A. urtica* and the polychaetes *Paraprionospio pinnata* and *Pectinaria californiensis* are among the most abundant species that have been consistently collected in shelf-wide surveys since the 1950s, Table 2-1 demonstrates that the makeup of the shelf assemblage shifts somewhat over time. The pelecypod *Cyclocardia ventricosa* was considered a co-dominant on the Santa Barbara shelf benthos based on its contribution to the biomass, but its abundance has decreased from about $72/m^2$ in the late 1950s (Jones 1969) to below $35/m^2$ in the late 1970s (Fauchald and Jones 1979a, 1979b; Word and Mearns 1979). The echiuran, *Listriolobus pelodes* was very abundant on the Santa Barbara shelf in 1959-60 and was considered a separate assemblage (Barnard and Hartman 1959). Its abundance has fluctuated considerably all along the mainland shelf since the assemblage was first described (Fauchald 1971, Pilger 1980, Stull *et al.* 1986a). Because of the degree to which its feeding, burrowing, and respiratory activities rework the sediments, *Listriolobus* has the potential to greatly affect the overall structure of the assemblage. Some species, such as the polychaete *Pectinaria californiensis*, may show significant seasonal fluctuations due to apparent adult migrations (Fauchald and Jones 1979a, SCCWRP unpublished). The polychaete *Spiophanes missionensis* has increased in abundance from $23/m^2$ in the 1960s to $161/m^2$ in 1985, to become second in abundance to *A. urtica* (Table 2-1). We will discuss the issue of temporal variability further below.

Table 2-1

ABUNDANCES (MEAN NUMBER/m^2) OF SELECTED SPECIES IN THE STATE (JONES 1969) AND THE BLM YEAR 2 SURVEYS (FAUCHALD AND JONES 1979b). "LUMPED" INDICATES THAT ALL *THARYX* WERE CALLED *THARYX* SPP. "TO SPECIES" MEANS THAT THARYX WAS IDENTIFIED TO SPECIES. A "." INDICATES THAT THE SPECIES WAS NOT FOUND IN THE STATE SURVEY.

SPECIES	STATE SURVEY	BLM SURVEY
Amphiodia urtica	359	621
Spiophanes missionensis (p)	23	278
ostracods	273	to species
Amphicteis scaphobranchiata (p)	94	16
Tharyx spp. (p)	to species	131
Mediomastus californiensis (p)	25	113
Prionospio sp. (p)	93	–
Mysella tumida (pe)	•	101
Tharyx multifilis (p)	62	lumped
Prionospio sp A. (p)	0	81
Axinopsida serricata (pe)	55	35
Axiothella rubrocincta (p)	•	73
Tharyx tesselata (p)	54	lumped
Paraprionospio pinnata (p)	45	51
Rhepoxinius bicuspidatus (a)	48	21
Glycera capitata (p)	18	40
Pectinaria californiensis (p)	45	75
Tellina carpenteri (pe)	•	39
Parvilucina tenuisculpta (pe)	12	16

p = polychaete
pe = pelecypod
a = amphipod

While earlier surveys, summarized in Table 2-1, show that a single, heterogeneous assemblage occurs on the Southern California mainland shelf, more recent studies have provided additional detail. We combined data from the SCCWRP Reference Surveys (1977, 1985, 1990) with 1985 monitoring data around major discharges to develop an overview of spatial and temporal patterns in benthic assemblages. We were restricted to these surveys because of the kinds of data integration problems described in Section 1. Ordination analysis was used to investigate and describe the major patterns in assemblage species composition and abundances.

Figure 2-1 shows the positions in the ordination space of reference and maximally affected (by outfalls) stations at each depth. The maximally affected stations are only maximum for the data used; in actuality these stations are moderately affected by the location outfalls. Table 2-2 shows the species that are associated with each of these sub-assemblages defined by depth and degree of outfall effects. In the different sub-assemblages from reference areas, there is a somewhat different rank order among the most common macrobenthic species, but *Amphiodia* is present in each.

The first two axes of the ordination space are correlated to depth (axis 1) and outfall efforts (axis 2). Other axes or dimensions of the ordination space (not shown) are correlated with time (axis 3) and sediment type independent of outfall effects and depth (axis 4). The fact that these different environmental factors correlate with separate ordination axes indicates that they are somewhat uncorrelated within the sampling area (the positions of the stations on the different ordination axes are independent or uncorrelated; Appendix 2A). This allows us to study community changes correlated with each of these environmental factors separately without confounding effects due to other environmental factors. This fortunate situation is no doubt due to the oceanography and geology of the Southern California shelf. The steep narrow shelf contributes to a strong gradient of community changes associated with depth. Strong longshore bottom currents on the shelf carry most materials from the outfalls along depth contours instead of across depth contours; thus the outfall effects are somewhat independent of depth (Figure 2-1). At any one depth on the shelf, there is a wide range of sediment types, allowing for the study of community changes associated with changes in sediment type.

Presumably, much of the change in assemblages is related to changes in sediment over depth (Fauchald and Jones 1976, Thompson *et al.* 1987). Similar changes occur, within the same depth range, when sediment type changes. For example, macrobenthic assemblages are quite different in areas of high sand content (>90%, as off Imperial Beach and in Santa Monica Bay). In these areas, *A. urtica* occurs in reduced

Table 2-2

MEAN ABUNDANCES (NUMBER/0.1 m^2) OF SPECIES FOUND IN THE SIX GROUPS OUTLINED IN FIGURE 2-1. SPECIES FOUND AMONG THE TEN MOST ABUNDANT IN ANY ONE OF THE SIX GROUPS ARE INCLUDED.

SPECIES	REFERENCE			OUTFALL		
	30 m	**60 m**	**150 m**	**30 m**	**60 m**	**150 m**
Amphiodia urtica	9	124	13	2	0	0
Myriochele sp.	3	40	1	0	0	0
Spiophanes missionensis	25	32	4	18	8	0
Spiophanes berkeleyorum	1	2	38	4	8	0
Amphiodia sp.	3	13	6	0	0	0
Phoronis sp.	2	14	0	0	0	0
Amphideutopus oculatus	9	2	0	0	0	0
Praxillella sp.	7	1	3	0	0	0
Rhepoxynius bicuspidatus	3	6	2	0	0	0
Pectinaria californiensis	2	20	4	12	4	0
Spiophanes fimbriata	1	0	9	0	0	0
Glottidia albida	13	1	0	5	0	0
Heterophoxus oculatus	1	6	2	0	0	0
Sternaspis fossor	3	2	4	0	0	0
Euphilomedes producta	0	2	6	0	0	0
Ampelisca brevisimulata	9	3	0	5	0	0
Maldane sarsi	1	1	4	0	0	0
Paradiopatra parva	0	0	5	0	0	0
Axinopsida serricata	0	7	2	1	5	0
Paraprionospio pinnata	9	2	3	3	4	7
Decamastus gracilis	0	0	0	0	0	2
Cerebratulus sp.	0	0	0	3	2	2
Notomastus tenuis	0	0	0	3	4	3
Macoma yoldiformis	2	0	0	13	2	0
Prionospio (minuspio) lighti	0	1	1	5	11	2
Leptochelia sp.	4	2	0	18	5	0
Spiochaetopterus costarum	3	2	1	11	13	0
Melinna oculata	5	0	0	24	0	0
Aora columbiae	0	0	0	0	20	0
Nereis sp.	2	0	1	9	14	5
Prionospio sp. A	5	4	1	13	22	3
Modiolus sp.	1	0	0	30	0	0
Tellina carpenteri	6	2	2	14	36	2
Capitella capitata	0	0	0	7	37	4
Euphilomedes carcharodonta	6	8	0	66	2	0
Mediomastus sp.	2	1	2	14	57	1
Parvilucina tenuisculpta	8	4	5	124	256	108
Tharyx sp.	2	1	1	155	465	384

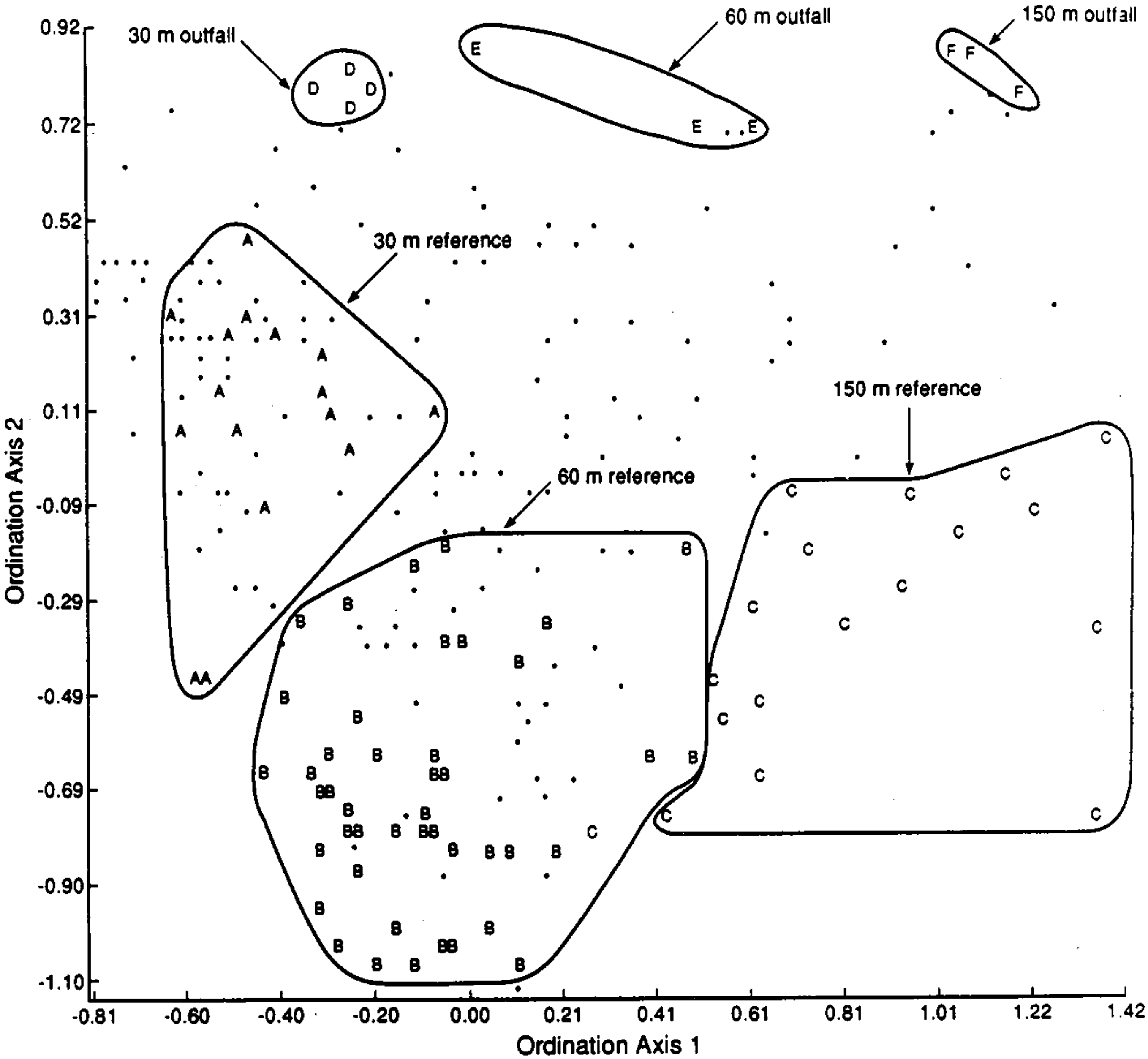


Figure 2-1. Ordination from analysis of SCCWRP reference stations (1977, 1985, 1990) and 1985 Southern California discharger data. A,B,C = SCCWRP reference sites 1977, 1985 and 1990. D,E,F = Stations affected by waste discharge. Reference and most-affected outfall stations at three depths are outlined. Note that depth is correlated with the first ordination axis, and outfall effects are mostly correlated with variation along the second ordination axis. Species found at the outlined stations are shown in Table 2-2.

densities or is absent. Instead, other species such as the gastropod *Micranellum crebricinctum*, the pelecypod *Tellina modesta*, or the ophiuroid *Amphipholis hexacanthus* may be dominant.

Near wastewater outfalls, benthic assemblages also change dramatically (Figure 2-1, Table 2-2) as a result of contamination, organic enrichment, increased rates of sediment deposition (i.e., physical disturbance), or ecological interactions such as predation and competition. In these altered habitats, *A. urtica* does not occur. In moderately affected areas, the pelecypod *Parvilucina tenuisculpta* and the polychaete *Tharyx* sp. become dominant (Table 2-2). In highly affected areas, the polychaete *Capitella capitata* becomes numerically dominant (Thompson 1982b, Word *et al.* 1977). In the sludge field in Santa Monica Bay, prior to sludge discharge termination in 1987, even *Capitella* could not survive well, and another polychaete *Ophyrotrocha* sp. A was dominant (City of Los Angeles 1989). More detailed information on the distribution and abundance of specific species is included in a subsequent section on indicators.

The designation of sites as "reference" sites is based on knowledge of the kinds and abundances of macrobenthos collected. As pointed out above, decades of research on the macrofauna off Southern California has provided a very clear idea of the kinds of macrobenthos that inhabit normal mainland shelf areas, and how they change nearer to outfalls. Regardless of sediment contaminant concentrations (which is not a biological effect per se) or toxicity test information (which may be difficult to interpret), off Southern California sites may be designated as a reference site based solely on the macrobenthic assemblage that inhabits the site. Further, as will be demonstrated, those changing macrobenthic assemblages along outfall gradients can be quantified using ordination methods. That information may then be used to set limits on acceptable deviations in macrobenthic assemblages for use in compliance monitoring.

The regional ordination analysis also provides a context for developing additional insight into the magnitude and character of temporal changes. Benthic communities in 1977 were somewhat different from communities, in the same habitats, in 1985 and 1990. Table 2-3 illustrates some of the changes in species abundance and rank order that occurred in reference areas over this time period. It is not clear exactly what causes such changes, but we can identify three main possibilities. The period from 1977 to 1985 saw not only the largest El Niño event of this century (1982-83), but also some of the largest winter storms in decades (again in 1982-83). The El Niño event had the potential for affecting reproduction, dispersal, and survival, as well as the supply of nutrients to the benthos. Large storms rework and resuspend sediments, thus altering the benthic habitat, sometimes severely. A third possibility is that some of the changes shown

Table 2-3

MEAN ABUNDANCES OF SPECIES (#/GRAB) FROM THE THREE SCCWRP REFERENCE SURVEYS. ONLY STATIONS THAT WERE SAMPLED IN ALL THREE SURVEYS AT 60 METERS ARE INCLUDED. SPECIES AMONG THE TOP TEN MOST ABUNDANT IN ANY ONE SURVEY ARE INCLUDED.

SPECIES	YEAR		
	1977	1985	1990
Amphiodia urtica	139	99	79
Myriochele sp.	0	54	261
Spiophanes missionensis	31	26	30
Amphiodia sp.	0	15	56
Pectinaria californiensis	20	3	19
Phoronis sp.	35	1	4
Prionospio sp. A	5	3	11
Heterophoxus oculatus	6	5	7
Rhepoxynius bicuspidatus	8	2	7
Axinopsida serricata	9	5	2
Euphilomedes carcharodonta	6	1	7
Paramage scutata	0	12	0
Parvilucina tenuisculpta	6	2	4
Sternaspis fossor	1	7	3
Tellina modesta	0	10	0
Mysella sp.	7	3	1
Euclymeninae sp. A	0	2	7
Tellina carpenteri	1	5	3

in Table 2-3 are an artifact of changes in taxonomic standards. Section 1 documented the potential severity of this problem and, to date, resources have not been available to thoroughly resolve all taxonomic uncertainties. It is most likely that temporal changes do in fact exist and that their actual magnitude is somewhat exaggerated by the lack of complete taxonomic standardization over time.

Indicator Species

Some species are so common that abundances are potentially useful in identifying reference sites on the Southern California mainland shelf. The brittlestar *Amphiodia urtica* and the clam *Parvilucina tenuisculpta* are two such species. We discuss these two as an example of how indicator species can help characterize reference conditions and identify changes due to human activities.

The relative abundances of *A. urtica* and *P. tenuisculpta* indicate the degree of contamination and/or disturbance due to wastewater outfalls at shelf sites. High abundances of *A. urtica* and low abundances of *P. tenuisculpta* indicate normal muddy shelf sites. As sites become moderately impacted, their relative abundances tend to reverse, with *P. tenuisculpta* becoming more abundant than *A. urtica* (Table 2-4). The following sections summarize distributional and natural history information relevant to their use as indicator species.

Amphiodia urtica has been called the most common and abundant invertebrate on the Southern California mainland shelf (Barnard and Ziesenhenne 1961). Their overall distribution is from Shumagin Island, Alaska in the north to Cedros Island and Tangola Bay, Baja California, in the south. They range in depth from 16 to 325 m. Off Southern California they occur all along the mainland shelf and on offshore insular shelves, ridges and banks, but they are consistently more abundant on the mainland shelf.

In several major shelf surveys, average abundances of this species varied widely, ranging from 78 to 1342/m^2 (Table 2-4). The highest density measured was 3630/m^2 (SCCWRP Station R56-60, 1977). The estimates in Table 2-4 illustrate the range of *A. urtica* densities typical of the mainland shelf assemblage. The variability in these densities in part represents *A. urtica*'s response to conditions at different depths and sediment types (Figure 2-2). It also reflects temporal variability and sampling error.

Table 2-4

ABUNDANCES OF INDICATOR SPECIES FROM VARIOUS SPECIES
(#1-1/10 m^2 VVG; #2-1/16 m^2 Box; #3-1/4 m^2 Campbell)

	3	2							1								
Survey Area Date Depths n =	AHF all 1956-61 13-253 176	BLMYR1 Hunt. Bch.	BLMYR1 Pt. Dume	BLMYR2 Coal Oil Pt.		BLMYR2 San Ped. Sh.		60 m Surv. all 1977	SCCWRP Replicate Study OC Shelf		SCCWRP Reference Survey					
		1975-76	1975-76	W1977S		W1977S		60 m	1980	1981	all 1985			all 1990		
		28-109 18	40-193 8	68 8 reps 8		32 8 reps 8		26	60 10 reps 10	44	30 12	60 13	150 13	30 6	60 7	150 7
<u>*Amphiodia*</u> *Mean/m^2* *Var. 2*	359 (f=87)	683 s(547.6)	560 s(452.8)	652 s(108.8)	1000 (96)	152 (45.3)	286 (93.9)	1342 s(981.9)	186 s(60.2)	82 (14.6)	83 s(70.5)	1023 (68.1)	146 (156)	78 ci(86)	1158 (688)	287 (242)
<u>*Parvilucina*</u> *Mean/m^2* *Var. 2*	12 r=(4-68)	452 s(150.2)	48 s(39.2)	22 –	12 –	10 –	2 –	90 (100.3)	25 c(54)	59 (50)	99 s(148.6)	45 (54.7)	12 (15.7)	38 s(44.4)	41 (54.3)	99 (178.8)

W = winter, S = summer

For variation estimes:

r = range
s = standard deviation
c = 95% con. int.
f = frequency of occurrence

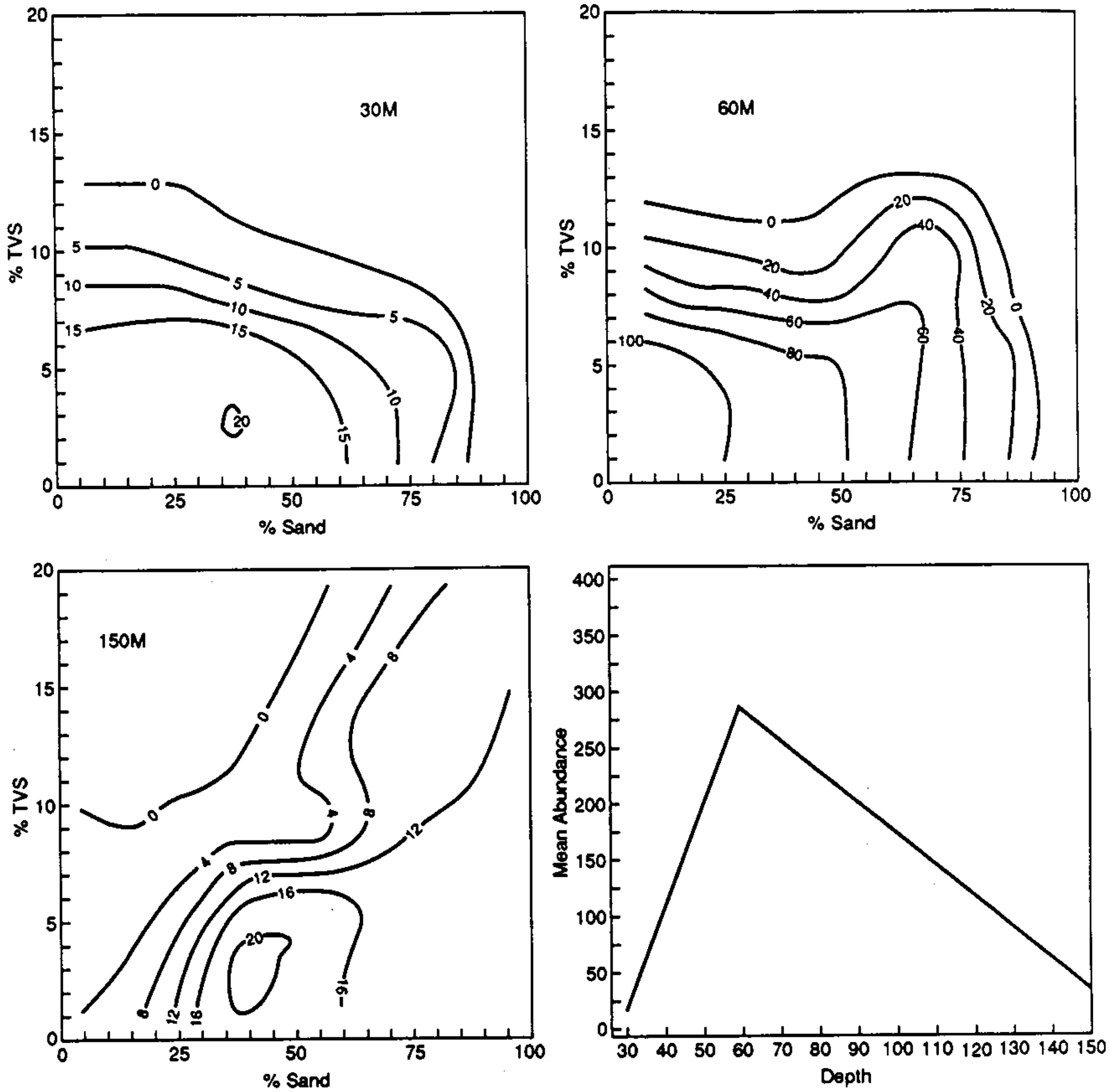


Figure 2-2. Observed relationship between the abundances of *Amphiodia urtica* (represented by the contours) and % TVS (total volatile solids), % sand, and depth. Contours are based on smoothed data from SCCWRP reference stations (1977, 1985, 1990) and 1985 Southern California discharger data. Stations adjacent to the high-energy outfalls (Orange County and Hyperion) were excluded because only responses in reference areas were of interest. The plot of *A. urtica* vs. depth is based on the mean of the top five abundances at the reference stations (to avoid confounding depth and outfall effect). *A. urtica* abundance in units of abundance/0.1 m^2. Note *A. urtica* seems to prefer lower % sand and % TVS at 30 and 60 meters, but can be found at higher % sand and % TVS levels at 150 meters.

Table 2-1 shows that *A. urtica* abundances can vary considerably over time, but the SCCWRP Reference Surveys suggest that this variability is somewhat reduced when samples from the same depth are compared (Figure 2-3). In this instance, abundance varied by a factor of two between surveys at the same depth. This is similar in magnitude to the changes observed from summer to winter in replicated samples from two sites on the shelf (Table 2-4, Coal Oil Point and San Pedro Shoals).

Amphiodia urtica burrows into the top 2 - 5 cm of sediment, completely covering their central disk with sediment. They may deposit feed from surface or sub-surface sediments or suspension feed from the water column. In the laboratory, *A. urtica* can be induced to switch from suspension to deposit feeding by varying the water current. If the current is increased, the number of arms dedicated to suspension feeding increases.

In spite of temporal shifts in assemblage structure, the data presented above show that it is possible to clearly distinguish relatively undisturbed reference areas from those affected by wastewater outfalls. This is because the changes caused by natural disturbances such as El Niños and storms do not mimic those caused by outfalls. Thus, anthropogenic changes are readily distinguishable even against the moving background of natural changes in reference assemblages. As the current speed is decreased and then stopped, the arms drop to the sediment surface and begin to deposit feed. This species is therefore capable of feeding facultatively in different modes and from different sediment strata. Its feeding mode *in situ* probably depends on such environmental variables as currents, suspended particles, and food availability (Thompson 1982a).

When deposit feeding, only the tips of their arms can usually be seen above the sediment surface. Typically only three or four arms are visible, while the other arms forage below the surface. One of the surface arms is usually dedicated to sediment transport from sub-surface to surface. Most of the transported material is presumed to be feces as this species has no anus. The result is a small mound of material around each of several arms on the sediment surface. They do not seem to mind the presence of other *A. urtica* very near by and often touch and intertwine arms with each other.

The diet of *A. urtica* from seven sites is summarized in Table 2-5. It was predominantly detrital aggregates and the proportions of food items that were similar in specimens from different areas on the mainland shelf and from offshore shelves and banks. The mineral particle size frequency distribution from the stomachs of 5 specimens showed that 87% of the particles were smaller than 10 μm, and were mostly clay minerals. By volume, however, particles up to 100 μm (fine sand) contributed most to gut contents. Mineral particles

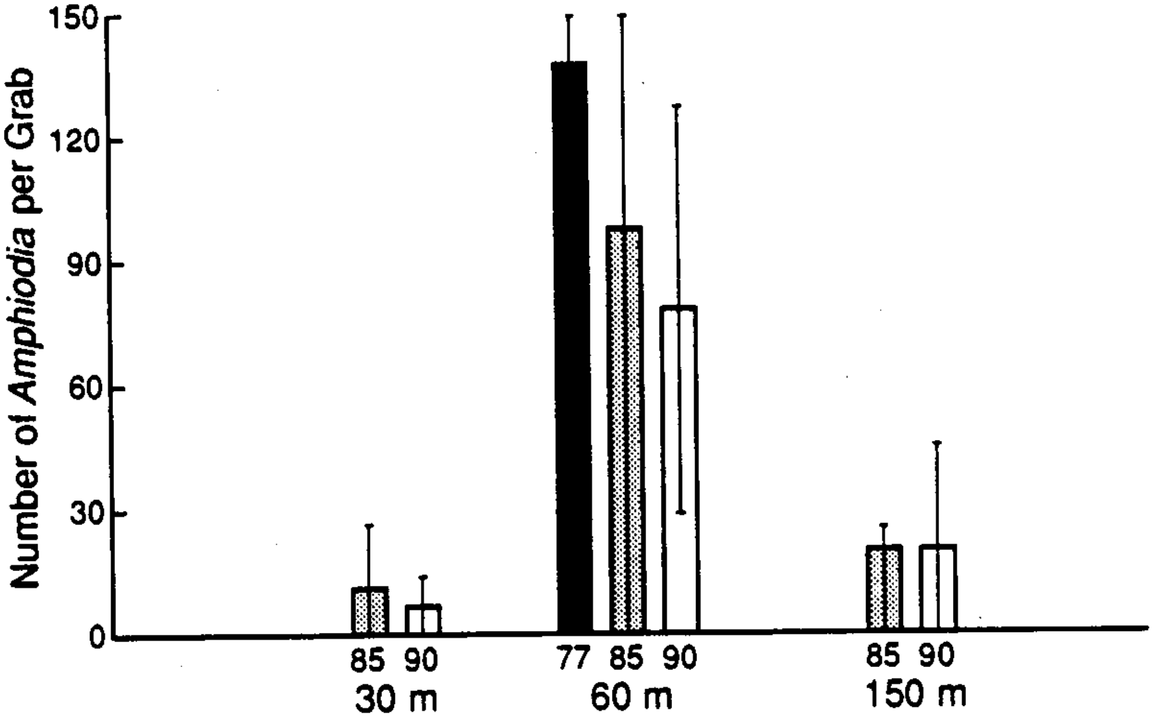


Figure 2-3. Mean abundance and 95% confidence limits of the mean for *Amphiodia urtica* from the three SCCWRP reference surveys in 1977, 1985 and 1990. Each depth shown separately. Based on n=6 at 30 and 60 meters, and n=7 at 150 meters.

Table 2-5

DIETS AND FEEDING STRATA OF SEVERAL KEY MAINLAND SHELF SPECIES (AFTER THOMPSON, 1982A).

Species	n =	Percentage of gut contents volume					
		Detrital aggregates*	Single mineral particles (> 10 μm)	Partic. organic material	Animal remains	Foraminifera	Feeding Stratum
A. urtica	22	93	7	0	0	T	F,S,Su
C. ventricosa	6	71	19	0	0	T	F,S
S. missionensis	12	53	48	0	0	T	S
L. pelodes	8	38	58	0	0	5	S
A. hexacanthus	3	37	63	0	0	0	F,S
P. tenuisculpta	14	100	0	0	0	0	F,S

* Aggregations of small mineral particles (< 10 μm), fine particulate organic material, and microbes.

F = filter feeder, S = sediment surface feeder, Su = subsurface feeder, T = trace in the stomachs were similar to *in situ* sediments, suggesting that this species does not feed selectively (Thompson 1982a).

There are no detailed studies of the reproductive cycle or life history of *A. urtica*. However, such a study is currently being conducted at SCCWRP. Size frequencies on the mainland shelf usually are unimodal, and juveniles may be collected at various times during the year. A study in Puget Sound, Washington (Lie 1968), in contrast, found bimodal size frequency distributions and spring recruitment. This same study estimated life expectancy at five years and growth rates at less than 2 mg (dry wt) per year.

The small bivalve *Parvilucina tenuisculpta* ranges from Cook Inlet, Alaska in the north to Ensenada, Mexico in the south (Jones and Thompson 1984). It is a member of nearly all benthic assemblages off Southern California, except in the deep basins. They range in depth from 13 to 702 m, but on the Southern California mainland shelf they live between 13 and 322 m, and are most abundant at depths between 100 and 150 m (Jones and Thompson 1984).

Average densities of *Parvilucina tenuisculpta* from reference areas on the mainland shelf are shown in Table 2-4. As with *A. urtica*, the variation in these estimates reflects *P. tenuisculpta*'s response to conditions at different depths and sediment types, as well as temporal variability and sampling error.

Parvilucina tenuisculpta prefers sediment with elevated organic content (Figure 2-4). It may become more abundant in two very different habitats: in areas of high upwelling, such as off Pt. Conception (up to $1880/m^2$), and in transitional areas moderately affected by wastewater outfalls (up to $3568/m^2$). In such transitional areas where densities are high, patches or aggregations of *P. tenuisculpta* are more densely packed (SCCWRP unpublished).

Because its abundance is generally lower than *A. urtica*'s, changes in *P. tenuisculpta*'s abundance over time are less dramatic. Density changes from summer to winter in replicated samples from two sites on the shelf (Table 2-4, Coal Oil Point and San Pedro Shelf) were not large (Jones and Thompson 1984). In addition, data from the SCCWRP Reference Surveys show that density changes, at the same depth, between 1977 and 1985 and between 1985 and 1990 are relatively small (Figure 2-5). Overall, average sampled abundances of this species on the shelf have increased from $12/m^2$ (State Survey) around 1960 to $186.5/m^2$ (BLM year 2

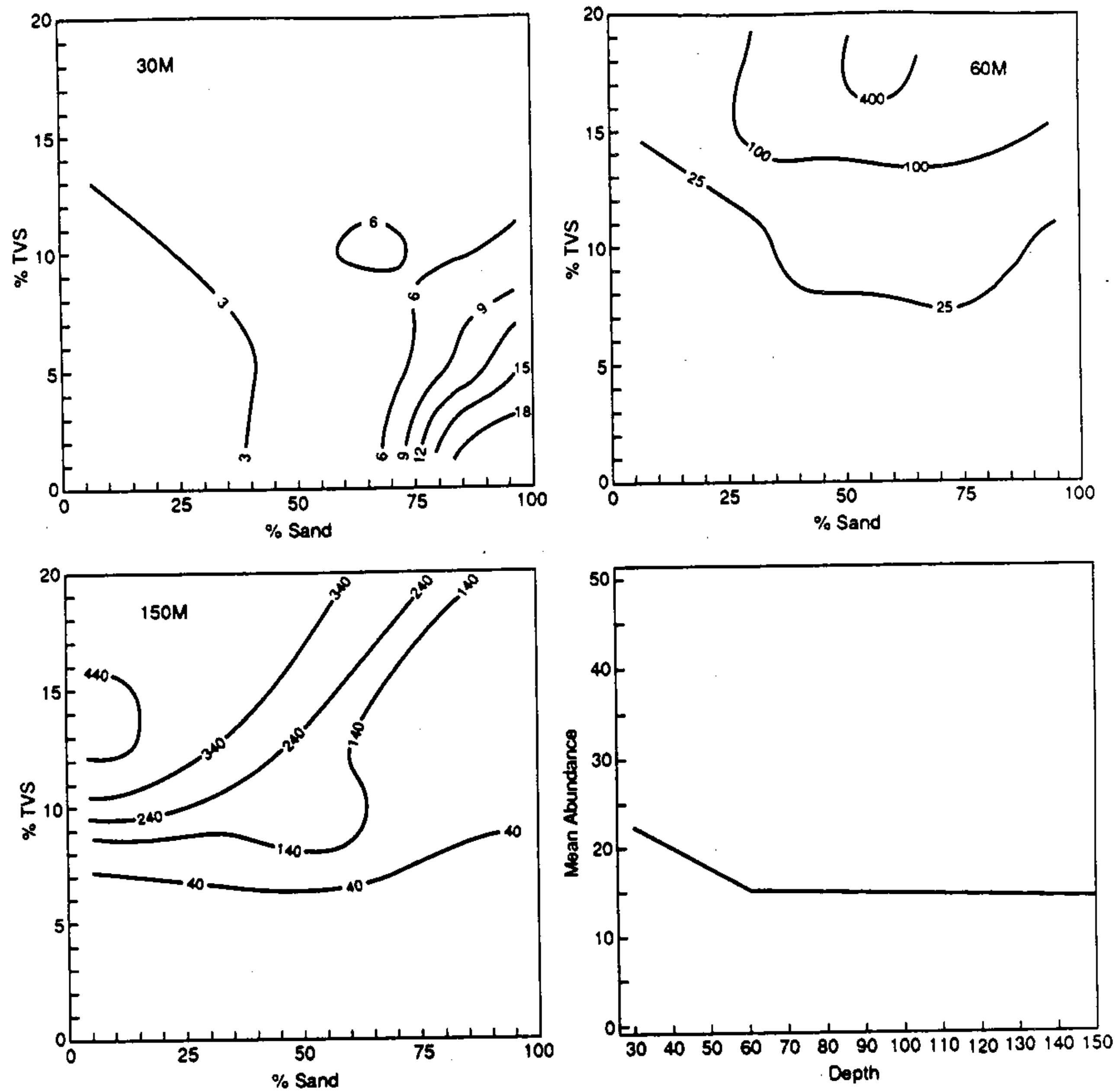


Figure 2-4. Observed relationship between the abundances of *Parvilucina tenuisculpta* (represented by the contours) and % TVS (total volatile solids), % sand, and depth. Contours are based on smoothed data from SCCWRP reference stations (1977, 1985, 1990) and 1985 Southern California discharger data. Stations adjacent to the high-energy outfalls (Orange County and Hyperion) were excluded because the availability of TVS to organisms is not well measured by the % TVS in the sediment in such environments (SCCWRP 1987). The plot of *P. tenuisculpta* vs. depth is based on the mean of the top five abundances at the reference stations (to avoid confounding depth and outfall effects). Abundances in units of abundance/0.1 m^2. Note that *P. tenuisculpta* seems to prefer higher % TVS at 60 and 150 meters.

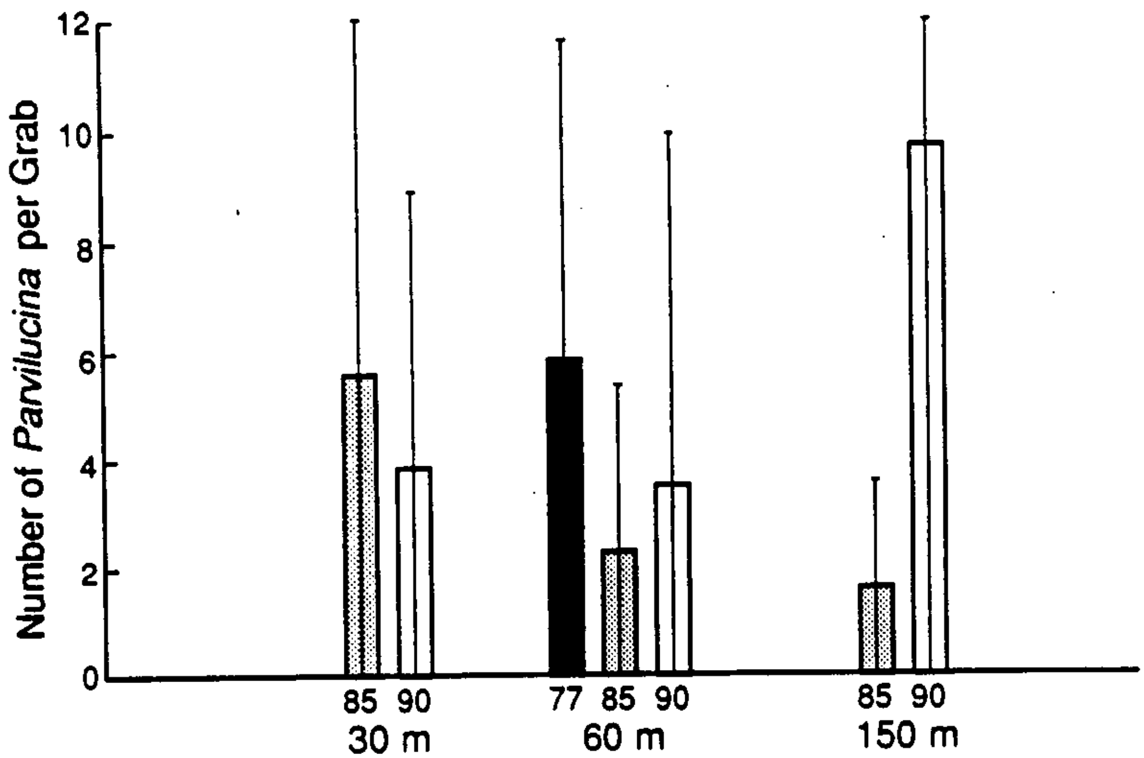


Figure 2-5. Mean abundance and 95% confidence limits of the mean for *Parvilucina tenuisculpta* from the three SCCWRP reference surveys in 1977, 1985, and 1990. Each depth shown separately. Based on n=6 at 30 and 60 meters, and n=7 at 150 meters.

survey) in 1977, demonstrating the potential for natural long term population fluctuations (Jones and Thompson 1984).

Parvilucina tenuisculpta possesses a very short siphon through which it feeds by filtering water and entrained particulates. Most specimens inhabit the sediment surface, but have been collected to depths of six cm below the sediment surface (Balcom 1980). They ingest mainly detrital aggregates and mineral particles up to 100 um in diameter (Table 2-5; Thompson 1982a).

Near wastewater outfalls, this species changes its biology. Not only does it become much more abundant, but it grows much larger than in reference areas due to becoming facultatively infected with an endosymbiotic bacteria that oxidizes sulfide to obtain additional metabolic energy (Felbeck *et al.* 1981).

In combination with data on the abundance of *A. urtica*, changes in abundances and biology of *P. tenuisculpta* could be used to help identify samples from areas beginning to show effects from waste discharge.

In addition to the abundances of individual indicator species and measures of assemblage species composition based on ordination analysis, certain other variables may potentially also be used to describe reference conditions. Four such measures are: number of individuals, number of species, species diversity, and biomass.

These variables, calculated from various surveys on the Southern California mainland shelf are summarized in Table 2-6. Variation in these estimates reflect differences in depth, substrate types, and sampling times, as well as additional unexplained spatial variability and sampling error. In addition, measures of the number of species and diversity are dependent on the size of the sample. Thus, comparisons among surveys that used different sized sampling devices are not valid for these two parameters. In reference areas off Southern California, these parameters are generally highest at the 30 m sites and decrease over shelf depths (Fauchald and Jones 1979a and 1979b; Thompson *et al.* 1987). Species diversity (H') values from the outer mainland shelf are among the highest in the region (mean = 3.2, range = 2.8 - 3.9). Biomass is extremely variable (up to 76%) due to chance collections of large motile invertebrates such as holothuroids, echinoids, and echiurans (Thompson, 1982b).

Table 2-6

VALUES OF NUMBER OF SPECIES, INDIVIDUALS, BIOMASS AND SPECIES DIVERSITY (H') FROM VARIOUS SAMPLES
(sample sizes same as Table 2-3)

	1	2						3								
Survey Area	**AHF**	**BLMYR1** Hunt. Bch.	**BLMYR1** Pt. Dume	**BLMYR2** Coal Oil Pt.		**BLMYR2** San Ped. Sh.		60 m Surv. all 1977	SCCWRP Replicate Study OC Shelf		SCCWRP Reference Survey					
Date Depths (m)	all 1956-61	1975-76	1975-76	W1977S		W1977S		60 m	1980	1981	all 1985			all 1990		
	13-253	28-109	40-193	68		32		26	60	44	30	60	150	30	60	150
n =	176	18	8	4 reps 8		4 reps 4			10 reps 10		12	13	13	6	7	7
		var=range		var = SD				var = SD	var = cv		var = 95% CI					
Species/grab	92-117	49 (36-65)	66 (54-70)	59 (7.1)	68 (1.3)	87 (5.7)	105 (8.4)	72 (3.7)	62.9 cv(19.1)	64.4 (11.6)	89.2 ci(10.8)	67.7 (12.5)	41.9 (8.1)	76.8 (24.3)	83.4 (20.6)	61.6 (11.8)
Individ./m^2	3093 -	4208 (1728-5760)	3776 (1920-5296)	2880 (381)	4224 (400)	4373 (966)	5096 (930)	4220 (340)	4096 (27.1)	3052 (14.1)	3238 (581)	3484 (757)	1548 (362)	2454 (422)	6254 (5735)	2672 (1141)
Biomass/m^2	up to 1100 -	- -	- -	108 (10.9)	170 (46.7)	205 (14.6)	141 (29.3)	73.8 (5.5)	96 (41.4)	88 (75.7)	157 (61)	114 (44)	81 (39)	79 (31)	152 (154)	56 (56)
H' (sample)	-	2.91 (1.84-3.67)	3.49 (3.44-3.57)	2.77 r(2.33-3.14)		3.72 (3.48-3.91)		3.12	-	-	3.89 (0.13)	3.12 (0.36)	2.75 (0.44)			

W = winter, S = summer

All variation estimes are standard deviation unless otherwise noted;

CI = 95% conf. interval
r = range
cv = coefficient of variation

Over time, these parameters appear to be less variable than the abundances of individual species. There was no significant difference in any of these parameters at each depth among the three SCCWRP reference surveys (Figure 2-6). These parameters seem to vary with substrate type, which appears to interact with depth (Figure 2-7). For example, at 60 m depth, number of species per grab is highest in sandy sediments with lower percent TVS. At 30 and 150 m, very different patterns are seen.

Number of species, number of individuals, species diversity, and biomass also change in sediments affected by wastewater outfalls. In transitional areas, species diversity, total abundance, and biomass actually increase in a well-documented response to organic enrichment. These values then decrease as contamination and enrichment increase nearer the outfall (Pearson and Rosenberg 1978, Swartz *et al*. 1985, see also Figure 3-6 in Section 3). Depending on the degree of severity of contamination and enrichment, these parameters can drop to levels much lower than those characteristic of reference areas.

The relatively low variability in these parameters, as compared to abundances of individual species, makes them attractive candidates for monitoring and compliance programs. However, they can be ambiguous and difficult to interpret. Such concerns are discussed at greater length in Section 3.

2.4 *Measures of Community Function*

Trophic Relationships

Information about feeding behavior, burrowing, and locomotion of benthic species could potentially be used to characterize reference conditions. Feeding behavior and trophic relationships reflect important ecological processes that often underly more visible evidence of assemblage structure. Additionally, trophic relationships are an important way in which organisms may accumulate contaminants. Thus, changes in these behaviors and relationships can provide insight into the ways in which human activities cause impacts in the benthos. Finally, burrowing is important for its potential effects on sediment resuspension and transport, and on bioturbation. These factors can influence the availability of contaminants and the suitability of the sediment as a habitat for other organisms. Despite the importance of such information, there is relatively little knowledge that is directly helpful in understanding and documenting changes between reference and impacted areas.

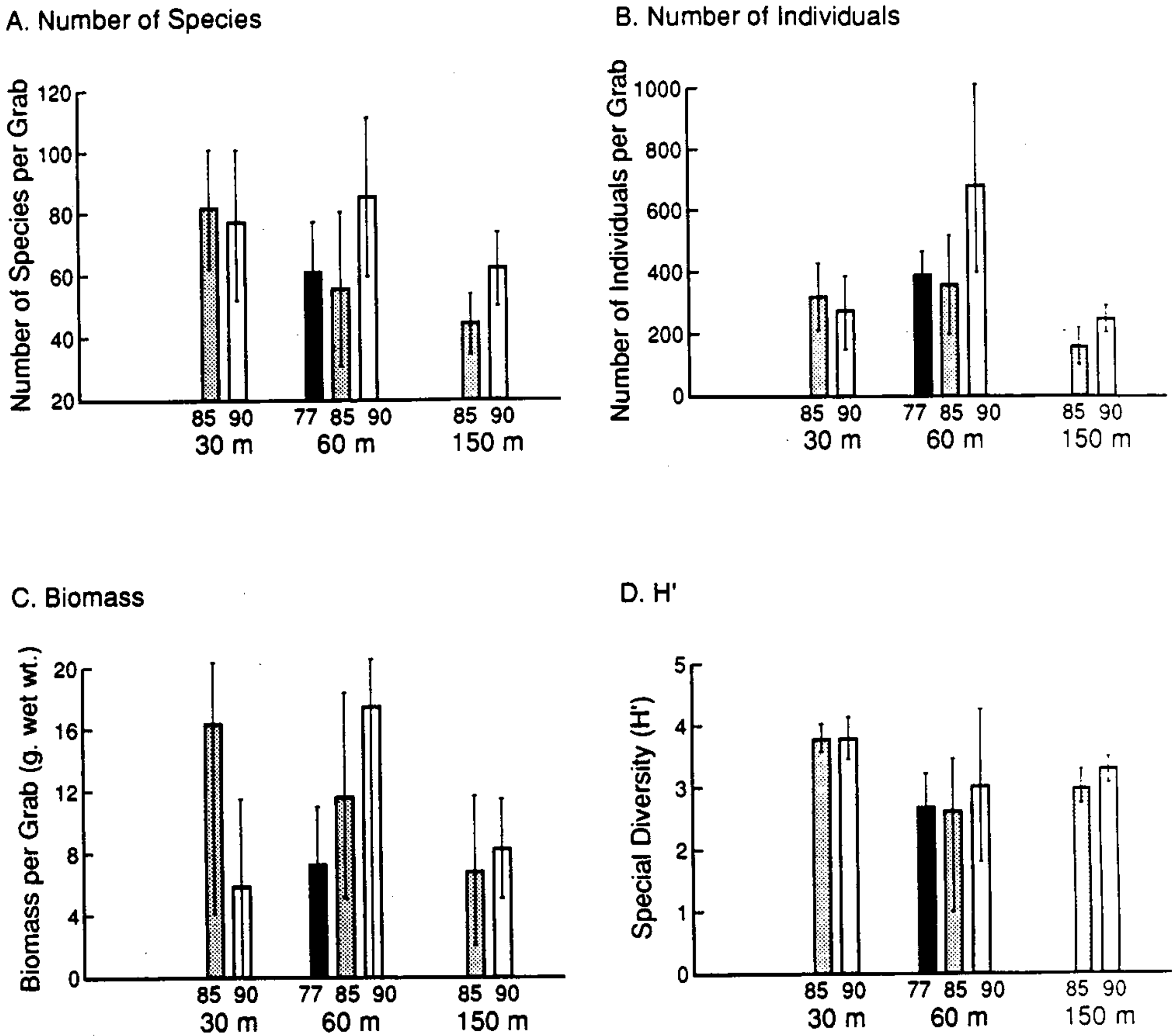


Figure 2-6. Mean value and 95% confidence limits of the mean for four community structure parameters. Data from the three SCCWRP reference surveys in 1977, 1985, and 1990. Each depth shown separately. Based on n=6 at 40 and 60 meters, and n=7 at 150 meters.

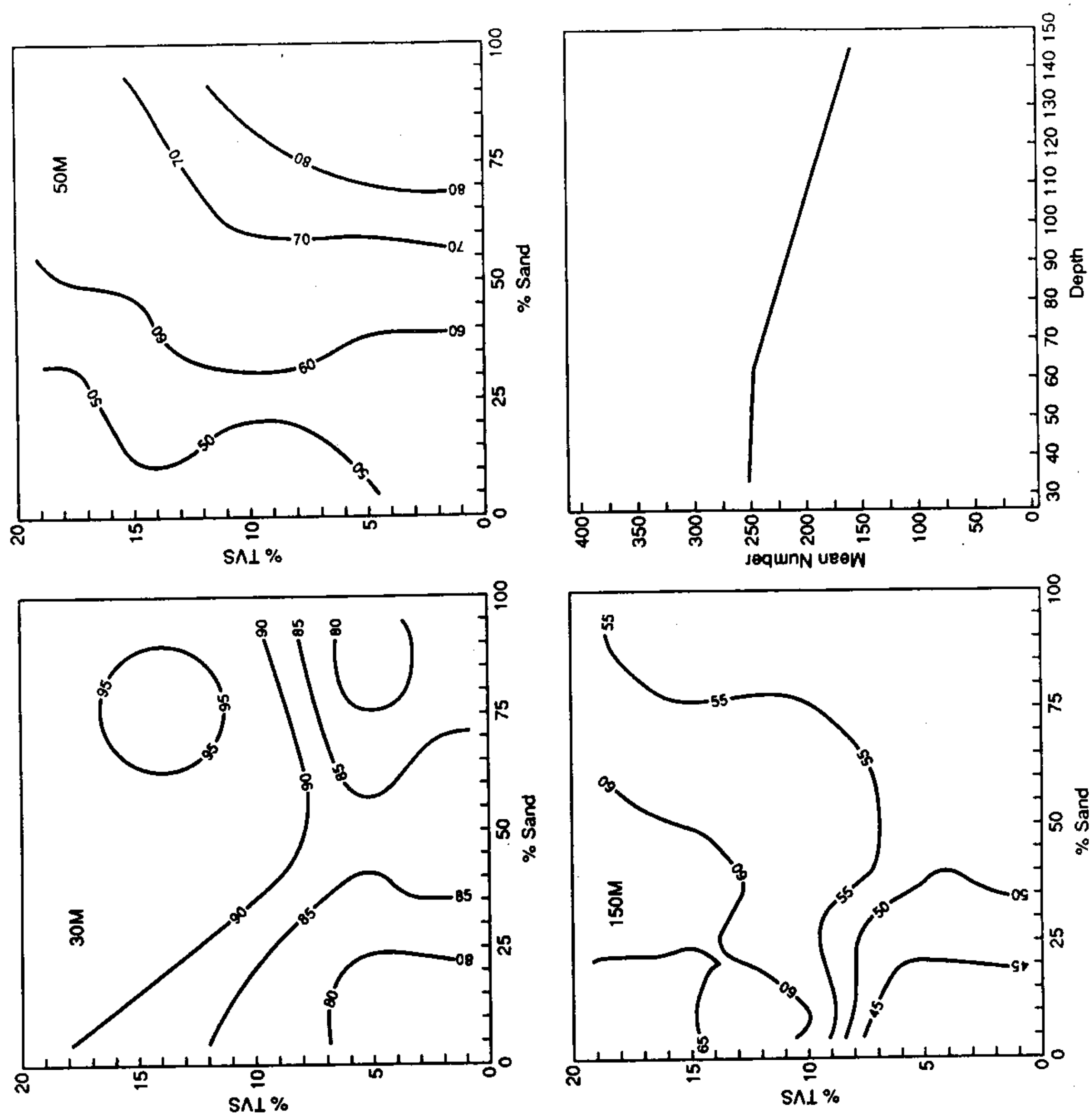


Figure 2-7. Observed relationship between the number of species (represented by the contours) and % TVS (total volatile solids), % sand, and depth. Contours are based on smoothed data from SCCWRP reference stations (1977, 1985, 1990) and 1985 Southern California discharger data. Stations adjacent to the high-energy outfalls (Orange County and Hyperion) were excluded because the availability of TVS to organisms is not well measured by the % TVS in the sediment in such environments (SCCWRP 1987). The plot of number of species vs. depth is based on the mean of the top five values at the reference stations (to avoid confounding depth and outfall effects). Note that number of species shows completely different relationships to sand and TVS at the different depths.

Many schemes have been developed for categorizing benthic organism into similar feeding and mobility types (Fauchald and Jumars 1979, Sanders *et al*. 1962, Thompson 1982a, Word 1978). While such exercises have heuristic value, there is considerable uncertainty associated with all these categorizations. The placement of a species into a category is usually subjective, and is based on the morphology of its feeding apparatus, its gut contents, or even an educated guess by the researcher. Categories are usually inflexible and do not allow for the fact that nearly all benthic species can switch their feeding modes or diets depending on the available source of food (Taghon *et al*. 1980, Thompson 1982a).

Each study has used a different set of species and habitats, and used a different set of categories (e.g., surface deposit feeder vs. burrowing detrital feeder, or suspension feeder vs. filter feeder). For example, Fauchald and Jones (1976) calculated the total number of <u>species</u> in each of 5 feeding/mobility categories (Table 2-7b). In 1977, they calculated the total number of <u>individuals</u> in a different set of 9 categories (Table 2-7c).

Few studies based on observation or experimentation have been conducted on Southern California species. Because of the behavioral flexibility of many benthic species, an understanding of feeding, diets, and sediment interactions requires considerable observation and experimentation at the individual species level. Again because of behavioral flexibility, such studies should document activities under varying environmental conditions. With these limitations in mind, we present a brief summary of the feeding and mobility types of Southern California benthos.

Diets and feeding modes of some of the most abundant Southern California mainland shelf species are summarized in Table 2-5. The most common and abundant shelf species ingest mostly detrital aggregates. Both *Amphioda urtica* and *Cyclocardia ventricosa* may feed either as surface detrital feeders or as suspension feeders. Only a few species (such as the polychaetes *Glycera capitata* and *Chloeia pinnata*, and the urchin *Lytechinus pictus*) ingest large quantities of particulate organic material or animal remains. Most mainland shelf species are filter feeders (about 42 %; Table 2-7b), but nearly as many species (31%) are surface detrital feeders. The mainland shelf assemblage obtains most of its food from detrital aggregates ingested at the sediment surface (Table 2-7a). Only about 18% of the assemblage are suspension feeders. About half of the species ingest Foraminifera, but they do not contribute large volumes to diets except in *Listriolobus pelodes* and *C. ventricosa* (Thompson 1982a).

Table 2-7a. PERCENTAGES OF MAINLAND SHELF ASSEMBLAGE (ABUNDANCE X BIOMASS) THAT UTILIZE EACH FOOD-STRATUM CATEGORY, BASED ON GUT-CONTENTS ANALYSES, BLMYR2 SAMPLES.
Sus = suspension feeder; Sur = surface feeder, Sub = subsurface feeder (Thompson 1982a)

	Food-Stratum Category														
Food	Detrital Aggregates			Single Mineral Particles			Particulate Organic Matter			Animal Remains			Foraminiferans		
Feeding Stratum	Sus	Sur	Sub	Sus	Sur	Sub	Sus	Sur	Sub	Sus	Sur	Sub	Sus	Sur	Sub
Percent	18	45	2	3	27	2	+	1	0	0	1	0	1	1	0

Table 2-7b. PERCENTAGES OF *SPECIES* IN MAINLAND SHELF ASSEMBLAGES IN EACH CATAGORY.
(BLMYR1; Fauchald and Jones, 1983); average of 31 sites from three shelf areas

Filter Feeders	Surface Detrital Feeders	Sub-surface Detrital Feeders	Predators	Scavengers
42	31	6	14	8

Table 2-7c. PERCENTAGES OF *INDIVIDUALS* IN MAINLAND SHELF ASSEMBLAGES IN EACH CATEGORY.
(BLMYR2; Fauchald and Jones, 1979b); four replicases from two sites, winter and summer pooled.

Filter Feeders		Surface Detrital Feeders		Sub-surface Detrital Feeders		Herbivores, Grazers, Scavengers		Carnivores
motile	sessile	motile	sessile	motile	sessile	motile	sessile	motile
4	9	48	12	7	7	<1	1	11

Mainland shelf species may ingest mineral particle sizes up to 350 µm. By size, clay minerals are always the most abundant particles ingested, however, by volume, average particle-sizes in each species are usually different. Differences in diets, particle ranges utilized, and feeding stratum (water, sediment surface, subsurface), may facilitate the coexistence of the numerous deposit feeding species in these benthic assemblages.

On the Southern California mainland shelf, about equal proportions (35 - 38%) of the benthic species are burrowing or tube dwelling surface-motile, 12% are sessile, 16% are surface-motile. How these life modes influence sediment processes on the shelf has not been studied; however, in other areas, the presence of polychaete tubes may increase sedimentation rates (e.g., Eckman *et al.* 1981). Additionally the feeding, motility, and life habits of benthic species may be different near outfalls than in reference areas; however, no empirical studies of these changes have been conducted.

Life Histories and Ecological Interactions

The life histories of individual species and ecological interactions among species (predation, competition) may also contribute to the formation of the assemblages observed on the mainland shelf. In theory, an understanding of these processes may facilitate the prediction and explanation of impacts due to human activities. such as wastewater discharge. However, very few studies of ecological interactions or life histories have been conducted.

The few existing studies of shelf species on life histories have furnished some information on reproductive seasonality (no strong patterns with season), on larval settlement, and on commensalism (Jones 1963, Baker 1975, Pilger 1977, Wicksten 1980, Anderson *et al.* 1985, Fauchald and Jones 1989a, SCCWRP unpublished). However, little research is available that ties such observations directly to the environmental changes thought to be responsible for human-induced impacts on assemblage structure.

2.5 *Conclusions*

Reference macrobenthic assemblages can be defined for the Southern California mainland shelf. These assemblages change in response to changes in environmental factors such as depth, sediment grain size, and organic content. They may change over time in response to events such as El Niños and major storms.

However, there is additional unexplained variability in space and time that can sometimes be very large. In spite of this variability, it is possible to identify consistent differences between reference assemblages and areas affected by wastewater outfalls. This is due in part to the fact that the changes induced by wastewater outfalls do not mimic natural changes in benthic assemblages. Thus, it is possible to detect the signal of human impacts, even against the noise of natural variability.

We have reviewed the use of several different methods for characterizing reference assemblages. In particular, ordination analysis proved useful for describing changes in assemblage species composition over space and time. Other measures such as abundances of indicator species, and community structure measures such as numbers of species, numbers of individuals, species diversity, and biomass can also potentially provide insight into assemblage changes due to wastewater outfalls.

Knowledge about assemblage composition, structure, character and magnitude of variation furnishes a basis for assessing the effects of contaminants on the benthos at a site of known depth, grain-size, and organic content. It also provides a framework for the development of more effective monitoring tools and approaches. This topic is addressed in Section 3.

SECTION 3 - RECOMMENDATIONS FOR USING REFERENCE SITE INFORMATION IN REGIONAL MONITORING PROGRAMS

3.1 *Introduction*

Two goals in the design of a regional monitoring program of this type include the definition of reference conditions, and the development of analytical techniques by which we can measure deviations from these reference conditions. In Section 2, we described the benthic communities found in Southern California reference areas. In Section 3, we address the latter goal by presenting analytical approaches for utilizing this information in a regional monitoring context. More specifically, we will delimit sampling stations exhibiting reference conditions, and propose methods to measure deviations from these conditions due to anthropogenic activities. Particular attention is paid to potential changes in the benthos due to outputs from the major sewage outfalls in the area. Some desirable properties of biological indicators that will be sensitive to these changes are discussed, but a detailed discussion of particular indicators is not presented, since the proposed analytical methodology should be applicable to a wide range of indicator variables. Finally, the implications of our results are discussed.

3.2 *Methods*

General Approach

Most of the scientific literature regarding sampling and statistical design for monitoring programs involves sampling potentially impacted locations over time (Green 1979, 1984, 1987; Skalski and McKenzie 1982, Bernstein and Zalinski 1983, Stewart-Oaten *et al.* 1986, Millard and Lettenmaier 1986, Underwood 1989, 1991; Faith *et al.* 1991). The more appropriate designs contain control or reference locations in the sampling plan. Since natural changes take place over time even in the absence of impacts, the reference locations are used to measure and account for these natural changes. Such designs are not generally appropriate for the present application, since they include sampling before the onset of potential impacts. Very little or no appropriate data are available from periods before the present outfalls began operations.

Our task is, therefore, to determine, without having the benefit of "before" samples, whether biological indicator values at a potentially altered location are different from the values expected in reference areas. We must be careful as indicator values will naturally vary among locations for reasons other than the potential impact in question (Hurlbert 1984). For example, indicator values can be affected by differences in location, habitat, time of sampling, historical and oceanographic conditions, depth, sediment grain size, organic content, proximity to larval sources, etc. Thus, when comparing a potentially impacted location with reference conditions, we must not confuse impact with changes due to these other factors (Bernstein and Smith 1991).

It will also be useful to quantify the degree to which impacted locations differ from reference conditions (when they do differ, in a statistical sense). Over time, the quantity and quality of the outfall effluents are changing. It will be of interest to observe changes in the benthos that may follow these changes in the effluents, especially whether the benthos has shifted toward or away from reference conditions. Finally, there may be levels of change that society deems acceptable. Quantification of the differences will enable us to determine if these acceptable limits are exceeded.

Definition of a Reference Envelope

To define which sampling locations (stations) should be considered "reference sites," we rely on the benthic community data (species composition and abundances) and known habitat characteristics. The ordination technique discussed in Section 2 is very useful for segregating potential reference locations from other locations, since reference locations (at least in Southern California) tend to be located in a particular region of the ordination space (Figure 2-1). As discussed in Section 2, reference areas will contain a characteristic suite of species that will enable their identification. In addition, reference locations should be sufficiently distant from outfalls and other known sources that may similarly affect the biological communities (e.g., areas of natural oil seepage).

To assist in the following discussion, we present results of an ordination analysis (Smith and Bernstein 1985, Bernstein and Smith 1986) with the SCCWRP 60 meter survey data taken in 1977 (Figures 3-1 and 3-2). These data are a subset of the data used in Figure 2-1, and are generally described in Section 1 of this report.

In this survey, there are sampling locations (stations) that are relatively far from outfalls and other potential impacts, and are known to contain the species in the reference assemblages described in Section 2. The area of the ordination space where these stations cluster is considered the reference region of the ordination space.

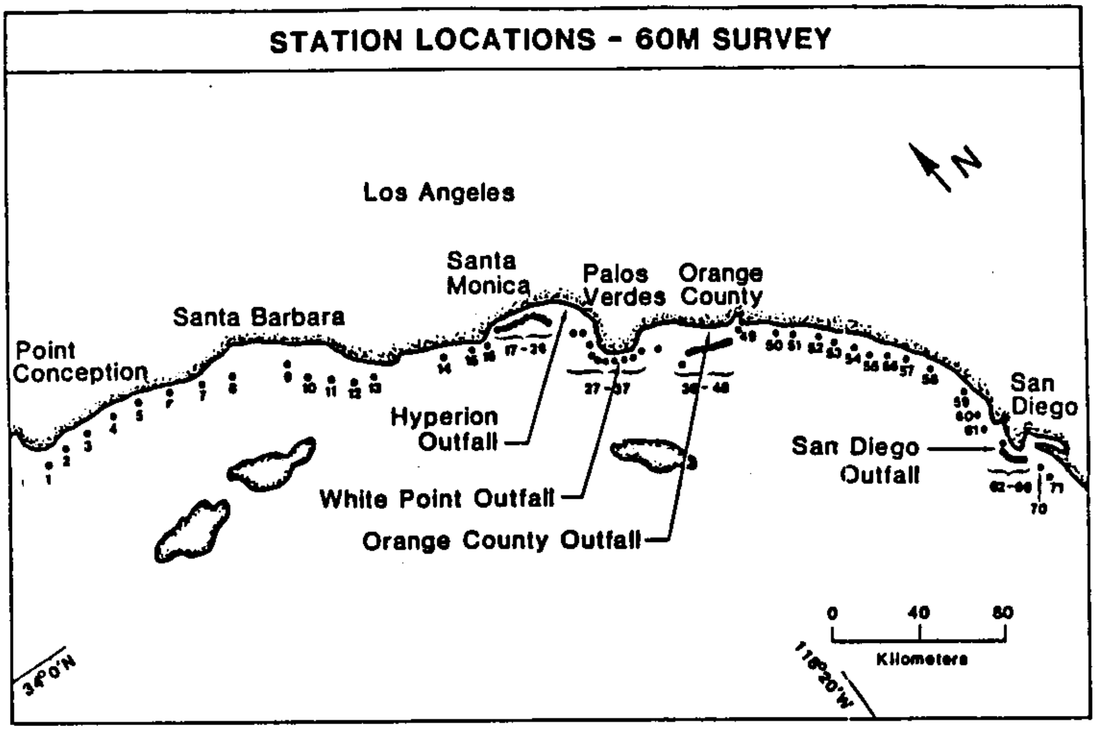


Figure 3-1. Station locations for the SCCWRP 60 Meter Control Survey, conducted in the spring and summer of 1977 (Word and Means 1979). Station 5M is at the terminus of the Hyperion 5-mile outfall. Figure from Smith and Bernstein (1985).

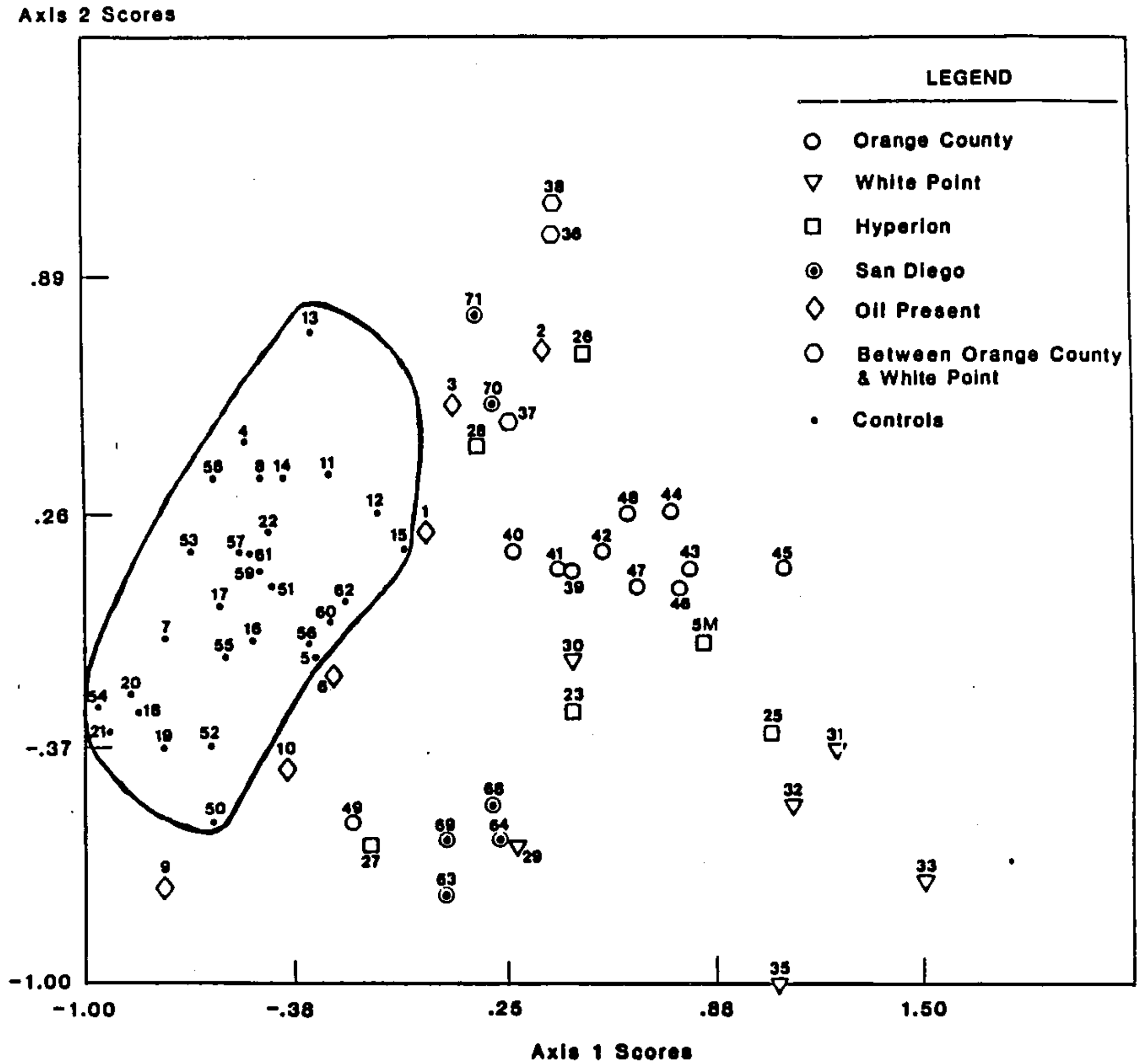


Figure 3-2. The first two ordination axes from analysis of the 60 Meter Control Survey benthic data. Station numbers identify the positions of each station in the ordination space. The stations within the reference envelope are outlined. Figure from Smith and Bernstein (1985).

Other stations that actually may be closer to the outfalls but fall within this cluster of points have a benthic community similar to the known reference stations; consequently, these other stations are also considered as reference stations. None of the stations falling within this cluster had sediment (or other) measurements indicating obvious contamination or enrichment effects. The reference stations chosen in this manner are outlined in Figure 3-2, and will be referred to as stations falling within the *reference envelope*. Bloom (1980) presents a similar idea relating to recovery of a community following disturbance, and Hughes *et al.* (1990, p 679) discuss a similar use of a "regional reference site ellipse".

Here we have chosen reference locations based mainly on benthic community characteristics at the stations, rather than strictly on distance from outfalls. This empirical approach is more useful, since up to a certain point, distance from an outfall is not a very precise predictor of outfall effects. Variations in the rate of effluent discharge, current directions and velocities will also affect the areal extent of such effects.

Assessing Deviations from Reference Conditions

For the purposes of discussion, let us assume that we have sampled the benthos at stations within the reference envelope and also have sampled Station X in the vicinity of an outfall. From the benthic data we measure biological Indicator I. We would like to test the null hypothesis that the value of Indicator I at station X is from the population of Indicator I values from the stations within the reference envelope, and therefore is probably not altered by the outfall effects.

Our recommended approach to testing this null hypothesis involves computing the upper or lower confidence bound for a chosen quantile (percentile) of the distribution of Indicator I values within the reference envelope. If the value of Indicator I at Station X is outside the confidence bound, then the null hypothesis would be rejected. In most cases, we would only be interested in whether the value of Indicator I at Station X is significantly lower or higher than the chosen quantile, thus we are usually dealing with a one-sided confidence bound. A conceptual explanation and justification of this approach follows. Computational details and underlying assumptions are given in Appendix 3A.

The null hypothesis of interest is: the indicator value at a test station is from the population of indicator values found within the reference envelope. Figure 3-3A illustrates the estimated distribution of indicator values within this population. The distribution of indicator values within the reference envelope is estimated

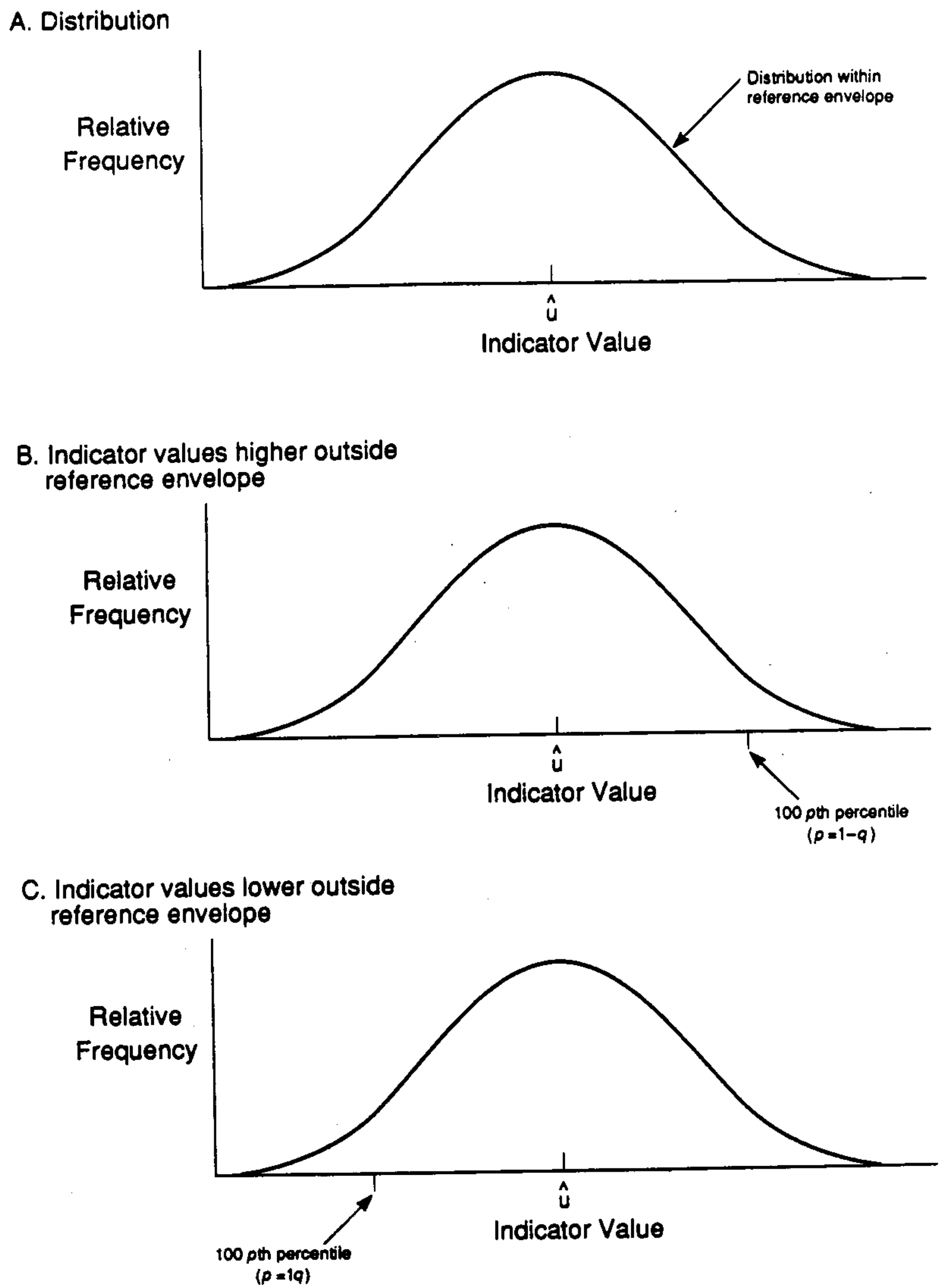


Figure 3-3. The use of percentiles to determine whether a test station is within the reference envelope.

from the indicator values measured at the stations within the reference envelope. We assume that this distribution is normal and can be estimated from the mean and variance of the station data within the reference envelope. Indicators that are not normally distributed can often be transformed to approximate normality (see below).

We would want to reject the null hypothesis for indicator values that have a relatively low probability (q) of being from this population. The chosen value of q leads to a threshold percentile or quantile ($100p$) of the distribution of indicator values. If we expect the indicator values to be lower outside the reference envelope, then $p=q$ (Figure 3-3B). For indicator values expected to be higher outside the reference envelope, $p=1\text{-}q$ (Figure 3-3C). For example, when using abundance of *Amphiodia urtica* as an indicator, we wish to reject the null hypothesis for *A. urtica* abundances having a probability less than or equal to 0.01 of being in the population within the reference envelope. The abundances of *Amphiodia urtica* are expected to be lower outside the reference envelope than within it, thus $p=q=.01$ (Figure 3-3B). For an indicator such as Index 5 (see examples) that is expected to increase outside the reference envelope, $p=1\text{-}q=1\text{-}.01=.99$ (Figure 3-3C).

The mean and variance on which the indicator distribution is based are estimated from the data; thus there will be uncertainty exactly what indicator value represents the $100p$th percentile of the underlying distribution. To account for this uncertainty, we will not use the computed $100p$th percentile of the distribution, but will instead use a $1\text{-}\alpha$ confidence bound (Figure 3-4A and 3-4B) of the $100p$th percentile. The true $100p$th percentile will be found within the computed confidence bounds $100(1\text{-}\alpha)$ percent of the time when n stations are repeatedly drawn randomly from the reference envelope population, where n is the number of stations used to estimate the distribution. In other words, the value of α is a specified type-1 error associated with the estimate of the $100p$th percentile.

To apply this method, the indicator value corresponding to the $100(1\text{-}\alpha)$ confidence bound of the $100p$th percentile is computed (see below), and test stations with indicator values outside this bound are assumed to be outside the reference envelope (Figure 3-4C).

We emphasize that this approach deviates fundamentally from the more common ANOVA or t-test approach of comparing the *overall mean* indicator value in a reference area with the mean value at a test station or area. This is because the *variability* of the overall reference-area mean declines as the number of stations within the reference area increases, and it will become easier to detect smaller and smaller deviations from

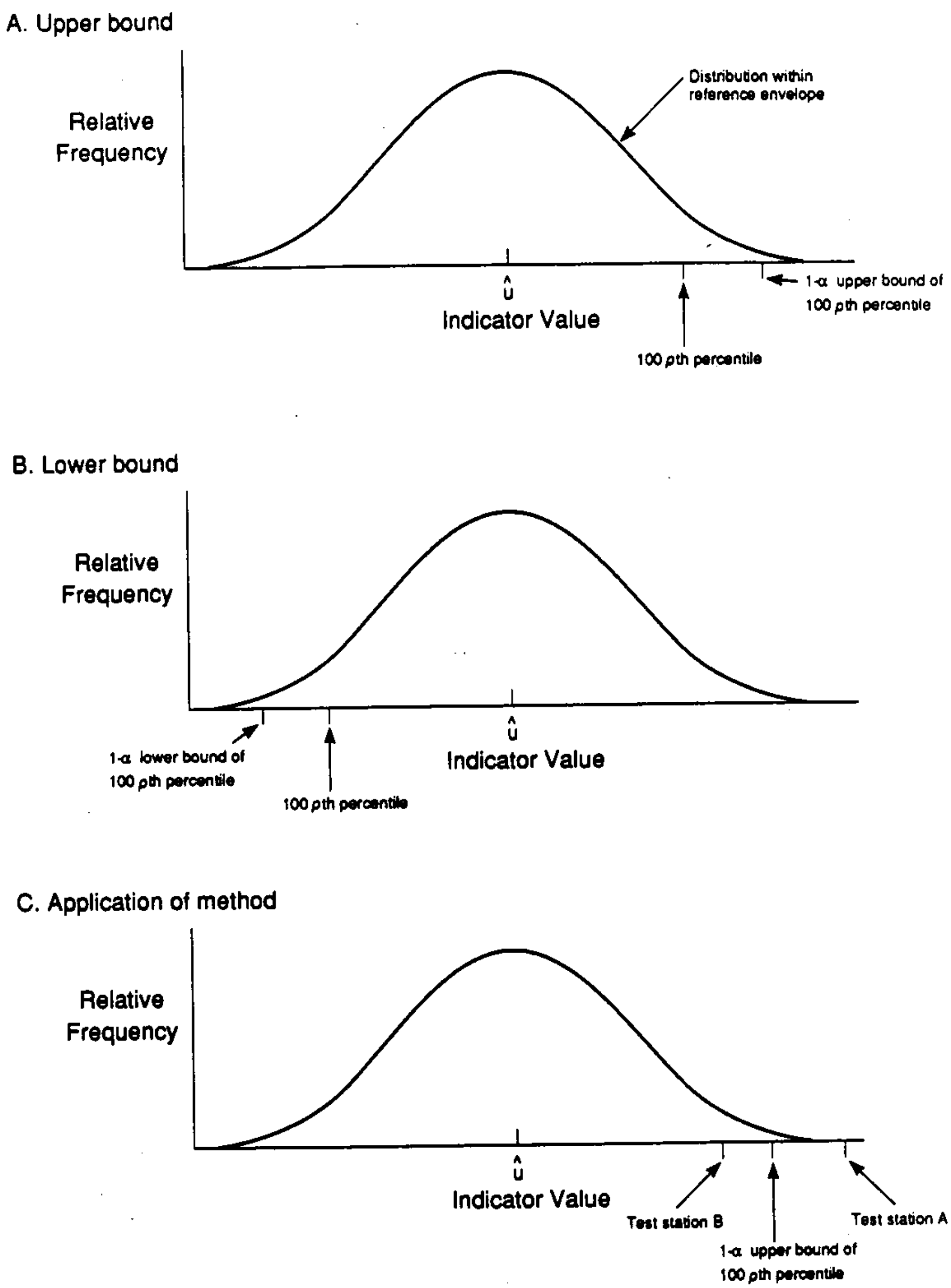


Figure 3-4. The use of confidence bounds of the 100*p*th percentiles to test for indicator values outside the reference envelope. A. Upper bound used when higher indicator values expected outside reference envelope (e.g., Index 5 values). B. Lower bound used when lower indicator values expected outside the reference envelope (e.g., abundances of *A. urtica*. C. Example for indicator variable expected to be higher outside the reference envelope. Test station B would be considered within the reference envelope and test station A would be outside the reference envelope.

the overall reference area mean. Instead we are testing whether the test station is within the population of reference envelope stations. Figure 3-5 contrasts our approach with ANOVA and shows how the ANOVA approach potentially leads to rejection of the null hypothesis for test stations similar to a large proportion of the stations within the reference envelope. This is unacceptable, since the whole point of the analysis is to determine if the test station is similar to stations within the reference envelope.

The confidence bounds are based upon the *between-station* variability of Indicator I values *within the reference envelope*. This is a very important point, since this variability captures the natural changes that would be expected in going from one location to another, as will be the case when comparing a test station (e.g., Station X) with the reference stations. The method as proposed assumes that all reference and test stations have the same number of replicates. In fact, with the data in the examples, one replicate was taken at each station. When there are multiple replicates at the stations, the between-station variance should be computed from the means of the replicates at each station. The method must be further developed to utilize means of different numbers of replicates at the different stations, because the between-station variance can change as the number of replicates averaged changes.

In cases where physical distance between stations in the reference area highly correlates with community differences, it would be preferable for the distances between the reference stations to be on a similar scale to the average distance between the reference stations and the test station. At least at 60 meters depth in the Southern California area, this correlation between inter-station distance and dissimilarity of benthic communities is only moderate. This can be seen in Figure 3-2, where geographically adjacent stations within the reference envelope are often close, but in some cases relatively distant in the ordination space. As discussed in the examples (Section 3.4), the differences in sediment characteristics at the stations appear to override the importance of physical distance in determining the dissimilarity of communities at different stations.

In many cases, the between-station variability within the reference envelope will be partially due to differences in habitat, time, geography, or other environmental factors not related to outfalls. The sensitivity and accuracy of the proposed test can be increased if these types of factors are held constant or partitioned out in the computations for the confidence bounds. A method for accomplishing this is given in Appendix 3A and demonstrated in the examples.

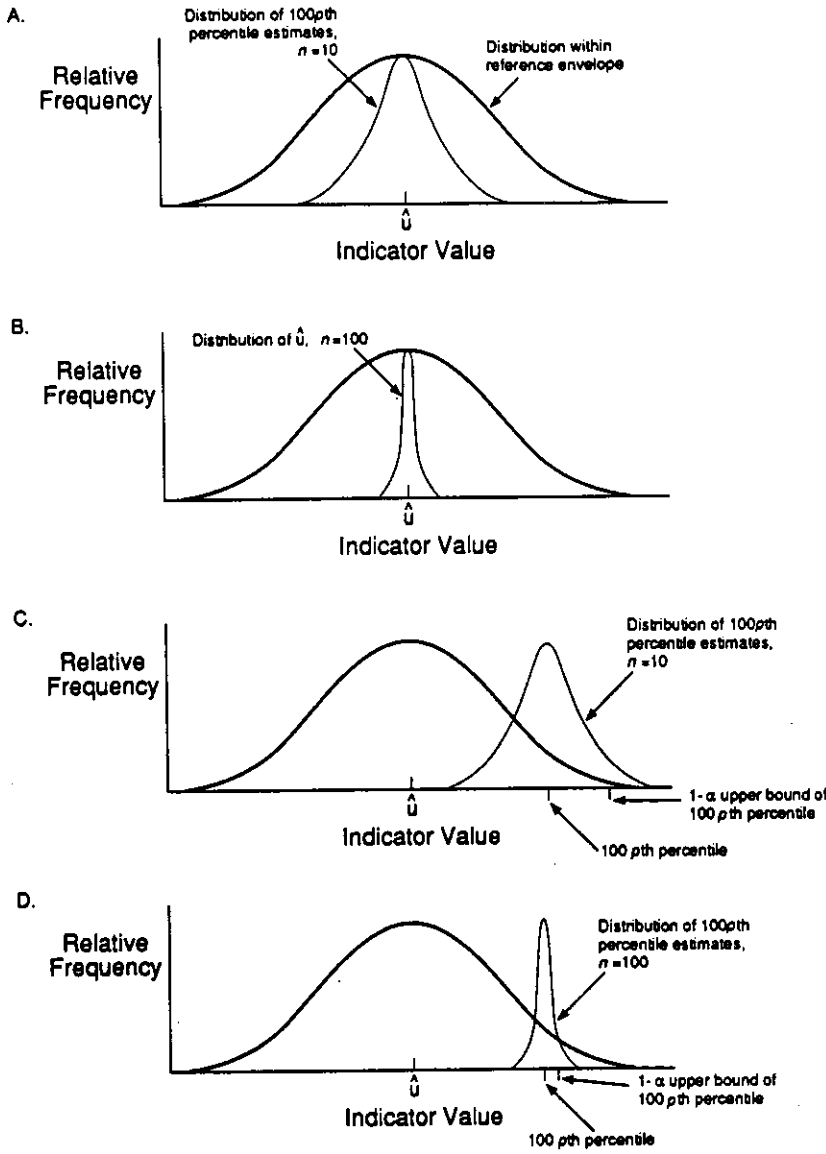


Figure 3-5. Contrast of the proposed approach with the ANOVA or t-test approach. A. Distribution of μ when $n = 10$. B. Distribution of μ when $n = 100$. C. Distribution of estimate of the 100*p*th percentile when $n = 10$. D. Distribution of the estimate of the 100*p*th percentile when n = 100. ANOVA or t-tests would be based on the distribution of μ, which is at the center of the reference envelope distribution. As n gets larger, the distribution of μ gets narrower, and the null hypothesis that the mean at a test station is no different from μ will be rejected for stations closer and closer to μ - even though many such stations may have indicator values similar to a large percentage of stations within the reference envelope. In contrast, the bound used with the proposed method is based on the distribution of the 100*p*th percentile estimates, which will be centered well toward the outer portion of the distribution of indicator values within the reference envelope. As n increases, the bound will be closer to the estimate of the 100*p*th percentile, which remains an outlier in the distribution of indicator values within the reference envelope. Thus, we would never reject the null hypothesis for test stations that resemble a large percentage of the indicator values within the reference envelope. This assumes, of course, that p is set to a value that corresponds to a tail region of the reference envelope distribution.

3.3 *Biological Indicators*

The discussion in Section 2 suggested some potential biological indicator variables. For example, changes in abundances of individual species (such as *Amphiodia urtica)*, or levels of community structure parameters (such as species diversity), or changes in position in an ordination space could indicate outfall effects. It is not our purpose to review all potential biological indicators, since the proposed approach should be generic in its application to a wide range of indicators. We only discuss some general principles that should apply in the choice and usage of indicator variables for outfall effects. Underwood and Peterson (1988), Keough and Quinn (1991), National Research Council (1986, 1990), Grassle and Grassle (1984), and Chapman (1991) provide further discussion on biological indicators.

Evaluation criteria for selecting biological indicators

Here we propose a few simple guidelines for selecting biological indicator variables. A good indicator variable should 1) have ecological interpretability, 2) have a linear or at least a monotonic (only increasing or decreasing in value) relationship with the outfall gradients, 3) be associated with sufficiently low variability to afford reasonable power with desired statistical tests, and 4) be practical in relation to the available resources and logistics.

For an indicator variable with good ecological interpretability, we will have a fairly good idea of why the observed changes take place in response to the outfall effects. In other words, we would know what the indicator is indicating. There are many potential aspects of an outfall that could affect an indicator. The presence of organic matter, sulfides associated with the breakdown of the organic matter, burial, toxic chemicals, changes in sediment size, or indirect effects could cause the changes in the indicator value. Unfortunately, for many of the more popular indicators, this knowledge of causes is lacking. For example, the information in Section 2 indicates that *Amphiodia urtica* would be a good indicator due to its areal distribution and abundance and sensitivity to outfall effects. However, little is presently known regarding the specific causes of this pattern. Ongoing experiments by SCCWRP to discover the specific sediment or water component(s) to which this species is sensitive should greatly enhance the ecological interpretability of this species as an indicator.

Indicator variables that change in a non-monotonic manner in response to outfall effects will have the same value at more than one position along the outfall gradient (Green 1987, Underwood 1989, Keough and Quinn 1991). Pearson and Rosenberg (1978) summarize the patterns of selected community parameters often observed along outfall gradients (Figure 3-6). The number of species, biomass, and total abundance all tend to change non-monotonically (increase to a peak and then decrease) along an outfall gradient. If this is the case, the number of species could have the same value at places within the normal, transition, or polluted zones (Figure 3-6). It is only when comparing the normal zone with the more highly polluted zones that these community parameters will be useful. Another potential indicator that varies non-monotonically along the outfall gradient is abundance of *Parvilucina tenuisculpta*. This species is most abundant where the outfall effects are moderate.

These types of indicators may at times be informative when used in conjunction with other indicators that provide information on the position along the outfall gradient, allowing us to structure different null hypotheses that are appropriate at different parts of the outfall gradient. For example, if we wanted to test whether a station was just moderately affected by an outfall, our null hypothesis would be that the number of species is not *elevated* in relation to the reference envelope. On the other hand, if we were interested in whether a station was grossly polluted, our null hypothesis would be that the number of species is not *depressed* in relation to the reference envelope.

The power of a statistical test is the probability that a desired level of effect will cause the rejection of the null hypothesis. The power depends on the variability of the dependent variable (indicator variable in our case), the number of observations (stations in our case), and the magnitude of the effect that we wish to detect. The greater the variability of the dependent variable (within the reference conditions), the lower the power. More observations and larger effect levels usually increase power. Thus, if the dependent variable is highly variable, we will need to take more observations and/or lower our standards as to the magnitude of change we wish to detect. This means that when all other properties of an indicator are equal, the less variable indicator will be more sensitive and cheaper to use. Limited budgets for monitoring often restrict the level of sampling, so dependence on imprecise indicator variables could be a waste of resources.

The preceding discussion on nonmonotonic changes along the outfall gradient is also relevant to the concept of power. The power of a statistical test comparing stations from the normal zone with the transition zone could be very low because the differences to be detected are very small or zero in some parts of the transition

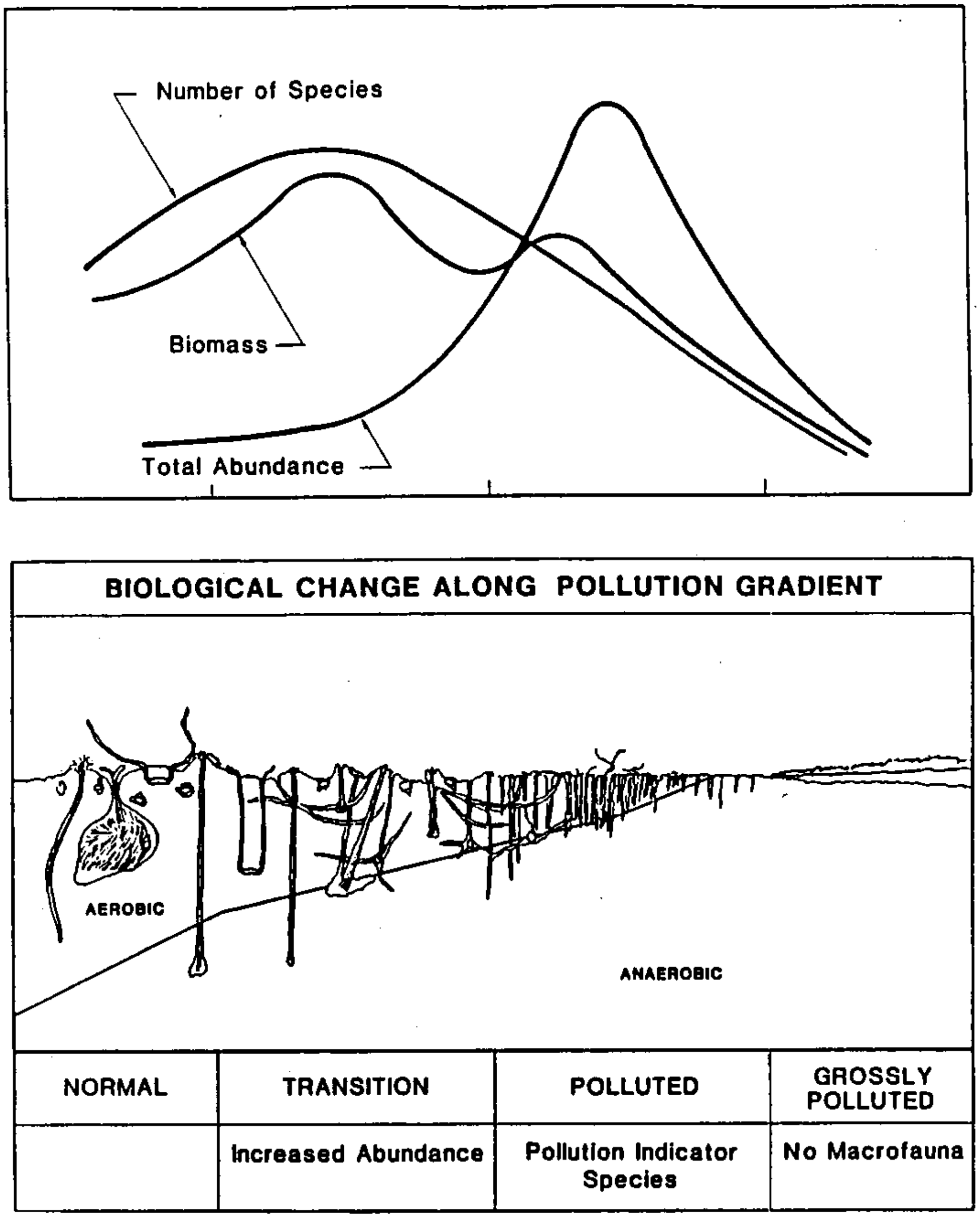


Figure 3-6. Generalized diagram of changes in fauna, sediment structure, and benthic community parameters along a gradient of organic enrichment (from Pearson and Rosenberg 1978).

zone (Figure 3-6). On the other hand, indicator variables whose values change more linearly along the outfall gradient will tend to be associated with greater power.

Fairweather (1991) and National Research Council (1990) discuss the importance of power in monitoring designs. Multivariate measures sensitive to changes in community composition are generally associated with greater power than other univariate measures such as abundances of individual species (Bernstein and Smith 1986, Smith *et al.* 1988, Warwick *et al.* 1990a, 1990b; Faith *et al.* 1991).

Finally, the levels of effort and resources associated with different indicator variables will vary. This means that we will always need to balance the benefits of a particular indicator with the costs associated with measurement of the indicator. For example, there is no need to allocate additional resources to measure the best possible indicator(s) if other much cheaper and simpler indicators will perform almost as well as the optimal indicators.

3.4 *Examples*

Population measure - *Amphiodia urtica*

Here we apply the methods described in Appendix 3A to abundances of *Amphiodia urtica* from the SCCWRP 60 meter survey (Appendix 3B). Since this species tends to decrease in abundance closer to outfalls, we will be interested in the lower confidence bound of the reference envelope distribution.

As discussed in Section 2.3, *A. urtica* is sensitive to changes in depth, sediment type and outfall effects, and also tends to change in abundance over time. If we can control for depth, time, and sediment type then we can estimate the loss in abundance due to outfall effects. With these data, time and depth are constant (one survey at 60 meters depth), so we need only control for changes in sediment type.

Two analyses are performed. The first will assume that the range of sediment types at the stations does not affect *A. urtica* abundances. The second analysis does not make this assumption, and variation in sediment characteristics is accounted for.

In the first analysis, all stations in the reference envelope are pooled in one distribution. Tests for the normality of this distribution were run with the SAS Univariate procedure (SAS 1990) using the Shapiro-Wilk statistic (Shapiro and Wilk 1965). The probability ($p=.07$) associated with the null hypothesis of a normal distribution indicated that a transformation would be beneficial to make the distribution more normal. Using the Box-Cox (1964) procedure, it was determined that a cube root transformation would be the optimal for transformation to normality. After the transformation, the probability associated with the test for normality increased considerably ($p=.70$), indicating that the transformation was successful.

The mean transformed value in the reference envelope is 5.108 and the standard deviation is 1.023. Using $p=.05$, $\alpha=.05$, $n=29$, and $g'(.95,.95,29)=2.232$, the lower bound of the transformed data is 5.108-(2.232)1.023=2.824. Back-transforming (cube of transformed value) gives a mean of 133.275 and a lower bound of 22.527. Thus, any station with an abundance of *A. urtica* less than 22.527 will be considered below the 5th percentile of the reference envelope distribution and therefore probably affected by the outfall(s). In Figure 3-7 the null hypothesis is rejected for all stations to the right of the line and accepted for all stations to the left. Thus, some of the stations beyond the defined reference envelope (dots) have sufficient abundances to possibly be unaffected by the outfalls (according to this indicator only).

Before we can run the second analysis, we need to examine the patterns of sediment characteristics in the data and set up strata that will hopefully control for the effects of sediment on *A. urtica* abundances. In Figure 3-8 the values for percent sand are superimposed at the corresponding station positions. In general, the percent sand is higher toward the top of the plot and lower toward the bottom (when the axis 1 values are somewhat constant).

Also, the TVS (total volatile solids) values are lower toward the top and higher toward the bottom when the axis 1 level is constant (not shown). We could hold the effective sediment type relatively constant by dividing the stations into three strata as delimited by the lines in Figure 3-8. These are called strata 1, 2 and 3 going from top to bottom in Figure 3-8.

We can now compute the reference envelope lower bounds for each stratum based on the pooled standard deviation. The Box-Cox procedure indicated that again a cube root was the optimal transformation for normality. We use the transformed data in this analysis.

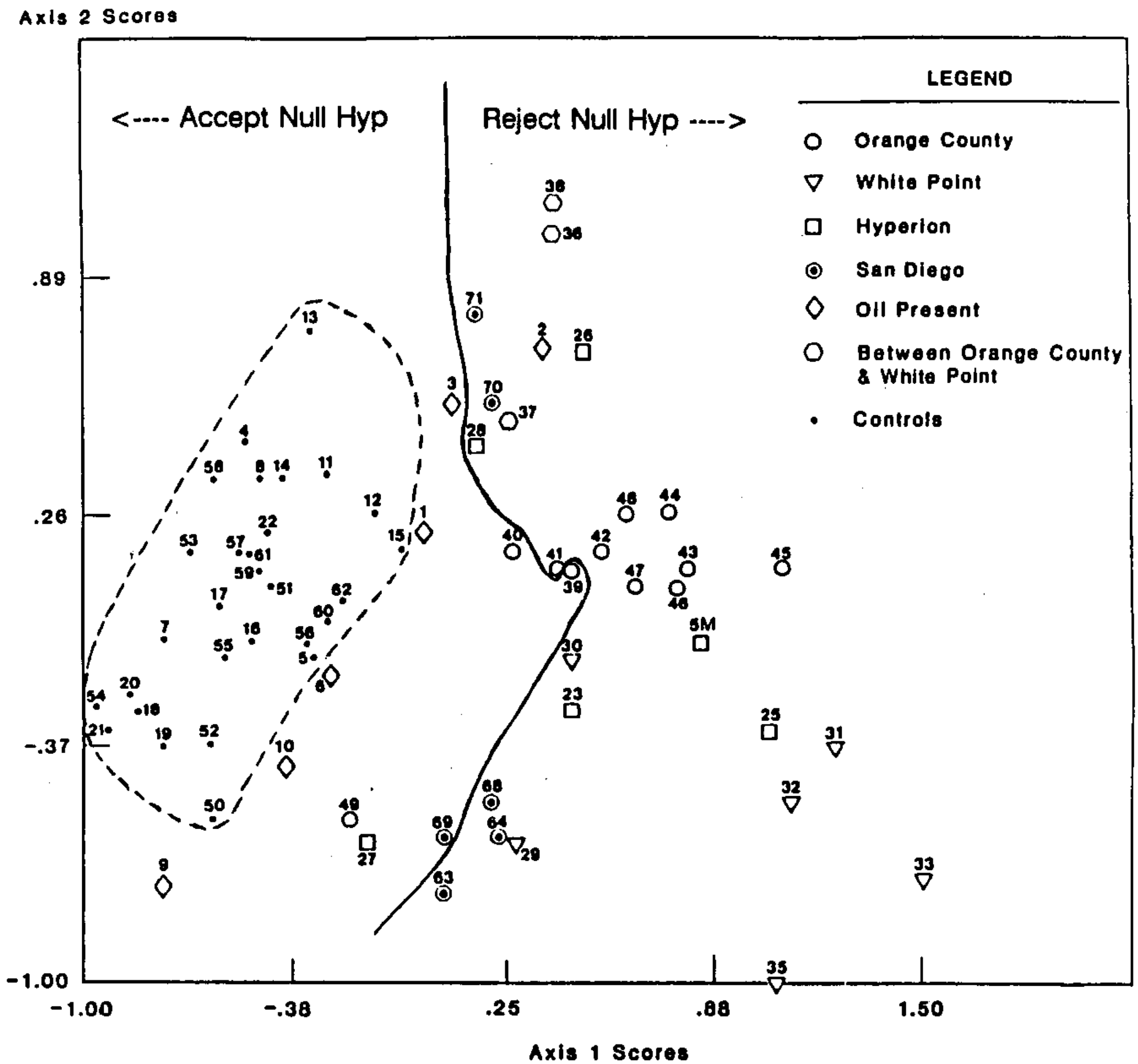


Figure 3-7. The first two ordination axes from analysis of the 60 Meter Control Survey benthic data. Null hypothesis accepted for all stations are to the left of the solid line, and rejected ones for all stations are to the right. The null hypotheses is that the abundance of Amphiodia urtica is not different from the population of values within the reference envelope. No stratification for sediment differences used. Dashed line outlines the reference envelope.

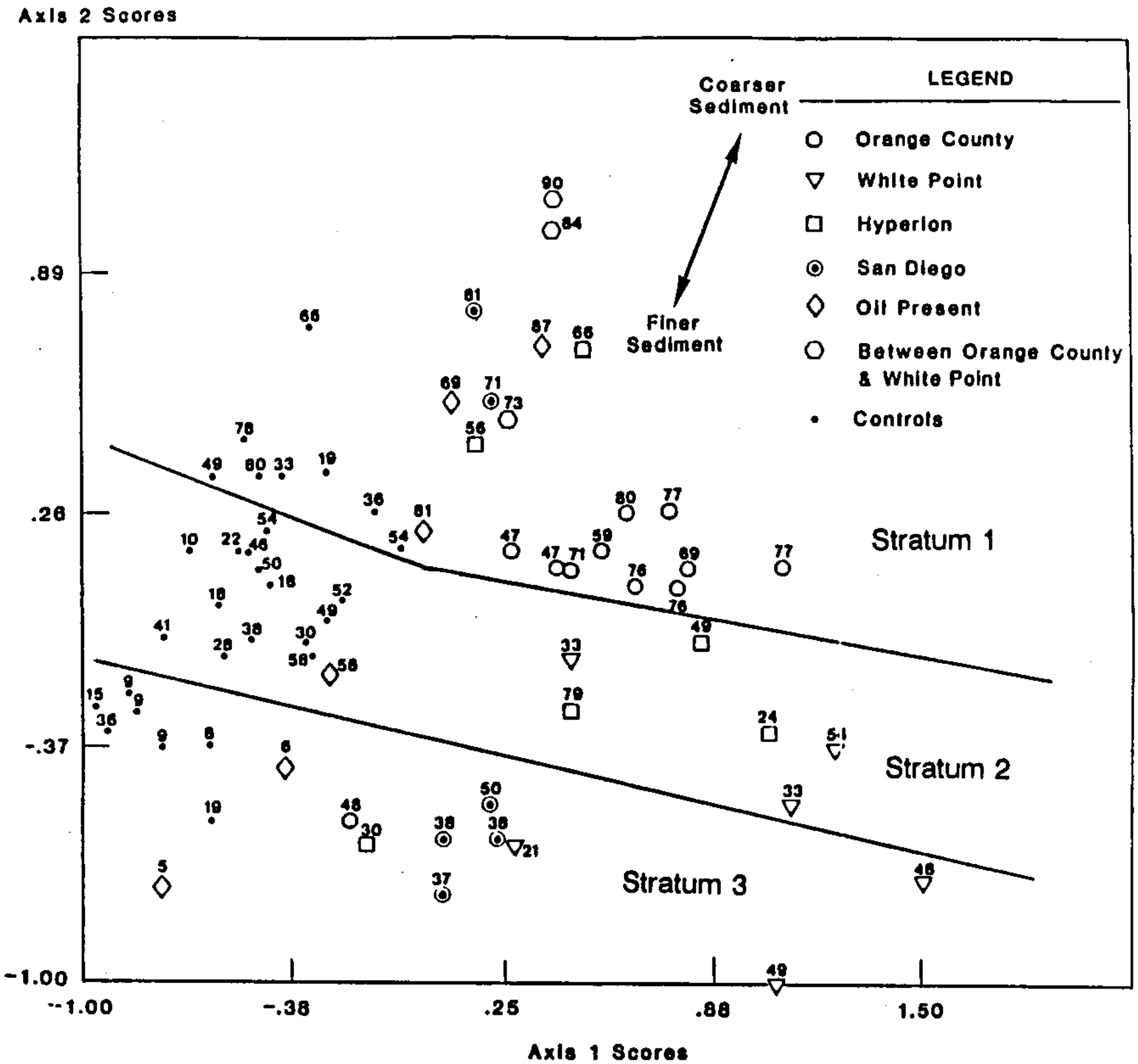


Figure 3-8. The first two ordination axes from analysis of the 60 Meter Control Survey benthic data. Measured percent sand values shown at corresponding station positions. Arrows indicate general direction of change of percent sand within the space when axis 1 position is relatively constant. It should be noted that since percent sand is only a crude measure of the effective (to the benthos) sediment type, we would expect a fair amount of variation in the percent sand within the strata.

The mean transformed values in the reference envelope are 4.24, 5.54 and 5.24 for strata 1-3, respectively. The pooled standard deviation is 0.889. Using $p=.05$, $\alpha=.05$, $n=26$, and $g'(.95,.95,26)=2.275$, the back-transformed lower bounds for the strata are 10.90, 43.512 and 33.139, respectively. Thus, any station with an abundance of *A. urtica* less than 10.90 in stratum 1, 43.512 in stratum 2, or 33.139 in stratum 3 will be considered below the 5th percentile of the reference envelope distribution and probably affected by the outfall(s). In Figure 3-9 the null hypothesis is rejected for all stations to the right of the line and accepted for all stations to the left. This result is slightly different from the first unstratified analysis. The null hypothesis is now accepted for stations 2 and 41, but rejected for stations 10 and 27.

The results for stratum 1 should be viewed with caution, since the sediment types for some of the test stations (e.g., stations 38, 36, 71) seems to be somewhat different when compared with stations within the reference envelope (Figure 3-8). Thus, for some of the stations within stratum 1, we are probably confounding outfall and sediment-type effects.

Changes in community composition - Index 5

Index 5 (Smith and Bernstein 1985) was developed to quantify the biological community gradient between reference and outfall areas utilizing ordination results. The order of stations along this gradient is somewhat evident in Figure 3-2, with reference stations toward the left and stations at the outfalls toward the right in the plot. To quantify positions on this gradient, a line is drawn between the mean location in the ordination space of the reference envelope stations and the mean location of the most affected outfall stations (Figure 3-10). The positions of each station on this gradient (Index 5 values) are the projections of the stations onto this line (Figure 3-11). With these data, stations with higher Index 5 values are more affected by the outfalls, and lower values in the reference envelope are presumably unaffected.

The raw Index 5 values are rescaled so that a value of one (1.0) corresponds to the peak in the number of species along the outfall gradient (Figures 3-6, and 3-12). This rescaling makes it possible to know something about the community characteristics at a station by knowing its Index 5 value. For example, the Index 5 values at stations around the San Diego outfall are around 1.0, which tells us that the community here is near the species richness peak, but the stations closest to the other outfalls are beyond this peak, and the number of species is declining. The rescaled Index 5 values are shown in Figure 3-13. This scaling of the Index could also be useful in comparisons among regions, since the Index would be scaled to a feature common to the different regions.

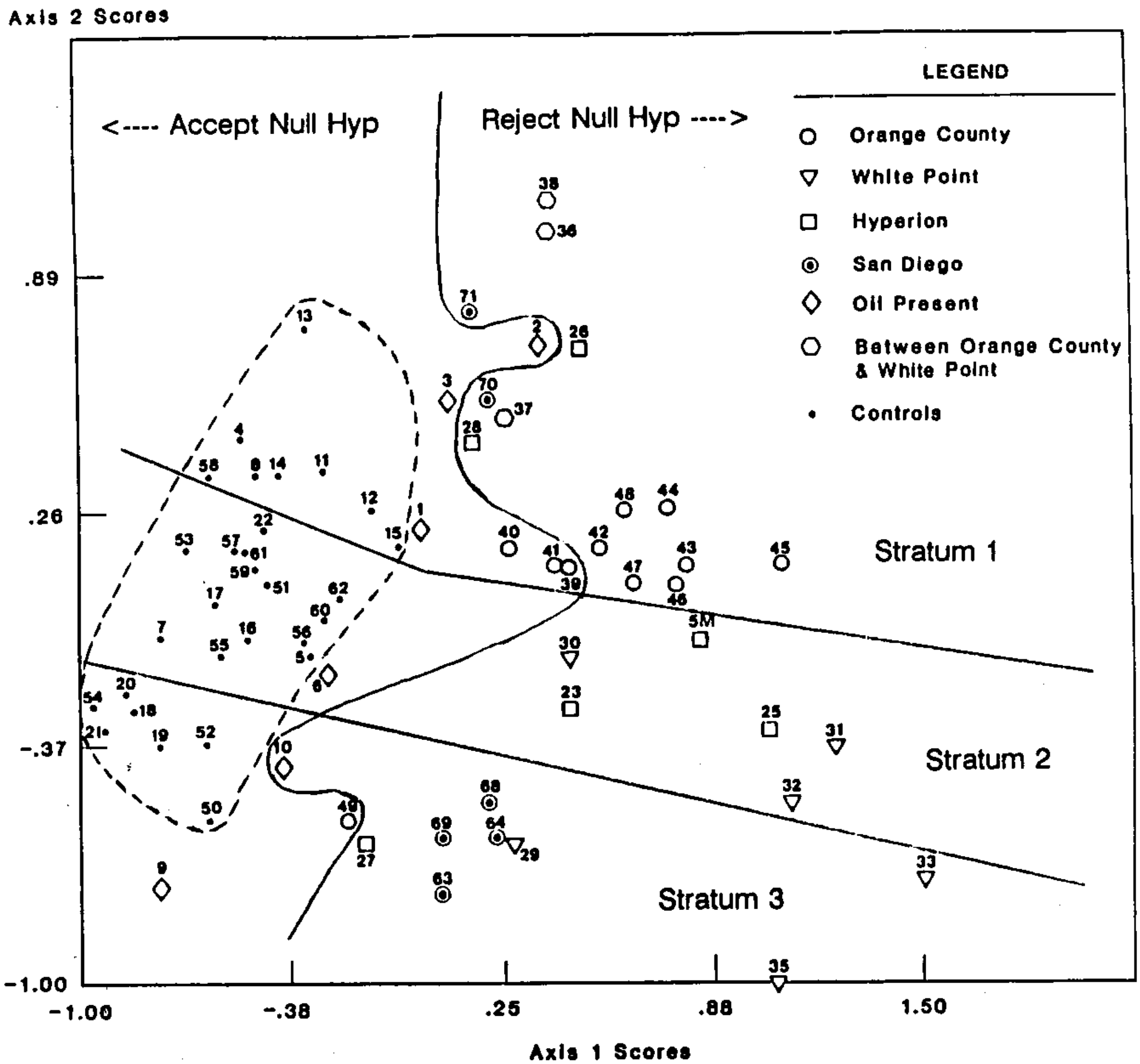


Figure 3-9. The first two ordination axes from analysis of the 60 Meter Control Survey benthic data. Null hypotheses accepted for all stations are to the left of the solid line, and rejected ones for all stations are to the right. The null hypothesis is that the abundance of Amphiodia urtica is not different from the population of values within the reference envelope. Stations stratified by sediment types for analysis.

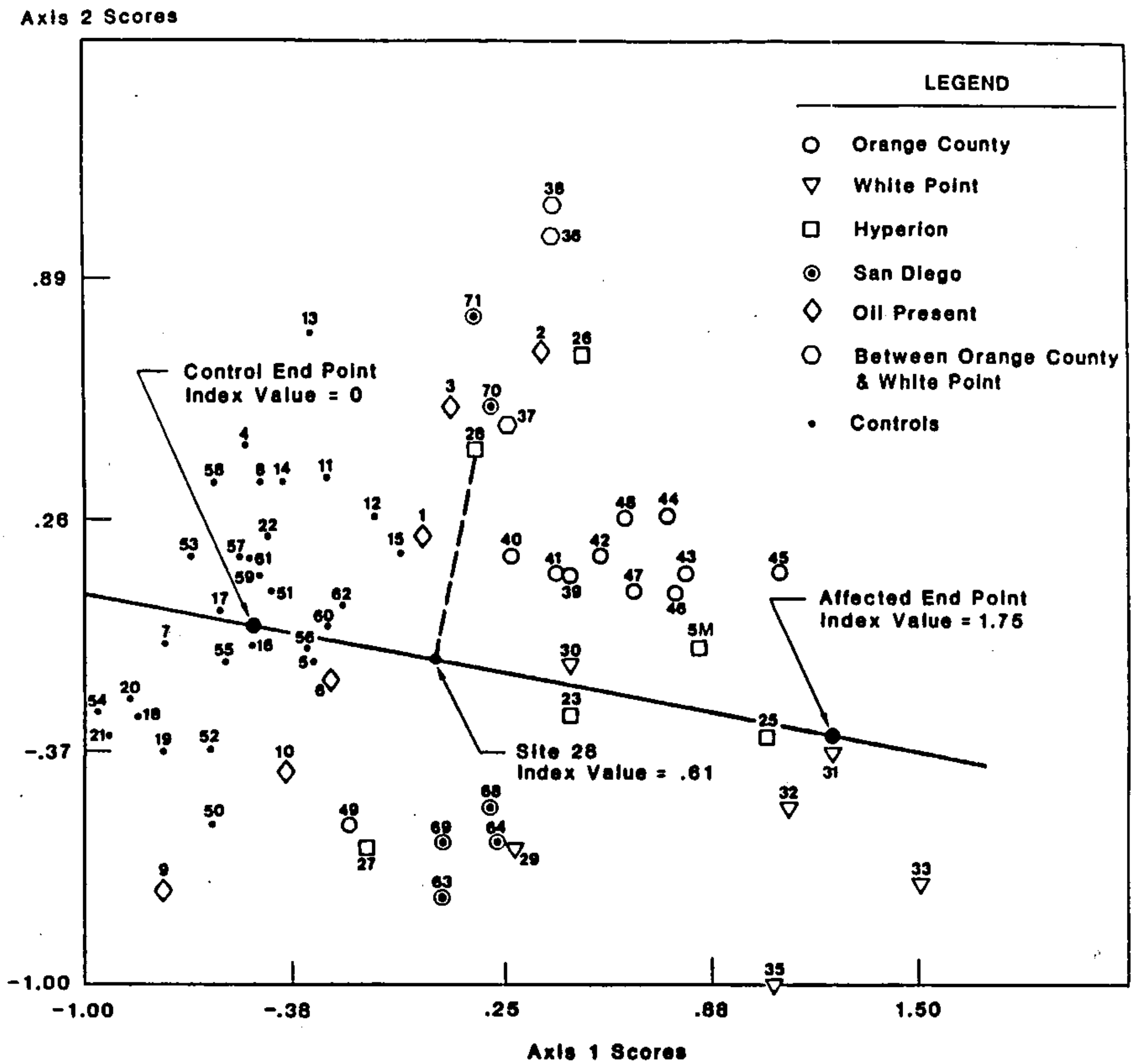


Figure 3-10. The first two ordination axes from analysis of the 60 Meter Control Survey benthic data. Line drawn between two end points indicating the mean position of the reference envelope stations and the mean position of the stations closest to the outfalls (excluding the San Diego outfall), respectively. Raw Index 5 values for the stations are the projections of the station positions onto this line. The projection for station 28 is shown as an example. Figure from Smith and Bernstein (1985).

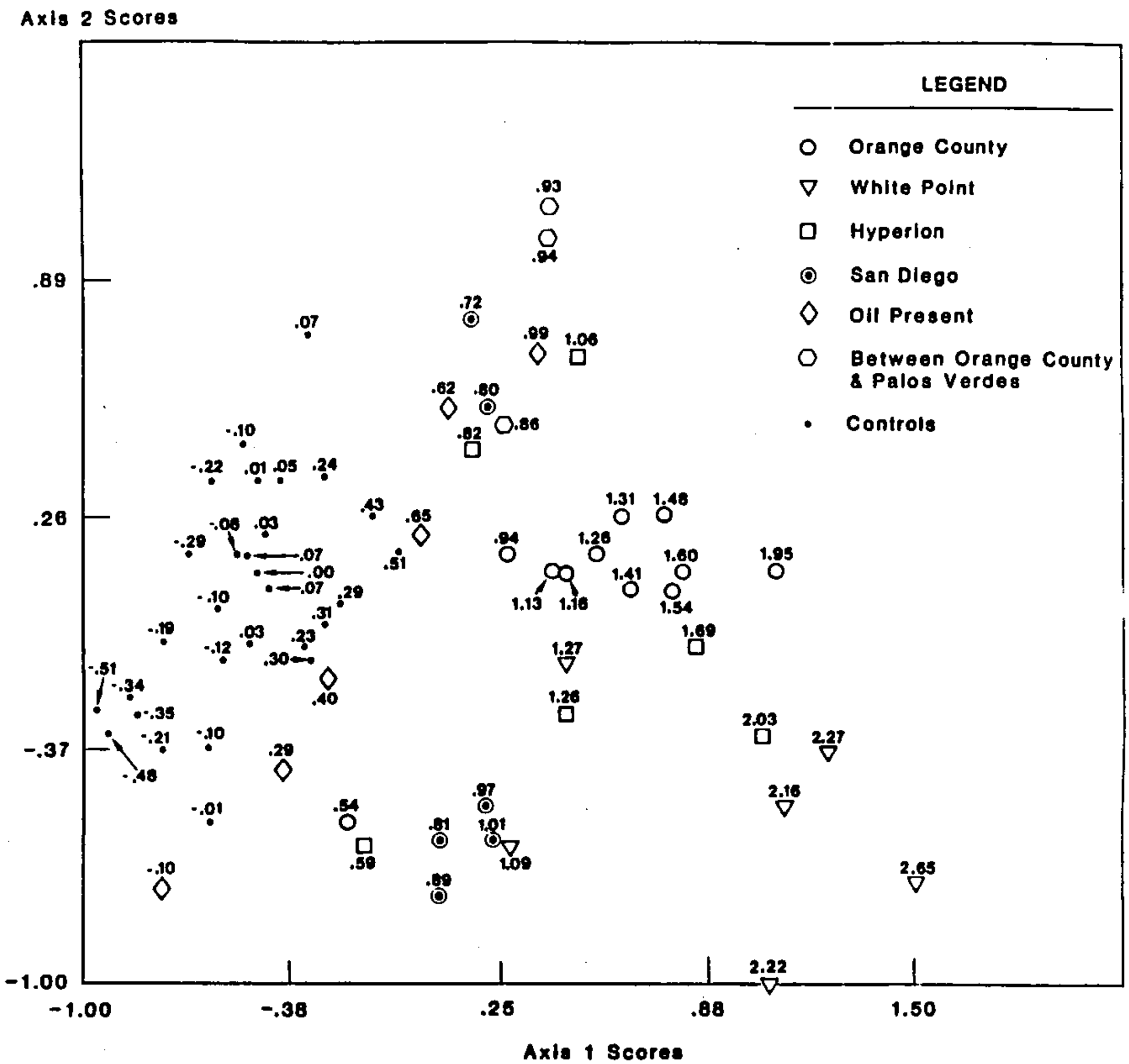


Figure 3-11. The first two ordination axes from analysis of the 60 Meter Control Survey benthic data. Scaled Index 5 values shown at each station position. Figure from Smith and Bernstein (1985).

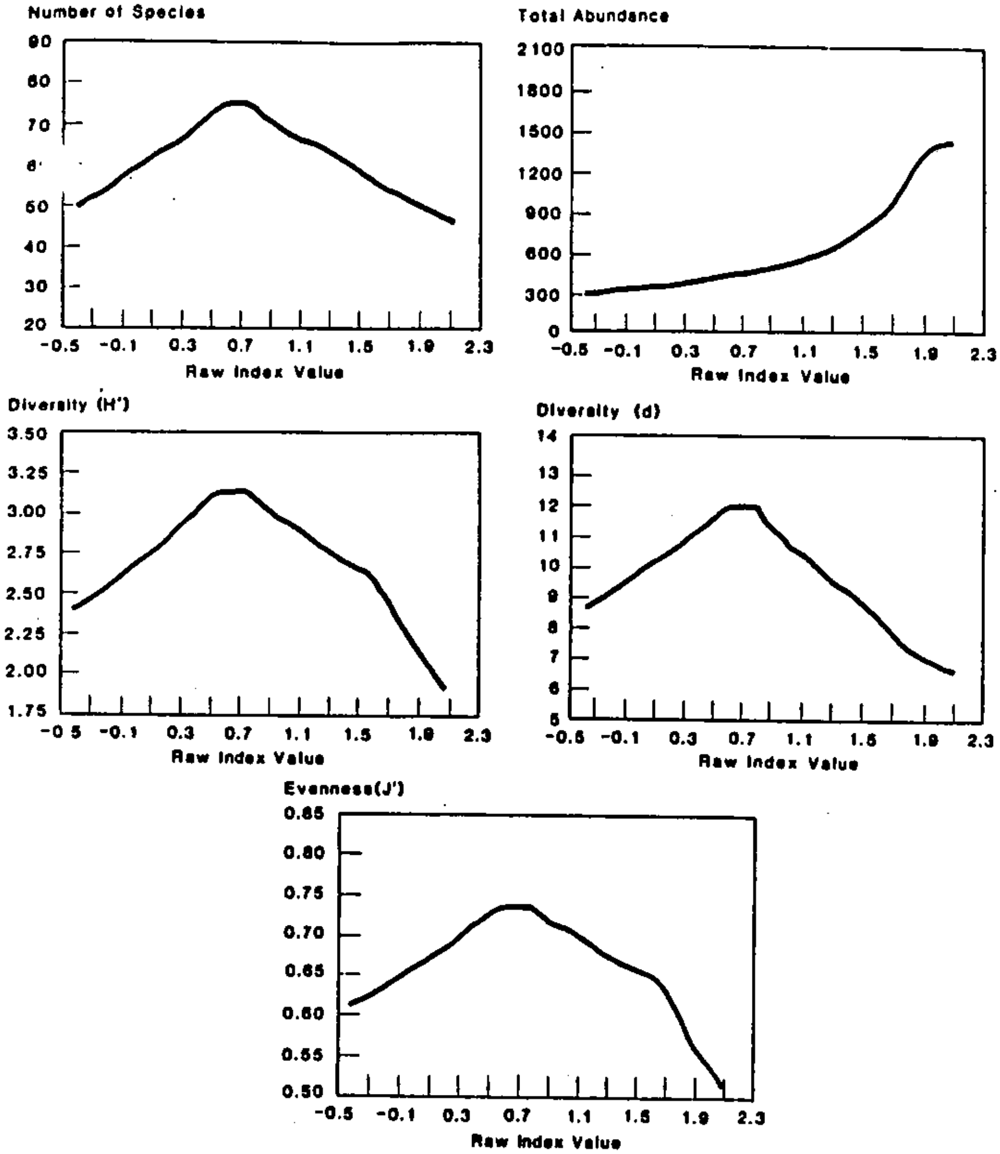


Figure 3-12. The pattern of selected community parameters along the outfall gradient define by the Index 5 values. The curves are smoothed representation of the data values, calculated with a weighted moving average. Note that the curves for species number and total abundance are similar to those predicted in Figure 3-6. The total abundance curve is not descending on the right because stations on the most extremely affected end of the gradient were not sampled. Sannon-Wiener diversity (H'), Gleason richness diversity (d), and evenness (J') also shown. The raw Index 5 values are rescaled so that a value of 1.0 corresponds to the peaks in species richness, diversity and evenness at around .7 raw index value. Figure from Smith and Bernstein (1985).

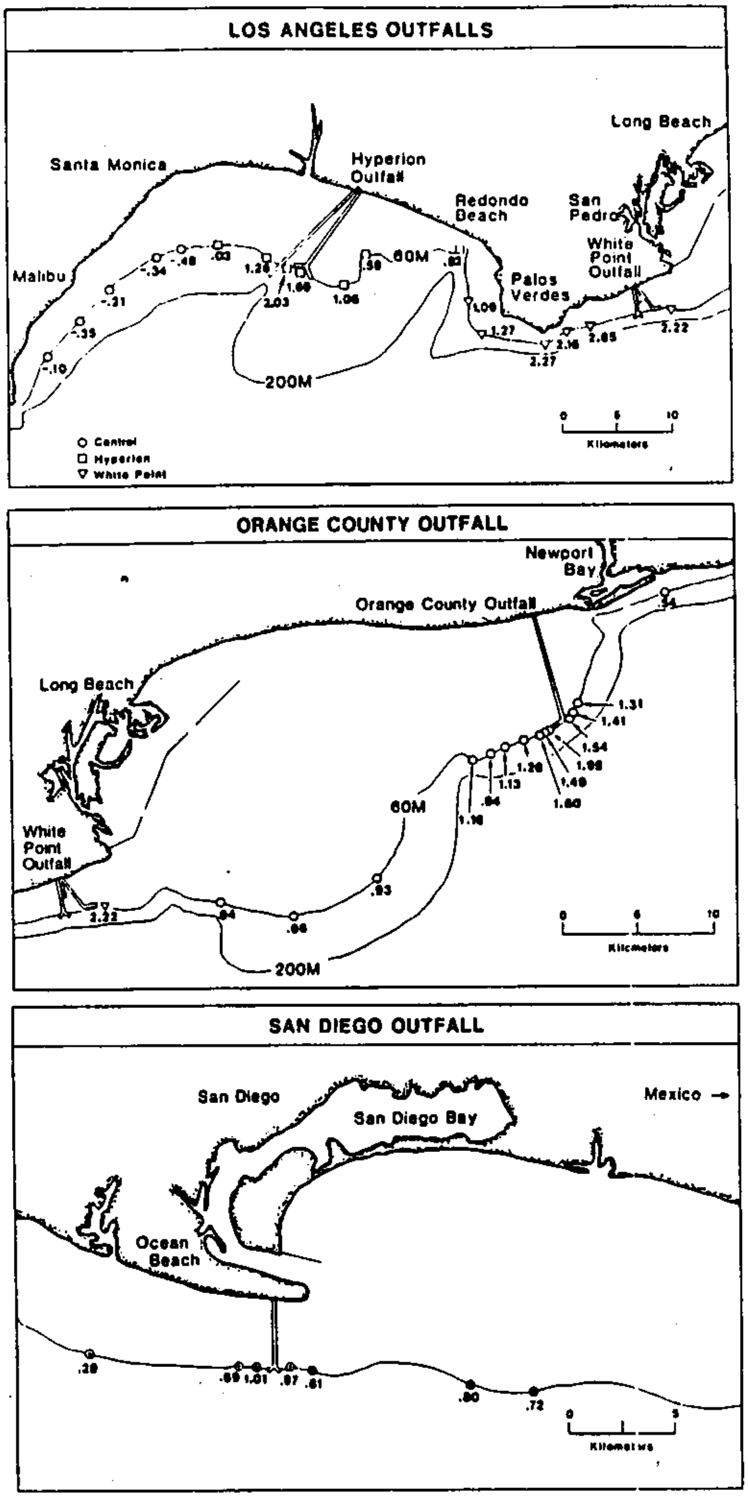


Figure 3-13. Scaled Index 5 values for stations in the vicinity of the major Southern California outfalls. Note the generally increasing values closer to the outfalls.

One very convenient feature of Index 5 *as applied to these data* is the fact that the sediment-type gradient (independent of sediment changes caused by the outfalls) is approximately at right angles to the outfall gradient quantified by Index 5 in the ordination space (Figures 3-8 and 3-10). This means that the Index 5 values are independent of the sediment-type gradient, and there is no need to stratify or adjust for sediment type when testing hypotheses with Index 5 values. We would expect this independent relationship between the sediment and outfall gradients to continue in future surveys, so this feature of the Index should remain with a similar sampling program.

Index 5 values at stations around the outfalls are higher than those in the reference envelope. Thus, we are interested in testing whether test stations are above the upper bound of the 95th percentile of the distribution of Index 5 values in the reference envelope. The test for normality of Index 5 values showed no indication of non-normality and the Box-Cox procedure showed that the optimal transformation was almost the same as no transformation at all. Thus, no transformation is applied to the Index values.

The mean Index value in the reference envelope is -0.021 and the standard deviation is 0.255. Using $p=.95$, $\alpha=.05$, $n=29$, and $g'(.95,.95,29)=2.232$, the upper bound of the 95th percentile is 0.549. Any station with an Index value greater than 0.549 will be considered beyond the reference envelope distribution and probably affected by the outfall(s). In Figure 3-14 the null hypothesis is rejected for all stations to the right of the line and accepted for all stations to the left. The null hypothesis is accepted for only four stations beyond the original reference envelope. It should be noted that some of the stations for which the null hypothesis was rejected are in the area of natural oil seeps, which may have organic enrichment effects that appear similar to outfall effects.

Index 5 as an indicator has some advantages over a population measure such as *A. urtica* abundance because it is based on changes in the abundances of several species simultaneously rather than a single species. This feature can give Index 5 values more sensitivity to environmental changes related to outfalls, and also lower variability when such environmental changes are not present (Bernstein and Smith 1986, Smith *et al.* 1988).

The fact that the Index values are not sensitive to the sediment-type gradient also reduces their variability under relatively constant outfall influences. The lower variability of the Index under constant outfall influence (i.e., no influence) is demonstrated by comparing the standard deviations of the indicators within the reference envelope. The transformed *A. urtica* abundances are almost three and a half times more variable than that of Index 5 (0.889

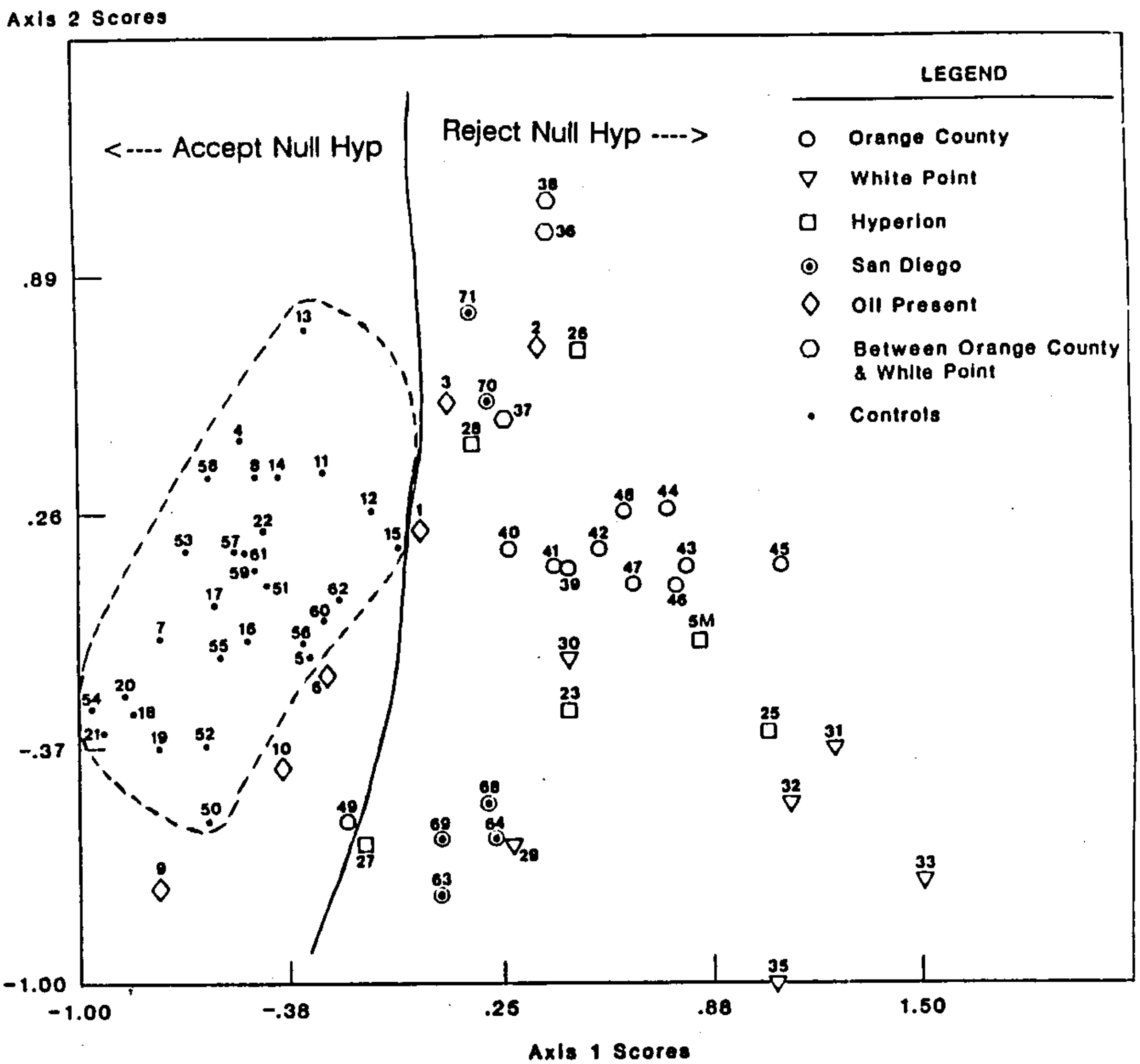


Figure 3-14. The first two ordination axes from analysis of the 60 Meter Control Survey benthic data. Null hypotheses accepted for all stations are to the left of the solid line, and ones rejected for all stations are to the right. The null hypothesis is that the Index 5 values are not different from the values within the reference envelope.

vs. 0.255, see above). This greater sensitivity and lower variability for Index 5 values lead to greater power when testing hypotheses. For a single species, we consider *A. urtica* to be quite a good indicator of outfall influences, and would expect most other potential indicator species to be even more variable and less sensitive. SAIC and EcoAnalysis (1984) compare the power associated with several potential indicator species, and show *A. urtica* abundances are generally associated with higher power in statistical tests (relative to other species).

3.5 *Discussion*

Current regulations require that beyond the "zone of initial dilution" (ZID) around an outfall, there must be a "balanced indigenous population" (BIP). Our interpretation of a "BIP" is that it represents the benthic community that would be present in the absence of the outfall in question. If we can assume that a community similar to that found within the reference envelope would be found in the absence of the outfall, then the proposed methodology should measure deviations from a BIP and therefore be a yardstick for compliance. As discussed below, this compliance should be based on a suite of indicator parameters rather than a single parameter.

The example analyses show that, at least in 1977 at 60 meters, if these methods were utilized with the two indicator parameters in a regional monitoring program to determine compliance, none of the major outfalls were in compliance. Since this time, most of the dischargers have been improving and continue to improve the quality of their effluent (as far as biological effects are concerned). As a consequence, some of the locations tested are and will continue to become closer to compliance. From this viewpoint, it will be useful to quantify how far out of compliance a station is, which will be the magnitude of the difference between the indicator value and the appropriate confidence bounds of the reference envelope.

It will be most informative and prudent to apply these methods to multiple indicator variables and examine the pattern of results (Chapman 1991). This approach should provide a perspective in which to judge the particular situation at each location. Environmental (abiotic) measurements at the stations should be included in this evaluation. The use of multiple indicators will also avoid overdependence on a single result that may be a chance event, as might be expected with multiple testing with several test stations. In our assessment of the results for multiple indicators we would give more weight to indicators that we knew were more sensitive and ecologically meaningful. In applying the recommended methods, it will be important to understand the relationships between external environmental factors and the indicator variables we choose to use (e.g., Figures 2-2, 2-4, and 2-7). If we do not account for the more important non-outfall factors such as causing variability of an indicator, the confidence limits can be relatively wide and the test of the null hypothesis can lack power. In addition, without this understanding, differences attributed to an outfall could be due to non-outfall factors. For example, if an indictor is very sensitive to sediment type, we would want to only compare a test station to stations within the reference envelope with a sediment type similar to the test station.

Our approach places emphasis on the distribution of indicator values in the reference area, making it imperative that there be sufficient sampling of reference areas. The current outfall benthic monitoring programs include stations only in the vicinity of the outfalls. The example results and other analyses indicate that at 60 meters, only a few of the stations in Santa Monica Bay (Hyperion monitoring program) are within our reference envelope. This small number of stations would be inadequate and not sufficiently diverse in sediment types for a regional monitoring program. Any future regional monitoring program should include increased sampling within reference areas. The data from these reference stations should be utilized by all dischargers in their analyses. In addition, the data from around the separate outfalls should available to all dischargers. This will enable analyses that include multiple levels of outfall effects. Such information would be especially useful for computing a measure like Index 5, which requires data from the more altered locations.

If the same data are to be used in the analyses for all the dischargers, the need for standardization of methods and taxonomy becomes critical. This should be an integral part of any regional monitoring plan (see Section 1).

The sampling within reference areas should emphasize areal and habitat-type coverage rather than replication at individual locations. Cuff and Coleman (1979) use optimization analysis to show that the most efficient use of resources when monitoring the level of an indicator in an area is usually to take single grabs at multiple locations rather than multiple grabs at fewer locations. This would be especially applicable to the present situation which uses the between-station indicator variability in the comparison of reference and affected areas. Thus, increased power and accuracy will mainly come from sampling more stations, not from replication at individual stations.

SECTION 4 - REGIONAL MONITORING RECOMMENDATIONS

This project has made valuable progress toward defining reference conditions in the benthos on a regional scale and toward developing an analytical approach for detecting deviations from these conditions. The insights gained from this work have clarified the additional issues that must be addressed to more completely develop the ability to define reference conditions and measure changes from these. Our recommendations are divided into six categories:

1. Make a commitment to a top-down approach to regional monitoring design in the Southern California Bight.

One of the major problems with the existing disjointed point source monitoring system is the difficulty in combining and integrating data from multiple monitoring programs. At the moment, there are two separate efforts underway to develop regional programs for Santa Monica Bay and the San Diego region. However, if these efforts are not integrated, and expanded to include the stretch of coast in between, they will merely recreate, at a larger scale, the current incompatibilities, and make it more difficult to retrofit programs for these subareas into the program for the entire region. Further, management questions and monitoring objectives should be clearly defined prior to developing the specifics of any monitoring program(s). Thus, we recommend that:

- regional monitoring be addressed in the context of the Southern California Bight as a whole.
- management questions for the bight as a whole, and for sub-regions within the bight, such as San Diego, be developed BEFORE attempting to design a monitoring program.
- monitoring objectives for the bight as a whole, and for sub-regions within the bight, be developed within the context of management questions. Use a of systematic approach such as that used for the Santa Monica Bay Restoration Project is recommended.
- a framework be constructed and used that will foster and support the cooperation needed to integrate the separate regional monitoring efforts.

2. Refine the proposed analytical technique to expand its applicability.

The proposed technique for detecting deviations from reference conditions was demonstrated with data from a single data set. Examples showed its utility using two different indicators of benthic conditions. However, in order to broaden its applicability and validate its use for compliance programs, several steps should be taken:

- investigate the use of the method with other ecological systems in addition to the benthos.
- determine the usefulness of the method in a situation where the influence of critical environmental gradients is more difficult to identify.
- develop more objective methods for defining the reference envelope. For the benthos, a wealth of background knowledge facilitated this process. However, this kind of information will not be available for all systems.
- extend the method for use in situations where varying numbers of replicate samples are available for stations within the sampling grid.
- further investigate the statistical properties of the approach to determine if there are any hidden pitfalls that need to be accounted for in its application.

3. Examine the properties of potential indicators in more depth to increase their utility.

Ultimately, the utility of any indicator depends on our ability to understand exactly what it tells us about the environment. For many of the traditional outfall indicators, such as *Amphiodia urtica*, our understanding is limited. At present, it is not always clear just which aspects of outfall-induced environmental changes cause changes in the indicators. For example, some of the data presented in Section 2 suggests that *A. urtica* responds to a complex interaction between depth and sediment type. As another example, research being carried out at SCCWRP suggests that different species respond differentially to distinct aspects of the changed environment around outfalls. Therefore, we recommend:

- additional analyses with existing data of the relationships among indicators and different aspects of the environment.
- laboratory investigations of the mechanisms underlying indicator responses.
- evaluation of additional potential indicators.
- investigation of how and whether using indicators in combination can reveal more about ecological changes.

4. Decide on the appropriate level of standardization for historical data.

Section 1 describes in detail the problems involved in combining data from benthic monitoring studies in the bight. Completely standardizing all historical data will be impossible. However, further standardization could be achieved and would result in important benefits. The level of standardization needed will depend on the kinds of questions the data will be used to answer. Thus, management and scientific questions should be clearly defined prior to any standardization effort. Some questions will be of more interest to managers, e.g., what were the visible changes associated with past improvements in discharge quality? Other questions will be of more interest to scientists or will be relevant to the specifics of monitoring design, e.g., what is the true amount of temporal variability in benthic communities?

We recommend that:

- managers and scientists decide on and prioritize a range of questions that could be addressed with historical data.
- knowledgeable scientists and data management experts estimate the amount of standardization effort involved in preparing the data needed for each question.
- resources be committed, in order of priority, to standardizing historical data.

5. Develop systems and procedures to ensure that all future data will be standardized.

In spite of the utility of existing efforts at data standardization (particularly SCAMIT), it is still difficult to combine data from different monitoring studies in the bight. This difficulty will only increase as efforts are made to develop truly regional datasets that combine data from different kinds of studies. The wide variety of research and monitoring efforts in the bight makes this a challenging goal to achieve. However, unless data and methods are truly standardized, it will continue to be impossible to efficiently make regional assessments. Therefore, we recommend that:

- working groups be set up to resolve inconsistencies among data and methods from different studies.
- managers at dischargers and other agencies that perform monitoring support these efforts by providing needed staff time and expertise.

- these efforts be carried out within the context of existing regional monitoring initiatives.

- regulatory agencies support these efforts by modifying the specifics of permits as needed to bring monitoring programs into accord with each other.

6. Design an efficient and effective monitoring program.

As documented in the National Research Council's examination of monitoring in the Bight (NRC 1990a), there are many ways in which monitoring's effectiveness could be increased. The existing programs were designed as point source monitoring efforts and the simple combination of all of these does not necessarily add up to an efficient regional program. In addition, there are many questions of sampling efficiency and sampling design that have never been adequately addressed in a regional context. Therefore, we recommend that:

- a regional monitoring design be developed, using the top-down approach recommended by the National Research Council in Managing Troubled Waters (NRC, 1990b).

- the statistical validity and efficiency of alternative designs be rigorously evaluated.

- historical data be used to develop estimates of spatial and temporal variability on scales that are relevant to regional monitoring.

- to the greatest extent possible, point source and regional monitoring requirements be balanced against each other in the same overall program.

- this design be developed within the context of ongoing regional monitoring efforts.

- regulatory agencies support these efforts by modifying the specifics of permits as needed to accommodate the requirements of the regional monitoring program.

SECTION 5 - APPENDICES

APPENDIX 2A: ORDINATION AS A METHOD OF DISPLAYING PATTERNS OF CHANGE IN COMMUNITY COMPOSITION

In this section, the concept of ordination is discussed only in sufficient detail for the understanding of the analyses utilizing ordination. We avoid presenting mathematical details that would detract from these main ideas.

The goal of the ordination analysis as used in this report is to concisely summarize patterns of community change among a set of samples in which benthic species were identified and enumerated. This technique will usually be sensitive and robust since the computations utilize quantitative information from multiple species.

The results of an ordination analysis are usually displayed as a bivariate plot containing points representing samples (e.g., stations at which benthic species were identified and enumerated). The relative positions of the sample points in the plot are consistent with the biological community differences among the samples. Thus, samples with very similar communities should be relatively close together in the plot, and samples with very different communities should be relatively distant in the plot. In reality, ordination results can involve more than two dimensions or axes, so the results may need to be displayed in multiple bivariate plots displaying different combinations of dimensions.

Ordination analysis can be a very useful tool for summarizing patterns of community change in a set of sampling stations, which are represented as points in a multidimensional ordination space. The distances between the points in the space are proportional to the community differences among the corresponding stations.

Before further explanation of ordination, we will discuss the dissimilarity index, which is usually the input into the ordination computations. The relationship between the dissimilarity index values and the ordination results are then demonstrated with an example.

Dissimilarity index

A dissimilarity index is used to compare the communities found at a pair of stations. The higher the dissimilarity index value, the more dissimilar the communities found at the two stations being compared.

The index computations are based on the species and their abundances at the stations. The dissimilarity index values for multiple stations are usually displayed in the dissimilarity matrix, which contains the dissimilarity index values for all pairs of stations. There are several potential dissimilarity indices that can be used with community data. The ordination analyses in Sections 2 and 3 utilized the Bray-Curtis dissimilarity index (Bray and Curtis 1957, Smith 1976, Smith *et al.* 1988, Clifford and Stephenson 1975, Boesch 1977). This index ranges from a value of zero (for two identical samples) to a value of one (for samples with no species in common).

Dissimilarity Index Values and Ordination

The concept of ordination is illustrated with a simplified hypothetical example. Figure 2A-1 shows an area where the benthos is sampled at five station locations indicated as 1-5 in the figure. There are only two environmental factors that will affect the community composition at a station, mainly depth and organic enrichment from the outfall. The benthic community changes constantly with depth, and near the outfall it changes with distance from the outfall. The contour lines around the outfall approximate the community changes due to the outfall, with the greatest community change nearest the outfall, and no community change beyond the outer contour.

First, let us assume that only stations 1-3 are sampled. Since none of these stations are near the outfall, we are not yet dealing with outfall effects. The dissimilarity matrix from the benthic data at these stations might appear as shown in Table 2A-1.

The dissimilarity matrix is consistent with constant community change with depth. The difference in depth between stations 1 and 3 is about twice the difference in depth between stations 1 and 2. Similarly, the dissimilarity of communities at stations 1 and 3 (0.4) is double the dissimilarity of communities at stations 1 and 2 (0.2). This dissimilarity matrix could be represented in an ordination space as shown in Figure 2A-2. Note that the dissimilarities match the distances among the corresponding stations in the ordination space, i.e., the distance in the ordination space between stations 1 and 3 is double the distance between station 1 and 2 (as is the case with the dissimilarities). This shows that stations sampled along a single environmental gradient that causes community change will be found approximately along a straight line in the ordination space. When we use all five stations, the dissimilarity matrix might appear as in Table 2A-2.

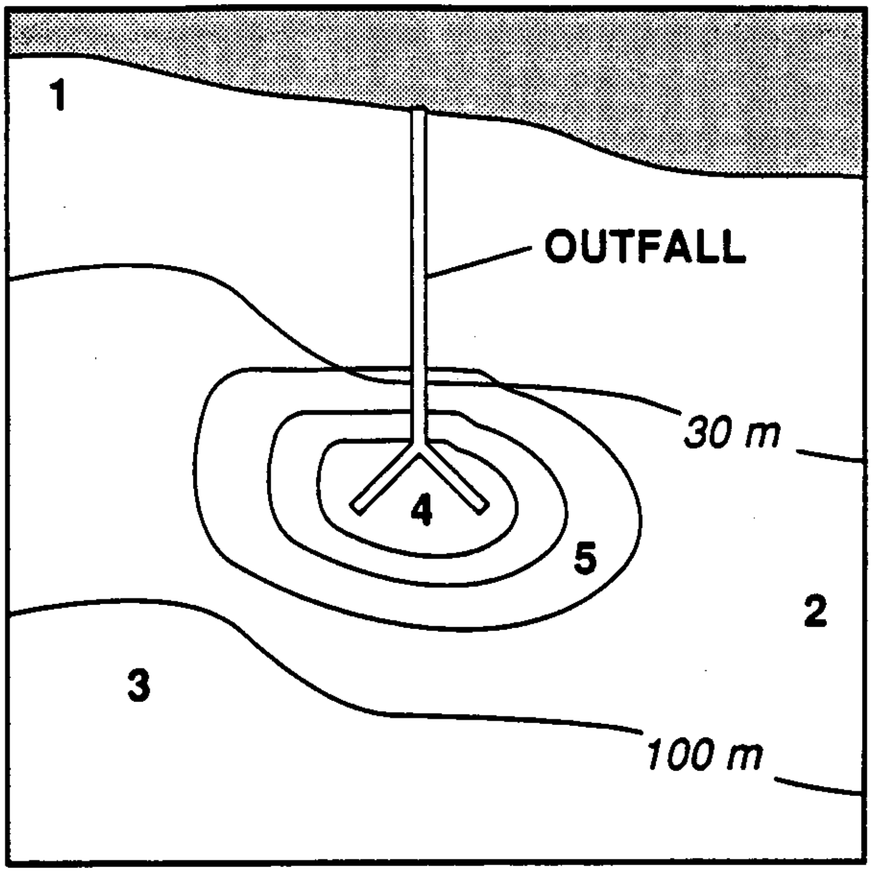


Figure 2A-1. Hypothetical example of a benthic survey with five stations taken at different depths and distances from an outfall.

Table 2A-1

DISSIMILARITY MATRIX FOR STATIONS 1-3

	Station 1	Station 2	Station 3
Station 1	0.0	0.2	0.4
Station 2	0.2	0.0	0.2
Station 3	0.4	0.2	0.0

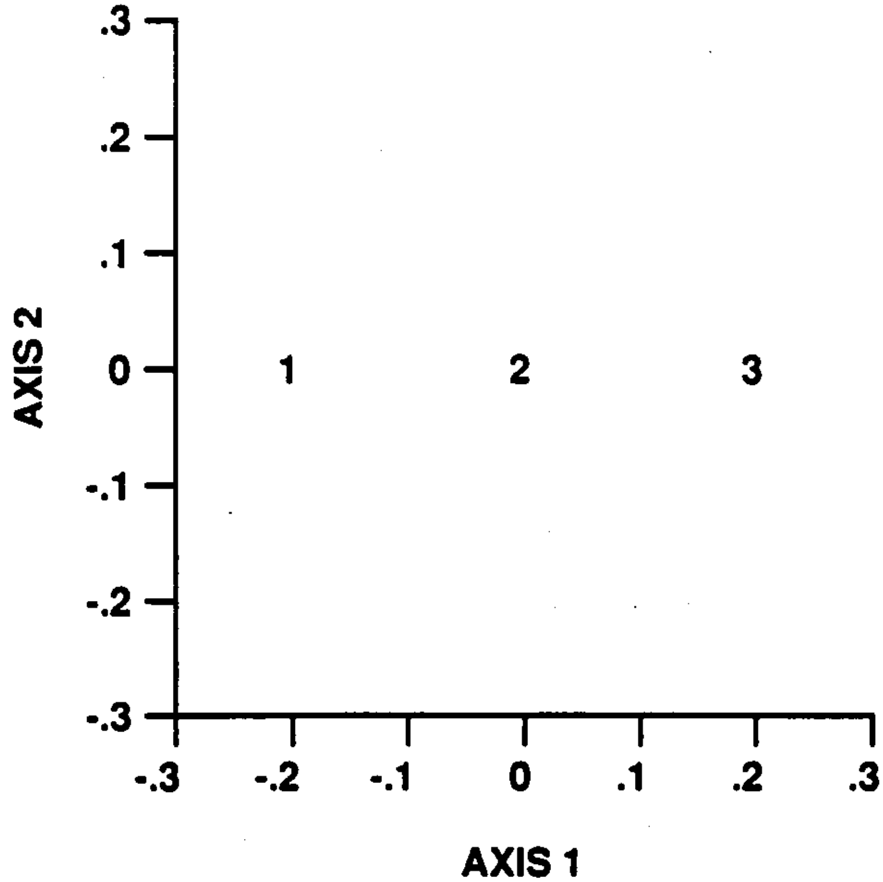


Figure 2A-2. Ordination space with stations 1-3, created from the dissimilarity matrix in Table 2A-1.

Table 2A-2

DISSIMILARITY MATRIX OR STATIONS 1-5

	1	2	3	4	5
Station 1	0.00	0.20	0.40	0.36	0.2
Station 2	0.20	0.00	0.20	0.30	0.14
Station 3	0.40	0.20	0.00	0.36	0.24
Station 4	0.36	0.30	0.36	0.00	0.16
Station 5	0.24	0.14	0.24	0.16	0.00

This dissimilarity matrix is consistent with constant change with depth and changes with distance from the outfall. The dissimilarities between outfall Stations 4 and 5 and the remaining stations 1-3 are greater than that expected from depth changes alone. These dissimilarities could be represented in an ordination space as shown in Figure 2A-3. The distances in the ordination between stations in the space are consistent with the dissimilarities in Table 2A-2. The effect of the outfall is seen by the positions of Stations 4 and 5, which are off in a direction approximately at right angles to the "line" represented by Stations 1-3 (representing community changes with depth).

Note that in Figure 2A-3, the projections of all stations onto axis 1, including the outfall stations, are consistent with the depth of the stations. For example, Stations 2, 4, and 5 are all at about the same depth and all project to about a value of zero on axis 1. Similarily, note that all projections of the stations onto axis 2 are consistent with outfall effects. For example, Station 4, the most affected station, has the highest projection (about 0.20), and stations 1-3, which are not affected at all by the outfall, have the lowest projections (about -0.1). This illustrates a very convenient feature often associated with ordination analyses. Patterns of community change associated with different environmental gradients (spatial or temporal) in the area sampled can be represented by point (station) patterns going off in different directions of the ordination spaces. This allows the analyst to study and differentiate community change due to different environmental causes. The value of this should be evident in the ordination results and their usage shown in Sections 2 and 3.

It should be noted that not all environmental factors causing community change in the data will be represented by changes in different directions of the ordination space. The environmental factors that will correlate with community changes represented in the ordination space in this manner will have to operate somewhat independently of other environmental factors causing major community changes. For example, in Figure 2A-1, depth and outfall effects are somewhat independent (uncorrelated) in the stations sampled. Stations at outfall depth could be highly affected by the outfall or not affected at all. In contrast, if changes in sediment size and water temperature both have a major impact on the benthic community, and both sediment size and water temperature were strongly correlated in the stations sampled (as would be expected with sampling over a large depth range), we could not distinguish patterns of community change in the ordination space due to these two environmental gradients.

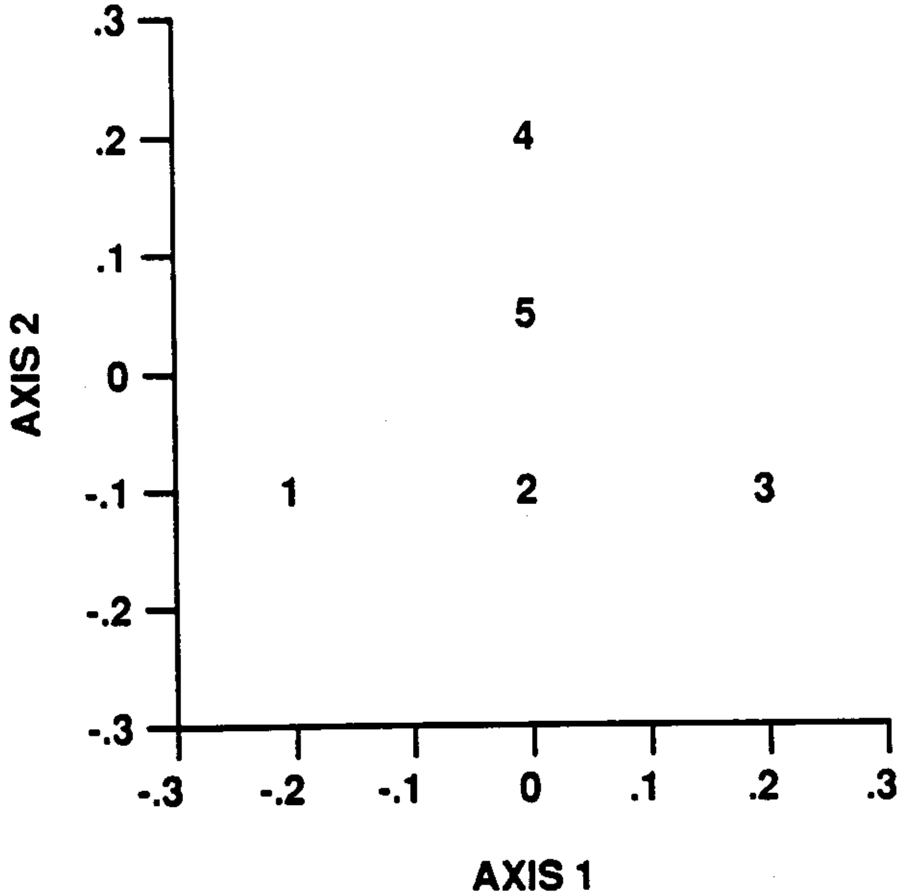


Figure 2A-3. Ordination space with stations 1-5, created from the dissimilarity matrix in Table 2A-2.

Ordination results consist of projections of the station points onto the various axes (or dimensions) of the ordination space. These projections are called scores, which can be plotted for interpretation (e.g., Figures 2A-2 and 2A-3). Each station will have a score on each ordination axis. The number of useable ordination axes in the space will depend on the number of independent trends of community change that exist in the data. The ordination axes are usually ordered to correspond to the "strength" of the various trends of community change in the data. Stronger trends will be associated with directions in the ordination space where the points are more spread out (i.e., corresponding to greater dissimilarities between points). Thus, axis 1 will represent the strongest pattern of community change in the data. Since the scores on the different axes are independent (uncorrelated), axis 2 will represent the strongest pattern of community change that is independent of that represented by axis 1, and so on for the remaining axes.

In reality there are multiple ordination techniques that can utilize a dissimilarity matrix to produce an ordination space. There are also important data manipulations that are often performed in an attempt to maximize the correspondence between the dissimilarity values and actual community changes. A more complete discussion of this topic can be found in Smith and Bernstein (1985), and Smith *et al.* (1988).

Methodological details for the ordination analyses presented in Sections 2 and 3 are as follows. Prior to the computation of the dissimilarity index values, the species abundance data at the stations were transformed by a square root and standardized by the species mean (of abundance values > 0). Dissimilarity index values > 0.80 were reestimated with the step-across method (Williamson 1978, Smith 1984, Bradfield and Kenkel 1987). The dissimilarity index values were then utilized with an ordination technique called local nonmetric multidimensional scaling (Sibson 1972, Prentice 1977, 1980).

APPENDIX 3A: THE COMPUTATIONS FOR THE CONFIDENCE BOUNDS USED TO TEST THE NULL HYPOTHESIS

If the distribution of station indicator values is normal, and we are interested in the null hypothesis is that an indicator value at the test station is not *lower* than that found in the reference area, then the 100(1-α) percent lower bound for 100*p*th percentile of the distribution of station values is

$$\hat{u} - sg'(1-\alpha;1-p,n), \tag{1}$$

where $\hat{u}$ is the mean indicator value in the reference area, s is the between-station standard deviation of indicator values, n is the number of stations in the reference area, α is the type-1 error level of the confidence bound, and $g'(a;b,c)$ is a value from Table 1 in Odeh and Owen (1980), Table A.12 in Hahn and Meeker (1991), or Table A3 in Gilbert (1987). See Hahn and Meeker (1991) and Gilbert (1987) for a discussion on statistical confidence intervals. Confidence bounds computed in this manner are equivalent to "tolerance intervals" (Hahn and Meeker 1991).

If we are interested in the null hypothesis that the indicator value at the test station is not *higher* than that found in the reference area, then the 100(1-α)% upper bound for *p*th percentile of the distribution of station values is

$$\hat{u} + sg'(1-\alpha;p,n). \tag{2}$$

For example, we have 5 reference stations with a mean value of 50.1 for Indicator I, a between-station standard deviation of 1.31, $\alpha=0.05$, $p=.10$, and $g'(.95;.90,5)=3.407$. Thus, the lower bound of the 10th percentile of the reference-station distribution of Indicator I values is 50.1 - (3.407)1.31 = 45.64 .

In this example, Indicator I would be a parameter, such as the abundance of *Amphiodia urtica,* that would be expected to decrease if affected by an outfall. If the indicator value at a test station fell below 45.64, we would conclude that there may be some outfall effect. The actual 10th percentile of this distribution is 48.42, but because of uncertainty in our estimates of the mean and standard deviation, the lower confidence bounds for this sample is adjusted to a lower number so that we will fail to bracket the real 10th percentile only 5%

of the time ($\alpha = .05$). The choice of p indicates how far into the lower tail of the normal distribution we wish to go before we reject our null hypothesis. This is a standard that would have to be set *a priori*.

Before using this technique, the distribution of indicator values in the reference area should be checked for deviations from normality. Most statistical software systems include tests for normality with graphical diagnostics (e.g., SAS Univariate Procedure, SAS 1990). If the distribution appears to be non-normal, the Box-Cox family of transformations could be applied to the data to find an optimal transformation to normality (Box and Cox 1964, Madansky 1988). Once the confidence bound is computed with the transformed data, it must be transformed back to the original units.

Alternately, distribution-free confidence bounds can be computed by methods described in Conover (1980), Gilbert (1987) and Hahn and Meeker (1991). Unfortunately, if p is relatively small (close to zero) or large (close to 1), then a large sample size (n) is required to compute these bounds. For example, if $p = .01$, over 100 reference station values would be required.

Controlling for non-outfall factors

Indicator variables can be affected by habitat or temporal differences such as sediment grain size, organic content, or depth that are unrelated to outfall effects. To avoid confusing such natural factors with outfall effects, it will be important to control for them in the analytical design. The simplest way to control for an effect is to hold it relatively constant in the data utilized to compute the confidence bounds. For example, the benthic communities are known to change considerably with depth, especially within the range of shelf depths that we are concerned with here. To control for depth, we could utilize data within a restricted depth range that corresponds to a test station. The same could be done for sediment types and time periods. The factors that must be controlled for can differ with different indicators.

A disadvantage of utilizing these restricted subsets of the data is that the sample size (n) gets smaller and the confidence bounds expand (giving lower power) as we eliminate data from the computations. One possible approach to retain more data would be to use analysis of covariance to mathematically adjust the data for differences in the pertinent non-outfall factors.

Another, much safer approach would be to estimate a pooled standard deviation (s) from within strata of stations defined by ranges of the variables that we are controlling for. The pooled standard deviation is computed as

$$s = [(\sum_{i=1}^{g} \sum_{j=1}^{m_i} (X_{ij}-\hat{u}_i))/\sum_{i=1}^{g} (m_i-1)]^{1/2}, \tag{3}$$

where x_{ij} is the indicator value at station j in stratum i, m_i is the number of stations in stratum i, $\hat{u}_i$ is the mean for stratum i, and g is the number of strata. This method would assume that the variances within the strata were homogeneous and that the within-stratum distribution of values was normal. Brown and Forsythe (1974) and Conover *et al.* (1981) discuss tests for equality of variance. Often transformations to normality also improve the homogeneity of variance (Box and Cox 1964). The mean ($\hat{u}$) used in formulae (2) and (3) for the confidence bounds would be that from the reference envelope strata matching the test station, and for use in equations (1) and (2),

$$n = \sum_{i=1}^{g} (m_i-1)-1, \tag{4}$$

where g is the number of strata, and m_i is the number of stations in stratum i. This expression for n was verified with Monte Carlo simulations. If some of the strata have significantly different variance (even after transformation) from the stratum matching the test station, then the offending strata should be left out of the pooling procedure.

An example will show the extra power gained from the pooling the variances from the different strata. Table 3A-1 shows indicator values from ten stations within the reference envelope. These stations are located within two depth ranges that are relatively homogeneous for this indicator. We have a test station that is found in the 50-70 meter range with an indicator value of 8.

First, we pool the variance within the depth strata and compute the lower confidence bound ($p=.10, \alpha=.05$) for the 50-70 meter stratum, which matches the depth range of our test station. Here the pooled $s=1.58$ (equation 3), $n=7$ (equation 4), and $g'(.95;.90,7)=2.755$. The lower confidence bound for the 10th

Table 3A-1.

INDICATOR VALUES AT TEN REFERENCE STATIONS.

Depth Range	
20-40M	50-70M
4	12
2	14
1	13
3	11
5	15

percentile is 13 - 2.755(1.58) = 8.647, using equation (1). Since the value of the indicator at the test station (8.000) is less than 8.647, we would reject the null hypothesis.

If we had not pooled the data from the two strata. The appropriate parameter values would be $s=1.58$, $n=5$, and $g'(.95;.90,5)=3.407$. The lower confidence bound for the 10th percentile is 13 - 3.407(1.58) = 7.617. In this case, the indicator value at the test station is greater than the lower bound and the null hypothesis would be accepted. The pooling allowed for a higher value of n and a more powerful test.

Assumptions

The assumption of normality for the indicator values at the reference stations has been discussed above. It is also assumed that the reference stations are located randomly in space (within reference areas). In Southern California, benthic programs most often employ a somewhat systematic sampling design. This should not be a problem if within the reference envelope, the following are approximately satisfied (Gilbert 1987).

1) There are no trends in the indicator value within and between the reference areas within the reference envelope.

2) There are no natural strata where the indicator values are locally elevated or depressed.

3) Indicator values are uncorrelated among the stations within the reference envelope.

When these assumptions are satisfied, we say that the population is "in random order". As noted above, we expect to encounter strata within the reference envelope, but we will only be comparing the test stations to a matching stratum that has no internal substrata, and will be computing the variation from within strata. Thus, we will satisfy second condition stated above.

In addition, it is assumed that there are no pertinent spatial periodicities in the indicator values. For example, if the indicator value peaked every ten miles along the coast, and were lower in between, systematic sampling stations ten miles apart would hit all low or all high indicator values. This situation would lead to serious biases in the mean and variances of the indicator value. Fortunately, such confounding periodicities are highly unlikely with the present application.

APPENDIX 3B: ABUNDANCES OF *Amphiodia urtica* IN SCCWRP 60 METER SURVEY

Station	Count	Station	Count	Station	Count	Station	Count
01	31	18	146	36	0	53	348
02	22	19	149	37	5	54	228
03	102	20	66	38	0	55	223
04	44	21	78	39	52	56	363
05	113	22	118	40	24	57	132
06	65	23	3	41	11	58	174
07	33	25	0	42	1	59	203
08	83	5M	0	43	0	60	182
09	33	26	1	44	2	61	117
10	30	27	26	45	0	62	180
11	97	28	8	46	0	63	16
12	41	29	1	47	1	64	2
13	42	30	0	48	1	68	16
14	89	31	0	49	36	69	56
15	89	32	0	50	148	70	38
16	161	33	0	51	207	71	1
17	201	35	0	52	258		

SECTION 6 - REFERENCES

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