CONTAMINANT RESIDENCE TIMES AND ECOSYSTEM RECOVERY RATES FOR SHELF AND ESTUARINE ECOSYSTEMS

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TABLE OF CONTENTS

TABLE OF CONTENTS (continued)

TABLE OF CONTENTS (continued)

LIST OF TABLES

LIST OF TABLES (continued)

LIST OF TABLES (continued)

LIST OF TABLES (continued)

LIST OF TABLES (continued)

LIST OF TABLES (continued)

LIST OF FIGURES

LIST OF FIGURES (continued)

LIST OF FIGURES (continued)

1. ABSTRACT

A study has been undertaken to place the concepts of Contaminant Residence Times and Ecosystem Recovery Rates into an overall framework suitable for assessing the impacts of various waste disposal strategies on coastal marine environments.

Residence time estimates for selected toxic organic compound classes and metals have been obtained using a computational model based on the definition of Residence Time (T) as:

$$T = \frac{A_x}{(dA_x/dt)}$$

or the inventory of a contaminant (A_x) divided by is mass rate of input to or removal (dA_x/dt) from an environmental compartment. The compartments modeled are the well-mixed water column and the sediment mixed layer. Calculations of (T) values were based on empirical data of ambient contaminant concentrations, and mass loading and removal terms. Five geographic areas differing in depths, pollutant loadings, and energy regimes were selected for study: Puget Sound Main or Central Basin, the Christiaensen Basin of the New York Bight Apex, the Lower Bay Complex of the Hudson-Raritan Estuary, the Palos Verdes Shelf Area of the Southern California Bight, and the Philadelphia Sewage Sludge Dumpsite. Residence time calculations were assessed for each system based on the variability in existing data on contaminant concentrations and on generic parameter (e.g., resuspension and sedimentation rates) values. Hypothetical loadings were computationally administered to each area to illustrate the utility of the modeling approach to address various waste disposal strategy alternatives.

A thorough review of existing studies relating pollutant loadings to extents of system perturbation and recovery time estimates was undertaken, considering benthic community ecosystems and contaminant body burden approaches. Based on these studies, recovery times were estimated for a variety of community types.

A conceptual framework relating the residence time of contaminants in sediments to these recovery rates was developed and focused on several scientific and scientific-regulatory approaches to linking the Residence Time and Recovery Estimates Segments.

The power of the modeling approach to identify critical data gaps and to guide future efficient research and monitoring studies was discussed as a main beneficial result of the study.

2. INTRODUCTION

2.1 Study Setting

The concepts of pollutant residence times and ecosystem recovery are closely interwoven. If a given pollutant input results in a biotic exposure which is sufficient to perturb an ecosystem, then this perturbation will persist until the forcing pollutant input is decreased in magnitude and duration or eliminated. Prolonged perturbation may result in ecosystem degradation. A system in chemical steady-state, that is one in which pollutant inputs equal pollutant outputs and in which concentrations are constant over time, will tend to remain a degraded state as well, if these ambient concentrations are above a threshold for system perturbation. If polluted inputs decrease, natural physical processes of recovery will begin and continue until the ambient pollutant concentrations are lowered. If these ambient concentrations meet the "criteria" for biological recovery then the onset of biological recovery may begin.

These are, of course, oversimplified generalizations, as specific numerical criteria for "perturbation", "degradation", and "recovery" are not clearly defined in marine science. A rigorous treatment of these topics demands careful definitions of these terms. The causal mechanisms for marine environmental "perturbations" are not well understood. If one considers, for example, the input of toxic organic compounds (e.g., polychlorinated biphenyls) or heavy metals (e.g., cadmium) to sediments adjacent to the sewage sludge dumpsite in the New York Bight Apex, can one attribute the documented environmental degradation of the benthos specifically to these pollutant inputs or to synergistic effects of combined pollutant inputs? Or, instead, are ecological perturbations due to habitat destruction resulting from sediment texture changes? Perhaps increased oxygen demand in the benthos due to non-toxic organic loadings has stressed the benthic ecosystem through oxygen depletion and/or hydrogen sulfide production to the point of observable ecosystem perturbation. Has increased nutrient regeneration from the benthos resulted in biological speciation changes in the overlying water or enhanced local productivity which, in turn, has resulted in anoxic or hypoxic conditions? In considering most coastal ecosystems, these answers remain for the most part, elusive.

In order to begin to understand linkages between 1) the concentration of pollutants and their persistence in a system, and 2) the perturbation and recovery of stressed and degraded ecosystems, one must first examine what is known about each separately.

2.2 Research Objectives and Study Design

A detailed study was undertaken to: 1) develop and apply a computational model to determine pollutant residence times in coastal environments, and 2) to evaluate the state of knowledge regarding the rates of the recovery of ecosystems in marine systems, after being perturbed by pollutant inputs. Studies of contaminant residence times in ecosystem compartments, and of the biological and chemical recovery and recoverability of these ecosystems are at the very heart of marine pollution research. Unfortunately, knowledge of how these complex processes operate and interact is quite poor.

The plan of execution of this program was to apply these concepts to detailed consideration of several important and varied marine environments in U.S. coastal waters. The overall design of the study was to conduct the above elements 1) and 2) of the study independently, until the elements could be conceptually merged in a synthesis activity.

The overall objectives of the study were then to:

1. Establish a computational model for determining the residence time of pollutants in the water column and in bottom sediments.

2. Develop residence time estimates for a number of representative pollutant classes in several selected estuarine and coastal marine systems.

3. Establish the computational model as a conceptual tool useful to explore various waste disposal strategies or hypothetical changes in pollutant loadings, in specific systems.

4. Thoroughly review, evaluate and summarize existing information on ecosystem recovery rates.

5. Conceptually link information on ecosystem recovery rates to pollutant residence time concepts, so as to allow for the use of the program's information in a number of hypothetical scenarios regarding waste input rates into marine systems.

Our overall technical approaches to above objectives 1 through 3 were based on a philosophy of going into a level of detail empirically justified by the data for each system, ecosystem component, and targeted pollutant. The development of complex theoretical models based on mathematical formulations was not our intent in this program. Our approach was to apply, where possible, empirically-derived information to the generation and application of computational models of pollutant residence times, and to relate these to ecosystem recovery concerns.

Our focus in the Contaminant Residence Time study was on the formulation of simple box models for the water column and sediment which:

1. Define for each system those physical, sedimentological, chemical and biological parameters needed to assess the mass rate of transfer, or time-integrated mass budgets of targeted pollutant classes into and/or out of the system or subsystem.

2. Are based on literature or estimated parameter values. The level of detail available from these literature values, in combination with parameter values assigned on the basis of intelligent estimates or extrapolated values, define the level of detail of the resultant box model formulated for each system. In developing and applying the model, we focus on the following questions: Are there sufficient data available to establish multicompartment water column models (i.e., surface water and bottom water) or will the box models include a single water column component linked to a single sediment component? On what level of detail can pollutant residence times be computed from these box models?

   Box model formulations were considered for establishing residence time estimates based on existing data. Application of these models was intended to define information gaps and data acquisition needs as well.

3. Provide a useable, valid, simplified schematic view of each ecosystem which can then be used for strategic planning for waste disposal, waste treatment and pollutant introduction strategies. The models were geared to provide residence time estimates for each targeted pollutant class in each environmental compartment (water column, sediment).

4. Enable estimates of environmental compartment recovery to be determined based on biological criteria established in the recovery time estimation part of the study.

Our technical approach to the Recovery Time Estimation study was to:

1. Define the concept of ecological recovery.

2. Obtain and synthesize existing information on various classifications of biological and chemical "recovery" of ecosystem compartments.

3. Estimate recovery times based on a series of well-studied chemical, biological and other measures.

4. Evaluate these recovery time estimates for the geographic area studied by considering differences and similarities between study areas, and by considering pollutant residence times estimated for each compartment within each study area as determined from the residence time study.

Once treated and presented separately, an approach to merging the topics was to be attempted, limited by our state of knowledge of the interaction of chemicals and biotal perturbations and recovery.

3. RESIDENCE TIME STUDY

3.1 Overview of Mass Balance Concepts

Several extensive studies have been conducted in recent years concerning the modeling of the fate of toxic substances in natural systems. Recent advances in these modeling effects are summarized in Dickson et al. (1982).

The most widely used and most useful concept in fate modeling studies is that of the system as a three dimensional box or a set of boxes corresponding to system components (e.g., water column, sediment) to be evaluated. These box "models" of actual system components then serve as useful, simplified conceptualizations of marine systems and can be used to inventory materials passing through or accumulating in them. Such inventories must consider the mechanisms of input, output, production, accumulation and depletion of substances, and the masses of materials associated with each mechanism. Thus, these mass or material balances represent an expression of the law of conservation of mass in accounting terms. The general principle of mass balance calculations is to establish a number of independent equations which consist of terms associated with mechanisms of input, output, accumulation, production and/or depletion of each accounting unit (e.g., amount or flux of sediment; amount or flux of PCB or metals).

Detailed mass balance box model approaches to studies of the fates of toxic substances are illustrated by the considerable efforts conducted in studies of the Great Lakes (e.g., DiToro et al., 1982; Thomann and DiToro, 1983). Thomann and DiToro (1983) conducted an extensive study of the fundamentals of the fates and fluxes of toxic substances in the Great Lakes. The authors developed a detailed physico-chemical model from mass balance principles incorporating mechanisms of particulate sorption-desorption, sediment-water and atmospheric-water interactions, and chemical and biochemical decay. Their focus was to apply a fate model, calibrated from ^{239}Pu data, to PCB and Cd transport. The general features of their detailed system model include:

1. Sorption-desorption mechanisms of the chemical with suspended particulates in the water column;
2. Similar mechanisms with solids in the sediments;
3. Loss of the chemical due to mechanisms such as biodegradation, volatilization, chemical and biochemical reactions, photolysis;
4. Transport of the toxicant due to advective flow transport, and dispersion and mixing;
5. Settling and resuspension mechanisms between sediment and water column;
6. Direct inclusion of external inputs that may be subject to environmental control.

The authors developed comprehensive mass balance equations for dissolved, particulate, water column and sediment segments. These equations can be greatly simplified if the system is considered to be completely mixed, all components being in equilibrium, and if rates of component decay, rates of diffusive pore water flux, and volatilization rates are small compared to the physical movement of materials within and between system components for a single completely mixed body of water.

3.2 Concepts of Pollutant Residence Times

In classic chemical oceanographic terms, the concept of the residence time of a chemical is defined as:

$$T = \frac{A_x}{(dA_x/dt)} \qquad (1)$$

where:

T = Residence time

A_x = Amount of chemical x in a given environmental compartment (inventory)

$\left(\frac{dA_x}{dt}\right)$ = Rate at which x is added to or removed from the compartment

and A_x is defined as:

$$A_x = C_x V$$

where:

C_x = Concentration of x in the environmental compartment

V = Volume of the compartment

and:

$$\left(\frac{dA_x}{dt}\right) = C_{x_1} Q_1 + C_{x_2} Q_2 + \dots C_{x_n} Q_n$$

$$= \sum (C_{x_i} Q_i)$$

where:

$\sum C_{x_i} Q_i$ = Sum of all individual flux terms which equal the total flux of contaminant x into or out of the compartment

The residence time of material in environmental compartment at steady-state is considered to be the average time that a unit of that material (i.e., a particulate or dissolved component in seawater, or a unit of seawater itself) resides in the compartment before it is removed by one or a combination of mechanisms (i.e., advection, settling, decay, volatilization for the water column; burial, resuspension and transport, diffusive flux, degradation and decay, biological ingestion for the sediment column). The formulation of residence time assumes that pollutant x is at **steady-state** and that the rates of addition to the compartment (mass loading) and removal from the compartment are equal. The residence time equation for mass loading thus becomes:

$$\left(\frac{dA_x}{dt}\right)_{in} = \sum (C_{x_i} Q_i)_{in} \tag{2}$$

and for removal:

$$\left(\frac{dA_x}{dt}\right)_{out} = \sum (C_{x_i} Q_i)_{out} \tag{3}$$

Assuming x is at steady-state, then:

$$\sum (C_{x_i} Q_i)_{in} = \sum (C_{x_i} Q_i)_{out}$$

Clearly, steady-state conditions apply to systems in equilibrium (e.g., the chemical composition of the world's ocean) and not to systems in the process of being degraded by introduction of pollutants or, for that matter, in the process of recovery. In these cases, pollutant inputs and outflows by combined physical, chemical, and biotransformative processes may not be in balance. However, for the purposes of establishing residence time estimates from conceptual process models, one must assume that a steady-state condition prevails. If the assumption holds for a given system, then (T) computed from input items should equal (T) computed from outflow terms. Furthermore, the concept of a residence time assumes that the pollutant is well mixed in the environmental compartment in a time appreciably shorter than the residence time and that the amount of component x, and the input or output of x does not change in 3-4 times this time period (Riley and Chester, 1971).

In addition to physical factors that add or remove materials (e.g., pollutants) from a compartment, other factors may control the persistence of pollutants in a particular system. Such control processes may include chemical and biodegradative transformations, including the partitioning between dissolved and particulate phases (e.g., pore water flux out of sediments) and the decay or degradation in each environmental compartment. The transformation processes affecting the ultimate concentrations and residence times of the pollutant, together with the source functions or the pollutant's mode, and rate of entry and exit into and from the system, define the conceptual models that must be established for each pollutant in each system.

3.3 Simplified Modeling Equations and Generic Parameters

Based on the preceeding principles and discussions we have applied the fundamental definition of residence time (T) in equation (1) to formulate simplified equations for fluxes in mass loading and removal processes for a single well-mixed water column and a single well-mixed sediment layer.

The use of these equations to calculate $(T)_{in}$ and $(T)_{out}$ values is rooted in the three basic assumptions discussed above.

3.3.1 Sediment Equations

3.3.1.1 Mass Loading

For the sediments, the final form of the mass loading expression is:

$$\left(\frac{dA_x}{dt}\right)_{in} = \left[C_{x(tsm)}\right]_{in} Q_{sed} + C_{x(od)}\ Q_{od}\ f + P_x \qquad (4)$$

where:

$\left[C_{x(tsm)}\right]_{in}$ = Concentration of x on particulate matter entering the sediment box compartment ($\mu g\ g^{-1}$ of tsm)

Q_{sed} = Sedimentation rate ($g\ cm^{-2}\ day^{-1}$)

$C_{x(od)}$ = Concentration of x on dumped or discharged material ($\mu g\ g^{-1}$ or $\mu g\ L^{-1}$)

Q_{od} = Rate of dumping ($g\ day^{-1}$ or $L\ day^{-1}$)

f = Fraction of dumped material settling on the bottom

P_x = Production of x within the sediment ($\mu g\ day^{-1}$)

3.3.1.2 Removal Processes

The outflux term equation for the sediments is:

$$\left(\frac{dA_x}{dt}\right)_{out} = C_{x(sed)}\ Q_b\ A_s + C_{x(sed)}\ Q_{rs}\ A_s + C_{x(pore)}\ Q_{pore}\ A_s + Z \qquad (5)$$

where:

$C_{x(sed)}$ = Concentration of x on surface sediment particles ($\mu g\ g^{-1}$)

Q_b = Burial rate ($g\ cm^{-2}\ day$)

A_s = Surface area of water column compartment (cm^2)
= Surface area of sediment compartment (cm^2)

Q_{rs} = Resuspension rate (μg sediment $cm^{-2}\ day^{-1}$)

$C_{x(pore)}$ = Concentration of x in sediment pore water within the mixed sediment layer ($\mu g\ L^{-1}$)

Q_{pore} = Flux of pore water (L cm^{-2} day^{-1})

Z = Decay rate due to degradation (μg day^{-1})

The above equations represent the simplified "model", essentially a computational model, which forms the basis of the contaminant residence time determinations presented in Sections 3.6 through 3.10 of this report. The specific application of this model to each system will be addressed in these sections.

3.3.2 Water Column Equations

3.3.2.1 Mass Loading

Using equation (2), the final form of the water column expression for mass loading becomes:

$$\left(\frac{dA_x}{dt}\right)_{in} = \left[C_{x(dis)}\right]_{in} Q_w + \left[C_{x(part)}\right]_{in} Q_w + Q_{x(dep)} A_s + C_{x(pore)} Q_{pore} A_s + C_{x(sed)} Q_{rs} A_s + C_{x(od)} Q_{od} \qquad (6)$$

where:

$\left[C_{x(dis)}\right]_{in}$ = Dissolved concentration of x entering the water column compartment (μg L^{-1})

Q_w = Advective flux of seawater (L day^{-1})

$\left[C_{x(part)}\right]_{in}$ = Concentration of particulate x entering the water column compartment (μg L^{-1})

$Q_{x(dep)}$ = Depositional flux of x on atmospheric particles or in rainout (μg cm^{-2} day^{-1})

3.3.2.2 Removal Processes

Based on equation (3) the expression describing the removal processes or the output flux term for the water column is:

$$\left(\frac{dA_x}{dt}\right)_{out} = \left[C_{x(dis)}\right]_{out} Q_w + \left[C_{x(part)}\right]_{out} Q_w + \left[C_{x(tsm)}\right]_{out} Q_{sed} A_s + Z \tag{7}$$

where:

$\left[C_{x(dis)}\right]_{out}$ = Dissolved concentration of x leaving the water column compartment ($\mu g\ L^{-1}$)

$\left[C_{x(part)}\right]_{out}$ = Concentration of particulate x leaving the water column compartment ($\mu g\ L^{-1}$)

$\left[C_{x(tsm)}\right]_{out}$ = Concentration of x on particulate matter leaving the water column compartment ($\mu g\ g^{-1}$ of tsm)

3.4 Pollutants Selected for Study

After consideration of a number of pollutant classes to be studied, a final set of known toxic organic compounds and heavy metals were selected for scrutiny. The toxic organic classes studied were the polychlorinated biphenyls (PCB) and polynuclear aromatic hydrocarbons (PAH), compounds widely recognized as persistent toxic chemicals of concern both from research and regulatory perspectives. Although each class actually represents a range of molecular components (i.e., PAH compounds from two to five aromatic rings or more; PCB compounds containing one up to ten chlorine substitutions on a biphenyl PAH backbone) including compounds with a range of solubilities, aqueous-solid partition coefficients (K_d), degradation rates, etc., we considered these classes as single entities primarily due to the sparsity of detailed compositional data on environmental samples. Thus, the modeling of these classes considers the class as a whole and individual residence time values may differ considerably from the class behavior. Although other synthetic organic compounds and classes may be of interest, their behavior, as it relates to residence times in environmental compartments, is assumed to be represented by PAH and/or PCB classes. Also, any reasonable attempt at modeling a pollutant class requires a great deal of reliable data which, as will be seen later is lacking in many systems even for the "popular" PAH and PCB classes.

Toxic heavy metals were also considered as a prime modeling target class due to their environmental and regulatory importance, and a more widespread available environmental data base on these elements. Two subclasses of metals were considered: the insoluble or reactive metals that have a large affinity for solid phases due to reactions in seawater and in sediment pore waters; and the soluble metals with greater solubility in seawater. The insoluble metals considered included lead, mercury and silver, while arsenic, copper and cadmium were representative of the more soluble metals. It was our intention to use a subset of these metals, representative of each subclass, to calculate residence time values. The choice of metals in each system depended on the availability of reliable data to be used in calculations.

Other classes of pollutants initially considered included the nutrients, phosphate, and nitrate/nitrite/ammonia, and pathogens. However, while we found that aspects of environmental data bases were substantial and reliable for parts of ecosystem components in selected systems (e.g., nutrients in the water column and pathogens in the sediments in the New York Bight), the data sets were limited in most respects and were deemed insufficient for use in this study. However, with additional data, these and other contaminants can be treated in a manner similar to the toxic organics and metals which are modeled in the sections that follow.

3.5 Selected Study Areas

Five coastal and estuarine areas were selected for study in the residence time segment of the program. These include:

1. The central or main basin of Puget Sound.
2. The Christiaensen Basin of the New York Bight Apex.
3. The Lower Bay Complex of the Hudson-Raritan Estuary.
4. The Palos Verdes shelf area of the Southern California Bight.
5. The Philadelphia dumpsite on the Continental Shelf off of Delaware.

Three basic criteria were used in selecting these study areas. First, it was very important to select a variety of coastal and estuarine regions to demonstrate the application of the residence time modeling strategy to a representative array of environments as shown in Table 3.5.1. The areas included high energy shelf areas in a state of "recovery", low energy deep depositional basins and heavily impacted shallow estuarine areas.

The second criterion was that a substantial, reliable data base existed for the targeted pollutant classes. The chosen study areas had received much research attention in recent years and, although significant data gaps certainly existed in each system, a sufficient quantity of data was available to evaluate pollutant residence times.

Table 3.5.1 Overview of Study Areas

Study Area	Location Type	Depth	Sedimentation Regime	Pollutant Mass Loading Rate	Primary Pollutant Mass Loading Type
Puget Sound Main Basin	Fjord-like; settling area	Deep; 200m	Active to moderate sedimentation; fine sediments	Light to moderate	Mixed; sewage, fallout, rivers
Christiaensen Basin (New York Bight)	Inner Shelf; topographical low	Moderate; 30m	Sediment exchange; net erosional; fine sediments	Moderate	Mixed; sewage sludge dumping, estuarine-shelf fluxes
Hudson-Raritan Estuary (Lower Bay Complex)	Lower estuary	Shallow; 10m	Heavy sedimentation; repeated dredging; mixed sediment types	Heavy	Mixed; riverine fluxes; atmospheric fallout; sewage effluents; input from shelf
Southern California Bight (Palos Verdes Shelf)	Inner Shelf	Moderate; 30-60m	High energy inner shelf	Heavy	Sewage discharge
Philadelphia Dumpsite	Mid-shelf	Moderate; 50m	High energy mid-shelf; ridge and swale topography	Nil to low (site no longer in use)	Prior sewage sludge dumping

Lastly, each area was selected due to environmental concerns focused on the system. Waste discharge practices within or adjacent to all of the selected study areas have been the subjects of much regulatory decision-making and environmental study over the past 10 years or so, and were considered by the program team to be generally lacking of an overall holistic view of contaminants in the system, such as that afforded by box model approaches to each system.

3.6 Puget Sound

The main basin of Puget Sound in Washington State was one of the geographical sites selected for the application of the residence time model to estimate residence times of contaminants in sediment and water. During the last 10 years Puget Sound has been the site of many geochemical and physical oceanographic studies. Consequently, data are available on the sources of contaminants, ocean water exchange rates, water and sediment chemistry, and sedimentation processes. Some of these data are available in summary reports by Dexter et al. (1981) and Romberg et al. (1984).

Puget Sound is a fjord-like basin that is connected to the Pacific Ocean via Admiralty Inlet and the Strait of Juan de Fuca (see Figure 3.1). The main basin is bounded on the north by Admiralty Inlet and to the south by the Tacoma Narrows. The basin is approximately 60 km long, 5 km wide, 200 m deep and is bounded by shallow sills at the northern and southern ends. General bathymetric features include relatively steep sides and a flat bottom.

Rivers and shoreline erosion are the major sources of sediment. Relatively rapid sedimentation occurs in areas of fine grain sediments on the basin floor. Little sedimentation occurs on the sills, sloping sides and shallow areas. The residence time of suspended particles in the water column is several weeks (Baker, 1984) and calculated residence time of water in the basin is several months (Section 3.6.2.1). Consequently, the basin is an excellent trap for sediments and contaminants which are associated with particles (Baker, 1984; Carpenter et al., 1985; Crecelius et al., 1983).

The main basin receives contaminants from the two major urban centers of Seattle and Tacoma. The Municipality of Metropolitan Seattle (Metro) recently completed a study of the sources, distribution and fate of contaminants in Puget Sound (Romberg et al., 1984). This study determined the mass loading of contaminants to the main basin and estimated the rate of accumulation of contaminants in sediments using age-dated sediment cores. The advective exchange of dissolved and particulate contaminants between the Sound and the Strait was also addressed.

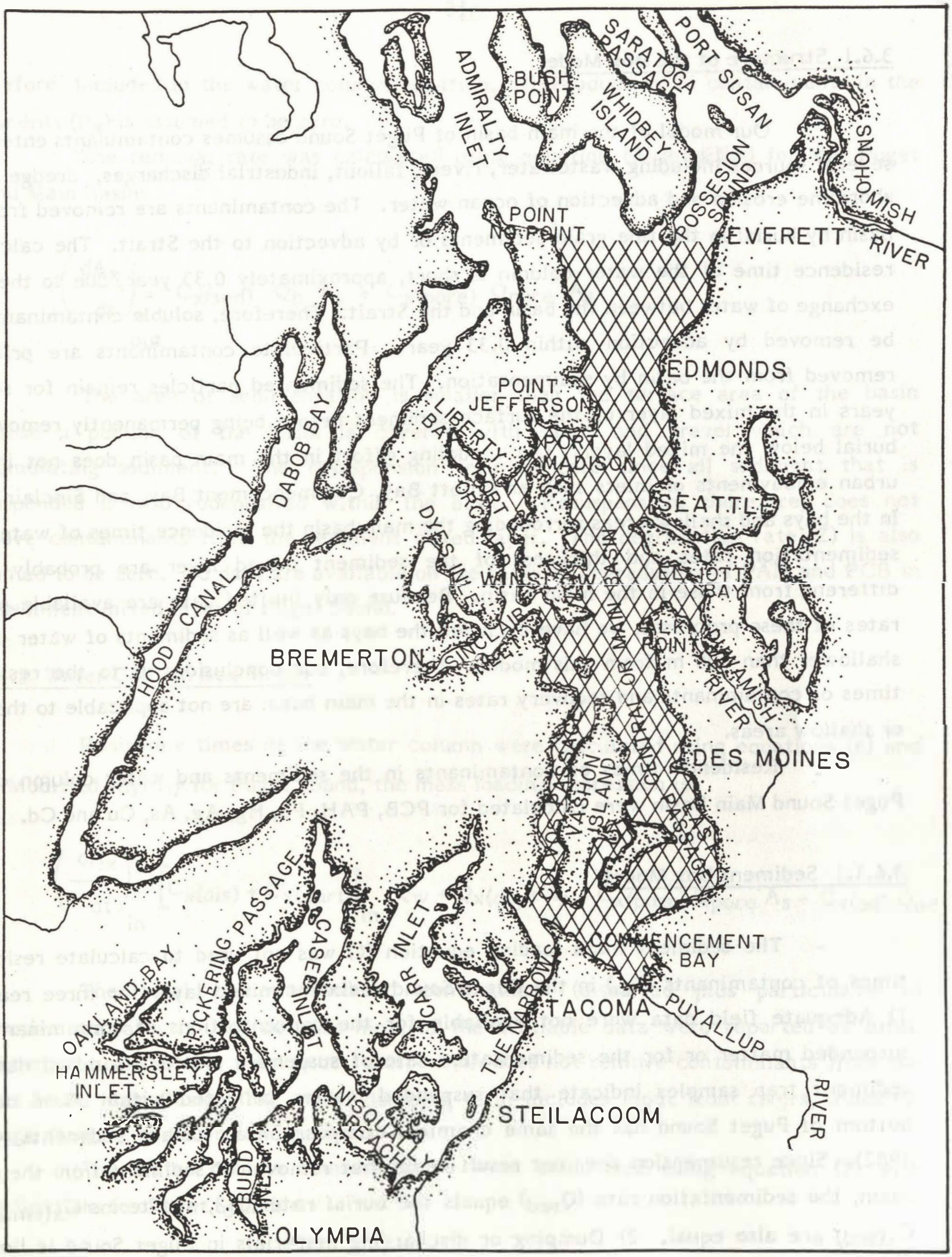

Figure 3.1 Area Map of Puget Sound. (From Dexter et al., 1981). The Area of the Box Model is From Southern Whidbey Island to the Narrows.

3.6.1 Structure of the Box Model

Our model of the main basin of Puget Sound assumes contaminants enter from several sources including wastewater, rivers, fallout, industrial discharges, dredge spoils, shoreline erosion and advection of ocean water. The contaminants are removed from the basin by burial in the fine grain sediments or by advection to the Strait. The calculated residence time of the water column is short, approximately 0.33 year, due to the rapid exchange of water between the basin and the Strait. Therefore, soluble contaminants will be removed by advection within 0.33 year. Particulate contaminants are primarily removed from the basin by sedimentation. The sedimented particles remain for several years in the mixed layer of the surface sediment before being permanently removed by burial below the mixed layer. Our modeling effort in the main basin does not include urban embayments or inlets such as Elliott Bay, Commencement Bay, and Sinclair Inlet. In the bays and shallow areas surrounding the main basin the residence times of water, the sedimentation rates and thickness of the sediment mixed layer are probably much different from those in the main basin. Because only limited data are available on the rates of these processes, we have excluded the bays as well as sediments of water depths shallower than 100 m from this model. Therefore, our conclusions as to the residence times of contaminants and recovery rates in the main basin are not applicable to the bays or shallow areas.

Residence times of contaminants in the sediments and water column of the Puget Sound Main Basin were calculated for PCB, PAH, Pb, Hg, Ag, As, Cu and Cd.

3.6.1.1 Sediment Box Model

The sediment mass loading equation (4) was not used to calculate residence times of contaminants (T_{in}) in the Puget Sound sediment mixed layer for three reasons: 1) Adequate field data were not available for the concentrations of contaminants on suspended matter or for the sedimentation rate of suspended matter. Limited data on sediment trap samples indicate that suspended matter collected within 50 m of the bottom of Puget Sound has the same chemical composition as surface sediments (Curl, 1982). Since resuspension does not result in the net removal of sediment from the main basin, the sedimentation rate (Q_{sed}) equals the burial rate (Q_b), and terms $C_{x(tsm)}$ and $C_{x(sed)}$ are also equal. 2) Dumping or discharging materials in Puget Sound is limited primarily to the water column rather than the sediments. These mass loadings are, therefore, included in the water column equation. 3) Production of contaminants in the sediments (P_x) is assumed to be zero.

The removal rate was calculated using equation (5) modified for the Puget Sound Main Basin:

$$\left(\frac{dA_x}{dt}\right)_{out} = C_{x(sed)}\ Q_b\ A_s + C_{x(pore)}\ Q_{pore}\ A_s$$

The area of sedimentation is smaller than the surface area of the basin because a portion of the basin is covered with sands and gravel which are not accumulating sediment. The resuspension term is zero since all sediment that is resuspended is also redeposited within the basin. Resuspension, therefore, does not remove contaminants from the sediment mixed layer. The degradation rate (Z) is also assumed to be zero. No data are available on the rates of degradation of PAH and PCB in the sediment mixed layer of Puget Sound.

3.6.1.2 Water Column Box Model

Residence times in the water column were calculated using equations (6) and (7). Modified slightly for Puget Sound, the mass loading expression is:

$$\left(\frac{dA_x}{dt}\right)_{in} = \left[C_{x(dis)} + C_{x(part)}\right]_{in} Q_w + Q_{x(dep)}\,A_s + C_{x(pore)}\,Q_{pore}\,A_s + C_{x(od)}\,Q_{od}$$

The total concentration of contaminant (dissolved plus particulate) in advective seawater is used because most of the available data were reported as total. Sediment resuspension is zero since resuspension does not remove contaminants from the Puget Sound Main Basin. The ocean dumping term includes input from rivers, industry, sewage dredging and shoreline erosion.

The water column removal rates were calculated using equation (7) with modifications for Puget Sound:

$$\left(\frac{dA_x}{dt}\right)_{out} = \left[C_{x(dis)} + C_{x(part)}\right]_{out} Q_w + \left[C_{x(tsm)}\right]_{out} Q_{sed}\,A_s$$

As in the mass loading calculation, the dissolved and particulate concentrations in advective seawater were combined for the sedimentation term, the concentration of contaminants in surface sediment is used for $C_{x(tsm)}$, and only the area of active sedimentation used for A_s. The decay rate is assumed to be zero.

3.6.2 Generic Parameters

3.6.2.1 Advective Transport

Puget Sound can be classified as a fjord-type estuarine system because of its steep sides, deep basin and relatively shallow sills. Circulation and mixing processes are controlled by fresh water input, density variations and the ocean source-water at the outside of the Admiralty Inlet sill, and turbulence generated by interaction between strong tidal currents and basin topography.

The circulation and mixing processes are complex due to the complexity of basin topography, variations in tidal range, seasonal density changes, wind effects and transfer of water between the other three basins of Puget Sound. The primary mixing is by tidal currents. The general conceptual circulation pattern is that dense bottom water from the Strait flows over the sill at Admiralty Inlet where it is mixed with less dense surface water and continues southward at depth through the main basin. This denser water travels to the Narrows where tidal currents mix it with the fresher surface water. This newly-mixed water flows northward to Admiralty Inlet where it either mixes with incoming ocean water or is transported into the Strait of Juan de Fuca and eventually enters the Pacific Ocean. The main basin is essentially a two-layer system with impact of ocean water at depth and surface water removal.

Recirculation at the entrance sill has been estimated to range from 50% to 67% resulting in flushing the main basin in 3-6 months (Cannon, 1982; Ebbesmeyer and Barnes, 1980). Friebertshauser and Duxbury (1972) estimated similar flushing rates for the main basin using a salt budget approach.

Ebbesmeyer et al., (1984) have developed a multicompartment circulation model for all basins of the Sound. We have used their net exchange transport rates of 6880 m^3 sec^{-1} for inflowing deep water and 7520 m^3 sec^{-1} for outflowing surface water. This model assumes 53% of surface water transported northward is recycled southward. We assume no interaction occurs with other basins. Using a volume of 7.8×10^{13} L, calculated from the area and depth of the main basin, the residence time of water in the main basin, assuming a net transport rate of 6880 m^3 sec^{-1}, is 0.33 year.

3.6.2.2 Sedimentation Rates

Sedimentation rates for the main basin range from 0.1-1.2 g cm^{-2} yr^{-1} and generally increase with decreasing grain size (Crecelius et al., 1983; Romberg et al., 1984; and Carpenter et al., 1985). We have calculated the mean sedimentation rates for three types of sediment (mud, sandy mud, and muddy sand) using sedimentation rates from Romberg et al., (1984) and the area of each sediment type based on maps prepared by Roberts (1979). The total area of the basin is 7.0 x 10^{12} cm^2 of which 61% (or 4.27 x 10^{12} cm^2) is accumulating sediment. The remaining 39% of the basin, which is covered with sand, gravel or steep slopes, is not accumulating sediment (Romberg et al., 1984). From the following data of Romberg et al. (1984) and Roberts (1979), the yearly sediment deposition in the basin and the average sedimentation rate are calculated:

Sediment Type	Sedimentation Rate	Area	Yearly Sediment Deposition in Basin
Mud	$0.73\ g\ cm^2\ yr^{-1}$	$\times\ 1.19 \times 10^{12}\ cm^2$	$= 0.87 \times 10^{12}\ g\ yr^{-1}$
Sandy Mud	0.52	$\times\ 1.19$	$= 0.62$
Muddy Sand	0.41	$\times\ 1.89$	$= 0.79$
		$4.27 \times 10^{12}\ cm^2$	$2.24 \times 10^{12}\ g\ yr^{-1}$

$$\text{Average Sedimentation Rate} = \frac{2.24 \times 10^{12}\ g\ yr^{-1}}{4.27 \times 10^{12}\ cm^2} = 0.52\ g\ cm^{-2}\ yr^{-1}$$

Sedimentation rates reported by Carpenter et al. (1985) for the deep region of the main basin average 0.57 ± 0.44 g cm^{-2} yr^{-1} for five mud cores and 0.38 ± 0.16 g cm^{-2} yr^{-1} for three sandy mud cores. Carpenter et al., (1985) also estimated the thickness of the sediment mixed layer for basin sediments: 3.4 ± 0.7 g cm^{-2} for mud; 6.2 ± 1.7 g cm^{-2} for sandy mud; 5.3 ± 0.6 g cm^{-2} for muddy sands. We have averaged the thickness of the three sediment types and used 4.9 g cm^{-2} (about 10 cm) for the thickness of the main basin mixed layer. Sediment porosity ranges from 0.5 - 0.9 (Lavalle et al., 1985). Multiplying this mixed layer thickness by the area of sedimentation (4.27 x 10^{12} cm^2) gives an inventory of 21 x 10^{12} g of sediment in the mixed layer. The residence time of the sediment in the mixed layer is:

$$\frac{21 \times 10^{12} g}{2.24 \times 10^{12}\ g\ yr^{-1}} = 9.4 \text{ years}$$

Thus, in approximately 10 years a new mixed layer of sediment will be deposited. However, mixing between the new and old sediments means the rate of recovery will be greater than the residence time. Since we have averaged rates and thickness over the area of the basin that is actively accumulating sediment, there will be specific sites in the basin that have much shorter or longer residence times. This will be discussed in Section 3.6.4 on sensitivity analysis.

3.6.2.3 Mass Loading of Contaminants

The mass loading of eight contaminants from seven sources to the main basin is listed in Table 3.6.1. We have used data from Romberg et al. (1984) which we believe are the best available mass loading estimates representative for the years 1980-1982. All mass loadings are assumed to enter the main basin. We have not tried to estimate fraction of dissolved versus particulate contaminant. Sewage includes both known outfall entering the main basin and combined sewage overflow (CSO). Storm water runoff is also included in sewage since much of the storm water either enters the treatment plants or CSO. The Metro Renton treatment plant enters the Duwamish River so mass loading from the Renton plant is reflected in the riverine inputs. Fallout includes both dry fallout and wet fallout. Industrial outfalls are those for which data are available through monitoring activities required by State of Washington NPDES permits. Loading from industries which discharge into municipal sewage plants is included in the sewage outfall loading. Only the Duwamish River, Puyallup River, and Lake Washington ship canal are included in "rivers". Shoreline erosion is an important source of sediment and some associated contaminants. Advection is defined as the annual transport of contaminants associated with three volumes of deep Strait water to fill the main basin three times (i.e., residence time of the water column is 0.33 year). Even though the Strait water is considered to be uncontaminated, there are still typical baseline concentrations of contaminants which can amount to a major source to the main basin due to the huge volume of seawater advected through the basin. For example, advection is the largest source of As, Cu, and Cd. These three contaminants tend to remain in solution and are transported back to the ocean within several months.

3.6.2.4 Sediment Chemistry

The concentrations of contaminants in Puget Sound surface sediments and baseline sediments are listed in Table 3.6.2. These concentrations are taken from the

Table 3.6.1. Mass Loading of Contaminants to the Main Basin of Puget Sound.

	Sewage and CSO Outfalls (kg yr^{-1})	Fallout (kg yr^{-1})	Dredge Spoils (kg yr^{-1})	Industrial Outfalls (kg yr^{-1})	Rivers (kg yr^{-1})	Shoreline (kg yr^{-1})	Advection (kg yr^{-1})	Total (kg yr^{-1})
PCB	57	n.d.	11	70	2	1	2.2	142
PAH	1,700	340	220	39	460	300	2.2	3,060
Pb	15,200	40,000	3,800	4,100	30,000	7,000	22,000	110,000
Hg	60	11	20	100	200	50	130	570
Ag	1,500	15	40	1,200	200	50	150	3,140
As	850	2,500	900	5,000	28,000	58,000	373,000	467,900
Cu	17,000	5,800	2,400	32,000	49,000	12,000	65,000	183,000
Cd	750	51	n.d.	750	400	300	21,000	23,300

n.d. = no data.

Table 3.6.2. Concentrations of Contaminants in Puget Sound Baseline Sediments, Surface Sediment, Mixed Layer Inventory, Net Annual Burial Rate and Percentage of Mass Loading Buried.

Contaminant	Pre-1900 Baseline Sediment ($\mu g\ g^{-1}$)	Surface Sediment ($\mu g\ g^{-1}$)	Mixed-Layer Inventory (kg)	Annual Burial (kg yr^{-1})	% of Mass Loading Buried
PCB	0.009	0.12	2,500	260	183
PAH	0.1	2.2	46,000	4,800	157
Pb	6	27	578,000	59,000	53
Hg	0.04	0.16	3,500	350	61
Ag	0.04	0.32	6,800	700	22
As	6	13	278,000	29,000	6.2
Cu	23	27	779,000	81,000	44
Cd	0.3	0.25	5,200	550	2.4

METRO Toxicant Pretreatment Planning Study (TPPS) Report (Romberg et al., 1984). Baseline sediments are those that have been age-dated to be deposited in the mid-1800's. The contaminant concentrations in surface sediment (mixed layer) are the weighted average for the three sediment types that are accumulating sediment. The mixed layer inventory is the sum of contaminants in each of the three sediment types (Table 3.6.2). These were calculated by multiplying the mass of the mixed layer by the average contaminant concentration for the corresponding sediment type, such as muddy sand. The annual burial of each contaminant in the main basin is calculated from the average sedimentation rate applied to 61% of the area of the main basin (Table 3.6.2). The column on the right in Table 3.6.2 is the percentage of the annual mass loading (Table 3.6.1) that is buried annually. Except for PCB and PAH these results are not unreasonable and will be discussed further in Section 3.6.5.

3.6.2.5 Contaminants in the Water Column

The concentrations of contaminants in the water column of Puget Sound and the Strait are shown in Table 3.6.3. These data were taken from Romberg et al., (1984). The annual transport of contaminants by advection of seawater either out of the Sound to the ocean or from the Strait into the Sound are calculated from the total concentration of contaminants in the water and the volume of water advected annually (Table 3.6.3). The inventory of contaminants in the water column is the product of water column concentration and the basin volume (Table 3.6.3). The advective flux (kg yr^{-1}) from the basin to the ocean is about three times the inventory since the basin water is assumed to be exchanged three times a year.

3.6.3 Residence Times in the Main Basin

3.6.3.1 Residence Times in the Sediment

The residence time $(T)_{out}$ of a contaminant in the mixed layer of the sediment (i.e., the sediment box) is determined by the time required to bury the contaminant below the mixed layer, resuspend and transport the contaminant out of the box, and diffuse soluble contaminants in pore water into the water column. The value of $(T)_{in}$ is a function of the mixed layer inventory and the rates of deposition. The (T) of a contaminant in the mixed layer of sediment is defined as either mixed layer sediment inventory (g cm^{-2}) divided by deposition (g cm^{-2} yr^{-1}) or sediment inventory divided by the combined rates of burial, resuspension, degradation, and pore water flux.

Table 3.6.3. Concentrations of Contaminants in Puget Sound and Strait Water Column, Advective Transport Rate and Inventory.

Puget Sound Water Column Chemistry

Contaminant	Dissolved (μg L^{-1})	Particulate (μg L^{-1})	Total (μg L^{-1})	Advective Flux to Ocean (kg yr^{-1})	Inventory Water Column (kg)
PCB	0.002	0.00013	0.0021	470	160
PAH	0.0045	0.05	0.05	12,000	3,900
Pb	0.01	0.15	0.16	38,000	13,000
Hg	0.0002	0.0012	0.0014	330	110
Ag	0.00005	0.001	0.0015	360	120
As	1.7	0.03	1.73	410,000	135,000
Cu	0.4	0.10	0.50	120,000	39,000
Cd	0.09	0.005	0.095	23,000	7,400

Strait of Juan de Fuca Deep Water

Contaminant	Dissolved (μg L^{-1})	Particulate (μg L^{-1})	Total (μg L^{-1})	Advective Flux from Ocean (kg yr^{-1})
PCB	0.01	0.00001	0.00001	0.022
PAH	n.d.	0.00001	0.00001	0.022
Pb	0.025	0.025	0.05	11,000
Hg	0.0001	0.0005	0.0006	130
Ag	0.0001	0.0006	0.0007	150
As	1.7	0.02	1.72	373,000
Cu	0.2	0.1	0.3	65,000
Cd	0.09	0.005	0.095	21,000

n.d. = no data

We have chosen to not include resuspension as a contaminant removal process because resuspension in the main basin of Puget Sound does not remove sediment or contaminants from the basin. Resuspension rates for the main basin fine grain sediment have been estimated to be approximately 3 g cm^{-2} yr^{-1} (Curl, 1982). However, studies of the transport of suspended matter indicate essentially all resuspended matter will be redeposited in the basin (Baker, 1984).

The flux of soluble contaminants from the sediment into the water column has been estimated from the concentrations of metals in pore water squeezed from surface sediments and assuming the diffusion coefficient for the metals will be the same as that reported by Curl (1982) for Mn in main basin sediments. The pore water fluxes are small compared to the burial rates of contaminants. The largest pore water flux value is for As which, at 4700 kg yr^{-1}, is about 15% of the burial rate (see Table 3.6.4). Furthermore, we assume that organic compound degradation rates are very small compared with other removal mechanisms since active degradation occurs only at or near the oxygenated sediment-water interface (Lee and Ryan, 1983).

The residence times of the contaminants in the mixed layer are on the order of 10 years (Table 3.6.4), based on a sediment inventory of about 5 g cm^{-2} and a sedimentation rate of 0.52 g cm^{-2} yr^{-1}. Thus, in about 10 years most of the contaminants will be buried below the mixed layer and removed from the marine environment. In Table 3.6.4 (T) was calculated both with and without the pore water flux. Pore water flux can slightly decrease the residence times of As, Cu, and Cd.

3.6.3.2 Residence Times in the Water Column

The residence times of contaminants in the water column of the main basin have been calculated from mass loading rates (Table 3.6.5) and also using removal processes (Table 3.6.6). The results for $(T)_{in}$ calculated from inventory and mass loading range from 0.04 years for Ag to 1.3 years for PAH (Table 3.6.5). The expected values for (T) are 0.33 years for the relatively soluble contaminants such as As and Cd, and 0.06 to 0.2 years for contaminants such as PCB, PAH, and Pb that are primarily associated with particulate matter. Since the water column has a residence time of 0.33 years and suspended matter has a residence time of about 0.06 years, the residence time of contaminants should fall in that range.

Table 3.6.4. **Residence Time $(T)_{out}$ of Contaminants in Puget Sound Main Basin Sediment Mixed Layer, Calculated from Pore Water and Burial Rate.**

Contaminants	Mixed-Layer Inventory (kg)	Burial Rate (kg yr^{-1})	Pore Water (µg L^{-1})	Pore Water Flux (kg y^{-1})	$(T)_{out}$ without Pore Water (years)	$(T)_{out}$ with Pore Water (years)
PCB	2,500	260	n.d.	n.d.	9.6	9.6
PAH	46,000	4,800	n.d.	n.d.	9.6	9.6
Pb	578,000	59,000	<0.1	24	9.8	9.8
Hg	3,500	350	0.004	1	10	10
Ag	6,800	700	0.002	0.5	9.7	9.7
As	278,000	29,000	20	4,700	9.6	8.2
Cu	779,000	81,000	4	980	9.6	9.5
Cd	5,900	550	0.04	10	10.7	10.5

n.d. = no data

Table 3.6.5 Residence Time $(T)_{in}$ of Contaminants in Puget Sound Main Basin Water Column, Calculated from Mass Loading.

Contaminant	Water Column Inventory (kg)	Mass Loading (kg yr^{-1})	$(T)_{in}$ (years)
PCB	160	142	1.1
PAH	3,900	3,060	1.3
Pb	13,000	110,800	0.12
Hg	110	570	0.19
Ag	120	3,140	0.04
As	135,000	467,900	0.29
Cu	39,000	183,400	0.21
Cd	7,400	23,300	0.32

Table 3.6.6. Residence Time $(T)_{out}$ of Contaminants in Puget Sound Main Basin Water Column, Calculated from Removal by Burial and Advection.

		Removal			
Contaminant	Water Column Inventory (kg)	Burial (kg yr^{-1})	Advection (kg yr^{-1})	Total Removal (kg yr^{-1})	$(T)_{out}$ (years)
PCB	160	260	470	730	0.22
PAH	3,900	4,800	12,000	16,800	0.23
Pb	13,000	59,000	38,000	97,000	0.13
Hg	110	350	330	680	0.16
Ag	120	700	360	1,060	0.11
As	135,000	29,000	410,000	439,000	0.30
Cu	39,000	81,000	120,000	201,000	0.19
Cd	7,400	550	23,000	23,550	0.32

3.6.4 Sensitivity Analysis

The term sensitivity analysis refers to the range of residence times that is calculated based on the uncertainty in the data used to estimate the residence times. The uncertainty in the sedimentation rate for the fine grain sediments of the main basin is ± 23% for the standard deviation of the mean based on cores we dated for Metro (Romberg et al., 1984). The standard deviation for the mean thickness of the sediment mixed layer is ± 20% based on data from Carpenter et al. (1985). The uncertainty in the chemical data is not easily estimated, especially for the mass loading data that were generated by many different agencies and also due to the many assumptions which were made for such data as average river flow rates or atmospheric fallout rates. The standard deviation for mean concentrations of contaminants in surface sediments is in the range of ± 24-49%. The variation in the water column chemistry data is generally greater than in the sediments.

The standard deviation associated with the residence time of Pb in the sediment mixed layer is ± 58% based on propagation of errors, assuming the standard deviations are ± 23% for the sedimentation rate, ± 20% for mixed layer thickness and ± 35% for Pb concentration in main basin surface sediments. Thus, the (T) for Pb in the sediment is 9.8 years for mean rates and concentrations. However, (T) can range from 4.1-15.5 years for one standard deviation.

3.6.5 Analysis of Model Results

Residence times can be calculated either by using input rates (mass loadings) or removal rates (burial and advection). If generic parameter data are accurate, and if the basic assumptions that govern (T) are valid, and the system is in steady-state, then both calculations should result in the same (T) for a given contaminant. Another check on the accuracy of the model is to compare the burial rate with the mass loading rate. The results in Table 3.6.2 indicate the burial of PCB and PAH are 183% and 157% of the mass loading rate, respectively. This indicates either the mass loadings are too low and/or the burial rate is too high or the system is not in equilibrium. The comparison between burial and loading rates for the metals shows that Pb, Hg, Ag and, to a lesser extent, Cu tend to be associated with particles and, therefore, will be removed by burial. Because As and Cd are primarily dissolved, only a few percent of these mass loadings are removed by burial.

The total removal rates (burial plus advection) of contaminants from the water column are shown in Table 3.6.6. By comparing these total removal rates with the mass loading rates (Table 3.6.1 or Table 3.6.5) it is apparent that removal of PCB and PAH exceed mass loading rates by about a factor of five. The Ag mass loading rate exceeds removal by three times. However, the other metals balance within about ± 10%. The high removal rates for PCB and PAH are probably due to over-estimation of the advection term. The major uncertainty in advection of these organic contaminants is again in the concentration of contaminants in the water column. Measurements of these contaminants at low concentration in seawater are extremely difficult to achieve. The major limitation is collecting a large enough sample without contaminating the sample.

The imbalance in the Ag budget may also be due to over-estimation of the mass loading term. We are more confident in the Ag removal rates than in the mass loading data because more data are available for Ag in seawater and sediment than in waste water.

The $(T)_{out}$ for all contaminants in the water column is in the range of 0.1-0.3 years when calculated using the total removal rates (Table 3.6.6). The main difference between the $(T)_{out}$ in Table 3.6.6 and $(T)_{in}$ in Table 3.6.5 is that the calculated (T) values for PCB and PAH are much shorter if removal rates are used.

We have estimated the percentages of total removal processes that are accounted for by burial (Table 3.6.7). The more soluble metals such as As, Cd and, to a lesser extent, Cu are removed more by advection while Pb, Hg, and Ag are removed primarily by burial. Except for PCB and PAH there is reasonable agreement between the percentage of mass loading removed by burial and the percentage of total removal processes that is due to burial. We believe the disagreement for PCB and PAH is due primarily to inaccurate water chemistry data.

3.6.6 Sediment and Water Quality For Hypothetical Loadings

The concentrations of contaminants in the surface sediments and water column of the main basin have been predicted for several hypothetical loadings (Tables 3.6.8 and 3.6.9). We have assumed that the basin has approached equilibrium for each loading. This may require almost a decade in the case of the sediment mixed layer. We also assume that the percentages of contaminant which are removed by burial and advection remain constant for different loading rates.

The concentrations of contaminants in surface sediment are for the early 1980s which correspond to the loadings in Table 3.6.1 noted as 1X in Table 3.6.8. Baseline

Table 3.6.7. Percentages of Mass Loading Rate and Total Removal Rate Buried in Sediment of Puget Sound Main Basin.

	Mass Loading[a] (kg yr^{-1})	% of Mass Loading[b] Removed by Burial	Total Removal Rate (Burial and Advection) (kg yr^{-1})	% of Total Removal[c] Rate Due to Burial
PCB	142	183	730	36
PAH	3,060	157	16,800	29
Pb	110,800	53	97,000	61
Hg	570	61	680	51
Ag	3,140	22	1,060	66
As	467,900	6.2	439,000	7
Cu	183,400	44	201,000	40
Cd	23,300	2.4	23,550	2

[a] Mass loading from Table 3.6.1

[b] Burial rate (from Table 3.6.2) divided by mass loading rate times 100

[c] Burial rate divided by total removal rate times 100

Table 3.6.8. Concentrations of Contaminants in Puget Sound Main Basin Mixed Layer Sediments if Hypothetical Loadings are Changed by Factors of 0, 1, 2 and 10 Times Loadings of 1981.

	μg g^{-1} dry wt.						
			Loading Factor				Secondary Treatment
	Puget Sound 1981	Baseline[a] (pre-1900)	0X[b]	1X	2X	10X	1X
PCB	0.12	0.009	0.0012	0.053	0.11	0.52	0.035
PAH	2.2	0.1	0.21	1.2	2.1	9.7	0.54
Pb	27	6	8.2	27	45	192	21
Hg	0.16	0.04	0.081	0.16	0.24	0.86	0.12
Ag	0.32	0.04	0.031	0.32	0.61	2.9	0.22
As	13	6	12	13	14	19	13
Cu	37	23	21	37	53	183	30
Cd	0.25	0.3	0.22	0.25	0.27	0.43	0.24

[a] Romberg et al. (1984).

[b] For 0X inputs of shoreline, advection and 50% of rivers were used.

Table 3.6.9. Concentrations of Contaminants in Puget Sound Main Basin Water Column if Hypothetical Loadings are Changed by Factors of 0, 1, 2, and 10 Times Loadings of 1981.

	Dissolved and Particulate µg L^{-1}						
			Loading Factor				
	Puget Sound 1981	Baseline Strait of Juan de Fuca	0X[a]	1X	2X	10X	Secondary Treatment 1X
PCB	0.002	0.00001	0.000002	0.00012	0.00017	0.00083	0.000056
PAH	0.05	0.00001	0.00034	0.0024	0.0034	0.015	0.00086
Pb	0.16	0.050	0.068	0.22	0.37	1.59	0.17
Hg	0.0014	0.0006	0.0047	0.00093	0.0014	0.0051	0.00071
Ag	0.0015	0.0007	0.0010	0.010	0.020	0.094	0.0074
As	1.73	1.72	1.79	1.89	1.99	2.80	1.82
Cu	0.50	0.30	0.25	0.45	0.64	2.22	0.36
Cd	0.095	0.095	0.087	0.095	0.103	0.166	0.091

[a] OX loading considers no anthropogenic inputs.

sediments are those deposited in the main basin pre-1900s. For the loading 0X (no anthropogenic inputs) only the inputs from shoreline erosion, advection and 50% of the river mass loadings are included. This condition is intended to predict sediment and water quality before industrial development began in the late 1800s. There is fair agreement between baseline and 0X, although most concentrations are higher for 0X loading, probably because more than 50% of the river loading is due to anthropogenic sources. The 1X loading is identical to the surface sediment for 1981 because this was used to set up the model. For the 2X and 10X loadings all the mass loadings except shoreline erosion and advections were doubled. Shoreline and advection were constant for all loadings.

3.6.6.1 Changes in Loading From Proposed Secondary Treatment

At the present time all sewage treatment plants that discharge directly into the main basin are primary treatment systems. These Metro treatment plants and those of other communities may be upgraded to secondary treatment in the future. We have estimated the resulting concentrations of contaminants in the water column and surface sediments of the main basin if all outfall now discharging to the basin were upgraded to secondary (Table 3.6.8 and 3.6.9). Data from the Metro TPPS (Galvin et al., 1984) were used to calculate the change in mass loading to the basin. Secondary treatment will increase the removal of metals from the present efficiency of 47%-77%, and the removal of extractables (including PCB and PAH) from present efficiency of 10%-78% (Table 3.6.10).

The changes in water and sediment concentrations for present (1981) mass loading and for a hypothetical loading with only secondary treatment are shown in the right-hand column of Tables 3.6.8 and 3.6.9. The improvement in water and sediment quality is not very dramatic, reducing organics by approximately 50% and some metals by 25%. As and Cd concentrations in the basin will not change since the mass loading of these two contaminants is less than 5% from sewage.

Secondary treatment will not result in a large improvement in environmental quality of the main basin because sewage is only one source of these contaminants and secondary treatment only reduces, but does not eliminate this source. Since the water and sediments of the main basin are not at present sufficiently contaminated to cause detectable changes in the marine ecosystem, upgrading to secondary treatment will only make a relatively small change in the concentration of some contaminants in an ecosystem which has not been shown to be stressed (Strand et al., 1985). Parts of urban embayments, including Elliott Bay, Commencement Bay, and Sinclair Inlet, have been

Table 3.6.10. **Mass Loading of Contaminants to the Main Basin of Puget Sound From Existing Primary Sewage Treatment Plants and Hypothetical Secondary Treatment Plants.**

	Primary Sewage Treatment		Secondary Sewage Treatment	
	% Removal Efficiency	Mass Loading (kg yr^{-1})	% Removal Efficiency	Mass Loading (kg yr^{-1})
PCB	10	57	78	14
PAH	10	1,700	78	410
Pb	47	15,200	77	6,500
Hg	47	60	77	26
Ag	47	1,500	77	650
As	47	850	77	370
Cu	47	17,000	77	7,300
Cd	47	750	77	320

shown to be both biologically degraded and relatively contaminated (Malins et al., 1984; Chapman et al., 1982; Galvin et al., 1984; Strand et al., 1985). Secondary treatment will not improve the quality of these bays since most primary effluent is discharged in relatively deep water outside the embayments. The discharge of contaminants to bays from combined sewage outflows, storm water runoff, industrial outfalls and nonpoint source will not be reduced by secondary treatment. In summary, secondary treatment will not significantly improve those areas of Puget Sound that are biologically degraded.

The results of the hypothetical loadings can be used to predict the past or future sediment and water quality of the main basin for greater or lesser loading rates than at present. If loadings were 10 times present, then sediments of the main basin would be similar to those currently found in some urban embayments where biological communities are altered (Malins et al., 1982; Malins et al., 1984; Strand et al., 1985).

3.6.6.2 Criteria for Hypothetical Loading

We have compared the concentrations of contaminants in the water column, pore water and surface sediments of the main basin with available water and sediment chemical criteria (Table 3.6.11). Table 3.6.11 also compares the contaminant concentrations in the main basin of Puget Sound for a hypothetical loading (10X the 1981 loading) with established water and sediment quality criteria. The water quality criteria for metals are 24-hour average seawater minimum criteria (EPA, 1980; 1984). The PCB value is from Tobin (1984) and the PAH value is for fluoranthene (Tobin, 1984). The concentrations of contaminants in sediment pore water were calculated using equilibrium partitioning coefficients (Kp). The Kp representing PAH is for fluoranthene, a PAH compound with an intermediate Kp. Tobin (1984) gave an organic carbon-water partitioning coefficient (Koc) of 4.5×10^4 which, when corrected for sediment with 5% organic carbon, has a Kp of 2.3×10^3. The same procedure was used for PCB employing the tetrachlorobiphenyl Koc of 1×10^5 (Tobin, 1984). The Kp for metals, which are typical of fine grain sediments, were calculated from our data for metals in Puget Sound pore water (Table 3.6.4) and sediments (Table 3.6.2). The pore water concentrations predicted at 10X loading are calculated using the Kp in Table 3.6.11 and surface sediment chemistry data at 10X loading.

The surface sediment criteria used in Table 3.6.11 are interim criteria for the Four-Mile Rock open water dredge spoil site in Puget Sound. These criteria are intended to be used to prevent further degradation of a disposal site in the main basin and should

Table 3.6.11. Comparison of EPA Marine Water and Sediment Quality Criteria with Predicted Concentrations of Contaminants in Water Column, Sediment Pore Water and Surface Sediment for Hypothetical Loading 10 Times 1981 Loading to Main Basin.

	Water Column		Sediment Pore Water		Surface Sediment	
	EPA Criteria Seawater (µg L^{-1})	10X Loading Puget Sound (µg L^{-1})	Kp Partitioning Coefficient	10X Loading Puget Sound (µg L^{-1})	U.S. EPA Region 10 Four-Mile Rock Interim Criteria (µg g^{-1})	10X Loading Puget Sound (µg g^{-1})
PAH	8[a]	0.00083	2.3×10^{3}[b]	4.3	11.2	9.7
PCB	0.014	0.015	5×10^{3}	0.1	0.61	0.52
Pb	8.6	1.6	$<2.7\times10^{5}$	<0.17	12.6	192
Hg	0.1	0.0051	4×10^{4}	0.002	1.1	0.86
Ag	2.3	0.094	1.6×10^{5}	0.018	n.d.	2.9
As	63	2.8	6.5×10^{2}	29	15	19
Cu	2.0	2.2	6.8×10^{3}	27	92	183
Cd	12	0.17	6.2×10^{3}	0.069	0.7	0.43

[a] Fluoranthene water quality criteria, Koc = 4.5×10^{4}.

[b] Tobin (1984) assumed sediment contained 5% organic carbon; PCB-4 Koc = 1×10^{5}.

n.d. = no data

not be used for other purposes. We have used them here for comparison since EPA has not yet developed marine sediment quality criteria.

The predicted concentrations of contaminants in the Puget Sound water column only slightly exceed the water criteria for Cu and PCB (Table 3.6.9). For pore water both Cu and PCB exceed the criteria for water by a factor of 10. The surface sediments will exceed the Four-Mile Rock criteria for Pb, As, and Cu assuming 10X loading rate. At the present (1981) loading rate neither water column nor sediments will exceed these criteria. However, Cu now exceeds the seawater criteria of 2 µg L^{-1} in pore water.

3.6.7 Summary and Research Needs

The residence times for contaminants in the main basin sediment mixed layer are approximately 10 years, with a 66% confidence range of 4-16 years. The residence times for contaminants in the water column range from 0.04-0.33 years. If the mass loading of a contaminant to the main basin is greatly reduced or eliminated, the recovery of the water column will occur in a year, while the surface sediments will require several decades to recover. Because there is considerable variation in both the sedimentation rates and thickness of the sediment mixed layer within the main basin, the period of sediment recovery may vary from several years to several decades.

The concentrations of several contaminants in age-dated sediment cores indicate that the surface sediments respond to changes in contaminant mass loadings (Crecelius and Bloom, 1985). Mixed layer residence times on the order of a decade can be inferred from profiles of PAH, PCB, heavy metals and radionuclides.

The greatest discrepancies in the modeling results are for PCB and PAH. Additional data are needed on the concentrations of these two contaminants in the water column before these discrepancies can be addressed.

Sediments in urban bays of Puget Sound contain much higher concentrations of contaminants than those of the main basin (Malins et al., 1984; Romberg, 1984; Strand et al., 1985). The urban sediments are a potential source of contaminants to the main basin if sediment resuspension or dredging operations transport particles out of the urban bays. Limited studies of sediment cores collected in Commencement Bay and Everett Harbor indicate that contaminants are being buried in these areas and will not be a long-term source of contaminants to the main basin (Riley et al., 1981; Crecelius et al., 1985). However, additional studies are needed to determine the potential for transport of bay sediments to the main basin.

3.7 CHRISTIAENSEN BASIN

The Christiaensen Basin (Figure 3.2) is defined as the area in the New York Bight Apex between 26 and 37 m depth north of 40°24'N (Freeland and Swift, 1978). The Basin is considered a topographical low, located directly north of the Hudson Shelf Valley of which the Basin is a shoreward extension. It has an area of 83 km^2 with an average depth of 30m. The geographical center of the Christiaensen Basin is approximately 5km southeast of Ambrose Light.

The proximity of the Christiaensen Basin to the New York Sewage Sludge and Dredged Material Dumpsites, Acid Waste Disposal Site and the mouth of the Hudson-Raritan estuary has lead to many investigations to determine if the Christiaensen Basin may be contaminated with pollutants originating from these areas. All of the above areas are sources of high levels of suspended particulate matter (SPM) and contaminants in the Christiaensen Basin are probably derived from SPM deposition. Other mechanisms of contaminant transfer probably do not occur, i.e., there is no authorized dumping of any type of material directly in the basin, nor is there any evidence of bedload movement of material into the basin from any disposal site (Freeland and Swift, 1978; Ingham et al., 1982).

Freeland and Swift (1978), using data from topographic surveys of 1845, 1930 and 1973, have concluded that the basin is an area of net erosion. This estimation runs counter to the fact that topographic lows generally tend to be net depositional areas. In addition, the Christiaensen Basin is floored with fine-grained sediment, a further indication that the Basin is an area of deposition. However, since there is no supporting information to indicate that the Christiaensen Basin is an area of net deposition, the conclusions of Freeland and Swift (1978) will be used in this modeling experiment.

The Christiaensen Basin must then be viewed as a depositional area during quiescent conditions and an area of net erosion during severe storm events, an approach also discussed by Ingham et al. (1982). Calculated deposition of 0.11 g cm^{-2} y^{-1} of sediment (Freeland and Swift, 1978) is offset by 0.25 g cm^{-2} y^{-1} sediment resuspended (Freeland and Swift, 1978) for a net resuspension rate of 0.14 g $cm^{-2}y^{-1}$. The upper layer of sediment then is constantly being reworked by storms during the course of the year. Sediment contaminants are contained in a fine layer of floc-like material similar in nature to SPM, and in material worked into the sediment mixed layer from this floc layer by bioturbation.

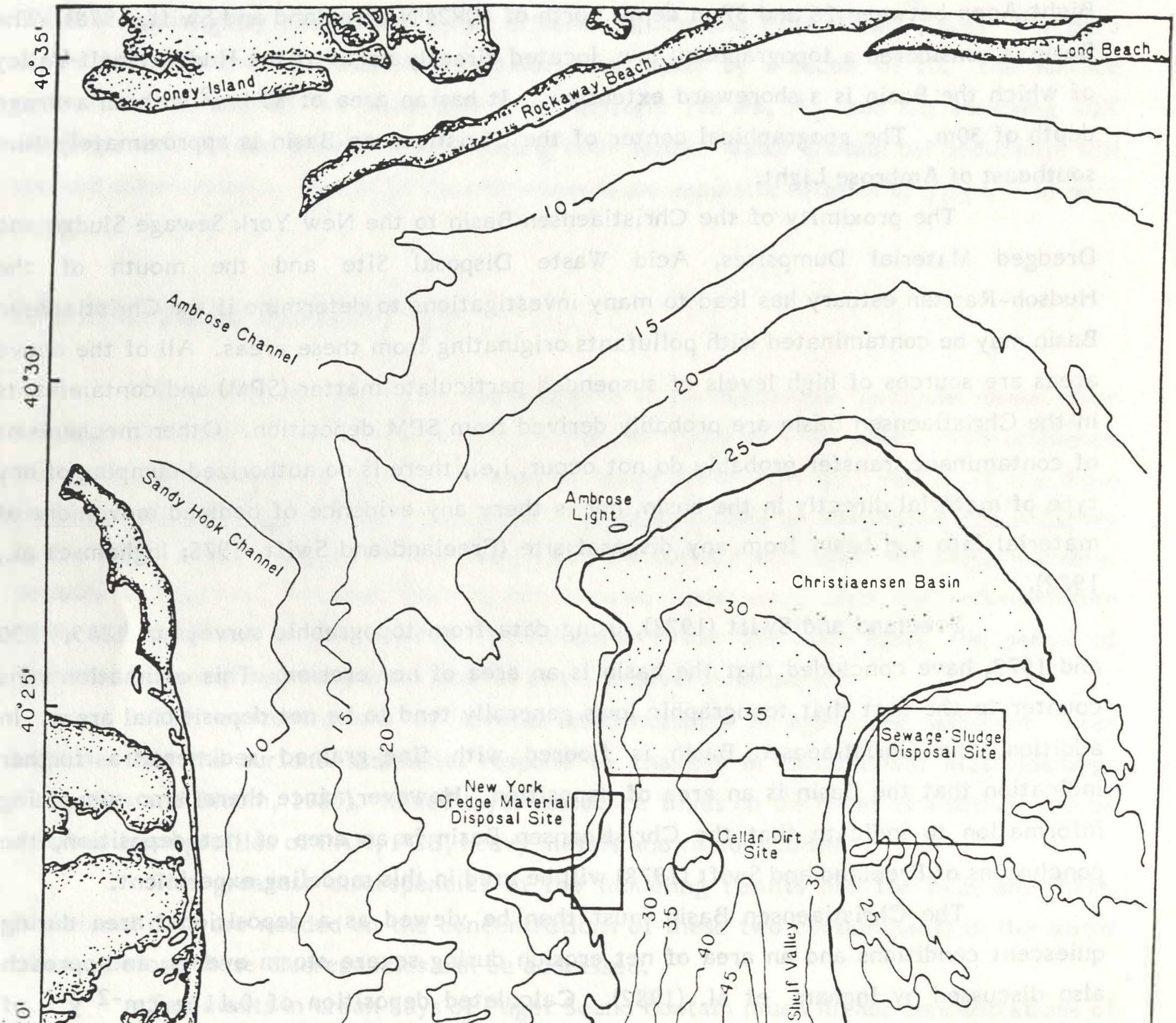

Figure 3.2 The Christiaensen Basin.

3.7.1 Structure of the Box Model

The box model for which we are applying the residence time model consists of an irregular area approximately 10.4 km wide, by 8 km long, with a uniform water depth of 30 m. The area of the box is 83 km^2 and the volume of the water column of the box is 2.5×10^9 m^3. The sediment box has an area of 83 km^2. The upper 5cm of sediment is the bioturbated layer generating a sediment mixed-layer volume of 4.1×10^6 m^3.

Residence times in the water column and sediment boxes for the Christiaensen Basin are calculated for PCB, PAH, Pb, Cu, and Cd. The inventory data are incomplete for the other contaminants and, therefore, residence times cannot be calculated with any degree of confidence.

3.7.1.1 Sediment Box Model

The general formulations of residence time equations used to calculate $(T)_{in}$ and $(T)_{out}$ have been presented in Sections 3.3.1 and 3.3.2.

For the Christiaensen Basin the mass loading or input term is reduced from equation (4) to:

$$\left(\frac{dA_x}{dt}\right)_{in} = \Big[C_{x(tsm)}\Big]_{in} Q_{sed}$$

Since there is no authorized dumping directly in the Christiaensen, f = 0 and the mass loading term becomes strictly a function of the sedimentation of contaminated particles. A certain fraction of the sludge dumped at the New York Sewage Sludge Site will be advected and deposited in the box. However, the available data afford no indication as to what fraction of the total sludge dumped at the dump site is mass loaded to the box, and f is assumed to be zero for the purposes of the calculation of residence times. The subject of sludge-derived contaminant mass loading in the Christiaensen Basin is discussed in Section 3.7.6 (Sediment and Water Quality for Hypothetical Loadings).

The removal expression modified for the Christiaensen Basin from equation (5) is:

$$\left(\frac{dA_x}{dt}\right)_{out} = \Big[C_{x(sed)}\Big]_{out} Q_{rs} \ A_s$$

Since there is no net sedimentation in the Christiaensen Basin, the output term is a function of the resuspension rate, not the burial rate. Also, since the burial rate is zero, the flux of pore water to overlying water is negligible ($Q_{pore} = 0$; Stumm and Morgan, 1981) and, therefore, pore water flux is not considered a removal process in the Christiaensen Basin sediment box. The microbial degradation rate is low for PAH ($t_{1/2}$ = 2 years; Lee and Ryan, 1983) and assumed to be negligible for other contaminants modeled. Therefore, for all contaminants studied except for PAH, the output term is strictly a function of the resuspension of contaminated sediment.

3.7.1.2 Water Column Box Model

Expressions for the residence time for contaminants in the water column are calculated from equations (6) and (7) presented for $(T)_{in}$ and $(T)_{out}$ in Section 3.3.1. Modified for the Christiaensen Basin the mass loading term becomes:

$$\left(\frac{dA_x}{dt}\right)_{in} = \left[C_{x(dis)}\right]_{in} Q_w + \left[C_{x(part)}\right]_{in} Q_w + C_{x(sed)}\ Q_{rs}\ A_s + Q_{x(dep)}\ A_s$$

For this study the net evaporation of surface water at the site is considered negligible compared to the flow of water through the site and Q_w is the same for water being advected into and out of the site. Pore water flux is negligible at the Christiaensen Basin and, therefore, no pore water contaminants are fluxed into site waters. Since there is no direct dumping in the Basin, the ocean dumping term drops out. Resuspended sediments are a source of contaminants to the water column and contribute to the water column mass loading. Thus, the input term is composed of two advection terms, an atmospheric flux term and a resuspension term.

The removal equation, modified for the Christiaensen Basin water column box is:

$$\left(\frac{dA_x}{dt}\right)_{out} = \left[C_{x(dis)}\right]_{out} Q_w + \left[C_{x(part)}\right]_{out} Q_w$$

Sedimentation in the box is negligible and, therefore, sedimentation is not a removal process for contaminants in the water column. Since the microbial degradation rate for all contaminants in the water column is much slower than the residence time of water being advected through the box, this term (Z) does not enter into the calculation of $(T)_{out}$.

Therefore, since the biodegradation and sedimentation terms are negligible, $(T)_{out}$ is calculated solely from the advection of contaminants out of the box.

3.7.2 Generic Parameters

3.7.2.1 Advective Transport

The main feature of New York Bight circulation is a relatively slow cyclonic (counterclockwise) flow to the southwest over most of the shelf with some indication of a clockwise gyre in the inner Bight (Hansen, 1977; Ingham, 1982). In addition, there is an exchange circulation characterized by a seaward flow of estuarine surface water, and landward flow of deeper waters between the Hudson/Raritan estuary and the open waters of the New York Bight. During conditions of high flood the effect of the seaward flow can be detected 64 km offshore in the Hudson Shelf Valley (Hansen, 1977). During such times the seaward flow of fresh water would also impact the water overlying the Christiaensen Basin. All of the above circulation features can be masked on a day-to-day basis by variable wind-driven currents and can be altered for larger periods during prolonged weather events (Hansen, 1977).

For the purposes of this study a southwesterly drift of water (3.7 cm sec^{-1}) over the box (EPA, 1981) is assumed. At this velocity, the residence time of water in the box is 3.4 days which becomes the upper limit of $(T)_{out}$ for contaminants in the water column.

3.7.2.2 Transport of Suspended Matter

The advective transport of suspended matter into the box is calculated to be 6.75×10^8 kg yr^{-1} assuming a 3.7 cm sec^{-1} southwesterly drift of water over the site and an SPM load of 2500 µg L^{-1}. Natural sedimentation in the box is calculated to be 0.11 g $cm^{-2}yr^{-1}$, whereas resuspension is calculated to be 0.25 g cm^{-2} yr^{-1}. The net resuspension rate in the box is then estimated to be 0.14 g cm^{-2} yr^{-1}. To maintain a sediment mass balance, advection must transport 7.91×10^8 kg yr^{-1} out of the box.

The bottom sediment resuspension rate of 0.25 g cm^{-2} yr^{-1}, calculated from the Freeland and Swift (1978) analysis of historical data, corresponds well with the estimation of Butman and Moody (1983) that an average of 5-10 storms per year will resuspend approximately 0.1 to 0.2 g cm^{-2} yr^{-1}. Because of the relatively high resuspension rate, sediment will constantly be resuspended and mixed with natural SPM before being advected out of the box. The surface layer of sediment (floc), therefore, is similar or identical in nature to SPM. However, sediment contaminant data used in this study are average values in the surface 5 cm of bulk sediment and, therefore, do not reflect SPM concentrations qualitatively or quantitatively.

In calculating the input of pollutants in the sediment box model, we assume that all the contaminants being deposited in the sediment mixed layer are contained on SPM from coastal and offshore water being advected through the site. These particulate contaminant fluxes to the New York Bight Apex may include river, atmospheric, sewage dumping, or dredged material dumping inputs. However, as the Christiaensen Basin is relatively isolated from direct inputs, the net result is that individual mass loadings are integrated into the "natural" SPM and the source of the contaminants is not identified in the data. The mass loading of contaminants from the Lower Bay and New York Sewage Sludge Site are addressed in Section 3.7.6.

3.7.2.3 Mass Loading of Contaminants

The mass loading of contaminants entering the water column box from the advection of natural SPM and atmospheric fallout is shown in Table 3.7.1. The atmospheric deposition should be viewed as input to the water column, not necessarily to the sediment in the box. The atmospheric contaminant inputs are generally much smaller than the contaminants advected on suspended sediment and are only of significance for PAH and Pb. Although SPM concentrations ranged from 900-3000 µg SPM L^{-1} in the data used for the study, the mass loading to the site is calculated for a rough mean of 2500 µg SPM L^{-1} in site waters.

3.7.2.4 Sediment Chemistry

The concentration of contaminants in Christiaensen Basin sediments, mixed layer inventory, and annual resuspension of contaminants from the mixed layer are presented in Table 3.7.2. Concentrations selected for the box model are mean concentrations from the data base. The standard deviation of the mean of certain contaminants equals sediment contaminant values and presents a difficulty in justifying a mean concentration. The mixed layer inventories are calculated assuming that selected box model sediment contaminant (mean) concentrations are distributed evenly throughout a 0-5 cm mixed layer of dry sediment density 0.6 g cm^{-3}. Since any uncertainty in sediment contaminant levels is directly reflected in contaminant inventories, that uncertainty is propagated directly into both $(T)_{in}$ and $(T)_{out}$ calculations. The sensitivity

Table 3.7.1 Mass Loading of Contaminants Entering the Christiaensen Basin.

	Contamination on Suspended Sediment (ug g^{-1})		Contaminant Advected[c] Suspended Sediment (kg)	Atmospheric[d] Deposition (kg yr^{-1})	Total Particulate Inputs[e] (kg yr^{-1})
	$\bar{x} \pm s$	Selected Concentrations Box Model			
PCB	0.22 ± 0.10[a]	0.2	135	0.06	135
PAH	2.5 ± 2.5[a]	2.5	1,690	240	1,930
Pb	41 ± 25[b]	40	26,900	2300	29,200
Cu	52 ± 40[b]	50	33,700	690	34,400
Cd	4.8 + 4.5[c]	5	3,400	41	3,440

[a] Boehm (1983b)

[b] Betzer (1978)

[c] Mean SPM concentration of 2500 µg L^{-1}

[d] Fallout and rainout calculated from data presented in Duce and Kester (1981) PAH data from Gschwend and Hites (1981)

[e] Equal to advected contaminants plus atmospheric flux

Table 3.7.2 Concentrations, Inventory and Annual Resuspension of Contaminants for Christiaensen Basin Sediments.

	$\bar{x} \pm s$ ($\mu g\ g^{-1}$)	Range ($\mu g\ g^{-1}$)	Selected Concentrations ($\mu g\ g^{-1}$)	Mixed Layer Inventory (kg)	Annual Resuspension (kg)
PCB	0.219 ± 0.113[a]	0.750 - 1.500[b]	0.20	500	44
PAH	15.8 ± 12.9[a]		16	40,000	3,360
Pb	63 ± 21[c] 68 ± 42[d]	9 - 242	65	163,000	13,600
Cu	55 ± 29[d] 103 ± 25[e]		75	187,000	15,800
Cd	1.3 ± 0.7[d] 3.3 ± 0.6[e]		2.3	5,750	480
Hg	1.0 ± 0.9[d]		1.0	2,500	210

[a] Boehm (1983a)
[b] MacLeod et al. (1981)
[c] Harris and Waschitz (1982)
[d] EPA (1981)
[e] O'Reilly and Zdanowicz, personal communication
[f] Concentrations selected for use in residence time calculations

of the calculated residence times to this type of uncertainty is discussed more completely in Section 3.7.4.

The annual resuspension of contaminants from the mixed sediment layer is calculated from the mean sediment concentration using the resuspension rate of 0.25 g cm^{-2} yr^{-1}. Since there is no net burial in the box, the burial term is zero and resuspension is generally the only mechanism for contaminants to be removed from the sediment mixed layer.

3.7.2.5 Contaminants in the Water Column

The concentrations of contaminants in the water column of the Christiaensen Basin are presented in Table 3.7.3. Particulate PCB and PAH data are from Boehm (1983b) and were taken at stations near the Christiaensen Basin. Particulate metals data are from Betzer (1978). For the purposes of calculation it is assumed that there is a uniform 2.5 mg L^{-1} suspended matter in the box; that total PCB and PAH are twice particulate levels; and that Pb, Cu and Cd partition between dissolved and particulate phases as in Puget Sound (presented in Table 3.6.3). The advective flux of contaminants into the box is calculated assuming a net drift of 3.7 cm sec^{-1} (8570 m^3 sec^{-1}). The inventory of contaminants was calculated by multiplying the total contaminant concentration by the volume of water in the box.

3.7.3 Residence Times in the Christiaensen Basin

3.7.3.1 Residence Times in the Sediment

The residence times (T_{in}) of contaminants in the sediment mixed layer have been calculated using deposition and resuspension rates from Freeland and Swift (1978), and mean concentrations for contaminants in the mixed layer (Table 3.7.4). Since the simplified $(T)_{out}$ residence time equation for PCB, Pb, Cu, and Cd contains only sediment contaminant levels and resuspension rates as variables, and since contaminant levels appear in both numerator and denominator, $(T)_{out}$ for these contaminants are the same. $(T)_{out}$ for PAH is shorter as a result of the microbial degradation of PAH in the sediments (Section 3.7.1).

With the exception of PAH the residence times calculated for $(T)_{in}$ are generally within the same order of magnitude as $(T)_{out}$ and average about 31 years. The mean residence time for all contaminants (except PAH) calculated from removal processes is 12 years.

Table 3.7.3 **Christiaensen Basin Water Column Chemistry.**

Contaminant	Dissolved (ug L^{-1})	Particulate (ug L^{-1})	Total (ug L^{-1})	Advective Flux (kg yr^{-1})	Inventory Water Column (kg)
PCB[a]	0.00055	0.00055	0.0011	300	2.7
PAH[a]	0.00625	0.00625	0.0125	3,310	31.2
Pb[b]	0.15	0.10	0.25	67,500	625
Cu[b]	7.6	0.13	7.7	2,079,000	19,250
Cd[b]	0.77	0.012	0.78	210,600	1,950

[a] Particulate PCB, PAH from Boehm (1983b). Dissolved levels assumed equal to particulate levels.

[b] Pb, Cu, Cd particulate levels from Betzer (1978). Dissolved Pb concentration set equal to particulate Pb. Dissolved Cu and Cd calculated assuming partitioning between dissolved and particulate phases the same as in Puget Sound waters (Section 3.6).

Table 3.7.4 Residence Times of Contaminants in the Sediment Mixed Layer of the Christiaensen Basin Calculated from Sedimentation and Resuspension Rates Using Mean Sediment and SPM Concentrations.

Contaminant	$(T)_{in}$[a] (years)	$(T)_{out}$[b] (years)
PCB	27.3	12
PAH	172	11.6
Pb	44.8	12
Cu	41.0	12
Cd	12.6	12

[a] Calculated assuming a deposition rate of 0.11 g cm^{-2} yr^{-1}.

[b] Calculated assuming a resuspension rate of 0.25 g cm^{-2} yr^{-1} and a degradation rate of 1.2 µg PAH cm^{-2} yr^{-1}.

3.7.3.2 Residence Times in the Water Column

The residence times of contaminants in the water column calculated from mass loadings are presented in Table 3.7.5. The calculated mean residence time of all contaminants is approximately 0.008 year (2.8 days) or slightly less than the residence time of water at the site. Since the only net removal process in the water column model is advective transport, the residence time of all contaminants calculated from removal processes is 0.009 year (3.4 days), the same as the residence time of water at the site. The similarity of $(T)_{in}$ and $(T)_{out}$ values is an indication of the overall predominance of advective transport in both mass loading and removal processes. Since advective transport and water column inventory are both functions of water column contaminant concentrations, any error in this measurement is cancelled out in the calculation of residence time.

3.7.4 Sensitivity Analysis

3.7.4.1 Sediment Mixed Layer

The calculation of residence times for the sediment mixed layer is subject to two types of uncertainties; those resulting from uncertainties in generic parameters and those specific to each contaminant. Uncertainties in generic parameters should affect all contaminant residence times equally and thus, primary attention must be focused on this aspect of the data set before confidence may be placed in any one contaminant's residence time. $(T)_{in}$ calculations are subject to both types of uncertainties. However, concentration and inventory data cancel out of the $(T)_{out}$ equations and, thus, $(T)_{out}$ is subject to uncertainties only in the generic parameters.

The simplified $(T)_{out}$ residence time equation contains only three generic parameters: the depth of the sediment mixed layer, the density of sediment in the mixed layer, and the resuspension rate. The depth of the sediment mixed layer is defined as 5 cm for the purpose of these calculations. The actual mixed layer depth can be quite variable depending on the sediment type and seasonality, and may encompass as much as 0-10 cm, or as little as 0-2 cm (B. Brown, personal communication). With such variation, the actual depth of the sediment mixed layer should be considered to be 5 cm ± 3 cm, a 60% uncertainty. The density of the mixed layer is defined as 0.6 g dry sediment cm^{-3}. The actual density may vary ± 0.1 g depending on the water content in the mixed layer. The resuspension rate is the only net removal process considered in $(T)_{out}$ calculations.

Table 3.7.5 **Residence Time (T_{in}) of Contaminants in Water Column of Christiaensen Basin Calculated from Mass Loading.**

Contaminant	Units Water Column[a] (kg)	Units Mass Loading[b] ($kg\ yr^{-1}$)	T_{in} (Years)
PCB	2.7	340	0.008
PAH	31.6	97,800	0.005
Pb	625	83,400	0.008
Cu	19,250	2,095,500	0.009
Cd	1,950	211,100	0.009

[a] From Table 3.7.3

[b] Advective flux (Table 3.7.3); Atmospheric deposition (Table 3.7.1); Annual resuspension (Table 3.7.2)

Sedimentation and burial are not considered removal processes based on Freeland and Swift's (1978) analysis of Christiaensen Basin topography. Since there is very little information to indicate the magnitude of uncertainty in Freeland and Swift's data, a conservative estimate considered in the sensitivity analysis is ± 100%. Since a low resuspension rate would be balanced by a complementing burial term, analysis of the uncertainty in the resuspension term is limited to an increase in the resuspension rate. Table 3.7.6 presents the effect of the uncertainty in generic parameter values expressed as calculated $(T)_{out}$ residence times.

Residence times for contaminants in the sediment mixed layer calculated as $(T)_{out}$ may vary from 5-19 years depending on the reasonable uncertainties of individual generic parameters. Combining uncertainties can lead to greater variation in calculated $(T)_{out}$. A $(T)_{out}$ of 2.4 years results from a mixed layer of 2 cm, sediment density of 0.6 g cm^{-3}, and a resuspension rate of 0.5 g cm^{-2} yr^{-1}. Although the best estimate of $(T)_{out}$ is 12 years, residence times based on generic removal processes may range from 2.4-19.2 years, almost an order of magnitude.

The biodegradation term adds another uncertainty to the calculation of $(T)_{out}$ for PAH. Lee and Ryan (1983) have shown that the potential half-life of PAH in mesoscale sediment experiments may vary from 5-21,000 days depending on the individual aromatic compound and seasonality. The same study found that the half-life varied from 16-470 days in estuarine sediments. Most biological activity was found to take place in the upper few millimeters of sediment, with low microbial activity below this layer. The calculation of the output rate, resulting from microbial degradation of PAH, is based on a half-life of 2 years with all microbial activity taking place in the upper 0.5 cm of the sediment mixed layer. Changing the depth of microbial activity to 0.1 cm or 1.0 cm changes $(T)_{out}$ to 11.2 and 7.5 years, respectively. Keeping the active layer in the upper 0.5 cm of sediment, but changing the half-life to 0.1, 1 and 10 years changes $(T)_{out}$ to 1.7, 7.5, or 11.2 years, respectively. Since not enough work has been done in this field to define this probable error with any certainty, the uncertainty of the output term due to microbial activity is best assessed by noting the above range in calculated $(T)_{out}$ values in response to changing the half-life and the depth of the biodegradation zone.

Contaminant residence times in the sediment mixed layer calculated from mass loading data are, in addition to uncertainties in generic parameters, subject to uncertainties in environmental measurement data such as sediment contaminant concentrations and suspended sediment contaminant concentrations. A review of the data (Tables 3.6.1 and 3.6.2) reveals considerable uncertainty for almost all contaminants. Examples of the effect of the uncertainty in PAH data are presented in Tables 3.7.7 and 3.7.8.

Table 3.7.6 Sensitivity Analysis of $(T)_{out}$ Based on Uncertainty in Generic Parameters in Christiaensen Basin Sediment Box.

Generic Parameter	$(T)_{out}$[a] (years)
Depth of Mixed Layer	
2 cm	4.8
5 cm	12.0
8 cm	19.2
Density of Mixed Layer	
0.5 g cm^{-3}	10
0.6 g cm^{-3}	12
0.7 g cm^{-3}	14
Resuspension Rate[b]	
0.25 g cm^{-2} yr^{-1}	12
0.50 g cm^{-2} yr^{-1}	6
Alternative Generic Parameters[c]	2.4

[a] Microbial degradation rate, Z = 0

[b] Alternative generic parameters: mixed layer depth = 2 cm; sediment dry density = 0.6 g cm^{-3}; resuspension rate = 0.5 g cm^{-2} yr^{-1}.

[c] Alternative parameters present a different set of generic parameters which could represent the state of the Christiaensen Basin sediment mixed layer.

Table 3.7.7 Effect of the Uncertainty in PAH Sediment Data on $(T)_{in}$.

Uncertainty Expressed as Standard Deviation[a]	$(T)_{in}$ (years)
$\bar{x} - s$	31.6
x	172
$\bar{x} + s$	312

[a] Refer to Table 3.7.2 for PAH sediment data

Table 3.7.8 Effect of the Uncertainty in PAH Suspended Sediment Data on $(T)_{in}$.

Uncertainty Expressed as Standard Deviation[a]	$(T)_{in}$ (years)
$\bar{x} - 0.95s$	34,500
x	172
$\bar{x} + s$	86

[a] Refer to Table 3.7.1 for PAH suspended sediment data

Uncertainties in sediment and suspended sediment contaminant concentrations can have a profound effect on the calculation of residence times. The residence time of PAH in the sediment mixed layer can vary from 31.6-312 years as PAH concentration in sediment varies from $\bar{x}$-s to $\bar{x}$ + s, respectively. Further, the residence time is shown to vary from 34,500-86 years corresponding to reported PAH suspended sediment concentrations.

PAH was selected for the sensitivity analysis because the PAH environmental data exhibited the most variability. However, other contaminant data exhibit similar uncertainties which translate to a similar wide range in calculated residence time values.

3.7.4.2 Water Column

Residence times for contaminants in the water column are also subject to generic and contaminant-specific uncertainties. However, the predominance of the advection term in both $(T)_{in}$ and $(T)_{out}$ calculations overwhelms the other terms, and a sensitivity analysis of water column residence times can be reduced to an analysis of the advection term.

The surface drift of 3.7 cm sec^{-1} is a conservative estimate of net currents at the Christiaensen Basin based on information in EPA (1982). The same document cites studies reporting mean flows of 15 cm sec^{-1} and 5 cm sec^{-1} in surface waters, and 2 cm sec^{-1} in bottom waters. Therefore, the flow through the box is subject to spatial and temporal variations, and assigning a mean net flow is subject to uncertainty. Assigning a one-half order of magnitude window about 3.7 cm sec^{-1} is probably a good estimate of the uncertainty in this term. This range includes net flow values of 1.2, 3.7, and 11.7 cm sec^{-1} through the box, yielding residence times calculated as $(T)_{out}$ of 10.7, 3.4, and 1.1 days, respectively.

3.7.5 Analysis of Model Results

Since contaminants are presumed to be in equilibrium with the sediment mixed layer, residence times calculated from contaminant mass loading should equal residence times calculated from removal processes. Microbial degradation and pore water flux are negligible for all contaminants and residence times calculated as $(T)_{out}$ are, therefore, essentially equal for all contaminants. Thus, not only should $(T)_{in}$ equal $(T)_{out}$ for each contaminant, but the residence times of all the contaminants selected for the study should equal each other.

Residence times calculated as $(T)_{in}$ range from 12.6 years for Cd to 172 years for PAH. The mean residence time (T_{in}) of four contaminants, excluding PAH, is 31.4 years which compares with the $(T)_{out}$ value of 12 years. The difference in the mean $(T)_{in}$ of 31 years and $(T)_{out}$ of 12 years is probably a function of the uncertainty in the generic parameter values. If the difference were due to contaminant-specific uncertainties, then one would still expect the mean $(T)_{in}$ to equal $(T)_{out}$. However, the fact that the mean $(T)_{in}$ is three times higher than $(T)_{out}$ indicates that deposition rates are actually higher than reported or that resuspension rates are lower.

$(T)_{in}$ for PAH is considerably higher than other $(T)_{in}$ values and 14 times higher than $(T)_{out}$, indicating that PAH inventory is an overestimate of actual mixed layer levels or that particulate PAH data underestimate actual levels. The PAH sediment concentration is a mean of 15 samples taken in three separate years (Boehm, 1983a). The mean PAH sediment concentration for each of two years was 9.5 $\mu g\ g^{-1}$ and 7.2 $\mu g\ g^{-1}$ (each $n = 5$). However, during the third year, mean PAH sediment concentration was 30.8 $\mu g\ g^{-1}$ ($n=5$), a level that considerably raised the mean calculated from all data. If the inventory were calculated from PAH levels from the two former surveys discarding the high levels, then the residence time, $(T)_{in}$, for PAH would be 89.9 years, a value closer to those of other contaminants.

Particulate PAH levels ranged from 0-370 ng PAH L^{-1} ($n = 8$; Boehm, 1983b) in the New York Bight. Such a wide range of contaminant levels indicates that the "natural" SPM in the Bight is not actually homogeneous, but is composed of suspended matter from different sources with different levels of contamination making the calculation of any mean concentration value difficult to justify without a much higher sampling frequency.

3.7.6 Sediment and Water Quality for Hypothetical Loadings

The advective flux of contaminants through the Christiaensen Basin is compared to the total mass loading of contaminants to the New York Bight Apex from sewage sludge dumping and flux of contaminants from the Hudson-Raritan estuary (Table 3.7.9). An analysis of the data indicates that assigning contributions of each mass loading to match the advective transport of contaminants at the Christiaensen Basin is an impossible task. Thus, for the purpose of analysis of hypothetical mass loadings, we have simply assumed two hypothetical baseline situations: 1) that sewage sludge dumping contributes either 10% or 2) 50% of the suspended matter advected through the site. A loading factor of 1X represents present day sewage sludge dumping activities which

Table 3.7.9 Mass Loading to New York Bight Apex from Advection of Estuarine/Riverine Water and Sewage Sludge Dumping Compared to Advective Flux of Contaminants Through Christiaensen Basin.

	Mass Flux to New York Bight Due To Advection (kg yr^{-1})		Advective Transport to Christiaensen Basin[c] (kg yr^{-1})
Contaminant	Sewage Sludge[a]	Lower Bay Complex[b]	
PCB	605	610	300
PAH	4,900	3,310	3,310
Pb	440,000	40,030	67,500
Cu	420,000	123,200	2,079,000
Cd	26,000	7,700	210,600

[a] PCB data from Boehm et al. (1985); other data from O'Connor et al. (1983).
[b] From Table 3.8.3
[c] From Table 3.7.3

presently contribute 2.7×10^8 kg yr^{-1} suspended solids contained in 7×10^6 m^3 yr^{-1} of wet sludge dumped (O'Connor et al., 1983). A loading factor of 0.1X represents a decrease by a factor of 10 in the advective transport of contaminants directly attributable to sludge dumping with other mass loads, such as atmospheric deposition and riverine/estuarine run-off remaining constant. A loading factor of 2X would represent a doubling of such loading.

The concentrations of contaminants in the sediment mixed layer have been calculated for several hypothetical loading conditions (Table 3.7.10). The concentrations were calculated assuming that the box reaches equilibrium under each loading condition and that the residence times of contaminants do not change under hypothetical loading conditions. Table 3.7.11 presents the concentration of contaminants in the water column at the Christiaensen Basin resulting from hypothetical loadings of 0.1, 1, 2, and 10 times 1981 levels. As with the sediment box, mass loadings directly attributable to activities such as dumping are assumed to be responsible for 10% or 50% of the total advective transport through the box. Under most loading conditions contaminant concentrations in the water column remain below EPA water quality criteria. Presently, total Cu concentrations at the Christiaensen Basin are above the EPA criterion. The levels remain above the criterion under all hypothetical conditions.

3.7.7 Summary and Research Needs

The residence times for contaminants in the sediment mixed layer at the Christiaensen Basin calculated as $(T)_{in}$ are in the range of 12-172 years whereas $(T)_{out}$ is uniformly 12 years. The residence times of contaminants in the water column range from 2-3 years, reflecting the residence time of water at the site.

To improve the quality and reduce the uncertainty of the residence times calculated for the Christiaensen Basin, future environmental research should be guided by the needs of the contaminant box model. Water quality data should include dissolved and particulate contaminant measurements as well as total particulate matter (SPM) information. Complete water data sets were not available for any contaminant modeled and gaps in the data were filled by extrapolation from other data sets. Water quality information was completely unavailable for Hg, As, or Ag in the data reviewed. Particular attention should be paid to these elements.

Sediment contaminant information was, in general, available but somewhat limited for the Christiaensen Basin. The wide differences in reported values, notably PCB, Cu, and Cd, made calculation of mean values difficult to justify. If possible, future

Table 3.7.10 Predicted Concentration of Contaminants in Christiaensen Basin Sediment Mixed Layer in Response to Hypothetical Loadings if Ocean Disposed Materials are Changed by Factors of 0.1, 2 and 10 Times Present Loadings, Assuming Present Loadings are 10% and 50% Total Mass Loading at Site[a].

	Christiaensen Basin 1981-1983 (µg g^{-1} dry wt)	Present Contribution = 10%[b] Total Mass Loading				Present Contribution = 50%[c] Total Mass Loading			
		Loading Factor				Loading Factor			
		0.1X	1X	2X	10X	0.1X	1X	2X	10X
			(µg g^{-1} dry wt)				(µg g^{-1} dry wt)		
PCB	0.20	0.18	0.20	0.22	0.38	0.14	0.20	0.27	0.90
PAH	16	14.6	16	17.6	30.4	11	16	21	64
Pb	65	59	65	72	124	46	65	85	250
Cu	75	68	75	83	143	44	75	108	383
Cd	2.3	2.1	2.3	2.5	4.4	1.3	2.3	3.4	12.1

[a] Calculations assume equilibrium achieved at each loading level.

[b] f = 0.1

[c] f = 0.5

Table 3.7.11 Concentration of Contaminants in the Christiaensen Basin Water Column in Response to Hypothetical Loadings if Ocean Disposed Materials are Changed by Factors of 0.1, 1, 2 and 10 Times Present Loading, Assuming Present Loadings are 10% and 50% of Total Mass Loading at Site[a].

	EPA Seawater Criteria (μg L^{-1})	Christiaensen Basin 1981-1983	Present Contribution = 10%[b] Total Mass Loading, Loading Factor (μg L^{-1}) 0.1X	1X	2X	10X	Present Contribution = 50%[c] Total Mass Loading, Loading Factor (μg L^{-1}) 0.1X	1X	2X	10X
PCB	0.019	0.0011	0.001	0.0011	0.0012	0.002	0.0007	0.0011	0.0018	0.0061
PAH	8	0.0125	0.011	0.0125	0.0317	0.0237	0.0068	0.0125	0.0168	0.068
Pb	8.6	0.25	0.23	0.25	0.27	0.47	0.14	0.25	0.38	1.38
Cu	2.0	7.7	7.0	7.7	8.5	14.6	4.2	7.7	11.6	42.3
Cd	12	0.78	0.71	0.78	0.86	1.48	0.42	0.78	1.17	4.3

[a] Calculations assume equilibrium achieved at each loading level. [b] f = 0.1
[c] f = 0.5

studies of sediment contaminant loading should be coordinated to provide a complete synoptic set of contaminant data from individual sediment replicates. This approach would unify the seemingly isolated studies that have provided all the data used in this study.

One of the greatest needs directed at improving the modeling results is that of defining sedimentation and resuspension rates at the site (i.e., generic parameter values). Age-dated cores along with sediment contaminant data profiles would greatly improve the state of knowledge in this regard. Further, resuspension of bottom sediments in this semi-enclosed basin is probably different from the remainder of the shelf in the New York Bight and site-specific studies would be needed to obtain better data than that available.

Lastly, the use of source tracers and contaminant pattern recognition to define the individual mass loading contributions of sludge dumping, dredged material dumping, riverine/estuarine input, and other inputs is needed before a comprehensive analysis of $(T)_{in}$ for both water and sediment can be made. This information is seriously lacking in the present data set, making an assessment of the effect of present mass loading on the Christiaensen Basin difficult to calculate and, further, making management decisions based on environmental data from the Christiaensen Basin difficult to justify.

3.8 Lower Bay Complex

The Lower Bay Complex of the Hudson-Raritan estuary is a triangular-shaped coastal estuary covering an area of 290 km^2 (Figure 3.3). The Lower Bay Complex includes the mouths of the Hudson River at the Narrows, Raritan River, and Jamaica Bay, and includes the Raritan Bay, Sandy Hook Bay, and Lower Bay. It is open to the Hudson River through a narrow opening called The Narrows and to the New York Bight through the Sandy Hook-Rockaway Point Transect.

The Lower Bay Complex receives inputs of many types of pollutants resulting from its use as a shipping waterway, sewage and industrial effluent receiving basin, fishing ground, and mining area. Approximately 100 m^3 sec^{-1} of treated and untreated municipal and industrial effluent are discharged into the Hudson-Raritan Estuary (Olsen et al., 1984), 5.5 m^3 sec^{-1} directly in the Lower Bay Complex (Mueller et al., 1982). In addition, chronic contaminant inputs from shipping, atmospheric fallout, riverine inputs, and coastal (urban) runoff contribute to contamination of the Lower Bay Complex.

The Lower Bay Complex is a relatively shallow (5 - 20 m) body of water with uneven bottom topography comprised of shoals, banks, and ship channels. Depending on the location within the bay system, areas within the Lower Bay Complex may be classified

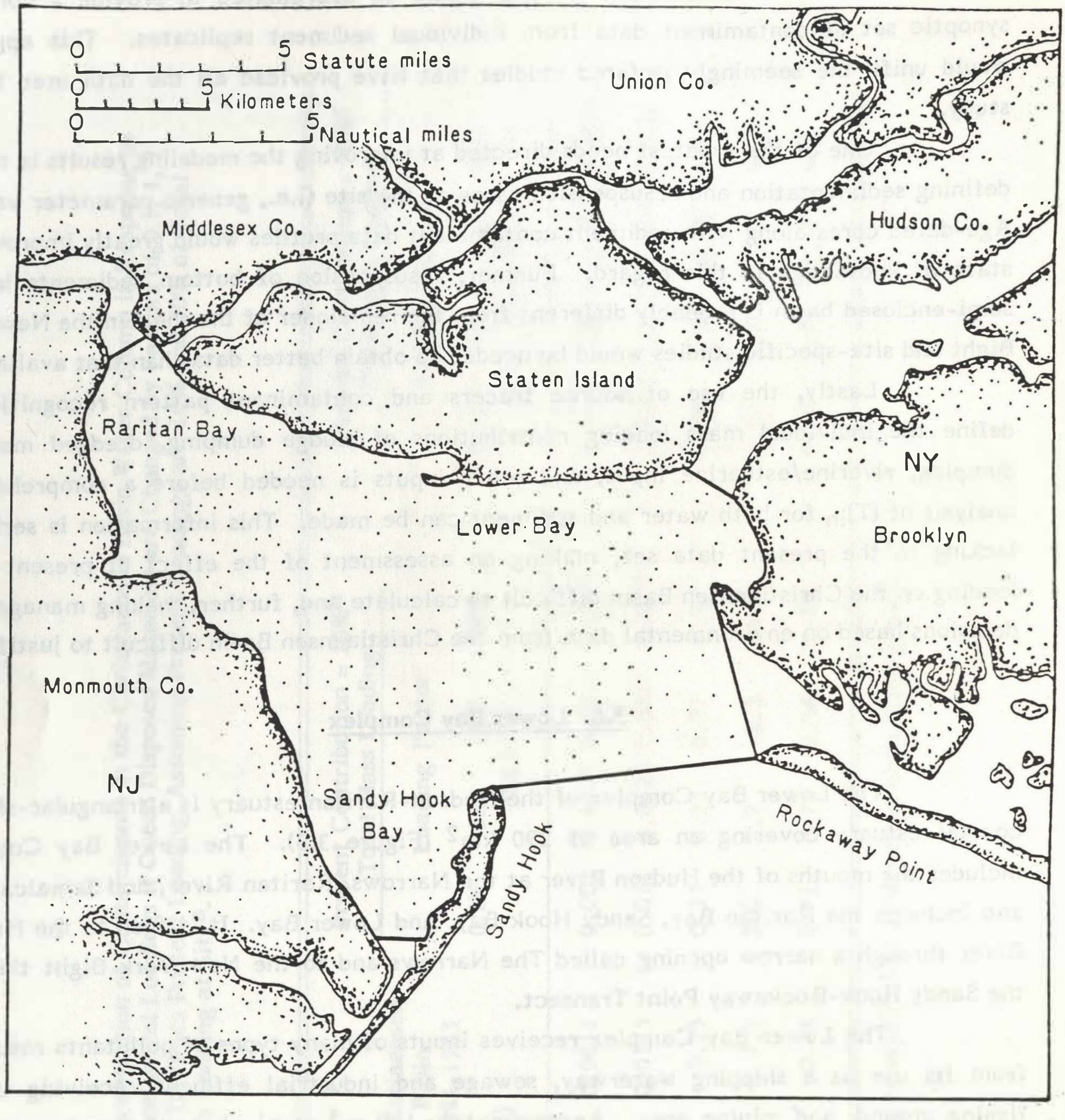

Figure 3.3 The Lower Bay Complex.

as either relatively high energy or low energy environments. The calculated sedimentation rate in dredged gravel pits is 2.2 cm yr^{-1}, whereas the calculated rate for dredged channel areas is 0.5 cm yr^{-1}, and off-channel areas is 0.15 cm yr^{-1} (Olsen et al., 1984).

3.8.1 Structure of the Box Model

The box model for the Lower Bay Complex consists of an irregularly shaped water column box approximately 19 km wide, 28 km long, and 10 m deep. The area of the box is 290 km^2 and the volume of the water column is 2.9 x 10^9 m^3. The area of the sediment box is also 290 km^2 and the thickness of the bioturbated mixed layer is assumed to be 5 cm. With a density of 0.7 g cm^{-3} the weight of sediment in the mixed layer (Olsen et al., 1984) is, therefore, 10.2 x 10^9 kg.

Residence times in the water column and sediment boxes for the Lower Bay Complex are calculated for PCB, PAH, Pb, Cu and Cd.

3.8.1.1 Sediment Box Model

Based on the general equation (4) for mass loading to the sediments, the expression modified for the Lower Bay Complex model is:

$$\left(\frac{dA_x}{dt}\right)_{in} = \left[C_{x(tsm)}\right]_{in} Q_{sed}$$

Since there is no authorized dumping in the Lower Bay Complex, $Q_{od} = 0$ and since the production of contaminants in the time scale of the model is assumed to be negligible (P_x=0), the mass loading term is strictly a function of the sedimentation of contaminated particles. Suspended solids present in wastewater discharged into the Lower Bay Complex will be deposited to some extent in the sediments of the box. However, this contribution is presumably accounted for in suspended sediment contaminant levels and deposition rates. The mass loading due to sewage effluent is considered directly in the water column box (Section 3.8.1.2) and in the section on Hypothetical Loading (Section 3.8.6).

The general removal equation (5) is used to calculate $(T)_{out}$ for the sediment model.

Major contaminant removal processes in mixed layer sediments are burial and resuspension. The burial term, calculated from net sedimentation rates in both dredged and non-dredged areas, includes the material removed by dredging. Since burial and dredging both effectively remove contaminants from the sediment box, they are combined as a single "burial" term. Resuspension, of the type that removes sediment from the mixed layer, occurs relatively infrequently during storm events. The resuspension removal process is not included in the simplified box model but, instead, is addressed in Section 3.8.4 (Sensitivity Analysis). Pore water flux of contaminants from mixed layer sediments will also be considered since the deposition rate exceeds the resuspension rate. The rate of pore water flux is equivalent to the burial rate (Stumm and Morgan, 1981). Concentration of contaminants in pore water is estimated from equilibrium partition coefficients for PCB, Pb, Cu, and Cd. The partition coefficient of PAH is assumed equal to that of PCB. The microbial degradation (Z) is very low for PAH ($t_{1/2}$ = 2 years; Lee and Ryan, 1983) and zero for other contaminants modeled. Therefore, for PAH, $(T)_{out}$ is calculated using all removal terms, while for the other contaminants all terms of equation (5) except for microbial degradation (Z) will be used.

3.8.1.2 Water Column Box Model

Residence times for contaminants in the water column are calculated from the general $(T)_{in}$ and $(T)_{out}$ equations (6) and (7), respectively.

In general, pore water flux is the least significant of the mass loading terms. However, it is included for comparison to other terms.

The removal rate equation for the Lower Bay Complex is the general equation (7) minus the Z term. For the calculation of the removal rate the microbial degradation rate (Z) is assumed to be much slower than the residence time of water in the box and is not considered in the calculation. Thus, the removal term is composed solely of advection and sedimentation terms.

3.8.2 Generic Parameters

3.8.2.1 Advective Transport

The predominant currents in the Lower Bay System are tidally driven. Tides and tidal current patterns in the box are complicated by the shape of the box, variation in freshwater discharge, and intricate connection of dredged waterways (Duedall et al., 1979). In addition, tidal and non-tidal velocities are highly dependent upon weather, and large variations can exist during storm events. The mean tidal range in the box is 1.45 m (Duedall et al., 1979) corresponding to a minimum daily tidal flow into and out of the box of 1.7 x 10^9 m^3 day^{-1}. In addition, tidal currents in the Upper Bay, Jamaica Bay, and Arthur Kill transport an additional amount of water through the box. The net non-tidal currents in the box are the result of freshwater discharge from the Hudson, Raritan, and Passaic Rivers. The maximum gauged discharge through the box is 1200-1800 m^3 sec^{-1} and occurs during the months of March, April and May (Duedall et al., 1979). Non-gauged flow also enters the estuary increasing net discharge above documented levels.

For the purposes of this study the mean flow of freshwater into the box is 534 m^3 sec^{-1} (Olsen et al., 1984) and the wastewater discharge directly into the box is 5.5 m^3 sec^{-1} (Mueller et al., 1982). From mass-balance equations (Klinkhammer and Bender, 1981) the calculated flow of marine water into the box is 180 m^3 sec^{-1} and the total flux out of the box to the New York Bight is 720 $m^3 sec^{-1}$.

3.8.2.2 Transport of Suspended Matter

The advective transport of suspended matter into the box from riverine sources is calculated to be 5.55 x 10^8 kg yr^{-1} assuming a mean particle density of 33 mg L^{-1} (Olsen et al., 1984) and mean riverine flow of 534 m^3 sec^{-1}. The transport of suspended matter from the Bight to the Lower Bay Complex is calculated to be 0.11 x 10^8 kg yr^{-1} assuming an SPM load of 2.0 mg L^{-1} (Boehm, 1983b) and a mean seawater influx of 180 m^3 sec^{-1}. Suspended matter introduced to the box from wastewater discharges is calculated to be 0.07 x 10^8 kg yr^{-1} assuming secondary effluent suspended sediment of 40 mg L^{-1} (Mueller et al., 1982) and a flow of 5.5 m^3 sec^{-1}. In addition, in situ productivity in the box is calculated to be 1.2 x 10^8 kg yr^{-1} (from Olsen et al., 1984). Natural sedimentation rate is taken to be 0.11 g cm^{-2} yr^{-1} in non-dredged areas (255 km^2) and 0.35 g cm^{-2} yr^{-1} in dredged areas resulting in an average net deposition of suspended sediment in the box of 3.9 x 10^8 kg yr^{-1}. To maintain a sediment mass balance, 3.0 x 10^8 kg yr^{-1} suspended matter is transported out of the estuary.

3.8.2.3 Mass Loading of Contaminants

The mass loading of contaminants entering the box from the advection of suspended matter, urban runoff, sewage effluent, accidental spills, landfill leachate, and atmospheric fallout is shown in Table 3.8.1. Inputs from sewage effluents, urban runoff, accidental spills, and landfill leachate contain both dissolved and particulate contaminant loads. Riverine and marine contaminant fluxes are calculated from levels of contaminants on naturally occurring suspended matter. Atmospheric fluxes of metals and PCB are calculated from dry deposition and fallout data, whereas the PAH flux was empirically determined by Gschwend and Hites (1981).

The calculated fluxes were determined as much as possible for inputs directly into the Lower Bay Complex box and, for this reason, are lower than values reported for the entire Hudson-Raritan Estuary (Mueller et al., 1982; Breteler, 1985). For instance, sewage effluent fluxes reflect improvements in sewage treatment in the New York-New Jersey Metropolitan area and reflect the relatively small discharge into the Lower Bay Complex compared to the volume discharged by the entire metropolitan region into the Hudson-Raritan Estuary.

3.8.2.4 Sediment Chemistry

The concentration of contaminants in Lower Bay Complex sediments, the sediment inventory, and annual deposition and removal rates of contaminants in the box are presented in Table 3.8.2. Due to the wide range of reported contaminant levels in the sediments, the inventory and burial rate are calculated for selected "mean" concentrations. Treatment of the uncertainty of the data will be addressed in Section 3.8.4.

The wide range of observed sediment contaminant concentrations is most likely a result of the variations in sedimentation rates within the Lower Bay Complex. The mean sedimentation rates of 0.35 g cm^{-2} yr^{-1} in dredged areas and 0.11 g cm^{-2} yr^{-1} in non-dredged areas actually are representative of a much larger range in actual sedimentation rates. For example, a typical sedimentation rate in areas of the Hudson-Raritan Estuary requiring extensive dredging is 6.3 g cm^{-2} yr^{-1}, whereas sediment deposition is negligible in wide areas of Raritan Bay (Olsen et al., 1984). Contaminant burial rates for the box model are based on an annual burial of 3.9 x 10^8 kg of sediment: 1.2 x 10^8 kg yr^{-1} in dredged areas and 2.7 x 10^8 kg yr^{-1} in non-dredged areas.

Table 3.8.1. Mass Loading of Contaminants to the Lower Bay Complex.

	Sewage[a] Effluent (kg yr^{-1})	Riverine[b] (kg yr^{-1})	Urban Runoff[c] (kg yr^{-1})	Atmospheric[d] (kg yr^{-1})	Other Direct Inputs[e] (kg yr^{-1})	Marine[f] (kg yr^{-1})	Total (kg yr^{-1})
PCB	700	220	72	73	63	4.4	1,130
PAH	4,300	3,060	-	4,550	-	60.5	11,970
Pb	89,000	36,700	59,300	16,450	530	726	202,700
Cu	78,000	45,200	49,640	2,400	470	890	176,600
Cd	5,900	4,780	3,040	330	70	0.9	14,120
Hg	260	-	168	10	14	-	452
Ag	-	-	-	30	36	-	66
As	49,000[c]	-	-	770	100	-	49,900

[a] Sewage flux 5.5 m^3 sec^{-1}; PCB data from Boehm et al.(1985); PAH estimated from riverine PAH data; Metals data from Mueller et al. (1982).

[b] Riverine particle flux 5.56 x 10^8 kg yr^{-1} from Olsen et al. (1984); PAH, PCB data from Boehm (1983b); Metals data from Betzer (1978).

[c] 20% of total metropolitan New York-New Jersey runoff estimated from Mueller et al. (1982).

[d] PCB data from Mueller et al. (1982); PAH data from Gschwend and Hites (1981); Metals data from Duce and Kester (1981).

[e] Accidental spills, leachate - 20% of metropolitan New York-New Jersey total (Mueller et al., 1982).

[f] Marine particulate flux 0.35 x 10^9 kg yr^{-1} from Olsen et al. (1984); PCB, PAH data from Boehm (1983b); Metals data from Betzer (1978).

Table 3.8.2. Concentration, Inventory, and Annual Burial of Contaminants, Lower Bay Complex.

	Concentration		Selected for Use in Box Model (μg g^{-1})	Mixed Layer[e] Inventory (kg)	Gross[f] Deposition Rate (kg yr^{-1})	Net Burial Rate (kg yr^{-1})	Pore Water Flux (kg yr^{-1})
	$\bar{x}$ + s (μg g^{-1})	Range (μg g^{-1})					
PCB	0.44[a] 0.11[b] 0.4[c]	0.18 - 0.70 0.003 - 2.03	0.4	4,080	381	156	0.004
PAH	20.7[a]	13.0 - 28.9	20.0	204,000	2,097	7,800	0.22
Pb	113 $\pm$ 115[d] 195[c]		113	1,152,600	25,380	42,900	6.1
Cu	160 $\pm$ 177[d] 280[c]		160	1,632,000	31,011	62,400	890
Cd	2.7 $\pm$ 2.8[d]		3.0	30,600	3,280	1,170	1.7

[a] Boehm and Fiest (1982)

[b] Stainken and Rollwagen (1979)

[c] Olsen et al. (1984)

[d] Greig and McGrath (1977)

[e] Mixed layer inventory = (concentration of pollutant in sediment mixed layer) x (density 0.7 g cm^{-3}) x (volume of mixed layer).

[f] Concentration of pollutant in SPM (μg g^{-1}) x (SPM deposition rate 3.9 x 10^{8} kg yr^{-1})

For all contaminants, deposition fluxes are a function of contaminant levels on SPM, whereas removal fluxes (resuspension, burial, pore water flux) are a function of sediment concentrations. For PAH, Pb, and Cu, sediment concentrations are higher than reported contaminant SPM concentrations which leads to a situation where more contaminant is being removed from the sediment mixed layer than is being added. Assuming equilibrium, removal processes should equal mass loading. The net difference in the two processes is another measure of the uncertainty of the data and is a matter that will be discussed more completely in Section 3.8.4.

3.8.2.5 Contaminants in the Water Column

The concentrations of contaminants in the water column of the Lower Bay Complex are presented in Table 3.8.3. PCB and PAH particulate levels from the Sandy Hook-Rockaway Point Transect were measured by Boehm (1983b). These data were taken on an ebbing tide and presumably reflect levels in the Lower Bay. Pb, Cu, and Cd particulate levels were also measured at the Sandy Hook-Rockaway Point Transect (Betzer, 1978). For consistency in mass balance considerations, all particulate contaminant levels were normalized to 13.4 mg SPM L^{-1}, whereas actual reported levels were lower (2.0 mg SPM L^{-1}; Boehm, 1983b and 5.7 mg SPM L^{-1}; Betzer, 1978). This approach was taken to normalize much of the sampling/analytical variability that is present in the water chemistry data set. The uncertainty of the water chemistry data is discussed in Section 3.8.4.

The advective flux of contaminants to the New York Bight from the Lower Bay Complex was calculated assuming a 720 m^3 sec^{-1} flux of estuarine water out of the box. The inventory of contaminants was calculated by multiplying the concentration of total contaminant by the volume of water in the box.

3.8.3 Residence Times in the Lower Bay Complex

3.8.3.1 Residence Times in the Sediment

The residence times of contaminants in the Lower Bay Complex sediment mixed layer are given in Table 3.8.4. The residence times are calculated as either $(T)_{in}$, the mixed layer sediment inventory divided by the deposition rate (function of contaminant levels on SPM) or as $(T)_{out}$, the mixed layer sediment inventory divided by the summed burial, resuspension and pore water flux (function of sediment levels only).

Table 3.8.3. Lower Bay Complex Water Column Chemistry.

Contaminant	Dissolved[a] ($\mu g\ L^{-1}$)	Particulate[a] ($\mu g\ L^{-1}$)	Total ($\mu g\ L^{-1}$)	Advective[b] Flux to Ocean ($kg\ yr^{-1}$)	Inventory[c] Water Column (kg)
PCB	0.014	0.014	0.028	610	78
PAH	0.073	0.073	0.145	3,310	424
Pb	0.28	1.04	1.32	25,400	3,250
Cu	11.8	1.08	12.9	294,300	37,600
Cd	1.44	0.12	1.56	37,540	4,800

[a] Particulate contaminant levels from Boehm (1983b) and Betzer (1978) normalized to 13.4 mg SPM L^{-1}, amount calculated from mass balance considerations (Section 3.8.2.2). Dissolved PCB and PAH levels assumed equal to particulate levels. Dissolved Pb, Cu, Cd calculated assuming partitioning between dissolved and particulate phases the same as in Puget Sound waters (Section 3.6).

[b] Flux = 720 $m^3\ sec^{-1}$.

[c] Inventory = (total concentration of contaminant) x (volume of water box).

Table 3.8.4. Residence Times of Contaminants in the Sediment Mixed Layer of the Lower Bay Complex Using Deposition and Burial Rates, and Mean Sediment and SPM Concentrations.

Contaminant	T_{in}[a] (years)	T_{out}[b] (years)
PCB	10.7	26.9
PAH	97	16.8
Pb	45	26.9
Cu	53	26.9
Cd	9.3	26.9

[a] Calculated assuming a gross sedimentation rate of 0.13 g cm $^{-2}$ yr^{-1} evenly distributed over Lower Bay Complex.

[b] Calculated assuming a net burial rate of 0.13 g cm^{-2} yr^{-1} evenly distributed over Lower Bay Complex and a PAH degradation rate of 1.5 x 10^{-6} g PAH cm^{-2} yr^{-1}.

For PAH, microbial degradation is also considered as a removal process. For other contaminants decay rate is set equal to zero.

An average sedimentation rate of 0.13 g cm^{-2} yr^{-1} corresponds to a sedimentation rate of 0.11 g cm^{-2} yr^{-1} in non-dredged areas, and 0.35 g cm^{-2} yr^{-1} in dredged areas of the box. For the purpose of the calculation, the deposition rate is equal to the sedimentation rate, and does not include material deposited and immediately resuspended. Olsen et al., (1984) indicate that most material deposited is resuspended over one tidal cycle.

No attempt has been made to distinguish between burial and dredging in the calculation of residence times. Since burial and dredging both effectively remove contaminants from the sediment box they are combined as a single "burial" term. The combined burial and resuspension rate of 0.38 g cm^{-2} yr^{-1} assumes a net sedimentation of fine-grained material of 1.2 x 10^8 kg yr^{-1} in dredged areas and 2.7 x 10^8 kg yr^{-1} in non-dredged areas.

The mean residence time of contaminants calculated as $(T)_{in}$ is 43 years which is a result of mean sediment and SPM inventories, and a deposition rate of 0.13 g yr^{-1}. This indicates that for current conditions it has taken approximately 40 years to achieve the contaminant levels found in the mixed layer inventory. The residence time calculated as $(T)_{out}$ is 27 years. Thus, in approximately 27 years, the present inventory of contaminants will be buried, dredged, or suspended and removed from the sediment mixed layer.

3.8.3.2 Residence Times in the Water Column

The residence times of contaminants in the water column have been calculated as $(T)_{in}$ from mass loading data (Table 3.8.5) and as $(T)_{out}$ using removal processes (Table 3.8.6). Residence time values range from 0.016 year for Pb to 0.34 year for Cd. The mean residence times are 0.13 year (49 days) based on mass loading and 0.08 year (29 days) based on removal processes. Based on volume and flow considerations the residence time of water in the box is 0.13 year.

The residence time of water in the box (0.13 year) is calculated by dividing the volume of water in the box by the advective flux of water out of the box. This residence time should be considered the upper limit of the residence times of contaminants in the water column. Actual $(T)_{out}$ residence times are expected to be less due to the net removal of contaminants from the water column to the sediment mixed layer as is shown

Table 3.8.5. **Residence Time (T_{in}) of Contaminants in Lower Bay Complex Water Column Calculated from Mass Loading.**

Contaminant	Water Column Inventory[a] (kg)	Mass Loading[a] (kg yr^{-1})	T_{in} (years)
PCB	78	1,130	0.07
PAH	424	11,970	0.035
Pb	3,250	202,700	0.016
Cu	37,600	176,600	0.21
Cd	4,800	14,120	0.34

[a] From Table 3.8.3

Table 3.8.6. Residence Time (T_{out}) of Contaminants in Lower Bay Complex Water Column Calculated From Burial and Advection Removal Process.

Contaminant	Water Column Inventory (kg)	Removal Burial (kg yr^{-1})	Removal Advection (kg yr^{-1})	Removal Total Removal (kg yr^{-1})	T_{out} (years)
PCB	78	156	610	766	0.10
PAH	424	7,800	3,310	11,110	0.038
Pb	3,250	42,900	25,400	68,300	0.048
Cu	37,600	62,400	294,300	356,700	0.105
Cd	4,800	1,170	37,540	38,710	0.124

in Table 3.8.6. $(T)_{in}$ calculated from a completely different contaminant flux data set is in good agreement with $(T)_{out}$.

3.8.4 Sensitivity Analysis

The calculation of contaminant residence times in the sediment mixed layer and water column of the Lower Bay Complex is subject to two types of uncertainties: those resulting from uncertainties in generic parameter values and those uncertainties specific to each contaminant. Since uncertainties in generic parameters will effect all contaminant residence times equally, they must be addressed before confidence may be placed in any one residence time. $(T)_{in}$ calculations are subject to both types of uncertainties. However, contaminant-specific data cancel out of $(T)_{out}$ equations and, thus, $(T)_{out}$ is subject to uncertainties only in generic parameters.

3.8.4.1 Sediment Mixed Layer

The simplified $(T)_{out}$ residence time equation for the sediment mixed layer (Section 3.8.1.1) contains four generic parameters: the depth and density of the sediment mixed layer used to calculate inventory, and resuspension and burial rates used to calculate the removal rate. Pore water flux is a minor contaminant removal term for the modeled pollutants and is not considered in the sensitivity analysis. The mixed layer is defined as 0-5 cm in this study. The actual mixed layer depth can be quite variable depending on sediment type (i.e., location in the box) and seasonality, and may range from 0-10 cm in areas with active polychaete colonies to as little as 0-2 cm in stressed environments (B. Brown, personal communication). For the purpose of sensitivity analysis, depth of the sediment mixed layer is 5 cm ± 3 cm. The density of the mixed layer is defined as 0.7 g (dry sediment) cm^{-3} from data of Olsen et al. (1984). However, Bokuniewicz and Ellsworth (unpublished manuscript, 1983) report a sediment density of 0.6 g cm^{-3}. Therefore, the uncertainty in the sediment density is estimated to be ± 0.1 g cm^{-3}. The burial rate is calculated from Olsen et al. (1984) and the uncertainty is estimated by the authors to be ± 35 percent. Infrequent storm events are known to resuspend the mixed layer. During these events, the material would be transported out of the estuary and, therefore, resuspension must also be considered a removal process. Such resuspension rates are unknown for the Lower Bay Complex but it is expected that they would exceed those calculated by Butman and Moody (1983) for the Continental Shelf

(Section 3.10). For the sensitivity analysis, resuspension rates of 0.25, 0.50, and 1.0 g cm^{-2} yr^{-1} are selected to estimate the uncertainty of this parameter.

Table 3.8.7 presents the effect of uncertainty in generic parameters on the calculation of $(T)_{out}$ for the Lower Bay Complex sediment mixed layer. $(T)_{out}$ varies from 3.6-14.6 years in response to reasonable uncertainties in individual generic parameters. Combining uncertainties can lead to an even greater difference from values in Table 3.8.4. A $(T)_{out}$ of 1.1 years results from a sediment mixed layer of 0-2 cm, a sediment density of 0.6 g cm^{-3}, and a resuspension rate of 1.0 g cm^{-2} yr^{-1}. Thus, although the best estimate of $(T)_{out}$ calculated from generic parameters is 9.2 years, the uncertainty of the data indicates that actual residence times may range from 1.1-14.6 years.

The uncertainty of the biodegradation term adds another uncertainty in the calculation of $(T)_{out}$ for PAH. As discussed in 3.7.4.1, half-lives of PAH in estuarine sediments range from 16-470 days and can display a wider range in controlled mesoscale experiments (Lee and Ryan, 1983). Most biological activity was found to occur only in the upper few millimeters of sediment with low activity below this region. For the calculation of removal rates, Z is 1.76 μg cm^{-2} yr^{-1} ($t_{1/2}$ = 2 yr) assuming a 0.5 cm active layer. Changing the microbiologically active layer to 0-0.1 cm or 0-1.0 cm instead of 0-0.5 cm, changes $(T)_{out}$ to 8.9 or 6.3 years, respectively. Alternatively, changing the half-life of PAH in the sediment to 0.1, 1, or 10 years changes $(T)_{out}$ to 0.9, 4.8, or 8.4 years, respectively. Since the actual uncertainty of microbial decay parameters is in the same range as discussed above, $(T)_{out}$ for PAH is not known with any better certainty. $(T)_{out}$ for PAH should be somewhat less than that for other contaminants, but how much less is not known.

$(T)_{in}$ values calculated from mass loading data are subject to contaminant-specific uncertainties in addition to uncertainties of generic parameters. Table 3.8.2 presents the uncertainty of sediment contaminant concentration expressed as range or standard deviation. For an extensive metals data set (Greig and McGrath, 1977) in the Lower Bay Complex (n = 130 samples) the standard deviation is greater than the mean. The large variation in metals data most likely reflects the range of depositional environments in the box and not analytical variability. The Lower Bay Complex is characterized by areas of extremely high depositional rates and areas in which resuspension is in equilibrium with the incoming flux of fine particles (Olsen et al., 1984). High depositional areas are characterized by a high organic content and relatively fine grain size compared to low depositional areas. Such high depositional areas are also characterized by high contaminant loadings compared to low depositional environments.

Table 3.8.7 Sensitivity Analysis of $(T)_{out}$ Based on Uncertainty in Generic Parameters in Lower Bay Complex Sediment Box.

Generic Parameter	T_{out}[a] (years)
Depth of Mixed Layer	
2 cm	10.7
5 cm	26.9
8 cm	43.0
Density of Mixed Layer	
$0.6\ g\ cm^{-3}$	23.1
$0.7\ g\ cm^{-3}$	26.9
$0.8\ g\ cm^{-3}$	30.7
Resuspension Rate[b]	
$0.25\ g\ cm^{-2}\ yr^{-1}$	9.2
$0.50\ g\ cm^{-2}\ yr^{-1}$	5.5
$1.00\ g\ cm^{-2}\ yr^{-1}$	3.1
Alternative Generic Parameters[c]	1.1

[a] Assuming microbial degradation rate (Z) equals zero

[b] Mean burial rate constant: $0.13\ g\ cm^{-2}\ yr^{-1}$

[c] Alternative generic parameters: mixed layer depth 2 cm; density $0.6\ g\ cm^{-3}$; resuspension rate $1.0\ g\ cm^{-2}\ yr^{-1}$

The variations in sediment contaminant concentrations express the uncertainty in calculating mean sediment inventories under such wide ranging sedimentary conditions. Contaminant concentrations on suspended sediment show the same degree of variability for most contaminants.

Tables 3.8.8 and 3.8.9 present the ranges of $(T)_{in}$ for Pb resulting from uncertainties in sediment and suspended sediment data. The residence time of Pb in sediment mixed layer can vary from 3.8-92 years as Pb concentration in sediment varies from $\bar{x} - 0.9s$ to $\bar{x} + 0.9s$, respectively. Furthermore, $(T)_{in}$ is shown to vary from 168-26 years corresponding to the standard deviation from the mean of reported Pb suspended sediment concentrations. Other contaminant data exhibit similar uncertainties which translate to a similar wide range in calculated residence time values.

3.8.4.2 Water Column

Residence times for contaminants in the water column are also subject to generic uncertainty. $(T)_{out}$ is calculated from the water column inventory and the combined removal terms, burial and advection. Since inventory, flux and burial terms are all related to particulate contaminant data, uncertainties in the contaminant-specific data cancel out. Contaminant burial rate is a function of contaminant levels on SPM and sediment burial rate calculated from radio-dated cores, whereas the advection term is a function of total water contaminant concentrations and advective flux of water out of the box. If advection were the only removal process, then the contaminant residence time would equal the residence time of water in the box (0.13 years). The uncertainty of the burial term is ± 35% (Olsen et al., 1984). For Pb this represents a range in $(T)_{out}$ from 0.039-0.061 years. The advective flux of water out of the box is estimated to be 534 m^3 sec^{-1} (Olsen et al., 1984). Based on a compilation of flow data (Duedall et al., 1979) the uncertainty in this estimation is ± 50%. Since advection is the main removal process for all but PAH and Pb, this uncertainty gives rise to a range in contaminant residence times. For Pb, the uncertainty in advection corresponds to contaminant residence times of 0.04 to 0.058 years.

3.8.5 Analysis of Model Results

Interpretation of model results must take into account the uncertainty of the generic parameters and contaminant-specific parameters. In the Lower Bay Complex, an extremely important removal term is the resuspension rate, a rate that is really not

Table 3.8.8 Effect of Uncertainty in Pb Sediment Data on $(T)_{in}$[a].

Uncertainty Expressed as Standard Deviation[b]	T_{in} (years)
$\bar{x} - 0.9s$	3.8
$\bar{x}$	45.0
$\bar{x} + 0.9s$	92.0

[a] Other contaminant-specific and generic parameters are the same as used in the calculation of $(T)_{in}$ for Pb on Table 3.8.4.

[b] Refer to Table 3.8.2. Standard deviation is greater than the mean. Therefore, 90 percent of the standard deviation is used for the purpose of calculation of x-s.

Table 3.8.9 Effect of Uncertainty in Pb Suspended Sediment Data on $(T)_{in}$[a].

Uncertainty Expressed as Standard Deviation[b]	T_{in} (years)
$\bar{x} - s$	168
$\bar{x}$	45
$\bar{x} + s$	26

[a] Other contaminant-specific and generic parameters are the same as used in the calculation of $(T)_{in}$ for Pb on Table 3.8.4.

[b] Pb or SPM $\bar{x} \pm s = 66.4 \pm 48.5$ μg Pb g^{-1}; Betzer (1978).

known with any great certainty beyond a factor of five. Since this term is also used in calculations of the gross deposition rate (resuspension rate plus burial rate = gross deposition rate) the uncertainty in the removal term is also transferred to the mass loading (T_{in}) calculations. Thus, there is considerable uncertainty in both $(T)_{in}$ and $(T)_{out}$ for all contaminants in the Lower Bay Complex (Section 3.8.4).

The uncertainty of the environmental data is discussed in Section 3.8.4. In addition to the reported error presented by each author, there is difficulty in reconciling data sets from different studies. In this regard, judgement must be used to identify the data to be used in the modeling study. For example, particulate metal data are that of Betzer (1978). Other metals data in the Lower Bay Complex water column include that of Mytelka et al. (1981) and Klinkhammer and Bender (1981). However, none of these studies presented a complete data set for all three metals and, for consistency in the modeling data, dissolved and total metals concentrations were calculated from partition coefficients instead of using other reported values. This approach limits the data set and essentially propagates any error in the original analysis. However, it does provide a common basis for interpretation. Furthermore, all water inventory and flux data were normalized to the concentration of SPM needed to balance the input and output terms of the SPM mass balance equation. This approach overestimates the actual particulate contaminant amounts on a per-liter basis for PCB and PAH by a factor of five and for metals by a factor of two. Since dissolved contaminant levels are calculated from particulate loads, they are also artificially elevated by this approach. However, such an approach was necessary for the model to present a coherent analysis of residence times.

The above limitations and uncertainty in the generic and contaminant-specific data make it imperative that the model results be analyzed for consistency. It is possible to test the model data by comparing total removal rates with mass loading rates. Assuming the system is in equilibrium, these two sets should be similar and total removal rates should not exceed mass loading rates. Total removal rates (advection plus burial) are presented in Table 3.8.10 along with mass loading data. In general, these two rates are in fairly good agreement with each other, with removal processes ranging from 41-105% of the mass loading rates. These are, in part, a result of our efforts to manage and choose data sets to achieve the balance between mass loading and removal processes. However, if the best estimates of loading and removal processes, and contaminant-specific data were not in reasonable agreement, such comparison would immediately identify the inconsistency.

Another method of testing the data is by comparing the burial rates to total removal rates. The percent of the total removed by burial should be similar for PCB,

Table 3.8.10. Percentage of Mass Loading Rate Removal from System by Burial and Advection in the Lower Bay Complex.

	Mass Loading[a] (kg yr^{-1})	Total Removal Rate[b] Burial, Advection (kg yr^{-1})	Percent of Mass Loading Removed by Burial and Advection	Percent of Total Removal Rate Due to Burial[c]
PCB	1,130	766	68	20
PAH	11,970	11,110	93	70
Pb	202,700	82,930	41	52
Cu	176,600	185,600	105	33
Cd	14,120	8,870	62	13

[a] Mass loading rate, Table 3.8.1

[b] Burial rates, Table 3.8.2; Advection rates, Table 3.8.3

[c] Burial rate/Burial rate + Advection

PAH and Pb, those contaminants with uniformly low solubilities. Relatively more Cu and Cd should be removed by advection as these contaminants are more water soluble. From Table 3.8.10 it can be noted that the amount of PCB removed by burial is low relative to Pb and PAH and that, in general, no trend can be determined by this analysis. A more complete data set including dissolved and particulate contaminant loadings would be needed to address this issue.

3.8.6 Sediment and Water Quality for Hypothetical Loadings

The concentrations of contaminants in the Lower Bay Complex sediment mixed layer have been calculated for several hypothetical loadings conditions (Table 3.8.11). For the purpose of this study it is assumed that the hypothetical changes in mass loading are the result of changes in strategy of domestic sewage discharge into the box, and that levels of contaminants on suspended matter are directly proportional to changes in mass loading. Although other sources of contaminants may change in response to changes in effluent discharge (e.g., marine flux) all other sources of contaminants are assumed to remain constant. It is assumed that the box has reached equilibrium under each loading and that residence times of contaminants do not change under hypothetical loading conditions. It can be seen that, for the hypothetical sewage effluent loading conditions, the contaminants all respond in a similar manner. Reducing effluent discharge into the Lower Bay Complex will have very little effect on sediment contaminant levels. Increasing discharge by a factor of 10X increases sediment contaminant levels 4-5 times present levels. Differences between contaminants reflect differing percentages of sewage effluent inputs in the total mass loadings (Table 3.8.1).

Responses of contaminants in the water column to changes in mass loading from sewage effluent are given in Table 3.8.12. Total contaminant levels are taken from Table 3.8.3 assuming mean particulate contaminant levels in the box of 13.4 mg SPM L^{-1}. Under these conditions PCB and Cu levels are presently above EPA water quality criteria. Reducing sewage effluent input into the Lower Bay Complex lowers PCB levels to below EPA thresholds. However, Cu levels still remain above the criteria. Doubling the effluent discharge into the bay increases the levels of all contaminants. However, only PCB and Cu remain above threshold criteria. Increasing mass loading from effluent to 10X present levels in addition brings Pb and Cd levels near marine water quality criteria. PAH level remains well below water quality standards.

Table 3.8.11 **Predicted Concentration of Contaminants in Lower Bay Complex Sediments (Mixed Layer) if Hypothetical Loadings are Changed by Factors of 0.1, 1, 2, and 10 Times Present Loading.**

	Lower Bay Complex 1981-1983 ($\mu g\ g^{-1}$ dry wt)	Loading Factor			
		0.1X	1X	2X	10X
		($\mu g\ g^{-1}$ dry wt)			
PCB	0.4	0.18	0.4	0.64	2.6
PAH	20	13.5	20	27.2	84.7
Pb	110	66.5	110	179	545
Cu	160	96.4	160	256	1053
Cd	3.0	1.9	3.0	4.9	14.3

Table 3.8.12 Predicted Concentration of Contaminants in Lower Bay Complex Water Column if Hypothetical Loadings are Changed by Factors of 0.1, 1, 2, and 10 Times Present Loading.

	EPA Marine Water Quality Criteria	Lower Bay Complex 1981-1983	Loading Factor			
			0.1X	1X	2X	10X
	(μg L^{-1})	(μg L^{-1})	(μg L^{-1})			
PCB	0.014	0.028	0.012	0.028	0.045	0.18
PAH	8	0.145	0.010	0.145	0.197	0.61
Pb	8.6	1.32	0.81	1.32	2.16	6.53
Cu	2.0	12.9	7.6	12.9	20.5	84.8
Cd	12	1.56	0.92	1.56	2.54	7.3

3.8.7 Summary and Research Needs

The residence times of contaminants in the sediment mixed layer of the Lower Bay Complex calculated as $(T)_{in}$ are in the range of 10-100 years, and 17-27 years calculated as $(T)_{out}$. The residence times of contaminants in the water column are 23 days based on mass loading and 29 days based on removal processes. The residence times in the water column box strongly reflect, although somewhat longer, the residence time of water in the Lower Bay Complex. The uncertainties of the data for both the generic and contaminant-specific parameters produce uncertainties in the calculation of residence times. The uncertainties in the data have a more significant effect on the sediment mixed layer residence times than on the water column residence times.

If research in the Lower Bay Complex were to be directed by this holistic modeling effort, then the most immediate needs would be filling gaps in the generic and contaminant-specific data sets. The greatest need with regard to generic parameters is obtaining reliable information on resuspension rates in the bay. Resuspension rate data used in the study is that of Butman and Moody (1983) and is an application of work performed on the Continental Shelf. Actual resuspension rates in the Lower Bay Complex are probably higher than those found on the shelf. Since this generic parameter effects both $(T)_{in}$ and $(T)_{out}$ calculations, it is probably the most important parameter that needs to be addressed.

Pb, Cu, and Cd concentration data have broad coverage in the Lower Bay Complex. However, these data are now 10 years old and similar surveys have not been conducted. Hg, As, and Ag sediment data are not available for the Lower Bay Complex. Sediment analysis for PCB and PAH is limited, and the available data are further limited by lack of standardization in reporting. For example, PAH measurements used in the modeling study are the sum of individual aromatic parent compounds and alkylated homologs (Boehm, 1983a; 1983b), whereas another investigator simply reports parent compounds (MacLeod et al., 1981). The ability to fill critical data gaps as highlighted by this modeling effort would be greatly enhanced if future contaminant monitoring programs were coordinated such that all contaminant and auxiliary analyses such as grain size and TOC were completed on the same sample set.

Generic parameters in the water column box are all reasonably well known. Riverine, estuarine, and effluent flows have been routinely monitored for years. However, contaminant-specific water column data in the Lower Bay Complex water column are relatively scarce. Again, a coordinated monitoring program to measure all contaminants of interest would place more confidence in the results of the model, and would increase the confidence that the scientific and regulatory community might have in the results of such "what if" exercises as those presented in Section 3.8.6.

3.9 Southern California Palos Verdes Shelf: Outfalls at Whites Point

The Palos Verdes Shelf in Southern California has been a waste water disposal site for the last 45 years. The two sewage outfalls of the Joint Water Pollution Control Plant (JWPCP) for the Los Angeles County Sanitation Districts are located offshore of Whites Point on the Palos Verdes Peninsula. These outfalls discharge approximately 1.4×10^9 L day^{-1} of advanced primary treated waste water generated by 3.5 million people and 70,000 industries or commercial establishments.

The Palos Verdes Shelf was selected as a site for the investigation of contaminant residence times and recovery rates because it is a relatively high energy environment on the Continental Shelf, and should have the potential for both rapid recovery and short contaminant residence times. Due to monitoring and research conducted by the Southern California Coastal Water Research Project (SCCWRP) and the County Sanitation Districts of Los Angeles County, an extensive data base is available for this area.

The results of previous monitoring and sedimentation modeling show that contaminants are transported mainly parallel to the shoreline and that most of the deposition of solids occurs upcoast (northwest) of the outfalls. Approximately 14% of the particles in the waste water settle within an area of 21 km^2, along a 16-km long section of the coast (Hendricks, 1982; Hendricks, 1983). Natural sedimentation rates on the shelf are low, on the order of 0.01-0.1 g cm^{-2} yr^{-1}, while resuspension rates are relatively high, 1-20 g cm^{-2} yr^{-1}. Near the outfalls the maximum sedimentation rate is predicted to be about 0.38 g cm^{-2} yr^{-1} (Hendricks, 1982). Profiles of contaminants in age-dated sediment cores indicate the sedimentation rates are relatively high near the outfalls but low 16 km upcoast, or 2 km inshore or offshore from the 60-m isobath (Stull et al., 1984). Stull et al. (1984) have also shown that there is a remarkable correlation between the improvement in both sewage effluent quality and sediment quality during the last decade. This indicates that the recovery rate, or response of the sediment quality to change in loading, is on the order of several years at this site.

3.9.1 Structure of the Box Model

The box model for the Palos Verdes Shelf consists of a rectangular water column box 1.3 km wide, 16 km long and 60 m deep (Figure 3.4). The long axis of the box is parallel to the coastline and centered on the 60-m isobath. The area of the box is 21 km^{2} and the volume of the water column in the box is 1.26 x 10^{9} m^{3}. The sediment box area is also 21 km^{2} and the thickness of the bioturbated mixed layer is 1.0 g cm^{-2}, which ranges from 1-5 cm thick (Hendricks, 1983). The mass of sediment in the mixed layer is 2.1 x 10^{8} kg^{3}.

Contaminants modeled for the Palos Verdes Shelf sediments and water column were PCB and DDT as well as Pb, Hg, Ag, Cu and Cd.

3.9.1.1 Sediment Box Model

The sediment mass loading equation (4) is not used to calculate (T_{in}) because 1) field data for contaminant concentrations on suspended matter are not available, 2) all ocean dumping is discharged to the water column rather than to the sediments, and 3) production of contaminants within the sediment is assumed to be zero.

The removal rate was calculated from equation (5) with modifications for the Palos Verdes Shelf:

$$\left(\frac{dA_x}{dt}\right)_{out} = C_{x(sed)}\ Q_b\ A_s + C_{x(sed)}\ Q_{rs}\ A_s$$

The pore water flux term is not used since pore water data are not available for the Palos Verdes area. This term is probably only minor based on the relative contribution of the term for other study areas such as Puget Sound or the Lower Bay Complex. Since the microbial degradation rate for PCB and DDT is very low ($t_{1/2}$ = < 2 years), it is assumed to be zero.

3.9.1.2 Water Column Box Model

The mass loading equation (6) was used to calculate residence time for the Palos Verdes Shelf water column. With site-specific modifications, the mass loading equation used to calculate $(T)_{in}$ is:

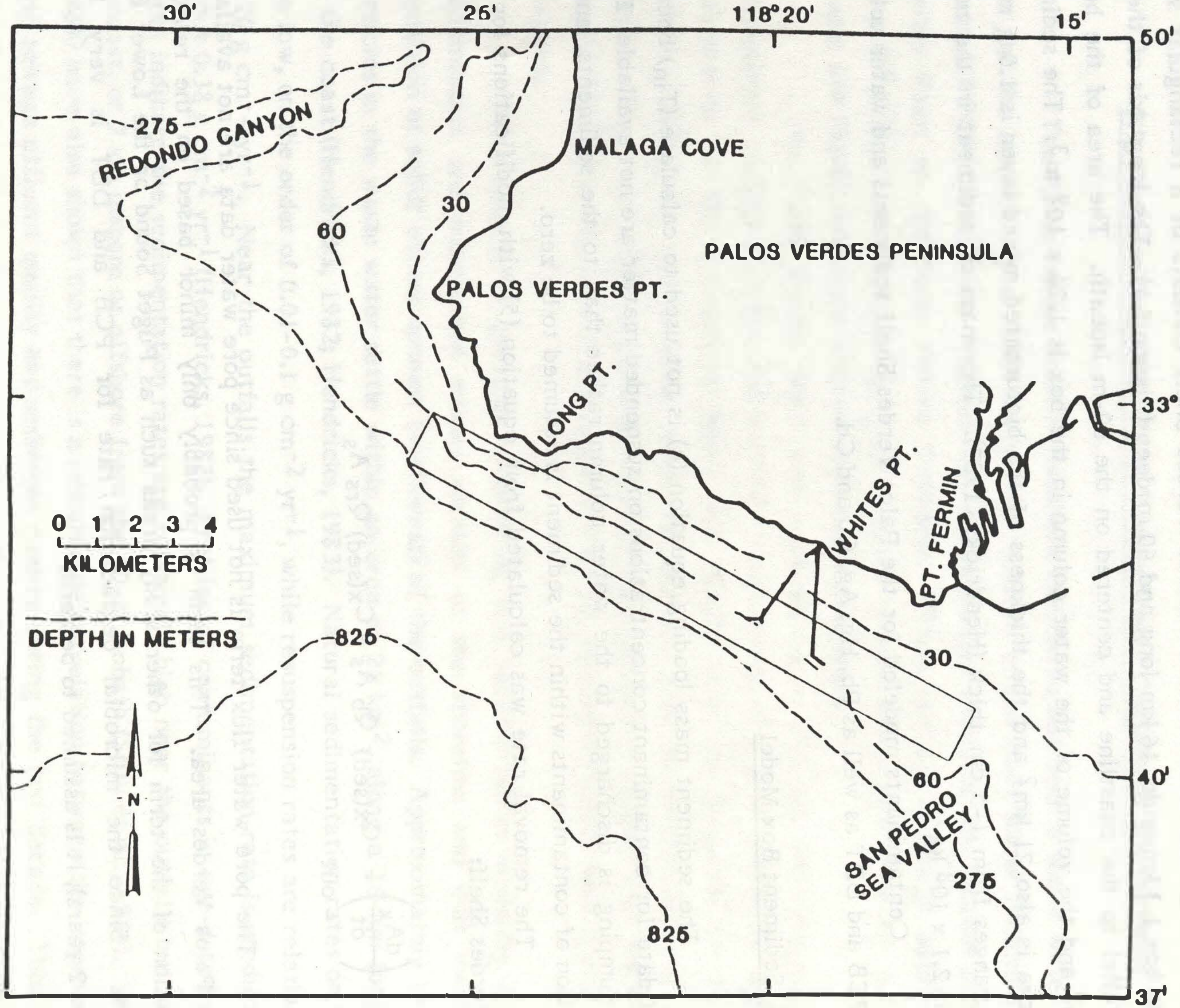

Figure 3.4 Palos Verdes Shelf Off Southern California. Area in Rectangle Used for the Box Model Calculations.

$$\left(\frac{dA_x}{dt}\right)_{in} = \left[C_{x(dis)} + C_{x(part)}\right]_{in} Q_w + Q_{x(dep)}\, A_s + C_{x(sed)}\, Q_{rs}\, A_s + C_{x(od)}\, Q_{od}$$

Because data are only available for total contaminant loadings, the particulate and dissolved terms have been combined. Since there are no pore water chemistry data, the pore water flux is assumed to be zero. Pore water flux of contaminants into Puget Sound (Section 3.6) is relatively insignificant and is, therefore, also expected to be insignificant for the Palos Verdes Shelf model.

Water column removal processes basically include advection and sedimentation. Equation (7) modified for the Palos Verdes Shelf model is:

$$\left(\frac{dA_x}{dt}\right)_{out} = \left[C_{x(dis)} + C_{x(part)}\right]_{out} \left[Q_w + C_{x(tsm)}\right]_{out} Q_{sed}\, A_s$$

The particulate and dissolved fractions are combined since only total concentration is available from the data. The degradation rate (Z) is assumed to be zero due to the short residence time of the water column in the compartments.

3.9.2 Generic Parameters

3.9.2.1 Advective Transport

The predominant current direction in the area of the box is long shore, upcoast from the southeast to the northwest. Median speeds reported by Hendricks (1983; 1980) ranged from 2-40 cm sec^{-1}. During the spring of 1981 the median speeds at the outfall were 10.2, 6.2, and 7.4 cm sec^{-1} for near-surface, mid-depth and near-bottom currents, respectively. The net velocity was approximately 1.0 cm sec^{-1}. Net motions at mid-depth during seven months of previous observations were 3.6 cm sec^{-1} (upcoast) and 5.5 cm sec^{-1} (Hendricks, 1983; 1980). Currents, however, occasionally transport sewage effluent downcoast, offshore and onshore. The contour lines of contaminants in the Palos Verdes shelf sediments show that the major deposition of solids occurs upcoast of the outfalls along the 60-m isobath.

For the purpose of calculating the residence time of contaminants in the water column, a net velocity of 5 cm sec^{-1} upcoast is assumed. This velocity will transport a parcel of water 4 km day^{-1}, thus traversing the deposition area in four days. Because the residence time of the water column is on the order of four days, the residence time of dissolved contaminants and particulate matter in the water is also not more than four days.

3.9.2.2 Transport of Suspended Matter

The advective transport of natural suspended matter into the box is calculated to be 1.22 x 10^8 kg yr^{-1}, assuming a 5 cm sec^{-1} current and a suspended particulate load of 1.0 mg L^{-1} (dry weight). The other source of suspended matter to the box is from the outfalls which discharge approximately 0.9 x 10^8 kg yr^{-1}. The suspended matter is removed from the box by advection and sedimentation. Sedimentation in the box is estimated by Hendricks (1983) to range from 0.01-0.35 g cm^{-2} yr^{-1} above the natural sedimentation of 0.02 g cm^{-2} yr^{-1}. Only approximately 14% of the volatile solids discharged by the outfalls are deposited in the box (Hendricks, 1983). We have assumed the average sedimentation rate in the box is 0.08 g cm^{-2} yr^{-1} which is the sum of 0.02 g cm^{-2} yr^{-1} of natural and 0.06 g cm^{-2} yr^{-1} due to sewage particles. The net sedimentation in the box is 0.168 x 10^8 kg yr^{-1}. To maintain a sediment mass balance, advection must transport 1.95 x 10^8 kg yr^{-1} out of the box.

Resuspension of bottom sediments is an important process on the Palos Verdes shelf. Hendricks (1983) has estimated that resuspension rates range from 1-20 g cm^{-2} yr^{-1}. We have used 10 g cm^{-2} yr^{-1} as the resuspension rate for our modeling. This means that the surface layer of sediment, probably only a few millimeters thick, is resuspended and deposited many times each year before the sediment is permanently buried below the mixed layer or advected out of the box. Because of the high resuspension rate, the composition of the sediment in the mixed layer is frequently mixing with the suspended matter that is advecting through the box. Consequently, the chemical composition of the surface sediment is identical to that of the suspended particles.

In calculating the chemical balance and residence times for the box model, we assume the inputs of contaminants are from the JWPCP outfalls, atmospheric fallout, rivers, and advection of "clean" coastal seawater which is transporting low concentrations of dissolved contaminants and 1 mg L^{-1} suspended "baseline" sediment. In the box contaminants and suspended solids are uniformly distributed, and the sedimentation rate is 0.08 g cm^{-2} throughout the box. Approximately 90% of the contaminants and solids that enter the box are quickly removed by resuspension and advection. However, because the outfalls are such a large source of contaminants, the water and sediments within the box are contaminated.

3.9.2.3 Mass Loading of Contaminants

The mass loading of contaminants entering the box from different sources is shown in Table 3.9.1. The 1980 annual mass emissions for the JWPCP outfalls is from Schafer (1982). The atmospheric fallout rates are from Young and Jan (1976), and Young and Heesen (1976). The fallout rates, measured during non-brush fire periods, were assumed to be uniform over the 21-km^2 area of the box. Surface runoff or rivers contribute a relatively small proportion of contaminants to the box. Young et al. (1980) estimated contaminant input to the Southern California Bight via surface runoff. We assumed that approximately 1% of the contaminants from surface runoff enter the box. Because 98% or more of the inputs of contaminants to the box are from the JWPCP, uncertainties in the runoff and fallout rates will not have significant effects on the total mass loadings.

3.9.2.4 Sediment Chemistry

The concentrations of contaminants in "baseline" shelf sediments, Palos Verdes shelf sediments, the sediment inventory and annual burial rate of contaminants are shown in Table 3.9.2. Baseline is defined as sediments not contaminated by anthropogenic sources. The "baseline" sediment metals data are from Katz and Kaplan (1981). The baseline PCB and DDT concentrations are taken from the Puget Sound model (Section 3.6).

The concentrations of contaminants in the Palos Verdes shelf sediment box were taken from Stull et al. (1984) and SCCWRP reports. Because the sediment chemistry changed by more than a factor of 10 over a distance of several kilometers, it is difficult to calculate the average concentration for the surface sediment in the box. Also, the sediment chemistry has been decreasing during the last decade, paralleling the decrease in JWPCP emissions. The sediment chemistry is our best estimate for the average of surface sediment in the year 1980.

The mixed layer inventory in Table 3.9.2 is the mass of contaminants in the top 1 g cm^{-2} of sediment through the 21-km^2 box, assuming the sediment chemistry in the previous column. The annual burial rate of contaminants assumes 0.08 g cm^{-2} yr^{-1} of sediment is removed from the sediment mixed layer.

The right column in Table 3.9.2 lists the percentages of total inputs of contaminants (from Table 3.9.1) that are buried annually. These percentages range from 3-26% and average 12%. This means that, on the average, 12% of the contaminants

Table 3.9.1. Mass Loading of Contaminants to Palos Verdes Shelf Box.

Contaminant	Sewage (JWPCP, 1980) (kg yr^{-1})	Fallout (kg yr^{-1})	River Runoff (kg yr^{-1})	Advected Baseline Suspended Sediment (kg yr^{-1})	Total Inputs (kg yr^{-1})
PCB	336	0.004	1	0.002	337
DDT	542	0.7	1	0.002	543
Pb	62,000	882	510	21	63,400
Hg	410	n.d.	3	0.105	413
Ag	5,170	0.17	3	0.84	5,173
Cu	98,000	67	300	18.9	98,400
Cd	10,300	3.2	14	0.84	10,300

n.d. - no data

Table 3.9.2. Concentrations of Contaminants in Baseline Shelf Sediments, Palos Verdes Sediments, Mass of Contaminants in Sediment Mixed Layer and Net Burial of Pollutants in Sediments.

Contaminant	Baseline Shelf Sediment (μg g^{-1})	Palos Verdes Shelf Sediment Box (μg g^{-1})	Mixed-Layer Inventory (kg)	Annual Burial (kg yr^{-1})	% of Total Inputs Buried
PCB	0.001	4	840	67	20
DDT	0.001	5	1,050	84	15
Pb	10	250	52,500	4,200	6
Hg	0.05	3	630	50	12
Ag	0.4	10	2,100	168	3
Cu	9	400	84,000	6,720	7
Cd	0.4	20	4,200	336	3

discharged to this box are permanently buried. The remaining 88% are advected out of the box and probably deposited in deeper water offshore and upcoast of Palos Verdes. There is good agreement between our estimate (12%) and Hendricks' estimate (14%) for the amount of volatile solids that would be deposited within a 21-km^2 area (Hendricks, 1983). Outside the 21-km^2 area, the sedimentation rate is near the baseline concentrations of 0.02 g cm^{-2} yr^{-1} and the sediment chemistry is approaching baseline, thus making it difficult to estimate deposition rates for contaminants.

The concentrations of pollutants in suspended sediment in the box were calculated assuming the total inputs (Table 3.9.1) are particulate and mixed uniformly with suspended solids in the water column of the box. These predicted concentrations are similar to those measured in surface sediment (Table 3.9.3).

3.9.3 Residence Times in the Palos Verdes Shelf

3.9.3.1 Residence Times in Surficial Sediments

The residence times of contaminants in the mixed layer of sediment have been calculated using four different sedimentation rates (Table 3.9.4). The residence time in three of the columns is for a net sedimentation rate or burial rate of 0.08 g cm^{-2} yr^{-1} which is the average sedimentation rate in the box. Resuspension rates used were 10, 1 and 0 g cm^{-2} yr^{-1}, and the average (T) values are 0.1, 1.5 and 18 years. The (T) for baseline sedimentation rate with no resuspension averaged 72 years. These results demonstrate that resuspension is the dominant factor in controlling the residence time in an area with low sedimentation rates.

The actual residence time for contaminants in the mixed layer of Palos Verdes shelf sediments is probably not longer than a few years. This is based on field data which demonstrate a good correlation between mass emission rates from the outfalls and the surface sediment chemistry (Stull et al., 1984). In order to account for (T) of a few years the resuspension rate must be on the order of a 1 g cm^{-2} yr^{-1}. Hendricks (1983) has modeled the physical processes near the outfalls and estimates that resuspension is much greater than 1 g cm^{-2} yr^{-1}. Bioresuspension may be of significance in deeper water or in sediments with lower volatile solids than for sediments near the outfalls. Bioresuspension rates for the shelf range from 0.3-2.0 g cm^{-2} yr^{-1} (Hendricks, 1984).

Residence times on the order of 0.1 year may be possible if physical energy is high but the surface sediments can only change composition as new suspended sediment is supplied. Using a mixed layer thickness of 1 g cm^{-2}, the sediment inventory for 21 km^2 is

Table 3.9.3. Concentrations of Contaminants in Suspended Sediment Assuming Total Inputs of Contaminants and Suspended Solids are Particulate, and Uniformly Dispersed Throughout Water Column of Box.

Contaminant	Predicted Concentrations in Suspended Sediment ($\mu g\ g^{-1}$)	Measured Concentrations in Surface Sediment ($\mu g\ g^{-1}$)
PCB	1.6	4
DDT	2.6	5
Pb	305	250
Hg	2.0	3
Ag	25	10
Cu	469	400
Cd	49	20

Table 3.9.4. **Residence Times for Contaminants in the Sediment Mixed Layer of the Palos Verdes Shelf for Different Sedimentation Rates.**

	Total deposition 10.08 g cm^{-2} yr^{-1} Resuspension 10.0 g $cm^{-2}yr^{-1}$ (years)	Total deposition 1.08 g cm^{-2} yr^{-1} Resuspension 1.0 g $cm^{-2}yr^{-1}$ (years)	Net deposition 0.08 g cm^{-2} yr^{-1} No Resuspension (years)	Baseline Net deposition 0.02 g cm^{-2} yr^{-1} No Resuspension (years)
PCB	0.25	2.5	31	120
DDT	0.19	1.9	24	100
Pb	0.08	0.8	10	40
Hg	0.15	1.5	19	30
Ag	0.04	0.4	5	20
Cu	0.08	0.8	10	40
Cd	0.04	0.4	5	20
Average	0.1	1.5	18	72

2.1×10^8 kg. Advection of natural particles will supply 1.22×10^8 kg yr^{-1}. In approximately two years all the surface sediment in the box can be replaced with new suspended particles advected through the box if resuspension is greater than 0.5 g cm^{-2} yr^{-1}.

3.9.3.2 Residence Times in the Water Column

The concentrations and inventories of contaminants in the water column of the Palos Verdes shelf have been predicted for the box model (Table 3.9.5). Since we were not able to obtain water column chemistry data from the shelf area we have used the same baseline water column concentrations used for the Puget Sound model. The baseline inventory (Table 3.9.6) is small compared to the inventory in the water column when the outfalls are operating. The predicted seawater concentrations of contaminants were calculated assuming all contaminants are mixed throughout the box and the (T) for the water column is four days. Because the actual current velocities near the outfall are highly variable, the actual concentrations will range from much higher than predicted during stagnant conditions to much lower when currents are strong.

3.9.4 Sensitivity Analysis

Our best estimates for the residence time of contaminants in the shelf sediments are several years. This is based on resuspension rates of 1 g cm^{-2} yr^{-1}, mixed layer thickness of 1 g cm^{-2}, and available chemical data for sediments and suspended matter. The chemical data for the sediments and mass loading are well known, probably within an uncertainty of $\pm$50%. The thickness of the mixed layer is not well known. However, it is not likely to be greater than 5 g cm^{-2} based on the profiles of contaminants in sediment cores which have subsurface peaks (Stull et al., 1984). Resuspension rates are also not well known and can range from 0.3-10 g cm^{-2} yr^{-1}. If the mixed layer were 5 g cm^{-2} (10-20 cm thick), then the inventory would be five times greater than the (T) in Table 3.9.4. If the mixed layer was 5 g cm^{-2} and resuspension rate was 1 g cm^{-2} yr^{-1}, then (T) would be 5 years which is probably the maximum that is reasonable considering the field data for the last decade. A reasonable range for mixed layer thickness is 1-3 g cm^{-2} and a resuspension rate of 0.3-5 g cm^{-2} yr^{-1}. These values will result in a (T) of approximately 1-10 years.

Table 3.9.5. Concentrations and Inventory of Contaminants in the Water Column of the Palos Verdes Shelf.

	Baseline Water Column Chemistry (μg L^{-1})	Baseline Water Column Inventory (kg)	Predicted Water Column Inventory for JWPCP 1980 Discharge (kg)	Predicted Seawater Concentrations for 1980 Discharge (μg L^{-1})
PCB	0.00001	0.012	3.6	0.003
DDT	0.00001	0.012	6.0	0.004
Pb	0.010	12.6	690	0.5
Hg	0.001	0.126	4.5	0.004
Ag	0.002	2.52	56	0.04
Cu	0.1	126	1200	0.9
Cd	0.05	63	175	0.13

Table 3.9.6. Palos Verdes Shelf Sediment Chemistry for Hypothetical Loadings.

	ug g^{-1} dry weight					
	1980 Field Samples	Baseline	Loading Factor			
			0X	1X	2X	10X
PCB	4	0.001	0.009	1.6	2.2	3.3
DDT	5	0.001	0.009	2.6	3.6	5.3
Pb	250	10	22	305	419	613
Hg	3	0.05	0.08	2.0	2.7	4.1
Ag	10	0.4	0.43	25	34	51
Cu	400	9	12	469	654	970
Cd	20	0.4	0.55	49	68	102

3.9.5 Analysis of Model Results

3.9.5.1 Sediment Mixed Layer

The mixed layer of sediment would assume a new equilibrium contaminant concentration within a few years if the discharges from the Whites Point outfalls stopped. The sediment would approach the chemical composition of the natural suspended matter transported over the shelf, assuming the resuspension rate remains at approximately 1 g cm^{-2} yr^{-1} and the mixed layer remains 1 g cm^{-2}. The supply of natural suspended solids is approximately equal to one-half the mass of the mixed layer. Therefore, the sediment particles in the mixed layer will be resuspended and exchanged with the natural suspended solids every two years.

Hendricks (1980, 1982, 1983, 1984) has shown that resuspension of bottom sediments is related to volatile solids (VS) concentrations. Sediments with high VS resuspend at a much lower current speed than sediments with low VS. As sediments recover, following a reduction in sewage discharge rates, there may be a point at which the physical resuspension rate decreases dramatically due to changes in sediment texture (T. Hendricks, personal communication). When this point in sediment recovery is reached, which may be in the range of 5-10% VS, the resuspension rate will decrease resulting in little exchange of sediment in the mixed layer with suspended solids. Since the natural sedimentation rate on the shelf is on the order of 0.02 g cm^{-2} yr^{-1}, it will take approximately 50 years for the chemistry of the mixed layer to approach that of suspended solids, assuming no resuspension. However, if the rate of bioresuspension is approximately 0.2 g cm^{-2} yr^{-1} then the mixed layer will approach equilibrium with the suspended solids in about 5 years (Hendricks, 1984).

3.9.5.2 Water Column

The movement of water through the box results in the replacement of the water column in four days. Since approximately 10% of the contaminants entering the box are retained in the sediment, the remaining 90% are advected out of the box and have a residence time of about four days in the water column. If the mass loading is changed, the water column should approach equilibrium in about four days. Because of resuspension, there will be a source of contaminants to the water column from resuspension of contaminated sediments. This source is significant and will have an impact on water quality for several years after the outfalls stop discharging.

3.9.6 Sediment and Water Quality for Hypothetical Loadings

3.9.6.1 Sediment Mixed Layer

The concentrations of contaminants in the mixed layer of sediment for four hypothetical loading rates are shown in Table 3.9.6. The first column of concentrations is for field samples collected in 1980. The baseline concentrations are predicted assuming no anthropogenic loading to the California Bight. The 0X loading is for zero loading from the outfalls, but fallout and rivers retain the same loading rates as 1X, which is the mass loading for 1980 (Table 3.9.1). For 2X and 10X the outfall loading is increased by two and ten times the 1980 rate while the other inputs are held constant.

The concentrations of contaminants in sediment for 0X loading factor are higher than for baseline because of the input from fallout and rivers. The predicted sediment chemistry for 1X agrees within a factor of two with the 1980 field samples for PCB, DDT, Pb, Hg, and Cu, but is 2.5 times higher for Ag and Cd. These discrepancies between predicted and measured can be due to many factors. The most likely factors are over-estimating the percentage of a contaminant that is buried and over-estimating the average sediment chemistry from the 1980 field samples.

3.9.6.2 Water Column

The water chemistry for five different hypothetical loading rates is shown in Table 3.9.7. The loading factor 0X is for no anthropogenic inputs and, therefore, the contaminant concentrations are baseline. The loading factors 1X, 2X, and 10X are for 1980 outfall loadings, two times and ten times, respectively, the 1980 loadings. The column labeled "Recovery" shows contaminant concentrations in the water column assuming the outfalls stopped discharging and with surface sediments the only source during a recovery period of three years. During the sediment recovery period, the suspension is contributing contraminants to the water column. If the recovery period were less than three years the water concentrations would be higher due to a greater resuspension rate of contaminants. During recovery the concentrations are intermediate between baseline (OX) and those predicted for 1980 loading (1X).

The predicated concentrations of contaminants in the water column pore water and surface sediments for 10 times the 1980 loading are compared to water and sediment quality criteria in Table 3.9.8. The source of these data is described in Section 3.6.7, except for the Kp of DDT which is from Tobin (1984).

Table 3.9.7. Palos Verdes Shelf Water Chemistry for Hypothetical Loadings.

	Loading Factor ($\mu g\ L^{-1}$)				
	0X	1X	2X	10X	Recovery
PCB	0.00001[a]	0.003	0.005	0.027	0.0011
DDT	0.00001[a]	0.004	0.008	0.04	0.0016
Pb	0.010[a]	0.5	1.0	5.1	0.2
Hg	0.001[a]	0.004	0.008	0.03	0.0013
Ag	0.002[a]	0.04	0.09	0.4	0.017
Cu	0.1[a]	0.9	1.7	8.1	0.37
Cd	0.05[a]	0.13	0.22	0.9	0.08

[a] Typical coastal baseline seawater concentrations.

Table 3.9.8. Comparison of EPA Marine Water and Sediment Quality Criteria with Predicted Concentrations of Contaminants in Water Column, Sediment Pore Water and Surface Sediment for Hypothetical Loading 10 Times 1980 Loading to Palos Verdes Shelf.

	Water Column		Sediment Pore Water		Surface Sediment	
	EPA Criteria Seawater $\mu g\ L^{-1}$	10X Loading Palos Verdes $\mu g\ L^{-1}$	Kp Partitioning Coefficient	10X Loading Palos Verdes $\mu g\ L^{-1}$	U.S. EPA Region 10 Four-mile Rock Interim Criteria $\mu g\ g^{-1}$	10X Loading Palos Verdes $\mu g\ g^{-1}$
PCB	0.014	0.027	$5x10^3$	0.66	0.61	3.3
DDT	0.001	0.04	$8x10^3$	0.66	0.007	5.3
Pb	8.6	5.1	$>2.7x10^5$	2.3	126	613
Hg	0.1	0.03	$4x10^4$	0.10	1.1	4.1
Ag	2.3	0.4	$1.6x10^5$	0.32	n.d.	51
Cu	2.0	8.1	$6.8x10^3$	143	92	970
Cd	12	0.9	$6.2x10^3$	16	0.7	102

[a] Tobin (1984) assumed sediment contained 5% organic carbon, PCB-4 = $1x10^5$ and DDT Koc = $1.6x10^5$.

n.d. = no data

The water column data are predicted to exceed the criteria for PCB, DDT and Cu. The pore water will exceed water column criteria for PCB, DDT, Cu and Cd. The sediments will exceed all the Four-Mile Rock criteria. However, the Palos Verdes sediments presently exceed the Four-Mile Rock criteria for all these contaminants. The predicted water column data for 1980 mass loading indicate only DDT exceeds the water criteria.

3.9.7 Summary and Research Needs

The residence times for contaminants in the mixed layer of the Palos Verdes shelf sediments are in the range of 1-3 years. The residence times for contaminants in the water column are several days. If the mass emission of contaminants to the shelf is greatly reduced, recovery of the water column will occur in several days, while the surface sediments will require several years to recover. Resuspension of surface sediments is the process that removes contaminants from the sediment mixed layer. Because of the uncertainty in the sediment resuspension rates for sediments with both various degrees of contamination and benthic faunal communities, it is difficult to accurately estimate residence times and recovery rates. The concentrations of contaminants in sediment cores indicate the sediment quality has improved as the mass emission rate decreased during the last decade (Stull et al., 1984). This rate of recovery may not continue if the rate of resuspension decreased dramatically as the sedimentation rate and volatile solids content decrease in the sediments near the outfalls. Although considerable progress has been made in modeling the transport and fate of contaminants discharged at Whites Point, continued research on modeling and field verification studies will lead to a better understanding of the fate of contaminants.

3.10 PHILADELPHIA SEWAGE SLUDGE DISPOSAL SITE

The Philadelphia Sewage Sludge Disposal Site was active for the period June, 1973 to November, 1980. During this time the site was used by Camden, New Jersey and Philadelphia for the disposal sewage sludge. The City of Camden dumped at the site from June, 1983 to November, 1976; whereas Philadelphia dumped from June, 1973 to November, 1980. For a short period of time in 1974 and 1975, the Modern Transportation Company dumped sewage collected from several northern New Jersey communities, but this was quickly terminated as other means of disposal became available. From September, 1968 to July, 1969 the site was also used by DuPont-Edgemoor for disposal of acid-iron waste (Muir, 1983). In total, approximately 4.11 x 10^6 metric tons of waste were dumped at the site (Muir, 1983).

The site (Figures 3.5 and 3.6) is located approximately 71 km southeast of the mouth of Delaware Bay between $38^{o}22.5'$ and $38^{o}20'$N, and $74^{o}10'$ and $74^{o}20'$W. Water depths at the site range between 40-55 m with a mean depth of 51 m (EPA, 1981). The bottom topography is primarily a ridge (topographic high) and swale (topographic low) system similar to features which exist on 75% of the Continental Shelf (Knebel and Folger, 1976). Ridge-to-ridge distances in the vicinity of the site range from 2-6 km (Swift et al., 1971). The alignment of the ridges at the site is mostly east to northeast at $20\text{-}30^{o}$ angles to the shoreline (Freeland and Swift, 1978).

The significance of the ridge and swale topography at the site lies in the fact that these features are the result of recent and continuous processes of current and wave surge transport (Swift et al., 1971). Data suggest that significant transport occurs only during storms capable of generating currents and waves of sufficient energy and depth penetration, rather than by the prevailing winds or currents (Butman and Noble, 1979). At this area of the shelf Butman and Moody, (1983) expect that there will be on the average 5-10 storms per year capable of resuspending bottom material and that this resuspended material will be transported 10 km cross-shelf and 100 km long-shelf during a winter season. Conservative estimates of turbidity data during these storm events indicate that SPM levels during storm events will rise by 20 mg L^{-1} in the bottom 10 m of water. Assuming 5-10 storms per year, this corresponds to 0.1-0.2 g cm^{-2} yr^{-1} of resuspended material at the Philadelphia Site. For the purposes of this study, we have used a value of 0.2 g cm^{-2} yr^{-1} and addressed the range of resuspension in our sensitivity analysis of residence time results.

The existence of the ridge-swale system on the Mid-Atlantic Shelf also implies that no net sedimentation or erosion has occurred in recent history (Swift et al., 1971). Thus, for the purposes of modeling this system, sedimentation is set equal to resuspension.

Previous monitoring at the site during sludge dumping operations has indicated that significant alterations to the water column chemical parameters are limited to increases in suspended solids (SPM) and phytoplankton nutrients. Release of dissolved trace metals could not be detected over background variations (EPA, 1981).

The geochemical effects of dumping on the sediments at the site and surrounding area have been studied extensively for the period of 1973-1979, the years of heaviest site usage. These studies revealed no temporal trends in TOC elevation or variations in sediment trace metal levels of Pb, Cu, Ni or Cr. Cd levels were generally below detection limits. Levels of all metals were within the range of those found elsewhere in the Mid-Atlantic Continental Shelf (EPA, 1981).

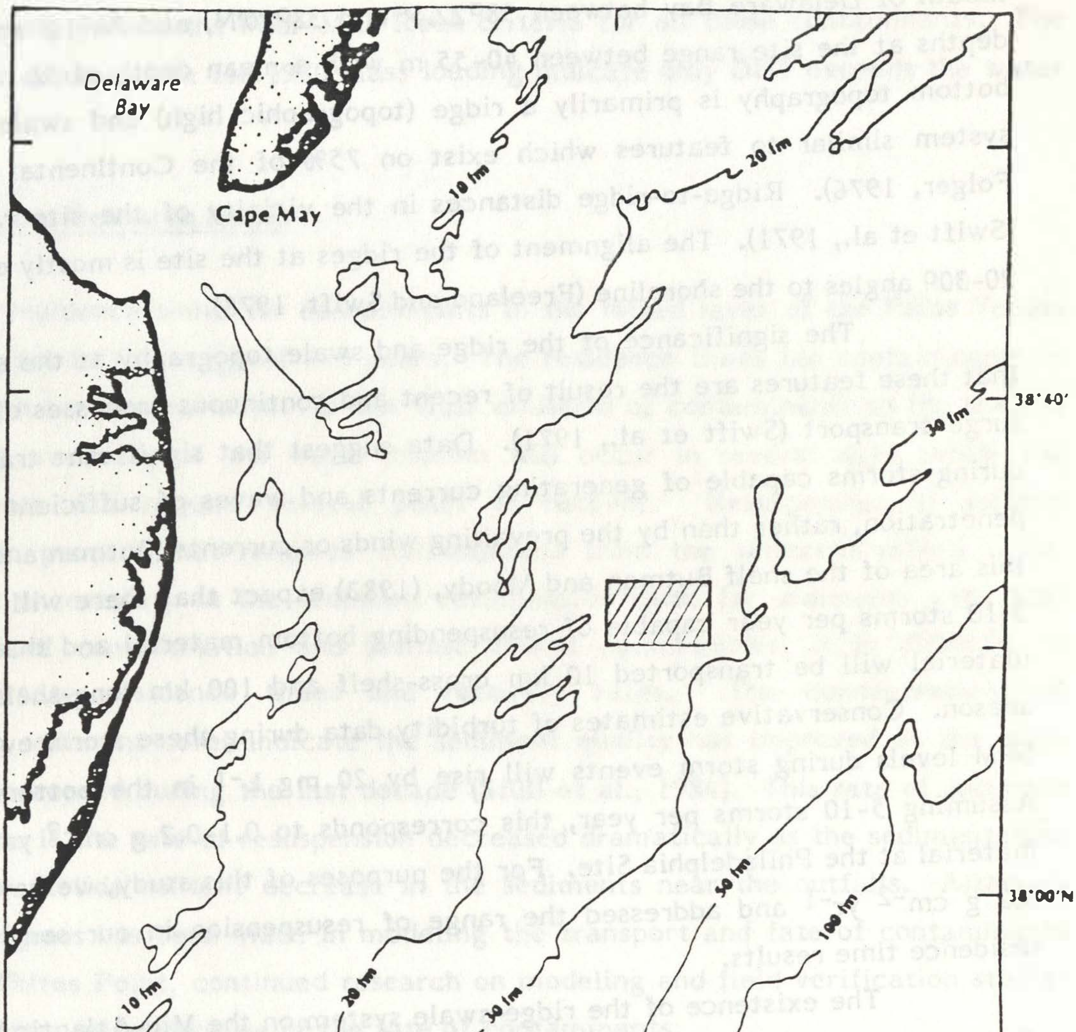

Figure 3.5 Philadelphia Sewage Sludge Disposal Site.

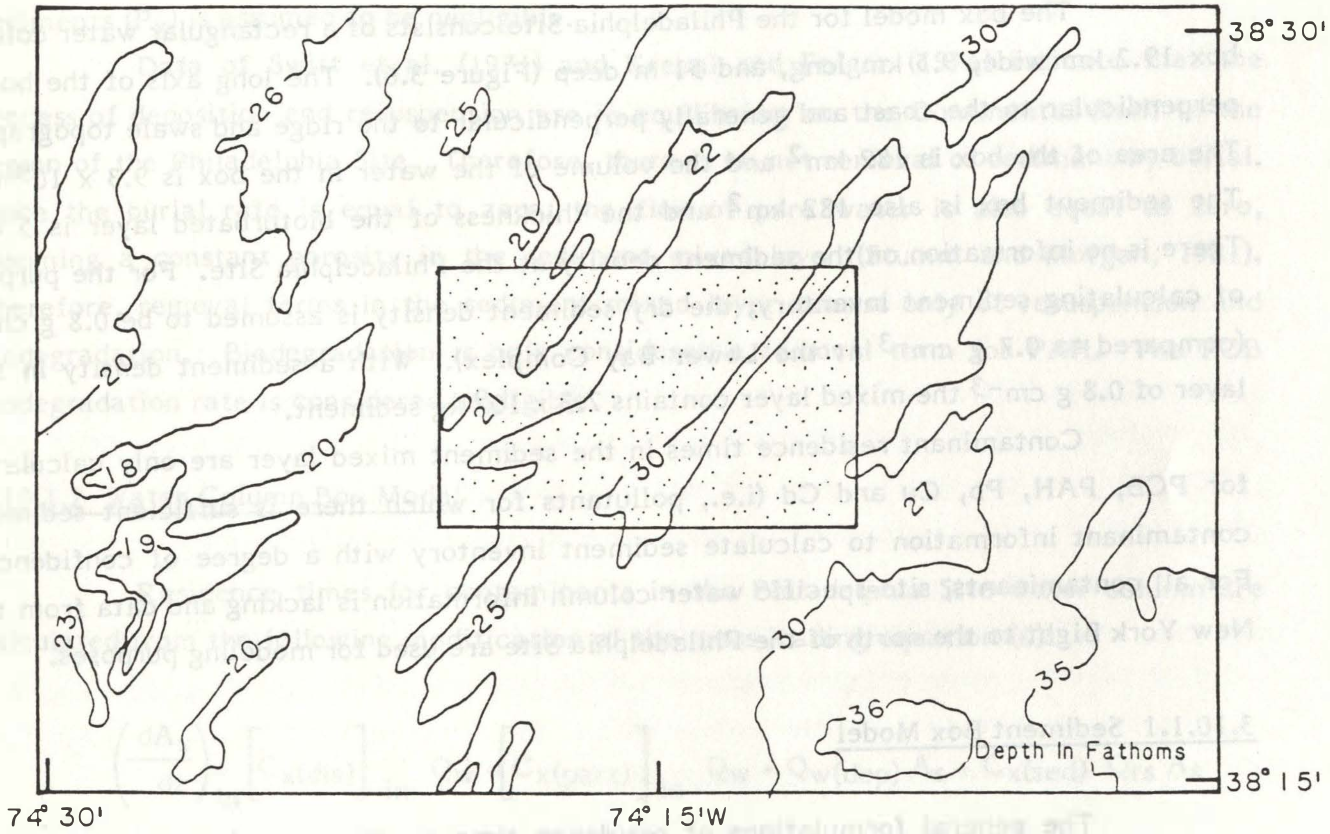

Figure 3.6 Bathymetric Detail of the Philadelphia Sewage Sludge Disposal Site.

The results of previous monitoring have suggested an impact of dumping upon sediments downstream (south) of the site, (Lear et al., 1979; 1981) in association with relatively sheltered swale sediments. However, other interpretations of the data indicate a normal co-variation between geochemical parameters and sediment chemical constituents (EPA, 1981).

3.10.1 Structure of the Box Model

The box model for the Philadelphia Site consists of a rectangular water column box 19.2 km wide, 9.5 km long, and 51 m deep (Figure 3.6). The long axis of the box is perpendicular to the coast and generally perpendicular to the ridge and swale topography. The area of the box is 182 km^2 and the volume of the water in the box is 9.3×10^9 m^3. The sediment box is also 182 km^2 and the thickness of the bioturbated layer is 5 cm. There is no information on the sediment density at the Philadelphia Site. For the purpose of calculating sediment inventory, the dry sediment density is assumed to be 0.8 g cm^{-3} (compared to 0.7 g cm^{-3} at the Lower Bay Complex). With a sediment density in this layer of 0.8 g cm^{-3} the mixed layer contains 7.3×10^9 kg sediment.

Contaminant residence times in the sediment mixed layer are only calculated for PCB, PAH, Pb, Cu and Cd (i.e., pollutants for which there is sufficient sediment contaminant information to calculate sediment inventory with a degree of confidence). For all contaminants, site-specific water column information is lacking and data from the New York Bight to the north of the Philadelphia Site are used for modeling purposes.

3.10.1.1 Sediment Box Model

The general formulations of residence time equations used to calculate $(T)_{in}$ and $(T)_{out}$ have been presented in Section 3.3.1. Modified for the Philadelphia Site, the equation for mass loading is:

$$\left(\frac{dA_x}{dt}\right)_{in} = \left[C_{x(tsm)}\right]_{in} Q_{sed}$$

and the removal equation is:

$$\left(\frac{dA_x}{dt}\right)_{out} = C_{x(sed)}\ Q_{rs}\ A_s + Z$$

Mass loading at the Philadelphia Site is strictly a function of suspended sediment deposition. Since there is no authorized dumping at the Philadelphia Site, the ocean dumping term drops out of the equation. Production of contaminants in the sediments (P_x) is assumed to be negligible.

Data of Swift et al. (1971) and Krebal and Folger (1976) indicate that the process of deposition and resuspension are in equilibrium on the Continental Shelf in the region of the Philadelphia Site. Therefore, there is no net removal of sediment by burial. Since the burial rate is equal to zero, the flux of pore water is also equal to zero, assuming a constant porosity in the sediment mixed layer (Stumm and Morgan, 1981). Therefore, removal terms in the sediment mixed layer consist only of resuspension and biodegradation. Biodegradation is only considered a removal term for PAH. The PCB biodegradation rate is considered negligible.

3.10.1.2 Water Column Box Model

Residence times for contaminants in the Philadelphia Site water column are calculated from the following modification of the mass loading equation (6):

$$\left(\frac{dA_x}{dt}\right)_{in} = \left[C_{x(dis)}\right]_{in} Q_w + \left[C_{x(part)}\right]_{in} Q_w + Q_{w(dep)}\ A_s + C_{x(sed)}\ Q_{rs}\ A_s$$

At the Phildelphia Site, the mass loading term consists of the flux of contaminants in advected seawater, atmospheric deposition, and resuspended bottom sediments entering the box. Since the pore water flux is negligible at the Philadelphia Site, this term drops out of the equation. Of the mass loading terms, the most significant is the advection term. The concentrations of pollutants being advected into the site are assumed to be the same as pollutants being advected out of the site.

Modified for the Philadelphia Site, the removal process equation is:

$$\left(\frac{dA_x}{dt}\right)_{out} = \left[C_{x(dis)}\right]_{out} Q_w + \left[C_{x(part)}\right]_{out} Q_w$$

Contaminant removal at the Philadelphia Site consists solely of the advection of contaminants out of the box. Since there is no net sedimentation this term drops out of the equation. Also, since the decay rate of contaminants is much slower than the residence time of contaminants in the box, the microbial decay term is not considered in the calculation.

3.10.2 Generic Parameters

3.10.2.1 Advective Transport

Ketchum (1953) and Norcross and Stanley (1967) have both cited a southeasterly drift of surface water at the site. Studies by Oostdam (1983) have shown a net easterly drift of 26-34 km day^{-1} at 18 m depth at the site, while recording a net bottom drift to the southwest of 10 km day^{-1}. In other studies Cook (1980) recorded net drifts of 8 km day^{-1} at the southwest corner of the site and 15 km day^{-1} at the northeast corner of the site.

For the purposes of this study a net drift of 11.5 km day^{-1} to the southwest is used to calculate water residence time of 1.0 day at the site. Thus, the residence time of a dissolved contaminant in site waters is one day, and the upper limit of particulate matter residence times at the site is also one day.

3.10.2.2 Transport of Suspended Matter

The advective transport of natural suspended matter is 4.9 x 10^9 kg yr^{-1} assuming a 11.3 km day^{-1} southwesterly current and a 1.2 mg kg^{-1} SPM load. For the purposes of this study, this natural SPM is considered the only source of SPM to the box. The SPM is removed from the box strictly by advection. Sedimentation in the box is set equal to resuspension. Both sedimentation and resuspension are calculated from the effect of storm events on the bottom 10 m overlying water and are set at 0.2 g cm^{-2} yr^{-1}. To maintain a sediment mass balance, advection of SPM out of the box must equal 4.9 x 10^9 kg yr^{-1}.

3.10.2.3 Mass Loading of Contaminants

The mass loading of contaminants entering the Philadelphia Site box is shown in Table 3.10.1. Site-specific particulate contaminant data are unavailable and data from the New York Bight are used for these inputs. PAH and PCB data are taken from Boehm's (1983b) study in the New York Bight, whereas Betzer's data (1978) from the Bight is used for particulate metals. Atmospheric fallout data for metals and PAH were calculated from atmospheric inventory, settling and rainout data presented in Duce and Kester (1981). The PCB deposition rate was calculated as direct gas exchange (Liss, 1973; Liss and Slater, 1974) using atmospheric PCB levels from Harvey and Steinhauer (1974). In all cases the mass loading of advected transport far exceeds atmospheric deposition.

3.10.2.4 Sediment Chemistry

The concentration of contaminants and contaminant inventory in shelf sediments at the Philadelphia Site is presented in Table 3.10.2. The data taken from Lear and O'Malley (1983) and EPA (1981) are site-specific, whereas the PAH data used is from the shelf sediment at the southern boundary of the New York Bight (Boehm, 1983a). Data for Ag and As are unavailable for the site or related areas. Due to the wide range of contaminant levels in site sediments, selection of one value to input in the model is difficult. The uncertainty in the contaminant-specific data is presented in Section 3.10.4. The mixed-layer inventory is the mass of contaminants in the upper 5 cm of site sediment assuming the selected sediment contaminant concentration from Table 3.10.2 and a sediment density of 0.8 g cm^{-3}.

3.10.2.5 Contaminants in the Water Column

The concentration of contaminants in the water column of the Philadelphia Site are presented in Table 3.10.3. Since there are no site-specific contaminant data available, shelf water quality data from the New York Bight are used for modeling purposes. Assumptions and references for the data set are discussed in Section 3.10.7. The advective flux of contaminants into the box is calculated assuming a net drift across the site of 11.5 km day^{-1} and a uniform distribution of contaminant in the water column. The inventory of contaminants was calculated by multiplying the total contaminant concentration by the volume of water in the box.

Table 3.10.1 Mass Loading of Contaminants at the Philadelphia Site.

	Contamination on Suspended Sediment (ug g^{-1})		Contaminant Advected[b] Suspended Sediment (kg yr^{-1})	Fallout & rainout[d] (kg yr^{-1})	Total Inputs (kg yr^{-1})
	$\bar{x} \pm s$	Selected Concentrations Box Model			
PCB	0.22 ± 0.1[a]	0.2	990	0.2	990
PAH	2.5 ± 2.5[a]	2.5	12,300	22	12,300
Lead	41 ± 25[c]	40	198,000	5,250	203,000
Copper	52 ± 40[c]	50	247,000	1,520	248,000
Cadmium	4.8 ± 4.5[c]	5	25,000	85	25,100

[a] Christiaensen Basin; Boehm (1983)
[b] Sedimentation at Site 0.2 g cm^{-2} yr^{-1}
[c] Betzer (1978)
[d] Calculated from atmospheric data presented in Duce and Kester (1981)

Table 3.10.2 Concentration, Inventory, and Annual Resuspension of Contaminants in Philadelphia Interim Dumpsite Sediments.

	$\bar{x} \pm s$ ($\mu g\ g^{-1}$)	Range ($\mu g\ g^{-1}$)	Selected Concentrations Box Model ($\mu g\ g^{-1}$)	Mixed Layer Inventory (kg)	Annual Resuspension (kg)
PCB	0.019 ± 0.009[a]	0.008 - 0.034 0.004 - 0.008[b]	0.020	146	7.3
PAH	15.8 ± 12.9[c]		16	117,000	5,824
Pb	2.8 ± 1.4[b] 3.3 ± 1.6[a]	0.7 - 6.7	3	21,900	1,090
Cu	0.65 ± 0.46[b] 1.5 ± 0.6[a]	0.3 - 3 0.93 - 2.9	1.1	8030	400
Cd	0.013 ± 0.009[b]		0.01	73	3.6
Hg	0.004 ± 0.003[b]		0.004	29.2	1.5
Ag	No data				
As	No data				

[a] Lear & O'Malley (1983)
[b] EPA (1981)
[c] Christiaensen Basin; Boehm (1983)

Table 3.10.3 Philadelphia Site Water Column Chemistry.

Contaminant	Dissolved	Particulate	Total	Advective Flux	Inventory Water Column
	(μg L^{-1})	(μg L^{-1})	(μg L^{-1})	(kg yr^{-1})	(kg)
PCB	0.00055	0.00055	0.0011	453	10.2
PAH	0.00625	0.00625	0.0125	5,150	116
Pb	0.15	0.10	0.25	103,000	2,325
Cu	7.6	0.13	7.7	3,174,800	71,080
Cd	0.77	0.012	0.78	321,200	7,250

3.10.3 Residence Times in the Philadelphia Site

3.10.3.1 Residence Times in Sediments

The residence times of contaminants in the sediment mixed layer, calculated from mass loadings (T_{in}) and from removal processes (T_{out}), are presented in Table 3.10.4.

The mean deposition rate is calculated from the data of Butman and Moody, (1983), estimating the resuspension occurring during storm events mid-shelf in the western North Atlantic. Deposition is set equal to resuspension for box model calculations. Since the residence time (T_{out}) equation for all contaminants (except PAH) contains only sediment contaminant levels and resuspension fluxes as variables, and since the fluxes are also a function of sediment contaminant levels, residence times for all contaminants calculated from removal processes are the same.

With the exception of PAH, $(T)_{in}$ of all contaminants is under two years. Residence times calculated from removal processes are a uniform 20 years except for $(T)_{out}$ for PAH which is lower due to the estimated rate of microbial decay in the sediment mixed layer. It is assumed that degradation rates for PCB are at least 10 times less than for PAH.

3.10.3.2 Residence Times in the Water Column

Since there is no net deposition out of the water column box, the residence time of suspended and dissolved contaminants is the same and equals the water column residence time at the site, assuming that any degradation processes taking place within the water column are negligible. Assuming an 11.5 km day^{-1} drift the calculated water column residence time is 0.98 days. Therefore, the $(T)_{in}$ and $(T)_{out}$ for all contaminants in 0.98 days.

3.10.4 Sensitivity Analysis

The ability of the model to predict contaminant residence times in the sediment mixed layer and water column of the Philadelphia Site is directly dependent upon the reliability of the data. Any uncertainty in the generic or contaminant-specific data will translate into the same uncertainties in calculated residence times. The fact that only PCB, Pb, Cu and Cd sediment levels are site-specific and that all other data used in the box model are from other regions leads to great uncertainity in the residence times calculated from input data.

Table 3.10.4. Residence Times of Contaminants in the Sediment Mixed Layer of the Philadelphia Site in Different Rates of Sedimentation and Resuspension Using Mean Sediment and SPM Concentrations.

Contaminant	T_{in}[a] (years)	T_{out}[b] (years)
PCB	1.8	20
PAH	129	13.4
Pb	1.5	20
Cu	0.42	20
Cd	0.04	20

[a] Calculated assuming a deposition rate of 0.2 g cm^{-2} y^{-1}.

[b] Calculated assuming a resuspension rate of 0.2 g cm^{-2} yr^{-1} and a degradation rate of 1.6 µg PAH cm^{-2} yr^{-1} (from Lee and Ryan, 1983).

Therefore, the most reliable residence time estimate is that calculated from $(T)_{out}$ removal terms which are not dependent on contaminant concentration data (i.e., numerator and denominators cancel). There is good agreement between investigators that, for the past 1000 years, the Mid-Atlantic shelf has been in equilibrium between depositional and removal processes (Swift et al., 1971; Butman and Noble, 1979). However, site-specific data are unavailable to define what short-term processes are occurring at the site over the last 10-20 years. Extrapolating resuspension data during storm events at other shallower sites on the Mid-Atlantic shelf is presently the only way of estimating resuspension and deposition at the Philadelphia Site. During any given year, true values of $(T)_{out}$ could differ from the estimate by the one-half order of magnitude. Table 3.10.5 presents the effect of this uncertainty on $(T)_{out}$.

Other uncertainties in the generic parameters include sediment density and depth of the sediment mixed layer. The calculation of contaminant inventories assumes a 5 cm mixed layer (0-5 cm depth) with a sediment dry density of 0.8 g cm^{-3}. As discussed previously, the mixed layer in these mid-Shelf sediments may include as little as 0-2 cm or as much as 0-8 cm. The dry density of sediment in sheltered swale regions may be closer to 0.6 g dry sediment cm^{-3}, similar to values at the Christiaensen Basin, whereas the density may be 1.0 g cm^{-3} in ridge regions. The effect of these uncertainties on sediment residence times are also presented in Table 3.10.5.

Since contaminant-specific data is mostly drawn from the New York Bight region, the reader is directed to Section 3.7.4 for specific analysis. In addition to the actual uncertainty of the contaminant data there is a further introduction of error simply in using data from the New York Bight in Philadelphia Site calculations. This is most obvious in the calculations of $(T)_{in}$ for PAH in the sediment mixed layer (Table 3.10.4). The PAH inventory was calculated from New York Bight sediment data, whereas the other contaminant concentrations were site-specific. The high value of $(T)_{in}$ for PAH is probably a result of the inflated PAH inventory. On the other hand, all $(T)_{in}$ values (except PAH) are considerably lower than $(T)_{out}$, suggesting that the rate of addition is greater than the removal process. This is most likely a result of using suspended sediment data from the New York Bight to calculate mass loading at the Philadelphia Site.

Water residence times and, hence, contaminant residence times are calculated assuming a net southwesterly water drift of 11.5 km day^{-1} over the site. Reported drifts range from 8-34 km day^{-1} in easterly to southwesterly directions (Section 3.10.2). Assuming that 8 km day^{-1} and 34 km day^{-1} represent the full uncertainty in this advective parameter, the calculated contaminant residence times are 1.4 and 0.33 days, respectively.

Table 3.10.5. Sensitivity Analysis of $(T)_{out}$ Based on Uncertainty in Generic Parameters at the Philadelphia Site.

Generic Parameter	T_{out} (years)
Resuspension Rate (g cm^{-2} yr^{-1})[a]	
0.063	63
0.2	20
0.63	6.3
Density of Mixed Layer (g cm^{-3})[b]	
0.6	15
0.8	20
1.0	25
Depth of Mixed Layer (cm)[c]	
2	8
5	20
8	32

[a] Depth of mixed layer 5 cm with a density of 0.8 g cm^{-3}

[b] Using resuspension rate of 0.2 g cm^{-2} yr^{-1}; depth of mixed layer 5 cm

[c] Using resuspension rate of 0.2 g cm^{-2} yr^{-1}; density of 0.8 g cm^{-3}

3.10.5 Analysis of Model Results

Assuming that the contaminants in the sediment mixed layer are in equilibrium with contaminants in the water column, residence times calculated from input terms (T_{in}) should equal those calculated from removal terms (T_{out}). In fact, for all contaminants except PAH, the calculated $(T)_{in}$ values are much smaller than corresponding $(T)_{out}$ values. The disparity most likely arises from the inadequate water chemistry data that are valid for the New York Bight, but not necessarily for the Philadelphia Site. Particulate contaminant levels at the Philadelphia Site which, when combined with sedimentation rates, form the mass flux or input to the sediments are most likely less than those reported in the New York Bight and used here for lack of water column chemistry data at the site. An inflated particulate contaminant term, in turn, inflates the contaminant flux into the box, thus decreasing residence time for any given inventory level. It appears, from the calculations in Table 3.10.4, that these data may overestimate the water column particulate contaminant levels by a factor of 10 for PCB and Pb, and up to a factor of 500 for Cd. In the other extreme, $(T)_{in}$ for PAH is inordinately high compared to $(T)_{out}$, an imbalance most likely caused by an overestimate in the PAH inventory at the site. New York Bight sediment PAH levels were used for the calculation of residence time at the Philadelphia Site. Site sediments should be much less contaminanted with PAH. If depositional rates are correct, then sediment PAH levels appear to be overestimated by a factor of six. Additional confirmatory data on water column PAH, PCB and metals particulate levels are much needed.

Sediment contaminant residence times for those contaminants modeled are probably best calculated as $(T)_{out}$ and are uniformly 20 years, except PAH which has a $(T)_{out}$ of 13.4 years. Given the uncertainty in generic parameters used to calculate $(T)_{out}$ the range of $(T)_{out}$ is 6.3-63 years.

The residence time of contaminants in the water column is uniformly 1 day, and, given the uncertainty of the data, actual residence times may range from 0.33-1.4 days. Hence, the advection of water through the site has an enormous impact on water quality, and it is not surprising that investigations of water quality at the Philadelphia Site during the years of active sludge dumping could find no impacts of dumping on water quality.

3.10.6 Sediment and Water Quality for Hypothetical Loadings

For the purpose of calculating the response of contaminants in the sediment mixed layer of the Philadelphia Site to hypothetical loadings, it is assumed that the dumping of sewage sludge is resumed at the site. Table 3.10.6 presents sediment

Table 3.10.6 Predicted Concentration of Contaminants in Philadelphia Site Mixed Layer Sediments if Hypothetical Loadings are Changed by 1, 2, and 10 Times the Maximum Annual Loading for Previous Years.

	ug g⁻¹ dry wt			
		Loading Factor		
	Philadelphia Site 1979-1981 (OX)	1X	2X	10X
PCB	0.020	0.020	0.021	0.03
PAH	16.0	16	16	17
Pb	3.0	8.9	11.9	62.1
Cu	1.1	1.98	2.86	9.8
Cd	0.01	0.02	0.02	0.07

OX = no loading, or present condition assuming steady state contaminant levels in response to 1, 2, and 10 times the maximum annual dumping that occurred at the Philadelphia Site. In 1976, 0.6 x 10^6 m^3 of liquid sludge containing 0.1 x 10^9 kg solids was dumped at the site (EPA, 1981). For the purpose of calculation it is assumed that dumping resumes and that 0.1 x 10^9, 0.2 x 10^9, and 1.0 x 10^9 kg solids are dumped at the site. It is further assumed that sludge solids are suspended upon dumping and homogeneously mixed throughout the water column, and that the system reaches equilibrium with the increased mass loading. Waste loadings in the hypothetical sewage sludge are the same as that reported for 1975/1976 (EPA, 1983) except for PCB and PAH loadings which are taken from O'Connor et al. (1983).

It can be seen from Table 3.10.6 that only sediment Pb and Cu are increased significantly by increased mass loading at the Philadelphia Site. Dumping would have to be increased to 10 times 1976 levels before sediment levels would increase beyond those found elsewhere on the Continental Shelf (EPA, 1981). Since a significant amount of Cu would be dissolved in the water column rather than sedimented, the Cu loadings are probably an overestimate. Also, since Pb levels in sewage discharges elsewhere have been reported less than half of early 1970's levels (SCWRP, 1984), Pb mass loadings based on 1975/1976 data are probably overstated and Pb levels in sediments would not increase to the extent shown in Table 3.10.6. The concentrations of other contaminants in sludge are similar to those of suspended sediments using data from the New York Bight, and dumping of sludge will have little effect on sediment contaminant concentrations of PCB, PAH and Cd.

The response of contaminant levels in the water column, assuming resumption of dumping 1, 2 and 10 times the maximum annual sludge dumped at the Philadelphia Site, is presented in Table 3.10.7. Since the residence time of water at the site is 0.98 day, the total mass load of contaminants in sludge was divided by 365. This mass load is assumed to be uniformly dispersed in the water column at the Philadelphia Site. No consideration was made for dissolved and particulate contaminant levels and all values are for total contaminant on a volume basis.

Table 3.10.7 shows that only Pb levels in the water column rise significantly above background levels. The changes of all other contaminant levels would be undetectable under hypothetical loadings conditions, given the natural variability of the measurements.

3.10.7 Summary and Research Needs

The residence times for contaminants in the mixed layer at the Philadelphia

Table 3.10.7 Concentration of Contaminants in Philadelphia Site Water Column if Hypothetical Loadings are Changed by 1, 2, and 10 Times the Maximum Annual Loading for Previous Years.

	ug L-1				
			Loading Factor		
	EPA Criteria Marine	Philadelphia Site 1979-1981	1X	2X	10X
PCB	0.014	0.0011	0.0011	0.0011	0.0012
PAH[a]	8	0.0125	0.0125	0.0125	0.0130
Pb	8.6	0.20	0.22	0.33	1.25
Cu	2.0	0.65	0.71	0.77	1.24
Cd	12	0.034	0.04	0.04	0.07

[a] Fluoranthene water quality criteria

Site sediments are in the range of 20 years calculated as $(T)_{out}$. The residence times for contaminants in the water column are approximately one day, the residence time of water at the site. Regarding the contaminants modeled, the shelf sediments at the site show no impact of the sludge dumping that occurred at the site in the 1970's. Since $(T)_{out}$ in the sediment box is 20 years it is expected that, if the sediments were initially contaminated by dumping activities, the contamination would still be found. Therefore, a conclusion to be drawn from the study is that the impact of sludge dumping on site sediments was initially limited in magnitude due to the constant turnover of site waters and that the sludge was diluted greatly before settling on the Continental Shelf.

If dumping or some other activity resumes at the site and requires future environmental monitoring, the monitoring program should focus on the needs of the contaminant residence time model as a guide for future measurements. In this regard, the monitoring program would have as a goal a complete data set needed to accurately determine residence times at the site and, hence, actual and projected consequences of dumping.

During the 1970's numerous environmental monitoring programs focused on identifying the environmental impacts of sludge dumping at the Philadelphia Site. In general, these programs focused completely on sediment contaminant levels and, for this reason, there are no site-specific water quality data resulting from these studies. The studies found little environmental impact of sludge dumping on sediment quality, partially because the relatively low mass loadings of sludge dumping and, in part due to the extremely high advective flux of water through the site. Statistically, the impacts of measured contaminants were negligible due to the high natural variability on the shelf (EPA, 1981). This type of monitoring strategy should not be repeated unless it is part of larger program encompassing all data needed to model the processes occurring at the site.

3.11 Summary and Critique of Residence Time Modeling Results

The computational model presented and used to calculate (T) values in the five study areas has strengths and weaknesses. The strength of the model is in its ability to view each system in a holistic, interrelated manner. With such a computational model environmental managers and others evaluating various waste disposal strategies can consider the effect of altering disposal strategies on the equilibrium, ambient toxic contaminant concentrations. Other contaminants (e.g., pathogens in sediments) can also be considered as they parallel toxic contaminant behavior (e.g., pathogens can be assumed to behave as sediment itself, or as sediment-associated refractory organic materials such as PCB). Also, by computing a (T) value for contaminants in sediments one can approximate the response time of the sediment to change in loadings (i.e., how long will it take before a new equilibrium concentration is achieved?). The model presented contains inherent checks on the computation results. The $(T)_{in}$ values calculated from mass loading data for each environmental compartment studied should approximate the $(T)_{out}$ values calculated from removal processes. When they don't, errors or uncertainties in generic parameter values or an ambient concentration data need be examined. Error in our basic assumption that the systems are in steady-state equilibrium may be responsible for discrepancies as well.

A great strength and important use of the model lies in the ability of the results to point towards critical data needs of environmental monitoring and research programs. For example, when one conceives of an environmental monitoring program, one of course designs a study around contaminants of interest, statistical considerations of replication, location of study sites, etc. However, this modeling exercise has shown that measurement of the generic parameters such as sedimentation rate and resuspension rate are equally important data needs in most areas. Without confidence in, for example, the resuspension rate, potential utility of the model to answer "what if" type questions regarding waste disposal strategies is severely limited.

Using a holistic view of environmental monitoring data acquired in the five study areas over the past decade, one concludes that the most viable and useful data set comes from the Puget Sound Central Basin. A good grasp of generic parameter values and mass loading data acquired over the past decade leads to confidence in calculated (T) values for all but PAH, where good ambient water column data are lacking. The sediment instantaneous resuspension rate is fairly well known, although fortuitously it is not needed for (T) calculations in the main basin due to the "trap" nature of the basin itself.

Similarly, the comprehensive nature of the Palos Verdes Shelf data base lends confidence and utility of calculated (T) values. An excellent mass loading data base, along with a reasonable knowledge of the generic parameter values, creates this confidence. However, gaps in this data base necessitated the extrapolation of Puget Sound ambient water column concentration data to the Palos Verdes Shelf. This extrapolation surely leads to some unquantifiable error.

The strengths of the computational model in providing for "what if" interrogatories and in guiding measurement programs also are the sources of the model's greatest weakness. That is, if the data used in (T) calculations is uncertain, the (T)-generated values are themselves highly suspect. In systems such as the Hudson-Raritan Estuary Lower Bay Complex and the Christiaensen Basin, $(T)_{in}$ values compare only marginally with $(T)_{out}$ values, and the sensitivity or uncertainty analysis reveals that (T) can vary considerably. These variations obviate the utility of the (T) values and the model to answer hypothetical loading questions.

The Lower Bay Complex and Christiaensen Basin systems are perhaps the most complex systems studied in this research program. Nevertheless, the great amount of monitoring data obtained over the last decades is severely limited in its systematic use by the sparsity of information on important generic parameter values, including resuspension rates in the Lower Bay Complex, and deposition and resuspension rates in the Christiaensen Basin. This study did not deal directly with ocean waste disposal inputs in the New York Bight, but instead dealt with them indirectly as part of the overall observed contaminant loading to the Basin. Had the model been directly reliant on data on the deposition of sewage sludge particles or resuspension of dredged material deposits and fluxes into the Basin, we would have found that these data were very poorly known. No one knows confidently if the Basin is net depositional or erosional and what the rates of these important processes are!

The model weakness (i.e., the dependence on empirical data) is revealed dramatically in the Philadelphia Site where the water column data base and knowledge of generic parameter values are both poor, in spite of considerable monitoring effort which focused on sediment contaminant concentrations.

However, as an overall framework, we believe that our modeling approach is extremely useful. We have chosen a level of detail in which individual physico-chemical processes are lumped into other parameters. We defend this approach of relying on a practical, computational model based on empirical observations rather than one based on detailed theoretical model which is dependent on theoretical expressions and laboratory-derived constants. A model is only as good as one's ability to verify it through field observations. Many of the "theoretical constants" used physico-chemical processes actually vary from site to site. A pointed example is the often-used "apparent" sediment-water partition coefficient (K_d) which is highly dependent on sediment type, organic coating, ambient seawater/porewater, total organic content, etc. The empirical approach of relying on actual data to compute (T) is, of course, a two-edged sword. One's (T) calculations live and die by the data base.

Therein, in our opinion, lies the model's real asset and utility. Scientists and scientific mangers are being asked to generate better information with fewer resources. If we are to satisfy the needs of the marine research community to understand "how the system works", with the needs of scientific and environmental managers to understand the environmental consequences (i.e., toxic loadings and ambient concentrations) of changes in waste disposal strategies, we maintain that the residence time model employed in this study can most efficiently guide the real needs of both groups and, hence, guide the utilization of our research and monitoring resources.

4. ECOLOGICAL RECOVERY

4.1 Introduction

There has been a growing focus in recent years on incorporating ecological concepts and criteria into federally mandated standards for marine environmental quality. With the constant improvement in analytical chemical methods, lower and lower concentrations of contamination can be detected. As a result, it has become apparent that some level of contamination pervades almost the whole marine environment. Strictly speaking, contamination becomes pollution only when it has a deleterious effect on populations and communities (Clark, 1982). The point at which contamination becomes pollution is different for different ecosystems, communities, and populations. As a result, major marine environmental protection legislation and the resulting environmental regulations require the maintenance of stability, diversity and balanced indigenous populations, and prohibit activities which will result in unreasonable degradation and/or irreparable harm to the marine environment.

These concepts are difficult to interpret and apply in a regulatory context, mainly because ecosystems are not static entities. All ecosystems are in a process of constant change with respect to community structure and function, even where the physical/chemical characteristics of the ecosystem are relatively constant (Green, 1984). Only the rate of change varies from one ecosystem to another.

All marine and estuarine ecosystems are subjected, at least periodically to disturbance and stress. Usually, the community structure of a particular ecosystem is controlled by these disturbances which disrupt the community and return the successional process to an earlier stage (Green, 1984). Different ecosystems respond differently to stress or disturbance. The ability of a community or ecosystem to weather a stress, disturbance or perturbation and return to normal afterward is defined as stability (Harrison, 1979; Santos and Bloom, 1980). The concept of stability can be subdivided into several components (Orians, 1974; Westman, 1978; Boesch and Rosenberg, 1981).

- Persistence is the constancy of the community over time, irrespective of perturbations;
- Resistance is the ability of a community to remain unaffected by perturbations;

- Resilience is the ability of the ecosystem or community to recover to some more or less persistent state following disturbance.

Resilience can be subdivided further into four components (Westman, 1978):

- Elasticity is the rapidity of restoration of a particular ecosystem characteristic to its pre-impact state;
- Amplitude is the magnitude or upper threshold of disturbance-mediated change from which the ecosystem is able to return to the pre-disturbance state;
- Hysteresis is the degree to which the path to restoration or recovery is an exact reversal of the path of degradation;
- Malleability is the degree to which the persistent state established after disturbance differs from the original state.

The characteristics of a marine community which determine its stability are poorly understood. The long-held belief that complex highly diverse communities are more stable than simple, low diversity communities has been shown to be untrue in many cases (Goodman, 1975; Harrison, 1979). Quite frequently, simple estuarine benthic communities are more resistent and more resilient to man-induced and unusual natural stresses than similar but more diverse communities found under more oceanic conditions (Sanders, 1968; Boesch and Rosenberg, 1981; Hyland et al., 1985).

Recently, Hyland et al. (1985) provided experimental evidence in support of the hypothesis that simple, low-diversity communities found in inconstant and unpredictable environments are inherently more stable than the diverse communities characteristic of more constant and predictable environments. They compared the response of two benthic communities in Narragansett Bay, Rhode Island to equivalent additions of No. 2 fuel oil at a nominal concentration of 10,000 ppm in experimental sediment chambers deployed in each of the two environments. The inconstant environment was a shallow estuarine marsh cove with a benthic community dominated by the polychaete Streblospio benedicti and the oligochaete Tubificoides brownae. The more constant environment was a marine area of the bay with fine silty sediments in about 9 m of water with a benthic community dominated by the bivalve Nucula annulata and the polychaete Mediomastus ambiseta.

During the first month after deployment of the chambers, the concentration of total petroleum hydrocarbons in both treatments dropped to about 5,000 ppm and remained at this level for the remainder of the one year experiment. Forty-eight hours after the oil was added, there were 33 and 50 percent fewer species in the marsh cove and bay samples, respectively than in the corresponding nearby natural bottoms. There was very rapid recruitment of species to the oiled marsh cove samples, so that after one month these samples contained 48 percent more species than found in the adjacent natural bottom. After twelve months, number of species in the oiled and natural bottom were approximately the same.

In the bay, recruitment to the oiled sediments was much slower. After four months, there were still fewer species in the oiled than in the natural bottom. After twelve months, the number of species in the oiled samples was only nine percent less than in the natural bottom.

In the marsh cove, 69 percent of the species present were resistant and responded to the oil by population growth, while 65 percent of the species present in the bay samples were sensitive to the disturbance and responded by population declines. Thus, benthic infaunal communities in the inconstant environment were much more resistant and resilient to pollutant disturbance than those in the more constant environment.

However, a precise definition of ecological recovery is difficult. Marine communities are not static, but instead are undergoing constant change. This process of change is called ecological succession. Odum (1969) has defined ecological succession in terms of the following parameters. "1) It is an orderly process of community development that is reasonably directional and, therefore, at least theoretically predictable. 2) It results from modification of the physical environment by the community; that is, succession is community-controlled even though the physical environment determines the pattern, the rate of change, and often sets limits as to how far development can go. 3) It culminates in a stabilized ecosystem, the climax, in which maximum biomass and symbiotic function among organisms are maintained per unit of available energy flow." Succession occurs in an ecological community following a perturbation that returns the community to an earlier successional state. Since estuarine and nearshore marine ecosystems are continually subjected to natural distrubance, they never reach the climax state, but are undergoing continuous change. At any given time, a marine ecosystem may be characterized by a mosaic of successional stages representing the evolving spatial pattern since the last disturbance (Gray, 1977; Rhoads and Germano, 1982; Green, 1984). Even supposedly stable deep-sea communities undergo change or succession, but on a longer time-scale.

Gray (1977) argues that, because of this complexity, Odum's definition of succession leading toward a climax community, called global stability, is not a realistic model for change following disturbance in most benthic ecosystems. Instead, marine ecosystems may exhibit neighborhood stability, characterized by multiple stable points or seral stages of different degrees of stability and longevity (May, 1977). In such a system, pioneer and final communities are predictable, whereas intermediate successional communities and sequences, some of which may be relatively persistent, are less predictable (Pearson and Rosenberg, 1978). In such a system, it is difficult to define explicitly what is the "normal" state of a given marine ecosystem and correspondingly uncertain when recovery from damage is complete.

Thus, recovery from a pollutant insult in estuarine and marine ecosystems must be viewed as a process rather than an endpoint. Recovery should be defined in terms of rate and degree of change in the community toward the pre-disturbance state, and not necessarily in terms of attainment of the same species-abundance-age-size structure of the community as existed before the disturbance. However, in comparing the resilience of different ecosystems, both the endpoint and the process should be considered.

At least in theory, the rate of recovery (elasticity) of a particular community from different disturbances of similar amplitude should be the same. However, disturbance due to pollutants, particularly in benthic ecosystems, tends to be longer-lasting than most types of natural disturbance, resulting in slower recovery. In the case of pollutant-mediated disturbance, three stages, partially overlapping in time, are involved in the recovery process: 1) disappearance of the pollutant from the ambient medium through dilution or degradation; 2) disappearance of the pollutant from the biota through metabolism or depuration; and 3) ecological succession toward the prestressed community state. The rate of the first two processes is dependent on the nature of the ecosystem, affected populations and pollutant. The first is discussed in detail elsewhere in this report. The characteristics of the second and third stages and the extent of overlap with the earlier stage are the subject of this review.

4.2 Benthic Recovery

4.2.1 Shallow Subtidal Benthos

4.2.1.1 Background.

Following an acute pollution event, such as an oil or hazardous waste spill, or following a substantial decrease or cessation of a chronic point or nonpoint discharge of contaminants into an estuarine or coastal marine ecosystem, ecological recovery will begin only after the concentration of contaminants is reduced to a level that can be tolerated by indigenous biota. Recovery in the water column may begin almost immediately as contaminants are diluted rapidly by dispersion, degradation, and sedimentation. In benthic, wetland, and shoreline regimes, contaminants tend to be more persistent and initiation of ecological recovery may be delayed and its rate, once started, slowed. Cato et al. (1980) identified six processes important to chemical recovery of estuarine sediments, processes which have been considered (Section 3) as part of the contaminant residence time calculations:

1. a decrease or cessation of pollutant discharge;
2. good mixing and exchange of the estuarine water mass;
3. mixing of contaminated with uncontaminated sediments due to physical (i.e. storm-induced resuspension and transport) and biological (i.e. bioturbation) processes;
4. chemical and biological transformation of pollutants in sediments resulting in their release into the overlying water column, or immobilization in stable bound forms;
5. the persistence of different pollutant chemicals;
6. sedimentation of uncontaminated sediments resulting in burial of the contaminated sediment layer below the zone of biological activity.

The authors considered that the burial of the contaminated sediment layer by deposition of clean sediment to be the most important physical process leading to initiation of the ecological recovery process. Of course, bioturbation, by causing vertical mixing of sediments, may decrease the effectiveness or delay the completion of the burial process.

Pearson and Rosenberg (1978) reviewed the responses of macrobenthic communities of estuarine sediments to chronic organic pollution discharges (i.e. sewage, pulp and paper mill wastes). They observed a characteristic gradient of response of the benthic macrofauna along a pollution gradient away from the point discharge (Figure 4.1). Sediments in the immediate vicinity of the discharge are devoid of benthic macrofauna. The first animals encountered are small opportunistic species. As one moves further away from the most organically enriched area, the number of species present increases slightly, while the abundance of individuals increases dramatically to a high point, called the PO-point or peak of opportunists. Further from the discharge point, the abundance of individuals decreases to a point, the ecotone, where biomass levels off or decreases as number of species increases and number of individuals decreases. The ecotone point on this pollution gradient is a transition zone characterized by a benthic community poor in species, abundance, and biomass. On the more organically enriched side of the ecotone, the benthic community has low diversity, being composed of a few pollution-tolerant opportunistic species. On the less organically enriched side of the ecotone, the different transitory assemblages, or seral stages, gradually approach the composition of the community in the unpolluted environment. The community at the ecotone contains species from both adjacent communities, but in reduced numbers. Beyond the ecotone, as organic enrichment of sediments along the pollution gradient decreases further, the number of species and total biomass increase to a peak and then decline to a slightly lower level characteristic of the natural unstressed community. This peak in species richness and biomass probably occurs at the point on the gradient where the organic enrichment of sediments has decreased to the point where it is relatively non-toxic but is high enough to provide nutrition to lower trophic levels upon which the benthic macrofauna subsist. Thus, the shape of the species, abundance, biomass (SAB) profiles along pollution gradients of toxic non-biodegradable contaminants (i.e. heavy metals or chlorinated organics) may be different from those described here for organic biodegradable pollutants.

The change in SAB relationships along an organic pollution gradient in space also may occur along a temporal pollution gradient. Following initiation of an organic pollution discharge, SAB relations of the benthic community near the discharge will change (move to the left in Figure 4.1) as the sediment organic pollution load increases. Conversely, if a chronic discharge is stopped or substantially decreased, the benthic community near the discharge will show a change over time in SAB relationships in the opposite direction (i.e. to toward the right in Figure 4.1).

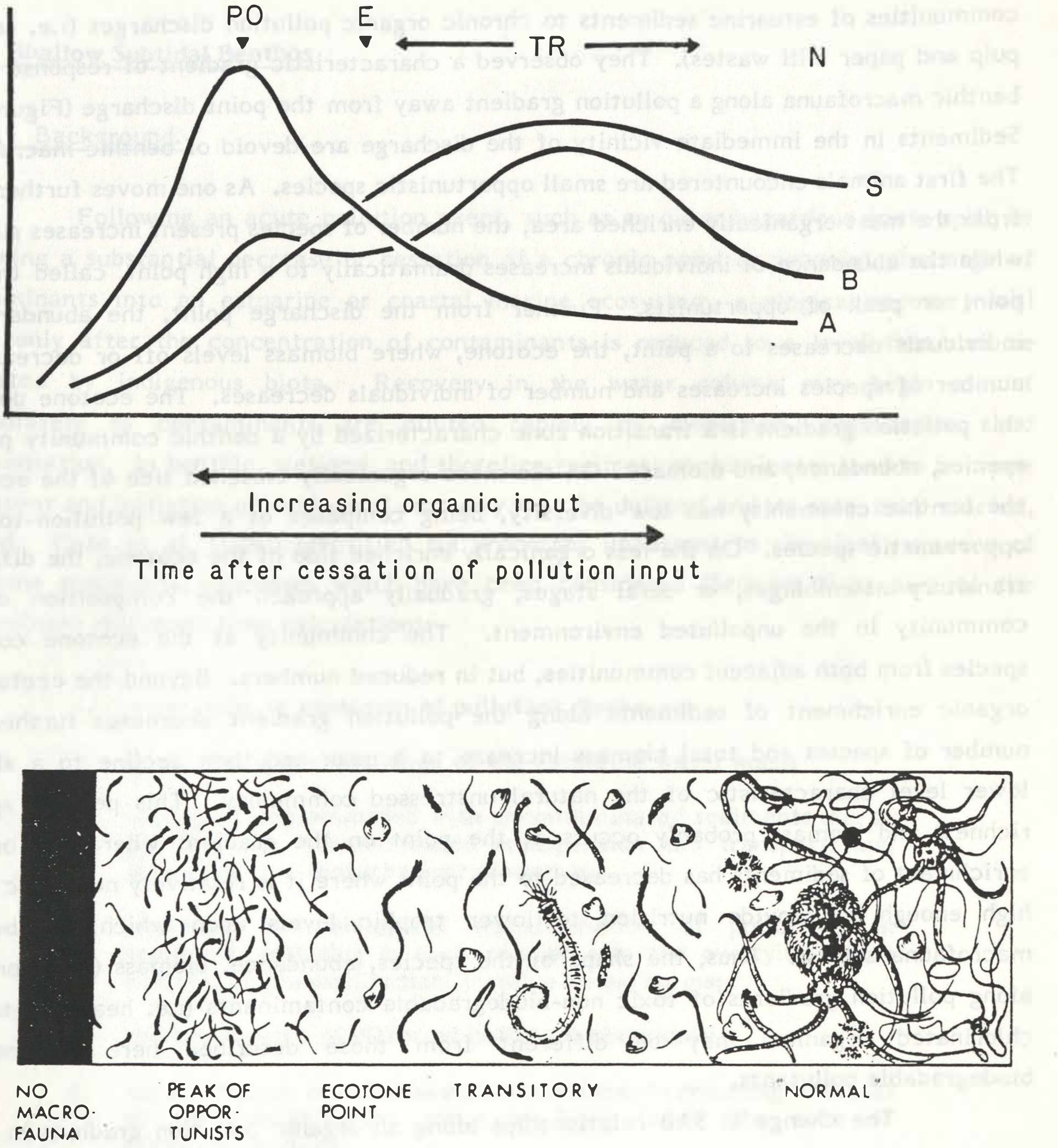

Figure 4.1 Generalized SAB diagram of changes in benthic infauna along an organic pollution gradient or over time after cessation of a pollutant discharge. S, species numbers; A, total abundance; B, total biomass; PO, peak of opportunists; E, ecotone point; TR, transection zone. The bottom part of the figure shows schematically the appearance of the benthic faunal community along the temporal or spatial pollution gradient (from Pearson and Rosenberg, 1976).

Similarly, following a natural disturbance which has resulted in the elimination or significant reduction of the normal macrofauna, the pattern of early succession of the benthic infauna is similar to that along an organic pollution gradient (Pearson and Rosenberg, 1978; Rhoads et al., 1978). In disturbed fine-grain sediments, particularly if they have been enriched with organic pollutants, the redox potential discontinuity (the RPD or point at which the oxidation/reduction potential drops below zero and sediments become anaerobic) moves toward or even to the sediment surface (Figure 4.2). The first colonizers of the disturbed area are small opportunistic tube-dwelling polychaetes or oligochaetes, such as Capitella capitata, or in some cases small short-lived molluscs like Mulinia lateralis (Rhoads and Germano, 1982). The pioneering species which colonize a disturbed bottom first will vary depending on the nature of the sediments, pollutant characteristics and concentrations, disturbance intensity, and types of larvae available for recruitment to the disturbed sediments. Most of the pioneers are very small and feed at the sediment surface or from the water column. Tube walls or shells isolate the colonists from the anoxic sediment. However, their activities gradually cause the RPD to sink deeper into the sediment. This allows larger longer-lived species to settle. These new recruits include increasing numbers of infaunal deposit-feeders which increase the depth of the RPD further through bioturbation. Finally, a more stable long-lived equilibrium assemblage develops which resembles the benthic fauna of similar nearby undisturbed areas. The rate of these changes is, of course, quite variable from site to site. Intertidal, shallow subtidal and estuarine sediments seem to recover more rapidly than deeper offshore sediments. This may be due to availability of larger numbers of pioneer organisms in shallow waters (Rhoads et al., 1978). Shallow water ecosystems frequently are characterized by frequent disturbance, such that a significant fraction of the normal benthic macrofauna consists of opportunistic pioneer species.

Boesch and Rosenberg (1981) reviewed several studies of recolonization of azoic sediment trays in different benthic environments. There was a very rapid recolonization of azoic sediments placed on the bottom in 14 m of water in Long Island Sound (McCall, 1977). The pattern of succession in the trays was similar to that outlined in Figure 4.1 with a rapid establishment of the peak of opportunists in ten days to two months, followed by establishment of a more stable community similar to that in the surrounding seafloor in 5-12 months. At a nearby dredge material dump site in 20 m of water, recruitment of benthic fauna to the dredge material pile composed of contaminated sediments from New Haven Harbor, Connecticut was slower (Rhodes et al.,

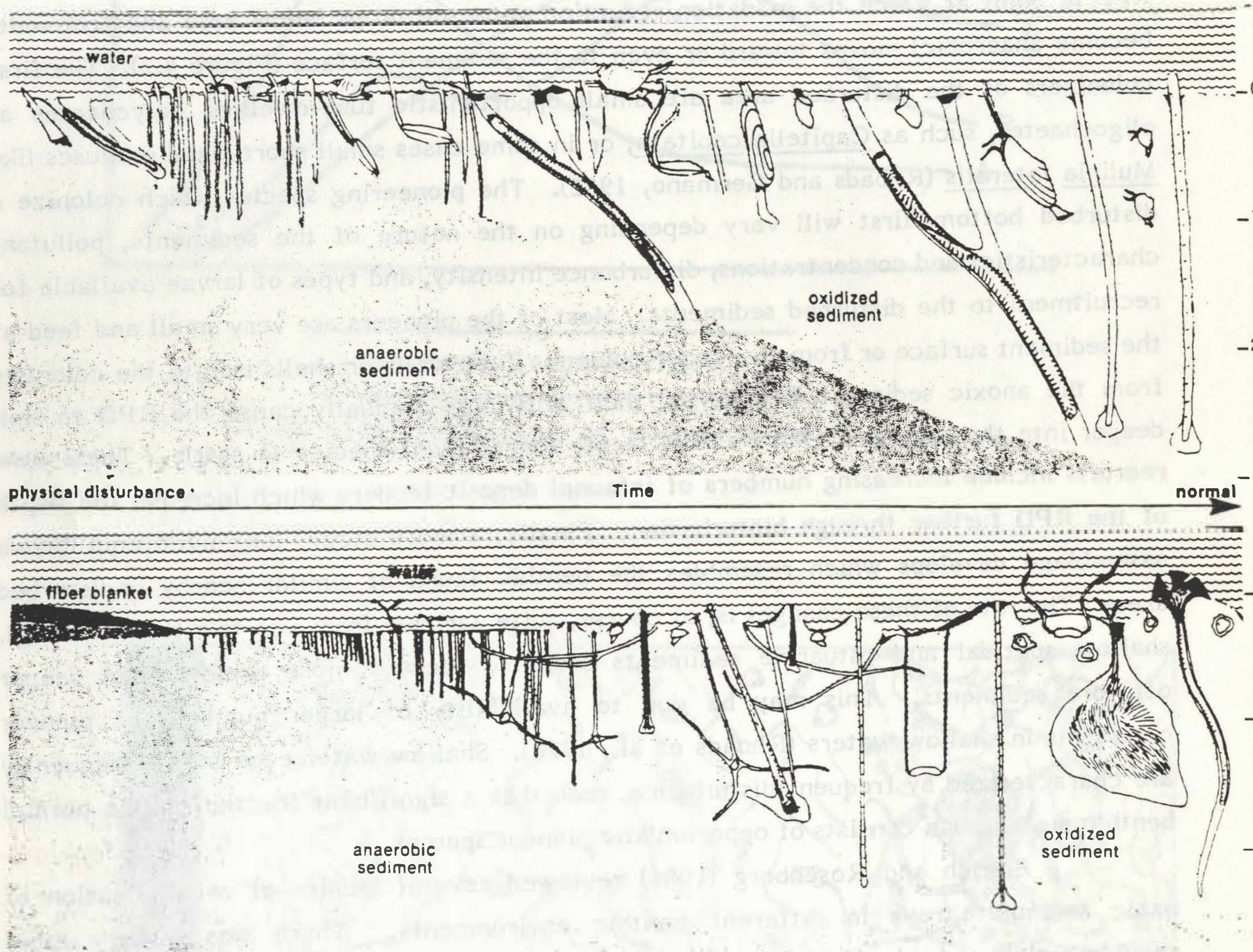

Figure 4.2 Schematic depiction of changes in benthic infaunal community structure over time following a single major disturbance and over distance from a chronic point source of organic pollution (from Rhodes et al, 1978).

1978). Peak density was not reached until 4-10 months following disturbance, while peak species richness was not reached for at least 12 months. Similarly, Grassle (1977) reported rapid increases in both density and species richness within two months in azoic sediment trays deployed on the bottom in 10 m of water in nearby Buzzards Bay, Massachusetts. Species richenss was 77 percent that in nearby undisturbed sediments after two months.

Azoic sediments placed on the bottom in 65m of water off New Jersey were colonized more slowly (Boesch and Rosenberg, 1981). After nine months, species richness in the sediment trays was just over half that in the natural community and density was less than one-fourth that of the natural community. In 1760 m of water off New England, species richness in sediment trays had only reached one-third and density only one-tenth that in the nearby natural communities after 26 months (Grassle, 1977). Based on these investigations, Boesch and Rosenberg (1981) concluded that macrobenthic communities in variable or frequently-disturbed environments are more resilient from severe disturbance than similar communities in more constant environments. The greater resilience of communities in frequently-disturbed environments is related to the fact that populations making up these communities include many species with high reproduction rate and short generation time (so called r-strategists). Such populations can recolonize disturbed habitats rapidly.

As stated above, when the disturbance is the introduction of a persistent pollutant into the ecosystem, recovery may be delayed until pollutant concentrations in the sediments drop to levels which can be tolerated by the colonizing macrofauna. In the case of biodegradable organic wastes (i.e. sewage, paper mill wastes), the major disturbance may be anoxia of bottom water and sediments due to the high biological and chemical oxygen demand (BOD and COD) of the wastes. In such cases, recovery may be delayed until sufficient organic material is oxidized, swept away or buried that the surficial sediments can be reoxygenated.

4.2.1.2 Case Studies.

A few specific examples will be given to show the types of recovery observed in benthic ecosystems following pollution abatement or acute catastrophic spills. The Saltkallefjord is a small arm of the Gullmarsfjord, an estuary on the west coast of Sweden near the junction of the Skagerack and Kattegatt. For nearly 80 years, the Saltkallefjord received wastewater from a sulfite pulp and paper mill at Munkedal, 5 km north of the fjord on the River Orekilsalv (Leppakoski, 1975). Between 1955 and 1966, the waste load increased continuously to about 55,000 m^3/day. In June 1966, the manufacture of sulfite cellulose was stopped and the plant was modernized, resulting in a ten-fold reduction in the waste discharge. The benthic fauna of the fjord had been studied extensively in the 1920s and 30s (Molander, 1928; Lindrath, 1935), in 1964-67 (Bagge, 1969a,b; Leppakoski, 1968, 1975) and in the 1970s (Rosenberg, 1972, 1973, 1976), providing excellent documentation of the degradation and subsequent recovery of an estuarine benthic ecosystem.

Discharge of the wastewater to the river resulted in an increase in organic carbon concentration and bacterial activity, low and fluctuating oxygen concentrations, and a decrease in pH of bottom water and sediments in the fjord.

Between 1928 and 1965, the dominant Syndosmya alba and Amphiura filiformis - A. chiajei communities disappeared from the upper part of the fjord. In 1964-1966, sediments in a small region near the mouth of the River Orekilsalv were completely devoid of benthic macrofauna (Figure 4.3). The boundary region adjacent to the unpopulated region was dominated by the opportunist polychaetes Capitella capitata and Scolelepis fuliginosa. Half way down the fjord, there was a distinct maximum in density of benthic fauna. Dominant species included the echinoderm Amphiura filiformis and the molluscs Montacuta ferruginosa and Thyasira flexuosa. Both biomass and abundance of benthic infauna decreased toward the mouth of the fjord.

Following cessation of wastewater discharges from the sulfite pulp and paper mill in 1966, the benthic fauna of the Saltkallefjord began a slow recovery, first at locations in the fjord farthest from the river inflow and then at locations near the river mouth (Figure 4.3). At Stations 11 and 12, one to two km from the river mouth, species recruitment occurred in 1968 (Leppakoski, 1975; Rosenberg, 1976). In 1969, three years after cessation of waste discharges, recovery also began at Stations 6 and 9 close to the outflow. Succession was intense in the area of the four stations undergoing recovery, and species abundance had reached the range encountered in the same area before pollution (1923-32) and at the relatively unpolluted Station 18.

The successive changes during recovery in the benthic faunal density were quite variable from station to station (Figure 4.4). The peaks in 1965, 1967, and 1969 were due to the great abundance of the polychaete Capitella capitata. The peaks in 1972 were due to the great abundance of Mysella bidentata and Scalibregma inflatum at the two inner stations and the polychaete Myriochele ocutata at the two mid-fjord stations.

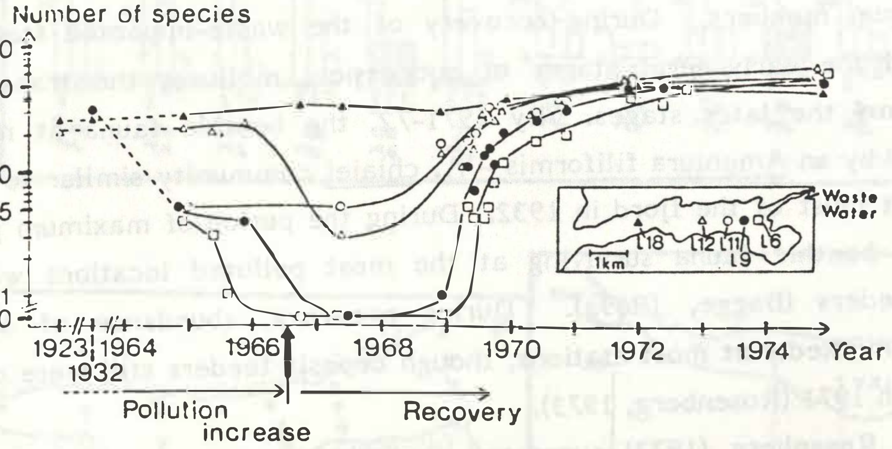

Figure 4.3 The succession of benthic infaunal populations of five stations in the Saltkallefjord Sweden. The arrow indicates the time when wastewater discharges from the sulfite pulp mill ceased (from Rosenberg, 1976).

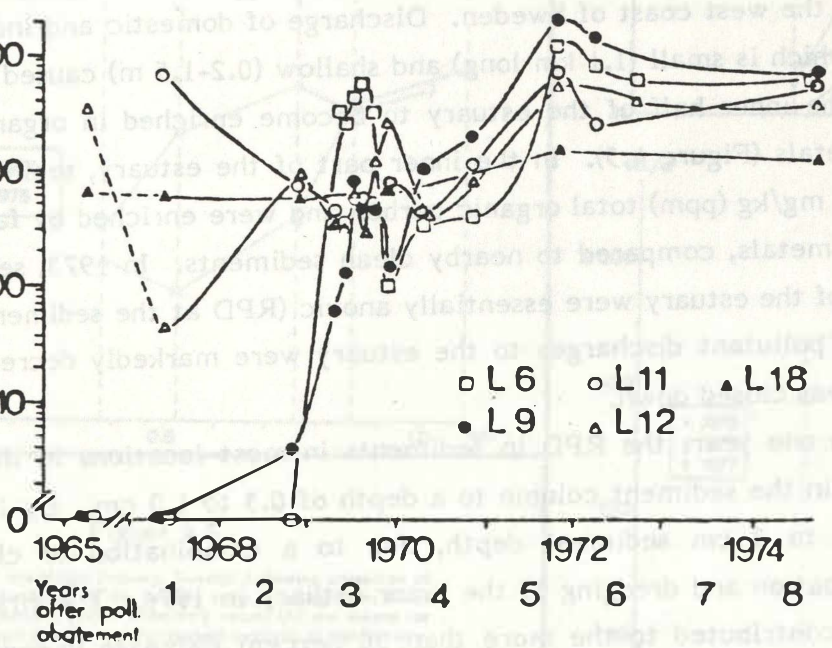

Figure 4.4 Dynamics of benthic infaunal abundance in the Saltkallefjord, Sweden following cessation of pulp mill wastewater discharge in 1966. Station locations are those indicated in Figure 4.3 (from Rosenberg, 1976).

The dominant taxonomic groups in the benthos of the Saltkallefjord were polychaetes, molluscs and echinoderms. At the reference Station 18, they were present in nearly equal numbers. During recovery of the waste-impacted stations, polychaetes dominated the early seral stages of succession, molluscs the transitory stages, and echinoderms the later stages. By 1971-72, the benthic fauna at most stations was dominated by an Amphiura filiformis - A. chiajei community similar to that encountered throughout most of the fjord in 1932. During the period of maximum pollution in 1965-1966, the benthic fauna surviving at the most polluted locations was dominated by deposit-feeders (Bagge, 1969a). During recovery, abundance of suspension-feeders increased markedly at most stations, though deposit-feeders still were dominant at most locations in 1971 (Rosenberg, 1973).

Rosenberg (1973) suggested that the more-or-less terminal seral stage of succession in the benthic community in the Saltkallefjord was approached five years after pollution abatement. However, less dramatic successional changes continued beyond five years, such that after eight years it was not possible to distinguish between a normal community and the one recovering from organic pollution (Rosenberg, 1976).

In another investigation, Cato et al. (1980) have investigated the recovery of the benthic ecosystem of the Valen estuary located a short distance south of the Saltkallefjord on the west coast of Sweden. Discharge of domestic and industrial sewage to the estuary, which is small (1.1 km long) and shallow (0.2-1.5 m) caused the sediments, particularly of the inner half of the estuary to become enriched in organic matter and several heavy metals (Figure 4.5). In the inner part of the estuary, sediments contained more than 80,000 mg/kg (ppm) total organic carbon and were enriched by factors of 13-50 in several heavy metals, compared to nearby clean sediments. In 1973, sediments in the inner two-thirds of the estuary were essentially anoxic (RPD at the sediment surface). In December, 1974, pollutant discharges to the estuary were markedly decreased when the treatment plant was closed down.

Within one year, the RPD in sediments in most locations in the estuary had moved downward in the sediment column to a depth of 0.5 to 1.0 cm. By 1978, the RPD had moved to 1 to 3 cm sediment depth, due to a combination of clean sediment deposition, bioturbation and dredging of the inner estuary in 1976. The first two factors also undoubtedly contributed to the more than 50 percent decrease in sediment organic carbon concentration between 1973 and 1975.

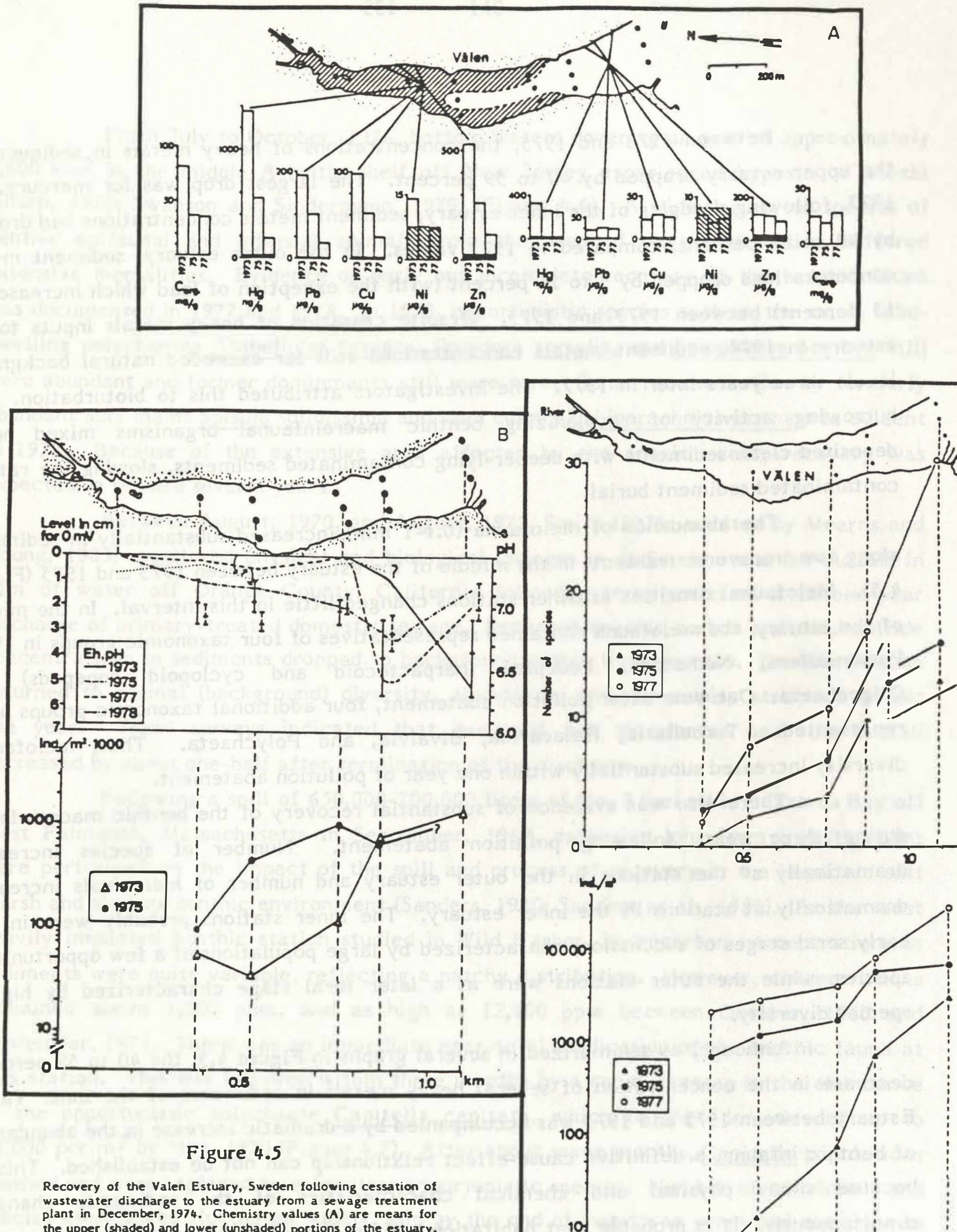

Figure 4.5

Recovery of the Valen Estuary, Sweden following cessation of wastewater discharge to the estuary from a sewage treatment plant in December, 1974. Chemistry values (A) are means for the upper (shaded) and lower (unshaded) portions of the estuary. The pH, Eh (B) and biological parameters and means for the five or six stations or transects indicated by solid circles. Lower left graph (B) is numbers of meiofaunal individuals per m^2; lower macrofauna per m^2 (from Cato et al, 1980).

Between 1973 and 1975, the concentrations of heavy metals in sediments of the upper estuary dropped by 40 to 59 percent. The largest drop was for mercury. By 1977, following dredging of the inner estuary, sediment metals concentrations had dropped by 49 to 86 percent, compared to 1973 values. In the outer estuary, sediment metals concentrations dropped by 3 to 26 percent (with the exception of lead which increased by 13 percent) between 1973 and 1975. Despite cessation of heavy metals inputs to the estuary in 1974, sediment metals concentrations still far exceeded natural background levels three years later in 1977. The investigators attributed this to bioturbation. The burrowing activity of recolonizing benthic macroinfaunal organisms mixed newly deposited clean sediments with deeper-lying contaminated sediments, slowing the rate of contaminated sediment burial.

The abundance of meiofauna (0.1-1 mm) increased substantially in sediments along two transverse transects in the middle of the estuary between 1973 and 1975 (Figure 4.5). Meiofaunal densities at other stations changed little in this interval. In the middle of the estuary, the meiofauna contained representatives of four taxonomic groups in 1973: Foraminifera, Nematoda, Podoplea (harpactecoid and cyclopoid copepods) and Oligochaeta. One year after pollution abatement, four additional taxonomic groups were represented: Turbellaria, Halacarida, Bivalvia, and Polychaeta. Thus, meiofaunal diversity increased substantially within one year of pollution abatement.

There also was evidence of substantial recovery of the benthic macroinfauna during three years following pollution abatement. Number of species increased dramatically at the stations in the outer estuary and number of individuals increased dramatically at stations in the inner estuary. The inner stations probably were in the early seral stages of succession, characterized by large populations of a few opportunistic species, while the outer stations were at a later seral stage characterized by higher species diversity.

Although, as summarized in several graphs in Figure 4.5, the 40 to 59 percent decrease in the concentration of several heavy metals in sediments of the inner Valen Estuary between 1973 and 1975 was accompanied by a dramatic increase in the abundance of benthic infauna, a definitive cause-effect relationship can not be established. This is because many physical and chemical characteristics of the sediments changed simultaneously. It is probable that habitation and colonization of sediments in the inner estuary in 1973 was prevented more by sediment anoxia than heavy metal contamination (Pearson and Stanley, 1979).

From July to October, 1976, bottom waters covering an area of approximately 8,600 km^2 in the middle Atlantic shelf off New Jersey experienced hypoxia or anoxia (Sharp, 1976; Swanson and Sindermann, 1979) (Figure 4.6). A wide variety of species of benthic epifaunal and infaunal animals from the area of oxygen depletion suffered extensive mortalities. Evidence of early but incomplete recovery of the benthic fauna was documented in 1977 and 1978. In 1978, opportunistic species such as the small tube-dwelling polychaetes Asabellides oculata, Polydora socialis, and Spiophanes bombyx still were abundant and former dominmants still were rare. Some recolonization by formerly abundant surf clams Spisula solidissima and sand dollars Echinarachnius parma was evident in 1978. Because of the extensive area affected by the anoxic event, recovery was expected to require several years.

Between August, 1970, and August, 1972, Smith (1974: as cited by Mearns and Young, 1983) monitored physical and biological changes in sediments near an outfall in 21m of water off Orange County, California following termination of a fifteen-year discharge of primary-treated domestic sewage. Sediment organic carbon and total sulfide concentrations in sediments dropped to background within three months. Benthic infauna returned to normal (background) diversity, abundance, and community structure within one year. Trawl surveys indicated that demersal fish abundances near the outfall decreased by about one-half after termination of the discharge.

Following a spill of 650,000-700,000 liters of No. 2 fuel oil in Buzzards Bay off West Falmouth, Massachusetts in September, 1969, extensive long-term investigations were performed on the impact of the spill and process of recovery in the impacted salt marsh and shallow benthic environment (Sanders, 1980; Sanders et al., 1980). At the most heavily impacted benthic station studied in Wild Harbor, hydrocarbon concentrations in sediments were quite variable, reflecting a patchy distribution. However, concentrations remained above 1,000 ppm, and as high as 12,400 ppm between October, 1969 and September, 1971. There was an immediate near-total eradication of the benthic fauna at this station. This was followed within three months by a rapid increase in the abundance of the opportunistic polychaete Capitella capitata, which reached densities of up to 95,000 per m^2 by May, 1970 (Figure 4.7). After about seven months, Capitella populations crashed and were replaced by a few less opportunistic species. Number of individuals and species abundances fluctuated widely, and by the end of two-three years had not risen to levels encountered there at the time of the spill.

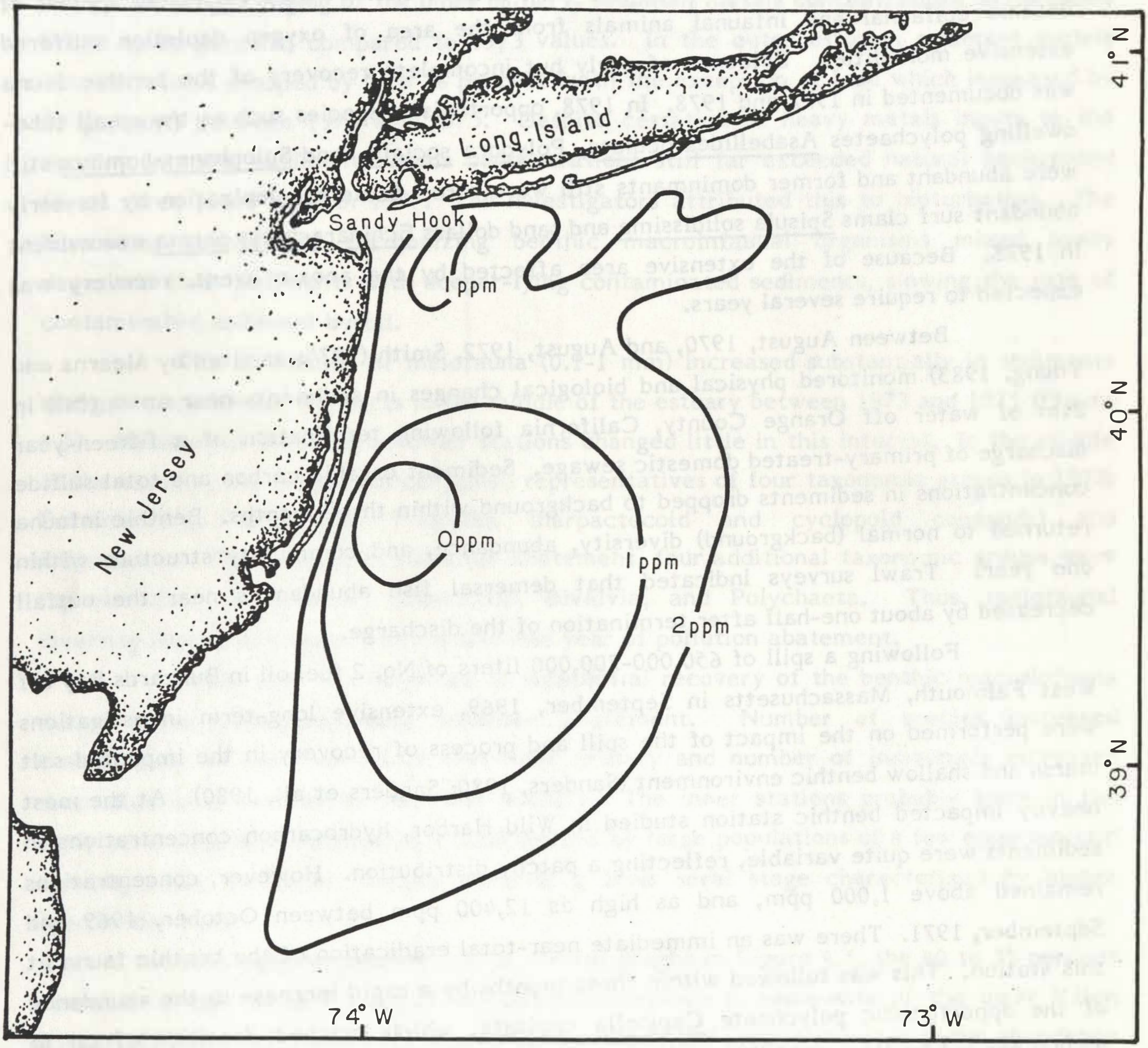

Figure 4.6 Area on the middle Atlantic Shelf affected by anoxic bottom water during the summer of 1976. Values are concentrations of oxygen in mg O_2/liter (ppm) (from Sharp, 1976).

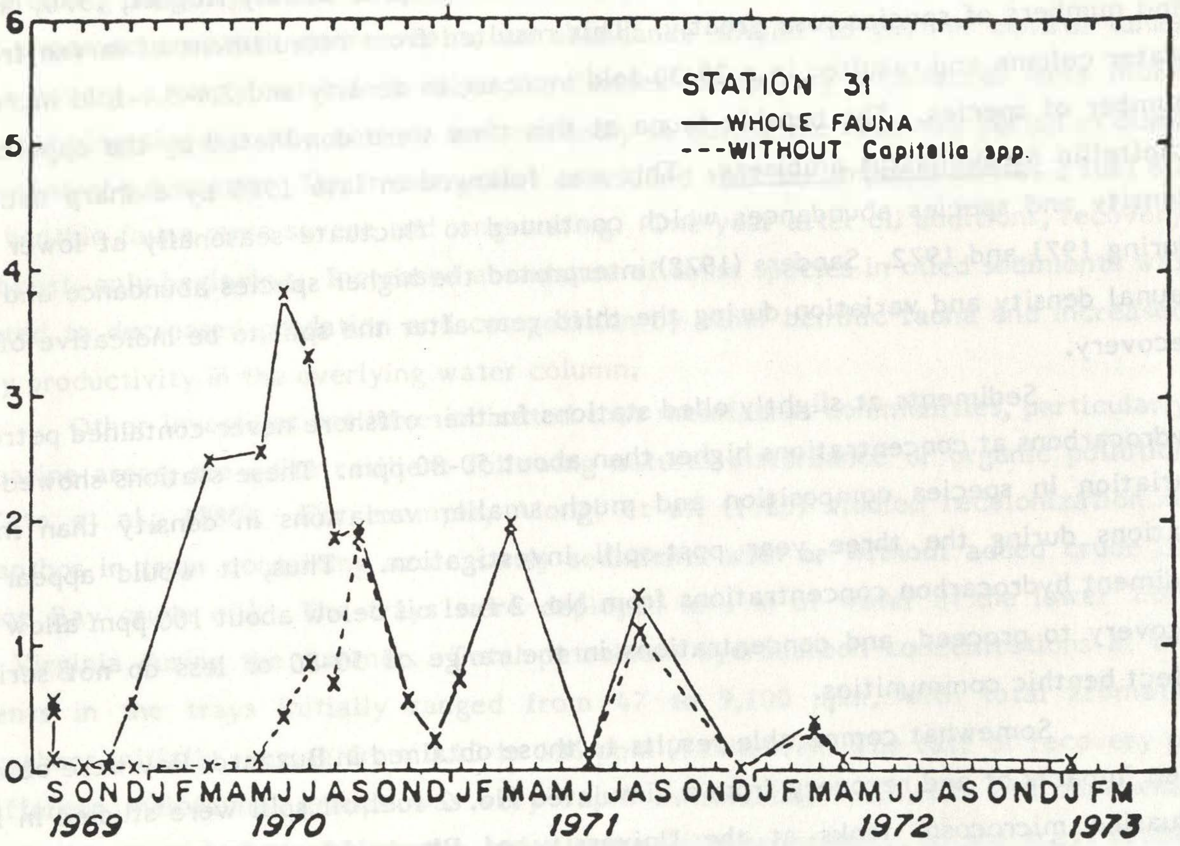

Figure 4.7 Density of benthic infauna with and without Capitella at a heavily oiled shallow water station in Wild Harbor, Massachusetts, following a spill of No. 2 fuel oil in September 1969 (from Sanders et al, 1980).

At stations just offshore from Wild Harbor, petroleum hydrocarbon contamination of sediments following the spill was much less severe than in the harbor. At Stations 9 and 10, sediment total petroleum hydrocarbon concentrations were in the range of 150-250 ppm in October-November, 1969, and had dropped to the range of 46-88 ppm by mid-late 1971. At these stations, number of species and faunal densities dropped to low levels during the seven-eight months after the spill. In July-August, 1970, densities and numbers of species rose greatly. This resulted from recruitment of larvae from the water column and resulted in a 20-30 fold increase in density and 2.4-3.4-fold increase in number of species. The benthic fauna at this time were dominated by the opportunistic capitellid *Mediomastus ambiseta.* This was followed in late 1970 by a sharp decline in density and species abundances which continued to fluctuate seasonally at lower levels during 1971 and 1972. Sanders (1978) interpreted the higher species abundance and lower faunal density and variation during the third year after the spill to be indicative of some recovery.

Sediments at slightly oiled stations further offshore never contained petroleum hydrocarbons at concentrations higher than about 50-80 ppm. These stations showed little variation in species composition and much smaller variations in density than inshore stations during the three year post-spill investigation. Thus, it would appear that sediment hydrocarbon concentrations from No. 2 fuel oil below about 100 ppm allow early recovery to proceed, and concentrations in the range of 50-80 or less do not seriously affect benthic communities.

Somewhat comparable results to those obtained in Buzzards Bay were obtained when impacts of and recovery from a simulated No. 2 fuel oil spill were studied in large estuarine microcosm tanks at the University of Rhode Island (Oviatt et al., 1982). Sufficient No. 2 fuel oil was added to the tanks twice weekly for four months to produce a hydrocarbon concentration in the water column of 90 ppb. After termination of hydrocarbon inputs, recovery of the microcosm communities was followed for one year. At the end of the oil dosing period, surficial sediments in the oiled microcosms contained a maximum of 806 ppm saturated and 109 ppm aromatic hydrocarbons. After one year, hydrocarbon concentrations in sediments had dropped to 100-200 ppm and 40-80 ppm aliphatic and aromatic hydrocarbons, respectively.

During the oil exposure period, and extending into the recovery period, the abundance of all but one species of benthic macrofauna, the polychaete *Chaetozone* sp., decreased. There was no change in species diversity, although one species, the amphipod

Ampelisca abdita disappeared completely during the oil exposure period and was only minimally present during recovery.

The benthic meiofauna also were severely affected by the No. 2 fuel oil exposure. The ostracods were the most severely affected meiofaunal group. Their numbers were severely depressed during the exposure period and showed only a small increase during recovery. Foraminifera and benthic ciliates increased in abundance during the oil exposure and then decreased to an abundance similar to that in control tanks during recovery. Newly settled juvenile polychaetes (temporary meiofauna) were more abundant during the exposure period, and markedly so during the recovery period in oiled than in control substrates. The investigators concluded that the impacts of No. 2 fuel oil on the benthic fauna were severe and long-lasting. One year after oil additions, recovery was, at best, only beginning. Increased abundance of some species in oiled sediments was attributed to decreased predation and competition by other benthic fauna and increased primary productivity in the overlying water column.

Other investigations have indicated that meiofaunal communities, particularly of estuarine areas, are quite resilient following natural disturbance or organic pollution (i.e. Cato et al., 1980). For example, Alongi et al. (1983) studied recolonization of meiobenthos in trays containing azoic sandy sediments with or without added crude oil (Prudhoe Bay crude oil). The trays were deployed in 1 m of water in the lower York River, Virginia during the summer. Total petroleum hydrocarbon concentrations in the sediments in the trays initially ranged from 47 to 9,100 ppm, with total aromatic hydrocarbons initially ranging from 0.43 to 136 ppm (Table 4.1). The rate of recovery of the different meiofaunal groups was very rapid in surficial sediments and somewhat slower in deeper sediments, below the redox potential discontinuity, in the trays (Table 4.1). Nematodes were the dominant meiofaunal taxon in all treatments, and also were the slowest to recover in the trays to abundance and community composition similar to that in study site control sediments. In fact, recovery of nematodes was not complete in 90 days in deep sediments in the middle and highest oiled trays. The results of this study indicate that recovery of benthic meiofauna was complete at 28 days at all depths in sediments containing 3,900 ppm total oil and 29 ppm aromatic hydrocarbons (mid oil, Table 4.1) initially and 1,285 ppm total oil and 15.75 ppm aromatic hydrocarbons at 28 days.

Similarly, Boucher (1980) reported little change in the abundance of meiofaunal nematodes and harpacticoid copepods in subtidal fine sand sediments in 19 m of water in the Bay of Morlaix, France, heavily impacted by the Amoco Cadiz oil spill (up

Table 4.1. **Initial recovery time (in days) of meiobenthos in the upper (above the redox potential discontinuity, usually 2 cm) and lower (below the redox potential discontinuity) portions of 5 cm sediment cores from oil-contaminated exposure trays in the York River estuary, Virginia (From Alongi et al., 1983).**

	Tray		Treatment					
	Control		Low Oil		Mid Oil		High Oil	
Taxon	Top	Bottom	Top	Bottom	Top	Bottom	Top	Bottom
Nematodes	7	25	6	26	11	a	8	a
Harpacticoid Copepods	5	1	4	3	5	5	4	1
Ostracods	3	2	3	3	4	5	3	2
Polychaetes	3	3	3	6	3	3	3	4
Others[b]	5	3	2	1	2	1	7	1
Total Fauna	5	25	5	26	8	26	16	a

a, Recovery not reached in 90 d.

b, Includes Foraminifera, Bivalvia, Oligochaeta, Ciliata, Hydrozoa, Amphipoda, Isopoda, Archiannelida, Turbellaria, Kinorhyncha, Cumacea, Tardigrada, and Gastrotricha.

to 3,800 ppm saturated petroleum hydrocarbons in sediments). However, there was evidence of a decrease in nematode diversity in the oiled sediment, due to an increase in the abundance of a few dominant species following the spill.

Two small estuaries on the north coast of Brittany, France, Aber Benoit and Aber Wrac'h, became heavily contaminated with Amoco Cadiz oil following the spill. Glemarec and Hussenot (1982a,b) investigated the recovery of the benthic macrofauna of the two estiaries following the spill. The estuaries contain a wide variety of different soft substrates, from fine intertidal muds to coarse sand flats, so it was possible to study the course of recovery of different types of sediment regimes.

During the three months after the spill, oil spread throughout the estuaries, and sediment petroleum hydrocarbon concentrations in most areas were in the 1,000-10,000 ppm range (measured by infrared analysis: Marchand et al., 1982). The exceptions were the coarse sandy areas at the mouths of the estuaries. There were immediate massive kills of the benthic fauna of the estuaries after the spill, and nearly all communities still were extremely depauperate eight months later (Figure 4.8). The coarse sand communities at the mouth of Aber Wrac'h were little affected by the spill and showed near normal seasonal fluctuations. The highest oil concentration measured in these coarse sand sediments was 1020 ppm in May, 1978, three months after the spill. In the subtidal dune-sand sediments off Aber Benoit, petroleum hydrocarbon concentrations never exceeded 50 ppm, and there was some evidence of only a temporary decrease in total abundance with no significant change in benthic community structure.

Muddy sediments at the heads of the estuaries were the most heavily oiled and most severely damaged. After eight months, highly opportunistic species, characteristic of early stages of recovery from organic pollution, began to appear and reached very high densities. Peak abundance of opportunists was not reached until 20-25 months after the spill. At most other locations in the estuaries, characterized by fine sands and muddy sands, dominant benthic fauna eight months post-spill included the less extreme opportunist species, primarily cirratulid and capitellid polychaetes (e.g., Chaetozone setosa, Heterocirrus spp., Polydora spp., Cirratulus cirratulus, etc.). After about 17 months, these opportunists were joined by a group of sensitive species which typically disappear following a perturbation and then reappear over a broader ecological range (e.g., Spio sp., Notomastus laetericeus, Phyllodoce spp., Nereis diversicolor, etc.).

Winter storms resulted in sediment resuspension and a decrease in some cases in sediment hydrocarbon concentrations. These were followed, especially in the second

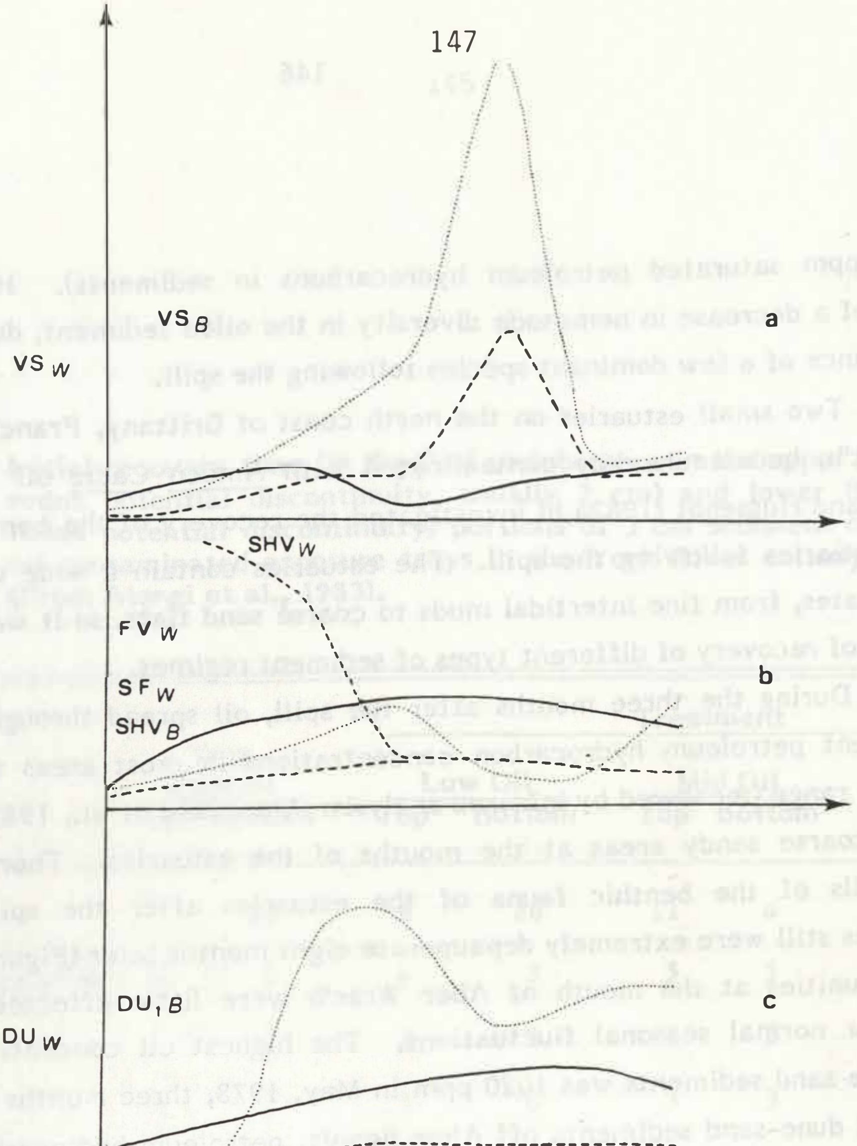

Figure 4.8 Changes in community parameters of benthic infaunal communities occupying several types of substrates in Aber Benoit and Aber Wrac'h, France following heavy oiling by the Amoco Cadiz oil spill. Solid line, number of species; dotted line, abundance of individuals; dashed line, biomass. $\mathrm{Du_B}$, $\mathrm{Du_W}$, medicine and fine sands in Aber Benoit and Aber Wrac'h, respectively; $\mathrm{SF_{B1}}$, $\mathrm{SF_W}$, fine sands; $\mathrm{FV_B}$, $\mathrm{FV_W}$, fine and silty sand; $\mathrm{SHV_B}$, $\mathrm{SHV_W}$, heterogeneous silty sand; $\mathrm{VS_B}$, $\mathrm{VS_W}$, sandy mud (from Glenmarec and Hussenot, 1982).

winter after the spill, by a decrease in opportunists and an increase in later successional stages. In Aber Benoit, the decline in opportunistic species, and replacement by later successional stages began after eight months in well-oxygenated dune sands, after 13 months in heterogeneous muddy sands, after 17 months in fine sands and sandy muds and after 25 months in the muddy sand areas at the head of the estuary. Two to two and a half years after the oil spill, the communities of the dune sand areas had recovered completely, while earlier successional stages still were evident in all other substrates. Communities in Aber Wrac'h recovered more slowly than communities in similar substrates in Aber Benoit. Glemarec and Hussenot (1982a,b) estimated that it would require three to four years for benthic communities of the Abers to return to normal or to a new equilibrium condition.

Cabioch et al. (1982) and Dauvin (1982) reported similar changes following the Amoco Cadiz oil spill in benthic communities occupying different sediment types in deeper coastal waters of the Bay of Morlaix. Coarse sandy sediments were little affected by the spill and their communities recovered rapidly. The fauna of fine muddy sand, particularly several species of amphipods of the genus Ampelisca, were severely impacted by the hydrocarbons and were very slow to recover. Significant recolonization of severely impacted sediments did not begin until the second annual cycle of recruitment (spring, 1980). There was further substantial recruitment in the third spring following the spill. The dominant fauna of the heavily impacted offshore fine sand sediments during the first year after the spill were opportunistic species of capitellid and cirratulid polychaetes. Approximately three years after the spill, the communities of the fine sandy sediments in the Bay of Morlaix had returned to the normal pre-spill range in terms of density, species richness, and biomass. Abundance of amphipods still was depressed in some areas.

Following the grounding of the Soviet tanker Tsesis on the Swedish coast of the brackish Baltic Sea south of Stockholm in October 1977, approximately 1,000 tons of medium grade (No. 5) fuel oil was released (Elmgren et al., 1983). Although about two-thirds of the oil was recovered mechanically, significant amounts of oil reached the rocky shores and the benthic environment.

Within 16 days after the spill, an estimated 95 percent of the dominant amphipods, Pontoporeia affinis and P. femorata and polychaetes, Harmothoe sarsi were eliminated from the benthos at the most severely impacted station. The deposit feeding clam Macoma balthica was resistant to the pollution but became heavily contaminated with oil (up to 2,000 ppm dry wt.). All meiofaunal groups except nematodes were reduced in abundance in the affected area.

In the summer following the spill (1978), there was a very heavy recruitment of juvenile Macoma balthica in areas where amphipods had been virtually eliminated. Not until the summer of 1979 did amphipods, Harmothoe sarsi, and harpacticoid copepods begin to return to sediments of the most heavily impacted stations, and oil contamination of M. balthica decrease to 1,000 ppm. By the third year after the spill, abundance of amphipods still was depressed but H. sarsi had returned to its pre-spill abundance. Oil concentrations in M. balthica had dropped to about 250 ppm. Because of the long life span of M. balthica, the authors predicted that the disturbed community structure at the heavily oiled stations would persist for several years. Full recovery could take five to ten years.

Leppakoski and Lindstrom (1978) studied recovery of benthic macrofauna in a small low salinity (5.4-6.7 °/oo salinity) harbor on the Finish coast of the Baltic Sea receiving effluents from an oil refinery, following nearly complete abatement of the oily waste discharges. The toxicity of sediments to benthic insect (Chironomus plumosus) larvae decreased markedly in the first three years following cessation of oily wastewater discharges. Number of species and individuals, and species diversity of benthic macrofauna increased sharply at stations near the former outfalls. However, three years after abatement, there was a trend toward reduced diversity, characterized by a marked increase in the dominance of the bivalve Macoma balthica, as occurred following the Tsesis oil spill. Thus, substantial but not complete recovery of the benthic community had occurred three years after near complete cessation of oily wastewater discharges to the harbor.

4.2.1.3 Summary.

Virtually all the investigations to date of the recovery of benthic communities from acute or chronic pollutant stress have been performed in estuarine and nearshore coastal waters. Under these conditions, the responses to and recovery from organic (sewage and pulp mill wastes) and petroleum hydrocarbon pollution have been remarkably similar. Where heavy metals were present in addition to organic pollutants, it was not possible to determine if the metals contributed to the magnitude of the benthic impact or affected the rate of recovery of the benthic ecosystem.

Factors which did seem to affect the rate of recovery include: water depth, to the extent that it influenced the amount of storm-induced sediment resuspension;

magnitude of pollutant accumulation in the sediments; sediment grain size (recovery was more rapid in coarse than in fine sediments); and availability of planktonic larvae for recruitment.

Fuel oils (No. 2 and No. 5 fuel oils) seem to produce more severe impacts and result in slower recovery than crude oil. This undoubtedly is due to the higher concentration in fuel oils than in most crude oils of light aromatic hydrocarbons, particularly the toxic naphthalenes and phenanthrenes (Neff and Anderson, 1981). Unfortunately, there are no published studies of the recovery of marine benthic communities from persistent non-polar organic pollutants such as polychlorinated biphenyls and chlorinated pesticides. This could be due to the lack of such studies or the lack of recovery of benthic ecosystems impacted by these pollutants.

Generally, the estuarine and coastal marine benthic ecosystems studied to date recovered rapidly from organic and petroleum pollution. Recovery times of a month (meiofauna) to 5-10 years have been suggested. This rapid recovery lends support to the conceptual model of Boesch and Rosenberg (1981) that estuarine communities are more resistant to stress than strictly marine communities because they are composed of more eurytolerant species. In addition, such communities are more resilient than oceanic communities because they include more opportunistic species acclimated to frequent disturbance.

4.2.2 Deep-Sea Benthic Environment

4.2.2.1 Background.

The deep-sea benthic environment is characterized by a very stable physical chemical regime: low temperatures, high salinity, and high dissolved oxygen concentrations (Sanders et al., 1965). Only within the past twenty years has it become apparent that the deep-sea benthic fauna has a high diversity, in many areas nearly as high as the diversity of shallow tropical seas and coral reefs (Sanders, 1968; Jumars, 1976). However, despite the uniform physical environment, there is evidence of substantial zoogeographic variation and small scale patchiness in benthic community structure (Jumars, 1976; Rex, 1983). Species richness and diversity tend to increase with depth on the continental slope and rise but then decrease markedly at abyssal depths greater than 4000 m (Hessler and Sanders, 1967; Sanders and Hessler, 1969; Jumars and Hessler, 1976; Rex, 1981).

One of the most important factors regulating community structure in the deep-sea benthos is the slow rain of organic nutrient particles, including plankton and fecal pellets from above, which may be an important nutrition source for the benthos (Wiebe et al., 1976).

With the exception of the benthic epifauna associated with deep-sea hydrothermal vents (Turekian and Cochran, 1981; Rhoads et al., 1981), and wood boring bivalve molluscs associted with pieces of wood on the bottom (Turner, 1973), life processes of the deep sea benthos appear to proceed at much slower rates than in shallow depths (Sanders, 1980). In situ respiration of benthic infauna (Smith and Teal, 1973) and rate of microbial turnover of organic matter (Jannasch, 1979) appear to be quite slow. Deep-sea bivalve molluscs grow very slowly, but live very long. The bivalve Tindaria callistiformis from 3500 m may live 150 years and in that time reach a length of only 6-8 mm (Turekian et al., 1975; Sanders and Allen, 1977). The species does not show evidence of gonadal development until 30-40 years old and may reach sexual maturity only once at 100+ years of age. Many other deep-sea bivalve species mature late in life and may spawn only once.

4.2.2.2 Case Studies.

Because the deep-sea benthos is highly diverse with a slow biomass production and turnover rate, it is likely to have low resistance and resilience to disturbance of the benthic environment (Boesch and Rosenberg, 1981). Most deep-sea benthic organisms lie on or within the top centimeter of the sediments, and appear not to be morphologically adapted for strenuous burrowing (Ozturgut, et al., 1981). At the site of the EPA Deepwater Dumpsite 106 in approximately 2,000 m of water, Rowe and Gardner (1979) estimated the sedimentation rate to be only 0.01-0.1 mm/year. In an assessment of potential impacts of coal fly-ash disposal at the 106 mile dumpsite, ERCO (1981, 1982) suggested that the epibenthic fauna would be able to escape or avoid accumulations of ash on the bottom, but the infauna would not. These benthic infaunal animals, because of their limited ability to migrate vertically would be smothered by deposition of even a few millimeters of material. Jumars (1981) observed substantial mortality of numerous benthic species in a bathyal community at a water depth of 1,200 m after one day of burial by sediments deposited by a manganese nodule mining experiment. Because of the slow rate of recruitment of benthic fauna to disturbed substrates in the deep sea (Grassle,

1977) and the slow rate of growth and reproduction of some deep-sea benthic fauna, recolonization and recovery of this habitat may require several years, and re-establishment of the age-size structure of the normal community may take decades.

4.3 Shorelines and Hard Substrates

There is a wide variety of different types of intertidal shorelines and shallow water hard substrates which may be subjected to chonic or acute pollutant stress. These include rocky shores, sand beaches, intertidal mud flats, salt marshes, sea grass beds, coral reefs, and mangrove swamps. Gundlach and Hayes (1978) classified several of these coastal habitats with respect to vulnerability to oil spills. The classification included consideration of persistence of the oil and magnitude of initial biological impact to the habitat, but not ecological resilience. Their ranking from least to most vulnerable was: exposed rocky headlands and wave-cut platforms; coarse-grained sand and gravel beaches; sheltered environments such as sheltered rocky shores, mud flats, salt marshes, and mangrove swamps. The rate of recovery of these coastal marine ecosystems will depend not only on the magnitude of the acute or chronic impacts of the pollutant and the rate at which it is cleansed from the habitat, but also the rates of recolonization and succession in the affected ecosystem. There have been several studies of the recovery of coastal ecosystems following acute oil spills or following cessation of chronic pollutant impacts. These studies indicate that coastal ecosystems vary even more than subtidal benthic ecosystems in resistence and resilience to pollutant-mediated perturbation.

4.3.1 Intertidal Sediments

4.3.1.1 Case Studies.

Rosenberg (1971) studied the recovery of the littoral fauna of the Saltkallefjord on the west coast of Sweden following closure of the sulfite pulp mill in July, 1966. In the upper part of the fjord near the pollutant input source, the number of species of littoral fauna increased from 18 in August, 1967 to 30 a year later. In 1967, shoreline vegetation in the inner fjord was very sparse and hard substrates were covered by a film of slime. This film had pretty much disappeared by 1968-1969, and there was an increase in shore grasses Zostera and Ruppia and algae Enteromorpha and Fucus.

Populations of several species of intertidal infauna migrated slowly up the estuary following pollution abatement (Figure 4.9). It has required one to three years for the major species of littoral fauna to reinvade the fjord to the extent where further migration was limited by low salinities (Rosenberg, 1976). Complete recovery probably required somewhat longer.

On February 4, 1970, the tanker Arrow went aground in Chedabucto Bay, Nova Scotia and spilled approximately 10,000 tons of Bunker C residual oil. Much of the oil was washed up along 300 km of the rocky and sandy coast. After six years, intertidal sediments in the spill area still contained 10-25,000 ppm petroleum hydrocarbons (Gilfillan and Vandermeulen, 1978). Shannon-Wiener diversity and macrofaunal biomass both were lower in the oiled compared to nearby clean sediments. Six to seven years after the spill, soft-shell clams from the spill site still were stressed and had reduced growth rates. Fewer mature adults were found at oiled than at control sites.

Following spills of crude or refined petroleum at Picnic Bay, Hong Kong (Wormald, 1976) and LaCoruna, Spain (Giere, 1979), meiofaunal communities of sandy beaches where some of the oil came ashore were almost totally destroyed. Within one year to fourteen months, the meiofaunal communities of the oiled beaches had experienced substantial, near complete recovery. Nematodes reappeared first, followed by harpacticoid copepods and polychaete worms.

Vanderhorst et al. (1980) studied the effects of Prudhoe Bay crude oil on recruitment of benthic infauna to trays of azoic sediment placed at different levels of the intertidal zone in different seasons in the Straits of Juan deFuca, Washington. Oil at a concentration of 2,000 ppm was mixed with the sediments in the experimental trays. In 15 months, the sediments lost 85 to 97 percent of the hydrocarbons originally present, depending on sediment grain size.

There was essentially complete recovery of benthic infauna in control (no oil) sediment trays in 15 months, in terms of species abundance, diversity, and composition compared to nearby natural sediments. Recovery of clean azoic sediments during three summer months was 69 percent with respect to numbers of species and 11 percent with respect to numbers of individuals. Relative recovery during three fall months was 82 percent with respect to numbers of species and 18 percent with respect to numbers of individuals. After 15 months, the experimentally oiled sediments had recovered by more than 90 percent with respect to numbers of species, but only 48 percent with respect to numbers of individuals. Level in the intertidal zone also had an effect on the rate of

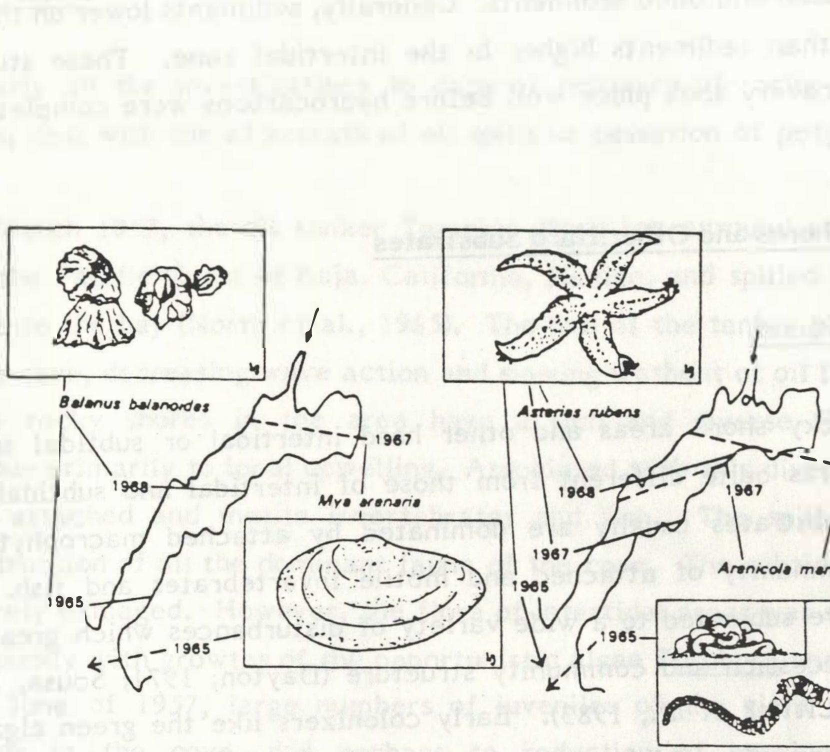

Figure 4.9 Movement of several species of intertidal benthic animals into the Saltkallifjord after closure of a sulfite paper mill in 1966. The wastes entered the fjord via the river indicated by the arrow (from Rosenberg, 1976, as redrawn by Cato et al, 1980).

recovery of clean and oiled sediments. Generally, sediments lower on the shore recovered more rapidly than sediments higher in the intertidal zone. These studies showed that substantial recovery took place well before hydrocarbons were completely released from the sediments.

4.3.2 Rocky Shores and Other Hard Substrates

4.3.2.1 Background.

Rocky shore areas and other hard intertidal or subtidal substrates possess faunas and floras quite different from those of intertidal and subtidal soft substrates. These hard substrates usually are dominated by attached macrophytic algae and an associated community of attached and motile invertebrates and fish. Exposed rocky shores often are subjected to a wide variety of disturbances which greatly influence the patterns of succession and community structure (Dayton, 1971; Sousa, 1979; Taylor and Littler, 1982; Littler et al., 1983). Early colonizers like the green algae Enteromorpha spp. and Ulva spp. may inhibit recruitment of long-lived perennial macroalgae such as the large red and brown algae. Grazing by various herbivores, such as limpets, sea urchins, and crabs may clear space for settlement of the perennial macroalgae and also for long-lived sessile invertebrates such as barnacles.

Acute or chronic pollution events, or subsequent cleanup attempts, may completely obliterate the rocky intertidal communities or return them to an earlier successional state, characterized by a low diversity of short-lived opportunistic species with rapid rates of biomass production and turnover (Littler and Murray, 1975; 1978). Following pollution abatement, recovery rate will depend on the rate of cleansing of the rocky shore of pollutants, and, more important, the availability from adjacent unimpacted subtrates or from the plankton of the right mix of species of fauna and flora to recolonize the available substrates.

On exposed rocky shores, removal of the pollutant chemicals usually is rapid, due to wave action, abrasion and photochemical ractions. In sheltered areas or boulder patches, pollutant chemicals may be quite persistent and recovery will be delayed.

4.3.2.2 Case Studies.

Nearly all the investigations to date of recovery of rocky shores and other hard substrates, deal with the aftermath of oil spills or cessation of petrochemical waste discharges.

In March 1957, the oil tanker Tampico Maru ran aground at the mouth of a small cove on the Pacific Coast of Baja, California, Mexico, and spilled part of its cargo of diesel fuel into the bay (North et al., 1965). The hull of the tanker blocked 3/4 of the entrance of the cove, decreasing wave action and slowing washout of oil from the cove.

The rocky shores in the area have a rich and diverse flora of attached macrophytes, due primarily to local upwelling. Associated with this diverse flora is a rich community of attached and motile invertebrates and fish. The spill caused a nearly immediate destruction of all the dominant fauna of the cove. The subtidal flora appeared not to be severely damaged. However, the flora of intertidal areas was severely damaged and replaced quickly with growths of the opportunistic algae Porphyra and Ulva.

By June of 1957, large numbers of juveniles of the giant kelp Macrocystis began appearing in the cove, due perhaps to reduction of grazing by herbivorous invertebrates or to increased protection of the cove provided by the wreck (Figure 4.10). The Macrocystis forest reached its maximum extent in 1961, more than four years after the spill, and subsequently declined, possibly due to return of grazing invertebrates. By 1963, the Macrocystis climax community had passed through at least two complete succession cycles, which in turn influenced development of associated flora and fauna.

Within three months after the spill, motile fish and invertebrates were beginning to return to the cove. Opaleye fish (Gerella sp.) and shore crabs (Pachygrapsus sp.) were observed in tide pools. After about five months, small barnacles Chthamalus fissus began reappearing on intertidal rocks. During the next six years, 45 additional species of animals re-established populations in the cove. Recruitment was small in the year after the spill, but increased sharply in the second year to the extent that the cove "looked" normal. However, further increases in species numbers and abundances of individuals occurred in 1960 and 1961. By 1964, conditions in the cove had not returned to those that existed there before the spill, due mainly to the unusually rich growth of Macrocystis. However, substantial recovery was realized within about six years of the spill.

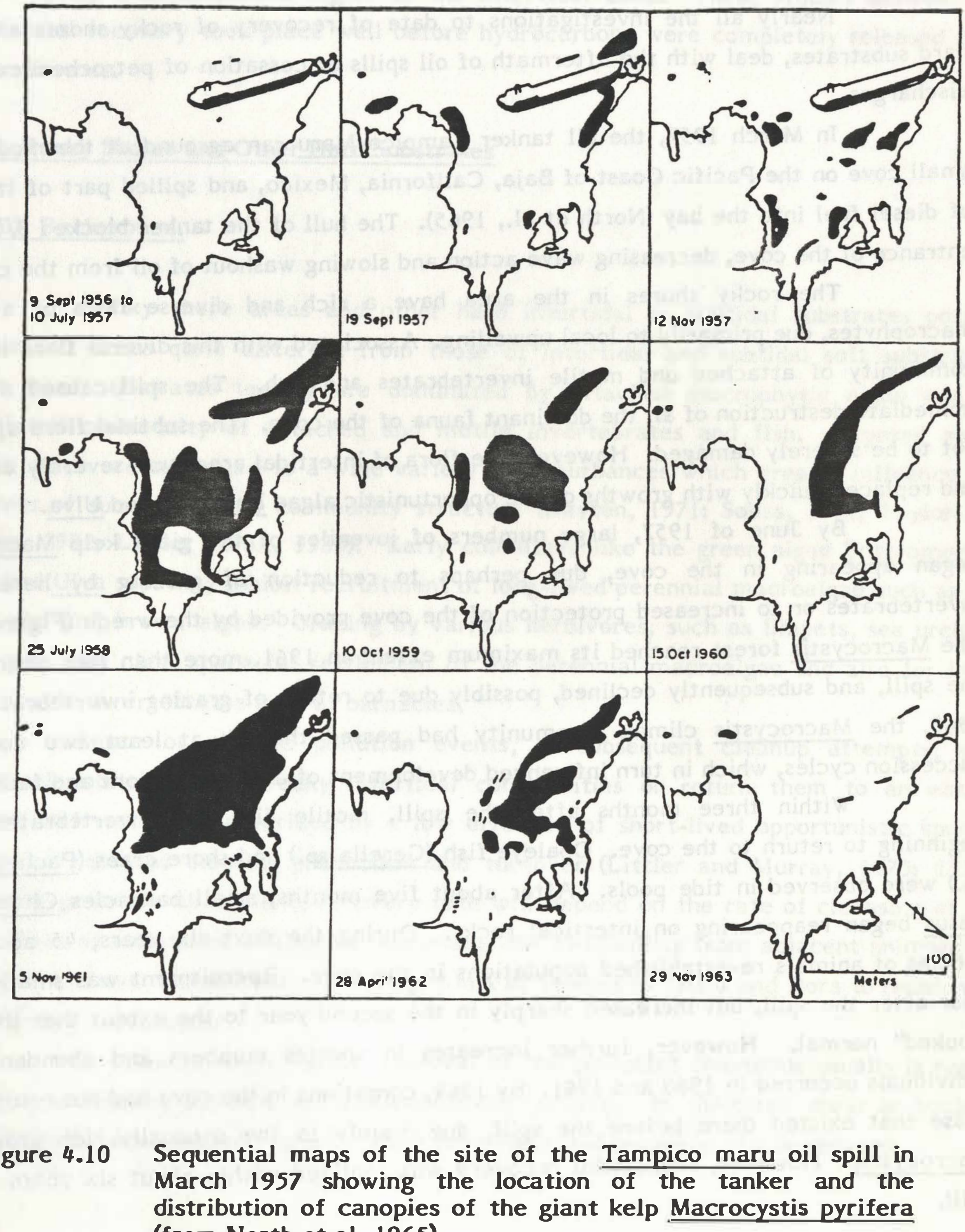

Figure 4.10 Sequential maps of the site of the Tampico maru oil spill in March 1957 showing the location of the tanker and the distribution of canopies of the giant kelp Macrocystis pyrifera (from North et al, 1965).

An important observation to come out of this study is that in highly structured, complex communities like those in this temperate marine cove, the course of succession following severe disturbance may lead to long-lived subclimax seral stages which have a different community structure than those in similar nearby undisturbed habitats. In such situations, it may not be appropriate to define recovery as a return to the same community structure as existed before disturbance.

Extensive forests of giant kelp, Macrocystis pyrifera, occur in nearshore coastal waters of southern California. The kelp, as well as associated red sea urchins Strongylocentrotus franciscanus, are harvested commercially. In 1928, kelp forests covered approximately 1,800 acres off the Palos Verdes Peninsula in Los Angeles County. Beginning in the mid 1940s, the kelp forests there began to deteriorate and in mid-1967 the last Macrocystis plants died. There was a good inverse correlation between the area of kelp canopy off the peninsula and the amount of waste solids discharged by the Los Angeles County Sanitation District outfall at White Point (Wilson, 1982) (Figure 4.11). The mechanism(s) by which the waste effluent caused the decline in the kelp forest is uncertain. Suggested causes include: siltation and burial of rocky surfaces by floc originating from both plankton, which increased due to waste-mediated eutrophication, and the suspended solids in the sewage itself; reduction in the depth of the euphotic zone by suspended particulates; toxic chemicals in the sewage; or overgrazing by sea urchins and other herbivores.

Since 1971, improvements in the Los Angeles County sewage treatment facilities have substantially reduced the discharge of suspended solids. In 1974, the kelp forests and associated biotic communities began to recover, and by 1982 approximately 700 acres of kelp forests had returned. Environmental changes contributing to the recovery of the kelp forest undoubtedly include improvements in water quality and clarity (i.e. decrease in turbidity) and reduction of burial of hard substrates necessary for settlement and growth of the kelp. In addition, commercial harvesting of red sea urchins has reduced the numbers of grazers allowing the kelp to re-establish itself. There also has been a slight but significant increase in local water temperature which could have contributed to the regrowth (Mearns, 1981). The erratic fluctuations in regrowth of the kelp undoubtedly are related to sporatic increases in populations of other grazers, including purple sea urchins Strongylocentrotus purpuratus and white sea urchins Lytechinus anomesus, which have denuded large areas of hard substrate otherwise suitable for settlement and growth of kelp.

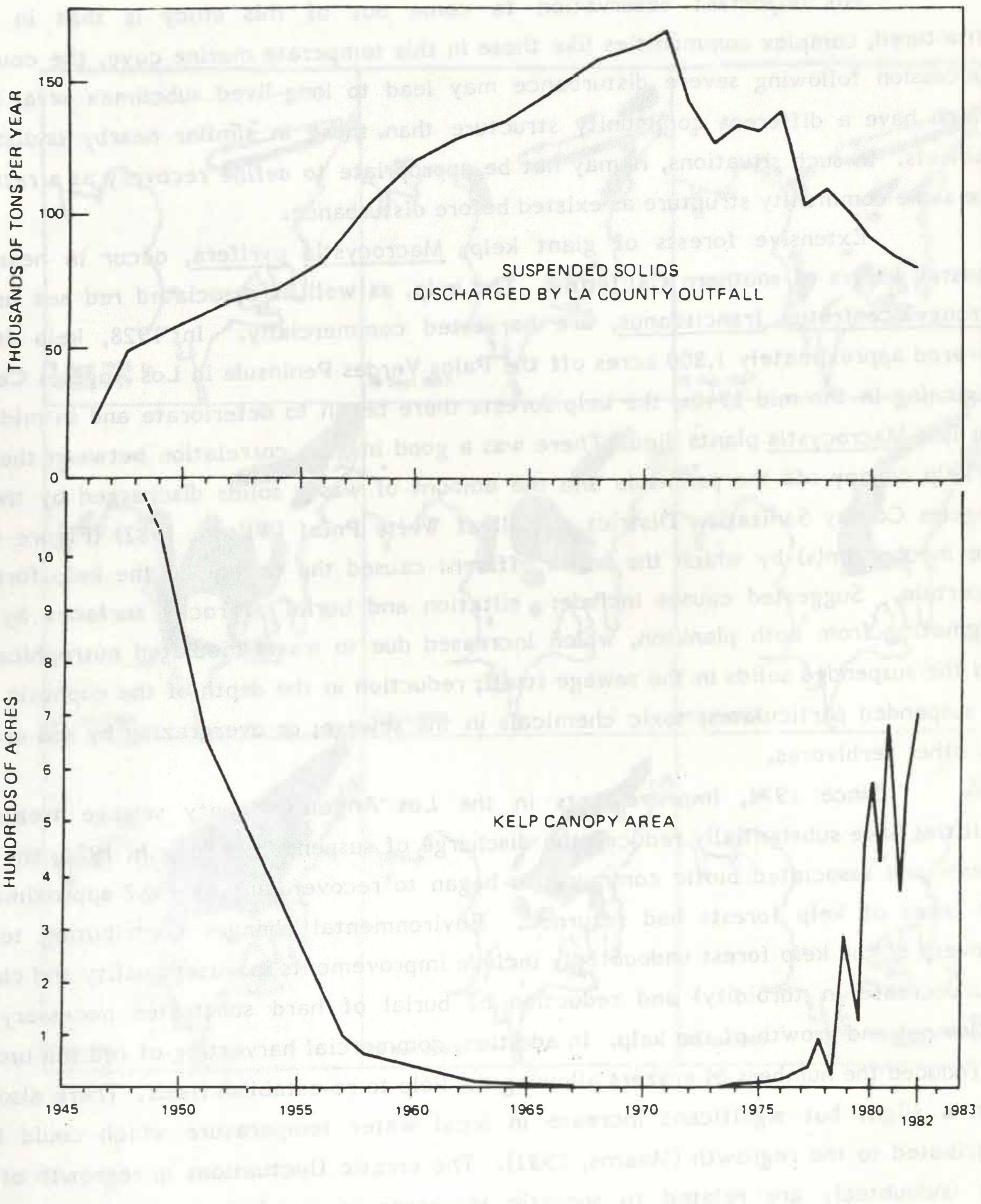

Figure 4.11 Comparison of changes in concentration of suspended solids in wastewater discharges from the Los Angeles County Sanitation District outfalls and extent of the coastal zone of the Palos Verdes, Peninsula, California occupied by kelp forest Macrocystis pyrifera (from Wilson, 1982).

Reish (1971) studied the recovery of the biofouling community on hard substrates in Los Angeles harbor following a state-mandated cessation of oil refinery waste-water discharges into the Dominguez Channel, which empties into the harbor. The volume and BOD/COD content of waste-waters were substantially decreased between February, 1968 and September, 1970. Before pollutant abatement, the only macroorganisms associated with the submerged dock surfaces included five species of polychaetes, dominated by Capitella capitata, blue-green algae, the green alga Enteromorpha crinita, chironomid larvae, and tubificid oligochaetes. Dissolved oxygen concentrations in the water at these locations generally was below 1.0 mg/l.

By October, 1970, shortly after cessation of the last major refinery wastewater inflows, dissolved oxygen concentrations had risen to the region of 3.8 to 4.2 mg/l and the blue-green algae-oligochaete and Enteromorpha-oligochaete assemblages had disappeared. At the most severely impacted stations, Enteromorpha was still the dominant macroalga, but at stations further from the pollutant source, Enteromorpha was replaced by Ulva lobata. In October, 1970, 34 months after the start and one month after the completion of the wastewater discharge abatement, the fouling community at stations in Dominguez Channel contained 13-18 species of macrobiota, while macrofouling communities in the harbor itself included 27-37 species. Reish (1971) speculated that Mytilus edulis would become the dominant biofouling organism within one year. Thus, initial phases of recovery of the biofouling community in Los Angeles harbor proceeded very rapidly after termination of the refinery discharge.

After a spill of 2,000-3,000 tons of gasoline in Lindsway Bay, Milford Haven, Great Britain from the tanker Dona Marika in 1973, there were massive mortalities of limpets Patella vulgata and barnacles along about 3 km of rocky shore (Dicks and Hartley, 1982). There was substantial recovery of these species after one year and virtually complete recovery of the rocky shore in six years.

In an earlier more massive spill nearby, 14,000 tons of Kuwait crude oil was stranded along 150 km of the coast of West Cornwall, England in March, 1967, following the grounding of the tanker Torrey Canyon on the Seven Stones Reef (Southward and Southward, 1978). During cleanup operations, 10,000 tons of toxic dispersants were used to clean the oil from the shore. The dispersants caused more damage than the oil itself. Most animals and some algae were killed on oiled rocky shores that were treated heavily with dispersants, whereas a few animals and most algae survived in places that were less heavily treated with dispersants.

On rocky shores, most oil was removed by the dispersants. That not removed right away was removed by wave action and weathering within one year.

The successional pattern of recovery was similar in all areas and resembled that which occurs following experimental scraping clean of rock surfaces on the shore (Figure 4.12). However, the rate of recovery was inversely related to the severity of impact, mainly of dispersants. Within four months after the spill, the opportunistic green algae Enteromorpha and Ulva began to appear on the intertidal rocks, and increased in coverage through the summer and fall. The rapid colonization and growth of these green algae was made possible by the nearly complete elimination from the rocky shore of grazing herbivores, such as limpets, top shells, and winkles. During the late summer and autumn, large brown algae Fucus vesiculosus and F. erratus began to succeed the green algae. By spring-summer, 1968, there was a very dense growth of green and brown algae covering the rocky shore. However, faunal abudance still was greatly reduced compared to pre-spill conditions. Barnacles which survived the oiling and dispersant application were smothered by the dense growth of algae. The surviving limpets Patella vulgata grew much more rapidly than normal because of the great abundance of food. They were joined in the spring by large numbers of juvenile limpets which had settled at low tide level during the previous winter.

The recruitment and growth of Patella was insufficient to prevent the continued heavy settlement and rapid growth of Fucus in 1968. However, the continued rapid increase in the Patella populations resulted first in a decrease in successful settlement of Fucus in 1969 and then a gradual aging and thinning of the growth of Fucus from 1970 on. The Patella population reached a peak in 1972 and then crashed as the algal food supply was depleted. Continued grazing by Patella resulted in a nearly complete elimination of Fucus from the rocky shore by 1974.

During the years when fucoid algae dominated the rocky shore, percent rock coverage of encrusting coralline algae Lithothamnion and abundance of dog-whelks Nucella lapillus increased dramatically. With the subsequent decline of the fucoid algae, abundance of whelks and coralline algae decreased and abundance of barnacles Chthamalus increased.

Lightly oiled, wave-exposed rocky headlands that received light dispersant treatment showed almost complete recovery after 5-8 years. Heavily oiled protected shores that received repeated applications of dispersants required 9-10 years or longer to recover. During recovery, subtidal algae such as Laminaria digitata and Himanthalia elongata extended further up the shore than before the spill.

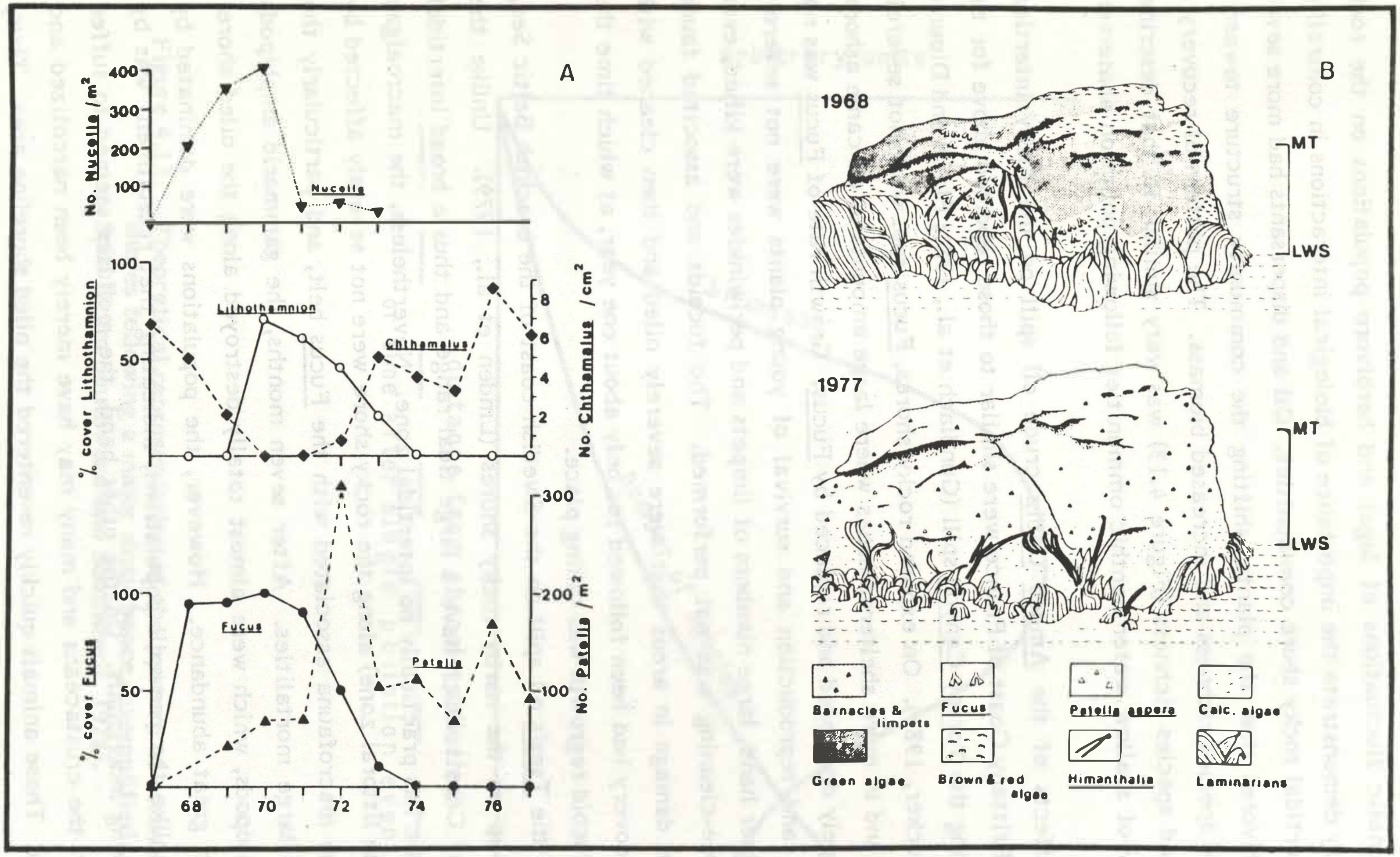

Figure 4.12 Fluctuations in populations of selected intertidal rocky shore plants and animals at mean tide level at Portleven between 1967 and 1977 (A). This habitat was heavily oiled following the Torrey Canyon Oil Spill in March 1967 and was subsequentially heavily treated with oil dispersants. (B) Pictorial representation of the recovery of plant and animal populations on wave-beaten intertidal rocks at Cape Cornwall, Great Britian following oiling by the Torrey Canyon Oil Spill. The upper picture represents the situation 13 months after all Herbivores had been killed by oil dispersants; the bottom picture represents the situation 9 years later in May 1977 (from Southward and Southward, 1978).

The dramatic fluctuations of lagal and herbivore populations on the rocky shore during recovery demonstrate the importance of biological interactions in controlling the structure of intertidal rocky shore communities. Oil and dispersants had more severe effects on the herbivores than the plants, shifting the community structure toward a condition of reduced species richness and increased biomass. The pattern of recovery in terms of biomass and species richness (Figure 4.13) was very similar to that described earlier for recovery of shallow water benthic ommunities following pollution abatement (Figure 4.1).

Initial effects of the Amoco Cadiz crude oil spill on the rocky intertidal communities of the Brittany Coast of France were similar to those reported above for the British Coast following the Torrey Canyon spill (Gundlach et al., 1981; Floc'h and Diouris, 1981; Topinka and Tucker, 1981). On exposed rocky shores, Fucus spp. were not severely affected by the oil, and in more sheltered areas where large amounts of oil came ashore, Ascophyllum was largely destroyed and replaced by Fucus. Growth rate of Fucus was not affected by the oil and reproduction and survival of young plants were not severely depressed. On the other hand, large numbers of limpets and periwinkles were killed, even where agressive shore-cleaning was not performed. The fucoids and associted fauna suffered more severe damage in areas that were severely oiled and then cleaned with water or steam. Recovery had been followed for only about one year, at which time the early phase of rapid fucoid regrowth was taking place.

Following the Tsesis oil spill on the Swedish coast of the brackish Baltic Sea, some of oil washed up on the narby rocky shores (Linden et al., 1979). Unlike the Cornwall and Brittany Coasts which have a large tidal range and thus a broad intertidal zone, the central Baltic has practically no intertidal zone. Nevertheless, the macroalgae which form a belt (the littoral zone) along the rocky shore were not severely affected by the oil. However, the macrofauna associated with the Fucus belt, and particularly the crustaceans, suffered large mortalities. After seven months, the gammarid amphipods and some species of isopods, which were almost totally destroyed along the oiled shore, were present again in great abundance. However, the populations were dominated by juvenile individuals, unlike the pre-spill populations, indicating that recruitment was by reproduction of surviving individuals. On the other hand, the molluscs seemed to suffer fewer mortalities than the crustaceans and many may have merely been narcotized and dropped to the bottom. These animals quickly re-entered the oiled shoreline area. Thus, faunal recovery and repopulation of the oil-impacted shore was accomplished by

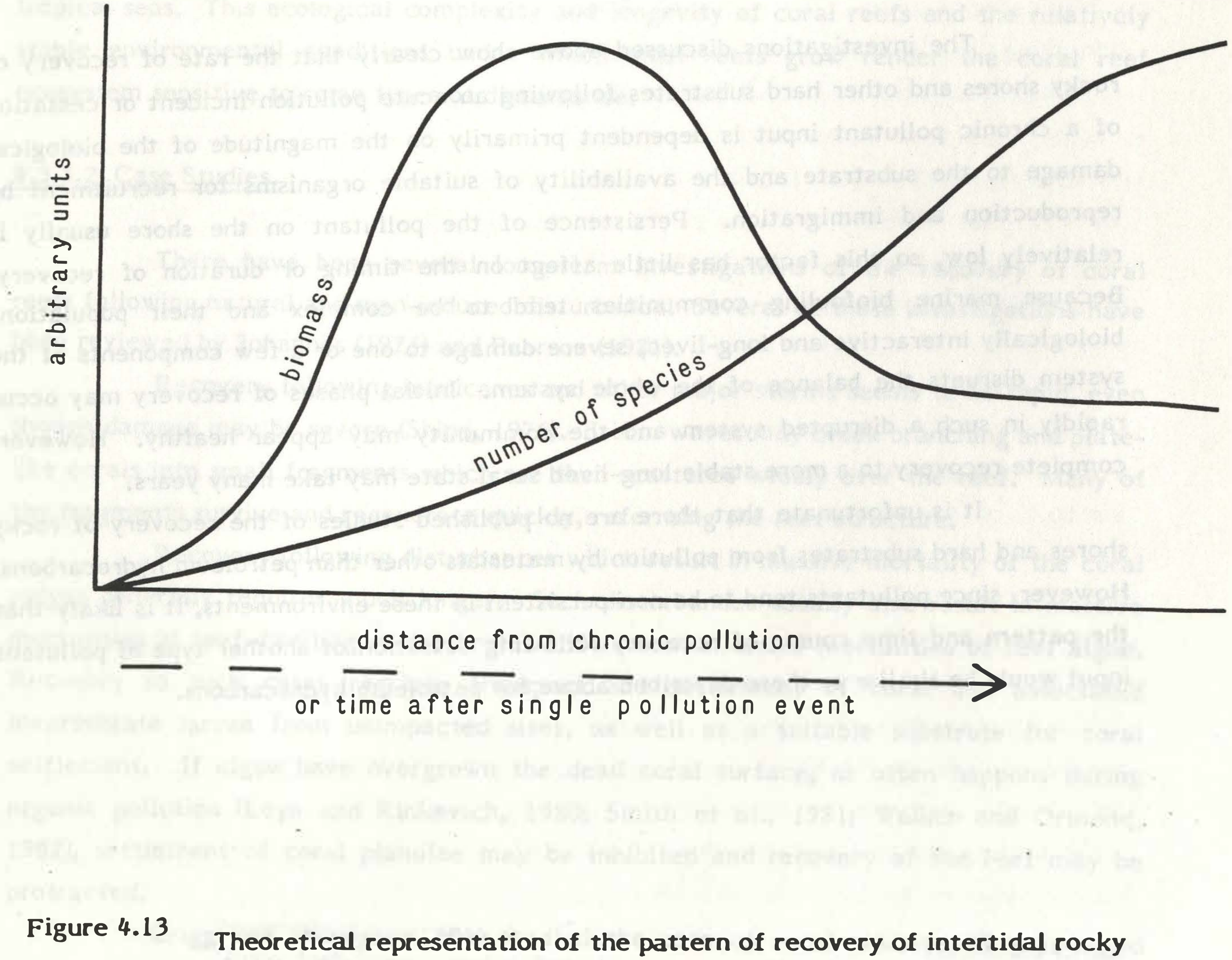

Figure 4.13 Theoretical representation of the pattern of recovery of intertidal rocky shores following a major disturbance, exemplified by the Torrey Canyon Oil Spill (from Southward and Southward, 1978).

immigration and reproduction. Recovery was not complete after one year and the authors speculated that complete recovery of all faunal groups on the rocky shore would require two to three years. Thus, in this spill, the magnitude of the impact of the oil on the rocky shore was not great enought to dissrupt the macroalgae-herbivore balance. As a result, recovery was quite rapid.

4.3.2.3 Summary.

The investigations discussed above show clearly that the rate of recovery of rocky shores and other hard substrates following an acute pollution incident or cessation of a chronic pollutant input is dependent primarily on the magnitude of the biological damage to the substrate and the availability of suitable organisms for recruitment by reproduction and immigration. Persistence of the pollutant on the shore usually is relatively low, so this factor has little affect on the timing or duration of recovery. Because marine biofouling communities tend to be complex and their populations biologically interactive and long-lived, severe damage to one or a few components of the system disrupts the balance of the whole system. Initial phases of recovery may occur rapidly in such a disrupted system and the community may appear healthy. However, complete recovery to a more stable long-lived seral state may take many years.

It is unfortunate that there are no published studies of the recovery of rocky shores and hard substrates from pollution by materials other than petroleum hydrocarbons. However, since pollutants tend to be non-persistent in these environments, it is likely that the pattern and time course of recovery following cessation of another type of pollutant input would be similar to those described above for petroleum hydrocarbons.

4.3.3 Coral Reefs

4.3.3.1 Background.

Coral reefs are among the most complex, productive, and diverse ecosystems on earth (Goreau et al., 1979). Close interactions among the members of the coral reef community insure the efficient utilization and recycling of the limited nutrients of tropical seas. This ecological complexity and longevity of coral reefs and the relatively stable environmental conditions under which most reefs grow render the coral reef ecosystem sensitive to some types of disturbance.

4.3.3.2 Case Studies.

There have been several long-term investigations of the recovery of coral reefs following natural and man-induced disturbance. Several of these investigations have been reviewed by Johannes (1975) and Pearson (1981).

Recovery following hurricanes and other major storms seems to be rapid, even though damage may be severe (Shinn, 1976). Storm waves may break branching and plate-like corals into small fragments which are then scattered widely over the reef. Many of the fragments survive and regenerate quickly, reforming the reef structure.

Recovery following disturbances which result in massive mortality of the coral polyps generally requires much longer. Such disturbances usually also result in massive mortalities of reef-dwelling invertebrates and somewhat lesser mortalities of reef algae. Recovery in such cases requires transport and settlement of coral and associated invertebrate larvae from unimpacted sites, as well as a suitable substrate for coral settlement. If algae have overgrown the dead coral surface, as often happens during organic pollution (Loya and Rinkevich, 1980; Smith et al., 1981; Walker and Ormond, 1982), settlement of coral planulae may be inhibited and recovery of the reef may be protracted.

Grigg and Maragos (1974) studied the rate of recolonization of submerged larva flows in Hawaii by reef-building corals. The authors compared the coral reef communities inhabiting six submerged lava flows of different ages with nearby undisturbed coral communities. In areas exposed to wave action, recovery of the coral community (in terms of number of species, species diversity, and percent substrate coverage by corals) following complete decimation by the lava required about 20 years. The normal coral community structure in such areas reflects the effects of frequent disturbance and is dominated by pioneer species. In sheltered areas, on the other hand, recovery may require more than 50 years. The sheltered coal reef community contains many slow-growing long-lived massive species of corals and is more characteristic of a climax coral community. There was some indication that species diversity in these recovering coral communities increases during succession and reaches a peak before the climax community is reached. Increased diversity also results in increased competition for space and limited resources, resulting in eventual elimination of the opportunistic species and a reduction in diversity as the climax community is approached.

Detailed studies were performed of recovery of two coral reefs in the northern Gulf of Aqaba, Red Sea, following a catastrophic low tide which killed 90 percent of the scleractinian coral communities in the area (Loya, 1975; 1976; Rinkevich and Loya, 1977; 1979; Loya and Rinkevich, 1980). One reef was in a nature preserve located 3 km south of an oil terminal which had been a source of frequent oil spillage. Counting only those spills large enough to "blacken" the coral reef nature preserve, ninety-five spills occurred in 1971 to 1973, with a frequency ranging from one to seven per month. There also was phosphate enrichment, causing eutrophication, in the nature preserve due to effluents from a nearby fertilizer plant (Fishelson, 1973a; Loya, 1975). The second reef, located 5 km south of the first apparently was completely free of any form of human perturbation. Three years after the catastrophic low tide, the second "clean" reef was flourishing with a high diversity of reef corals. However, on the chronically polluted reef, almost no recolonization had occurred. In addition, structure of the coral community at the clean reef had returned to a composition similar to that before the catastrophe, while a markedly different community structure persisted at the polluted reef. The authors suggested that the chronic oil pollution interfered with normal reproductive processes in the corals and thereby prevented normal recolonization of the reef. The possibility that other factors, including phosphate pollution and differences in hydrographic conditions at the two sites, might have contributed to differences in recovery rates at the two sites were not considered.

The reef flat communities of the northern Red Sea are naturally stressed by salinity and temperature. Coral reef diversity in the Red Sea decreases from south to north as one approaches the northern limit of normal coral distribution. The reef flats around Eilat and Aqaba are dominated by rapidly growing opportunistic corals such as

Stylophora pistillata (Fishelson, 1973b), and therefore may be more resilient than coral reef communities in more stable tropical environments.

4.3.3.3 Summary.

It is apparent from the few studies discussed above, that under the right conditions, recovery of coral reefs following even a severe disturbance, may begin fairly quickly, probably within 1-2 years. Before coral planulae will settle on a new scoured-clean surface, a period of about one year seems to be needed to "condition" the surface (Grigg and Maragos, 1974). The exact nature of the conditioning process is not known. As with rocky shores, if grazers are decimated by the disturbance, a rapid overgrowth of available hard substrates by filamentous algae or soft corals may inhibit settlement of coral planulae (Pearson, 1981), at least until grazers return and clear space for coral settlement. Complete recovery may require ten years or more. However, in most cases, it is difficult to clearly define the climax coral community, making uncertain the judgement of when recovery is complete.

4.3.4 Salt Marshes

4.3.4.1 Background.

Very extensive salt marshes occur along the U.S. Atlantic and Gulf Coasts and along large stretches of the Alaskan Coast. These marshes are important nursery areas for many food-chain or commercial fisheries organisms (Shenker and Dean, 1979; Weinstein, 1979). Although marshes in different latitudes resemble one another superficially, there are important differences. For instance, New England marshes tend to be small with a peaty, high organic carbon substrate, whereas marshes south of New Jersey and along the Gulf Coast tend to be more extensive with fine-grained mineral soils lower in organic carbon (Nixon and Oviatt, 1973). These differences in soil type, particularly percent organic carbon, influence retention of pollutant chemicals in the marsh and thus the rate of recovery of the marsh following pollution abatement or following an acute pollution event, such as an oil spill. For instance, recovery of a salt marsh near Falmouth, Massachusetts was incomplete more than twelve years after a small spill of diesel fuel (Teal and Howarth, 1984), whereas direct application of a light crude oil was without effect on the flora and fauna of a Louisiana salt marsh (DeLaune et al., 1984).

4.3.4.2 Case Studies.

Baker (1973) studied the effect of successive oilings at monthly intervals with Kuwait crude oil on the recovery of salt marsh vegetation of coastal South Wales, Great Britain. Up to four oilings caused only short-term (one year) damage to the lower marsh, Spartina anglica community. After 12 oilings, the Spartina was nearly completely eliminated, but began to return after two years, mainly through vegetative colonization from nearby untreated areas.

The mid-marsh Puccinellia maritima community was unaffected by up to four oilings. After eight or twelve oilings, the Puccinellia was nearly completely destroyed and there was little or no recovery after four years. The upper marsh Juncus maritimus community was damaged severely by two or more oilings. Plots receiving two or four oilings showed only modest recovery after four years; those receiving eight or twelve oilings were still devoid of plants after four years. Another marsh plant, Agrostis stolonifera recovered rapidly and actually increased its total percent coverage in the Juncus zone, even after twelve oilings.

These studies demonstrated the wide range in sensitivity of different marsh grasses to oil pollution and the variable rate of recovery following disturbance. Since recovery of the marsh vegetation was primarily by vegetative growth from surviving plants, recolonization of bare marsh is likely to be very slow if large areas of marsh have been impacted.

These conclusions drawn from small-scale experimental oilings of salt marsh plots are amply verified by extensive field studies of recovery of a salt marsh near Southampton, England following substantial reduction in petroleum hydrocarbon-contaminated wastewater discharges from an oil refinery (Baker, 1971; Dicks and Iball, 1981; Dicks and Hartley, 1982).

In 1951 and 1953, the refinery-petrochemical complex began discharging wastewater at two locations in dendritic creeks of the salt marsh. Between 1963 and 1972, the combined discharges into the marsh amounted to about 672,000 m3 of wastewater per day containing 16.8-20.2 metric tons of oil. After 1972, effluent volume was decreased slightly and petroleum hydrocarbon concentration in the effluent was reduced by more than one-half, so that between 1974 and 1980 the effluent was reduced by stages to 504,000-552,000 m^2 per day containing 5.0-7.7 tons of oil.

This 2-3 fold reduction of oil input to the marsh was sufficient to allow substantial recovery of the marsh flora and fauna (Figure 4.14). Between 1953 and 1970, approximately 600,000 m^2 of the marsh had been completely denuded of vascular plants by the effluents. Before the pollution, the marsh was characterized as a Spartinetum, dominated by the common cordgrass Spartina anglica. The sediments were a soft mud. The Spartinatum is a long-lived intermediate seral stage in the succession of salt marshes in this region. Earlier successional stages include a period of stabilization of the intertidal mud flat by filamentous green algae, followed by the Salicornetum, characterized by opportunistic salt-tolerant annual grasses such as Salicornia spp. and Sueda maritima. Although Salicornia is sensitive to oil, it is propagated rapidly by waterborne seeds in denuded areas.

The wastewater effluent returned the successional stage of the marsh to the filamentous green algae stage. Following pollutant abatement, recolonization by Salicornia and some other species that are spread by seeds, including the perennial aster Aster tripolium, was quite rapid (Figure 4.14). Spartina anglica, a perennial grass with extensive rhizomatous roots, spread back into the damaged marsh slowly at first from the edges of the impacted zone, but more rapidly after about 1977. There was a good inverse relationship between the total concentration of oil in the effluent and the effluent index (effluent volume x oil concentration, divided by 10,000) on the one hand and an area of the damaged marsh recolonized by Salicornia and Spartina between 1970 and 1980. Transplantation studies showed that persistent damage in the still denuded area was due primarily to the characteristics of the wastewater still being discharged and not to accumulation of petroleum hydrocarbons in the sediments. Sediment total aliphatic hydrocarbon concentrations in the marsh sediments in 1979-1980 ranged from 340-17,750 ppm. Spartina was present at loctions containing up to at least 1,000 ppm sediment hydrocarbons, while Salicornia was growing in all but the most heavily contaminated sediments.

The animal communities of the areas of the marsh which had formerly been denuded of vegetation were dominated by opportunistic oligochaetes (Enchytraeidae and Tubificidae) in 1979. Sediments containing up to 5,875 ppm weathered oil (as aliphatics) contained apparently healthy populations of Salicornia and oligochaetes.

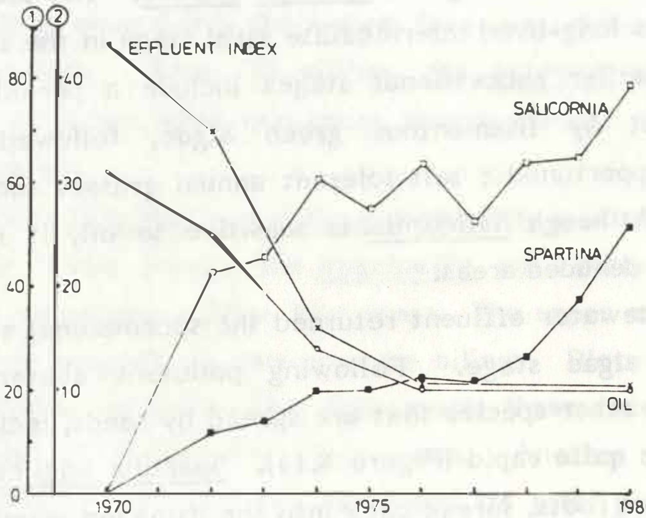

Figure 4.14 Comparison of long-term patterns of effluent quality index (effluent volume times oil concentration, divided by 10,000), and concentration of oil in the effluent from an oil refinery on the one hand and areas of damaged marsh recolonized by Spartina anglica and Salicornia spp. (as percentage). Curves for effluent index, Spartina and Salicornia use vertical axis 1; the curve for oil concentration was vertical axis 2 (from Dicks et al, 1981).

Thus, an approximately 50 percent reduction in the amount of oil discharged in refinery wastewater effluents to the salt marsh resulted in a fairly rapid repopulation of damaged areas by marsh grasses and oligochaetes. In areas at the furthest extent of effluent damage, where recolonization had been occuring for 10 years, the marsh appeared healthy with a diverse community of plants and animals. Nevertheless, there still were important differences in species composition of the recovered marsh compared to nearby unoiled Spartina marsh. Final recovery could require several more years and/or further reduction in oily wastewater discharges.

Bender et al. (1980) performed experimental spills of a light southern Louisiana crude oil in salt marsh plots in the York River, Virginia, an oligohaline-mixohaline (salinity, 0.2-18.5 $^{o}/oo$) estuary. The oil markedly reduced the biomass production of Spartina alterniflora during the year after the experimental spill. Growth of the marsh grass still was reduced in the second year. However, by the third year, the standing crop of marsh grass was equal to or greater than that in control plots, and the investigators concluded that recovery of the marsh was nearly complete. Populations of benthic animals in the marsh itself and in marsh creeks also were severely affected by the experimental oiling. These benthic populations also showed signs of substantial recovery after three years.

Not all components of the salt marsh community are affected equally by spilled oil. Meiofauna were little affected by an experimental oiling of a Louisiana Spartina alterniflora salt marsh with southern Louisiana crude oil (Fleeger and Chandler, 1983). No oil-induced mortality of any species of meiofauna could be detected even at oil concentrations greater than 13,000 ppm. Abundance of some species increased in oiled sediments, possibly due to increased microbial activity.

Krebs and Burns (1977) studied the effects of a spill of diesel fuel in Buzzards Bay, Massachusetts on populations of fiddler crabs Uca pugnax in the Wild Harbor marsh. The abundance of Uca in the marsh was reduced at all stations at which sediment hydrocarbon concentrations exceeded 200 ppm At sediment concentrations above 1,000 ppm, crab populations were nearly completely eliminated. At the most heavily oiled stations, there was little recruitment of juvenile crabs even after seven years, as long as sediment hydrocarbon concentrations exceeded 1,000 ppm. At stations where sediment hydrocarbon concentrations dropped to the 500-1,000 ppm range, recruitment was better, but crab abundance was still greatly reduced compared to clean areas after seven years. At stations at which sediment petroleum hydrocarbon concentrations dropped below 100 ppm, there was a normal abundance and size distribution of *Uca* after seven years. The authors concluded that petroleum concentrations in sediments in the range of 100-200 ppm were toxic to juvenile overwintering crabs. Much of the toxicity of the spilled diesel fuel to *Uca* and other marsh biota was attributed to high concentrations of aromatic hydrocarbons, particularly alkyl naphthalenes which were very persistent in the marsh sediments (Teal et al., 1978).

Another diesel fuel spill nearby, four years later caused severe damage to salt marshes at Winsor Cove, Massachusetts (Hampson and Moul, 1978). The stands of *Spartina* and *Salicornia* which were present in the marsh at the time of the spill were nearly completely absent three years later in 1977. In three years, there was little evidence of re-establishment of the lower marsh either by reseeding or rhizome growth. In the upper marsh, *Saliconria virginica* did begin to re-establish itself after two years. In the absence of *Spartina* in the lower marsh, populations of ribbed mussels *Modiolus dimissus* became more abundant than in reference areas after two or three years, even though they were destroyed in large numbers by the spill.

The denuded areas of marsh were covered with a blue-green algal mat during 1975-1977. The opportunistic green alga *Enteromorpha* was found sparingly in the marsh. Even after three years, benthic meiofauna of the oiled marsh were greatly reduced, the extent of reduction being correlated with petroleum hydrocarbon concentrations in the peaty sediments. These two studies show that northern temperate salt marshes recover slowly from spills of toxic refined petroleum products. This, undoubtedly, is due primarily to the persistence of petroleum hydrocarbons, particularly the toxic aromatics, in the peaty marsh sediments, and not to any particular sensitivity of the marsh biota which are very similar to those of more southern, more resistant salt marshes.

Following the Amoco Cadiz oil spill, large areas of the salt marshes at Ile Grande on the north Brittany coast were destroyed during the cleanup effort (Seneca and Broome, 1982). In some areas, only the above-ground marsh vegetation and associated oil had been removed. In other areas, the entire marsh surface, including the root mat, had been removed to a depth of more than 30cm. Attempts were made to restore these areas by replanting marsh grasses. After three years, the restoration had met with some success in areas within 30cm of the elevation of the natural marsh in the vicinity.

In those areas that were not destroyed by the cleanup effort, there were varying degrees of recovery of the marsh grasses in the three years following the spill. Recovery ranged in high marsh areas barely affected by the oil to practically no recovery at all (5-20 percent) in some low-lying areas adjacent to major marsh creeks which had been heavily oiled (Levasseur and Jory, 1982).

Overall recovery of the marsh vegatation in the areas studied was estimated at about 35 percent by the end of 1981. The two main recovery processes were regeneration or regrowth of perennial grasses from surviving rhizomes, and germination of seeds of annual grasses in the denuded areas. There was little germination of seeds of perennial species. Reseeding was very slow in areas exposed to strong tidal influences and in areas that had been beaten down by foot traffic. Some species, such as Juneus maritimus, which survived the initial oiling, subsequently showed evidence of decline and die-out, possibly due to accumulation of hydrocarbons in the sediments. In some areas, these perennial species were replaced by annuals, so that the floral composition of the marsh changed. The process of marsh recovery seemed to accelerate after 1980, but undoubtedly would require several more years to complete.

In marked contrast to the severe damage and long recovery time following and discharges to temperate marshes in Massachusetts, England and France, recovery of a salt marsh in Galveston Bay, Texas following a spill of Bunker C residual oil was very rapid (Webb, et al., 1981). All the above-ground biomass of the oiled Spartina alterniflora marsh was removed along creek channels during the cleanup effort in November. Regrowth of the Spartina was complete by the following spring. In other areas of the marsh propper, oil partly or completely covered the marsh grass. The oil killed only the above-ground portions of the grass. These plants all produced new growth the following spring and appeared to be completely normal. However, more recently, Alexander and Webb (1985) showed that when Spartina in Galveston salt marshes was completely covered with No. 2 fuel in either November or May, complete recovery of the plant biomass required at least twelve months.

4.3.4.3 Summary.

Salt marsh ecosystems seem to be somewhat resistant to moderate or one-time pollutant-mediated disturbance. However, if the acute or chronic pollution causes severe damage, then resilience is low and recovery may take many years. Marsh grasses are sensitive to the level in the intertidal zone at which they occur. Thus, activities which lower the level of the marsh surface will produce long-term changes in the marsh. This may include use of heavy equipment to remove contaminated surface layers of the marsh, as occured in the Ile Grande marshes on the north coast of Brittany following the Amoco Cadiz spill (Long et al., 1981). In low southern marshes with soft muddy soils, people just walking in the marsh while cleaning up spilled oil manually may compact the soil and cause longer-lasting damage than the oil itself (Lindstedt-Siva, 1979).

4.3.5 Seagrass Beds

4.3.5.1 Background

Extensive beds of seagrasses (aquatic flowering grasses) occur in the shallow subtidal zone of the U.S. Atlantic, Pacific, and Gulf Coasts. Zostera marina is found on the Pacific Coast from Port Clarence, Alaska to Agiopampo Lagoon, Mexico, and on the Alantic Coast from southern Greenland to Cape Fear North Carolina (Thayer et al, 1975). It is replaced south of Cape Fear and through the Gulf of Mexico by turtle grass, Thalassia testudinum and other tropical and subtropical seagrasses. Seagrass beds are extremely productive. Eelgrass itself may produce approximately 300-600 g dry biomass/m^2/year, excluding root production (Wood et al., 1969). In addition, it may support an equal biomass of epiphytic organisms. Eelgrass detritus is an important nutritional source for many nearshore invertebrates and fish.

Despite the importance in the coastal marine ecosystem of seagrass beds, little is known about their resistance and resilience with respect to pollution and other anthropogenic disturbances. Seagrass beds generally overlie anaerobic sediments. Wood (1959) reported that when Zostera was removed from an area, the sediments became aerobic, and recovery of the eelgrass was impaired. However, Marshall (1970) reported that when Zostera was removed by hand from small experimental plots, it recovered completely within one season. Dredging activities along the Texas coast near a seagrass bed containing Thalassia and Halodule resulted in decreased light penetration, increased sedimentation, and reduced chlorophyll content and productivity of the grasses (Odum, 1963). During the following growing season, plant production was greater than before dredging in areas not completely smothered by sediments. The Thalassia was killed by burial under 30 cm of sediment.

4.3.5.2 Case Studies.

Seagrasses are sensitive to high water tempratures, and Thalassia beds at the mouth of the cooling water discharge canal of the Turkey Point, Florida power plant are destroyed each summer, only to return each fall (Zieman, 1970).

Following the March 1973 Zoe Colocotronis spill of Venezuelan crude oil into coastal waters of southern Puerto Rico, several tropical shoreline habitats including turtlegrass beds, were heavily oiled (Nadeau and Bergquist, 1977). During the spill, the Thalassia leaves in some areas of the seagrass flats were completely coated with oil and oil was driven into the coralline sandy sediments by the wave action. The top growth of heavily oiled Thalassia died and was removed by wave action. Ten months later, there was evidence of seagrass regrowth. Two years later, the seagrass beds had shown substantial recovery. The macro epifauna and infauna communities of the seagrass flats were severely damaged by the spill. Many of the epibenthic animals had returned within thirteen weeks, but infaunal abundance and diversity remained dipressed. No long-term observations of faunal recovery in the seagrass flats were made.

Other investigations have suggested that although seagrasses themselves appear to be moderately tolerant of pollution, epibenthic and infaunal communities associated with seagrass beds may be adversely affected by pollutants that do not directly affect the seagrasses. For instance, adverse effects on the epibenthic invertebrate and fish faunas of seagrass beds have been reported in a Thalassia bed in Appalachee Bay, Florida receiving pulp mill effluents (Heck, 1976) and in a Zostera bed in Spencer Gulf, South Australia receiving metaliferous effluents from a lead smelter (Ward and Young, 1982).

Following the Amoco Cadiz crude oil spill, the oil slick spread over the extensive Zostera marina beds of Roscoff, France. Although the seagrass wasn't severely damaged, the associated epi- and in- fauna were (Den Hartog and Jacobs, 1980; Jacobs, 1980). During the first few weeks after the spill, the leaves of some plants had black "burned" patches. The damaged leaves were shed and quickly replaced. Of the epibiota, the gastropods were hardly affected, while cumaceans, tanaeids, and echinoderms suffered massive mortalities at the time of the spill. Their populations had returned to their former abundance and diversity within one year. The amphipods, though not abundant in the seagrass bed before the spill, were rich in species. Of the 26 species of amphipods present before the spill, 21 had not been seen in the year after the spill. The pre-spill dominant, Pherusa fusicola, reappeared in small numbers during the autumn after the spill. In addition, the opportunist Gammarus locusta, formerly absent, appeared in the seagrass bed during the fall and winter after the spill.

Following the spill, there was a sharp decrease in numbers of both individuals and species of benthic infauna in the seagrass bed. The smaller crustaceans and echinoderms were almost completely eliminated. Recovery began rapidly. By early 1979, less than a year after the spill, numbers of individuals and species had returned to near pre-spill values. Only the filter-feeding amphipods did not show significant recovery within one year.

4.3.5.3 Summary.

This limited number of studies would seem to indicate that seagrass communities are relatively tolerant to pollutant stress, and recover rapidly following disturbance that does not remove or destroy the roots of the grasses. However, where the grasses have been destroyed, recovery may be very slow. There has been a gradual and progressive decline in submerged aquatic vascular plants in Chesapeake Bay during the last twenty years (Orth and Moore, 1981, 1983). In polykaline and mesohaline areas, the dominant grasses are Zostera marina and Ruppia maritima. Although the causative agents for the decline are not known, increased concentrations of primary nutrients (from sewage) and increased water turbidity (from phytoplankton growth and land erosion) seem to be contributing factors. There has been some evidence recently of vegetational recovery in areas containing sparse patches of grass. However, in areas that are completely devoid of vegetation and far removed from sources of propagules, recovery has been minimal and is likely to be very slow.

4.3.6 Mangrove Swamps

4.3.6.1 Background.

Mangroves are trees or bushes that occur on sheltered shores of estuaries and coasts in tropical and subtropical climates. In the continental United States, they are restricted to areas of the Florida and Gulf Coasts. Where they occur, mangroves are an important source of shelter and detrital nutrition to a rich and diverse community of mangrove-associated biota. Mangroves usually grow in anaerobic muds and their roots receive oxygen through specialized stilt roots or pneumatophores.

4.3.6.2 Case Studies.

As with seagrass communities, the mangrove community appears to be relatively resistant and resilient to moderate pollutant-mediated disturbance. However, if the disturbance is sufficient to kill the trees, then recovery may be very slow. Torential rains in October, 1958 killed much of the epibenthic fauna associated mangrove forests in Kingston Harbor, Jamaica (Goodbody, 1961). Two months later, there was limited recolonization of submerged roots by sponges, serpulid, and sabellid worms, barnacles, and bivalve molluscs. After three months, several other taxa had returned and recovery was well-advanced.

Getter et al. (1981) reviewed the impacts of five oil spills on mangrove forests of the Gulf of Mexico and Caribbean Sea. The spills differed substantially in the physiography of the spill site and the nature and amount of oil spilled. The types of responses observed included defoliation, leaf deformation, and stunting, adventitious growth of pneumatophores, lenticel expansion, adult tree mortality, seedling deformation and mortality, and changes in the abundance and distribution of mangrove community fauna and flora. For instance, following oiling of mangroves in Machos Creek, Puerto Rico, the trees were heavily defoliated within two months,. They remained defoliated 18 months later. The defoliation allowed more light to penetrate to the forest floor, elevating temperatures and salinities there. Under these conditions, numerous seedlings germinated and began to grow in the oiled substrate, but they had low survival. Stunting and leaf deformities were common. Recovery has been very slow.

In March, 1973, crude oil discharged from the grounded tanker *Zoe Colocotroni* seriously damaged red and black mangrove forests in the Bahia Sucia, Puerto Rico. In a small area, mangroves, *Rhizophora mangle* and *Avicennia nitida* died in the three years following the spill. After three to four years, the epifaunal and infaunal communities associated with the roots of surviving mangroves had shown considerable recovery (Nadeau and Berquist, 1977). Gilfillan et al. (1981) reported that five years after the spill, the black mangrove community had recovered completely. However, in the red mangrove forest, the number of individuals and species still was reduced. Substantial recovery had occurred despite the fact that sediments in the area contained up to 80,000 ppm of highly weathered oil.

In January, 1975, the tanker Showa Maru spilled approximately 54,000 barrels of oil in the Singapore Straits. The oil severely damaged mangroves on the shores of nearby Indonesian islands (Baker, 1982). Two years later, large numbers of dead or damaged mangroves Rhizophora spp. were observed behind seaward fringes of Sonneratia. Areas containing dead mangroves usually also had sediments with high concentrations of petroleum hydrocarbons. Abundance of snails and crabs was greatly reduced in the severely impacted areas. The low abundance of mangrove seedlings in the impacted area indicated that recovery had just begun more than two years after the spill.

Chaw (1984) recently reviewed several studies of impacts of spilled oil on mangrove ecosystems. The most frequent effects recorded were large-scale mortality of mangrove epifaunal and infaunal invertebrates, defoliation of mangroves, and death of young and occasionally mature mangroves. When the mangroves themselves were not severely damaged or killed outright, recovery was rapid. In areas where mangroves were killed, little recovery was observed for up to seven years.

4.4 The Pelagic Environment

4.4.1 Background.

Pollutants spilled in or discharged intentionally to the ocean first came in contact with organisms living in the water column. The water column is the permanent or temporary habitat of a wide variety of organisms including phytoplankton and zooplankton which drift with the prevailing water currents and nekton which are free-swimming fish and invertebrates. Included among the zooplankton are the holoplankton, which complete the full life-cycle as members of the plankton, meroplankton, which are larvae and other early life stages of marine invertebrates and fish that later mature to become part of the nekton or benthos, and neuston which float on the water surface. These pelagic organisms, especially the meroplankton, are generally considered to be quite sensitive to pollutants.

However, pollutants are not persistent in the water column. Where mixing is adequate, pollutants are diluted and dispersed to very low concentrations very rapidly. If a pollutant discharge plume does not differ substantially in density and viscosity from seawater, its ingredients can be expected to be diluted many thousand-fold within hours of discharge. In addition, many pollutants tend to adsorb to suspended particles (both inert and biological) and settle with them to the bottom. Some organic pollutants, such as aromatic petroleum hydrocarbons, may be lost rapidly from the water column by evaporation or photo-, chemical-, and biological- oxidation (Neff, 1979).

As a result of these processes, planktonic organisms in the open ocean may be exposed to a plume of rapidly diluting pollutant chemicals for only a short period of time. Nektonic animals may avoid the plume altogether. In estuaries or other enclosed bodies of water, pollutants may be more persistent in the water column and chronic exposure of water column organisms is possible.

4.4.2 Phytoplankton.

Results of several laboratory, mesocosm, and field studies have indicated that in the presence of persistent pollutants including sewage, metals and chlorinated hydrocarbons, average cell size of the phytoplankton community decreases (O'Connors et al., 1978; Eppley and Weeler, 1979). Greve and Parsons (1977) have hypothesized that an increase in the abundance of small relative to large phytoplankton (decrease in the diatom/flagellate ratio) may increase the number of trophic levels and thereby decrease fish production. They provide evidence that the nanophytoplankton-based food chain leads to gelatenous zooplankton (ctenophores and medusae), while the microphytoplankton-based food chain leads to young fish. This shift from micro- to nano- plankton is not always seen after pollution incidents, such as oil spills (e.g., Lannergren, 1978), and may be caused or influenced by a wide variety of natural environmental factors (Parsons and Takahashi, 1973; Eppley and Weiler, 1979). Thus phytoplankton cell size probably is not a good indicator of pollutant impact or recovery of the water column.

Following the Tsesis oil spill along the Swedish coast of the Baltic Sea, the biomass and production of phytoplankton, already dominated by microflagellates, increased (Johansson, 1980). There was no change in phytoplankton production per unit biomass, suggesting that the increases were due to decreased grazing by zooplankton. Near the wreck site, the biomass of zooplankton declined sharply during the first few days after the spill. Within about five days, there was near complete recovery of zooplankton biomass and diversity, even though visible contamination of rotifers and copepods continued for about three weeks.

The spring after the spill, the spawning frequency of herring in the impacted area was lower than in nearby reference areas (Nellbring et al., 1980). Though this could not be attributed unequivocally directly to the oil, a decrease in hatching success could be. The decreased hatching success was due to a fungus infection which usually is controlled by gammarid amphipods which were virtually eliminated from the area by the oil. Thus, residual effects of the oil pollution persisted in the pelagic ecosystem for at least six months after the spill.

4.4.3 Zooplankton.

Following the Amoco Cadiz oil spill, the zooplankton community, including meroplankton, more than about 6.4 km offshore was unaffected (Laubier, 1978). However, there was a depression of phytoplankton abundance and growth for several weeks after the spill in the immediate vicinity of the wreck and in nearby estuaries which became heavily contaminated by the oil (Cabioch et al., 1981). Further away, phytoplankton production was stimulated following the spill. In nearshore waters and particularly the estuaries, zooplankton abundance was substantially decreased about three weeks after the spill. No recovery was apparent seventy days after the spill. However, by June, zooplankton communities in the estuaries had a normal species composition, but still depressed numbers, suggesting that recovery was underway.

Capuzzo and Lancaster (1981) studied the effects of discharge of American Cyanamid waste at the 106-mile deepwater dumpsite off New Jersey. Zooplankton collected from the waste plume within several hours after the dump showed reductions in feeding rate, metabolism and egg production compared to controls. These changes persisted when copepods were returned to clean seawater. However, the offspring of exposed animals were normal. Since only a small fraction of the zooplankton community is entrained in the waste plume, and generation time of the zooplankton is short, the impacts on zooplankton of waste dumping at the 106-mile dumpsite were thought to be minimal and of short duration.

4.4.4 Mixed Plankton.

Oviatt et al. (1984) performed experiments in the large marine mesocosm tanks at the Univesity of Rhode Island to assess the impact of grossly contaminated sediments on the overlying water column. Tanks containing sediments from the grossly polluted Providence River were compared to mesocosms containing measurably polluted sediments from central Narragansett Bay, and relatively unpolluted sediment from Rhode Island Sound.

After five months, water column communities, including phytoplankton, holoplankton, and meroplankton, were similar in the three treatments. After one year, phytoplankton and zooplankton biomass was slightly higher in the Providence River treatment than in the other two treatments. Based on the results of this experiment, the authors predicted that removing pollutant inputs to an estuary like the Providence River-Narragansett Bay system will, after a short time, result in a relatively clean and biologically normal water column. Pollutant materials in sediments will tend to stay there for long periods of time and, thus, will have little effect on the overlying water column.

4.4.5 Pelagic and Demersal Fish.

The conclusions of Oviatt et al. (1984) are amply illustrated by the rehabilitation of the Thmes River estuary, England (Gameson and Wheeler, 1977; Andrews and Rickard, 1980). The Thames River estuary has a long history of pollution, primarily from sewage effluents from the City of London. The Thames fishery began to decline approximately 150 years ago, and by 1950, the only species of fish able to survive in the inner estuary were eels. Between 1955 and 1964, extensive measures were taken to improve sewage treatment and reduce pollutant loads to the river. This resulted in a gradual rise in the dissolved oxygen concentration at half tide in summer from less than 10 percent saturation in 1948-1960 to about 50 percent saturation in 1975 at a location 30 miles below London Bridge.

The number of fish species taken in the river rose sharply from two in 1964 to 49 in 1960, and then more gradually to 100 in 1977-1979. Thus, by 1977, all the freshwater and euryhaline fish species indigenous to the Thames system and 95 percent of marine species indigenous to the southern North Sea had been seen in the river. The two species which occurred in the river in small numbers in 1969 (flounder Platichthys flesus and eel Anguills anguilla) increased dramatically in numbers there by 1976. More important, sensitive species like the tideway smelt Osmerus eperlanus and herring Clupea harengus returned to the river in large numbers.

There also have been changes in the benthic macrofauna of the inner Thames. Since 1974, there has been a shift from a depauperate community dominated by tubificid worms to a more diverse community dominated by the polychaete Nereis diversicolor and the amphipod Corophium volutator. Among the epibenthos, large numbers of gravid shrimp Crangon crangon and prawns Palaemon longirostris and P. serratus have returned to the Thames. There was a very rapid recolonization of the waters near the Beckton sewage work in 1976 when its effluent quality was improved and drought conditions allowed saltwater intrusion further than usual up the river.

4.4.6 Summary.

Thus, it is apparent that, although water column organisms are sensitive to pollutants, a combination of rapid dispersion, dilution, and degradation of pollutants in the water column on one hand and ralatively short life-cycles and rapid turnover times of many plankton species on the other, allow water column communities to recover very rapidly following acute pollution events or abatement of chronic pollutant inputs. Recovery of pelagic fish populations, of course, may require several years. The size of most fish stocks vary widely from year to year due to a wide variety of factors (Jones, 1982). The primary cause of these fluctuations, however, appears to be recruitment. Eggs and larvae are the most sensitive life stages of fish, and many factors, natural and pollutional, will influence their survival and thus annual recruitment. Thus, it is usually difficult or impossible to attribute even large changes in the size of fish stocks to pollution or pollution abatement (Spaulding, et al., 1983). An exception is in a situation like the cleanup of the Thames River, which allowed many species of fish to return to waters where they had not occurred for many years.

4.5 Chemical Recovery (Body Burdens)

4.5.1 Background.

There is a strong tendency for marine organisms to accumulate in their tissues certain inorganic and organic chemicals from seawater, food, and sediments. If the accumulated chemicals are present at higher than normal concentrations (i.e. metals) or are not natural components of the marine environment (i.e. synthetic organics and petroleum hydrocarbons), they may be accumulated by marine organisms to concentrations which are toxic to the organisms themselves or to consumers of the organisms, including man. Thus, spillage or intentional discharge to the ocean or coastal waters of chemicals and wastes may lead to contamination of the biota of the receiving waters. Following an acute pollution event, such as an oil spill, or following cessation of a chronic waste discharge, an important part of the subsequent recovery of the affected ecosystem will be the gradual release or metabolic degradation of accumulated pollutant chemicals by the resident biota as ambient levels of the pollutant decrease. Elevated body burdens in organisms from pollutant-impacted environments may contribute to reproductive impairment (Conan, 1982; Cross et al., 1984), slowing the rate of ecosystem recovery through reproductive-recruitment. In most cases, however, pollutant depuration by the indigenous biota of the damaged eocsystem contributes relatively little to the process of ecological recovery. The biological communities characteristic of the recovered ecosystem will not be present in the early phases of recovery, and ecological succession will involve recruitment of new populations from outside the damaged area by reproduction and immigration. Nevertheless, contaminant levels in the organisms inhabiting a pollutant-damaged ecosystem are a useful index of the stage and state of ecosystem recovery. Therefore, rates of release of different types of pollutant chemicals by marine animals following their return to clean seawater will be reviewed and evaluated here in relation to the process of ecosystem recovery.

4.5.2 Pollutant Release Rates.

Mean rates of release (expressed as half-time, T_{50}, or time in days required to release 50 percent of an accumulated body burden when the organism is returned to a clean environment) of several types of pollutant metals, aromatic hydrocarbons and chlorinated hydrocarbons are summarized in Table 4.2. It is quite apparent that some metals and chlorinated hydrocarbons are quite persistent in the tissues of marine animals. Aromatic (petroleum) hydrocarbons appear to be much less persistent. This is due to several factors. Many marine animals are able to metabolize aromatic hydrocarbons to more polar metabolites which are more readily excreted or in some cases bind tightly to tissue macromolecules (Neff, 1979; Lee, 1981; Stegeman, 1981). Many chlorinated hydrocarbons also are metabolized, but not as rapidly as polycyclic aromatic hydrocarbons.

In addition, in those species such as bivalve molluscs which do not readily metabolize polycyclic aromatic hydrocarbons, rate of hydrocarbon release following return to clean seawater seems to be inversely related to duration of exposure to hydrocarbon pollution (Neff and Anderson, 1981). Thus, animals, particularly bivalve molluscs, chronically exposed to oil in the field, often release the accumulated body burden of hydrocarbons much more slowly than animals which have been exposed for a short period of time in the laboratory.

Table 4.2. Estimated time required for freshwater and marine animals to release 50 percent of accumulated pollutant chemicals (half time, T_{50}) following return to a clean environment in the laboratory or field.

Compound	Species	T_{50} (Days)	Reference
Metals			
Cadmium	Lobster, Homarus americanus	NR	McLeese et al., 1981
	Oyster, Crassostrea virginica	120	Mowdy, 1981
		NR	Zaroogian, 1979
	Sandworm, Nereis virens	NR	Ray et al., 1980
Copper	Mussel, Mytilus edulis	120	Roesijadiætal., 1985
	Oyster, Crassostrea cucullata	60	D'Silva & Quasim, 1979
Lead	Mussel, Mytilus edulis	60	Schulz-Baldes, 1979
Inorganic Mercury	Mussel, Mytilus edulis	84	Roesijadi et al., 1985
	Crab, Cancer magister	30	Sloan et al., 1974
Silver	Oyster, Crassostrea virginica	NR	Greig & Wenzloff, 1978
	Mussel, Mytilus edulis	NR	Roesijadi et al., 1985
Arsenic	Oyster, Crassostrea virginica	NR	Okazaki & Pancitz, 1981
	Mussel, Mytilus galloprovincialis	32	Unlu & Fowler, 1979
	Clam, Scrobicularia plana	57	Langston, 1984
Aromatic Hydrocarbons			
Naphthalene	Oyster, Crassostrea virginica	2	Lee et al., 1978
	Mean for Several Species of Bivalves	4.8	Neff et al., 1976; Fucik et al., 1977
	Shrimp and Amphipods	0.3	Cox et al., 1975; Neff et al., 1976; Anderson et al., 1979
Anthracene	Oyster, Crassostrea virginica	3	Lee et al., 1978
Fluoranthene	Oyster, Crassostrea virginica	5	Lee et al., 1978
Benz(a)anthracene	Oyster, Crassostrea virginica	9	Lee et al., 1978
Benzo(a)pyrene	Oyster, Crassostrea virginica	18	Lee et al., 1978
	Copepod, Calanus plumchrus	8	Lee, 1975
Chlorinated Hydrocarbons			
Polychlorinated Biphenyls	Sandworm, Nereis diversicolor	17-50	Fowler et al., 1978
Aroclor 1242	Clam, Cerastoderma edule	10a	Langston, 1978
	Mussel, Mytilus edulis	8	Calambokidis et al., 1979
Aroclor 1254	Mussel, Mytilus edulis	23	Calambokidis et al., 1979
	Copepod, Acartia tonsa	0.5-1.0	McManus et al., 1983
	Clam, Cerastoderma edule	NR	Langston, 1978
Aroclor 1260	Mussel, Mytilus edulis	39	Calambokidis et al., 1979
Trichlorobiphenyl	Clam, Cerastoderma edule	5	Langston, 1978
	Mussel, Mytilus edulis	3	Calambokidis et al., 1979
Tetrachlorobiphenyl	Clam, Cerastoderma edule	6	Langston, 1978
	Lobster, Homarus americanus	28	McLeese et al., 1980
Pentachlorobiphenyyl	Clam, Cerastoderma edule	17	Langston, 1978
	Sandworm, Nereis virens	61-186	Goerke, 1984
Hexachlorobiphenyl	Mussel, Mytilus edulis	50	Calambokidis et al., 1979
	Lobster, Homarus americanus	119	McLeese et al., 1980
Trichlorobenzene	Trout, Salmo gairdneri	0.4	Melancon & Lech, 1980
Hexachlorobenzene	Trout, Salmo gairdneri	40-50	Niimi & Cho, 1981
Trichlorophenol	Minnow, Pimephales promelas	0.5	Call et al., 1980
Hexachlorocyclohexane	Brine Shrimp, Artemia salina	2.5	Canton et al., 1978
Kepone	Oyster, Crassostrea virginica	5.2	Morales Alamo & Haven, 1983
DDT	Marine Zooplankton	11	Harding & Vass, 1977
Dieldrin	Oyster, Crassostrea virginica	75	Mason & Rowe, 1976
Endrin	Oyster, Crassostrea virginica	67	Mason & Rowe, 1976
	Lobster, Homarus americanus	4	McLeese et al., 1980

For animals which encounter pollutant chemicals primarily via the water (pelagic animals and filter-feeders), the T_{50} values listed in Table 4.2 may give a rough indication of the rate at which pollutants will be released following cessation of a pollutant discharge or an acute spill. This is the case when the pollutants are removed from the water column rapidly following pollution abatement. However, if there is pollutant wash-out from sediments or other reservoirs, low concentrations of the pollutants may persist in the water column for long periods of time, slowing the rate of pollutant release by the resident fauna and flora.

Sediment-adsorbed pollutants are much less bioavailable to marine organisms than pollutants which are dissolved or dispersed in the water column (Neff, 1982; 1984; Breteler and Saksa, 1984). In several laboratory studies reviewed by Neff (1984), bioaccumulation factors for polycyclic aromatic hydrocarbons, polychlorinated biphenyls and pesticides from sediments (concentration in animal/concentration in sediment) by marine invertebrates ranged from 0.06 to 20. In two field studies where contaminant concentrations in sediments and biota from the same location were measured, bioaccumulation factors from sediments ranged from 0.001 to 4.46 for polycyclic aromatic hydrocarbons (Boehm, 1982) and 0.02 to 51 for polychlorinated biphenyls (Stainken and Rollwagen, 1979). The greater variance in the field data reflect the greater uncertainty about sediment contaminant levels to which the benthic biota analyzed were actually exposed, because of patchiness of sediment contamination. In any event, the boaccumulation data indicate that contaminant concentrations in benthic biota generally will be similar to contaminant concentrations in the sediments on or in which they reside. As a result, the long-term pattern of decline in the concentrations of pollutants in benthic and relatively non-migrating demersal biota following pollution abatement or an acute pollution incident ususally will parallel quite closely the pattern of decline in the concentrations of the pollutants in the surficial sediments of the site (Figure 4.15). However, the relationship between levels of sediment contamination and contamination of benthic/demersal biota will be different for different pollutants and different animal species. Connor (1984) recently examined field data from several contaminated marine sites and showed that the ratio of the bioaccumulation factors from sediment for chlorinated aromatic hydrocarbons and polycyctic aromatic hydrocarbons (chlorinated aromatic hydrocarbon biota/sediment ratio, divided by polycyclic aromatic hydrocarbon biota/sediment ratio) decreased for benthic and demersal species as the mixed function oxygenase activity of the species decreased. Thus, bivalve molluscs, which have little mixed function oxygenase activity, will have organic contaminant body burdens which

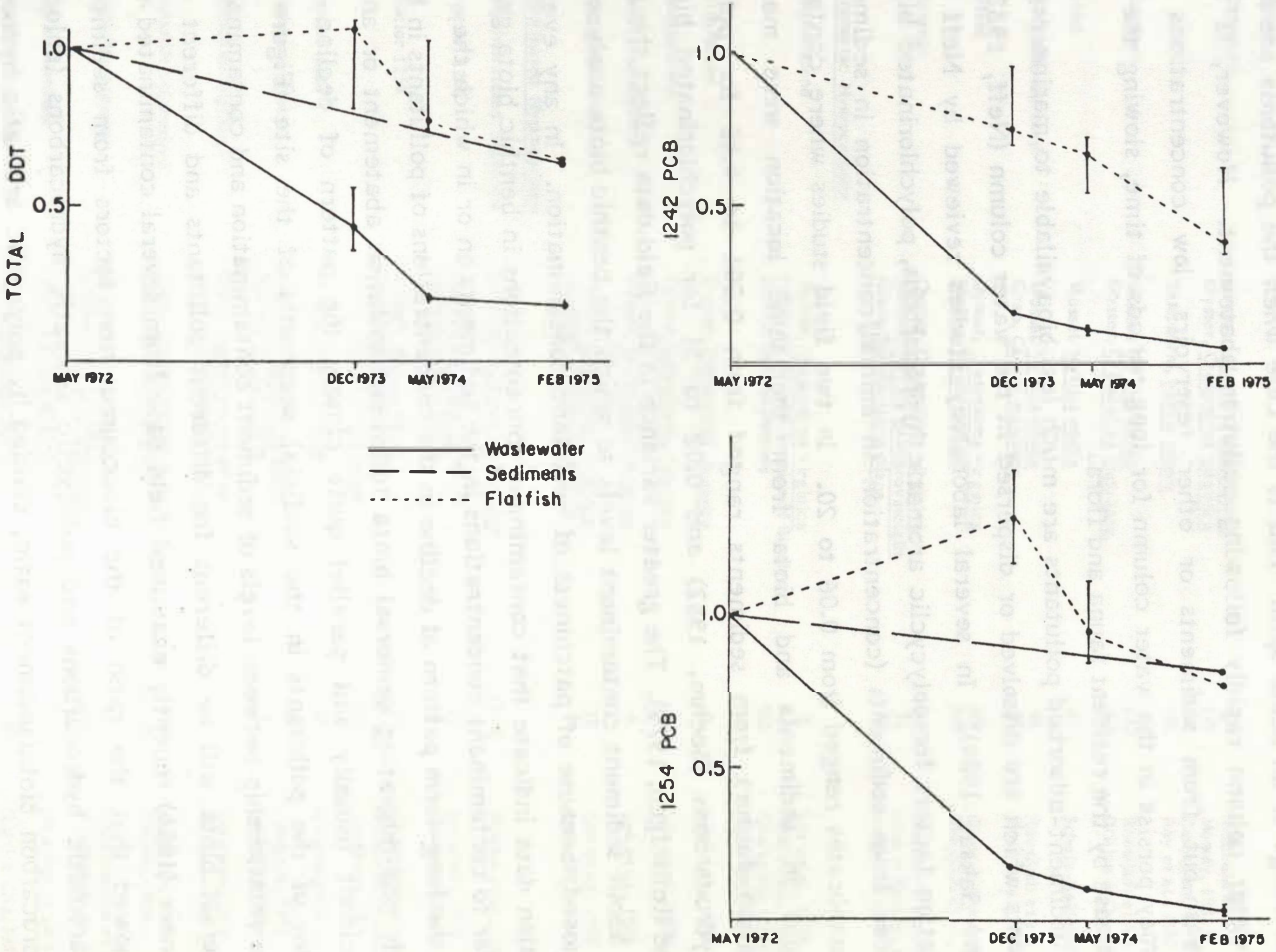

Figure 4.15 Normalized concentrations of PCBs and DDT in wastewater from the Los Angeles County Sanitation District Outfalls, and in surficial sediments and muscle tissue of Dover Sole near the Outfalls between 1972 and 1975 (from Young et al, 1977).

better reflect the organic contaminant burdens of the sediments in which they reside, than will demersal fish which have a very active mixed function oxygenese system. In fish there is selective retention of the less-readily metabolized chlorinated aromatics.

4.5.2.1 Field Studies.

Roesijadi et al. (1981) reported the concentrations of mercury in mussels Mytilus edulis in Bellingham Bay, Washington near a chlor-alkali plant. Before 1970, the plant released an estimated 4.5 to 9.0 kg mercury per day to the bay. In 1970, the discharge was reduced to 0.23 kg per day, and in 1974 it was reduced further to 0.05 kg per day. Mercury concentrations in the sediments bagan to decline after 1970 from a high of about 100 ppm with a half-time of 1.3 years. In the mussels, mercury concentrations dropped from about 1.2 ppm in 1970 to 0.6 ppm in 1973 (half, time of three years), to 0.12 ppm in 1978.

Following the Amoco Cadiz oil spill, oysters Crassostrea gigas from the heavily contaminated Aber Benoit and Aber Wrac'h near the spill site contained 540-552 ppm total petroleum hydrocarbons and 38 ppm total resolved polycyclic aromatic hydrocarbons (alkyl naphthalenes-benzo(a)pyrene) nine months after the spill (Neff and Haensly, 1982; Neff, et al., 1985). Body burdens of polycyclic aromatic hydrocarbons in the oysters dropped gradually and erratically to about 10-11 ppm 27 months after the spill. Each winter, winter storms would resuspend contaminated sediments and recontaminate the oysters and other fauna of the estuary. The dominant polycyclic aromatic hydrocarbons in the tissues of contaminated oysters were alkyl phenanthrenes and dibenzothiophenes. Pyrogenic aromatic hydrocarbons (fluoranthene - perylene) also were present in the oyster tissues but did not vary much with time after the spill.

The demersal fish, the plaice Pluronectes platessa was abundant in the estuaries before and after the spill. However, they contained only low concentrations of petroleum hydrocarbons after the spill. Most of the aliphatic and aromatic hydrocarbons in the plaice muscle and liver tissues were biogenic.

The hydrocarbon data demonstrate convincingly the dramatic differences in patterns of petroleum hydrocarbon contamination of oysters and plaice from the same oil-contaminated areas. The oysters contained high concentrations of both pyrogenic and petrogenic aromatic hydrocarbon spanning a wide molecular weight range. They released the accumulated hydrocarbons slowly and were continually recontaminated by hydrocarbons remobilized from contaminated sediment. The plaice on the other hand contained little petrogenic aromatic hydrocarbons. These differences undoubtedly reflect the markedly different capabilities of bivalve molluscs and telost fish to matabolize and activity excrete petroleum hydrocarbons. Interestingly, the oysters exhibited few and relatively minor sublethal biochemical and histopathological responses to the massive petroleum hydrocarbon accumulation, whereas the plaice from the Abers exhibited a wide variety of biochemical and histopathological lesions for at least 27 months after the spill (Neff and Haensly, 1982; Haensly et al., 1982; Stott et al., 1983). The polar metabolites of polycyclic aromatic hydrocarbons are much more reactive than the unmetabolized parent compounds and probably were responsible for the sublethal responses observed in the plaice.

Following the Tsesis oil spill on the Swedish coast of the Baltic Sea, Boehm et al (1982) monitored hydrocarbon contamination in populations of the epifaunal filter-feeding bivalve mollusc Mytilus edulis and the infaunal deposit-feeding bivalve Macoma balthica from the oil spill site for up to one year after the spill. Concentrations of total aromatic hydrocarbons in mussel tissues then decreased gradually to 100 ppm after ten months and then rose again to 440 ppm at twelve months post-spill. At less heavily impacted stations, aromatic hydrocarbon concentrations in mussels returned to background levels within one year. The phenanthrenes and dibenzothiophenes were more persistent in the mussel tissues than the naphthalenes and alkyl benzenes. The alkylated phenanthrenes and dibenzothiophrenes were retained to a much greater extent than the parent (unsubstituted) compounds. A series of trimethyl benzenes, characteristic of the spilled oil, also was quite persistant in the mussel tissues.

Macoma balthica from the spill site accumulated aromatic hydrocarbons to maximum concentrations of 460 to 1700 ppm within two months after the spill. Aromatic hydrocarbons then were released very gradually from the M. balthica tissues to low values of 380 to 1100 ppm in June, eight months after the spill, the bivalves then reaccumulated aromatic hydrocarbons during the summer to concentrations as high as 2,000 ppm in August. The reaccumulation of aromatic hydrocarbons eight to ten months after the spill probably was related to remobilization and transport of sediment-bound hydrocarbons. Hydrocarbon profiles in Macoma tissues resembled those associated with contaminated sediments and sediment trap particles. As in Mytilus, the most persistant aromatic hydrocarbons in Macoma were the alkylated phenantherenes, dibenzothiophenes and benzenes. Thus, as might be expected, the pattern of aromatic hydrocarbon accumulation and release by the filter feeder Mytilus edul[illegible]cted the pattern of hydrocarbon contamination of the water column, while [illegible]attern of aromatic hydrocarbon accumulation and release by the deposit feeder Macoma balthica reflected the pattern of hydrocarbon contamination of the sediments.

Boehm et al (1985) reported similar results following the Baffin Island Oil Spill (BIOS) experiment. The long-term fate and effects of chemically dispersed and undispersed crude oil were compared in an arctic marine environment on the north shore of Baffin Island, Northwest Territories, Canada.

Following the discharge of dispersed oil, total hydrocarbon concentrations in the water column as high as 50 ppm were sustained for the six hour duration of the discharge (Figure 4.16). High concentrations of dispersed oil persisted in the water column of the oiled bay for many hours and eventually spread to nearby bays. The dispersed oil retained an unweathered composition. However, the chemically disprsed oil did not persist long in the water column and sediment contamination by dispersed oil was minimal. Oil concentrations in sediments never exceeded about 6 ppm in the dispersed oil-treated bay. They returned to near baseline level within one year after the spill and then rose again by the second year, possibly through introduction of oil from outside the bay. Total oil concentrations in both the filter-feeding molluscs Mya truncata and Serripes groenlandicus, and deposit-feeding molluscs Macoma balthica and astarte borealis increased to a maximum one day to two weeks after the experimental spill and then decreasd to low, but higher than prespill concentrations 9 months, one year and two yars after the spill. Interestingly, despite the minimal contamination of the sediments by the dispersed oil, the deposit feeders, Macoma and Astarte accumulated the highest concentrations of petroleum hydrocarbons, 800 and 500 ppm respectively. Petroleum hydrocarbon concentrations in the sea urchin Strongylocentrotus droebachiensis did not reach a peak until nine months after the spill (760 ppm), and had dropped to about 50 ppm three months later.

In contrast, following the discharge of the undispersed oil, little oil got into the water column and much of the oil eventually was driven onto the beach by the prevailing winds. The maximum petroleum hydrocarbon concentration in the upper one meter of the water column under the oil slick was 2 ppm. However, low concentrations of petroleum hydrocarbons of 1 ppb or less persisted in the water column of the bay that received undispersed oil for up to two years after the experimental spill, probably due to continual reentry of beached oil into the water. The initial contamination of subtidal sediments by undispersed oil was minimal. Howevr, contamination of subtidal sediments increased gradually to about 400 ppm two years after the experimental spill as oil eroded from the beach.

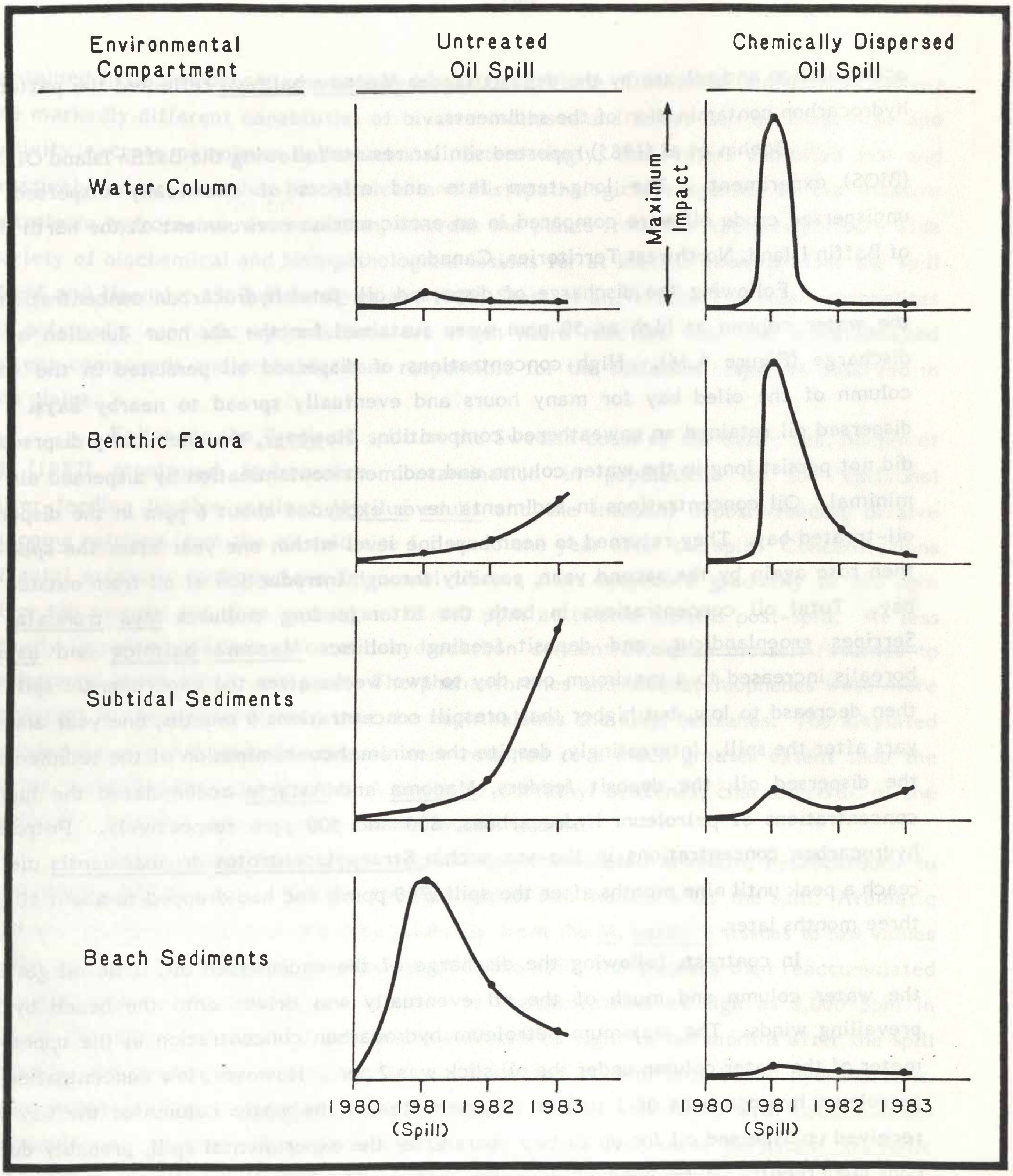

Figure 4.16 Summary of the comparative fates of chemically dispersed and undispersed crude oil during the Baffin Island Oil Spill (BIOS) experiment (from Boehm et al, 1985).

The bivalve molluscs accumulated 100-400 ppm petroleum hydrocarbons within two weeks after the experimental spill. However, they still were contaminated with oil and concentrations actually increased in Macoma and to a lesser extent Mya in the first and second years after the experimental spill. Thus, the undispersed oil was much more persistant than the dispersed oil in the benthic infauna.

4.5.3 Summary.

Laboratory and particularly field studies reveal that heavy metals and monpolar organic contaminants are released relatively slowly from contaminated marine animals following an acute pollution event or cessation of a chronic discharge. However, if recontamination of the environment from persistant pollutant reservoirs, such as intertidal sediments, does not occur, reduction of contaminant body burdens to low near-background levels may require one to a few years. Contaminant release is likely to be most rapid in water column animals, especially very small zooplankton. Benthic and monmigratory dimersal animals are unlikely to release contaminants from their tissues at rates significantly greater than the rate of decrease in contaminant levels in the surficial sediments where the animals live. The exception is polycyclic aromatic hydrocarbons in tissues of animals with a very active mixed function oxygenase system. Such animals may contain low levels of unmetabolized polycyclic aromatic hydrocarbons dispite high levels of contamination of their environment.

4.6 Ecological Recovery-Synthesis

In the numerous investigations discussed above, recovery times ranging from a few days to one hundred years or longer have been reported or predicted following pollutant-mediated disturbance to different marine ecosystems (Figure 4.17). As a general rule, shallow water coastal and inner estuarine ecosystems which are subjected to repeated natural disturbances, such as wave or storm action, ice scour, temperature and salinity extremes, never reach the stable long-lived climax community. Instead, they are maintained in a subclimax seral stage dominated by short-lived opportunistic species. In such ecosystems, pollution is just another disturbance, and recovery is rapid following pollution abatement. In more constant and stable envionments, persisent, long-lived climax or sub-climax communities may develop. These communities often lack resilience

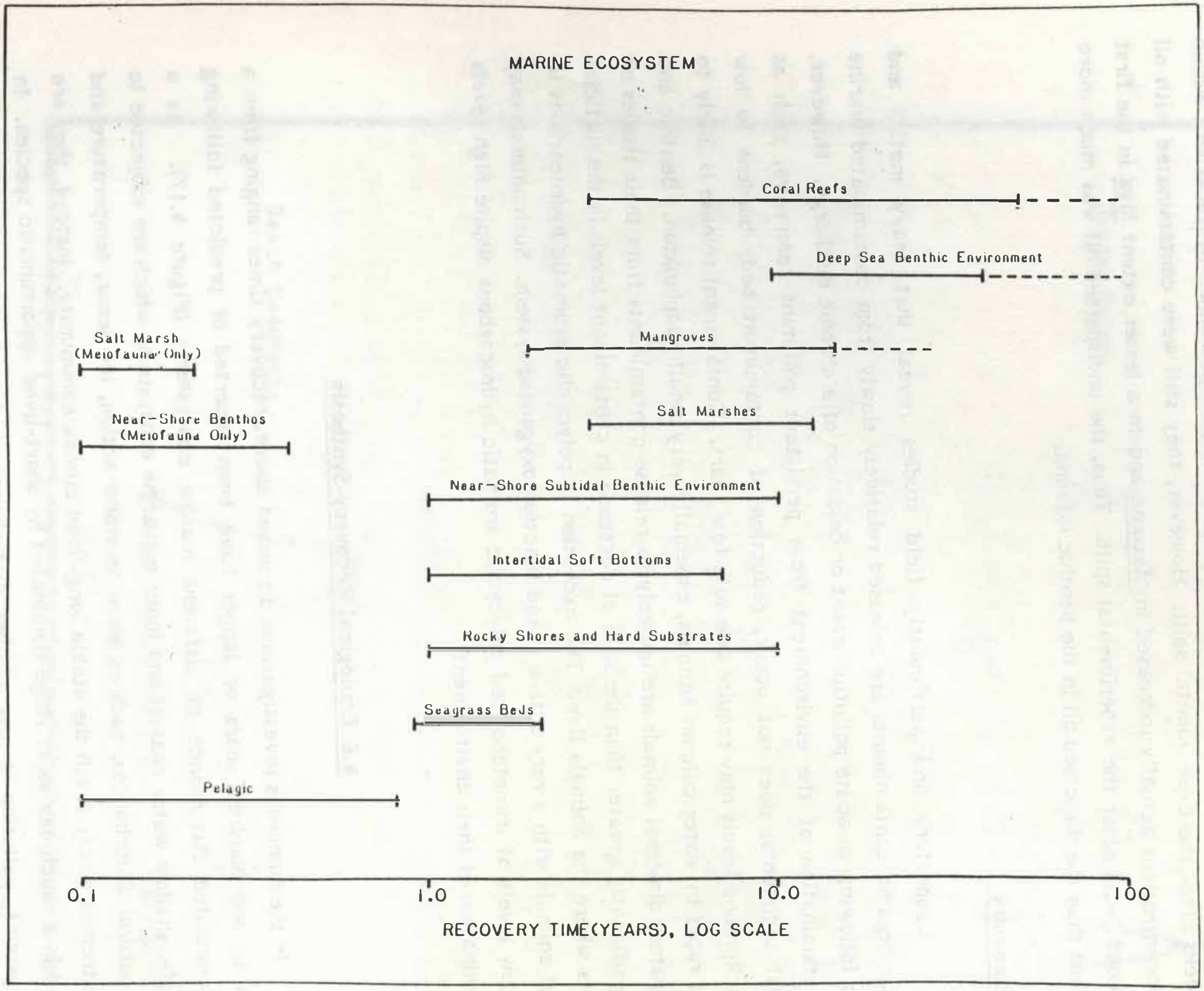

Figure 4.17 Summary of the estimated range of time in years required for ecological recovery of different marine ecosystems or ecosystem components. Dashed lines indicate that recovery may take longer than predicated, but data are not available.

and recover slowly from disturbances, including pollution. There actually have been only a few instances where complete recovery of such ecosystems has been documented. Thus, the actual time required for recovery of these stable long-lived ecosystems is not known with any degree of certainty. This uncertainty is indicated by the dashed time lines in Figure 4.15.

In all ecosystems, the rate of recovery is influenced to varying degrees by the magnitude of the disturbance. This is particularly true in those ecosystems in which the dominant biological entity in the community also represents the structural framework upon which the rest of the community depends. Included in this category are coral reefs, mangrove forests, seagrass beds, and salt marshes. If the pollution or other disturbance destroys or damages only the biota associated with the biological framework, but not the biological framework itself, recovery may be quite rapid. On the other hand, if the framework elements are destroyed, recovery may be protracted or in extreme cases, there may be no restoration of the former community structure. Destruction of marsh grasses, sea grasses, or mangroves may result in large-scale erosion or other alterations in the substrate, rendering the location unsuitable for recruitment and regrowth of the plants. Massive deposition of sediments on a coral reef may render the reef surface unsuitable or unavailable for settlement of coral planulae, resulting in more or less permanent dstruction of the reef. Such systems can be said to exhibit a high degree of hysteresis and moderate to high maleability. That is, the path of recovery in such systems is different from the path of degradation and may well lead to a stable long-lived seral stage quite different from that which existed there before pollution.

Two factors that complicate comparisons of the rate of recovery of different marine ecosystems are differences in the types of parameters used to measure disturbance and recovery, and differences in our ability to differentiate between natural variability in an ecosystem and changes due to disturbance and recovery. For instance, if meiofaunal community structure is used as an index of ecosystem recovery, then recovery of benthic, intertidal, and salt marsh habitats from pollution proceeds rapidly. If regrowth of the dominant marsh grasses is used as the sole measure of marsh recovery, then the process of recovery appears more rapid than if re-establishment of the complete salt marsh community is measured. The latter measure, of course, is the most valid, as well as most difficult, measure of true recovery. This becomes particularly apparent when examining recovery of exposed rocky shores, generally considered very resilient habitats. Quite frequently following a major disturbance such as an oil spill, the rocky shore returns to a normal appearance very rapidly due to massive regrowth of macroalgae.

However, if the grazers have been destroyed, the system is quite unstable, and it may take many years before the more stable pre-spill community structure is re-established and the ecosystem can be said to have recovered.

In many naturally disturbed shallow coastal environments, natural temporal and spacial variability in the community structure make it impossible to detect anything but major pollutant-mediated changes. In such systems, the final stages of recovery are difficult or impossible to differentiate statistically from natural ecosystem variability. In addition, such ecosystems may appear more resistant than they actually are for the same reason.

Several interesting observations emerge from this review. The response to stress hypothesis which originally was proposed and experimentally verified for estuarine benthic communities (Boesch and Rosenberg, 1981; Hyland et al, 1985) seems to be valid as well for a wide variety of other marine and estuarine systems. This hypothesis states that benthic communities found in constant and predictable environments are less stable (resistant and resilient) following unusual disturbances than lower-diversity communities found in more inconstant unpredictable environments. A similar differential stability is seen in such ecosystems as rocky shores, intertidal substrates, salt marshes, coral reefs, and possibly mangrove swamps. Such ecosystems in more exposed locations, more variable natural environmental regimes, or near the geographic limit of normal distribution seem to be more resilient than similar ecosystems in more stable, protected and geographically optimal conditons. Extension of the response to stress hypothesis to different marine and estuarine ecosystems certainly warrants further observational and experimental scrutiny.

The pattern of recovery of disturbed marine and estuarine ecosystems, described for the estuarine benthos by Pearson and Rosenberg (1978) in terms of temporal changes in numbers of species, abundance of individuals and biomass (the SAB curves, Figure 4.1), appears to apply to other ecosystems as well. Similar patterns are seen during recovery of most other major types of estuarine and marine ecosystems examined to date.

The investigations discussed above, in which recovery of coastal ecosystems was documented following cessation or partial abatement of wastewater discharges, demonstrate convincingly that these abatement measures can be extremely effective in initiating recovery and restoration of the receiving water ecosystems. These studies also showed, as might be expected, that there is some relationship between the magnitude of the decrease in polluted wastewater discharge and the rate and degree of subsequent ecosystem recovery. However, two important questions remain: Is there a minimum amount of waste discharge abatement required before some degree of ecosystem recovery is initiated; and is there some acceptable level of wastewater discharge at which more or less complete recovery of the receiving water ecosystem can occur?

The answers to both questions involve consideration of the so-called assimilative capacity of the ecosystems under consideration, which in the present context is equivalent to ecosystem resistance. Obviously, real recovery cannot begin until the rate of waste discharge is decreased to the point where there is a net decrease with time in the concentration of pollutants or other waste-derived impact-causing agents in the ecosystem under consideration. That is, rate of loss of pollutant chemicals from the ecosystem by dilution, wash-out, degradation, and burial must exceed the rate of input of pollutant chemicals to the ecosystems in the waste stream. For pollutants with a high BOD and/or COD, waste flow must decrease to the point where the system can become oxic.

The second question is a more difficult one, and has been the subject of much debate (i.e. Goldberg, 1979; Bascom, 1984). If we make the reasonable assumption that a level of contaminant input which is compatible with complete recovery also is one that will cause no persistent environmental damage, then the question becomes: Is there a level or rate of waste discharge which will cause no environmental damage? Although there have been many claims that discharge to the ocean of such materials as metals in sewage effluents (Bascom, 1982, 1983), waterbase drilling muds from offshore oil rigs (National Academy of Sciences, 1983), and coal fly-ash at the 106-mile dumpsite (ERCO, 1981, 1982) results in minimal and very localized impacts on the marine environment, these conclusions have been questioned. The concern is that there may be insidious long-term impacts of such discharges that we can't anticipate and therefore do not know how to measure. Carefully designed long-term monitoring programs would be required to detect such impacts. What the studies discussed above do demonstrate is that, depending on the nature of the receiving water ecosystem, complete cessation of the waste discharge may not be required before the system can recover to an advanced level of productivity and diversity not greatly different from the pre-discharge condition.

However, as stated earlier, there is a notable lack of information about the recovery of marine ecosystems of any kind following disturbance by an acute or chronic input of highly persistent and toxic synthetic organic chemicals such as polychlorinated biphenyls and some pesticides. Ecosystem damage from such discharges may persist for long after the discharge is terminated. For instance, source control of DDT in waste water discharged by the Los Angeles County sewage treatment plant at Palos Verdes, California was initiated in 1970 (Mearns and Young, 1983). As a result, DDT mass emissions in the combined municipal wastewaters of southern California decreased from 21,700 kg in 1971 to 218 kg in 1983 (Schaefer, 1984). Nevertheless, sediments, benthic invertebrates, and demersal fish from the vicinity of the Palos Verdes outfall continued to contain very high concentrations of DDT and DDT metabolites (Brown et al., 1984). Contamination of the fish is sufficient to cause reproductive impairment and other sublethal effects. Thus, in this case, abatement of a pollutant discharge has not yet, after 13 years, resulted in recovery of the impacted eocsystem. Recovery of benthic ecosystems from persistent pollutant impacts may require burial of the polluted sediments by deposition of several centimeters of clean sediments. Sufficient burial, in the absence of bioturbation, may require a decade or more in all but the most active depositional basins (Cato et al., 1980).

5. RESIDENCE TIMES AND RECOVERY RATES

Perhaps the only way to synthesize our determinations of contaminant residence times in various systems and our estimation of ecosystem recovery rates is to relate the residence times of contaminants in sediments to benthic recovery rates. The interrelated, simplified, conceptual process can be rationalized as follows:

> Given a system in equilibrium (i.e., steady-state with respect to contaminant inputs and outflows), a certain sediment contaminant level, or a linear (or non-linear) combination of contaminant levels, has created a stressed or degraded benthic community structure. Supposedly, benthic recovery can begin at some lesser level of contaminant or linear combination of contaminants. If the contaminant loading of the system is modified so that the new calculated equilibrium sediment contaminant concentration, or combination of contaminants, is below the "threshold of recovery" (TOR), then recovery can begin. The overall process conceived is then a sequence of contaminant lowering and recovery.

According to our calculations, the residence of sedimentary contaminants (T_{sed}) in the Puget Sound Main Basin or the Lower Bay Complex is 10 years. According to the recovery graph for coastal (Figure 4.17), recovery time (RT) of the system will be approximately 1-10 years. Thus, the time from decrease in mass loading (i.e., a new loading that results in a TOR value in sediment) to "recovery" of the benthic community will range from approximately 11-20 years (i.e., Total Recovery Time (TRT) = $T_{sed(TOR)}$ + RT. Note that:

1. This conceptualization depends on a linear process of contaminant lowering and recovery, and thus represents a worst case. If recovery begins as soon as mass loading decreases, then TRT = RT); and
2. The approach relates to the recovery from a steady-state, equilibrium mass loading (e.g., levels of metals or PCB in the sediments of the Christiaensen Basin or Lower Bay Complex), and not to events such as acute oil spill impacts. In areas where oil from spills is well-mixed into the sediment mixed layer such as in the Abers of Brittany, France, following the Amoco Cadiz spill, the approach would hold.

A similar approach can be taken to conceptual recovery from body burdens of chemicals. Let's say the criteria for recovery are taken to be, for example, a regulatory limit (e.g., FDA limits for consuming fish or shellfish). Sediment contaminants and benthic animal contaminant loads are assumed to be in equilibrium according to Karickoff (unpublished):

$$\frac{C_{x\ (sed)}}{\text{TOC}} = K \frac{C_{x\ (animal)}}{\text{Total Lipid Content}}$$

where K is a constant dependent only on the partitioning behavior (i.e., octanol-water partitioning) of the chemical. A similar construction can be presented for metals:

$$\frac{C_{x\ (sed)}}{\text{Metal Oxide Concentration}} = K \cdot C_{x\ (animal)}$$

Note that both of these areas are subject of intensive research as part of EPA Sediment Quality Criteria initiative (C. Zarba, EPA Criteria and Standards Division).

Any changed mass loading resulting in a sediment concentration that, in turn results in an equilibrium concentration within the animal that is below the regulatory limit, defines the recovery sequence. The TRT value would equal the T_{sed} plus the time to achieve equilibrium in the animal or the integrated rate of depuration.

Perhaps the most direct regulatory tie-in between mass loading, (T) and recovery relates to determined sediment quality criteria values, such as those discussed in Section 3.6. This discussion took into account: 1) published EPA Water Quality Criteria; 2) a best estimate for the partitioning of solid phase sediment contaminant concentration and sediment pore water; and 3) the relationship of calculation pore water levels to water quality criteria values. (P. Boehm, Battelle, is developing an Index of Sediment Contamination for NOAA based on this concept initially put forth by Pavlou and Weston; 1984).

A summary of recent approaches to determining "Sediment Quality Criteria" is presented below. Once these regulatory approaches are evaluated and actual "criteria" are established, a regulatory basis for defining the degradation-recovery threshold and, hence, the recovery time will be established.

The Environmental Protection Agency (EPA) has, during the past several years, begun to focus on establishing "safe" levels of toxic organic compounds and trace elements in sediments. Approaches to establishing sediment quality criteria, by which bottom sediment contaminant level could be judged as safe or unsafe (i.e., equivalent to levels deemed to indicate "recovery") have taken several different forms:

The Background Approach

Through this methodology, sediment criteria would be established with reference to measured contaminant concentrations in sediment from a specified reference location at which contamination is considered to be very low. Rather than attempting to address the difficult toxicological questions inherent in other approaches, this approach sets limits in relation to contaminant levels in reference areas. The reference area could be adjacent to the study area or could represent pre-industrial levels obtained through the analysis of deep core samples. If background levels are non-detectable, as would be the case for synthetic organics, one could take the state-of-the art analytical detection limits as the pristine background level. The criteria would be established at some permissable level of enrichment above background. The "safe" enrichment level would be based on sparse toxicological data or some arbitrary level of enrichment.

Criteria based on a background approach have been adopted by EPA Region X, as they were forced to set criteria vis-a-vis dredged material disposal in Puget Sound. The advantages of such an approach are:

- As the toxicity of individual sediment-associated contaminants is not known, some sort of background approach is the only present way to establish interim chemical criteria.
- Data on contaminant levels in sediments are widely available.
- In terms of establishing an Index of Degradation, an end point can be set against which data can be indexed.

The disadvantages of this approach are:

- The background approach is not based on any rigorous technical information and, therefore, is subject to much controversy.
- The sediment criteria and the Index would be highly site-specific, determined by the arbitrary choice of background site or depth strata.
- The background approach to establishing sediment criteria and, hence, the Degradation Index may be overly conservative and overly restrictive in a regulatory sense.

The Direct Water Quality Approach

Through this approach it is hypothesized that the interstitial waters are basically an extension of the overlying water column, but better represent the true exposure levels of organisms to bioavailable contaminants. Through this methodology the end point in any indexing or sediment quality criterion is the EPA established water quality criteria. Contaminant concentrations in interstitial waters are measured and compared directly with these laboratory-derived criteria levels. This approach assumes that sediment-associated contaminants are not available and, hence, have no potential effects through the direct ingestion of sediments. Although this point needs clarification and further study, it does appear that there is potential for increased contaminant uptake by certain marine organisms (e.g., polychaetes, deposit-feeding bivalves directly from sediment).

A real problem in the application in such an approach is the lack of an adequate data base on toxic contaminants in interstitial water and the limited methodology for obtaining valid, unaltered interstitial water samples for chemical determinations.

The appeal of this approach to establishing the sediment contaminant index is twofold.

- Water quality criteria values or endpoints for many contaminants are available and are based on widely-used methodology.
- The approach is applicable to a wide array of contaminants as it is based on direct measurements.

The disadvantages are:

- The index and/or criteria can't be developed for those contaminants for which water quality criteria data are lacking.
- Sediment ingestion and contaminant uptake routes are not considered.
- Direct interstitial water contaminant measurements are lacking and are difficult to make (but not impossible).

Equilibrium Partitioning (Sediment-Biota)

Through this methodology a permissable body burden of a contaminant is established based on correlation of contaminant body burdens and toxicity or reduced fecundity. Thermodynamic equilibrium relationships between sediment contaminant levels and body burdens are established, and permissable sediment contaminant levels are back-calculated. In this approach the endpoint is the concentration of contaminant in the sediment which, at equilibrium, ensures that the body burden limits are not exceeded.

This approach has been advocated by EPA-Narragansett and the U.S. Army Corps of Engineers. As described by Peddicord (unpublished), the sediment-biota equilibrium approach relies on the following equation:

$$\log S = (\log C_{tx} - 0.28) + \log TOC$$

where S is the sediment endpoint (permissable level), 0.28 is an empirically-derived consant, C_{tx} is the permissible tissue concentration on a lipid weight basis, TOC is the total organic carbon content of the sediment on a mass fraction basis. Only hydrophobic or neutral organic compounds are presently amenable to this approach (e.g., PCB). Metals and water soluble organics are not considered.

This approach is based on the fugacity approach of Karickoff and MacKay wherein equilibrium is defined at the point at which the driving forces between organic pools in sediment (TOC) and animals (lipids) are equal. The distribution coefficient between sediment and biota is considered to be a constant and independent of the compound being considered. Two other inherent assumptions are that all lipids have similar affinities for hydrophobic or neutral organics, and the organics are predominantly associated with the lipid fraction.

The permissible level of contaminant in tissues is not presently toxicologically based. It must be recognized that there is no direct relationship between toxicity and body burden because it cannot be assumed that animals normally acquire contaminants until they die. Only FDA limits are available for a limited number of contaminants and do not serve as biological endponts vis-a-vis ecosystem degradation.

Thus, the advantages to this approach are:

- If permissable tissue contaminant levels could be developed from burden/effect studies, then direct implications of sediment contaminant levels would form the basis of the criteria development and index formulation.

The disadvantages are:

- The present state of knowledge of burden/effect relationships is poor. FDA limits, even if they were set for many contaminants, (which they are not) have little relevance to environmental quality.
- Little is known about the variation in bioaccumulation factors with species or lipid composition.
- Much more intensive research is required to establish "endpoints".

Sediment Bioassay Approach

There is a certain attractiveness to the approach of determining safe levels of contaminants in sediments by exposing animals to contaminated sediments and measuring mortality, sublethal effects or bioaccumulation. Indeed, through a 10-day exposure period, dredged material disposal in ocean waters has been regulated by such laboratory bioassays.

Sediment bioassays could potentially be used as aqueous bioassays were in setting criteria. Clean sediments could be spiked with known amounts of individual compounds and used in bioassays. The difficulty arises from the fact that any actual toxicity of natural sediments cannot be directly attributed to chemical causative agents, and developing such relationships in the laboratory through spiking of combinations of contaminants to examine the additivity of toxicities would be a long process. Different sediment types (grain size, mineralogy, major element levels such as iron or total organic carbon) would have to be tested as the bioavailability/toxicity of chemicals would be related to the physical-chemical state of the contaminant-different animals would have to be tested.

As causative mechanisms for sediment toxicity are not known, setting an Index endpoint (degraded=1) would be very difficult.

The advantages of establishing an Index of sediment toxicity would be:

- o The Index would account for all possible routes of uptake.
- o The analogy with the widely accepted aqueous bioassay approach to setting water quality criteria is attractive.

Disadvantages relate to the facts that:

- o The causative agent(s) would not be known so one would have to assume that all contaminants acted in concert.
- o Standardized techniques and actual data on the relation of contaminant levels to sediment toxicity are not available.
- o A significant amount of careful laboratory work would have to be carried out and sediment spiking would have to be demonstrated to be equivalent to natural sediment contaminants.

Sediment-Water Equilibrium Partitioning

Using this formulation, the concentration of contaminant in sediment would be related to the level in interstitial water through equilibrium partitioning relationships. Once calculated (rather than measured as is the case for the Direct Water Quality Approach). This interstitial water value would be compared with existing EPA Water Quality Criteria (i.e., indexed through a ratio). This formulation is based on the assumption that when water quality criteria are applied to interstitial water, adequate protection for infaunal animals is afforded.

Again, as in the Direct Water Quality Approach, the use of these relationships enables a large toxicological data base to be used without additional research.

The advantages of this approach may be summarized as follows:

- o Use of a large existing toxicological data base developed in setting water quality criteria enables sediment criteria to be established, against which existing contaminant levels can be indexed for those compounds and elements for which water quality criteria have been established.

- A firm theoretical basis exists for this approach which may be refined on the basis of direct laboratory studies and on field verification.

The disadvantages of the approach are:

- Where water quality criteria are non-existent for organic compounds or metals, certain contaminants can't be indexed.
- Direct ingestion of contaminated sediment is not taken into account.
- Strictly speaking, contaminant equilibrium may not always exist between sediment and interstitial water, but in any event represents a maximum level of exposure.

These areas are all under study at present. Once sediment quality criteria are established, "recovery" can be defined in a regulatory sense by these values. Thus, one can calculate a mass loading which will result in sediment concentrations below the sediment quality value (a contaminant-specific, or composite value). The recovery time then should just be the T_{sed} value, given the decreased mass loading that results in a sediment contaminant value less than the sediment quality criterion.

Obviously, the integration of the Residence Time and Recovery concepts is not straight-forward and progress in this area is probably going to be achieved most rapidly from guidance from the U.S. Environmental Protection Agency's sediment quality criteria development process.

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