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Ocean Disposal of Municipal Wastewater: Impacts on the Coastal Environment

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Edited by
Edward P. Myers
National Oceanic and
Atmospheric Administration

Elizabeth T. Harding
Editorial and Production Manager
MIT Sea Grant College Program

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KEY TO ABBREVIATIONS

8

Public Health Consequences of Coastal and Estuarine Pollution: Infectious Diseases

Victor J. Cabelli
Morris A. Levin
A.P. Dufour

INTRODUCTION

Historically, the earliest requirements for the quality of coastal and estuarine waters were directed against the potential for infectious disease among the users of these resources, specifically that resultant from the contamination of the waters with pathogenic microorganisms present in human fecal wastes. Furthermore, the decrease in reported outbreaks of infectious disease in the United States associated with the use of fecally polluted coastal and estuarine waters for food, and the limited number of outbreaks due to recreational use (Cabelli, 1977, 1978a; Verber, 1981), indicate that existing water quality guidelines and standards have been largely effective. This has been so in spite of the many different infective agents which can be recovered from sewage or sewage impacted waters (McKee and Wolf, 1963; Melnick, 1976; Feachem et al., 1978). However, outbreaks of an acute, infectious gastroenteritis continue to occur from transmission through freshwater recreational activity, shellfish consumption and drinking water (McCabe, 1977; Centers for Disease Control, 1982a; Verber, 1981a); and a single group of viral agents, the Norwalk-like viruses, has increasingly been shown to be the etiological agents (Appleton and Pereira, 1977; Murphy et al., 1979; Gerba, 1981; Baron et al., 1982; Gunn et al., 1982; Kaplan et al., 1982). Furthermore, more recent information clearly shows that many "sporadic" cases of this disease do occur consequent to swimming in sewage polluted marine waters (Cabelli et al., 1979, 1982); these however, generally are not reported.

Six outbreaks of gastrointestinal disease associated with swimming in freshwater were reported to the Centers of Disease Control in 1980 (four due to shigellae and two of unknown etiology); and, of these, five came from waters which met the current USEPA guidelines (Centers for Disease Control, 1982a). These

findings not only raise the question of the adequacy of the existing microbiological guidelines but also the conditions under which these somewhat benign gastroenteritis could be replaced by more serious diseases such as infectious hepatitis and typhoid fever. In fact, swimming- or shellfish-associated outbreaks of shigellosis (Rosenberg et al., 1976; Centers for Disease Control, 1982a, infectious hepatitis (Portnoy et al., 1975), gastroenteritis (Kaplan et al., 1982; Gunn et al., 1982; Baron et al., 1982) and even cholera (Blake et al., 1980) and cholera-like disease (Wilson et al., 1981) but not typhoid fever (Verber, 1981) have occurred in the United States within the past decade.

If, for no other reason, the common source cholera and cholera-like disease outbreaks are important as a reminder that: 1. exotic infectious agents can be (and probably are being) introduced into our aquatic environments as a consequence of commerce with and travel to countries in which these agents are endemic; 2. it is our coastal and estuarine waters (including the Great Lakes) which are most vulnerable to such contamination from discharge of sewage (frequently untreated) from seagoing vessels; 3. existing water quality guidelines and monitoring programs based upon fecal indicators are by their very nature largely ineffective as indices of the health risks from such "relatively small", non-point, intermittent sources of pollution or from introduced or native autochthonous agents, such as the 01 and non-01 cholera vibrios; and 4. that one barrier to secondary transmission of these agents in coastal or even inland communities has been and continues to be, the "safe" disposal of domestic sewage.

Moreover, it is now obvious that:

-- Existing guidelines and standards, even when properly applied in appropriate situations, cannot and do not assure "complete safety", but rather some risk of infectious disease which was detectable at the time the guideline was formulated or is unknown because the guideline was largely based on attainability;

-- measurable risks of infectious disease are being accepted under existing health effects guidelines and standards for the quality of coastal and estuarine waters;

-- the data in support of the existing microbiological guidelines for shellfish-growing and recreational water quality guidelines and standards were neither developed nor can be used in the context of risk analysis; and

-- ecologic, economic, and social factors will have to be considered if guidelines and standards are to be developed based on the acceptability rather than the detectability of risk.

Finally, there are other aspects of the infectious disease potential from the pollution of coastal waters which have not received enough attention. Included are diseases among ecologically and economically important marine fauna, and health problems which could be generated by new or expanded technologies (the dissemination of organisms with multiple antibiotic resistance or recombinant DNA, thermal pollution, etc.).

One objective of this chapter is to review the state of knowledge on how pollution affects the risk of infectious disease in people who use coastal and estuarine waters for food and recreation. The second is to examine current measures that would bring these risks within acceptable limits. A third is to identify data requirements and research needs.

TRANSMISSIBLE DISEASES AND THE SOURCES FOR THEIR ETIOLOGICAL AGENTS

Potentially, a large number and wide variety of infectious and biotoxigenic agents can cause human disease when polluted coastal and estuarine waters are used as sources of food and recreation. Included are species of viruses, bacteria and fungi, protozoan and metazoan parasites, and biotoxins produced by several species of bacteria and dinoflagellates. In actuality, only a few of these infectious agents have been implicated in cases or outbreaks of disease clearly associated with such activities. These diseases, their etiological agents, and the sources of the agents, and the routes of transmission are shown in Table 8.1. The most common of these diseases in the United States at present is an acute, infectious gastroenteritis (Cabelli, 1978a, 1980; Verber, 1981), potentially caused by any of several bacterial and viral agents. Characteristically, bacterial pathogens could not be isolated from the ill individuals, although recent information would suggest that a few of the outbreaks

Table 8.1 Microbial, Infectious and Biotoxigenic Diseases Transmitted by the Recreational and Shellfish-borne Routes.

Agent	Disease	Route of transmission[a]		Source of the agent[a]			
				Feces		Sewage	Water
		Rec.	Shell.	Human	Animal		
Bacteria							
Salmonella sp.	Typhoid and paratyphoid[b] fevers, salmonellosis	+	+	+	+	+	-
Shigella sp.	Bacilliary dysentry	+	-	+	-	+	-
Pseudomonas aeruginosa[c]	Otitis externa, skin infections[d]	+	-	+[m]	+[m]	+	+[n]
Aeromonas hydrophila[c]	Infected wounds	+	-	+[m]	+[m]	+	+[n]
Vibrio vulnificus	Infected wounds[e]	+	-	-	-	-	+
Vibrio parahaemolyticus	Gastroenteritis[f]	-	+	+[m]	-	+[o]	+[n]
Vibrio cholerae (01)	Cholera[g]	-	+	+[o]	-	+[o]	?
Non-01 *V. cholerae*	Cholera-like disease	-	+	+[m]	-	+[o]	+
Leptospira sp.[c]	Leptospirosis (Weil's disease)	+	-	+[m,p,o]	+[p]	-	-
Campylobacter sp	Gastroenteritis	-	+[h]	+[m]	+	+[o]	-
Clostridium botulinum	Food poisoning (botulism)[i,j]	-	+	-	+	-	+
-	Gastroenteritis[k]	?	+	+[q]	+[q]	+[q]	+[q]
Mycobacterium marinum	Infected wounds	+	-	-	-	-	+
Viruses							
Hepatitis A.	Infectious hepatitis	+	+	+	-	+	-
Norwalk-like	Acute, infectious non-	+	+	+	-	+	-
Human rotavirus	bacterial gastroenteritis	+[l]	?	+	-	+	-
Adenovirus, types 3 and 4	Pharyngo-conjunctival fever[c]	+[r]	-	+	-	+	-
Coxsackievirus	Pleurodynia, others	+[s]	-	+	-	+	-

Agent	Disease	Route of transmission[a]		Source of the agent[a]			
		Rec.	Shell.	Human	Animal	Sewage	Water
Protozoa							
Naegleria sp. (pathogenic)	Primary amoebic meningoencephalitis	+	-	-	?[t]	+	+
Bird shistosomes	Swimmer's and clam digger's itch	+	-	-	+	-	-
Algae							
Gonyaulax sp.	Paralytic shellfish[i] poisoning	u	+[v]	-	-	-	+

[a]Water = multiplies in environmental waters; Rec. = recreational (swimming); Shell. = shellfish consumption; Animal = lower animals. All agents in human feces also assumed to be present in sewage.

[b]Rare for recreational route; none for shellfish route since 1959. [c]Primarily in freshwater.

[d]Primarily from hot tubs and whirlpool baths. [e]Less frequently by *V.parahaemolyticus* and *V. alginolyticus* strains.

[f]Specific toxigenic strains. [g]Only since 1973 by 0-1 strains. [h]Only two shellfish-borne outbreaks.

[i]Toxin in food. [j]Problem in food processing. [k]Possibly caused by enteropathogenic *E.coli* and *A. hydrophila, Yersinia enterocolitica* and the protozoan, *Giardia lamblia;* much less frequent than viral form.

[l]Inferred from prospective epidemiological bathing beach study. [m]Other source more important.

[n]Density probably influenced by nutrient loading. [o]Not a significant source in US. [p] Urine not feces.

[q]Varies with potential agent. [r]Characteristically associated with use of swimming pools, not natural water bodies.

[s]Two questionnable outbreaks in freshwater. [t]Seed from birds suggested. [u]Upper respiratory symtoms from other less toxigenic dinoflagellates, *Prorocentrum* sp.

[v]Shellfish poisoning is also due to other dinoflagellates with different toxins.

In addition, there a number of other pollution associated agents which could cause swimming or shellfish associated disease, although there is *no evidence* that they have done so. They include the bacteria *Staphlococcus aureus, Klebsiella, Clostridium perfringens;* most of the enteroviruses; amoeba such as *Entamoeba histolytica* and a number of exotic multicellular parasites.

may have been due to V. parahaemolyticus, enteropathogenic E. coli and Campylobacter species. However, it has been long suspected that in most of the outbreaks the agent was a virus. Two groups of viruses have been shown to be the etiological agents of an acute, infectious, non-bacterial gastroenteritis, the human rotavirus and the Norwalk-like viruses (Blacklow et al., 1972; McNuty, 1978). As noted above, the latter group of viruses has been identified increasingly as the etiologic agents in outbreaks of shellfish-, swimming-, and drinking water-associated gastroenteritis; and the former is a prime candidate for the gastroenteritis reported in the prospective bathing beach epidemiological studies (Cabelli, 1980).

The most serious disease in terms of long-term health effects is infectious hepatitis. However, there have been no reported outbreaks of this disease associated with swimming in marine waters; and, although there have been a number of major shellfish-borne outbreaks of infectious hepatitis in the US over the past three decades (Verber, 1981), there have been only two small outbreaks reported in the period 1974-1979. Nevertheless, under certain conditions (notably, epidemics in the population whose fecal wastes pollute the waters), the potential for infectious hepatitis or some of the other diseases shown in Table 8.1 could become realities. In addition, a number of infectious diseases could be, but are not, spread via the water-borne routes in the United States because the agents have not been introduced into the population or environment in sufficient numbers, or have been largely eliminated therefrom. This potential can become real as it has in several shellfish-associated outbreaks of classical cholera along the Gulf Coast since 1973 (Blake et al., 1980; Centers for Disease Control, 1981). It remains to be resolved whether these classical (01) strains of Vibrio cholerae, like the non-01 strains, have found an ecologic niche in coastal and estuarine waters wherein they multiply. In any event, both the 01 and the non-01 strains have been isolated from coastal and estuarine waters, bottom sediments, and shellfish along the Gulf Coast of the US (Madden et al., 1982).

The largest number of these infective agents by far have their source in human and, to a lesser extent, lower animal fecal wastes. They reach marine environments directly from point source disposal of separate and combined wastewater discharges, from piped or barged disposal of sewage sludge and, to a much lesser extent, from non-point sources such as urban and rural storm water run-off. The infectious agents may

also be carried into estuarine and coastal waters indirectly by rivers and streams which receive such discharges, or by the translocation of dredge spoils.

A second potential source is sewage itself in that some infective agents, which are infrequently isolated from "normal" human feces or are present in very low densities (Dufour, 1977), can be regularly recovered from domestic sewage and certain industrial effluents at rather high densities (Dufour and Cabelli, 1976; Miescier and Cabelli, 1982). The four agents in question, Pseudomonas aeruginosa, Aeromonas hydrophila, Acinetobacter calcoaceticus, and Klebsiella species, are opportunistic pathogens. The first two have been responsible for swimming-associated illness in fresh and brackish waters (Hoadley and Knight, 1975; Hanson et al., 1977). Although all four have been isolated from coastal and estuarine waters, their densities decrease with increasing distance from known pollution sources (Cabelli et al., 1974, 1976; Dufour, 1977; Lacroix, personal communication) and there is no evidence that they multiply in coastal waters. Members of the genera Pseudomonas and Aeromonas also have been implicated in disease among freshwater fauna (Sniezko, 1970).

A third source of infective agents for man or marine fauna is the marine environment itself in situations where the agent multiplies in the water, irrespective of the source of the inoculum. In addition to the non-01 cholera vibrios responsible for shellfish-borne outbreaks of cholera-like disease, Vibrio parahaemolyticus (marine food-borne infections) (Fujino et al., 1974), V. vulnificus (wound infections) (Ryan, 1976), and certain toxigenic dinoflagellates (Fortuine, 1975) fall into this category, as do most of the agents of faunal disease (Sniezko, 1970).

CRITERIA, GUIDELINES AND STANDARDS

Definitions of health effects criteria, guidelines and standards and their relationships to each other have been set forth in an earlier publication as has the need for the development of criteria which can be used in the context of risk analysis (Cabelli, 1978b).

A health effects, water quality criterion developed for use with a microbial indicator system is defined as a quantifiable relationship between the density of the indicator in the water and human health risks involved in uses of the water for food and recreation. Above all, it is a relationship upon which a judgment can be made. A water quality guideline

derived from the criterion is the judgment made, that is, a suggested upper limit for the density of the indicator in the water which is associated with unacceptable health risks. The concept of acceptability implies that there are social, cultural, economic, and political as well as medical inputs to the derivation and that these may vary in time as well as space. A water quality standard obtained from the criterion is a guideline fixed in law. Since the criterion is a dose-response type relationship, once the acceptable level of risk is determined, it can be extrapolated to a given indicator density.

HEALTH EFFECTS WATER QUALITY INDICATORS

There have been a number of reviews on health effects water quality indicators (Hoadley and Dutka, 1977; Cabelli, 1978b; Pipes, 1978). In general, the proposed indicators can be placed in one of four categories: 1. fecal indicators; 2. enteric pathogens whose source is exclusively fecal; 3. human pathogens whose source is the aquatic environment because they can multiply therein; 4. environmental parameters which index the ability of these pathogens to multiply in the water.

A list of potential indicators is presented in Table 8.2 Historically, enteric diseases such as cholera, typhoid fever, and dysentery were of major concern; fecal contamination of water, primarily drinking water, was the source of the agent; and fecal indicators were used to index the health hazards involved. With reference to the risk of infectious disease via the recreational and shellfish routes, fecal contamination remains of paramount importance. The fecal indicators most commonly used are the ones suggested at the turn of the century, E. coli, Streptococcus faecalis and Clostridium perfringens, or groups which include these species. There are a number of constraints on the use of fecal indicator systems and on the interpretation of the results obtained. These, along with the advantages and disadvantages of each indicator system, have been discussed in earlier publications (Cabelli, 1977; 1978b) and will be discussed later in this review.

Enteric pathogens are rarely used as water quality indicators in the United States for a number of reasons, including the large number of potential agents, problems in methodology, their fluctuating densities, and the poor data bases available for extrapolating the risk of disease from measurements of their densities in environmental waters. On the other

Table 8.2 Potential Health Effects Water Quality Indicators for Recreational and Shellfish Growing Waters.

Bacteria

- Coliforms
- Fecal coliforms
- *Escherichia coli*
- Enterococci
- *Clostridium perfringens*
- Bifidobacteria
- *Pseudomonas aeruginosa*[a]
- *Aeromonas hydrophila*[a]
- *Vibrio parahemolyticus*[a]
- Non-01 vibrios[a]
- *V. cholerae*

Viruses
- Coliphage

Fungi
- *Candida albicans*

Chemicals
- Coprostanol

Algae
- Gonyaulax sp.[a]

---

[a]Aquatic microorganisms responsible for cases or outbreaks of waterborne disease. Nutrients whose levels in the water correlate with their densities also are potential indicators. Enteric pathogens not included; salmonellae and enteroviruses have been suggested.

hand, with those pathogens whose source is the marine environment, we have no choice but to enumerate the pathogen itself or to index its potential presence from measurements of the conditions and/or nutrient levels which lead to its multiplication.

TARGET AREA CRITERIA

Recreational Use

One of the most important justifications for pollution control programs for coastal and estuarine waters is the maintenance of the recreational resource. This is particularly true near our large, coastal, metropolitan areas where swimming and beach-related activities are among the few forms of outdoor recreation available to the public. From a 1960 survey, the Outdoor Recreation Resources Review Commission found that swimming was second only to "walking for pleasure" in outdoor recreational activities and was likely to rank number one by the turn of the century. A later survey in 1965 by the Bureau of Recreation (BOR) concluded that swimming was expected to be the most popular outdoor activity by 1980. BOR found that outdoor swimming "participation occasions" increased 44 percent between 1960 and 1965, while the population of individuals 12 years old and older increased 8 percent. Between 1965 and 1980, BOR expected that swimming would increase 72 percent while the population was expected to increase 29 percent. Expressed in terms of individuals, BORs 1965 survey found that 49 percent of the population 12 years old and older went swimming outdoors that year, an increase of 15 percent since 1960 (National Technical Advisory Committee (NTAC), 1968).

Exisiting criteria and guidelines. Heretofore, there have been no health effects, water quality criteria for recreational waters in the sense used here. However, there are guidelines and standards which have been developed and promulgated against the possibility of a swimming-associated infectious disease. In the United States, these usually have been expressed as limiting concentrations of total coliforms or fecal coliforms (McKee and Wolf, 1963; Mechalas et al., 1972). The most commonly used guidelines and standards in the United States are a geometric mean fecal coliform density of 200 per 100 ml of the water and a density of 400/100 ml not to be exceeded in more than 10 percent of the samples examined (Table 8.3). These guidelines were originally developed by the National Technical Advisory Committee to the Federal Water Pollution Control Administration (1968) and are

Table 8.3 Total and Fecal Coliform Standards[a] for Primary Contact Marine (including Great Lakes) Recreational Waters

State	Year Rev	Water Type	Total Coliform Limit per 100 ml Average	Total Coliform Limit per 100 ml Percentile	Total Coliform Limit per 100 ml 1 Sample	Fecal Coliform Limit per 100 ml Average	Fecal Coliform Limit per 100 ml Percentile	Fecal Coliform Limit per 100 ml 1 Sample
Alabama	77	SW[b]				LM[c] 100		
Alaska	79	SW				Mean 20	90%≤40	
California	78	SW	Ave 1000	80≤1000	≤ 10,000	EPA[e]	EPA [e]	
Connecticut	76	SW	Med[d] 700	90% ≤ 2300		EPA[f]	90% ≤ 500[f]	
Delaware	75	SW				EPA		
Dist. of Columbia	Pro[g]	SW				EPA	EPA	
Florida	74	SW	LM 1000	80≤1000	≤2,400	EPA	EPA	≤800
Georgia	77	SW				LM 100		
Hawaii	74	SW	Med 1000	90%≤2400		EPA	EPA	
Illinois	75	FI				EPA	EPA	
Indiana	78	FW				EPA		≤400[h]
Louisiana	77	SW				EPA	EPA	
Maine	77	SW	Med 70	90%≤230		Med 100	90%≤200	
Massachusetts	78	SW	Med 700	90%≤1000				
Maryland	74	SW				EPA	EPA	
Michigan	73	FW				EPA		
Minnesota	77	FW				EPA	EPA	
Mississippi	77	SW				EPA	EPA	
New Hampshire	77	SW			240			

State	Year Rev	Water Type	Total Coliform Limit per 100 ml Average	Percentile	One Sample	Fecal Coliform Limit per 100 ml Average	Percentile	One Sample
New Jersey	74	SW				EPA		
New York	74	SW	Med 2400	80%≤5000		EPA		
North Carolina	77	SW				EPA	80%≤400	
Ohio	78	FW				EPA	EPA	
Oregon	Pro	SW						
Pennsylvania	Pro	SW				EPA		
Rhode Island	77	SW	Med 700	90%≤2300		Med 50[c]	90%≤500[c]	
South Carolina	77	SW				EPA	EPA	
Texas	76	SW				EPA	EPA	
Virginia	77	SW				EPA	EPA	
Washington	77	SW				Med 14	90%≤43	
Wisconsin	78	FW				EPA	EPA	
Puerto Rico	76	SW				EPA	80%≤400	
Virgin Islands	73	SW				LM 70		
Trust Territory	73					EPA	EPA	
American Samoa	73					Ave 100	90%≤200	
Guam	76					Ave 200	EPA	

[a]Does not include caveats, special requirements and limitations, etc. of various states;

[b]seawater (estuarine and coastal); [c]log mean; [d]median; [e]US Environmental Protection Agency guidelines, geometric mean not to exceed 200/100 ml and 90% of samples not to exceed 400/100 ml;

[f]guideline; [g]proposed; [h]in one month; [i]freshwater (included for states bordering on the Great Lakes).

essentially those now recommended by the USEPA (1976). They were derived from data obtained by Stevenson and his associates (1953) from two prospective epidemiological studies, one conducted at two beaches on Lake Michigan in Chicago and the other comparing the symptom rates among bathers at a polluted stretch of beach on the Ohio River and a nearby chlorinated swimming pool in Kentucky. Total coliform densities associated with "detectable effects" were extrapolated to fecal coliform densities, and these in turn were adjusted to yield the guidelines as described. In the context of the present discussion, it is of interest that health effects were not observed in a third study conducted at two saltwater beaches (Stevenson, 1953). The NTAC guidelines and standards have been criticized for the paucity of epidemiological data which support them (Henderson, 1968; National Academy of Sciences-National Academy of Engineering, 1972; Moore, 1975).

Guidelines and standards used in Europe and the Mediterranean countries were reviewed in two World Health Organization documents (World Health Organization, 1975, 1977). They range from essentially none at all to limiting E.coli densities to 100 per 100 ml of water. Those guidelines proposed by the Council of the European Communities (1976) are shown in Table 8.4.

The arguments for and against health effects, recreational water quality standards based upon fecal indicator densities were presented in a symposium held in England in 1974 (Moore, 1975; Shuval, 1975) and reviewed more recently by one of the authors (Cabelli, 1978b).

Development of criteria. Potentially, information required for the development of the criteria can be obtained from four sources: 1. case and outbreak reports, 2. retrospective epidemiological studies, 3. prospective epidemiological studies, and 4. predictive models.

Case and outbreak reports of swimming-associated enteric disease, when they do occur, have been understandably only marginally productive as sources of information for the development of water quality criteria for the reasons given elsewhere (Cabelli, 1978b). The major reason is that information on the quality of the water and the nature of the responsible source of pollution at the time the exposures took place is rarely available. In addition, "sporadic" cases of non-reportable illnesses -- these can be quite numerous as seen from the findings of prospective

Table 8.4 Bacterial Standards[1] for Bathing Water (European Economic Community)

Parameter	Guideline	Limit	Sampling Frequency
Total coliform[2]	500	10,000	F[5]
Fecal coliform[2]	100	2,000	F
Fecal streptococci[2]	100	---	D[6]
Salmonella[3]	---	0	D
Enteroviruses[4]	---	0	D

[1]These standards are to take effect in 1985. The member states agree to endeavor to observe the guideline (which requires that 90% of the values be less than the stated value) when no limit is indicated. 95 % of the samples must conform to the Limit. Ref: Official Journal of the European Communities No. L31/L S.2.76.

[2]Organisms/100 ml.

[3]Organisms/liter.

[4]PFU/10 liters

[5]Fortnightly.

[6]Samples taken on evidence of deterioration of water quality.

epidemiological studies to be described later in this chapter -- are not reported to health authorities. Moreover, a statistical association to swimming in polluted waters, much less a mathematically definable illness-water quality relationship, is difficult to establish from small outbreaks. Two notable exceptions were the shigellosis (Rosenberg, et al., 1976) and Norwalk-like virus gastroenteritis (Baron et al., 1982) outbreaks. Both occurred at freshwater beaches in state parks. Although the association to swimming was clearly established, in neither case was there definitive information on the quality of the water at the time of the outbreak. The etiological agents for most of the reported outbreaks associated with swimming in marine waters are not of fecal origin (Cabelli, 1978a).

There has been a single retrospective epidemiological study, the one reported by Moore and associates in 1959. In a study conducted at English and Welsh coastal communities, they found no greater association of swimming in sewage polluted waters among children with clinical poliomyelitis than in a group of paired controls. In addition, they could find few, if any, cases of salmonellosis which were swimming-associated. Their negative findings may have been due to the relative insensitivity of the epidemiological tool. A more likely explanation is that they described a reality with regard to these two diseases. Only one poorly-documented outbreak of water-borne poliomyelitis has been reported since that time, and it was via the drinking water route, not recreational use (McCabe, 1977). There have been only two reported outbreaks -- three cases-- of swimming-associated salmonellosis since the one in Australia in 1958 (Flynn and Thistlewaite, 1965), and both were associated with swimming in grossly polluted waters (see Cabelli, 1980)

As noted earlier, the prospective epidemiological studies conducted in the 1950s by Stevenson and his associates did reveal detectable health effects, but only for fresh waters.

The output from predictive models (Dudley et al., 1976) has not been especially useful with regard to the development of criteria. The form of the output was appropriate; the problem was the paucity and quality of the input data.

The USEPA has completed a program consisting of a number of prospective epidemiological-microbiological studies designed to develop the necessary health effects, water quality criteria for marine recreational

waters. The experimental design has been published (Cabelli et al., 1975), as have the overall findings from the program (Cabelli et al., 1979, 1982; Cabelli, 1980, 1981a, 1981b). A criterion has been developed from the multi-year studies conducted at two different geographic locations in the United States and single-year study at a third (Cabelli et al., 1982). The salient findings from this program were as follows: First, an acute, relatively benign gastroenteritis with a short incubation period and duration (Cabelli, 1981a) was the most common swimming-associated, sewage pollution-related illness (Cabelli, 1980). This is consistent with the information from reported outbreaks of water-borne disease. Second, of the potential water quality indicators examined, enterococcus densities in the bathing waters were the best correlated to the swimming-associated rates of gastro-intestinal symptoms (Table 8.5) (Cabelli et al., 1979, 1982). Equally important, E.coli was the second best indicator; and the densities of fecal coliforms -- the microbial indicator system most frequently used in recreational and shellfish growing water guidelines and standards -- were poorly correlated with the incidence of swimming-associated gastroenteritis (Figure 8.1). Third, regression lines were obtained from the US studies which predict the swimming-associated rates for total gastrointestinal symptoms and the "highly credible" portion (gastroenteritis) (y) from the mean enterococcus density in the water (x). The enterococcus and E.coli lines and the 95 percent confidence limits for the former are shown in Figure 8.2. Fourth, the recreational water quality criterion developed from the USEPA program is shown in Figure 8.3. It is the X on Y regression line which predicts the mean enterococcus density in the water from a swimming-associated gastroenteritis rate judged acceptable (Cabelli, 1980, 1981b). The correlation coefficients for the line and its 95 percent confidence limits are also shown in Figure 8.3.

The Y on X regression equation shown in Figure 8.2 can be used to predict the gastroenteritis rate and, with beach usage information, the seasonal number of cases of this disease being accepted at a given marine bathing beach. However, the distribution of enterococcus densities in the water is required for the prediction; and this information is not available for most beaches. The microbiological monitoring data, and the guidelines and standards against which these data are usually examined, are generally stated in terms of total and fecal coliform densities. For the reasons which can be seen from Table 8.5 and Figure 8.1, these

Table 8.5 Correlation Coefficients for the Swimming-associated Gastroenteritis Rates Against the Mean Indicator Densities for 1973-1975 Trials Conducted at New York City Beaches

Indicator	Correlation Coefficients (r)	Number of Points (N)
Enterococci	.96	9
E. coli	.56	9
Klebsiella	.61	11
Enterobacter/ Citrobacter	.64	13
Total Coliforms	.65	11
C. perfringens[a]	.01	8
P. aeruginosa	.59	11
Fecal Coliforms	.51	12
A. hydrophilia	.60	11

[a] No data for 1973.

Figure 8.1 Comparison of indicator-illness relationships for enterococci, E. coli, fecal coliforms, and total coliforms obtained from the New York City prospective, bathing beach epidemiological study. Data grouped by beach and year. Solid line and circles - total gastrointestinal symptoms. Broken line and triangles - highly credible gastrointestinal symptoms (gastroenteritis). Correlation coefficients (r) as shown.

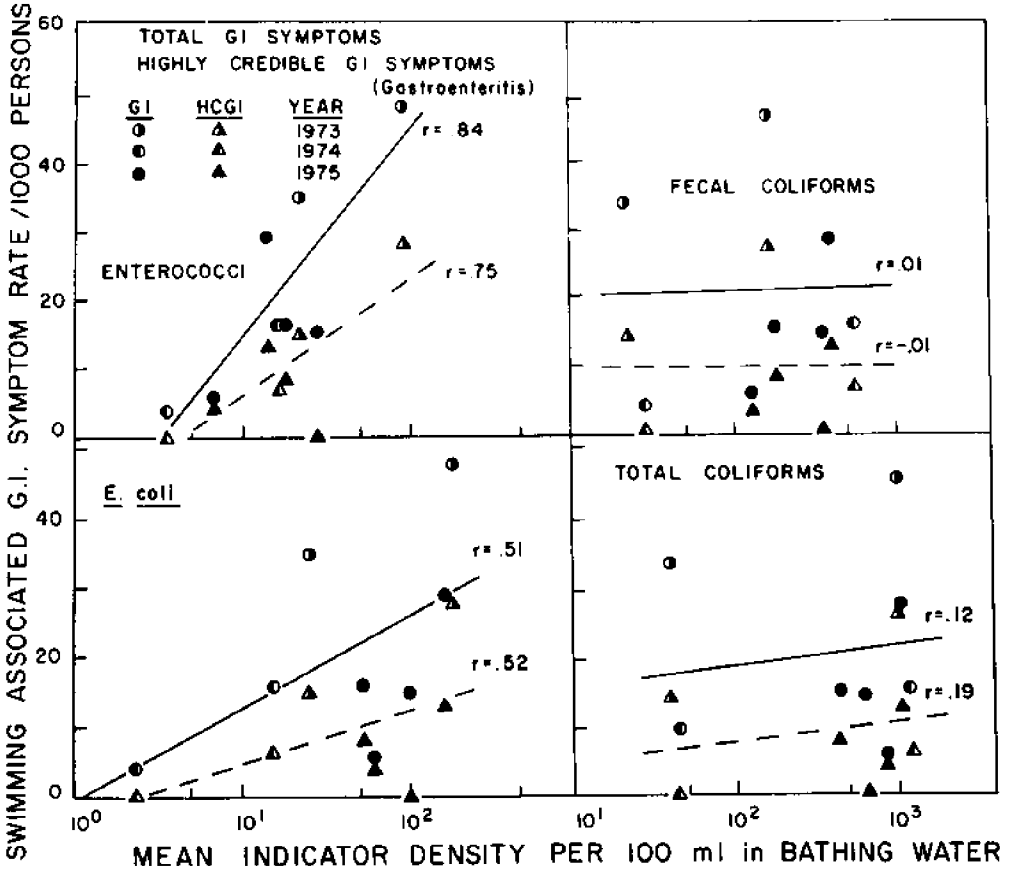


Figure 8.2 Y (rate for highly credible gastrointestinal symptoms) on X (mean indicator density in the bathing water) regression lines for enterococci and E. coli obtained from all the US bathing beach epidemiological studies. Data grouped by trials with similar indicator densities. See Cabelli, 1980 for more details.

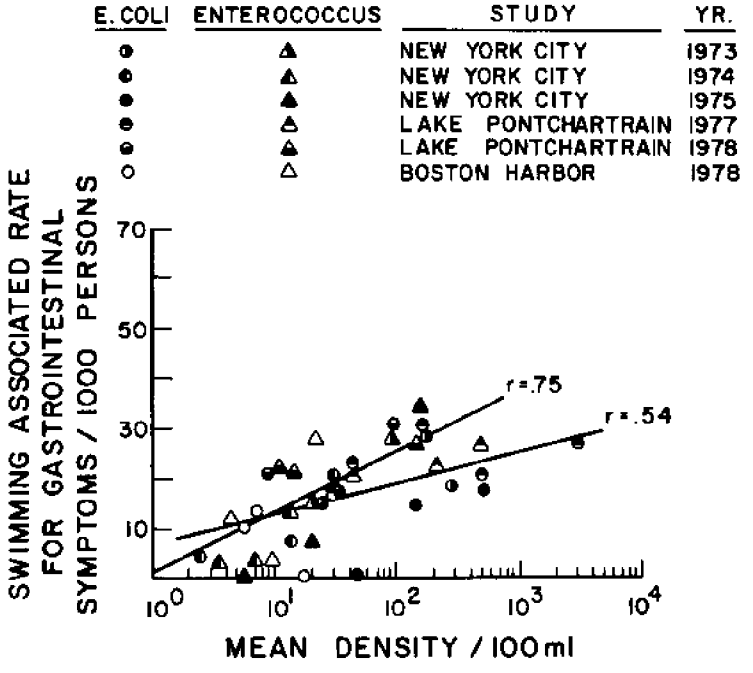
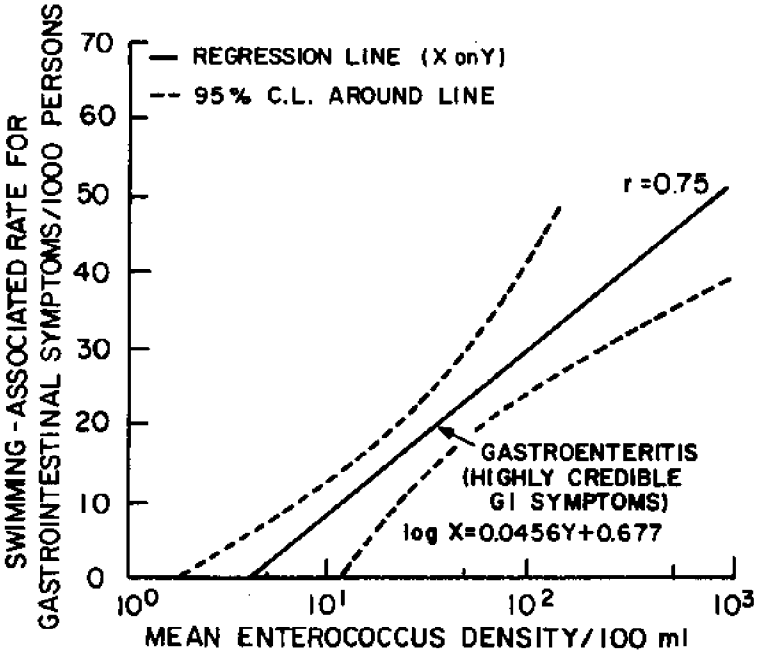


Figure 8.3 Health effects recreational water quality criterion developed from USEPA epidemiological program. It is the X on Y regression line (see Fig. 2) which predicts the mean enterococcus density in the water from some acceptable swimming-associated gastroenteritis rate.



data cannot be used in a predictive sense directly; and their extrapolation to enterococcus densities would improperly assume similar sources and biological decay characteristics for the two indicator systems.

The illness-indicator relationships obtained from the USEPA program are subject to a number of limitations. These have been considered in an earlier review (Cabelli, 1978b). One of these limitations has special relevance to the present discussion. Although the data base used in the development of the illness-indicator relationship was obtained with regard to both spatial and temporal variability, the relationship can change with the level of gastroenteritis in the population whose wastes contaminate the water, and in all probability, with the level of immunity to the specific etiological agents in the swimming population.

It can be seen from Figure 8.2 that the swimming-associated gastroenteritis rate-enterococcus density regression line differs from a typical dose-response curve in two ways. First, the slope of the line is extremely shallow, although it intercepts the x-axis at a very low enterococcus density. For example, if the regression line is extrapolated to the enterococcus density in raw sewage, 3×10^5/100 ml (Miescier and Cabelli, 1982), the predicted swimming-associated gastroenteritis rate would only be 6.7 percent. Second, the regression line for the $\log_{10}$ transformed enterococcus densities appears to be linear and not sigmoid. If the extrapolation of the relationship describes a reality, at least for the observed gastroenteritis, marked increases in pollution beyond the last data point shown (5×10^2 enterococci/100 ml) would produce only modest swimming-associated gastroenteritis rates, presumably because of a limited pool of susceptible individuals in the swimming population. As noted earlier, this leaves another important question to be answered, the conditions under which the relatively benign gastroenteritis is replaced by more serious enteric disease. An alternative explanation is that the regression line obtained is the first portion, as it were, of a sigmoid curve; and the consequence would be an accelerated increase in the rate of swimming-associated gastroenteritis with increasing pollution. The first explanation is clearly supported from gastroenteritis-enterococcus density relationships obtained from a similar prospective epidemiological study conducted at some Alexandria, Egypt beaches (Cabelli, 1980). Some of these beaches were subject to

nearby, raw sewage discharges resulting in mean enterococcus densities in the water of about 10^4/100 ml.

A 13 percent attack rate among bathers who immersed their heads in sewage-polluted, fresh bathing waters was obtained from the analysis of a swimming-associated shigellosis outbreak (Rosenberg et al., 1976). The fecal coliform density in the water, determined after the outbreak, was 17,500/100 ml. Most of the affected individuals required hospitalization. For obvious reasons, these findings do not bear on the central issue, the nature of the gastroenteritis-enterococcus density relationship. However, they do provide an insight, tenuous as it is, on the conditions which could result in more serious enteric disease.

The findings from the 1979 swimming-associated disease outbreak at two beaches on a freshwater lake in Michigan (Baron et al., 1982; Koopman et al., 1982) are more relevant to the central issue. As in the prospective epidemiological studies, the observed illness was a relatively benign gastroenteritis with a short incubation period, an acute onset, and a short duration. The causative agent was a Norwalk-like virus, one of the viral agents thought to be responsible for the observed illnesses in the epidemiological studies (Cabelli, 1981a). The reported attack rates for two of the study populations in the outbreak were 78 and 72 percent. The median, fecal coliform density as determined from 12 water samples collected on the day before and the day after the outbreak was somewhat less than 200/100 ml. However, a sanitary survey revealed no direct municipal wastewater discharges into the lake. The attack rates for the outbreak are completely inconsistent with -- are almost two orders of magnitude greater than -- that which would have been predicted from the water quality data using the illness-indicator relationships obtained in the prospective epidemiological studies. The high attack rates could have been produced by a relatively small, fecal discharge near the beach, e.g., overflowing septic tanks, boat wastes, the bathers themselves, and even the wastewater discharge from a small community, if a high percentage of the individuals contributing to the discharge were shedding the virus. It can be seen that the illness-indicator relationships in such situations have virtually no predictive value. The converse of this conclusion is a major but often over-looked limitation on criteria, guidelines and standards based on fecal coliforms. That is, they do not apply to waters impacted by small fecal sources such as those noted above.

Faunal Growing Areas

Fishery products provide the world community with an abundant and inexpensive source of protein and other nutrients. In 1975, almost 154 million pounds of seafood were harvested (Guthrie, 1975). There has been a 40 percent increase in the per capita consumption of fish products in the United States over the years 1960 to 1976 (US Department of Commerce, 1976).

The marine fauna of major concern in terms of the potential for the transmission of infectious disease are molluscan shellfish because they are consumed raw or partially cooked (steamed), they are located in the benthos, and they are filter-feeders able to concentrate particulate and soluble materials from their environments (Wood, 1976; Gong et al., 1957; Janssen, 1974; Schulze-Baldes, 1975). Concentration factors of seven- to ten-fold have been shown for various microorganisms and factors up to 17,000 for radionuclides. Other marine shellfish, e.g., crabs, and even finfish, have been implicated in outbreaks of three diseases, V. parahaemolyticus gastroenteritis, botulism, and cholera-like disease caused by non-01 V. cholerae. However, since these etiological agents are autochthonous, their presence in the fish, even in relatively large numbers, must be assumed; the barrier to disease transmission is proper processing of the product, especially heating and refrigeration where applicable; and outbreaks of disease must be attributed to breakdowns in one or more of these processes.

Emphasis in this section will be on infectious and biotoxigenic diseases associated with the consumption of molluscan shellfish. This is being done not only for the reasons stated above but also because in large part, these diseases are associated with fecal pollution and their reduction is amenable to classical pollution control technology. The diseases transmitted were listed earlier in this chapter and are documented elsewhere (World Health Organization, 1974; Guthrie, 1975). They are caused by bacterial, viral, and parasitic agents as well as toxins such as paralytic shellfish poison (Fortuine, 1975; Levin, 1978). Except for sea mussels and sea clams, most of the shellfish resource is in estuarine and nearshore coastal waters impacted to some extent with municipal sewage.

Existing criteria guidelines and standards. Criteria, guidelines and standards for the harvesting and sale of fish and shellfish have been developed in the United States and other countries. These are

intended to protect the consumer from ingestion of dangerously contaminated material. In the United States, the regulatory program for minimizing the potential for shellfish-borne disease is a cooperative one between the federal government, local agencies, and commercial interests (US Department of Health Education and Welfare (USDHEW), 1965). The cornerstone of this program is the classification of shellfish-growing areas and the prohibition of shellfish harvesting from areas classified as unacceptable. Moreover, at least with regard to the potential for infectious disease, the classification is based on the sanitary quality of the growing area water rather than the shellfish. In a sense, this practice is more analogous to process rather than product control, although some states also have market standards for the shellfish themselves. Undoubtedly, control at the source, the shellfish harvest area, is the most practical and effective means of regulating the sanitary quality of the shellfish consumed because of three considerations: the large number of independent harvesters (diggers), the diverse market channels by which the shellfish reach the consumers, and the popularity of so-called recreational harvesting (for one's own use). Thus, in addition to area classification, the effectiveness of the regulatory program is heavily dependent on enforcement activities to prevent shellfish harvesting in prohibited areas, although most states also require that lots of shellfish entering commercial channels be identified (tagged) as to when, where, and by whom they were harvested. These efforts notwithstanding, it is generally accepted that illicitly harvested (bootlegged) shellfish do enter commercial channels. Moreover, this possibility is almost always presented as one explanation for shellfish-borne outbreaks of infectious disease.

Development of guidelines and standards. The early history of frequent outbreaks of typhoid and paratyphoid fever associated with consumption of shellfish served as the basis for the current standard. A consensus was gradually reached in the period 1914 to 1925 from the analysis of outbreaks as reported to the US Public Health Service and several states. The consensus was that public health would be protected in instances where not more than 50 percent of the one ml portions of water collected from the harvest area contained coliform organisms. This guideline was established by a series of committees which met in the period 1925 to 1937. Subsequent mathematical analysis demonstrated that this was the

equivalent of approximately 70 coliform organisms/100 ml, and a study was initiated to provide data to substantiate the 70 number. This study, conducted in Raritan Bay in 1941 (Kehr and Butterfield, 1941), provided the only justification for that guideline. The authors attempted to develop a model which would predict an acceptable coliform level in terms of the probability of contracting typhoid fever from the consumption of raw shellfish. They assumed that the survival characteristics for Eberthella (Salmonella) typhosa and and coliform organisms were equivalent and that the ingestion of one S. typhosa cell would result in disease, an assumption not supported by later findings (Hornick et al., 1970). The input data included coliform and salmonella densities in sewage from various locations (data obtained from literature values in the United States and other countries) as well as coliform and salmonella densities in waters both heavily and moderately impacted with sewage. In addition, water and shellfish samples from Raritan Bay were examined; and an estimate of the dilution ratio in the Bay was developed. Their conclusion was that a 70 coliform/100 ml limit on shellfish growing waters would provide the necessary protection for the consumer. This limit as a median value and a 90 percentile value of 120/100 ml were established as the microbiological standard for shellfish harvesting waters by the National Shellfish Sanitation Program (USDHEW, 1965). Since the Kehr and Butterfield (1941) justification, there has been no concrete evidence to support the standard even though these authors strongly indicated a need to expand and support their data base. The total coliform values were extrapolated to median and 90 percentile fecal coliform limits of 14 and 53 per 100 ml, respectively. The data came from a comparison of the densities of these two indicators near the "closure lines" at a number of locations throughout the United States (Hunt and Springer, 1974).

Indicator systems. The coliform group of microorganisms (or some segment of the group) is accepted in the United States and most European countries as the most appropriate indicator of the risk of infectious disease among shellfish consumers, although in two countries examination for Salmonella species is also required (Table 8.6). Whereas total or fecal coliform densities in the growing water are used in assessing the sanitary quality of shellfish in the United States, most European countries monitor the shellfish themselves using E.coli as the indicator. However, as can be seen in Table 8.6, there is no agreement on the allowable level of this indicator organism in the shellfish. Sampling regimens and/or assay procedures also vary greatly from country to

Table 8.6 International Standards for Shellfish and Shellfish-Growing Waters

Country	Parameter Tested: Total Plate count	E. coli	Salmonella	Total Coliform
Denmark	100,000/g	(Type 1) <5/g	<5/g	Not Done
France (shellfish)	Not Done	For Cooking <2/ml[1] Raw Use <1/ml	0/25 ml	Not Done
Italy (water)	Not Done	<2/100 ml[2] 6/100 ml	Not Done	Not Done
(shellfish)	Not Done	160/100 ml[3] 500/100 ml 60/100 g	Not Done	Not Done
Netherlands (shellfish)	Not Done	0-4/ml Acceptable 5-15 Temporary Prohibition >16 Prohibition	Not Done	Not Done
United Kingdom (shellfish)	Not Done	0-4/ml Acceptable 5-15 Temporary Prohibition >16 Prohibition	Not Done	Not Done
United States (water)	Not Done	Not Done	Not Done	Median 70/100 ml <10% of sample exceeding 230

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[1]Data is expressed in terms of the volume of shellfish before dilution and homogenation.

2 <2/100 ml in 90% of samples/year.
76/100 ml in less than 10% of samples/year.

3 <160/100 ml in 90% of samples/year.
>500/100 ml in less than 10% of samples/year.
600/g = market standard

country and from place to place within countries, although this is being standardized with regard to the European Economic Community (Council of European Economic Communities, 1979).

The most cogent argument in favor of the classification of shellfish growing areas based upon the coliform levels in the shellfish is that this is the material actually ingested. However, three arguments can be used in favor of classification based on coliform levels in the water. First, the assay is simpler. Second, during the winter in temperate zones, coliforms cannot be recovered from the northern quahog (Mercenara mercenaria) even when the animals are harvested from grossly polluted waters. Nevertheless, outbreaks of infectious hepatitis have been reported from quahogs so harvested. This is probably due to the poorer survival of the indicator relative to the pathogen in the shellfish (Cabelli and Heffernan, 1970a). Third, because of the dynamics of uptake and elimination of particulates by the shellfish, the coliform level at any given time is probably an average of that in the water over the preceeding 6 to 24 hours. This negates one of the strongest features of the classification program, the requirement for sampling under the "most unfavorable conditions" (i.e., selection of sampling sites and times after identification of the hydrographic conditions resulting in the least dilution of pollutants) (USDHEW, 1965). Unfortunately, this requirement is often ignored; and sampling is conducted on a random basis or at specific sites and times which permit convenient laboratory analysis.

The biotoxin of most concern with regard to consumption of molluscan shellfish is saxotoxin or paralytic shellfish poison (PSP). It is produced by some dinoflagellates of the genus Gonyaulax and accumulated by shellfish which graze on them. Blooms of the toxic dinoflagellates are generally limited geographically, occur seasonally during certain years, and do not appear to be related to human pollution. Quantitative analysis for paralytic shellfish poison is conducted by many countries. Assaying the level of paralytic shellfish poison present in molluscs is universally accomplished by use of the mouse bioassay procedure (American Public Health Association, 1970). A level of 80 micrograms per 100 g of test tissue is universally accepted as the maximum level for human consumption (Wood, 1976).

Effectiveness of microbiological standards. The microbial standards for shellfish growing waters have

been attacked by public health officials as being too ineffective (Portnoy et al., 1957) and by shellfishermen as too restrictive (Villani and Berle, 1977). These opposite opinions are based on the same set of facts -- continuing outbreaks of disease associated with consumption of shellfish. Those who assume that all shellfish which reach the consumer -- including those implicated in outbreaks of disease -- have originated in acceptable harvesting waters feel very strongly that the current standard is too ineffective since it permitted those outbreaks. Individuals who support the existing standard or look on it as being too restrictive generally contend that the problem lies in the enforcement or classification since they assume that the implicated shellfish were harvested from closed or improperly classified areas. Because of the difficulties in tracing the shellfish implicated in such outbreaks to specific harvest areas, the paucity of water quality data at the time the shellfish were harvested, and the ever present possibility that "bootlegged" shellfish entered commercial channels, it is virtually impossible to resolve these differences from the information in outbreak reports. Moreover, as noted earlier, fecal indicator standards by their very nature cannot be expected to be operative when small, immediate sources of pollution are involved (Cabelli, 1978b).

It is unlikely that the issue of the overall effectiveness of the standards will be resolved without recourse to a controlled, prospective epidemiological study. Moreover, if the findings from the bathing beach prospective epidemiological program (Cabelli et al., 1982) can be used as a guide, the rates for sporadic cases of disease obtained from such a study, in contrast to those from the analysis of reported outbreaks, will provide a more sensitive and definitive test of the effectiveness of the standard. However, some insights can be obtained from the comparison of the trends in the reported outbreaks of the shellfish-borne infectious diseases in the United States over the past several decades. There have been no reported outbreaks of shellfish-borne typhoid or paratypoid fever since 1959. On the other hand, shellfish-borne outbreaks of other infectious diseases have occurred since that time, especially those due to the two viral diseases associated with human fecal contamination, acute, infectious, non-bacterial gastroenteritis and infectious hepatitis (Verber, 1981). This is understandable in part from the circumstances under which the coliform standard was developed, as noted earlier. That is, the calculations made by Kehr and Butterfield (1941) were based on the

levels of salmonellae in sewage at that time, and it was assumed that the ID_{50} (the number of viable cells required to infect 50 percent of the exposed individuals) for S. typhosa was one viable cell. However, the ID_{50}, as later determined from human volunteer studies, was shown to be about 10^6 viable cells (Hornick et al., 1970); and the incidence of typhoid fever in the general population (hence, the number of salmonellae discharged into sewage) has been decreasing since that time. The conclusion that gross contamination with municipal wastewaters would be required to produce an outbreak of shellfish-borne typoid fever is supported by the fact that the breakdowns in the control program (e.g., bootlegging, improper growing area classification, wastewater treatment plant malfunctions, etc.), which have been used to explain recent outbreaks of gastroenteritis and infectious hepatitis, have not produced outbreaks of typhoid fever since 1959. Moreover, for the reasons to be discussed later in this review, it could reasonably be argued that the unequivocal prohibition against harvesting shellfish in the "shadow of an outfall" (USDHEW, 1965) and the more widespread secondary treatment and disinfection of municipal wastewaters were probably more instrumental than the microbial standard in eliminating outbreaks of shellfish-borne typhoid fever.

There is no data base to indicate that the coliform standards, as limiting as they are, provide adequate protection against the possibility of shellfish-borne outbreaks, much less sporadic cases, of the two viral diseases. Moreover, it can be said with assurance that the microbial standards provide an adequate margin of safety only against the possibility of infectious disease from the contamination of shellfish growing waters with municipal wastewater containing relatively small numbers of bacterial agents of relatively low infectivity, i.e., salmonellae. Some other limitations of the standards are worthy of note. First, the potential for diseases due to autochthonous bacterial agents, such as V. parahaemolyticus, and the non-01 vibrios can not be indexed by a fecal indicator. Second, for the reasons noted earlier with reference to recreational water use, fecal indicator standards do not apply to waters subject to small-point or non-point sources of fecal pollution. Third, as seen from some recent outbreak reports (Centers for Disease Control, 1982b) and inferred from the results of the bathing beach epidemiological study (Cabelli, 1981), there is a good possibility that the degree of protection provided by the coliform standard against the potential for viral gastroenteritis is only marginal. Certainly, it is less than that provided against typhoid fever since viruses are generally more

environmentally resistant than the coliforms, viral gastroenteritis is more prevalent in the population, the viral agents are excreted in larger numbers in the feces of ill individuals, and the available data would suggest that these viral agents are highly infective (see Cabelli, 1981a).

Issues and Conclusions

It would appear that, with reference to the risk of infectious disease consequent to the recreational use of sewage polluted coastal and estuarine waters, the first of many needed steps has been taken. The USEPA has developed a water quality criterion which can be used in the context of risk analysis; and this criterion should serve as the foundation for the case-by-case development of realistic effluent guidelines for the discharge of municipal sewage wastes. However, many questions remain to be answered and much more research is needed. One of the more immediate requirements is a mathematical model for predicting the beach-specific, swimming-associated gastroenteritis rates from the distribution of enterococcus densities in the bathing waters. The acceptability of these rates on a case-by-case basis could then be used as a feed-back loop in the formulation of water quality guidelines and standards.

The etiology of the "sporadic" cases of gastroenteritis reported in the prospective, bathing beach, epidemiological studies needs to be established. Of the two suspected agents, the human rotavirus and the Norwalk-like viruses, the former is the more likely candidate for the majority of cases because of the findings with regard to age distribution and the role of immunity (Cabelli, 1981a). On the other hand, the human rotavirus has been implicated in only four drinking water-borne disease outbreaks while, as noted earlier, the Norwalk-like viruses have been shown to be the etiological agents in an increasing number of outbreaks transmitted via the swimming, shellfish, and drinking water routes. Enumerative methods for these agents are needed to produce the data on environmental levels of the agents which, with more defined infectivity data, could be used to resolve these seemingly anomalous findings and to develop the predictive models such as those proposed by Dudley et al. (1976). Such models could be useful in predicting the level of disease in the "discharging population" at which the rather benign gastroenteritis observed in current studies is replaced by more serious enteric disease. Moreover, if the gastroenteritis agents are

as prevalent, hardy and/or infective as the epidemiological data would suggest (Cabelli, 1981a), they themselves may be useful water quality indicators. This is not true of the other enteric pathogens. Isolation and enumeration methods for the shigellae and certain other bacterial pathogens are needed to better utilize the data from disease outbreaks in the refinement of the criteria and in making the predictions noted above.

If the viral, gastroenteritis agents are as resistant to the killing effects of chlorination and transport in marine waters as suspected, a water quality indicator system more resistant than the coliforms to these "environmental" effects probably will be needed for monitoring shellfish-growing as well as recreational waters. The 1975 outbreak of infectious hepatitis associated with shellfish harvested from Louisiana waters (Portnoy et al., 1975) and the absence of fecal coliforms in shellfish collected from polluted waters during the winter in temperate zones (Cabelli and Heffernan, 1970a) would suggest this to be a critical need. Enterococci are an improvement; highly chlorine resistant coliphage (McBride, 1980) may better simulate the viral pathogens in this regard.

Outbreaks of shellfish-borne enteric disease attributable to the contamination of the growing area waters with municipal sewage have been declining over the past 7 to 8 years with one exception, nonbacterial gastroenteritis. Undoubtedly, the elements of the shellfish sanitation program noted earlier and, especially, increased secondary treatment and disinfection of wastewater effluents -- in many cases as a response to the rather restrictive microbial growing area standards -- have been largely responsible. The issue to be addressed is the effectiveness of the microbiological standards in maintaining the risk of the shellfish-borne gastroenteritis at some acceptable level. For the reasons stated earlier, this issue cannot be resolved by the retrospective analysis of reported outbreaks. It can and should be addressed by a prospective epidemiological study modeled after the one used for recreational waters. This approach could also be used to consider two related questions, the adequacy of the coliform indicator systems (Portnoy et al., 1975) and the existence of sporadic cases of shellfish-borne infectious hepatitis (Koff et al., 1967). Moreover, the criteria developed from this effort could be used with regard to the acceptability rather than the detectability of risk.

Even more restrictive standards may be required to reduce the risk of shellfish-borne enteric disease, probably viral gastroenteritis but possibly infectious hepatitis as well, to some acceptable level. Relative to the value of the shellfish resource, the cost of pollution abatement needed to meet the standards may be prohibitive at times. The costs include not only the economic resources needed for advanced wastewater treatment and disposal but also potential ecologic effects of disinfection. A response to this contingency is in place. It is the controlled purification of the shellfish, depuration (USDHEW, 1965). Relatively short and economically feasible treatment periods have been found to reduce to acceptably low levels the fecal coliform levels in shellfish harvested from marginally polluted, "restricted" areas (Heffernan and Cabelli, 1970; Cabelli and Heffernan, 1970b). However, the effectiveness of depuration and the treatment period required to achieve acceptable reductions in the naturally occurring levels of the gastroenteritis and hepatitis viruses remains a moot point. More research and an indicator system that better reflects the reduction in these viral agents are needed.

It was emphasized earlier that recreational water and shellfish growing area quality criteria, guidelines and standards based upon fecal indicators, including those developed in the USEPA program, cannot be applied in certain specific situations. Aside from an increased awareness of these limitations by operating officials, there is a need to develop alternative monitoring and control strategies for such situations. There is no feasible monitoring strategy for waters contaminated with the fecal wastes of relatively small numbers of individuals (less than about 2000) since the levels of fecal indicators of the pathogens themselves have no predictive value (see Cabelli, 1978b). However, control measures and technology are available for three such situations, small boat wastes, overflowing or improperly designed disposal systems for waterside dwellings, and disposal systems for small communities. For the first, prohibition of such discharges in the vicinity of recreational and shellfish resources is required. The control strategy for the second situation is the identification and correction of the defects. The third condition requires an alternative disposal method or a treatment and disinfection regimen which will reduce relatively high densities (possibly as many as 10^7/ml of sewage) or relatively "resistant" viral pathogens to a level consistent with some acceptable risk of water-related illness. A fourth situation, fecal contamination from

the bathers themselves, relates to small ponds, bays and coves subject to minimal water exchange. Where feasible, the control technology is that used with swimming pools, restrictions on bather density and even disinfection.

The risk of disease due to autochthonous microbial pathogens, those whose source is the aquatic environment itself, can only be indexed by the pathogen density or the conditions which lead to its multiplication. Water quality criteria are not available for most of these agents, although one has been suggested for V. parahaemolyticus and one does exist for paralytic shellfish poison. The following questions need to be answered for each such agent. First, as seen from case and outbreak reports, is there a sufficient risk of water-related disease or subsequent secondary transmission to warrant water quality standards and the monitoring efforts required for area classification? Second, if so, are there temporal, spatial, or ecologic limitations on the multiplication of the agent which can be used as an alternate basis for the implementation of monitoring efforts and/or control technologies?

The third limitation on the use of microbial indicators relates to the discharge of sewered or unsewered urban and rural storm water runoff containing house, farm, and even wild animal fecal wastes. The existing microbiological criteria, guidelines and standards for target areas (recreational and shellfish-growing waters) and their pollution sources (sewage effluents or sludge) are based upon indicators which are present in both human and lower animal fecal wastes. Contamination of the water with human fecal wastes carries with it a greater risk of disease than that of lower animal wastes. Therefore, the predicted risk of disease from the use of waters impacted by non-human fecal sources will be overstated by criteria, guidelines and standards using fecal indicators. Fecal indicator systems which are more human specific and epidemiological data are needed to resolve this question. The development of appropriate disposal strategies for the combined sewer overflows from large metropolitan centers continues to be one of the most difficult problems in control technology.

There is yet another alternate control strategy for the three general situations where fecal indicator standards are not applicable and shellfish are the only potentially impacted resource. It is routine depuration of the product irrespective of the fecal indicator density in the water. This alternative is

especially attractive in dealing with the problem of the autochthonous bacterial pathogens since there is no issue of the effectiveness of viral removal, and depuration would be required only with shellfish harvested from endemic foci of the agent. More research will be needed to establish product control and monitoring procedures for all three applications.

There are no criteria and guidelines against the possibility of pollution-related infectious diseases among marine fauna. This possibility should be addressed in both its major aspects, the effect of nutrient pollution on the proliferation of the infective agent in the water and the effect of toxicants on host resistance to disease.

The potential for the dissemination of plasmids coding for multiple antibiotic resistance and of recombinant DNA via these environmental routes continues to be addressed relative to the health risks and consequences involved. A review of these efforts is beyond the scope of this chapter.

Finally, as noted earlier, the most important function of target criteria, such as those developed for recreational waters, is their use in the development of guidelines and standards for the disposal of wastes via ocean or estuarine outfalls. The research needs for this application will be discussed in the next section.

EFFLUENT GUIDELINES

Existing Guidelines and Standards

The Federal Water Pollution Act Amendments of 1972 (P.L. 92-5000) established a requirement that secondary treatment be applied to all public sewage wastes discharged into coastal and estuarine waters by 1983. In 1973, bacterial limits were added to the requirement for secondary treatment, as follows: the geometric mean density in effluent samples collected in a period of 30 consecutive days shall not exceed 200 fecal coliforms per 100 ml, and the geometric mean in effluent samples collected in a period of seven consecutive days shall not exceed 400 fecal coliforms per 100 ml. These specific fecal coliform limits virtually assured some type of disinfection since secondary treatment alone cannot reduce fecal coliforms to the 200/100 ml density. Furthermore, there was little doubt that chlorine would be the disinfectant of choice because no other comparably economical means of

disinfection was readily available. The bacterial limits were imposed to ensure the effectiveness of chlorination in rendering the treated effluent relatively free of enteric pathogens. In 1976, the fecal coliform limits and the requirement for universal disinfection were rescinded because of health effects that might occur due to organohalide by-products formed from chlorination of the effluents and because of protential ecological effects of chlorine on vertebrates and invertebrates in the immediate vicinity of sewage treatment plant outfalls.

The rescinding of the fecal coliform limit and, by inference, the requirement for universal disinfection cannot be interpreted as a mandate for totally discontinuing chlorination. Along the Atlantic and Gulf coasts, disinfection of sewage effluents usually is required to meet the microbiological guidelines for potentially impacted recreational waters and shellfish-growing areas. Rather, the change was meant to discourage overchlorination by allowing local agencies to determine the need and extent of disinfection on a case-by-case basis. This notwithstanding, it can be seen from some rather simple calculations that the fecal coliform water quality guideline for recreational waters (200/100 ml), much less that for shellfish-growing waters (14/100 ml) cannot be met without disinfection or some alternative, such as advanced treatment or long-distance offshore disposal. That is, the combination of primary and secondary treatment provides about a 15-fold reduction in the fecal coliform density, from an average of about 8×10^6/100 ml in the plant influent to about 5×10^5/100 ml in the secondary treated effluent (Miescier and Cabelli, 1982). Reasonably efficient initial dilution of the effluent at the "mixing zone" will produce a further 50-fold reduction to a density of about 10^4/100 ml. This leaves a 2 to 3 log reduction to be accomplished by disinfection at the plant or by dilution, sedimentation and biological decay during transport between the source and the potential targets. Since the latter option is unavailable in many cases, there will be a continuing requirement for chlorination or some other means of reducing the level of infectious agents in sewage effluents.

The calculations in the previous paragraph of necessity use the existing USEPA and FDA microbiological guidelines stated as fecal coliform limits. As noted earlier, both the choice of the indicator and the specific limits are being questioned on the basis of recent epidemiological data. Thus the

calculations probably understate the need for disinfection or alternate treatment and disposal strategies. Moreover, it is assumed that chlorination, as monitored by coliform or fecal coliform densities, reduces enteric pathogens to a level where they do not represent a health hazard. This assumption has been questioned, however, since many viruses may be more resistant to chlorine than the coliforms. Scarpino and his co-workers (1972) have shown that E. coli is much more sensitive to chlorine than Poliovirus and Coxsackievirus. E. coli is not only more sensitive to chlorine but also to chloramines as reported by Kjellander and Lund (1956). There are two consequences of these findings if, in fact, they describe a "real world" situation. The first is the need for a more chlorine resistant indicator, one that approaches the reputed resistance of the viral agents of hepatitis A and gastroenteritis. The second questions the effectiveness of chlorine in killing viral agents in sewage at chlorine concentrations that are environmentally acceptable. Alternative measures being considered are: 1. the use of a more effective and ecologically more acceptable means of disinfecting sewage effluents and; 2. the discharge of the effluents through ocean outfalls to areas that are distant enough from recreational or food resources.

Disinfectants other than chlorine have been considered. Both ozone (Burleson et al., 1974; O'Donovan, 1965) and bromine chlorine (Jackson, 1974) have been shown to be more effective against viruses than chlorine. However, there are drawbacks to the use of these disinfectants. They are more expensive than chlorine, and are much more difficult to apply to waste effluents. However, because of recent improvements in application technology and concern with the ecological effects of chlorination, there has been renewed interest in ozonization and ultraviolet light treatment for wastewater disinfection.

The second alternative (long, relatively deep, ocean outfalls that achieve high dilutions) has been used rather successfully by coastal communities in southern California to meet the local standard for recreational waters. This standard requires that no more than 20 percent of the water samples are to contain total coliform densities in excess of 1000/100 ml. However, there are data from both laboratory and in situ studies indicating that the behavior of the coliforms does not reflect that of the viral pathogens in that the viruses survive longer, especially when particulates are present and the differential survival operates over longer transport times. Based on

laboratory studies, Morris and his co-workers (1976) reported that enteric viruses from sewage survived three to six times longer in seawater than coliforms. Berg and Metcalf (1978), in their review of indicators of viruses, describe studies showing that the indicator bacteria-to-virus ratio decreases with exposure time in both freshwater and seawater environments. Vasl (1978) found that the T_{90} values for E.coli, as obtained from open ocean studies conducted offshore from Tel Aviv, Israel, were appreciably less than those for the enteroviruses. Equally important, those for the enterococci and enteroviruses were quite similar. The hepatitis outbreak from the consumption of raw oysters taken from waters in Louisiana which met the coliform standard (Portnoy et al., 1975) lends some supportive evidence to the premise that enteric viruses survive longer than indicator bacteria in marine environments. However, most of this evidence is circumstantial; and there is a need to verify the inferred relative decay rates by in situ studies at other locations.

Another issue is related to the adsorption of bacteria and viruses to particulate material. Organic colloids have been reported to prolong the survival of both bacteria and coliphage in seawater. Bitton's and Mitchell's (1973, 1974) data indicate that the differential die-off of particle protected viruses relative to particle protected E. coli was greater than that observed between non-protected virus and non-protected E. coli by a factor of 3:3. Similar viral data reported by Gerba and Schaiberger (1974) showed that the addition of clay particles to natural seawater extended the time to obtain a 4-log reduction of coliphage T2 from 6 days to approximately 26 days. Their results also indicate that the particles adsorb large numbers of viruses. If a similar aggregation of large numbers of pathogens on particles occurs in sewage effluents, it poses a potential problem for two reasons. First, the survival period would be prolonged, increasing the probability that viable pathogens might reach a user resource; and second, the high densities of pathogens at a single focus increases the likelihood that an infectious dose might reach a potential host.

As noted earlier, relative to their densities in the influent to sewage treatment plants, the levels of pathogenic and indicator microorganisms are not appreciably decreased by primary and secondary treatment. However, there are two aspects of sewage treatment that could influence the potential for infectious disease. The first is the generally recognized effect of BOD and suspended solids on

chlorine demand and disinfection efficiency respectively. The second is the less frequently perceived influence of the suspended solids level in sewage effluents on the potential for water-borne disease from agents with high infective doses. For example, the ID_{50} and ID_{30} for Salmonella typhosa as reported from human volunteer studies were about 10^7 and 10^5 organisms respectively (Hornick et al., 1970); the densities of salmonellae (all species) in recreational waters are relatively low (they did not exceed 0.045/100 ml at the "barely acceptable" beach in the New York City study) (Cabelli et al., 1975); and the ingested water during a swimming experience is estimated at 10 to 50 ml (Streeter, 1951). It is obvious that the probability of contracting salmonellosis through swimming in these waters should be infinitesimal since the ingested dose would have been 0.0045 salmonellae. Even if the Salmonella densities were increased by two orders of magnitude to correspond to the increase in the mean fecal coliform density from that at the "barely acceptable" beach to that found in the Mississippi River shigellosis outbreak (Rosenberg et al., 1976) of about 17,500 fecal coliforms/100 ml, the ingested dose (that in 10 ml of water) would only be 0.44 salmonellae. The probability of disease again would be extremely small. This is consistent with the etiology of the outbreak, Shigella and not Salmonella, and the fact that the ID_{50} for the former (Levine et al., 1973) is several logs lower than the latter. Even if the salmonella density were increased by another two orders of magnitude to simulate the situation of a non-immune individual swimming near the "boil" of a raw discharge (such as in 1921, when the New Haven Harbor swimming-associated outbreak of typhoid fever occurred (Ciampollini, 1921), the ingested dose would be about 50 salmonellae. This is about the level that did not produce any infections in 14 human volunteers and two logs less than that which produced an ID_{30} response in humans (Hornick et al., 1970). This very "iffy" analysis raises two related questions, suggests one possible explanation, and has implications for sewage treatment.

The questions are, under what circumstance could there be an outbreak of swimming-associated salmonellosis and what were those which led to the Australian (Flynn and Thistlewaite, 1964) and New Haven outbreaks. One explanation is the greater probability of infection if the cells are not uniformly dispersed in the water but rather are clustered on particles. In the New Haven Harbor outbreak, the individuals probably were swimming in waters heavily contaminated with raw sewage rich in particulate matter; and in the Australian outbreak, the heavy currents along the shore

could have carried the particles to the beach from the broken outfall. The implication is that suspended solids removal is very important in yet another way as regards the risk of disease. Obviously the highest concentration of the most heavily salmonella-populated particles would be in the benthos "in the shadow of an outfall." This is probably one of the reasons why typhoid and paratyphoid fever were more frequently associated with shellfish consumption than with swimming in the period from 1930 to 1960 when there was no prohibition against harvesting shellfish "in the shadow of an outfall" and why there has been no shellfish-associated typhoid fever since 1959.

Translation of Target Area Guidelines into Effluent Standards

The translation of target-area guidelines into effluent standards requires a transport model constructed from hydrographic data that characterizes a particular receiving water. Characteristics, such as vertical density distribution, currents, and dispersion, are important elements of any such model. Once the transport model has been determined, the physical, chemical, and biological factors which affect the physical and biological fate of indicators and pathogens can be applied to it so as to predict the densities at the target from those at the boil. Conversely, the model can be used to determine the effluent density and, hence, the type of treatment and disinfection and the length of the outfall to comply with a given target-area guideline.

The two physical factors which affect the biological decay of indicator and pathogenic microorganisms are water temperature and sunlight. Water temperature affects their survival directly (Greenberg, 1956) and indirectly via the affect on potential predator blooms (Sieburth, 1965). Temperature also establishes thermoclines which generally retard the shoreward transport of effluents from deep-ocean outfalls. Sunlight has been shown by many to be a highly significant factor in the disappearance of bacteria from seawater (Hollaender, 1943; Gameson and Saxon, 1967; Gameson and Gould, 1974; Bellair et al., 1977). The effect of the visible range of sunlight is related to its intensity and therefore to the depth of the water.

Chemical input factors such as nutrients and antibiotic substances do not appear to play a major role in the die-off of indicators and pathogens that

are discharged from ocean outfalls. The occurrence of nutrients in the water usually results in reduced biological decay rates, and it has been suggested that chelation of toxic heavy metals may be the reason (Jones, 1964). Other workers have observed a slight initial growth which delayed the die-off effect (Foxworthy and Kneeling, 1969). Antibiotic substances from algae may also affect the disappearance of indicators and pathogens (Sieburth, 1964). This antibiosis effect usually occurs during algal blooms which, in turn, are dependent on nutrient loading and temperature. Because of the sporadic nature of these blooms, the antibiotic factor probably has only marginal importance. Although predation of indicators and pathogens by bacteriophage, bdellovibrios and marine protozoa and competition from other microorganisms have been observed in the laboratory (Mitchell, 1968), they appear to be relatively minor factors in open coastal and estuarine waters.

Irrespective of the mechanisms involved, if disinfection is to be avoided, biological decay must produce a 1- to 2-log reduction in the densities of the currently used indicators (total or fecal coliforms). This is necessary if the levels in primary or even secondary effluents (10^6 - 10^7/100 ml) are to be reduced to those acceptable at the target (10^2 - 10^3/100 ml) for recreational waters with only a 3-log reduction by dilution and sedimentation (Odgaard, 1976). Classification of the bathing beaches on Staten Island and Coney Island as practiced by the City of New York was used by one of the authors (Cabelli, 1982) as another approach towards determining the need for wastewater disinfection. Two of the beaches, South and Graham on Staten Island, are posted as unsafe for swimming. During the summers of 1981 and 1982, water samples were collected at weekly intervals at these two beaches and at a number of others from Cape May, N.J. around to the Shinnecock Outlet, Long Island, N.Y. These were assayed for enterococci. For a given beach, each enterococcus density was extrapolated to a corresponding predicted, swimming-associated illness rate; and a 90 percentile value was obtained from the frequency distribution of the rates for each beach. Those for the two posted beaches were 6.51 and 7.28. The enterococcus densities in primary and secondary sewage, 2.0×10^5 and 8.7×10^3, respectively (Miescier and Cabelli, 1982), were reduced 1:10, 1:50 or 1:100 to account for initial dilution at the boil and then 1, 2 or 3 orders of magnitude to account for the combination of biological and physical decay during subsequent transport. These density estimates then were extrapolated to the corresponding predicted

illness rates as shown in Table 8.7. A general notion of the reductions required during initial dilution and subsequent transport were obtained by comparing the predicted rates in Table 8.7 to those at the two posted beaches. It would appear that, in the absence of the option for long, deep outfalls, the required reductions could not be achieved with primary and even secondary treated sewage without recourse to some "disinfection."

Table 8.7 Predicted Mean Swimming-Associated Gastroenteritis Rates/1000 persons for Residual Enterococcus Densities at Two Beaches Posted as Unsafe on Staten Island, NY

Treatment	Initial Dilution	Predicted Illness Rate (Y)[1] After Transport Reduction of 90%	99%	99.9%
Primary	1:10	40.6	28.3	16.1
	1:50	32.0	19.8	7.5
	1:100	28.3	16.1	3.9
Secondary	1:10	23.8	11.6	0.0
	1:50	15.3	3.0	0.0
	1:100	11.6	0.0	0.0

[1]Gastroenteritis predicted from the formula $Y = 12.25 X + 0.073$, where $X = \log_{10}$ enterococcus density/100 ml and Y is in cases per 1000 swimmers.

There may be an even greater need for disinfection than that implied from the foregoing calculations. First, the calculations were based on the need to achieve the fecal and total coliform guidelines recommended by the USEPA and used by the City of New York, respectively. As shown earlier (Figure 8.1), both indicators were poorly correlated to the incidence of swimming-associated illness; and this was probably due, in part, to their poor survival relative to that of the viral pathogens. Second, as seen from the results of the epidemiological studies, measurable, although possibly acceptable, rates of swimming-associated gastroenteritis occur at beaches considered acceptable according to both microbiological guidelines. Third, when a shellfish resource is to be protected, even greater pathogen and, hence, indicator reductions will be required because of bioaccumulation of the organisms, possibly shorter distances between the source and the targets and the combination of the

benthic location of the resource and the effect of particulates as noted earlier.

While the above considerations do not preclude long, deep outfalls as an alternative to disinfection, they do raise the possibility that different inputs to the models for calculating outfall distances may be required so as to increase both physical and biological decay during transport.

New Approaches and Research Requirements

In general, two approaches have been used to study the survival of enteric microorganisms in marine environments, *in situ* transport studies and controlled laboratory experiments. Transport studies are conducted by adding a conservative tracer (usually Rhodamine B or radioactive bromine) to the sewage before it is discharged into the ocean and then assaying the plume periodically to determine tracer and microbial densities. Both of these approaches are used to predict the extent of bacterial decay over distance or time. The apparent reduction of microorganisms with time is due to sedimentation, dilution and die-off.

The *in situ* transport experiments, although they result in data that are relevant to the area under study, do not produce decay rates that are transferable to other aquatic environments. Mitchell and Chamberlin (1974), in their review of the biological decay data from 69 *in situ* transport studies conducted at various locations around the world, found that the T_{90} (the time required for a one log (90 percent) reduction in the indicator or pathogen density) values ranged from 30 minutes to 8 hours. Harremoes (1974) suggests that this type of variation might be the result of variability in and, in some cases, the poor quality of the methods used to enumerate bacterial densities and to measure physical dilution. Undoubtedly, this accounts for part of the great variation in T_{90} values. However, it is also quite probable that much of the variation is due to "backyard" effects, that is, to unique sets of conditions characteristic of a given environment which influence the biological decay of enteric microorganisms. The existence of significant "backyard" effects would dictate that *in situ* biological decay data must be collected under conditions as similar as possible to those where the resultant decay constants are to be applied, preferably at the same "general" location. Unfortunately, for obvious reasons, this is difficult if not impossible to do in most situations.

Another possible source of variability in the biological decay coefficients obtained from *in situ* transport studies comes from under- or overestimates of the effect of sedimentation (usually the former) due to the nature of the tracer added to the sewage. The estimate of the biological decay coefficient will vary to the extent that the physical behavior of the tracer, especially as regards sedimentation, does not parallel that of the indicator and pathogen containing sewage particulates. The use of a conservative tracer already present in sewage, such as *C. perfringens* spores (Bisson and Cabelli, 1980) should not only resolve this problem but also simplify the procedure.

The alternative to *in situ* transport studies is to mimic the anticipated conditions with laboratory studies conducted in static or flow-through systems. The data from laboratory studies are usually introduced into physical transport models as decay coefficients for the physical, chemical, and biological factors which are thought to affect biological decay under natural conditions. However, the predictability of much of the data so obtained is also open to question because the actual situations often are poorly simulated. For example, survival is studied in seawater rather than in sewage-impacted seawater, in static rather than in continuously diluting systems, with pure cultures of *E. coli* rather than sewage as the source of the indicators and pathogens, and, at times, with unrealistically high densities of test organisms (this is especially true of studies with viral agents). Finally, the applicability of decay rates derived from models may be inappropriate because of the "backyard" effect mentioned above; and the validity of using land-based tanks to simulate decay in open water has not been substantiated.

It would appear that *in situ* transport studies are the only means for obtaining reliable data on enteric microorganism decay in the vicinity of ocean outfalls. The drawbacks of these types of studies are: the data obtained are applicable only to the particular "backyard" where they are collected; such studies are very expensive to conduct; and they require the presence of an existing outfall or some other means of discharging large quantities of sewage offshore. What is needed, therefore, is an economical method for obtaining biological decay data that are comparable to those obtained through *in situ* transport studies. Relatively impermeable plastic bags, filled with sewage appropriately diluted with seawater and partially or totally submerged in the waters to receive the wastewater effluent, have been used. However, while

subject to changes in ambient temperature and sunlight, they are like static chambers in most other aspects. Two systems that might meet the above criteria are *in situ* environmental diffusion chambers (McFeters and Stuart, 1972; Sieburth and Lavoie, 1976; Vasconcelos and Swartz, 1976) and large onshore tanks (Akin et al., 1971) in which the sewage is continuously diluted in locally obtained seawater. The credibility of such systems for producing data reflective of *in situ* decay would have to be verified through parallel studies with *in situ* transport experiments. If proven reliable, either method would provide an economical, transportable means of obtaining meaningful data from the "backyard" in question.

Undoubtedly, there will be an increasing need to "fine tune" wastewater treatment and disinfection and outfall location in order to achieve a realistic balance between the costs involved and adverse effects -- ecological effects of chlorination on one hand and the risk of infectious disease on the other. This requires a mathematical model (in effect, a reasonably reliable and consistent indicator-illness relationship) which will permit microbiological guidelines at potentially impacted targets to be extrapolated from deliberate decisions as to the acceptable risk of infectious disease among the users of these resources. Such a model (criterion) is now available for bathing beaches; a similar one is needed for shellfish growing areas. The second requirement is the technology needed in obtaining more realistic, reliable and site-specific biological decay coefficients for the appropriate indicator organism -- the coefficients to be used in the translation of the target area guidelines into effluent standards on a case-by-case basis. Much of the required technology, including the chambers, conservative tracer, and methods for assaying the indicators and pathogens, is now available. It needs to be tested and applied.

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9

Public Health Risks of Toxic Substances

Theodore J. Kneip

INTRODUCTION

Concern over the presence of toxic materials in municipal wastewater discharges to both marine and fresh waters has been stimulated by a continuing series of environmental crises, such as those caused by mercury discharges to Minamata Bay, Japan, or PCBs to the Hudson River. A resulting belief of many is that no natural system can assimilate any quantity of toxic materials, and that discharges will inevitably have public health implications. These feelings have been strengthened by many reports of the presence of toxic substances in public water supplies.

Discharges of toxic materials to water can result in exposures of large populations with potentially significant health effect. This has been demonstrated in a marine system by the Minamata Bay problem (D'Itri, 1972) and in fresh waters by Itai-Itai (ouch-ouch) disease (Friberg et al., 1974), and methyl mercury exposures of Canadian Indian tribes (D'Itri and D'Itri, 1977). However, in each of these instances the discharge was directly related to some form of large-scale industrial activity, rather than to any municipal waste source. No cases have been reported where toxic materials in municipal discharges have produced observable public health effects, but continuing research programs may uncover subtle, heretofore unnoticed problems.

Despite the difficulties often encountered, large-scale epidemiological studies continue to be done in efforts to ascertain causes of disease or disability due to chronic, low-level exposures of large populations to toxicants (Epidemiology Subcommittee of the Safe Drinking Water Committee, 1978). Proper design of epidemiological studies involves decisions on health data to be sought and on definitions of potential causes. Consideration of potential risks due to toxic materials can help to

assure improved design and adequate performance of future epidemiological efforts. It is the objective of this chapter to provide a basis for ranking possible exposures of human populations to toxic materials in ocean discharges of municipal wastes, and to estimate the probability of health effects from the projected exposure. This evaluation can be of use in defining potential risks that should be examined from the standpoint of acceptability or non-acceptability.

PRINCIPLES OF TOXICITY EVALUATION

Toxic materials introduced into the ocean can reach humans through three major exposure routes: contact, inhalation, and ingestion. Direct contact can occur during swimming, fishing, and other recreational activities; however, it is doubtful that concentrations of toxic materials could reach levels in recreational areas necessary to produce effects, such as contact dermatitis. Skin contact with pure compounds may be required to produce such an effect and unknown compounds are tested first at a dilution of 1 part to 100 parts of water, then at increasing concentrations. It follows that concentrations of toxic or irritating substances sufficient to cause problems through skin contact are probable only where untreated industrial discharges might occur. Evaluation of exposures could be undertaken through calculating hypothetical concentrations of irritants to assign some probability to the risks of dermatitis or mucous membrane irritation where discharge data were available.

Inhalation risks are likely to be low as well, but no data are available on concentrations of discharged toxic or surface active materials in ocean spray. Calculations of concentration factors between bulk and surface waters, measurement of aerosol mass concentrations and aerosol compositions at beaches may also need to be undertaken in the future to evaluate this exposure route.

Ingestion exposures can take direct or indirect routes. As ingestion of seawater is both accidental and normally very limited in quantity, this route will not be considered here. The route of major concern, and the most critical exposure route to be considered, is the potential contamination of seafood. As bioconcentration may occur for many toxic elements and compounds, transfer of contaminants to and through the food chain is of most concern.

Three major techniques for assessing potential risks were mentioned in Chapter 1: critical pathways, specific activities, and mass balances. Of these the "specific activity" approach is applicable only to radionuclides with both radioactive and stable isotopes present in the environment. Both the critical pathway and mass balance approaches require relatively detailed knowledge of sources, transport, diffusion, flushing times, sinks, etc. to allow calculation of probable concentrations to be expected in the system and ultimately of human exposures. The methods differ in detail, but seek to identify exposures of populations at risk for comparison to known dose response or acceptable risk levels. Application of these techniques for any substantial list of toxicants is a task for a committee of experts, as many rational assumptions will be required to select most probable values for the many factors involved.

Where data exist for the concentrations of toxic materials in food organisms, overall risk assessments also are possible by comparisons with previously established limits. For instance, comparisons can be made to acceptable or tolerable levels of toxic substances in food published by the US Food and Drug Administration (FDA, 1978) and the World Health Organization (WHO) and Food and Agriculture Organization (FAO) of the United Nations (FAO-WHO, 1972). Selection of substances for comparison can be based on the methods of a panel of experts who assessed potential toxicant threats to the New York Bight (NOAA, 1979). This area of the ocean is of critical importance as it receives five-sixths of all of the United States non-dredged waste materials found in ocean discharges (NOAA, 1976).

For the materials selected in the New York Bight study (NOAA, 1979) and for materials selected and measured in the Southern California Coastal Waters Research Project (SCCWRP, 1975) or in studies of Chesapeake Bay, San Francisco Bay and Puget Sound, direct comparisons will be made between measured and acceptable concentrations in food where limits are available. In this atttempt at risk estimates, diet fractions will not be estimated; rather an overall assumption will be made that target populations with unusually high seafood intakes exist in coastal areas. Where limits or acceptable concentrations have been stated for fish or shellfish specifically, comparisons will be make directly to these.

AVAILABLE DATA BASES

It is obviously impossible to examine all materials listed in the NIOSH "Registry of Toxic Effects of Chemical Substances" (NIOSH, 1976) for their potential risks. Several efforts to bring toxic and carcinogenic materials into focus in relation to water quality or other criteria on the potential for human exposures have resulted in lists of elements or compounds of concern. A selection of such substances is given in Table 9.1. Both the US Environmental Protection Agency (EPA) and National Science Foundation (NSF) have attempted to emphasize compound classes in an effort to reduce the magnitude of the problem of risk analysis. Classes are listed in Table 9.2 for each of these sources. Compounds or compound classes on these lists which are discharged in significant amounts through municipal sewage systems will be identified for further evaluation in this chapter.

Municipal Wastewater Inputs

In the review of municipal sewage systems provided by Mueller and Anderson in Chapter 2 of this volume, estimates are provided for effluent concentrations of some organic chemical species as well as toxic elements. Consideration is given to the effectiveness of various removal processes, costs, and potential environmental problems of alternate disposal routes.

The data from an earlier study by Mueller et al. (1976) showed that municipal discharges generally accounted for less than 20 percent of toxic metal inputs to the New York Bight, while barge dumpings accounted for 19 percent (for zinc) to 82 percent (for cadmium) of total toxic element inputs. Coastal zone runoff inputs were also significant. Revised data for mercury concentrations in municipal wastewaters in Chapter 2 reduce this input by a factor of 5 to 6 from the loadings calculated in the 1976 report.

An evaluation of these reports results in a conclusion that dredge spoil inputs to the Bight are the single most important source of toxic elements. However, this can be seriously misleading as no evidence exists as to the sources of the toxic materials in the dredged sediments or to their bioavailability. With the combined sewer systems common in the New York area, municipal effluent discharges, and particle laden storm water overflows may well be the major sources of such contamination.

Table 9.1 Toxic Substance List

Chemical	EPA[1]	NSF[2]	OSHA[3]	NY Bight[4]
Acenaphthene	+	-	-	-
Acetonitrile	-	+	-	-
Acrylonitrile	+	+	-	-
Aldrin/Dieldrin	+	-	-	+
Allyl chloride	-	+	-	-
Aniline	-	+	+	-
Antimony & cmpds.	+	-	-	-
Arsenic & cmpds.	+	-	+	+
Asbestos	+	-	+	-
Benzene	+	+	+	+
Benzidine	+	+	+	+
Benzo(a)pyrene	-	+	+	-
Beryllium & cmpds	+	-	+	-
Cadmium & cmpds	+	-	+	+
Carbon tetrachloride	+	+	+	+
Chlordane	+	-	-	+
Chlorinated benzenes	+	+	-	+
hexachlorabenzene		+		
Chlorinated ethanes:	+	+	+	+
1,2 dichloroethane	+			+
1,1,1,-trichloroethane	+	+		+
hexachloroethane	+			+
Chloroalkyl ethers	+	-	+	+
chloromethyl ether	+	-	+	
chloroethyl ether	+	-		
mixed ethers	+	-		
Chlorinated naphthalene	+	-	-	
Chlorinated paraffins: (35-64% chlorine)	-	+	-	-
Chlorinated phenols:	+	-	-	+
trichlorophenols	+	-	-	
chlorinated cresols	+	-	-	
2-chlorophenol	+	-	-	-

Table 9.1.2

Chemical	EPA	NSF	OSHA	NY Bight
Chloroform	+	+	+	+
Chloroprene	-	+	-	-
Chromium & cmpds	+	-	+	+
Copper & cmpds	+	-	-	-
O-Cresol	-	+	-	-
Cyanides	+	-	-	-
DDT & metabolites	+	-	+	+
Di (2-ethylhexyl) adipate	-	+	-	-
Diarylarylenediamines (mixed)	-	+	-	-
Dichlorobenzenes:	+	+	-	+
1,2 Dichloro benzene	+	+		
1,3 Dichloro benzene	+	-		
1,4 Dichloro benzene	+	+		
Dichlorobenzidine	+	+	+	+
Dichlorodifluromethane	-	+	-	-
Dichloroethylenes:	+	-	+	+
1,1 dichloroethylene	+	-		+
1,2 dichloroethylene	+	-		+
7,16 - Dichloro-indanthrone (vat Blue #6)	-	+	-	-
1,4 Dichlorophenol	+	-	-	+
Dichloropropane	+	+	-	-
Dichloropropene	+	+	-	-
Dimethylamine	-	+	-	-
Dimethylammonium chloride				
Bis (hydrogenated tallow alkyl)	-	+	-	-
2,4 - dimethyphenol	+	-	-	-
Dimethyl terephthalate	-	+	-	-
N,N - Dimethyldodecylamine/oxide	-	+	-	-
Dimethylpolysiloxane (Silicone fluid)	-	+	-	-
Dinitrotoluene	-	-	-	-
Di (2 ethylhexyl) phthalate	-	+	-	+
2,2 - Dithiobis (benzothiazole)	-	+	-	-

Table 9.1.3

Chemical	EPA	NSF	OSHA	NY Bight
Dodecylbanzene sulfonic acid, sodium salt	-	+	-	-
Dodecyl mercaptan	-	+	-	-
Endosulfan & metabolites	+	-	-	+
Endrin & metobolites	+	-	-	+
Ethylbenzene	+	+	-	-
Ethyl chloride	-	+	-	-
Ethylene	-	+	-	-
Ethylene dibromide	-	+	-	-
Ethylene dichloride	-	+	-	-
Ethylene oxide	-	+	-	-
Ethylenimine	-	+	+	-
Fluoranthene	+	-	-	-
Fluorescent brighting agents #28	-	+	-	-
Formaldehyde (37% by weight)	-	+	-	-
Haloethers:	+	+	+	+
chlorophenyl-phenyl ether	+	-		
bromophenylphenyl ether	+	-		
bis(dischloroisopropylether	+	+		
bis-(chloroethoxy)methane	+	-		
polychlorinated dipheny (ethers)	+	-		
bis (2-chloroethyl) ether	+	+		
Halomethanes:	+	+	+	+
methylene choride	+	-		
methylchloride	+	+		
methylbromide	+	-		
bromoform	+	-		
dichlorobromethane	+	-		
trichlorofluromethane	+	+		
dichlorodifluoromethane	+	-		
methyl bromide	-	+		
Heptachlor & metabolites	+	-	-	+
Hexachlorobutadiene	+	+	-	-
Hexachlorocyclohexame (all isomers)	+	-	+	+
Hexachlorocyclopentadiene	+	-	-	-
Hexamethylamine tetramine	-	+	-	-
Isophorone	+	-	-	+
Lead and cmpds	+	+	+	+
tetraethyl lead	+	+	-	

Table 9.1.4

Chemical	EPA	NSF	OSHA	NY Bight
2-Mercaptobenzothiazole	-	+	-	
Mercury and cmpds	+	-	-	+
2-Methoxyethanol	-	+	-	-
4,4-Methylene (bis)-2-chloraniline	-	+	-	-
Methylene chloride	-	+	-	-
Naphthalene	+	+	-	-
Nickel and cmpds	+	-	+	-
Nitrilotriacetic acid, trisodium salt	-	+	-	-
Nitrobenzene	+	+	-	-
Nitrophenols:	+	-	-	+
2,4 dintrophenol	+	-	-	
dinitrocresol	+	-	-	
Nitrosamines	+	-	+	-
p-Nonylphenyl manganese phosphate (mixed)	-	+	-	-
Nonylphenol, ethoxylated	-	+	-	-
Penthachlorophenol	+	-	-	+
Perchloroethylene	-	+	-	-
Penol	+	+	+	+
N-(1,4-Dimethypentyl)-N-phenyl-p-phenylene diamine	-	+	-	-
Phthalate esters	+	-	-	+
Polychlorin ted biphenyls (PCBs)	+	+	+	+
Polyhalogenated biphenyls	+	+	+	+
Polyethylene glycols (MW 400)	-	+	+	-
Polynuclear aromatic hydrocarbons				
benzanthracenes	+	-	+	+
benzopyrenes	+	+	-	
benzofluoranthene	+	-	-	
chrysenes	+	-	-	
dibenzanthracenes	+	-	+	
indenopyrenes	+	-	-	
Polyurethane resins (diisocyanate)	-	+	-	-
Polyvinyl chloride	-	+	-	-
Selenium and cmpds	+	-	-	+
Silver and cmpds	+	-	-	+
Sodium isopropylxanthate (Acyclic xanthic acid salts)	-	+	-	-

Table 9.1.5

Chemical	EPA	NSF	OSHA	NY Bight
Sulfolane	-	+	-	-
Tetraacetic acid, tetric sodium salt (Ethylene dinitrilo)	-	+	-	-
Tetrabromoethane Halomethane	+	+	-	-
2,3,78-Tetrachlorodibenzo-p-dioxin (dioxin) (TCDD)	+	+	+	-
Tetrachloroethylene	+	-	-	+
Tetrais (hydroxymethyl) phosphonium	-	+	-	-
Thallium and cmpds	+	-	-	+
Toluene	+	+	-	-
Toluene diisocyanate (TDI)	-	+	-	-
Toxaphene	+	-	-	+
Trichloroethylene	+	+	-	+
Tri(2-chloroethyl) phosphate	-	+	-	-
Tricresyl phosphate	-	+	-	-
Tris (2,3-dibromopropyl) phosphate	-	+	-	-
Tricresyl phosphate	-	+	-	-
Vinyl chloride	+	+	+	+
Vinylidene dibromide	-	+	-	-
Vinyl toluene	-	+	-	-
Xylenes (mixed total)	-	+	-	-
Zinc and cmpds	+	+	-	-
Zinc di(butylhexyl) phosphorodithiote	-	+	-	

Note: + present
- absent

1. US Environmental Protection Agency, July 10, 1978.
2. Nelson, N. and B. Van Duuren, 1975.
3. OSHA, October 4, 1977.
4. NOAA, MESA, 1979.

Segar and Cantillo (1976) have pointed out that the calculated input in the estuarine flow (volume multiplied by concentration) is much lower than Mueller's estimates for Cu and Zn. They attribute the difference to the possible transfer to sediments and transport as dredge spoil as noted above. This reinforces the possibility that the major source of some elements may actually be municipal discharges, although the route of input may be indirect.

Considerable information on the discharge of toxic materials to the southern California Bight has been summarized in a report of the Southern California Coastal Water Research Project (1977). Data are included on concentrations, annual quantitites of toxic materials discharged, and sources of the various toxic materials. Trends in the discharges and changes in sources are discussed in the report with emphasis on the causes of decreases or increases in the discharges. Control of industrial sources has greatly reduced discharges of total DDT and total PCB compounds by the southern California municipal systems. Toxic metal levels show year-to-year fluctuations, but no overall trends.

Water Quality

Major studies have been carried out in the New York Bight, southern California coastal waters, Puget Sound, San Francisco Bay, and Chesapeake Bay. However, data on toxic substances have not been collected in a uniform manner, since the objectives of the studies have varied with locations. The generally useful data from reports on studies of toxic materials are summarized in Table 9.3 for the New York Bight (Alexander et al., 1978) and Southern California Coastal Waters (SCCWRP, 1975).

In reviewing water column data in the New York Bight, Segar and Cantillo (1976) have concluded that elevated concentrations of the metals manganese, cadmium, and zinc are influenced by the estuarine inputs; furthermore, the dredged material dump site could not be shown to be a significant factor. The cadmium in the estuarine flow is rapidly diluted to background values, but limited sampling pattern density prevents a definite conclusion on the manganese pattern. The maximum concentrations reported are below those designated as hazardous, but indicate a need for concern, particularly as they may not represent the highest values attained in immediate dump site locations.

Table 9.2 Classes of Toxic Materials Listed or Considered

Class	EPA	NSF	NY Bight
Synthetic/ Organics	a. Toxic b. Reactive c. Chlorinated d. Pesticides	High Production, Large Discharges, Hazardous	Industrial Synthetic Organics, Significant Discharges, Hazardous
Toxic	Yes	Yes	Yes
Carcinogens	Yes	Yes	Yes

*OSHA listed only suspect carcinogens. New York Bight Panel choices were by consensus of the panel.

Table 9.3 Seawater Trace Element Concentrations in Study Areas Ranges of values, µg/l

		New York Bight Apex		Southern California Coastal Water***	
		1*	2**	Control	Plume
Cd	Soluble	- to 3.1	0.05-11	0.05	0.06-0.08
	Particulate	<0.1	--	0.02	0.09-0.18
Cu	Soluble	<0.2-20	<0.6-29	0.01	0.26-0.61
	Particulate	0.3-3.0	--	0.02	0.89-1.4
Pb	Soluble	<0.5-8	--	--	--
	Particulate	<0.5	--	--	--
Ni	Soluble	<0.4-72	0.5-7	0.29	0.72-1.2
	Pariculate	<0.2	--	0.02	0.18-0.26
Zn	Soluble	1-120	--	--	--
	Particulate	--	--	--	--

* Alexander et al., 1978. Sandy Hook Rockaways Transport.
** Segar, D.A. and A.Y. Cantillo, 1976.
*** Southern California Coastal Waters Research Project, 1975.

The southern California studies show that the increases in metal concentrations in discharge plumes are largely in particulate form, with only copper and nickel showing increases in concentrations of dissolved species. It was estimated that all values would approach background at several times the dilutions of 100 to 350 actually observed in the study (SCCWRP, 1975). As no longer-term trends in discharge rates can be observed for the metals (SCCWRP, 1977), background concentrations should exist other than in those areas immediately affected by discharge plumes. It is important to note, however, that the municipal discharges remain the principal sources of metals in the southern California coastal waters, with the exception of lead which has a major source in dry fallout (SCCWRP, 1977).

Sedimentary and Biological Fates

Several of the US coastal ocean studies that have been cited have reported concentrations of toxic materials in sediments in discharge areas. The strong tendency for particles to adsorb both trace elements and organic compounds is cited by Gross, Boesch and Roberts, and Morel and Schiff in Chapters 3, 5, and 6 of this volume. Most toxic materials are much more concentrated in sediments than in water. The transport and distribution of suspended or resuspended solids in the ocean is extremely important to an understanding of distributions and effects of toxic materials.

Elements and organic compounds can be differentiated by the fact that the elements are conservative. A sufficiently detailed effort in a system study can provide mass balance data through determination of the element sought. Such studies on organic materials are often impossible because loss of material through biochemical or chemical transformations occurs in different chemical species, as discussed in Chapters 5 and 6. Complete mass balance studies require definition of all transformed species and far more complex and expensive analytical efforts for the organic species.

As discussed in Chapter 7, the bioconcentration of organic materials has been related to the relative solubility of such compounds to water and lipid or organic solvent phases. The predictive capabilities of such an approach have been further discussed by Chiou et al. (1977) and Metcalf (1977). While this approach is generally useful in estimating the

likelihood that bioconcentration will occur for a given chemical, accurate prediction of the final concentration in the biota is not possible, as only the most general relationships have been established.

Laboratory studies of the partitioning between water and particulate phases, and of photochemical, oxidative, or biochemical transformations serve to indicate possible pathways of removal of organic compounds. However, these studies have not provided the data required to understand rates and routes of transfer or disappearance in as large and complex a system as the ocean. We must still select the materials of importance and plan the critical measurements in samples needed to answer the proper questions. In this case the question is: What are the risks to the human populations who consume seafood?

RISK ESTIMATES

The estimates of human population risks reported here will be based on the assumption that existing FDA or WHO estimates for tolerated exposures or concentration guidelines are adequate to prevent known chronic health effects. In view of the difficulties of estimating concentrations in biota from discharge quantitites and calculations of transport, degradation, and bioconcentration factors, measured values for concentrations of toxic materials in biota will be compared with the published limits.

This approach is limited to evaluation of data for the ocean areas which have been studied, and the limited number of toxic substances which have been measured to date in edible organisms. It is a serious complication that the municipal discharges may not be the major sources of the toxic materials measured.

Selection of Substances of Concern

Estimates of potentially serious problems can be made by calculating maximum dissolved concentrations based on average total discharges and average diluent water volumes. Mueller et al. (1976) estimated the average new water flow through the Bight at 5.8×10^5 cubic meters per second (m^3/s) (5×10^{10} cubic meters per day (m^3/d)) and O'Connor, et al. (1977) estimated 34,000 cubic kilometers per year (km^3/yr) (10^{11} m^3/d). A corresponding estimate for the Bight apex is 5×10^9 m^3/d from a volume of 48 km^3 and a 10-day residence given by Han (personal

communication). The values in Table 9.4 represent concentrations calculated from the mass load data of Mueller (1976), by assuming the value of 10 x 10_{10} m^3/d represents the average flow for the Bight and 5 x 10^9 m^3/d the average flow for the apex. The results of these calculations indicate that cadmium, chromium, and lead are possible problems, while other toxic elements have low ratios of calculated concentrations to seawater values. In the cases of copper and zinc, changes in accepted seawater concentrations could affect the apparent ratios.

Table 9.4 Calculated Concentrations of Several Elements for the New York Bight

Element	Input	Concentrations** Bight	Apex	Seawater*** Concentration
	MT/d	µg/l		µg/l
Cd	2.4	0.024	0.48	0.02-0.1
Cr	5.0	0.05	1.0	0.04-0.6
Cu	13.8	0.14	2.8	1.0-3.0****
Hg	0.1	0.001	0.02	0.05-0.2
Pb	12.7	0.13	2.6	0.02-0.03
Zn	33.0	0.33	6.6	1.0-10.0****

* Mueller et al., Chapter 2, this volume; and Mueller et al., 1976; units are metric tons per day (MT/d).

** Calculated using 10 x 10^{10} m^3/d for the new water volume in the Bight and 5 x 10^9 m^3/d for the Apex.

*** Summarized from Saila, S.B. and D.A. Segar, 1979.

**** Continuing trends in values reported as "true" seawater concentrations could lower these ranges.

Chlorinated compounds such as PCBs are found at concentrations of about 1 microgram per liter (µg/l) in sewage influents where many of the elements are at

concentrations of about 100 µg/l (Mueller and Anderson, Chapter 2, this volume). The resulting concentrations after dilution would be on the order of 0.001 µg/kg (micrograms per kilogram) or 1 nanograms per kilogram (ng/kg) in the ocean, assuming dissolution and dispersion proportional to that of the trace elements. This would be at or below the limit of detection for such materials by methods in routine use.

As these calculations assume total dissolution and complete homogeneous dispersion, they are not conservative for materials which enter on or transfer to solids, or for concentrated inputs such as sludges. To the extent that inputs are not uniformly distributed, the average calculated values will be in error. The calculations serve as a guideline for priorities, but cannot be used to eliminate any toxic material from consideration. Additional criteria are necessary for adequate selection of priority hazardous materials.

An evaluation of chemical pollutants in the New York Bight (NOAA, 1979) resulted in the selection of priority toxic materials on the basis of several criteria. Quantities reaching the Bight, persistence, potential for bioconcentration, and mammalian toxicity were the principal factors used. These criteria were applied to five categories of toxicants: metals and metal compounds, petroleum hydrocarbons, halogenated hydrocarbons, artificial radionuclides, and industrial synthetic organics.

The substances studied were selected from several lists of potential hazards and categorized as perceived major threats, potentially significant threats, and substances not requiring priority attention. The results are given in Table 9.5. The panel did not compare hazards between categories.

The Southern California Coastal Water Research Project has studied toxic metals, chlorinated pesticides, PCBs, chlorinated benzenes, and low molecular weight halogenated hydrocarbons. The substances selected for study are the result of a systematic ranking of priority toxicants, although the methods of ranking are not discussed in recent reports (SCCWRP, 1976; 1977).

The emphasis placed in many reports on the lack of data on a large number of hazardous organic materials clearly demonstrates the inadequate data base for risk estimates. NOAA's Office of Marine Pollution Assessment has begun broad scan analyses of a number of

Table 9.5 Priority Ranking of Chemical Pollutants in the New York Bight

Group	Metals	Petroleum Hydrocarbons	Halogenated Hydrocarbons	Artificial Radionuclides	Industrial Synthetic Organics
Major Threats	Cd Hg*	Polynuclear Aromatic Hydrocarbons	Chlorinated Pesticides	Pu	Polychlorinated Biphenyls (PCBs)
Potential Threats	Tl	Petroleum Hydrocarbons	Chlorobenzenes Chlorophenols LMHHs**	--	Benzidenes Diphenylhydrazine Halogenated diphenyl, ethers Isophorone
No Current Priority	As Cr Pb Se Ag	--	--	--	Haloalkyl ethers Nitrobenzenes Nitrophenols Phenols Phthalate Esters

* Based on earlier discharge estimates which were probably 5 to 6 times too high for sewage effluents.

** LMHH = Low molecular weight halogenated hydrocarbons

materials in the rankings to support clarification of the risk estimates for these materials (O'Connor, personal communication).

A generalized approach descriptive of the information and comparisons needed to evaluate toxic hazards is given in Table 9.6 and 9.7. This approach emphasizes many of the points discussed above, accounting for toxicity, transformations, bioavailability, and other factors related to critical pathways for such contaminants.

The FDA has established action levels for food stuff concentrations for a number of materials considered hazardous to human health. In addition, FAO-WHO have proposed provisional weekly tolerable intakes of several toxic elements for adults (see Table 9.8). Reports by individual authors and the FDA have discussed the significance of the levels of toxic elements in the US diet (Kolbye et al., 1974; Mahaffey, 1977; FDA, Bureau of Foods, 1975, 1977). The general level of lead was found to be sufficiently low as to be of no concern for adults. However, continuing concern

Table 9.6 Selection of Materials of Concern
(Estimate degree for each item)

1. Toxicity
   a. Human
   b. Aquatic biota

2. Partitioning Behavior (n-octanol/water)

3. Discharge Estimates

4. Transport (Dispersion-Dilution)

5. Availability to Biota

6. Transformation to Nontoxic or to Toxic Forms
   a. Physico-chemical
   b. Biological

7. Estimate of Concentration Factors for Biota
   a. From Items 2-6
   b. From field studies

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Table 9.7 Hazard Definition

1. List estimated or measured concentrations in biota due to municipal sewage inputs.

2. List specific or general regulatory or recommended limits (FDA, WHO, etc.)

3. Rank in order of estimated hazard considering:
   a. Excess of 1 over 2
   b. Degree of toxicity

4. Estimate population exposures
   a. Diet fraction estimates for seafood
   b. Increased toxic material exposure due to ingestion of contaminated seafood

5. Rank in order of population exposure hazard and degree

was expressed about lead exposures of very young children (0-3 years), and considerable efforts have been made to reduce lead concentrations in infant foods. Estimated median cadmium intake in the US diet were essentially unchanging at 30-35 µg/day for the adult diet (USFDA, 1975); however, this is within a factor of two of the WHO-FAO provisional tolerated intake and has been noted as a concern. Also, it is important to note that the use of average levels doubles the estimated US intake to a value equal to the WHO provisional value.

Generally only the toxic elements cadmium, lead and mercury have been considered by the various bodies responsible for protection of human health. In part this selection has been due to known uses and the potential for contamination of foods, and in part because humans are known to have been poisoned by ingesting these elements. The possibility that chlorinated compounds are carcinogens has made control of these materials of great importance, and known uses which directly contaminate foodstuffs are strictly controlled.

The WHO-FAO expert committee on food additives has stated that the concept of an "acceptable daily intake" (ADI) is not appropriate for materials such as toxic metals (FAO-WHO, 1972), since the apparent margin of safety based on available data is relatively small for many populations. The ADIs for cadmium, lead, and mercury have been set as "none." The ADI has been replaced by the term "provisional tolerable weekly intake" which recognizes that foods are contaminated, but intake levels exist at which no effects have been observed. Weekly values are used to account for daily fluctuations and the term "provisional" is intended to indicate the need for continuing review as data are accumulated. The values given in Table 9.8 comparing WHO and FDA limits are for use in evaluation of adult diets, and are not applicable in the case of infants or children.

These values make clear that extremely small amounts of toxic materials are considered hazardous. Concentrations in food of less than 0.05 to a few µg/g are normal, however, for some or all of these toxic elements and have not been known to cause overt toxic symptoms. On the other hand, foods generally contain less than 1 ng/g (µg/kg) benzo-a-pyrene (BaP), a known carcinogen (IARC, 1973). Cooked or smoked fish and meats were found to contain concentrations ranging from 2 to as high as 50 ng/g (µg/kg) depending mainly on the method of preparation.

Table 9.8 Guideline Values for Toxic Materials in Foods

Element or Compound	WHO-FAO* Provisional Tolerable Weekly Intakes		USFDA (#8) Action Levels
	µg/person	mg/kg body weight	µg/g, in food
Cd	0.4-0.5	0.0067-0.0083	--
Pb	3	0.05	-- (Condensed milk)
Hg Total	0.3	0.005	1.0: Seafood, edible portion
Methyl Mercury as mercury	0.2	0.0033	--
Chlorinated Pesticides	--	--	0.03-10: Various Compounds & Foods
PCBs	--	--	5.0: Fish

*FAO/WHO Report, 1972.

These levels are of potential concern for such carcinogens even though they are three orders of magnitude lower than the concentrations of toxic elements.

Toxic Elements

Data on trace element concentrations in organisms taken in studies of the New York Bight, southern California coastal waters, and in San Francisco Bay provide substantial information for comparison to guidelines and to general surveys. The data from these studies are compared with surveys of the National Marine Fisheries Service (NMFS) and FDA in Table 9.9.

The data for the New York Bight show somewhat higher concentrations for arsenic, lead, mercury, and silver compared with the NMFS survey, yet they fall in the ranges observed by FDA for the general US food supply. One very elevated value for cadmium is exceptional, but that value is related to a specific area in Long Island Sound and unrelated to municipal discharges. The SCCWRP data provide internal controls

Table 9.9 Trace Element Concentrations in Selected Organisms, µg/g Wet Weight

	NMFS[1]		NY Bight[2]		So. Calif.*[3]		San Francisco Shellfish[4]		USFDA Ranges[5]
	Finfish	Shellfish	Finfish	Shellfish	Control	Outfall	Control	Test	(All Foods)
Sb	0.5-0.9	0.8-1.0 (crustacea)	--	--	--	--	--	--	--
As	2.5	4.0-5	1.5-8.4	2.0-15	--	--	--	--	--
Cd	<0.1	0.1-0.3	0.05-0.4	0.1-0.4 (2.9)	0.34	0.95	0.33	0.66-1.4	0.008-0.68(99%) (35%<0.02)
Cr	0.1-0.6	--	0.1-0.5	0.3-1.0	0.35	0.05	0.1	0.3-6.6	--
Cu	0.1-2.0	Higher	0.07-7	0.4-95	0.16	0.41	2.0	1.1-48	--
Pb	0.3-0.7	0.5-0.8	0.3-1.2	0.7-1.3	<0.01	<0.01	0.9	0.4-19	0.021-2.99(99%) (35%<0.1)
Mn	<0.3	--	--	--	--	--	--	--	--
Hg	<0.3	<0.3-<0.1	0.03-0.5	0.05-0.5	--	--	0.04	0.01-0.15	1.0 (Action level)
Mo	<0.1	--	--	--	--	--	--	--	--
Ni	0.1	0.2-0.4	--	--	0.12	0.22	--	--	--
Se	0.4-0.9	--	0.06-0.9	0.03-1.7	--	--	--	--	--
Ag	<0.1	0.5-0.7	0.01-0.2	0.1-1.8	0.008	0.026	--	--	--
Sn	0.4-0.8 µi	--	--	--	--	--	--	--	--
V	<0.5	0.2-0.5	--	--	--	--	--	--	--
Zn	2.-20	--	3.4-31	6.9-920	22.0	25.0	58.0	8.5-157	--

*Scallop muscle data, mean value.

[1]Hall, R.A., E.G. Zook and G.M. Meaburn, 1978.
[2]Hall, R.H., et al., 1978.
[3]Coastal Water Research Project, 1976.
[4]US Environmental Protection Agency, 1974.
[5]USFDA, Bureau of Foods, 1975.

that are more useful in determining the effects of the discharges. The studies in southern California waters for 1975-76 (SCCWRP, 1977) examined six invertebrates and six fish for cadmium, chromium, copper, mercury, nickel, lead, silver, and zinc in outfall and control areas. Differences were for the most part relatively small, with the invertebrate organisms elevated in outfall versus control samples for 30 of 62 paired comparisons, and lower in 21 and the same in 11 of the 62 paired comparisons. Concentrations in fish were higher for outfall areas in 34 cases, lower in 24, and the same in 30 of 88 paired comparisons. Although several invertebrates appeared to concentrate most of the elements at outfall areas as compared to control areas, no toxic element concentrations approached regulatory limits.

The San Francisco study shows contamination for all elements studied when compared with internal controls. Both cadmium and lead are elevated when compared with NMFS and FDA results. Elevated coliform bacteria were present in the waters and in the organisms, where elevated trace element concentrations were found. This evidence suggests that municipal sewage discharges or mixed industrial-municipal discharges are responsible for all of these contaminants in the area.

The data of Hall et al. (personal communication) for the New York region confirm earlier reports that the highest concentrations of copper, cadmium, and zinc are found in organisms taken in Long Island Sound. The patterns indicate a source on the Sound to the east of New York City, but unrelated to dumping areas. This may represent industrial contamination such as is known to exist in the sediments of Bridgeport Harbor, but the actual source has not been determined.

Data obtained from the Chesapeake Bay and Puget Sound have not as yet provided an adequate basis for evaluation of food organism contamination by toxic elements. Two reports of studies in Puget Sound agree generally with the more extensive data for the New York Bight and southern California waters (Schell and Nevissi, 1977; Pavlou and Dexter, 1979).

Summarizing the discussed data on toxic elements, the SCCWRP studies clearly link contaminated organisms to outfall sources; however, the same cannot be said for the New York area. As noted by Mueller et al. (1976) and Mueller and Anderson in Chapter 2 of this volume, several major inputs of trace elements

exist to the Bight, and Segar and Cantillo (1976) concluded that the patterns of trace element concentrations in the apex did not clearly indicate discharge-located relationships. The Environmental Protection Agency (EPA) has concluded from the other data that some evidence exists of elevated concentrations of trace elements in organisms around the sewage sludge dump site (USEPA, 1978); however, human exposure from this source should not occur since the shell fisheries in this area are closed because of bacterial contamination. This is also the case in San Francisco Bay.

Toxic Organics

For regulatory purposes, the FDA has surveyed domestic and imported foods for the chlorinated and organophosphorus insecticides since the 1960s. A review of the data for 1969 versus 1973 is available (USFDA, Bureau of Foods, 1973) with pertinent results from this review discussed in the following paragraphs.

The early focus in the testing was on the chlorinated pesticides, and reliable data were believed to have been developed in the 1960s. The findings of Jensen (1966) that a series of peaks often seen in such testing were due to PCBs led to further analytical developments. The revised methodology was used to survey foods in 1973, and PCBs were found frequently in commodities from milk, meat and fish resources. Fish and fish byproducts, at 64.2 percent and 45.9 percent positive, were the most frequently contaminated foods. In addition, the maximum concentration of 123 µg/g found in fish was almost 100 times larger than the value of 1.6 µg/g reported for milk, the second highest value for unprocessed food products tested.

Among imported fish samples, PCBs (Aroclor-1254) were sixth in frequency of positive findings at about 9 percent with Dieldrin, TDE, DDT, BHC and DDE ranging from 10 to 20 percent. Aroclor-1260 was eighth at about 3 percent. Domestic fish samples had Aroclor-1254 second at almost 20 percent of positive findings with DDT, TDE, dieldrin and DDE ranging from about 12 to 22 percent. While only some 15 percent of imported samples showed concentrations exceeding 0.2 µg/g chlorinated materials, and only a few percent over 1.4 µg/g, the corresponding domestic frequencies were 24 percent and greater than 15 percent.

Sharp downward shifts in the fraction of positive findings were observed between 1963-1969 and 1973 for DDT and its metabolites TDE and DDE, with some increases in dieldrin, BHC, HCB and chlordane. An apparent shift from 0 to greater than 20 percent positive for PCBs reflects the absence of PCB measurements for 1963-1969. Some of the downward shift in the DDT group may also reflect earlier errors due to calculation of PCB peaks as DDT compounds.

Data from FDA Region II (New York) are similar to that of the national averages for PCBs and differ only slightly for the pesticides. FDA Region IX (California) data show no PCBs, over 26 percent DDE positive, but lower values than national averages for several other pesticides.

Samplings were classified as survey or selective, with the latter intended to assure identification and quantification necessary for determination of potential violations of action levels. In the latter series, Regions II and IX were at 28.6 and 30.0 percent violations, with only Region III at 30.8 percent and Region I (Boston) at 14.3 percent also exceeding 10 percent. The EPA (Butler, 1973) has also carried out a monitoring program on pesticides to detect trends and locations which might indicate the presence of controllable sources. Both the New York and California areas were noted as among the most contaminated sources of molluscs collected for this study.

In San Francisco, the municipal sources do not appear to contribute pesticides or PCBs in sufficient amounts to cause serious contamination, while the southern California SCCWRP studies have clearly demonstrated the positive association of DDT and PCB contaminated biota with municipal discharges. FDA data on pesticides and PCBs in Region II (New York-New Jersey) indicate a slightly elevated frequency of contaminated commercial fish but, like the toxic element problem, the cause cannot be specifically resolved as to source.

The SCCWRP efforts have focused on municipal discharge sites and compared the data to control areas and evaluated trends related to discharge control efforts. Comparison of discharge rates, sediment concentrations and concentrations in bottom dwelling fish by SCCWRP (1975, 1976, 1977) have shown discharges to be the source of the DDT and PCB materials in sediments and fish. Decreases in discharge rates from 1972 through 1975 were not

paralleled by decreases in sediments and fish. Total DDT and metabolite concentrations were in the range of 10 to 25 mg/wet kg in fish and total PCBs from about 1 to 3 µg/wet kg.

Data for 1976 (SCCWRP, 1977) showed discharges of total mono- to hexachlorobenzenes (CB) totaling 10,170 kg/yr versus 1,240 kg/yr total DDT and 1310 kg/yr total PCBs. Concentrations in the fish muscle (Dover sole) were less than 0.02 mg CB/wet kg, 11.3 µg DDTs/wet kg and 0.73 mg PCBs/wet kg. Thus, bioconcentration is much less important for the chlorinated benzenes as compared to the pesticides and PCBs. Preliminary data on SCCWRP studies of low molecular weight halogenated hydrocarbons (LMHHs) (SCCWRP, 1977) show these materials to be 100 times more concentrated than PCBs or DDTs in sewage, but data have not as yet been reported for organisms.

The San Francisco study also evaluated chlorinated compounds, but found these to be relatively low and within FDA limits. In that study of shellfish, PCBs ranged from 0.004 to 0.09 µg/g, and dieldrin was found at undetectable to 0.004 µg/g levels.

The fish in the Husdon River estuary are contaminated with PCBs believed to be mainly from an upriver industrial source. Commmercial striped bass fishing in the river has been closed for several years because of levels exceeding the FDA guideline of 5 µg/g total PCBs. Accumulated PCBs in sediment beds of the river and estuary may continue to contaminate these fish for years to come. Whether this source contributes to the FDA findings is an open question.

To date, there is no national compilation of typical concentrations of any suspect carcinogenic organic compounds in seafood. A pilot program in Puget Sound (MacLeod, 1976) found 60 to 1,000 ng/g dry tissue for several selected polynuclear hydrocarbons, or about 6-100 ng/g wet. This is comparable to the IARC levels reported for smoked food concentrations of BaP, but analytical quality control has not as yet been confirmed. A report by Pancirov and Brown (1977) indicates organisms from the New York Bight contain BaP at levels of less than 0.5 to 3 ng/g wet weight. They found those levels comparable to values reported by a number of other investigators.

An organized survey for hazardous synthetic organic chemicals in seafood organisms has been undertaken as part of the Marine Ecosystems Analysis

(MESA) Program (O'Connor, personal communication) based on the conclusions of the Panel on Chemical Contaminants in the New York Bight (NOAA, 1979).

Data on known carcinogens are simply not available. The polynuclear aromatic hydrocarbons exist in suspended particulate matter in the air of many cities at concentrations of about 1-5 ng/m^3 or at a ratio of about 0.001 to 0.005 times the concentration of lead. The exact concentration or ratio depends on the compound chosen and period of measurement. These materials originate from fuel burning sources, are present in very fine particles, and are relatively water insoluble; thus they would be expected to follow the general pathway of lead. Accordingly, they should be found in sludges, sediments, and dredge spoils. Degradation rates, transfers to biota, and other mechanisms of removal or dispersion are largely undefined. These and other important factors are less well known for any other carcinogenic compounds likely to be present.

CONCLUSIONS

Existing municipal discharges are clearly implicated in the toxic element contamination of seafood resources in limited geographical areas. In no case have mercury concentrations been shown to exceed FDA action levels. The lead levels in the New York area exceed those reported by the NMFS survey, and are of concern, but are in the same range as foods surveyed by the FDA. Cadmium concentrations do not appear very different from those of the NMFS survey or the FDA data. Except for San Francisco Bay, the effects of the discharges appear minimal. Correction of the New York ranges by removal of the Long Island Sound data would further reinforce this conclusion.

The data on pesticides and PCBs indicate that industrial (SCCWRP, 1977) or agricultural (Butler, 1973) sources are likely to be considerably more important than municipal discharges as sources of these materials to the oceans. Ongoing studies of these, and a number of other organic chemicals in the New York Bight, may help to clarify the significance of the municipal source in that area.

Values for concentrations of toxic elements in seafood, that are attributable to municipal discharges and where fisheries are not closed due to elevated *E. coli* levels, are in the range reported for the general US diet studies of the FDA. The FDA has not

established guidelines or action levels for any element except mercury at 1 µg/kg wet in edible flesh. Levels established by the FDA are based on average US calculated diet fractions and measured toxic element concentrations, and the concentrations noted in this study are probably of little significance to the general public or the overall seafood supply.

By similar calculations based on dietary compositions, Lowenberg and Kneip (1979) have shown that an increase of lead in leafy vegetables from 0.14 µg/g to 0.62 µg/g could produce a significant increase in lead intake for a one year-old child. The increases noted in concentrations of lead in shellfish or invertebrates in the New York Bight could possibly be of significance to children if specific populations were consuming large amounts of fish. Further evaluations should be undertaken where adequate data can be obtained on both dietary habits and levels of toxic materials for specific groups of people with potentially high seafood intakes.

Data for PCBs generally fall well below FDA action levels for seafood, while SCCWRP data for total DDTs in flatfish muscle tissue range from 15 to 26 mg/kg wet tissue (ng/g) from 1972 to 1975 (SCCWRP, 1976) and are reported as 11.3 mg/kg wet tissue in 1976 (SCCWRP, 1977). These values all exceed the 5 ppm (mg/kg) allowed for raw fish by FDA (1978). No indication has been given that the last value is a decrease or a result of the control of industrial inputs.

The data relating toxic material discharges from municipal systems to seafood resources does not indicate a need for drastic measures. Current EPA policies requiring control of toxic materials at industrial sources should reduce the loading of most toxic materials in municipal wastes over a period of years, and are encouraged. The use of the catalytic muffler will result in a phasing out of the use of lead in gasoline and should remove the major source of this element in sewage sludge. The successful control of major toxic material inputs to municipal systems in the southern California area indicates that similar programs should successfully control these problems in other areas. If the discharges and airborne emissions of toxic elements are thus controlled, only dissipative uses of such materials need be of further concern.

Dissipative uses could include widespread use of toxicants for pest control, which is a practice prevalent in densely populated areas of large cities. A review of pesticides used and possible fates may be helpful in evaluating potential hazards from this source.

A second category for further attention is already under study in the New York Bight. Polynuclear aromatic hydrocarbon compounds are produced in fuel burning sources and are present in all dusts deposited from airborne materials. Many other biologically active materials are also present, based on studies available to date (Daisey, 1980). The conclusions of the Panel on Chemical Contaminants in the New York Bight should be reviewed at some time in the future in terms of possible fallout or runoff discharges of these materials, which would then contribute to the concentrations in combined sewage discharges. Comparisons to date obtained in the ongoing studies of the New York Bight would afford further evaluations of hazard potentials.

Proposed land disposal or incineration of sewage sludges also will be safe if the toxic materials are removed prior to disposal. However, hazards are known to exist if such control is not achieved, and these hazards have been less carefully studied than ocean disposal. Mueller and Anderson (Chapter 2) have emphasized the problem of controlling toxicants in emissions from a sludge incinerator, and active studies exist in both USDA and FDA on the potential effects of disposal of contaminated sludges on crop lands. Cadmium is taken up by food crops under certain soil conditions and metal concentrations (Chaney et al., 1977). The data in various studies indicate that purely domestic wastes are not a problem, but sludges from sewage treatment plants with industrial sources can be serious hazards because of cadmium contents. No studies are known to exist of the typical concentrations of organics, their reactivities, or fates in land disposal of such sludges.

Data for San Francisco Bay and Long Island Sound, while somewhat fragmentary, indicate more serious problems may exist in these more confined bodies of water. Officer and Ryther (1977) have considered the problem of discharges and concluded that discharges of toxicants should not occur in these types of water bodies. In evaluating the general problem, they concluded source control to be best, with ocean discharge probably preferable to secondary

treatment. They note, however, the serious lack of data evident from this report on the fate and effects of the toxic materials in oceanic systems.

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10

Uncertainty and Policy Formation

Talbot Page

INTRODUCTION

In this chapter the focus shifts from science to policy. The preceding chapters pull together a large amount of scientific information related to wastewater and sludge disposal in coastal waters, but at the same time they suggest that what is known is but a small fraction of what needs to be known for confidence in making some of the most important policy decisions. Compared with other transport mechanisms such as air pollution and groundwater contamination, we know little about the effects and importance of toxic chemicals in coastal water wastewater and sludge disposal. There are large gaps in our knowledge of the more traditional materials as well -- the natural organics, nutrients, and metals. We can measure changes in worm populations where sludge is discharged, but we know little of the long-term implications.

The nature of uncertainty varies from one aspect of the management problem to another, from toxics to pathogens to the other materials. In some areas, such as toxics, uncertainty is very great and decision making resembles gambling. In other areas, the appropriate alternatives are more clear-cut because despite the remaining uncertainties, the alternatives are basically few and simple and the weight of evidence leans strongly to one side.

The purpose of this chapter is to consider how variations in uncertainties associated with coastal disposal of wastewater and sludge can be taken into account in making policy and decisions. The starting point for the chapter is the realization that limited data condemn the policymaker to making decisions under uncertainty. Postponing some action until more is learned is a decision under uncertainty, as is taking precautionary intermediate steps. The same is true for defining and undertaking further research.

Yet decisions must be made without all the facts. We have built ocean outfalls and treatment plants and have existing and emerging policies concerning the level of treatment, pretreatment, disposal, and source control. How such policies can be understood and assessed and, if possible, improved is the subject of this and the following two chapters. This chapter attempts to build a framework for policy analysis. Chapter 11 considers the legal institutions which provide means for implementing policy options as well as constraining them and Chapter 12 evaluates specific policy choices.

UNCERTAINTY AND THE POLICY APPROACH

The best place to begin a policy analysis is with questions about resource commitment. How and where to commit resources are the "ultimate" policy questions for agencies and legislatures concerned with wastewater and sludge disposal into coastal waters.

Questions related to resource commitment are also important for another reason. While there is much to be said for accumulating knowledge for its own sake, from the point of view of policy formation, the wealth of scientific information in the preceding chapters is useful only to the degree that it helps resolve questions of where and how to commit resources and helps structure scientific information for decision purposes. By itself, scientific inquiry is likely to be an open-ended process, with investigation of each interesting question leading to three or four more interesting questions. To structure this information for actual use, we start with a list of policy questions which have to be decided at a point in time, by default if not by choice. Questions of resource commitment can be divided into an A and a B list.

List A

1. How much regional variation should there be in the treatment of wastewater and sludge? Previous policy in the 1972 Federal Water Pollution Control Act (FWPCA) favored uniform treatment. Section 301(h) of the 1977 amendments allows for some regional variation, but how much should there be?

2. How advanced should the treatment be? Generally, more advanced treatment produces more sludge. Where do we stop?

3. What should the balance be between disposal into the ocean and into other media? How much, if any, wastewater should be treated and used for groundwater recharge and irrigation? How much sludge should go into landfills and incinerators?

4. What should the balance be between treatment at the sewage plant and pre-treatment by industrial firms before disposal into the municipal sewage system? This question applies to metals and toxic chemicals and differently in each case.

5. How much source control should there be? This question applies especially to toxic chemicals. The only alternatives are initial restriction on production or chemical transformations that result in harmless molecules. Some chemicals may be sufficiently hazardous to require control at their source.

6. What research and monitoring should be sponsored by government agencies?

While these questions will help organize the subsequent analysis, it is not possible to answer them directly. They depend to a large measure on judgments about seriousness of wastewater and sludge disposal effects. These questions about <u>effects</u> make up List B:

<u>List B</u>

1. How bad is the toxics problem? Obviously the response to this and other questions on effects will vary with the location and particular situation.

2. How severe is the problem of pathogens and communicable disease?

3. How much damage is there to recreation and to other uses of coastal waters; how much aesthetic effect is there?

4. How severe is the stress to marine populations?

These questions in turn depend upon interpreting the existing scientific information.

Policy analysts and scientists tend to work in opposite directions. To organize and focus their inquiries, policy analysts begin with the ultimate, practical questions of decision (List A) and work back to effects (List B). They then must consider what is known about marine transport, chemical, and biological systems. The scientists tend to work forward from effects beginning with the physical inputs to the coastal waters and modelling causal flows moving forward in time.[1] In this chapter the treatment of uncertainty, and correspondingly, the value of information, is derived from the policy analyst's approach. Both approaches are needed; they contribute to each other as sketched in Figure 10.1.

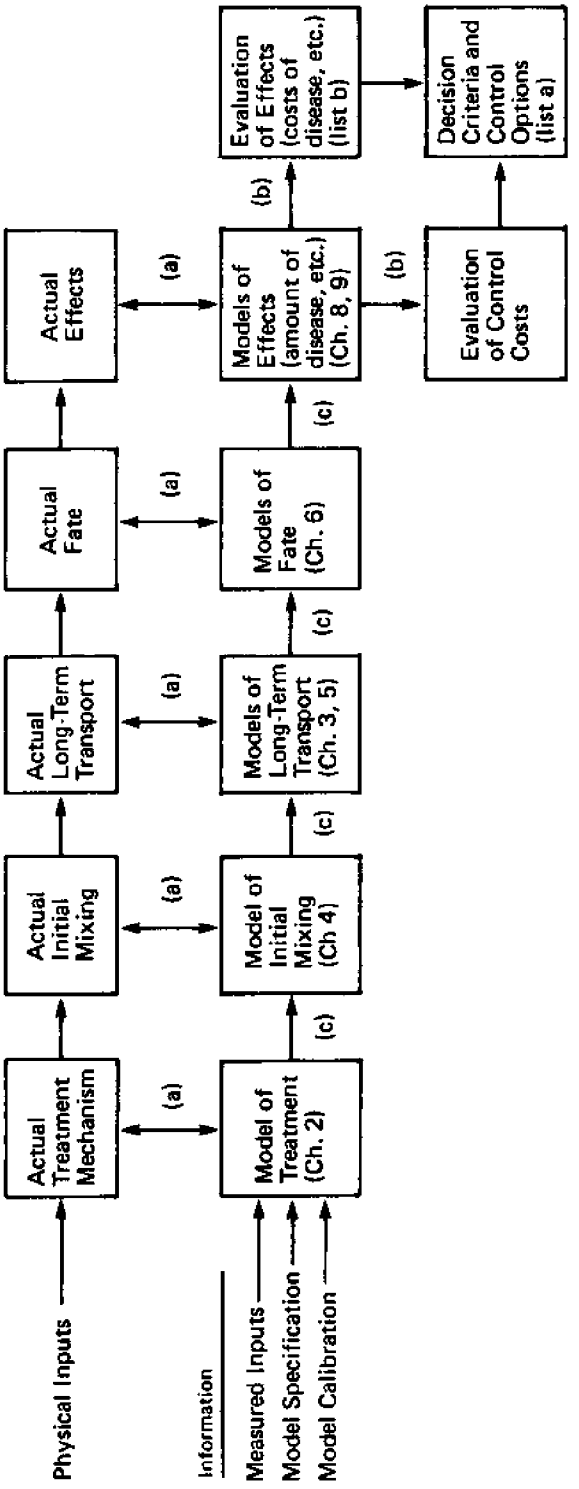
In terms of Figure 10.1 the scientists' approach, followed in much of the book, begins in the upper left corner and works down toward the lower right corner. Actual mechanisms cannot be understood directly, but scientists try to make their models correspond to the actual mechanisms as closely as possible to pursue knowledge for its own sake. Much attention is devoted to tightening the correspondences and developing detailed submodels.

Cost-benefit analysis focuses on the evaluation of effects and the costs of control. There are few cost-benefit studies of wastewater disposal and most of the attention appears to focus on quantification of recreational benefits from waste treatment (Freeman, 1979a) and the engineering costs of conventional sewage treatment (Mueller, Chapter 2 of this volume). A narrowly focused cost-benefit analysis accepts the predicted effects as more or less given and concentrates on the problems of evaluation (Freeman, 1979b).

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1 An engineering approach which attempts to optimize over a whole system, like the policy approach, may also begin with general questions of costs and benefits (list A) and move toward specific ones of modelling. The problem, of course, is to resolve questions like those in list A sufficiently to define an objective function which can be optimized. When the objective function is narrowly and exogeneously prescribed, for example, "design the least expensive tertitary treatment plant meeting the following specifications...", the approach becomes suboptimization.

Figure 10.1 Phenomena, models, and evaluation.



Although concerned with the evaluation of decisions, this chapter does not attempt a narrowly focused cost-benefit analysis. A broader approach for analysis of wastewater and sludge disposal is adopted because of the dominating role of uncertainty. How we "allocate" or treat existing uncertainties and the steps taken toward their resolution have major, and sometimes highly influential, effects on actual decisions. For example, DDT in wastewater discharges into the southern California Bight was not regulated until large volumes of DDT were discovered in the wastewater, about 1970. Once discovered, regulation by source controls did not wait for a quantified cost-benefit study of the damage to marine ecosystems. In a similar way, whether or not a chemical is banned depends more on the strength of evidence of its carcinogenicity and potency than on a fine-tuned argument on "the value of life." The basic question to be addressed is how to evaluate the costs and benefits of effects and possible control measures when there are uncertainties in predicting effects and understanding marine mechanisms.

Given the complexity of marine mechanisms and the enormous number of uncertainties, it is fair to ask at the outset what might be gained from a policy approach that attempts to work back from potential decisions in the lower right corner of Figure 10.1 toward the phenomena of the upper left.

Three types of benefit can be distinguished:

1. In a few cases the analysis might be cut and dried -- effects and control possibilities are well enough understood so that the expected net benefits of alternative decisions can be estimated quantitatively, and a single course of action selected that offers the highest net benefits.

2. In other cases the uncertainties are sufficiently pervasive so that explicit calculation of expected net benefits from alternative decisions is not helpful. Nonetheless, "islands of relative certainty" rise above the general sea of uncertainty. These islands are not built upon underwater mountains of evidence, like normal islands, but on broad understanding of what appear to be the largest and most important effects, in their qualitative evaluation of costs and benefits. In such cases, the existing evidence is not conclusive in a scientific sense, but has been accumulating in favor of the relative certainty, and it appears that further resolution of uncertainty will strengthen the

evidence. For example, concern with toxic chemical discharge into ocean water existed a decade ago (Brooks, 1971). Research in the intervening period has strengthened our concern, and it appears likely that future research will further strengthen it. Several islands of relative certainty with their policy implications are tentatively identified in this chapter and discussed in more detail in Chapter 12.

3. Perhaps most important, the policy approach even when not yielding specific decisions may offer some guidance as to the process of policy formation.

In other words, how one thinks about the various uncertainties associated with effects has a large impact on the control measures adopted. What is needed is a framework flexible enough to encompass the range of uncertainties associated with wastewater and sludge discharge. This range can be described within four classes of materials discharged.

- Class I includes nutrients and natural organic material in the form of suspended solids, ammonia and other natural oxygen-demanding materials. These materials naturally cycle through ecosystems in large quantities.

- Class II includes pathogens: bacteria and viruses.

- Class III includes heavy metals such as lead and cadmium in far greater concentrations than appear in natural systems.

- Class IV includes toxic chemicals which by affecting the genetic code (genotoxic) may cause cancer, teratogenic effects, and other diseases. A large number of synthetic organics and radioactive materials fall into this class.

As a crude generalization, uncertainty increases from Class I to IV. We know most about the mechanisms and effects of the natural materials in Class I. For example, models can predict with some confidence the effects of discharge of BOD on dissolved oxygen concentrations. The diffusion and die-off rates for pathogens have also been well studied, as the control of pathogens has been a primary concern in sewage treatment. However, there are important uncertainties still remaining in our understanding of pathogens, as

noted in Chapter 8. One such uncertainty is how well coliform, an indicator bacterium, matches the die-off rates for the pathogens of real concern, both in the treatment process and in the marine environment.

The presence of heavy metals in sewage became a concern when it was found that large concentrations could temporarily upset the sewage treatment process. Also, until the 1970s it was thought that the addition of metals to seawater background levels was small with little or no bioaccumulation in the food chain. But it is now known that waste discharges of metals are proportionally higher to the background levels (Morel and Schiff, Chapter 6). Moreover, we now know that some metals, e.g., methyl mercury, can be biologically activated. Once activated, some become toxic chemicals. Like a number of other toxic chemicals, some bioconcentrate upward in the food chain. The methylization of mercury came as a surprise, and there are important remaining uncertainties as to which metals, under what conditions, can bioactivate.

But probably the largest uncertainties are associated with the enormous number of industrially produced chemicals that find their way into wastewater and sludge. We know little about the long-term toxicity of all but a relative handful of the 75,000 chemicals in active commerce, many of which are new chemicals, synthesized after World War II.[2] We do not know how many of these chemicals, in what quantities, find their way to municipal wastewater systems and thence to coastal waters. We do not know the relative importance of wastewater systems as a conduit compared with other forms of transport and dispersion. Of the relatively small number of chemicals that have been tested adequately, perhaps several hundred for cancer and fewer for other long-term effects such as teratogenicity, we have only

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2 The inventory of the Toxic Substances Control Act (Section 8(b)) contains about 65,000 chemicals in active commercial use. This inventory does not list a number of pesticides and other chemicals controlled by other laws. The estimate of 10,000 additional chemicals itself is highly uncertain, and indeed the number on the inventory is not definite, due to problems of listing. Thus there is considerable uncertainty about the number of chemicals in active commerce.

rudimentary knowledge of toxic potency, with estimates often ranging over several orders of magnitude. For these largely unknown and uncharacterized chemicals the key questions for coastal disposal are: 1. how much is being discharged? 2. how toxic? 3. how persistent? 4. how likely to bioaccumulate, ultimately as a pathway to human exposure? 5. what effect on marine populations?

Several factors help explain the somewhat crude generalization of increasing uncertainty from Class I to IV:

- Numbers. In Class I there are only a few aggregate indexes of concern, such as BOD and SS; in Class II a relatively small number of important diseases is of concern; in Class III, though a small number of metals, a larger and undetermined number of reaction products is involved; in Class IV there is a far larger number of chemical candidates, to begin with, often at very low concentrations (ppb and below).

- Pathways. Knowledge about pathways decreases as we move from Class I to Class IV. Class I materials move along well-known natural pathways, while Class IV pollutants have long half-lives and move through a series of many complex steps.

- Latency. Some of the effects of Class I materials are almost immediate; some pathogen effects occur within a few days of discharge; some bioactivated metals effects take several years to manifest themselves; the usual latency for carcinogens is 20 to 40 years from the time of exposure (not including the time of bioaccumulation); and for mutagens, the latency may be a generation or more.

- Visibility. Somewhat associated with the other factors, the effects of Class I materials are typically more apparent than those of Class IV.

Zero-Infinity Dilemmas

The costs and benefits associated with a control alternative also vary over a wide range. At one end of the range, the underlying effects may be reasonably well understood and the costs and benefits of controlling the effect may be quantified with some confidence. This is true with smells and other simple aesthetics. In such cases, uncertainty plays a minor role and control can be regulated so that incremental costs equal the incremental benefits. Such cases can

be called cases of classical pollution, and are most amenable to classical cost-benefit analysis, where uncertainty plays a minor role.[3]

At the other end of the range (Class IV), neither effects nor the underlying mechanisms are well understood. Nonetheless, it is useful to compare the potential environmental cost with the precautionary cost of control. A term used in the area of nuclear safety, a "zero-infinity dilemma", describes a situation where there is some small estimated probability of a catastrophic cost. Thirty years ago methylization of mercury appeared highly unlikely; yet the cost of control of mercury discharge (basically source control) was small compared with the damage done by the bioactivated chemical. Those observations lead to a second, crude generalization -- control problems for Class I pollutants often tend to have classical pollution characteristics; the pollutants in Class IV a "zero-infinity" characteristic. One reason is that Class I pollutants often affect recreational and aesthetic values, while Class IV pollutants are associated with dread disease, as are some of the pathogens of Class II. In addition, many of the effects associated with Class I pollutants are readily reversible, whereas the effects of Class III and IV are more irreversible.[4]

Several characteristics taken together, which define a notion of environmental risk apply to Class IV pollutants. These characteristics are: strong uncertainty about mechanism, relatively modest cost of precautionary action compared with the potential

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[3] Even for the "simple" cases, however, quantitative cost-benefit analysis is difficult in practice, and sometimes there are legal obstacles to trading off control options, across media. See Freeman (1979a) for a review of cost-benefit studies of water pollution contol.

[4] The effects of a Class II pathogen producing cholera are more reversible than the effects of a Class IV chemical producing cancer, in the sense that the effects are longer term for the latter. Once cholera is controlled, the full effects of disease reduction are almost immediate. But once PCBs are controlled it will take decades, perhaps centuries, before they are destroyed or their negative effects mitigated.

severity of the environmental harm, low probability of harm, irreversibility of potential harm, latency of harm, external costs, internal benefits of risk, and wide dispersal of risk (collective risk) (Page, 1978). The large degree of ignorance, which leads to pervasive uncertainties throughout the modeling process, has already been mentioned. So has relative irreversibility, which exacts a higher cost than insufficient precaution would have and requires long correcting time. Latency and "zero-infinity" are also part of environmental risk. All these characteristics contribute to difficulties in managing environmental risk and in establishing institutions and incentives that lead to the "proper" amount of control.

Two other important characteristics which tend to apply to all pollutants also add to the difficulty.

First, the costs are external. The costs of Class IV pollutants are widely diffused in time and location; the costs are borne not by the generators of the expense, but by the public at large. Second, benefits are internal. The benefits accrue to the private firms generating the risks, in the form of lower operating costs. Sewerage agencies which do not take control measures, also benefit in lower operating costs. How these characteristics of internal benefit and external cost complicate the control process is illustrated by a comparison of Class III and Class IV pollutants. The characteristic of external costs applies less strongly to Class III. The main reason metals came to be controlled by pretreatment was they exacted costs that were borne internally by the treatment plant, rather than externally by the public. Thus it was to the advantage of the treatment plant operators, who were accountable for the internal costs of the plants, to control metals. Metals were affecting plant operations and raising treatment costs. A similar incentive for control does not exist for Class IV pollutants, which do not directly increase the costs of operating a treatment plant. In contrast with the classical pollutants of Class I, pollutants of Class IV tend to cause greater environmental risk.

MAXIMIZING NET BENEFITS

The goal in this section is to apply a decision analysis which is flexible enough to encompass the spectrum of uncertainty and the cost-benefit range

from balance to extreme imbalance. The analysis is driven by a maximization of expected net benefits, tailored to the situation where information is incomplete, but can be improved through research or testing (Weiner et al., 1979; Pauker and Kassirer, 1980; Weinstein, 1979; Page, 1981). Examples of decisions which fit into this framework include the following. The decision to extend the Orange County outfall from its original design length; the decision whether or not to increase the capacity of the treatment system to better accommodate storm-water overflows in San Francisco; and the decision to ban dieldrin, an example of control by source reduction.

Hypotheses and the Costs of Being Wrong

We begin with a hypothesis about the existence or non-existence of some adverse environmental effect. One such hypothesis is that some particular chemical is a human carcinogen which when released at a certain rate to seawater at some location will bioconcentrate to 10 ppm in certain fish. The evidence may be fragmentary and weak for this hypothesis and the likelihood of it may seem low. Another hypothesis is that with an extension of an outfall to 8 km into deep water in a particular region the shoreline coliform count will exceed 10 per milliliter (/ml) less than 5 percent of the time. So much may be known about the short-term diffusion around the outfall, and the other mechanics of the problem, that this hypothesis might be held true, or false, with a much higher degree of confidence.

Generally, hypotheses can be stated in one way and then restated in the opposite way. To be definite, subsequent hypotheses will be stated about the existence of some adverse environmental effect. By setting up hypotheses that assume the existence of adverse environmental effects, we are able to interpret consistently the cost of a false positive as equal in magnitude to the resource cost of precautionary control. Note that the cost of a false positive is borne only when we act to mitigate a potential environmental harm, which happens to be nonexistent. The cost of a false positive is the cost of one type of wrong decision. Because we are dealing in relative costs we can count the cost of precautionary control as one unit.

The cost of a false negative arises when we reject the hypothesis of an adverse environmental effect, but the effect exists. We can count the full

environmental cost of this harm as D + 1 units. If the correct decision had been made the D + 1 cost of environmental harm could have been avoided, but at a resource cost of 1 unit, for the precautionary action. Thus the net saving by undertaking the precautionary action is the difference, or D units. Thus we count D as the cost of a false negative. For Class III and IV pollutants the cost of a false negative may be several fold the cost of a false positive; in other words D may be "much" higher than 1. Alternatively, D is negative when the cost of control is higher than the environmental harm prevented. The cost of a false negative is the cost of the second type of wrong decision.

It may seem more comprehensive and balanced to consider 4 cases: the cost of precautionary action when the environmental hypothesis is true (1 unit); the cost of no action when the environmental hypothesis is false (0 units); the cost of not taking precautionary action when the environmental hypothesis is true (D + 1 units); and the cost of taking precautionary action when the environmental hypothesis is not true (1 unit). This explicitly takes into account the direct costs of right decisions as well as the costs of wrong decisions. But in this chapter we are measuring the costs of wrong decisions in the opportunity sense, as the cost of the wrong decision relative to the cost that could have been avoided if the right decision had been taken. Thus the cost of a false positive is (1 minus 0) or 1, and the cost of a false negative is (D + 1 - 1) or D. In this way the costs associated with right decisions are implicitly taken into account, and the analysis made more compact. Whether all the cases are analyzed explicitly or only the wrong decisions are analyzed in an opportunity cost sense, the strategies of minimum expected cost are the same and the approaches equivalent.

Likelihood of Hypotheses, Using Existing Information

After considering all the existing information, not all hypotheses appear equally likely. For an analysis of expected net benefits, some evaluation of likelihoods is needed. And here lies one of the principal differences in the approaches sketched in Figure 10.1. Traditionally many scientists prefer not to evaluate the relative likelihoods of hypotheses in terms of quantitative, subjective probability assignments. The usual reason given for this reluctance is that too little is known about the truth

or falsehood of the hypotheses to provide the grounds for numerical statements of likelihood. If pressed for quantification of some effect, instead of approaching the matter by developing subjective probability assignments for the underlying hypotheses, a different approach is taken. "Most likely" or "conservative" hypotheses are adopted; point estimates are calculated as though the hypotheses were true; and then a "safety factor" or "margin of error" is applied at the end. Decision theorists counter argue that this approach precludes the examination of minor hypotheses, which may have important implications for precautionary control (it makes the decision tree too narrow); provides no basis for determining how large the factor of safety should be; and concentrates on the propagation of measurement error while neglecting error in the specification of models where the most critical uncertainties often lie (National Academy of Sciences, 1980; Leamer, 1978).

For our purposes the matter is not either/or. Full quantification of subjective probability assignments of the likelihoods of hypotheses is not necessary any more than is the full quantification of costs and benefits of the various effects and control alternatives. The analysis proceeds as though such quantifications could be made, but then is interpreted, as much as possible, qualitatively, in terms of more or less likely, more or less costly. In fact, there is no point in full quantification of assessed likelihoods of various hypotheses, when it is not possible to quantify fully the costs and benefits of the alternative effects. We note in passing, though, that there is a growing literature on why some sort of evaluation of the relative likelihoods of hypotheses is necessary, how such evaluations can be done, and how accurate and consistent they are (Raiffa, 1970; Fischhoff, 1981; Morgan, 1979; Savage, 1971; Hogarth, 1975).

Here for the exposition we will focus on just one hypothesis at a time and write the assessed odds against it as N:1. For an environmental risk, N is definitionally greater than 1, the hypothesis of harm being considered less likely than the benign alternative, perhaps many fold less likely (N several fold bigger than 1). For a classic pollutant, N is typically less than 1, with the hypothesis more likely than not. Similarly, for a classical pollutant the costs of wrong decisions, either way, are often roughly comparable (especially for the marginal decision), and even when the adverse hypothesis is true the net saving from a precautionary decision, D,

is "near" 0, perhaps well between 1 and -1. But for an environmental risk, the net saving from a precautionary decision, D, is large compared with its cost, still counted as 1 unit. In other words, by definition, for classic pollutants both N and D are small, for ones of environmental risk both N and D are large.

New Information, Probability of Error

In their general setting, classical and environmental risk pollutants both require evaluating existing information, defining research issues, and "updating" the existing information from research and statistically interpretable tests of the data. Although for a given decision, with its researchable questions, there are likely to be many research issues as well as many tests of them. We can think of the many tests as one composite test, which when undertaken will either lead us to accept or reject the hypothesis in question.

With pervasive uncertainties the new information is unable to resolve the question with certainty. For a given empirical test, the test may erroneously suggest that the hypothesis is true when it is not. This outcome is a false positive.[5] The probability of a false positive is known as the significance level of the test, and is customarily denoted α. For well-understood tests this probability is easily quantifiable, and customarily tests are designed so that α is 5 percent, an "acceptably low" probability of a false positive. For new information not gained through a statistically designed test there is still some probability that the information will indicate the hypothesis is true when it is not, but this probability is not easily quantifiable (unless the new information is so overwhelming we might assign a 0 probability of a false positive).

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5 A test false positive can be distinguished from the false positive of precautionary control in the previous discussion. But when tests are followed one mistake leads to the other the two concepts of false positive become linked.

As for the other type of error, the test may erroneously suggest that the hypothesis is false when it is true. This outcome is known as a false negative. The probability of a false negative is customarily denoted β. For well-understood test this probability is also directly quantifiable, although its calculation depends upon the magnitude of the effect (for the calculation of α the magnitude of the effect is assumed zero) and thus is sometimes more difficult. Again, for information gained outside the structure of a formal statistical test, there will still be some probability of new information suggesting that the adverse environmental effect is absent when it exists (a false negative), but this probability is not easily quantifiable.

At this point we can state another important difference between the two approaches sketched in Figure 10.1. In the science-first approach it is customary to first set α to some predefined level, often 5 percent. Then for a given research and testing budget a statistical test is chosen to minimize β subject to α equal to the preset level. Notice in the selection of the test and α and β that evaluation of the relative costs and benefits plays no role. In the policy approach, the relative costs and benefits play a role as important as α and β. Maximizing net benefits is equivalent to minimizing the expected costs of wrong decisions, which is done by minimizing the sum of the probabilities of false positives and false negatives, weighted by the costs of false positives and false negatives. This equivalence is noted and used in the section The Maximization, below.

Together α and β characterize the information content of a statistical test or proposed new research. It would be reassuring if for a given decision we could set up the relevant hypothesis and empirical test of it so that both α and β were small, both equal to or less than 5 percent. Although this appears possible with some classical pollutants, for which information is more complete, it appears unrealistic for typical environmental risk pollutants, for which information is highly incomplete. As we shall see more clearly later, when numerical calculations are presented, the difference between the science-first and the policy-first approaches to uncertainty in new information matters less for classical pollutants than for environmental risk pollutants.

Cost of New Information

Finally, to complete the framework of expected cost, we need to compare the cost of acquiring new information with its value. The value is measured by its effect in decreasing the risk of too much precautionary behavior and the risk of too little. For a particular decision, the cost of new information from additional research and testing is counted as T units, where a unit is the cost of a false positive.

The Maximization

Once the decision under consideration is specified; the relevant hypothesis stated; the existing information incorporated into an assessment of the odds or likelihood of the hypothesis; and new research specified which would help resolve the uncertainty; there are basically three possible decisions.

First, one concludes that the research would not provide information sufficient to justify its costs and the decision maker acts on the basis of accepting the hypothesis as true (that the harmful environmental effect exists). The higher the cost of a false negative the more appropriate this decision.

Second, one concludes that the research would not provide information sufficiently valuable to justify its cost, and the decison maker acts on the basis of rejecting the hypothesis. The higher the cost of a false positive (unnecessary precautionary action) relative to the cost of a false negative, the more appropriate this decision.

Third, the additional research is undertaken, the hypothesis accepted or rejected, and precautionary action taken or not taken on the basis of the new information combined with the existing information.

It is possible for the first decision to be appropriate sometimes even in the extreme case when the resource cost of gathering additional information is 0. Such situations can arise when the cost of a false negative is high. Even if the resource cost of gathering new information is 0, there is a risk of a false negative as long as β is greater than 0. By deciding to act precautionarily the potential cost of a false negative is avoided at the cost of a false positive, which occurs whenever the hypothesis is false. Since the assessed probability of the hypothesis being false is $N/(N + 1)$ and the cost of a

false positive is valued at 1 unit, the expected cost of this decision is just

$$N/(N + 1) \qquad (10.1)$$

(Expected cost is the probability of harm times the potential magnitude of the harm. Where there are several possible severities and probabilities these products are summed, as in the third case below.)

At the other extreme the cost of precautionary control is high in comparison with the cost of a false negative. Gathering new information and relying on it entails the risk of a false positive, which can be avoided altogether if the new information is not sought and the hypothesis is rejected on the basis of existing information. In this case the cost of a false negative is borne whenever the hypothesis is in fact true. With the probability of the latter event assessed at $1/(N + 1)$ and the cost of a false negative valued at D, the expected cost is

$$D/(N + 1) \qquad (10.2)$$

In between the extremes we gather the new information and condition action on the basis of the combined new and old information. We also bear the resource cost of the test, which we count in units T, relative to the cost of control (the cost of a false positive). The expected cost of this decision is

$$D\beta/(N + 1) + N\alpha/(N + 1) + T \qquad (10.3)$$

In principle the approach of cost minimization directs us to select the smallest of (10.1), (10.2) and (10.3). In selecting (10.3) it is possible to consider tests of different cost (T), and with various probabilities of false positives and false negatives.

For a given cost of the test, α and β "trade off"; that is, if we insist on a lower α, then we must live with a higher β, and vice versa. It is possible to bring both down simultaneously, by increasing the cost of the test or research project, but there are limits to the gain in test information even with very expensive tests.

In practice, it is hard to quantify all these variables and trade-offs. Nonetheless some quantifications are possible, as we illustrate below. Moreover, the framework of expected cost helps guide qualitative inferences which will also be discussed below. The geometry of this framework is outlined in the appendix at the end of this chapter.

Levels of Uncertainty

A basic purpose in setting up the framework is to provide for essential roles for uncertainty in the analysis. Uncertainty can be analyzed at several levels: uncertainty concerning existing information, as to the likelihood of the hypothesis, or uncertainty as to the proper assessment of N; uncertainty as to the completeness of new information, or uncertainty as to and and the power of new research to resolve the existing uncertainties; and uncertainty as to the magnitudes of the potential harms and costs of precautionary controls. For some very well understood effects in Class I pollutants the first two types of uncertainty may not be important, and may drop out of the analysis. To model the non-probabilistic case, set $N=0$. In this case there is no need to acquire new information about environmental effects; the only question is whether $D > 0$ or $D < 0$. If $D > 0$, the cost of environmental harm is greater than the resource cost of control, and the preventive action is worth the cost. If $D < 0$, the cost of prevention is more than the cost of the environmental harm, and the control is not worth its cost. Thus in this limiting case, attention shifts from probabilistic and statistical aspects of decision making to the deterministic ones, estimating the engineering costs of control and measuring the environmental harm (quality of recreation diminished and so on). Much of standard cost-benefit analysis is focused in this direction.

ILLUSTRATIONS

A few examples may help illustrate some of the foregoing ideas. The first example is rather simple and its common sense solution does not require the formalization of expected value maximization of the last section. It serves more to illustrate some of the preceding definitions. In the second example the formalization of expected benefit maximization contributes to a clearer understanding of acceptable risk and its use in setting standards for pollution control. The third example suggests the likely asymmetries for Class IV pollutants.

Design of a Treatment Facility

In the design of an ocean outfall, one of the basic engineering decisions is its length from shore. The analysis involves a trade-off between marginal

environmental benefits of greater distance from shore (and greater depth) and the increased cost of construction and operation. If the chosen length is fairly long, so that shoreline impacts (like coliform counts) are already predicted to be low, the incremental benefits from a small increase in length may be small and hard to measure. But the incremental costs may also be small, because the pipe laying equipment is already at the working site, the difficult surf zone section has to be completed anyway, and the incremental length is the easiest part to build. Thus installing more pipe may cost only one half the average unit cost. In contrast, remobilizing later to extend the pipe length (if it turns out to be necessary) may cost 3 times the average cost now or 6 times the present incremental (marginal) cost because of the need to mobilize the equipment again and rearrange the diffuser section.

In this case the designer may decide as a precaution to extend the outfall a little further because it is relatively cheap insurance in case the water quality does not come out as favorably as predicted. In other words, the cost of a false positive equals the incremental cost of extending the outfall. If it turns out that the extra length is needed to protect the water quality, this is a true positive. (In this model, water quality is "adequately" protected by minimizing the sum of environmental costs and management costs.) If it turns out that the extra length is not needed, this is a false positive. The cost of a false positive represents the money that could have been saved had there been perfect information from the beginning.

From the point of view of the sewerage agency concerned only with internal costs and meeting mandated standards, the cost of a false negative can be defined directly. If the outfall were not extended and has to be extended later, this would be a false negative. The cost of a false negative is the amount that could have been saved had there been perfect information and the outfall been built right, at the beginning. The cost of a false negative is 5 units, counting as 1 unit the incremental cost of extension at the original time of construction and 6 units as the incremental cost of extension later. (Net saving is D = 6 -1 = 5.)[6]

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6 There are several complicating factors. First, it can be asked: Is not the cost of remedy overstated, due to postponement of construction and

However, the agency, whether it is local, state, or federal, may be interested in more than minimizing the expected construction and operating costs, subject to meeting existing water quality standards. It may be concerned with how well the standards reflect the true potential environmental damage from waste discharge. And in principle, the cost of a false negative is defined in terms of the potential environmental damage from insufficient precautionary action, rather than the cost of remedy if remedy is later required. Nonetheless, the cost of remedy compared with the cost of original construction is relevant. Estimating the potential environmental cost is likely to be much more difficult and uncertain than estimating the incremental costs of construction. And thus the ratio of incremental construction costs (5 in this case), can be viewed as the cost of insurance against the largely unquantified environmental risks, where the higher the ratio the lower the cost of insurance.

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the discounting of future construction costs? Ordinarily one would say yes, but the matter is complicated by the fact that the D should "really" be defined in terms of potential environmental costs, and in some sense the remedy costs are a proxy. Second, some of the potential environmental costs will begin to accrue right away, under a false negative, so that these should not be discounted. Third, the problem of discounting becomes deeper when considerations of intertemporal equity are taken into account (see Page, 1980; and Ferejohn, 1978). And fourth, there is the problem of establishing incentive compatibility.

Suppose for the sake of illustration that the incremental environmental harm, if the extension is not built, is 3 units. Because the cost of extension at the time of original construction is only one unit, the extension is worth undertaking. However, there is an incentive for the sewerage agency to minimize construction costs, subject to meeting predicted water quality constraints. This situation produces an incentive for the agency to predict low environmental impacts. Once the outfall is built too small, the incremental cost calculation goes the other way. Now it costs 6 units to control 3 units of environmental damage, and the agency can legitimately argue that the incremental construction is no longer worth the cost.

In a case like this, the decision to extend the outfall clearly depends on how much confidence the engineer has in the water quality analysis. Since it is not a precise art, it is attractive to run the risk of overbuilding the outfall when D (defined in terms of internal costs) is as high as 5. On the West Coast (see Table 4.1 in Chapter 4) outfalls have been made very long. For example, the Orange County Outfall extends 27,400 ft (8352 m) off the Orange County coast. The outfall was made probably 5,000 ft longer than necessary (Brooks, personal communication) to meet the regulatory requirements, but it was felt that the assurance of high performance and other benefits made it an attractive expenditure. Other factors entered into the decision also, such as the tendency for the requirements to get tighter during the lifetime of the outfall, which may be as long as 75 years.

A second design example suggests another pattern of potential costs and benefits. The problem is what to do about combined sewer overflows, such as those faced in many large eastern cities and the city of San Francisco on the West Coast. Because of the large volume of street runoff going into the sewers during rainstorms, the normal dry-weather treatment plants are completely inadequate to handle the suddenly increased flow when it rains. Therefore, overflows of untreated wastewater mixed with street runoff may

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Thus an agency with an incentive to minimize its internal construction and operation costs has an incentive to underbuild at the time of original construction. To offset this incentive, the agency might be told beforehand that remedy is likely to be required. The state or federal agency may develop the following policy. It announces to the sewerage agency before there is any construction that if it turns out that the environmental costs are 3 units, the sewerage agency will be required to retrofit, even at the cost of 6 units. With this policy, there is greater incentive for the agency to build the outfall longer in the first place, for a net saving of 3 to 1 or 2 units. (See, for discussion of rules of remedy, in another context but still for expected cost minimization, Kronman and Posner (1979).) The cost of remedy would be a consideration in setting up such a policy rule. Establishing a policy rule to minimize expected costs across the whole decision process, as in this case, is an example of the "principle-agent problem."

occur in coastal waters, but this happens relatively seldom during the year (e.g., 4 percent of the time in San Francisco where the rainy period is confined to the winter months). Because of the transitory nature of these overflows, they do not appear to be very damaging ecologically and are often regarded primarily as aesthetic nuisances. Coliform counts may exceed requirements for a day or two during and after the rain.

But in fundamental contrast with the previous example, the precautionary cost of control is roughly comparable to the cost of remedy, if control is postponed. That is, later control appears to be no more expensive in real dollars than proceeding at present with schemes of large storage vaults, pumping plants and greatly increased treatment plants.

This leaves us to compare the potential costs of environmental harm with precautionary costs of control. The cost of precautionary control, for storm water runoff, seems to be very high (e.g., about $1.5 billion capital cost alone for San Francisco). If indeed storm water runoff constitutes primarily a problem of transitory, aesthetic nature, the cost of a false negative will be low relative to the cost of a false positive. To the extent that storm water runoff poses a toxics problem, the question is how much the proposed treatment would control toxics. If little, then there is little precautionary benefit for toxics, and the cost of a false negative, for this type of control is retrofitting an existing system, it appears that D may be close to 1 or even negative.

In the case of a storm water overflow, the immediate effects are well understood because the overflows have already been ongoing for many years, and continuation of the present method of operation is not very risky in terms of encountering large unknowns, for the short-term effects.

A long-run solution for cities with combined sewers is to replace them gradually over a long time with a system of separate sewers, realizing that sooner or later sewers have to be replaced on something like a hundred-year cycle.

In this second example, even though the environmental effects are still uncertain, there is less motivation to proceed than there was in the first example, because D appears to be smaller in the second example. As we have seen, D is by no means the only consideration, but it is useful to distinguish among

different patterns of costs of false positives and false negatives, in different decision problems. It is also worth noting that EPA has required some cities to adopt very expensive management programs for controlling wastewater overflows, such as in San Francisco, so that this analysis of potential costs is of direct practical importance.

Acceptable Risk

Health standards for water quality and hence effluent discharge depend directly on a concept of acceptable risk. The science-first approach leads toward one notion of acceptable risk, and the policy-first approach another. The two approaches treat both uncertainty and evaluation differently, as will be shown in terms of a statistical example. One conclusion to be drawn from the example is that these differences are less important for their policy implications for Class I and II pollutants than for Class IV pollutants.

In the science-first approach attention is focused on the role of a false positive. First it must be shown that, with a high degree of confidence, some harmful effect is associated with a pathogen or chemical. Once that is done, an ample margin of safety is allowed to cover for the uncertainties associated with the magnitude of effect and pathways to exposure. The final result is an "acceptable daily intake" or a "tolerance limit." Generally, in the determination of risk, the cost of a false negative is not a consideration, nor its probability, nor the cost of a false positive. The analysis turns upon the probability of a false positive, and then the safety factor, which is more or less arbitrarily chosen. In a sense, the uncertainties which arise at each stage of the analysis are saved up until the end, where they blossom in the margin of safety.

"Acceptable daily intakes" (ADI) and "tolerance limits" should be interpreted with caution. First, as already noted, of the many thousands of chemicals of concern, only a small fraction have been tested. If an ADI or "tolerance limit" does not exist for a chemical, this does not imply that the chemical is safe in any exposure, although the chemical is often treated as though this were true. Second, some of the ADIs and tolerance limits are primarily based on tests for acute toxicity and then projected into limits for chronic toxicity (or genetic toxicity). But the relationship between acute toxicity and genetic

toxicity can vary widely. Third, the validity of a large number of the tests is now uncertain. For one example, the largest testing firm in the world, Industrial Bio-Test, went bankrupt amid charges of fraud and incompetent testing, leaving in its wake some 600 tolerances with questionable legitimacy. Fourth, the interactions or synergisms among chemicals in combination may be far more important than the chemicals acting separately. Generally the ADIs and tolerance limits are based on tests of chemicals in isolation. Yet there are cases where a non-carcinogen potentiates a carcinogen a thousand-fold. These sorts of interactions will be missed entirely in many of the tests leading to ADI and tolerances. But these possible interactions are clearly important for risk assessment of wastewater and sludge disposal, partly because of the large numbers of chemicals in the waste stream and partly because of the large number of chemicals in the receiving waters. Thus for all these reasons, it is important not to take ADIs and tolerances at face value as though they provided sufficient protection with virtual certainty, because of their safety factors.

We consider here one source of uncertainty which arises in the course of testing. For testing chemicals of suspected carcinogenicity, the most respected and widely used test is some form of the National Cancer Institute (NCI) bioassay. It is useful to discuss this test here for several reasons: The probabilities of the false positives and false negatives are well defined and can be calculated explicitly; the nature of the trade-off between α and β becomes clearer; and the difference between uncertainty within a model and uncertainty as to model specifications is illustrated. And it will illustrate why the difference between the science approach and policy approach is more important for Class IV pollutants than for Class I.

The NCI bioassay is a more or less standard test, involving 50 rodents (usually mice or rats) in a control group, and 50 in each treated group. Often there are two treatments, at different doses of the chemical under investigation. The two sexes are typically treated in separate groups; often two strains, or two different species, are used; and about 20 sites on the animal are studied for pathological response over the lifetime experiment. The Fisher exact test is usually used to interpret the number of cancers in the treated and control groups. For many of the most important test species and strains, the background level of cancer in the controls is about 2

percent for most sites (background rates for breast cancer in women and lung cancer for both men and women are several times this).

If a chemical is a potent carcinogen for which there is little other toxic effect, the Fisher exact test is very likely to detect the carcinogen (the false negative probability β is low). Suppose, for example, that a chemical has no acutely toxic effects and can be fed to animals to elevate the risk of cancer to be 15 times larger than its original background rate of 2 percent, which is the same rate as in the controls. In this case the expected percent of cancers in the treated group is 32 percent (15 x 2 + 2). For such a potent carcinogen there is only a 3 percent chance of a false negative ($\beta = .03$). This calculation assumes that the Fisher exact test is set in the traditional way with the probability of a false positive α equal to 5 percent.

However, when the chemical has acute toxicity as well as long-term cancer effects, it may not be possible to give high enough doses to achieve an excess cancer risk of 15 fold. Suppose we are dealing with the same carcinogen but it has acutely toxic effects, and we have to cut down the dose to one third or one quarter its previous level. At the lower dose level, the excess risk of this carcinogen is much smaller, perhaps 5-fold increase over the background level, with an excess risk of 10 percent. The chance of a false negative in this case is 54 percent.

This example suggests why it is so important to conduct analyses of statistical power, when dealing with toxic chemicals and other problems of environmental risk.[7] In many cases the probability of detection (β is the probability of non-detection), depends very sensitively on the magnitude of the effect being investigated, as we have just seen. A similar sensitivity of β with respect to the magnitude of the environmental effect has been shown in a study (Harris, Page and Reiches, 1977) on cancer and drinking water (this study is discussed in the next

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7 "Statistical power" and "probability of false negatives" are equivalent concepts. The first is the probability of detection, the second the probability of non-detection. They are both functions of the magnitude of effect being investigated, and for each magnitude of effect they sum to one. Statistical power is $1 - \beta$.

section). In testing, a negative result can mean either that there is no effect or that the effect exists but is undetected. There is simply no way to tell which inference is more appropriate unless β is calculated, as a function of the magnitude of the effect of concern. In other words, there is virtually no meaning in a negative result without calculation of β (a power analysis). In spite of this fact few studies of environmental effects in coastal waters -- or elsewhere -- undertake analyses of statistical power.

For risks to human populations small increases over background rates can be very important. For example, it is not uncommon for an excess risk of 10 per million to be considered unacceptably high. (Just where one draws the line depends on cold-blooded balancing of the benefits of the chemical and magnitude of effect, along with consideration of the ethics in the imposition of grave harms.) Because detectability is such an important problem in the investigation of potentially toxic chemicals, the National Cancer Institute, the Environmental Protection Agency, and the National Toxicological Program require maximally tolerated doses, as well as lower doses, in long-term bioassays, to decrease the probability of false negatives. But there are, of course, limitations to the sizes of doses which can be given, often because of acute toxicity, and we are not able to achieve low βs for existing bioassays, for many of the effects of concern.

Masking of chronic effects by acute effects is especially important when large numbers of people are to be subjected to very low doses. This of course may be the case for chemicals discharged into ocean waters and subject to dispersal and bioaccumulation. There is some evidence that chronic effects of genetic toxicity become linear with very low doses, while acute effects exhibit "safe thresholds." If true, then, as the dose is decreased but the number of people exposed increases in proportion, the importance of the chronic effect increases relative to the acute effect.

The question also arises as to whether low α can actually be achieved, due to multiple testing. Traditionally, for a given chemical under test the probability of a false positive is set for one site and 1 dose. Even if there is no effect at all, it appears that with 20 sites and 2 doses there is an excellent chance of a false positive somewhere in the 40 experiments. Salzberg (1977) raised this question

and calculated that the chance of a false positive for the composite test would be about 65 percent, if the chemical were considered to be a carcinogen upon a positive for one or more sites (the calculation assumes some statistical dependence among sites). The calculation led to a re-examination of the uncertainties associated with the specification of the statistical model itself. The Fisher exact test presumes no knowledge about the cancer incidence in the controls. Thus the assumption of the model assumes more uncertainty than in fact exists. The historical rates of cancer in the controls is well known, and this provides information on the probability of cancer of the controls. When this information on the background rate of cancer in the controls is taken into account, it is found that the true probability of a false positive is much lower than it would be if there were no information as to the spontaneous cancer rate in the controls. Fears, Tarone and Chu (1977) recalculated the probability of a false positive for a single site and dose when this information is taken into account and then found upper bounds on α for the whole experiment of 20 sites and 2 doses. These upper bounds were between 4 and 10 percent. The recalulations do not affect β.

The first point to be made here is that analyzing the uncertainties in the specification of models is highly important, and at times may be even more important than analyzing the uncertainties within specified models. The second point is that in presetting α to 5 percent there may be an acceptable trade-off between α and β, for potent carcinogens without competing toxicity, for acute effects of chemicals, and for many pathogens. For these cases it may be possible to achieve α and β both equal to or less than 5 percent. But for weak carcinogens with wide exposure, or for carcinogens with competing toxicity, presetting α to 5 percent is not acceptable without further calculation. It is not acceptable because it may imply a β of 50 percent or more. If the cost of a false negative is high compared with the cost of a false positive, then a decision approach suggests that β be decreased even at the price of increasing α. How much β should be decreased depends on the properties of the test, in particular how fast β can be decreased for a given increase in α; it depends as well on the relative costs of wrong decisions. (See the Appendix for the relationship.)

The science-first approach begins with mechanisms and effects and avoids judgments about costs and benefits associated with effects. For this

reason it is compatible with presetting α to 5 percent and then minimizing β. The procedure avoids considering the costs and benefits of effects. For Class I and Class II pollutants the procedure may lead to β also 5 percent or less; for Class IV pollutants the procedure may lead to very high β. The policy-first approach begins with consideration of the costs of wrong decisions. For Class I and Class II pollutants trading off α and β to minimize the expected cost of wrong decisions may lead to a very similar choice of α and β as in the science-first approach. But for Class IV pollutants β tends to be larger for a given α and the minimum expected cost trade-off of α and β is likely to diverge more from the α and β defined by the science-first approach. This means that the 2 approaches are likely to differ most for Class IV pollutants in their implications as to which precautionary decisions are worthwhile.

To summarize: the science-first approach focuses on the probability of a false positive and on proving the existence of an effect. In the policy-first approach all four ingredients -- the cost and probability of a false positive and the cost and probability of a false negative -- are symmetric and equally important for a concept of acceptable risk. Where the cost of a false negative is likely to be higher than the cost of a false positive, the probability of a false negative (β) is more critically sensitive for the decision than the probability of a false positive (α). When both α and β can be made small, perhaps both 5 percent or less, this difference in approach is not likely to matter very much. But when there is more uncertainty in the new information (for example, the sum of α plus β is 50 percent or more), and when there is considerable uncertainty as to the likelihood of the underlying hypothesis, then the two approaches are likely to diverge in their implications as to which effects are "positive" and which are to be taken seriously for decision purposes.

Comparison with Drinking Water Carcinogenicity

There are now some 30 or more studies of the potential cancer hazard from chemicals in drinking water (for a review see Crump and Guess, 1980). In contrast, there appear to be few studies of the potential cancer hazard from chemicals discharged into salt water. A comparison of the two potential hazards suggests some parallels as well as obvious differences. It also provides an illustration of asymmetries typical in environmental risks.

In 1974 there were 64 organic chemicals identified in drinking water; by 1981 there were over 700. These 700 represent, by weight, only about 15 percent of the organic fraction of drinking water. Some of these chemicals come from agricultural runoff, some fall out from the air, and many come from industrial discharges through municipal sewage systems. Presumably the same chemicals are to be found in wastewater and sludge discharges into coastal waters, but concentrations are likely to be larger in wastewater and sludge, because in drinking water the chemicals are identified "at the tap" rather than at the discharge pipe.

In addition, a large (but unknown) number of halogenated chemicals are synthesized by the chlorination process, acting on organic precursors. In the case of chlorinated drinking water, a principal feedstock appears to be naturally occurring organics in soil humus; in the case of sewage chlorination, Class I pollutants. In a mass balance of chlorination in drinking water, only about 10 percent of the total chlorine content has been identified in product chemicals. Brominated species are formed as well, due to chlorine's greater chemical activity and the presence of bromide ions. Although there appears to be less attention paid to the characterization of halogenated compounds in treated wastewater, the same situation presumably exists in the latter case, but in intensified form, for two reasons. First there is far higher organic content in wastewater and second, larger concentrations of chlorine are used in wastewater treatment.

The potential human health effect has been investigated by means of two conceptually different methods. In one method -- an epidemiologic method -- human populations have been studied, comparing cancer rates in people with high exposure to industrialized and chlorinated water with cancer rates with low exposure. In the second method, animal bioassays are used on single chemicals and the results extrapolated to estimates of human risk. Epidemiologic investigation is now in its third stage. In the first stage, whole populations were studied, with aggregate measures of exposure (for one of the first attempts at this see Page et al., 1976). These studies were primarily useful in constructing hypotheses. For example, they identified cancers of the gastrointestinal tract as of special concern and they suggested that, at least from present cancers resulting from exposures 30 to 40 years ago,

chlorination might have been a more important factor than the aggregate presence of synthetic organics of that time.

In the second stage, case control studies were conducted. These studies characterized exposures much more precisely, on an individual-by-individual basis. While a number of questions about dose-response relationships and some confounding variables remain, the case control studies have tended to confirm and sharpen evidence for a carcinogenic risk from drinking water (Crump and Guess, 1980). One of the main problems with the existing case control studies is that they are too small in scale to detect excess risks of 5 to 20 percent over the background as statistically significant at the 5 percent level ($\alpha = 0.05$). These are the excesses suggested by both the ecologic and the case control studies. In the third stage, larger case control studies are being undertaken to increase the statistical power of the tests. But even at this third stage, with sample sizes of 3000, the trade-off of α and β is unfavorable. To detect an increase of risk of 15 percent over background, with traditional probability of false positive set at 5 percent, there is still an 80 percent probability of false negative ($\alpha = 0.05$, $\beta = 0.80$).

In the second method, chemicals are identified in the water, along with their concentrations. Carcinogenic potency is taken from animal studies such as the National Cancer Institute's bioassays.[8] This effect is then extrapolated from the high dose of the experiment to the low dose in the water and extrapolated across the species lines from rodents to humans, taking into account differing metabolisms, body weight and lifetimes. The early extrapolation studies focused entirely on chloroform. Later studies included extrapolations on all the chemicals identified in drinking water as carcinogens for which animal studies exist, but chloroform remains the principal chemical of concern for drinking water.

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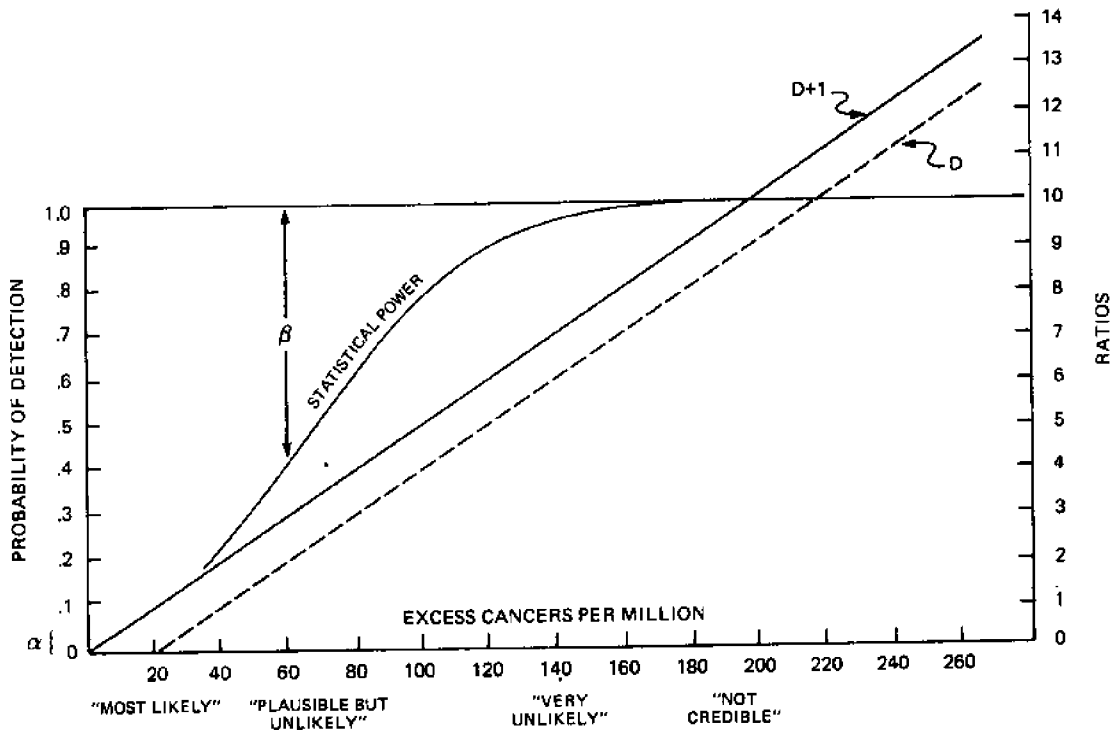
8 One definition of potency is the daily dose in milligrams per kilogram (mg/kg) of body weight leading to cancer in 50 percent of the treated animals, corrected for competing mortality and cancer in the controls.

Quantitative estimates arising from two independent data bases provide a check of one on the other. Both suggest a carcinogenic risk. Both approaches are highly uncertain in their quantitative estimates of excess risk and neither proves beyond a doubt that a non-negligible risk exists. But most interesting, the two approaches yield risk estimates that are within about an order of magnitude of each other, roughly in the range of 0.5 to 5 percent excess over background. Given the uncertainties associated with each method, probably at least an order of magnitude or two for each, the two methods agree within their range of resolution. The rough agreement between the two methods is much more suggestive than the implications of either method taken by itself. (See Crump and Guess, 1980, for a comparison of the two approaches.)

A second observation is that the risk estimated from the animal studies is considerably lower than that from the epidemiological studies. This could be due to chance, or uncertainty in the specification of the models. It could be due to the fact that the extrapolations only take into account the small fraction of chemicals yet identified. Or it could be due to the neglect of chemical interactions in the extrapolation approach. (There are tests for toxicity of wastewater taken as a whole but these are almost always for acute effects.) The comparison of risk estimates in drinking water suggests that longer term effects --such as cancer synergisms-- may be important, and therefore possible chemical interactions deserve much more consideration in the setting of standards for coastal waters.

Analysis of the implications of one of the regression equations in an ecologic study of drinking water suggests asymmetries which may be typical for environmental risks associated with ocean discharge of potentially toxic chemicals. (It appears that there are few, if any, corresponding analyses of risk for coastal discharge, so we are left with the fresh water comparison.) Figure 10.2 shows a regression relating parish (county) cancer rates of the gastrointestinal tract, for white males, in Louisiana, to the percent of the parish drinking water from the Mississippi River and other contaminated surface water compared with less contaminated groundwater (W), occupational exposure (I), and urbanization of the parish (R). The statistical analysis is discussed in more detail in Harris et al. (1977). Even though the drinking water variable is highly significant (indicating that a complete switch from ground to surface water is

Figure 10.2 Probability of detection (statistical power) and ratios of costs and benefits (Page and Bruser, 1981).



NOTES FOR FIGURE 10.2:

Statistical power, which is read on the left scale, is the probability of detection as a function of the magnitude of the effect. One minus the power is β, the probability of non-detection. The probability of (erroneous) detection, when there is no effect, is α.

D is the ratio of the cost of a false negative to the cost of a false positive. D+1 is the health damage. Both are functions of the magnitude of effect, and they are read on the right scale.

The regression equation analyzed is

$$M = 49 - 12(R) - 1.8(I) + 62(W)$$
$$\quad (9.9)\ (-4.4)\ \ (-3.8)\ \ (5.0)$$

where

M Gastrointestinal Cancer White Males
R Urbanization
I Median Family Income
W Percent Mississippi Water
t Values in Parentheses

The judgments of likelihood in quotes are qualified versions of N, the subjective odds against the environmental hypothesis. These were informal judgments, made by commentators, at the time of the original research, and never quantified.

The annual costs of control, by granular activated carbon, are estimated for a city of 1,193,000; trihalomethane concentration 250 ppb; GAC 90 percent effective removed; benefit per cancer prevented 500,000 steady-state comparison.

associated with 60 extra cancers per million population per year), the statistical power associated with this point estimate is very low. Under the assumption that the underlying model is correctly specified -- an assumption requiring examination because models are almost always misspecified in some ways -- the regression has the following interpretation. If the number of excess cancers is 140 per million, then there is about a 95 percent chance that the regression will find a drinking water effect, by producing a coefficient for the drinking water variable positive and statistically significant at the traditional level ($\beta = 0.05$ or $p < 0.05$). But if the excess mortality is 62 per million, which happens to be the point estimate of the regression, there there is only about a 40 percent probability of detection. The basic reason for low statistical power is that there are large numbers of people at risk, but the risk for each is small (the "binomial variance" problem). The "binomial variance problem" would also apply to any effort to quantify the risks from chemicals discharged in wastewater and sludge in coastal waters. In addition, the problem would apply in intensified form, due to the greater dispersal and greater numbers of people, and other species, ultimately exposed if there is bioaccumulation. The limitation of statistical power from binomial variance for this regression equation is graphed on Figure 10.2. As would be expected, the larger the actual effect, the greater is the statistical power.

The cost of precautionary control, computed for a city of 1.2 million considering drinking water filtration by granular activated carbon, was estimated by EPA to be $10.3 million annually. This cost-benefit analysis is discussed in more detail in Page et al. (1981). The annual benefit of precautionary action depends linearly on the magnitude of the cancer risk, the effectiveness of the treatment, and the value attributed to each cancer prevented. Thus the ratio of benefit-to-cost is a linear function of the size of the carcinogenic effect and is graphed on Figure 10.2. It can be seen that if there were 100 or more excess cancers per million then the chance of detection would be about 80 percent ($\beta = 0.20$); moreover, with this large an effect the benefits of control would be roughly 5 times the costs. However, the principal area of interest is not at 100 excess cancers per million. In accordance with the existing information at the time, the likelihood of such an extreme effect would be considered extremely low. The principal area of interest is where statistical power is low -- in the range of 0

to 30 excess cancers, which at the time was considered increasingly unlikely at the high end, but still plausibly credible. But where the initial assessment of likelihood becomes higher, the statistical power diminishes, and the cost-benefit ratio becomes smaller.

The problem can be compared with searching for a needle in a haystack. First of all, the needle may not be in the haystack at all (the low probability of the adverse hypothesis, or high N). But if the needle is in the haystack, it is hard to find (low statistical power in the area of interest, or high β). But if the needle is present and we miss it, we pay a high price (a high cost of the effect compared with the cost of precautionary action), or high D. Clearly in such a case it is important to analyze carefully the sources of uncertainty and to search for statistical models and tests that are sufficiently powerful to be relied upon, especially in the interpretation of negative results.

IMPLICATIONS FOR THE PROCESS OF POLICY FORMATION

The policy approach begins with questions with decision consequences, and then works back to the health and environmental effects of coastal discharge, and thence to the scientific models which predict the effects. In this approach, the value of information is the flip side of the cost of uncertainty. With uncertainty there is the risk of a wrong decision, from either a false positive or a false negative. The value of some new research or new piece of information is the reduction in the expected cost of wrong decisions. Even though our understanding of biological and chemical systems is too fragmentary to follow this approach in a fully quantitative way, it still suggests specific directions for the process of policy formation, some of which are discussed here.

Linkage

In order to evaluate the consequences of some decision, it is necessary to link the outputs of one submodel to the input of another, from waste treatment (Chapter 2) to the initial mixing (Chapter 4) to the effects (Chapters 7, 8, 9) to the control options (Chapters 11, 12). This book is a step toward linkage, as it brings together large amounts of relevant material from a wide variety of sources. The policy approach outlined in this chapter suggests continued and further attention on the linkage of the

submodels. In terms of Figure 10.1, this means further attention on links (c) and especially (b). This attention is in addition to (a), which is sometimes the consuming focus of independent scientific studies, in their concern for submodel development and validation. Dominance of the science-first approach may help explain the relatively minor discussion in this volume of two subjects -- fish tumors and the mussel-watch program. Both are mentioned as important. But each could deserve a whole chapter discussing their meaning. Because of the pervasive uncertainties these chapters would have to be speculative, defining questions and establishing hypotheses to be investigated by existing knowledge and proposed research. In the policy-first approach these chapters would come at or near the beginning and would serve in the search for links (b), (c), and (a) of the following chapters. For a substantive but basically descriptive discussion of shellfish toxicity see National Academy of Sciences (1980a); of fish tumors, Kraybill et al. (1977).

From the science-first approach it is perfectly reasonable to postpone speculative evaluation of potential harms when the basic facts themselves are speculative. The science-first approach concentrates first on the determination of the facts, leaving the evaluations until later when then they are less speculative. But a policy-first approach starts with speculative questions of evaluation.

There are several reasons for concern about fish tumors and other biological indicators:

1. In recent work by Ames and by Messelson, it is becoming apparent that different species react similarly to carcinogens, and they are likely to yield similar rankings of potency, in orders of magnitude. Besides the natural repugnance in eating cancerous meat from fish, tumor-bearing fish are likely to contain carcinogens dangerous to humans just as they proved to be dangerous to the fish. It is believed, for example, that a principal route of human exposure to PCBs is from fish consumption, especially fish from particularly contaminated bodies of fresh water.

2. It is possible that toxic chemicals may disrupt fish and other aquatic populations, affecting aggregate food resources.

3. Fish and other aquatic organisms, including plants, may provide an "early warning system" for later ecosystem disruptions. Early warning systems

are particularly important in the investigation of potential hazards to coastal discharge of wastewater and sludge, due to possible long dormancies of materials deposited in bottom sediments. Two decades ago it was thought that the mercury discharges would safely remain on the bottom. Methylization came as a surprise. A decade ago it was thought that groundwater contamination would be a minor problem due to natural filtering in the soil. In the 1980s serious groundwater contamination may become another surprise. It is useful to anticipate and investigate potential surprises before their causative factors become irreversibly beyond control.

4. Fish and other organisms allow for the study of the effects of chemicals in combination.

Beginning with the speculative questions as to the meaning of the potential effects on human health and on marine populations helps to formulate hypotheses and research agendas to make the evaluations less speculative. In doing so, it forces us to link the various submodels and effects models together. Key areas of uncertainty are identified, for which resolution would establish the links more strongly. The point <u>is</u> that starting with the evaluative questions helps focus the scientific activity along lines most useful for decision purposes. The point <u>is not</u> that the existing evidence of fish and shellfish contamination already proves that there is an enormous problem requiring immediate remedial action. When the effects are better understood, it may turn out that the contamination problems are small, requiring only a little control here and there; or it may turn out otherwise.

Combining Science and Policy Approaches

The "ultimate" policy questions listed at the beginning of the chapter do not spring from a vacuum. In large part, the existing science suggests the policy questions worth worrying about. Thus, the process of policy formation needs to encourage both approaches simultaneously, one beginning with science questions, the other with policy questions.

It is sometimes suggested that the work of the policy analysts and decision makers be kept separate from the work of the scientists, perhaps because the questions of evaluation might taint the science. A division of labor is fine, but only insofar as the divided labors fit together in a common framework.

The one offered here is that of minimizing expected cost. In this framework it is not possible to define a positive test result without making evaluations. This is because α and β trade off. Nor is it possible to define a notion of acceptable risk without at least qualitative cost and benefit evaluations. The assessment of N is a scientific judgment; the assessment of D an economic one. Yet the two play entirely symmetric roles in parts of the minimium expected cost calculation as illustrated in the appendix at the end of this chapter. This observation is not surprising because in a definition of expected value, N plays the role of a probability, D the value of a consequence (expected values are probabilities times outcome values). In the absence of observations, past definitions of a positive test have been attempted on "scientific grounds alone" and there have been similar attempts to define acceptable risk, and hence standards, as a scientific matter.

Taking these observations into account means that the communication links between the scientists and the policy analysts need to be considerably improved. In the past, sometimes it has been considered sufficient to report a test positive or negative at some significance level, usually $\alpha = 5$ percent. This has given way to reporting p values and confidence intervals. It is necessary to take the next step to report β values, for different magnitudes of effect of concern, as a routine matter (confidence intervals yield different information from β values). It is necessary for the scientist to report further information on uncertainty as well. The α and β values and the results of the test are computed under the assumption that the underlying statistical and physical model is correct. Generally, there are uncertainties as to the specification of the model, and these uncertainties must be reported as well. A test of whether or not enough information is being communicated from the scientist to the decision maker is met when the decision maker can himself define what is a positive test. Going the other way, the policy maker has the responsibility of communicating to the scientist which areas of uncertainty would have the highest payoff in resolution. There are a countless number of interesting scientific questions to pursue, and the scientist needs information as to which are the most important for decision purposes. As a part of this communication, the costs of wrong decisions need to be evaluated, in at least a rough and perhaps qualitative way, and communicated to the scientist.

Surprises are more likely when we act as though existing models were true, when in fact there is considerable uncertainty associated with most model specifications. Thus better communication of the uncertainty of model specification helps guard against surprises. Other potential surprises -- besides the classic example of methylization of mercury -- might be identified if specification uncertainties were more intensively characterized and communicated.

A vital need which becomes apparent from the policy-first approach is the need to generalize. There are simply too many chemicals to consider each on a one-by-one basis. Generalization is difficult when the underlying phenomena are not clearly understood; however, scientists have responded to this need in a creative and useful way. Chapter 6 summarizes and extends generalizations which help carry us beyond a one-by-one approach to chemicals. Particularly useful, for decision purposes, are generalizations based on an effect of concern, such as the octanol-water partition coefficient, which is a predictor of bioaccumulation. A great deal of work is now being undertaken on generalizations which would predict which chemicals are likely to be genotoxic. Another important effort in generalization, one which so far has received little resource commitment, is to investigate the general principles by which chemicals are likely to act synergistically.

IMPLICATIONS FOR SPECIFIC POLICIES

There will be further discussion of specific policies in Chapter 12, but some islands of relative certainty can be tentatively identified here. In searching for a "relative certainty" it is useful to look for places where the evidence, uncertain as it is, has tended to strengthen in a particular direction over time.

More on Toxics

Evidence has increased in the last two decades that toxic chemicals should be taken more seriously as a potential problem. This chapter suggests that traditional approaches to standard-setting and acceptable risk are less appropriate to Class IV pollutants than to the others, and thus the traditional approach is most in need of rethinking in this area.

Redirection of Secondary Treatment

To some extent requirements for secondary treatment were established to control toxic substances. However, it is becoming increasingly clear that the most effective way of controlling toxic chemicals is by pretreatment, production controls, and source reduction. To some extent, requirements for advanced treatment arose in response to the goal of minimizing discharge of all classes of pollutants into ocean waters. But in the past decade the costs of other forms of disposal of the increased volume of sludge from advanced treatment have been better understood. Incineration of sludge and landfill are seen to be increasingly more expensive and the concomitant environmental problems more apparent, thus leading to a rethinking of the reasons for and costs of advanced treatment. One otherwise attractive alternative, land use of wastewater and sludge for irrigation, soil conditioning and fertilization, largely depends on the control of Class II, III, and IV pollutants. And again, the only effective control of Class III and IV pollutants appears to be pretreatment or source control.

More Trade-offs in the Receiving Media

Legislation to control wastes was enacted, in large part, on a case-by-case basis, with ocean and river disposal considered separately from land disposal, and separately from incineration (air disposal). In dealing with the problem of waste treatment and disposal compartmentally, it was often required that the waste, in each case, go elsewhere. But in considering the problem as a whole, it is clear that it all goes somewhere, except for source reduction or treatment that transforms problem materials. In the past decade there has been increasing interest in comparing and trading off the costs of disposal in one medium versus another. This is true for all four classes of pollutants and further resolutions of uncertainty are likely to strengthen the case for managing wastes on a multi-media basis.

More Regionalization

The area of least uncertainty, in modeling the life cycle of wastewater discharges, is in the diffusion and transport of materials. A great deal is known about the way plumes move and how basins flush, or don't flush (Chapters 3, 4, 5). Regional differences

in mixing and transport are well understood and have reached the point of fine tuning. It is useful to take advantage of our knowledge of regional differences in designing treatment and discharge systems.

APPENDIX

In minimizing the expected cost of wrong decisions, we have 3 alternative strategies: gather no further information but take precautionary action on the basis of existing information, with expected cost

$$N/(N + 1); \tag{10.1}$$

gather no further information and take no precautionary action on the basis of existing information, with expected cost

$$D/(N + 1); \tag{10.2}$$

and gather further information, and act on the basis of that, in conjunction with existing information

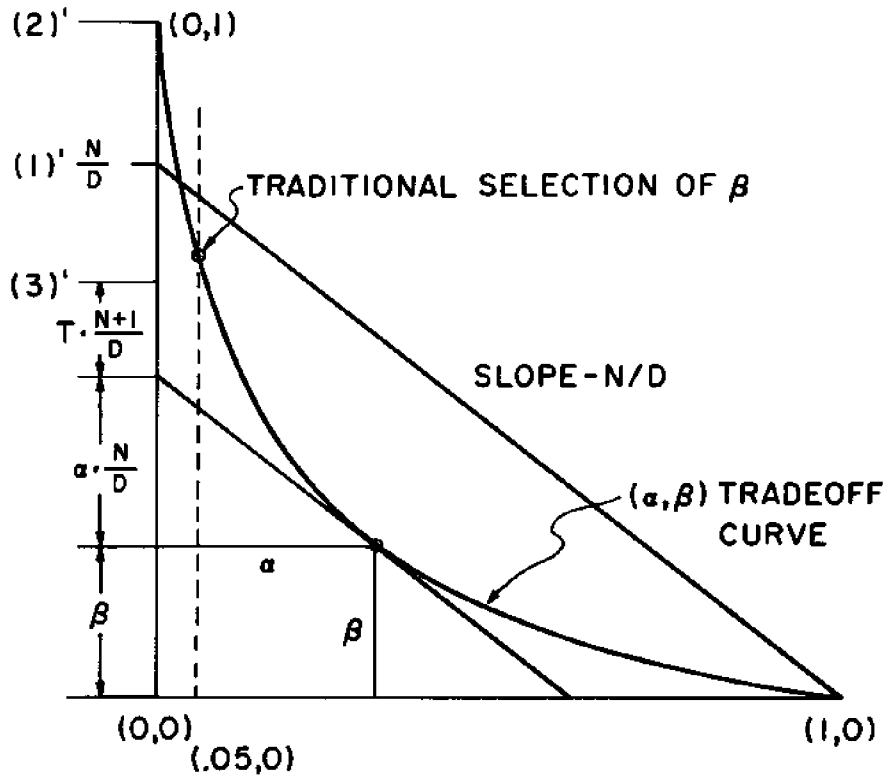
$$D\,\beta/(N + 1) + N\,\alpha/(N + 1) + T \tag{10.3}$$

In considering (10.3) we pick the most advantageous trade-off of (α,β), the pair leading to minimum (10.3). For a given cost of the test the trade-off opportunities are shown in Figure 10.3. By increasing the cost of the test (T), the whole trade-off curve can be pushed in toward the origin. The best scale of the test, or research program, is achieved when the gains from pushing the trade-off curve inward are just offset by the increased cost of gathering the new information (increased T).

For expository purposes, we are discussing a simple hypothesis and a simple alternative (a hypothesis with a single environmental effect, or no effect at all). In this case the trade-off curve defines the "most powerful test" for each significance level. In the more complicated case of many hypotheses and many alternatives the approach of minimizing expected cost is directly generalizable.

To show the relative rankings of the three strategies, and the best choice for the (α,β) trade-off, we change the scale by multiplying each of the expected costs in (1), (2), and (3) by $(N + 1)/D$. We seek the minimum of (1), (2), and (3) by seeking the minimum of

Figure 10.3 Geometry of cost minimization.



$$N/D \qquad (1)'$$

$$1 \qquad (2)'$$

$$\beta + \frac{N}{D}\alpha + T(N + 1)/D \qquad (3)'$$

By drawing the line segment from (1.0) with slope -N/D, (1)' shows up on the y-axis as marked (1)'. For comparison (2)' is just a unit up on this same axis. To compare the best (3)' with the (1)' and (2)', draw the tangent with slope -N/D to the trade-off curve. The α and β at this tangency comprise the cost minimizing trade-off of α and β for the given test cost T. Where this tangent hits the y-axis add T(N + 1)/D, to locate (3)'. Of the three strategies the one lowest on the y-axis is the least cost minimizing alternative.

It can be readily seen that any other choice of α and β other than the one picked out by the tangent with slope -N/D would have a higher expected cost than for (3)' defined by the tangency solution (the tangent leads to the lowest (3)' on the y-axis). In particular note that the traditional way of selecting the (α, β) trade-off is very unlikely to pick out the cost minimizing combination. In the traditional selection of the (α, β) trade-off, α is chosen to be 5 percent and β is chosen as whatever is compatible with this preselection. In other words, where vertical dashed line through x-axis at 0.05 crosses the (α, β) trade-off curve, the trade-off of a α and β is chosen.

The geometry of Figure 10.3 shows how the factors discussed in the chapter combine to determine the expected cost: the existing information translated into a statement of the level of suspicion about the adverse hypothesis (N); the relative costs of precaution and the potential risk (D); and the sharpness of the test or additional information in resolving uncertainty, summarized in the (α, β) trade-off curve. Figure 10.3 shows that in one part of the cost minimization N and D play entirely symmetric roles, and in another part they play different roles. Figure 10.3 suggests that it is not worth obtaining fine tuned information on D if we are very vague as to N and vice versa. Moreover, Figure 10.3 shows the difference between the approach of cost minimization and the one of classical hypothesis testing.

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11

Ocean Discharge of Municipal Wastes: Legal and Institutional Aspects

James E. Krier

INTRODUCTION

The aim of this chapter is to provide an overview and analysis of the regulatory structure governing disposal of municipal wastes. More particularly, the wastes of concern here are those contained in the effluent and sludge that result from operation of publicly owned treatment works (POTWs). While our main focus will be on waste discharge into ocean waters, we must of necessity give some attention to the other environmental media -- land and air -- as well. The wastes in question must go into one medium or another; an assessment of controls on waste discharge into the oceans that paid no heed to the implications of those controls for the other media would be seriously incomplete. Put another way, the environment is a system, and sensible environmental programs must take this into account -- they must be reasonably systematic. This is not to say that all possible trade-offs among media must somehow be identified, valued and optimized; the costs of exercising such precision would be unreasonably high. It is to say that, where manageable, environmental programs should take at least rough account of the more obvious and important interrelationships among air, land, and water as potential (and inevitable) disposal sites. As it happens, the subject under consideration here appears to be particularly susceptible to rough and ready judgments of this sort, but the opportunities for making them have been largely neglected. Federal controls on municipal waste disposal are far more fragmented, less systematic and integrated, than need be the case.

As the last sentence suggests, the emphasis here is on federal controls. In small part this is simply a matter of limits: assessment of a large number of state and local programs is beyond the chapter's ambitions. Primarily, though, the focus simply reflects the existing situation: present controls are

predominantly federal -- carried out by the federal government or, more commonly, run by state and local governments according to federal directions. This is true, at least, if one is concerned, as we are, with the central thrust of present policy, as opposed to the details of day-to-day operations. So we address federal law, especially its most generally important provisions.

The chapter begins with a sketch of the long history of events that led up to the present federal controls on ocean disposal of municipal wastes. Many of the themes in that history are more or less typical of the development of environmental policy generally; they are also suggestive of some shortcomings in the existing regulatory structure. The history, then, is more than an antiquarian exercise; it is relevant to where we are today, and why.

Following the historical background is a discussion of the most important federal controls on municipal waste disposal -- effluent and sludge alike. The purpose at this point is not a detailed description, but rather a general overview of a fairly complicated system of regulation -- an overview that reveals its central characteristics. We will then be prepared to assess the system. While doing so we shall look at some provisions of present law more closely than was the case in the overview.

Opportunities for fruitful reform of the present regulatory approach are, of course, one important concern of this book. While reforms will be suggested in this chapter, for the most part their elaboration is saved for Chapter 12.

HISTORICAL BACKGROUND

The story of municipal waste disposal and its control is interesting -- and typical. Like most histories of environmental problems (Krier and Ursin, 1977), the one of concern here suggests that the following matters played especially important parts in the sequence of developments: 1. the state of knowledge (including ignorance and uncertainty); 2. the state of technology (as problem cause and problem solution); 3. the periodic occurrence of episodes; and 4. a concern with costs (both direct and indirect). Moreover, this story, like most of its kind, reveals a gradual move of control from lower to higher levels of government (from local to state to federal).

The Development of Sewers

Before the second half of the 19th century, sewers for municipal waste removal did not exist, and treatment of any sort was virtually unknown (Hines, 1966. See also Tarr and McMichael, 1977a, 1977b, the basis for most of what follows, and Wagner, 1977 for brief historical overviews). There were a few "sewers", really just ditches and gutters, for storm and surface waters; local ordinances usually prohibited the use of these for waste disposal. By common practice, industrial wastes went directly into the nearest watercourse. Most human wastes were deposited in cesspools and privy vaults; when these receptacles filled up, the wastes were collected and dumped onto vacant land or into nearby waters. There was some spreading of wastes on farmlands as fertilizer (Wolman, 1977). Local laws regulated the building and cleaning of cesspools and privy vaults, but they were seldom enforced. The city health departments in charge of enforcement had limited funds and powers; usually only the threat of an epidemic could move them to act. Even then, just how to act was not at all clear. Medical opinion at the time was divided on the cause of epidemics; the contagionists believed the cause to be contact with diseased individuals; the anti-contagionists blamed decomposing fecal matter. Only the latter point of view supported strong enforcement of the privy vault and cesspool laws.

Between the years 1850-1880 there occurred a transition from cesspools and privy vaults to water-carriage or sewer systems as means for waste collection and disposal. Conventional wisdom attributes the transition to health concerns; more particularly, sewers were adopted in order to avoid the cholera and yellow fever epidemics that had recurred throughout the first half of the century. While the epidemics undoubtedly played a part, their role should not be exaggerated. To view health concerns as the exclusive cause suggests, for one thing, general agreement on a theory of disease, and this almost surely was not the case. Later in the century, as we shall see, disease was better understood, the understanding was far more widely shared, and health concerns were a very important factor (perhaps the primary one) influencing waste disposal techniques. But that this was so in 1850 is doubtful. The dominant cause for the transition to sewers probably related to the earlier introduction of a new water supply technology.

Prior to the 1800s, American cities obtained their water from such local sources as wells and springs. With the rapid urbanization that accompanied the 19th century, however, came a need for new supplies -- local sources were inadequate in quantity and quality (the later thanks in part to primitive waste disposal practices) to meet an increasing demand. A new technology, the waterworks, developed to meet the need by bringing water in from sources outside the cities. Philadelphia had such a system as early as 1801. By 1860 there were waterworks in the 16 largest cities and many smaller ones; by 1880 the total number of supply systems was almost 600.

As a consequence of the new technology, water consumption increased by a factor of 25 to 35 over the years 1800-1865. Caught up in the business of creating new supplies, cities neglected to provide for removal of the vastly increased amounts of water. Reliance was placed instead on the old disposal systems -- cesspools, privy vaults, and gutters. These, however, proved inadequate, and overflows became commonplace. Another new technology, the water closet, aggravated the problem. The water closet was an old idea, but one dependent for operation on the running water made possible by waterworks. The wastes from the water closets were run into the already overburdened cesspools and privy vaults (connection to storm sewers was prohibited, though done at times on the sly). The net result (gross would be a better word) of the two new water technologies was the need for a third -- a system of water carriage, or sewerage, for waste removal. Tarr and McMichael (1977a) summarize the sequence nicely:

> The introduction of a new technology -- running water -- resulted in the adoption of another new technology -- the water closet. The unanticipated result of the adoption of these technologies was to upset the existing system of waste disposal and to create problems of both sanitary nuisance and higher out-of-pocket costs for cesspool cleaning...Increasingly, in the period after the Civil War, as more cities installed running water closets, urban policymakers became convinced that the only method to deal with the wastewater problem was to adopt another new technology -- the installation of sewers that transported wastes by water carriage.

Sewerage, then, was a product more of technological opportunities than a concern with

health. Sewer systems developed out of convenience and were designed in a state of high ignorance; mostly to save costs, partly because so little was known about disease, the new water carriage technology was employed in a way that solved one set of problems only to create another. Efforts to solve the new problems -- efforts guided by a gradually increasing understanding of water pollution and its health consequences -- led directly to present day treatment programs.

Combined versus Separate Sewers

The decision for sewers did not exhaust all choices facing municipalities. Two different systems -- separate and combined sewers -- were available, and there was considerable debate about their relative merits. Combined sewers, with a single pipe carrying both storm water and wastes, had one outstanding virtue: they were cheaper than separate systems dependent on two pipelines, one for each sort of water. The cost advantage of combined systems was sufficiently attractive that virtually all of the early sewers (built mostly in the years 1860-1880) were of this sort. The sewers discharged directly into waterways, reflecting the prevalent attitude that running water purified itself.

Separate sewers became more common in the years after 1880, though combined systems, it appears, continued as the leading technology (at least in terms of total amounts of waste transported). Health hazards alleged to be associated with combined sewers were, for a short time, a factor favoring adoption of separate systems; health hazards actually associated with both technologies later led to new treatment practices.

In 1880 George Waring, a well-known sanitarian, designed a separate system for Memphis, Tennessee, as a means to avoid recurrences of the yellow fever epidemic that had hit the city a few years earlier. Waring attributed the health problem to lethal sewer gases given off by human wastes as they decomposed. He believed that the problem could be prevented by moving wastes out of the city quickly, while they were still in a "fresh condition." A separate system was the fastest means to transport wastes, hence it was the desired technology.

Waring promoted his system with avidity and some success: by 1892 over 20 towns and cities had adopted it. It is probable, however, that cost had as much to

do with these adoptions as did health concerns, for the following reasons. Combined systems were cheaper than separate sewers if the latter used two pipes -- one for wastes and the other for storm waters. However, a small separate pipe for wastes alone (leaving storm waters to gutters and the like) was cheaper than a combined system. Waring believed that a storm water pipe was unnecessary; thus cost considerations alone would support a choice for his system. But the same sort of cost considerations weighed against Waring's approach in places where surface removal of storm waters was unacceptable -- namely, large cities. Here Waring's approach would necessarily require the expensive alternative (relative to a combined system) of two separate pipes, and thus would likely be disfavored unless a city considered the health advantages claimed by Waring were worth the added expense. But cities considered otherwise. Indeed, Waring's theory regarding the role of sewer gases had been questioned virtually from the outset, and by the early 1890s professional opinion was of the view that combined and separate systems were equally effective in health terms. The choice between them thus came to turn on other factors, primarily cost. Small towns opted for the Waring system with a small separate pipe for wastes only, larger cities for combined sewers. Whatever the carriage technology, ultimate disposal typically followed the long-established practice: wastes were usually directed to the nearest available waterway.

"Cities disposed of their sewage through dilution in watercourses because it was convenient, cheap, and because the technology of sewage treatment was expensive and often uncertain" (Tarr and McMichael, 1977a). An added factor was the poor understanding of relationships between water pollution and adverse health effects. As the germ theory of disease gained acceptance, however, public health opinion came to doubt, rather generally by the 1890s, the safety of discharging sewage into waterways. Professional doubts found confirmation in the rising typhoid death rates in downstream cities that drew water from rivers receiving sewage from upstream sewer systems. The result was a new interest in sewerage purification, and in state legislation to control waste disposal practices.

Early State and Federal Legislation

Early state water quality legislation did not reflect a high degree of centralization: with a few exceptions, the general pattern until 1920 consisted

of local agencies established under state enabling legislation. Only as the uncoordinated activities of these "singularly local" agencies proved insufficient did actual control authority go to central state agencies (usually departments of health), a transition largely accomplished by about 1930 (Hines, 1966).

Protection of health as opposed, for example, to recreational interests was the exclusive concern of early local and state agencies (Hines, 1966; Tarr and McMichael, 1977b), and sewage treatment prior to disposal was the means selected to realize it. The choice reflected sharp disagreement between the agencies (most of them run by public health professionals) and sanitary engineers. The latter favored stream dilution as the primary means of purification, with the added step of water supply filtration where necessary for safe drinking water; the former favored sewage treatment and water filtration, so as to have a "double standard of protection." Engineers considered sewage treatment unduly expensive, especially with regard to the combined sewers carrying the major portion of municipal wastes. Good water, in their view, was the problem of downstream cities, which could filter water at a far lower cost than upstream cities could treat sewage. The engineers prevailed, as local and state agencies tended to back down on treatment demands, at least as to large cities. A chief consequence was that the construction of sewage treatment plants remained a low priority during the 1920s. "Between 1910 and 1930, for instance, new sewerage was provided for 25 million people, but sewage from only 13.5 million was treated" (Tarr and McMichael, 1977b).

Treatment plants received much more emphasis after 1930, initially as the result of New Deal public works programs. The federal government offered millions of dollars for construction, and states and localities accepted eagerly. While a concern with pollution was probably not the primary motivation behind the public works programs, improved water quality was nevertheless one of their central consequences. "As a result of the New Deal expenditures, combined with the reduction in economic output caused by the Depression, the years 1930 to 1940 marked the first actual reduction in water pollution in this country" (Davies, 1970).

The New Deal programs laid the foundation for major federal legislation that was to follow; they did not, however, mark the national government's first involvement with water pollution. Congress prohibited

the dumping of refuse into New York Harbor in 1886. Legislation in 1899 (the Refuse Act) went further by regulating discharges of wastes into navigable waters. Both Acts were concerned with impediments to navigation, rather than with pollution, though they were to be applied in behalf of water quality many decades later. The federal government began stream pollution investigations in 1910, and in 1912 the United States Public Health Service was authorized to investigate pollution of navigable waters; the concern was with waterborne diseases like cholera and typhoid. By the mid-1920s, and probably as a result of the success of such water supply treatment processes as filtration, federal attention was redirected to a new problem: oil pollution. Ships followed the practice of discharging oil into offshore ocean waters, damaging shellfish, polluting beaches, and creating fire hazards around harbors. The Oil Pollution Act of 1924 prohibited oil dumping into navigable coastal waters (Davies, 1970; Hines, 1967).

The years after 1924 witnessed considerable congressional attention to water pollution, but until 1948 no final legislation. Some bills got through one house of Congress but not the other; some passed both houses but died in conference; one found its way out of Congress only to be vetoed by President Roosevelt in 1938. Despite his action, Roosevelt endorsed the idea of a federal pollution control program (he had vetoed the bill for reasons having to do with his view of proper relations between the legislature and the executive). He wanted a federal agency, and federal grants and loans, to promote, coordinate, and support state and local water pollution control efforts. Roosevelt's enthusiasm generated the introduction of more legislation between 1940-45, but a Congress caught up with the war gave the bills little attention. A preoccupation with other matters also marked the period immediately after the war. The water pollution problem had grown worse, thanks to the intensity of wartime industrial activities, "but pollution control legislation more or less became lost in the great mass of postwar legislative measures" (Hines, 1967, p. 809). But this situation lasted for only a short time. A Federal Water Pollution Control Act (FWPCA), the first important federal legislation in the field, became law in 1948.

We can better understand the 1948 Act, the ancestor of today's federal program, if we see how it fit into the pattern of state and local control that existed at the time. As mentioned earlier, water pollution control was a distinctly local matter well

into the 20th century. State legislation existed, but generally only as a means to enable local controls. The shortcomings of the local approach -- cities could not control upstream sources, and had little incentive to control in favor of towns downstream -- led gradually to centralized (state) control authorities. By 1930 most states had a central authority; by 1948 all did. Usually the authority was a health department, and usually, as recounted above, the department was impatient with filtration by downstream users as the sole means to achieve pure water. The central concern during these years was with health. According to Hines (1966):

> A typical state act of this period conferring pollution control responsibility on the department of health described the types of pollution conditions subject to regulation, and made the health department responsible for investigating suspected pollution situations and for causing the institution of proceedings to abate the activities creating the pollution. Often the statute contained a pollution prevention measure in the form of a permit requirement for construction or change of waste disposal facilities.

After 1930, treatment works for municipal wastes became an increasingly important part of the total picture of control.

The Federal Water Pollution Control Act of 1948, then, became law in a context of much state and local activity. Consistent with this, and with the views of Roosevelt expressed earlier, the legislation declared the federal role to be one of support for ongoing efforts. Federal assistance took such forms as research and dissemination of information, investigations (at local request), and loans for treatment works (the Senate bill had provided grants for this purpose, but the largesse was curtailed by the House). The federal government did have some (narrow) enforcement authority, however: the Surgeon General could act to abate certain interstate pollution conditions endangering health and welfare. He could do so, though, only at local request, and only then after observing a bulky, time-consuming set of procedures.

Federal Legislation after 1948

In the years after passage of the Federal Water Pollution Control Act of 1948, there emerged two significant (though not necessarily related) trends. The first of these regarded activities at the state level. Beginning about 1950, state water pollution control efforts shifted from their narrow focus on health effects to a broader concern with industrial, agricultural, recreational and municipal uses. One sign of the change in emphasis has been the steady demise of state health departments as the agencies in charge of water pollution control; independent pollution control agencies of various sorts have generally taken their place (Hines, 1966; Davies, 1970). A second sign of the change is the wide range of authority in the new agencies to develop comprehensive plans, conduct investigations and research, establish water quality standards and effluent standards for multiple uses, issue and enforce permits, and approve plans for waste treatment facilities.

The second trend, and the one most important for our purposes, had to do with federal activities. The 1948 Act marked the first stage in a gradual reallocation of water pollution control authority from the states to the national government. Just as, at an earlier time, responsibility had moved from localities to state capitals, it has since largely shifted from the states to Washington. Here, as in the case of other environmental media (Krier and Ursin, 1977), increasing centralization has been the norm. It bears mention at this point that there is a large and important difference between state centralization on the one hand and federal centralization on the other; the former implies less uniformity of treatment. Partly this is because individual states have tended, more than has the federal government, to adopt an area-by-area, or variable approach to water pollution control; partly it is because the total pattern of state control would be variable even if each state adopted its own uniform standards, unless all states adopted identical standards. The states have not done so: "The state pollution control programs, as reflected in the statutes that create them, reveal about the amount of local variation that should be expected in regulation of a problem that varies significantly among the states in its causes and effects" (Hines, 1966). We save for later a discussion of the advantages of nonuniform treatment, and of the implications of those advantages for the shape of the federal program.

By briefly considering the evolution of federal water pollution control policy from 1948 to the present, we can describe the major pieces of legislation and, at the same time, document the increasing degrees of centralization, federalization, that they reflect.

In 1956 Congress amended the 1948 Act in ways which (though the amendments reaffirmed the proposition that states had primary responsibility for water pollution control) made clear that the federal role was to be significantly expanded. The 1948 legislation had provided only temporary authority -- it had been due to expire in 1953 but was subsequently extended to 1956. The 1956 amendments were permanent. They changed the cumbersome enforcement provisions of the 1948 Act by adding a conference procedure and removing some of the constraints on federal initiative that had previously existed. Of more interest here, the POTW construction loan program of the 1948 law (a program which, by the way, had never been funded) was replaced by grants-in-aid with authorizations of $50 million annually. The grants-in-aid provision was controversial. Eisenhower considered but did not exercise a veto of the 1956 amendments on that ground; four years later he did veto a bill which would have doubled the grant authorization. Another controversial provision never found its way into the 1956 legislation. A bill proposed in 1955 would have given the Surgeon General power to establish water quality standards for interstate waters. That provision was killed in the Senate, and the balance of the bill did not move out of Congress prior to the end of the session. A new bill (which eventually became the 1956 amendments) was introduced the following year, but by that time the idea of federal water quality standards had been abandoned.

Declining interest in water quality standards, together with Eisenhower's 1960 veto of increased construction grant authorizations, were the last significant victories of those opposed to an expansive federal role in water pollution control. A national conference on water pollution held in late 1960 urged active federal involvement in general, and substantial federal financial support of POTWs in particular. The following year a Senate select committee implicitly endorsed the latter. John F. Kennedy, the new president, was of similar views, a fact reflected in an administration bill submitted in 1961. The bill, with some changes, became law the same year. It amended the 1948 and 1956 legislation by, among other

things, transferring federal water pollution control responsibilities from the Surgeon General to Health, Education and Welfare (HEW); establishing a program of research on municipal treatment; extending federal enforcement authority; and increasing the authorization for construction grants up to a maximum of $100 million annually in the years 1964-67 (the ceiling on grants for individual projects was also raised).

Having accomplished this much, there were those in Congress who soon wanted more. The states, they felt, were playing only a reluctant part in protecting the nation's waters, and the Public Health Service (the agency within HEW responsible for water pollution control) was not doing much to push them along. To remedy these deficiencies, a new figure, Senator Edmund Muskie, had especially ambitious plans: a step-up in the construction grant program; creation of a Federal Water Pollution Control Administration within HEW; federal water quality and effluent standards for interstate and navigable waters.

Muskie's proposed bill, especially as it regarded water quality standards, was controversial. It passed the Senate easily enough in the year it was introduced, 1963, but spent much of the next year in the House. When it finally cleared committee in 1964 there was too little time for House action. Muskie introduced a new bill in 1965, and an amended version worked its way through Congress and became law. The Water Quality Act of 1965 created the new agency Muskie had proposed two years earlier, increased once again (to $150 million annually) the authorization for construction grants, doubled the ceiling for individual grants, authorized research and development grants for problems related to combined sewers, slightly extended federal enforcement authority, and provided for water quality standards. The standards, it should be noted, were left in the first instance to the states, each of which was to file a letter of intent to the effect that it would set water quality standards for interstate waters by 1967, subject to federal approval. If a state defaulted in some way (set inadequate standards, for example, or none at all), the federal government could, through an elaborate set of procedures, establish standards of its own. The standards, whether set at the state or federal level, consisted of water quality criteria plus a plan to implement the criteria. A body of water would be classified in terms of certain uses (water supply, recreation, agriculture, industry, etc.), criteria appropriate to the particular uses in

question would be specified (in numerical or narrative terms), and a plan to implement -- achieve -- the criteria would set forth abatement measures required of the sources using the body of water in question.

In 1966 the federal water pollution control program changed once again, partly by virtue of executive action and partly through new legislation. First, the Federal Water Pollution Control Administration created just a year earlier was transferred from HEW to the Department of Interior. Interior already had jurisdiction over a range of water resource concerns, and the move was made in order to achieve greater coordination of the total federal effort; moreover, Interior had (it was said) a conservation orientation that "matched well the broadened concept of water quality demands reflected in the 1965 Water Quality Act" (Hines, 1967). There may have been another motivation behind the transfer. The Democratic administration was less interested than the Congress in a state-federal partnership and had, indeed, prepared legislation which would take a regional approach to pollution control, an appproach that would leave less initiative and authority to individual states. The transfer from HEW to Interior was consistent with the new approach, since "the latter was much less committed to working through the states than HEW had been" (Davies and Davies, 1975).

The second change in the federal water pollution control program was accomplished through legislation. As just remarked, the administration wished to move toward regional control, and it proposed legislation to that effect. Congress, and Senator Muskie in particular, disliked the idea, favoring instead the approach of state-set water quality standards, subject to federal approval, that had been adopted in 1965. At the least, Muskie thought, that approach should be given a chance. He produced a bill which watered down virtually to nothing the administration's regional proposal, but which at the same time stepped up federal spending for POTW construction. It was essentially Muskie's view which prevailed. The Clean Water Restoration Act of 1966, among other things, increased construction grant authorizations to a total of $3.55 billion for fiscal 1967-71, and increased the maximum federal share of POTW construction costs from the existing 30 percent to a maximum of 55 percent (under specified conditions).

The federal funding provisions just described were intended to give states incentives to set water quality standards and to contribute state funds (on a

matching basis) to local construction of POTWs (Hines, 1967). These and earlier provisions, together with administrative practices in the Federal Water Pollution Control Administration, also had a more particular effect: that of encouraging secondary treatment. According to a recent report (National Academy of Sciences (NAS), 1978), the federal chairman of conferences held under the procedures adopted by the 1956 amendments to FWPCA (see above) "recommended that secondary treatment...be required everywhere and sought agreement from the state water pollution control agencies." These recommendations were hardly law, but the Federal Water Pollution Control Administration took them seriously in making construction grants; aid might be denied a project proposal that contemplated a lower (e.g., primary) level of treatment. Later, the Water Quality Act of 1965 provided an additional basis for secondary treatment. The Act, recall, required state water quality standards, subject to federal approval. As matters developed, a state risked disapproval, not to mention loss of funding, if it set standards for receiving waters based on less than secondary treatment of municipal wastes.

We are now virtually in position to sum up the historical background and move on to a discussion of the present regulatory regime. With one exception, what legislation remains to be addressed -- and not all of it relates directly to water pollution -- is best considered in the context of the present rather than the past. Even the exception need only be mentioned, for the sake of completeness. In 1970 the Water Quality Improvement Act became law. It dealt with two problems, oil spills and thermal pollution, which had gained in importance over the years, yet had been more or less neglected by Congress (the Torrey Canyon episode in 1967 helped call attention to the oil pollution problem in particular; the Santa Barbara oil spill in 1969 added impetus). The Water Quality Improvement Act, passed after a long period of legislative skirmishing, placed liability for most oil spills on vessel owners; as to thermal pollution, it established procedures to assure that facilities operated by federal licensees would not emit undue amounts of heat into navigable waters.

Summary of the History

In the years through 1970, four factors mentioned at the outset of this section -- knowledge, technology, episodes, and costs -- played an important part in shaping the pattern of municipal waste disposal

practices. Early disposal techniques reflected utter ignorance of the health or other effects of sewage. Changes in these techniques, to sewerage in particular, were probably motivated not by health concerns but by the incompatibility of existing waste collection systems with the increased flows brought on by waterworks. Thus the introduction of one new technology (waterworks) stimulated the introduction of another (sewerage). Sewer design and operation were affected to some degree by (divided) opinions about the health implications of alternative approaches, hence the debates on combined versus separate sewers, and on sewage treatment versus water filtration. But by and large, these debates were resolved in favor of those approaches that entailed the lowest (direct) costs. Nevertheless, there now ran through the arguments on all sides of the questions an acknowledgment of the adverse health effects that could result from untreated sewage.

Increased awareness of health implications, together with the influence of programs to recover from the Great Depression, led to heightened sewage treatment efforts (efforts that were, we should note, distinctly technological in character; throughout the period canvassed, technical fixes were increasingly seen as the solution to waste disposal problems, and this remains largely the case today). Accompanying these efforts were a broadened concern with water quality, a concern that ranged beyond health considerations, and a broadened locus of authority -- a locus that eventually moved, in important ways, to Washington. By 1970 the federal government occupied a fairly conspicuous position with regard to water pollution control. The 1948 legislation had begun a tradition of federal financial support for POTW construction in particular, and subsequent Acts (bearing various titles, but in each case serving to amend the original FWPCA of 1948) expanded the federal presence in this and other regards. Yet one could still describe the federal role as relatively modest when compared, say, to the Clean Air Amendments of 1970, legislation that went so far as to establish federal air quality and emission standards. In this perspective, water quality legislation lagged behind; there was merely provision, after all, for state water quality standards subject to federal approval, a procedure established by the Water Quality Act of 1965.

This situation was soon to change. Even in 1969 it was clear, according to standard accounts (see e.g., Davies and Davies, 1975), that FWPCA needed revision and expansion once again. Progress under

existing water quality legislation had been disappointing. The "environmental crisis" had generated pronounced public concern about pollution generally, and the new Republican administration had adopted environmental quality as a central issue. In early 1970 Nixon called for federal effluent standards to limit industrial and municipal water pollution, and for an increase to $1 billion annually for POTW construction funds. Congress responded slowly but nevertheless dramatically. In late 1971 the Senate passed a measure that went far beyond Nixon's proposals, a fact established by the Senate bill's policy declarations calling for elimination of pollutant discharges into navigable waters by 1985, with greatly improved levels of water quality as an interim goal. To realize these ambitions, the bill proposed stringent effluent standards to be set by the federal government (thus greatly diminishing the states' role), plus an authorization of $14 billion over four years for construction of POTWs. Enforcement was to be based on a permit system similar to one developed by the federal government in 1970 in its (fairly successful) effort to use the Refuse Act of 1899 as an additional means of water pollution control. Administration would be carried out by the new Environmental Protection Agency (EPA) established in 1970 and assigned most environmental protection functions previously exercised by other federal agencies.

Despite administration opposition to the levels of spending contemplated by the Senate bill, the House reached even further into the federal budget, passing (in 1972) a measure which, though weaker than the senate bill in several respects, raised the authorization level for waste treatment grants to $18.4 billion. Together with other authorizations, the total price tag of the House bill was almost $25 billion (compared to a total of $18 billion for the Senate bill). The compromise worked out in conference retained the House figure and was subsequently approved by Congress. Nixon vetoed the bill and this in turn was overridden, resulting in enactment of the Federal Water Pollution Control Act Amendments of 1972. The legislation, together with subsequent amendments, is the centerpiece of present water quality controls. We consider its most important provisions below.

OVERVIEW OF THE PRESENT SITUATION

We turn now from past to present, with two changes in emphasis. First, while the historical narrative in the preceding pages dealt with water generally, the main concern in the balance of this chapter is ocean waters in particular. But second, while the history considered only water, the discussion that follows must give some attention to land and air as well. The present federal program constrains disposal of municipal wastes -- effluent and sludge -- into the oceans; it also limits, however, disposal onto land or into the air. The sense of controls to protect one medium is to some degree dependent on the controls intended to protect the other media; the system must be assessed more or less as a whole. This section gives a brief overview of the federal program as it bears on ocean waters, land, and air respectively. More detailed treatment of some particularly important provisions is taken up in the course of the assessment in a later section. Table 11.1 sketches the legislation to be discussed.

Water

Of chief concern with respect to protection of ocean waters (from both effluent and sludge) are: 1. the Federal Water Pollution Control Act Amendments of 1972, together with 1977 amendments (commonly called The Clean Water Act), and 2. the so-called Ocean Dumping Act and Ocean Dumping Convention. (The Safe Drinking Water Act, intended primarily to protect groundwater used for public supplies, will be discussed in connection with land, inasmuch as it limits disposal onto land.)

The Federal Water Pollution Control Act Amendments of 1972 and the Clean Water Act of 1977. These two pieces of legislation are best considered together. The 1972 Act accomplished a major restructuring and expansion of the federal role in water pollution control. The 1977 Act amended the federal program in a large number of particulars. Most of the amendments, however, concern minor details; the basic structure of the 1972 legislation has been essentially preserved.

The Federal Water Pollution Control Act Amendments of 1972 contained a raft of far-reaching provisions; the legislation has been called "the most sweeping environmental measure ever considered by the Congress" (Davies and Davies, 1975). Congress was

Table 11.1 Federal Legislation on Municipal Waste Disposal

Date	Title	Medium Protected	Relevant Provisions
1972	Water Pollution Control Act Amendments of 1972 (Pub. L. 92-500)	Water	Technology-based effluent limitations; controls on toxic pollutants; secondary treatment and beyond for POTWs; permit program to assure compliance with effluent and water quality requirements; pretreatment requirements for sources discharging into POTWs; expanded financial support for POTW construction.
1977	Clean Water Act of 1977 (Pub. L. 95-217)	Water	Extensions of technology-based effluent limitation deadlines; increased emphasis on toxic pollutants; extensions and waivers of POTW secondary treatment requirements in certain cases; pretreatment credits of POTW removal of pollutants; changes in and expansion of federal financial support for POTW construction, including special incentives for innovative technology.
1972	Marine Protection, Research, and Sanctuaries Act of 1972 (Ocean Dumping Act, Pub. L. 92-532)	Water	Permit program for ocean disposal of wastes.
1977	Amendments to Ocean Dumping Act (no title, Pub. L. 95-153)	Water	Prohibition of ocean dumping of municipal sludge after 1981.

(Table 11.1 - continued)

Date	Title	Medium Protected	Relevant Provisions
1975	Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter (Ocean Dumping Convention)	Water	Similar to Ocean Dumping Act
1976	Resource Conservation and Recovery Act of 1976 (RCRA, Pub. L. 94-580)	Land	Assistance for solid waste management plans; prohibition of open dumping; conversion of open dumps; regulation of hazardous wastes; promotion of improved solid waste practices.
1974	Safe Drinking Water Act of 1974 (Pub. L. 93-523)	Groundwater	Controls on ground injection of wastewaters and other waste disposal practices.
1976	Toxic Substances Control Act of 1976 (Pub. L. 94-469)	All	Controls on introduction, distribution, use, and disposal of toxic chemicals.
1977	Clean Air Act Amendments of 1977 (Pub. L. 95-95)	Air	Air quality and emission standards; controls on waste incineration; emission-offset requirements for nonattainment areas; conditions on POTW construction grants.

aware that the law, thanks to its complex and demanding requirements, might well give rise to problems unforeseen at the time of passage. Accordingly, it created a National Commission on Water Quality to assess the new program and recommend any appropriate mid-course corrections. Based in part on the Commission's report (National Commission on Water Quality, 1976) and also on a comprehensive set of proposals by the EPA (with the approval of the Carter administration), the House and Senate approved bills which, after conference, became the Clean Water Act of 1977. As mentioned above, the Act amended the 1972 legislation in many respects, but nevertheless kept its basic structure intact. The following summary describes the 1972 provisions and the 1977 amendments of most importance to us here -- those that bear, directly or indirectly, on ocean disposal of municipal wastes. (For extended discussion of the 1972 legislation see, e.g., Rodgers, 1977; Zener, 1974. On the 1977 amendments see, e.g., Boytko, 1977; Hunciker, 1977; Hall, 1978.)

As we saw earlier, federal water pollution control legislation prior to 1972 relied on ambient water quality standards as the primary regulatory mechanism. The new federal program, begun in 1972, has not abandoned water quality standards; now, however, they play only a secondary role. Effluent limitations based on technology rather than the quality of receiving waters have become the mainspring of the federal approach. Under the terms of the Federal Water Pollution Control Act Amendments of 1972, existing industrial sources (other than those discharging into POTWs) were to achieve effluent limitations based on "best practicable control technology currently available" (BPCT) by 1977, and standards based on "best available technology economically achievable" (BAT) by 1983. New sources were to meet standards of performance reflecting "best available demonstrated control technology, processes, operating methods or other alternatives." In addition, special standards for toxic pollutants -- based not on technological factors but rather on environmental effects -- were to be established.

Regarding POTWs, those in existence as of mid-1977, or approved for federal financing before mid-1974, were to adopt "secondary treatment", and all POTWs were to employ "best practicable waste treatment technology" (BPWTT) by mid-1983. These standards applied to sludge as well as effluents.

Effluent standards are set by the EPA pursuant to legislative guidelines. The EPA also enforces standards, primarily through the issuance of National Pollution Discharge Elimination System (NPDES) permits. The NPDES permits, which may be issued by states with permit programs meeting federal requirements, specify the limitations with which direct dischargers (industrial or municipal) must conform in order to be in compliance with the law. Limitations relate not only to the effluent standards discussed above, but also to water quality standards and ocean discharge guidelines. Regarding water quality standards, the procedure that had existed prior to 1972 -- state-set water quality standards subject to federal approval -- was essentially retained. Under the 1972 legislation, however, if technology-based limitations were more than sufficient to meet water quality requirements, still those effluent limitations were, generally speaking, to be met. If, on the other hand, the 1977 limitations were insufficient to meet state water quality standards, or if the 1983 limitations would not achieve so-called fishable-swimmable water quality, then further controls were required. Moreover, sources discharging into ocean waters were made subject to ocean discharge guidelines promulgated by the EPA.

A different set of controls and procedures than those described above (pertaining to direct dischargers) was designed for sources discharging into POTWs. Such sources were not required to obtain a permit; rather, the POTW's permit contains provisions regarding introduction of effluents into the treatment works. Moreover, pretreatment standards applicable to sources discharging into POTWs were intended to protect against pollutants which would interfere with, pass through, or otherwise be incompatible with POTW treatment processes. As new source performance standards were promulgated for direct dischargers, equivalent pretreatment standards would be established to put sources on an equal footing and to ensure effective controls. Finally, dischargers into POTWs were required to pay a share of the capital and operating costs of treatment works. The general idea behind all these provisions was to protect and maintain POTWs, and to put roughly on a par -- in terms both of costs and environmental effects -- industrial sources that discharge directly into receiving waters, and those that discharge into treatment works.

As indicated at the outset of this discussion of the Federal Water Pollution Control Act Amendments of 1972, the legislation continued, indeed expanded, the financial support for POTW construction that has been a feature of federal law since 1948. The federal share was once again increased, from the previous maximum of 55 percent to a new figure of 75 percent, provided, among other things, that alternative waste management techniques were considered, and that the new treatment works would employ "best practicable waste treatment technology" and allow, to the extent practicable, later application of advanced technology.

The 1977 amendments have changed the foregoing in a number of respects. Primarily they reflect a more pronounced concern with toxic pollutants, heightened interest in alternative waste treatment techniques, reassessment of some of the stringent effluent limitations of the 1972 legislation, and reaction to delays in federal funding of POTW construction. Regarding direct discharges from industrial sources, provision has been made for extensions of the 1977 and 1983 deadlines, and toxic pollutants have been brought more directly into the revised framework. Industrial sources which failed to meet the 1977 requirements could get extensions until 1979 if they acted in good faith, made commitments to achieve compliance as soon as possible, and had the necessary facilities under construction. Extensions to 1983 are also available in several situations: where discharge into treatment works had been planned, construction of the works has been delayed, and the works themselves have received extension of their own treatment requirements; and where, for sources that cannot meet these requirements, the EPA concludes that discharge into a POTW is the best means of control.

Changes regarding the 1983 requirements for industrial point sources are more complicated and involve a new strategy regarding toxic and other pollutants. Under the new strategy, pollutants are classified into three categories -- conventional, toxic, and nonconventional. Conventional pollutants (e.g., BOD, suspended solids) are to be controlled to a degree that reflects "best conventional pollutant control technology" (a standard more or less in between the original 1977 and 1983 requirements), and the applicable deadline is extended to 1984. As to toxic pollutants, the 1977 amendments require that sources achieve limitations based on BAT (the original 1983 requirement) by 1984, or, with respect to toxics subsequently listed, within three years after

limitations are promulgated. Moreover, the EPA may specify "best management practices" regarding toxic pollutants. Regarding nonconventional pollutants -- those not included in either of the two categories just discussed -- the original BAT requirement has been retained but the 1983 deadline extended to 1984 or three years after limitations are established, whichever is later (but no later than 1987). Moreover, several variance provisions may ease these requirements.

Effluent limitations regarding POTWs have also been changed, but to a more limited degree, by the 1977 legislation, partly in response to federal funding delays, partly to take account of the assimilative capacity of certain ocean waters. The 1977 requirements may be extended to 1983 where POTW construction could not be completed by the 1977 deadline, or where the federal government failed to make financial assistance available in time to meet that deadline. An extension may involve interim effluent limitations and a schedule for working up to secondary treatment. Moreover, the 1977 secondary treatment requirement may be waived by the EPA (with state approval) in cases of existing discharges into marine waters if certain requirements are met (the requirements concern water quality, monitoring, pretreatment, toxic pollutants, and the volume of waste discharged). The legislative history of the Clean Water Act of 1977 suggests that this waiver provision might well be limited to ocean waters on the West Coast. The provision has the effect of overruling in part a federal court decision (Pacific Legal Foundation vs Quarles, 440 F. Supp. 316, Central District of California, 1977) that found illegal Los Angeles' practice of discharging into the ocean effluent and sludge that failed to meet secondary treatment requirements. (The 1977 amendments make clear that disposal of sludge into waters is covered by NPDES requirements, and require that guidelines for sludge disposal and utilization be developed.) It is the position of the EPA that POTWs may not discharge sludge into ocean waters, whether or not secondary treatment requirements are waived.

It should be noted that the foregoing extensions and waivers apply only to the 1977 requirements originally established by the 1972 legislation; the 1983 requirements still apply. Thus, compliance with BPWTT and the various water quality standards might be required as of 1983. The fact that BPWTT is presently

defined as the virtual equivalent of secondary treatment makes ambiguous the significance of the 1983 deadline.

The foregoing discussion concerns direct dischargers. Regarding sources that discharge into treatment works, the 1977 amendments have accomplished several significant changes. Pretreatment standards may be modified in cases where a POTW removes all or some of a toxic pollutant (the source in essence gets a credit for the amount removed) if the resulting discharge from the POTW does not violate the effluent limitation for the pollutant in question, and if the POTW's sludge is not contaminated as a result of the treatment process. Permit and enforcement provisions as they regard pretreatment have been strengthened, though it remains the case that dischargers into POTWs are not themselves directly covered by the NPDES program. An 18-month moratorium was placed on the POTW construction cost recovery system established for industrial users by the 1972 legislation. Moreover, small industrial users have been exempted from the system. In addition, with regard to residential users, operation and maintenance costs may be recovered through ad valorem taxes rather than user charges applicable to most nonresidential users.

Finally, concerning federal financial support of treatment works, the 1977 amendments have made a number of changes designed to promote the use of new technologies -- in particular, to encourage reduction, reuse, recycling, and land disposal of effluent and sludge. Additional federal funds have been authorized, the federal share for innovative and alternative treatment processes has been increased from 75 to 85 percent, and grant applicants are required to evaluate such processes fully.

The Ocean Dumping Act and the Ocean Dumping Convention. The Ocean Dumping Act is the common name for the Marine Protection, Research, and Sanctuaries Act of 1972. Ocean dumping of wastes -- disposal, that is, from vessels as opposed to outfalls -- has been covered by federal law since 1886 but, until recently, in only a fragmentary way. Ocean waste disposal increased markedly in the 1950s and 1960s, much of this being accounted for by sewage sludges and industrial wastes from sources in the northeastern coastal area of the United States, an area where land disposal has proved expensive and impractical (Lettow,

1974; Lumsdaine, 1976). The increased dumping led directly to the Ocean Dumping Act, and probably to the Ocean Dumping Convention as well.

The Ocean Dumping Act requires federal permits for the disposal of wastes into ocean waters. Under the act, the EPA develops criteria to govern dumping permits and dumping sites. The criteria are to take into account, among other matters, the need for dumping, availability of alternative disposal methods, and effects on water quality. In principle, ocean dumping is not entirely foreclosed by the Act, but only controlled. In practice, however, ocean dumping of municipal wastes was prohibited after 1981 by virtue of an amendment that required cessation of "sewage sludge" dumping:

> For the purposes of this amendment, the term "sewage sludge" means any solid, semisolid, or liquid waste generated by a municipal wastewater treatment plant the ocean dumping of which may unreasonably degrade or endanger human health, welfare, amenities, or the marine environment, ecological systems, or economic potentialities...
>
> Any wastes from a municipal wastewater treatment plant that can meet the EPA environmental impact criteria for ocean dumping are not precluded by the amendment. However, at this time it is not feasible to treat such wastes to the extent that they would no longer be classified as sewage sludge according to the definition given in the amendment. Therefore, this amendment will bring about the termination of all ocean dumping of sludges from municipal wastewater treatment plant discharges by the end of 1981.
>
> This amendment was stimulated by congressional concern that those municipalities currently dumping sewage sludge in the ocean were not moving rapidly enough toward implementing alternatives which would permit them to stop ocean dumping as soon as possible and that a statutory requirement would provide an additional incentive to accomplish this (USEPA, 1977).

The Ocean Dumping Act and the Clean Water Act overlap to some, but not an unmanageable, degree; the criteria developed under the two acts are essentially the same. The Ocean Dumping Act does not apply to outfalls governed by the Clean Water Act, but some provisions of the latter do apply (simultaneously with

the Ocean Dumping Act) to ocean dumping by vessels. The practice of the EPA is to treat all instances of such dumping under the Ocean Dumping Act alone. For possible changes in direction regarding administration of controls over disposal into the oceans under the Ocean Dumping Act and the Clean Water Act, see Federal Register 45:65942-65954, October 3, 1980.

The Ocean Dumping Convention (Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter) was signed by the United States in 1972 and became effective in 1975. (For a history of the negotiations leading up to the Convention, see Lettow, 1974). The Convention is not self-executing; party nations must enact implementing legislation. "The Convention tracks the original United States Act so closely that only minor technical amendments to the Act are required for the United States to implement the Convention" (Lettow, 1974).

Land

Protection of land resources from adverse effects of waste disposal practices is, in contrast to the case of water, still to a considerable extent the province of state and local governments. States and localities quite typically have controls on dumping of wastes, on land filling, and on other land applications of solid and liquid wastewater products. Nevertheless, in recent years federal legislation has been enacted that bears (more indirectly than directly) on land disposal of municipal wastes.

The Resource Conservation and Recovery Act of 1976 (RCRA). The most important piece of federal legislation -- though one, as we shall see, that to some degree is likely to relate only indirectly to municipal waste products -- is RCRA. RCRA represents the latest amendments to the federal Solid Waste Disposal Act of 1965 (amended by the Resource Recovery Act in 1970); the act is unusual in that it became law without the usual conference committee procedure (for the legislative history of the act see Kovacs and Klucsik, 1977).

The central purposes of RCRA are to provide assistance for solid waste management plans, to prohibit open dumping and to convert existing open dumps to facilities that do not pose environmental risks, to regulate hazardous wastes "from the cradle to the grave," and generally to promote improved solid waste collection, disposal, recovery, and reuse. With

the exception of the hazardous waste provisions, RCRA relies more on federal influence than it does federal mandates. The financial assistance it makes available is to serve as an incentive to states and localities voluntarily to develop appropriate solid waste management plans. Areas that fail to comply receive no assistance but are not otherwise subject to federal regulatory authority.

To say this, however, is not necessarily to say that RCRA lacks an effective bite. If a state or locality has independent designs to improve solid waste management, and if federal financial assistance would make it possible to realize these, then the funding incentive can be a strong one. Whether this will usually be the pattern, however, is an open question. Compliance with RCRA will be expensive. By some estimates, for example, 94 percent of present disposal sites do not meet the standards for sanitary landfills needed to qualify for federal assistance (Kovacs and Klucsik, 1977), involving large outlays for land acquisition, open dump conversion, and disposal facility construction. Yet, in contrast to the POTW construction grant program of the Clean Water Act, no federal funds are generally available for these purposes; federal assistance is limited, in essence, to financial support of planning and for dealing with emergencies. State and local governments, then, can be expected to look carefully at the high capital costs entailed in qualifying for limited federal benefits. In the view of some, the failure to provide funding for capital costs "will prevent the achievement of the lofty goals set by Congress, and it is merely a matter of time before the federal government will have to play a more active role in assuring that properly designed landfills on suitable sites will in fact be constructed" (Manko and Katcher, 1978).

There are other provisions in RCRA that may induce compliance notwithstanding its possibly weak funding incentives. The ban on open dumping is an example. The open dump provisions calling for closure or upgrading by 1983 are a part of a larger program to improve management of municipal solid waste by improved disposal, conservation, and reuse practices. The idea is to avoid, for instance, surface and ground-water contamination. The hope is that states and localities will comply with the open dumping provisions "voluntarily"; their failure to do so results in a loss of federal funds. This is not, however, the only sanction. Although there is some disagreement on the issue (see, e.g., Kovacs and

Klucsik, 1977), citizens may be able to enforce the prohibition on open dumps through RCRA's citizen suit provisions. Moreover, the EPA may bring an action to close any dump presenting an immminent and substantial danger.

Another example of an instance in which there is substantial and direct federal regulatory authority concerns hazardous wastes. Regarding such wastes -- those that, because of their quantity, concentration, or other characteristics, may seriously threaten health or the environment -- the EPA is to establish minimum standards for generation, transportation, treatment, storage, and disposal. The standards are mandatory and the EPA has enforcement authority (states may carry out their own programs if they meet federal requirements).

It should be noted that the hazardous waste provisions could apply to many municipal wastes, including some sludges (sludge from waste treatment plants is explicitly included in the Act's definition of solid waste, and all of RCRA's provisions are thus applicable to it). Generally speaking, however, the EPA has chosen to exclude POTW sludges from the hazardous waste provisions of RCRA, treating them instead under sludge disposal guidelines promulgated pursuant to the Clean Water Act, addressed to all aspects of sludge management, and implemented through NPDES permits. The effect of the strategy is to give the federal government direct control over land application and disposal of municipal sludge. Challenges to the strategy have thus far been unsuccessful. Hazardous industrial sludges -- those resulting from pretreatment, for example -- are regulated under RCRA; under certain circumstances POTW sludges will be also (BNA, 1980:11:684).

Other legislation. With regard to land application and disposal of municipal wastes, two other federal statutes deserve brief mention (although they are unlikely to put significant constraints on municipal waste disposal). First, the Safe Drinking Water Act of 1974 contains various provisions intended to protect groundwater. The Act has particular bearing on the common practice of injecting industrial wastewaters into the earth. Other provisions, however, will limit some municipal waste disposal practices in order to protect aquifers from contamination. Second, the Toxic Substances Control Act of 1976 contains a program of controls on the introduction, distribution, use, and disposal of high-risk chemicals. The Act bears primarily on

manufacturers. It could, however, feed into pretreatment or waste disposal requirements (under RCRA or the Clean Water Act, for example) in various ways. Indeed, there is an acknowledged overlap between the Clean Water Act and all the legislation described in this section, and Congress has directed the EPA to coordinate the measures and avoid duplication insofar as possible. The agency's decision to coordinate municipal sludge disposal by integrating RCRA and Clean Water Act requirements under the permit program of the latter Act alone is an example of its efforts in this regard.

Air

As in the case of water quality, th federal presence in the air pollution field has grown over the years from a support position to a program of mandatory (and stringent) requirements. The latest stage of this process is found in the Clean Air Act Amendments of 1977, under which the federal government establishes (or in some instances, requires the states to establish) air quality and emission standards. One effect of the standards is to constrain methods and amounts of waste incineration. Depending on the case, expensive control tehnologies will be required, and new combustion facilities locating in areas not in compliance with air quality standards may be obliged to offset (on a greater than one-for-one basis) emissions from other facilities in order to receive permission to operate. In addition, the Act provides the EPA with authority to condition POTW construction grants on compliance by the facility with emission standards, and on compliance by the state in question with provisions relating to prevention of significant deterioration and non-attainment areas. Several states have already had federal grants for POTW construction cut off for failure to comply with these provisions.

ASSESSMENT OF THE PRESENT SITUATION

The foregoing does not exhaust the list of legislation and regulations, state and federal, that pertain to municipal waste disposal, but it is sufficient for our purposes. (For discussion of the application of the National Environmental Policy Act's planning requirements to construction grants and new source NPDES permits, see Zener, 1974. On other federal, state, and local controls see NAS, 1978). This section describes some problems in the regulatory

framework prior to 1977, notes some improvements since that time, and suggests some possibilities for constructive change.

Problems

Municipal wastes, thanks to a long and rather haphazard process of legislative accretion, are presently subject to a formidable and poorly integrated list of controls that constrain waste disposal from all sides. The Clean Water Act and the Ocean Dumping Act and Convention place sharp limits (and in some cases prohibitions) on disposal of municipal effluents and sludges into water generally and the oceans in particular. Other legislation, RCRA especially, curtails land disposal and application options. The Clean Air Act Amendments require extensive technological controls prior to disposal of sludge by combustion. The virtue of these constraints is that they protect all environmental media -- water, land, and air -- from the adverse effects of waste disposal; their vices are high costs, unnecessarily stringent controls (in some cases), and the absence of a balanced systematic approach.

That compliance with the provisions, described in this chapter, will be very expensive, has not, to our knowledge, ever been denied; a thorough accounting is hardly necessary here. It is enough to note, as examples, estimates that municipal compliance with Clean Water Act requirements will cost on the order of $100 billion (Hunciker, 1977; for discussion of other cost estimates see Chapter 2 herein and Luken and Pechan, 1977). The costs of cleaning up hazardous waste sites -- only one requirement of RCRA -- have been estimated by various sources within the EPA to be from $22 to $42 billion, and the agency's deputy administrator has "noted that state governments do not have the resources or money to pay for containment and clean up of these sites" (BNA, 1979:9:2085). (The "Superfund" legislation recently enacted by Congress (Pub. L. No. 96-510) will provide some funding assistance on an emergency basis.)

High costs of compliance would be far less disturbing were one assured that the program of controls involves no waste -- does not, that is, require unnecessary or unjustifiable expenditures. Unhappily, the opposite is probably the case. Focusing on the federal water pollution control program alone, there is a large literature that has brought into serious question many of its

requirements. For example, the no-discharge goal was found by the National Water Commission to be unrealistic and no more sound than a no-development goal for controlling land use (National Water Commission, 1973), the 1983 fishable-swimmable goal to entail costs not worth the benefits to be achieved (cf. Ackerman et al, 1974), the time schedule of the program to be shortsighted and counterproductive (Tripp, 1977), and the geographically uniform effluent controls to be inefficient and, regarding post-1977 requirements, also unjustified in terms of water quality improvements (Luken and Pechan, 1977). The secondary treatment and BPWTT requirements for POTWs in particular were criticized as overlooking the assimilative capacities of ocean waters (Officer and Ryther, 1977; Luken and Pechan, 1977), an unsurprising conclusion given that POTW control requirements, generally speaking, have been designed with freshwater conditions in mind. One study taking an early look at the federal water quality program as a whole found that it promised to be generally inefficient (Ackerman et al., 1975), a particularly discouraging note given the undesirable distribution of water quality improvement costs (see in particular Collins, 1977 with respect to the POTW construction grant program).

Of course, not all observers would agree with all the criticisms suggested above, though it bears mention that many of them were endorsed, at least implicitly and to some degree, by the mid-course corrections recommended in the final report of the National Commission on Water Quality (1976). Moreover, the remarks above were directed at the 1972 amendments of FWPCA (as will be some of the remarks below), and some (but not all) of the shortcomings noted have been corrected by the Clean Water Act of 1977. The point here, however, is to provide the backdrop for considering those corrections and suggesting further ones in the next two sections. Before doing that, though, we must sketch one other very general and very important problem area that only magnifies those discussed above.

The problem is poor integration -- a multitude of controls each of which tends to ignore the other. We suggested at the outset of this chapter the need for a systematic approach to controlling municipal waste disposal, one that takes account of interrelationships among environmental media, considers the implications of one regulatory program in terms of other regulatory programs, and provides for sensible trade-offs. The challenge of a

systematic approach is a large one, but the federal program prior to 1977, and to a lesser extent since 1977, has failed to an unnecessary degree.

For example, one unquestioned result of the treatment requirements of federal water pollution controls is dramatic increases in the amounts of industrial and (in particular) municipal sludges. Large increases in municipal solid wastes generally are expected over the next ten years, especially in the major metropolitan areas of the United States (BNA, 1979:9:2255). POTW sludges can be expected to suffer the greatest increases, for the secondary treatment required by the federal program can double or triple sludge production as compared to primary treatment (BNA, 1978:9:1175; Tripp, 1977). The quick technological fix of the FWPCA amendments, in short, resolved the water quality problem only to create a sludge problem (Tripp, 1977); legislation since the 1972 amendments has compounded the latter by limiting or prohibiting sludge disposal into the oceans or air, or onto the land, all without apparent attention to obvious interrelationships among waste products and media. According to a General Accounting Office (GAO) report (BNA, 1978:9:1175):

> Sludge volumes are expected to increase from the current five million tons annually to above 10 million tons annually by 1987.
>
> Some generally accepted sludge disposal methods are being phased out or becoming increasingly regulated by the government.
>
> Landfill sites are dwindling, use of incineration is being restricted because of air pollution standards and fuel shortages, and ocean dumping has endangered human and marine life and will be phased out by December 1981.

The EPA "fully concurred" with the GAO report.

Another, more general interrelationship apparently overlooked by Congress has to do with the construction grants program. Not only has it had the effect, together with other provisions of the water pollution control program, of encouraging reliance on conventional and very expensive technologies (e.g., secondary treatment) that tend to foreclose later adoption of alternative, less capital intensive controls (e.g., advanced primary treatment), it has also stimulated urban development that has the dual

effect of limiting land disposal sites and increasing treatment demands and water pollution problems (Tripp, 1977; Sewers, Clean Water, and Planned Growth, 1977).

There are other obvious examples of the unsystematic approach of the federal program. Present law prohibits ocean discharge of sludge, for instance, and the Ocean Dumping Act prohibits sludge dumping after 1981. The Act's prohibitions will have particular impact on major urban areas of the East Coast, areas that rely on dumping precisely because they are short of alternative disposal sites and techniques. (Moreover, there may be a certain peculiarity in blanket prohibitions of sludge discharge and dumping when viewed in light of the Clean Water Act's waiver provisions regarding ocean disposal of wastewaters.)

The absence of effective integration in the federal program is hardly, of course, a secret. It was commented upon (with particular regard to municipal sludge) by the NAS (1978), which called for a "holistic, multimedium" approach of integrated management that gives full attention to trade-offs (see also Ackerman et al., 1974). While these observations were made without consideration of the Clean Water Act of 1977, the situation since passage of that legislation has not changed substantially (though there have been some improvements, as noted below). In interviews with EPA personnel, for example, opinions like the following were expressed: the present federal water quality program implicitly assumes that land and air disposal of wastes is generally preferable to water disposal, yet there is no systematic basis for the assumption; detailed trade-offs comparing the environmental impacts of land versus air versus ocean effluent have rarely been considered; there is a clear need for "organic" legislation as opposed to the media and mission orientations of present law -- orientations that do not permit the EPA to take sufficient account of trade-offs. In addition, a more careful phasing of the deadlines of existing programs is required.

Present controls, in short, are costly, and the high costs are aggravated by (in some cases) unnecessary uniformity and strictness, and by a general absence of integrated attention to the environmental system as a whole. Again, however, this is not to say that there have been no improvements since 1977; all in all the trend underway is promising, though there is room for further change. We discuss these points in the next two sections.

Improvements

It is probably too early to state with utter confidence that the situation regarding controls on disposal of municipal wastes is sure to improve, for many important developments are just underway, or only being talked about. Still, there have been encouraging signs since 1977.

As described earlier, the Clean Water Act of 1977 has amended the federal water pollution control program in a number of constructive ways. Easing of the technological standard for some controls (e.g., conventional pollutants), extensions of original deadlines for most control requirements, and broadened opportunities for waivers and variances will lighten the financial burden of water pollution control measures. At the same time, in many cases this loosening up will have no adverse effect on water quality -- because, for example, a variance may be granted only upon a showing that water quality standards will be met. In other instances, loosening up -- of originally scheduled deadlines, for example -- will afford additional time in which to assess the merit of advanced technological controls (such as BAT) in the face of contentions that they impose high costs but achieve, at best, only minimal benefits (Luken and Pechan, 1977).

Developments along these lines with respect to POTWs in particular are even more encouraging. They suggest that the federal mind is moving away from reliance on quick, but often simplistic and counterproductive, technological fixes and toward alternative treatment techniques. The 1977 Act requires (and encourages with increased funding) careful attention to innovative controls, and the EPA is an ethusiastic supporter of this new emphasis (Hunciker, 1977). Just as it is promoting alternative treatment, so too, very recent activities suggest, is the agency giving greater scrutiny to construction of capital intensive controls. EPA's policy, it appears, will be "to require a complete justification in terms of water quality or public health benefits for all projects proposed with treatment more stringent than secondary." In addition, "water quality standards more stringent than" federal criteria "must be fully justified" (BNA, 1979:9:2073. For GAO criticism of the EPA's earlier approach, see BNA, 1978:9:441. But on the possibility of EPA requiring more demanding water quality standards, see BNA, 1978:9:435).

Developments in POTW controls since 1977 also reveal important new attention to the assimilative capacity of ocean waters. As described in a previous section, the Clean Water Act of 1977 in its section 301(h) provides for waivers of secondary treatment requirements for ocean discharges in cases where characteristics of receiving waters permit lesser treatment without adverse effects on quality. The provision not only will reduce direct control costs with no loss of benefits, but will at the same time result in dramatic reductions in the amount of sludge that would otherwise be produced.

The section 301(h) modification provisions are especially sensible given that the uniform treatment requirements made applicable to all point sources by virtue of the 1972 FWPCA amendments were ill-suited to the particular case of POTWs. The congressional purpose behind uniform (and strict) treatment requirements was probably threefold: to force technology; to maintain and improve water quality; and to avoid the problem of states competing for industry by reducing effluent standards. Yet technology can be forced virtually as effectively by requiring stringent controls of some as by requiring them of all, so long as the research and development stimulated are sufficient to generate tehnological advances which can then, in a new iteration, be required as appropriate. Water quality can be protected by provisions permitting less than the tightest controls only upon a showing of no (or minimal) adverse effects -- the approach of section 301(h) and some other variance provisions of the 1977 Act. And the problem of state competition for industry, however persuasive it might be in the case of industrial sources, simply has no application to POTW location. Congress apparently recognized these points, with regard to the analogous problem of thermal pollution from power plants, in the exceptions to uniform treatment provided by section 316 of the 1972 amendments to FWPCA (see Zener, 1974); it has now done so with regard to ocean discharges from POTWs. Whatever the merits of geographical uniformity in treatment requirements for industrial sources (for criticism see, e.g., Luken and Pechan, 1977), application of the uniform approach to POTWs discharging into ocean waters was clearly misguided. Section 301(h), then, is a wise extension of the reasoning behind the section 316 provisions of the 1972 amendments. (Whether the extension should reach further is considered in the next section.)

The foregoing discussion should not be taken to suggest that recent developments reflect only a backing-away and loosening-up. On the contrary, the last few years have seen a new emphasis on treatment (including pretreatment) of toxic pollutants. Past neglect in this area brought justified criticism (see, e.g., Ackerman et al., 1974), and heightened efforts are a welcome development. From our perspective, effective pretreatment programs are especially important for two reasons: they expand POTW sludge disposal opportunities; and they reduce pressure to push POTW effluent control requirements to secondary treatment and beyond.

In light of our discussion of geographical uniformity above, it should be noted here that in the case of toxic pollutants a centralized program of uniform controls backed up by strong federal enforcement powers, the thrust of present policy, makes good sense. The reasons for this conclusion have been suggested by Ackerman et al. (1974):

> The major effort should be concentrated on the national level. First, a national staff could be large enough both to engage in the broadly based research effort required and to coordinate the research currently going on... Second, many discharges of esoteric materials may affect the inhabitants of distant regions, especially when the substances are ingested by fish subsequently harvested for market. Local governments cannot be expected to respond adequately to the important interests of these distant consumers. Third, if one believes it desirable...[to] err on the side of caution in the regulation of potentially hazardous discharges, it would seem wise to avoid a situation in which regions or states are tempted to relax their poison regulations to attract firms to the area.

(See also the discussion of risk and uncertainty in Chapter 10 herein; and Voytko, 1977, discussing local difficulties in enforcing pretreatment requirements.)

A final encouraging development to be mentioned here concerns various steps, undertaken by divergent authorities, toward more integrated management of municipal wastes; the steps, though partial, are promising. We have already mentioned that both Congress and the EPA are acutely aware of needs in this area, and are at work to do something about them. Thus Congress has called for coordination of the interrelated requirements of legislation like

the Clean Water Act, RCRA, and the Toxic Substances Control Act. The EPA has moved to coordinate various sludge disposal controls under the umbrella of sections 402 and 405 of the Clean Water Act. This could be a mixed blessing, resulting in relaxed requirements for land application of industrial sludges (See BNA, 1978:9:1212-14). The agency is also taking steps to consolidate permit procedures under four of the programs considered in this chapter (the Clean Water Act, RCRA, the Safe Drinking Water Act, and the Clean Air Act; see BNA, 1979:10:225-26). The Supreme Court has recently rendered a decision that enhances opportunities for land disposal of sludge on a regional basis, foreclosing narrow local decisions that limit disposal sites (City of Philadelphia vs New Jersey, 11 E.R.C. 1771, US Supreme Court 1978; see also Hardage vs Atkins, 12 E.R.C. 1043, 10th Circuit, 1978). A United States District Court has permitted the use of Clean Water Act section 208 (area-wide waste treatment management planning) funds for air quality and solid waste planning, noting that "the problems of air, solid waste, and water pollution are enormous, and only a coordinated attack on those and other environmental problems has any possibility of success" (BNA, 1978:9:1242; see also BNA, 1978: 9:913-14, on EPA efforts to coordinate media planning under section 208).

Possibilites for Constructive Change

To note the improvements above is not to say there is no room for further change; this section mentions several possibilities with particular regard to ocean disposal of municipal wastes. (Some of the observations here will be examined in more detail in Chapter 12.)

Despite promising developments in the last few years, there remains a clear need for better integration of controls on waste disposal. The "organic legislation" mentioned previously -- a systematic, multimedia approach to waste management -- has yet to be realized, and the result, as would be expected, is poor coordination. This state of affairs may reflect little more than the fact that the job of integration across all media, sources, and wastes is an exceedingly complex one, and one for which experience has left legislative and administrative bodies ill prepared. Under these circumstances, a more modest approach to integration, one consciously designed as a pilot program useful for generating information and ideas, is advisable.

Municipal waste management in particular recommends itself for such a program. The municipal waste problem involves all media and all levels of government. Its management poses a number of opportunities for effective trade-offs. Finally, as described earlier, a number of steps are already underway to bring more sense and harmony into control of municipal waste disposal. Further steps should be considered while there is still momentum behind the move for integration -- steps that might provide more insights into the job of achieving integrated waste management on a broader basis later.

Baldly stated, an effective municipal waste disposal management strategy would concentrate on essentially two goals: 1. removal, usually by pretreatment, and confined disposal of toxic pollutants; and 2. recycling, reuse, and varying disposal of POTW effluent and sludge (varying disposal based on the nature of the waste and especially on the relative availability of water, land, and air in particular areas). In terms of present controls, such a strategy suggests the importance of careful attention to pretreatment requirements, and the need for an open mind regarding geographic variability in nontoxic waste disposal requirements. (See, e.g., the testimony of N. Brooks in US Congress Hearings 1978, commenting among other things on the West Coast's relatively abundant ocean water waste disposal capacity and limited atmospheric waste disposal capacity, in contrast to the opposite situation on the East Coast.)

Pretreatment, of course, has recently received renewed emphasis and effort, but further progress is essential to effective municipal waste management (pretreatment, by removing undesirable and incompatible pollutants from industrial wastes before they reach a POTW, expands opportunities for safely and easily disposing of the effluent and sludge that leave the treatment works). Yet some observers foresee delay, rather than smooth progress, in pretreatment efforts, primarily because the present program "saddles the POTWs with the burden of administering an extraordinarily complicated set of differing requirements for each type of industrial user, an almost hopeless task for municipalities in view of the limited manpower available to them for this purpose" (Quarles, 1978). In addition, very stringent pretreatment requirements may interfere with cooperative relationships between POTWs and their users, leading to further difficulties and delays (cf. Voytko, 1977).

The provision in the Clean Water Act of 1977 allowing a credit to POTW users equal to the amount of pollutants removed by the treatment works may help alleviate some of these difficulties, but other measures should also be considered. One in particular would be to abandon those features of present pretreatment controls which in essence impose specific, uniform requirements on the various classes of industrial users -- requirements that probably are, to some sources, unnecessarily stringent and wasteful. An alternative approach, and one not dramatically different from the philosophy behind the pretreatment credits of present law, would establish effluent limits for a POTW itself, then permit the treatment works to allocate the total load among its users in the most efficient manner in light of its and the users' treatment systems. This would build more flexibility into the program than presently exists, and with no cost in terms of water quality. As with present law, provision could be made that any allocation arrived at by a POTW did not unduly limit options for reusing or disposing of its sludge.

The approach to pretreatment sketched above would among other things acknowledge the relationship between treatment requirements and water quality. In 1972 Congress made a fundamental change in the direction of water pollution control policy -- away from emphasis on the quality of receiving waters, and toward emphasis on stringent effluent controls (whether or not needed for improved water quality). Since 1972 the pattern has been shifting, however slowly, back to water quality concerns. The most important instance of this, for our purposes, is the provision in section 301(h) of the Clean Water Act permitting modifications of POTW secondary treatment requirements. The final regulations for implementing that provision reflect a fairly hospitable attitude toward modifications (Federal Register 44:34784-34832, June 15, 1979). Prior to enactment of section 301(h), but after the section had been proposed, the EPA indicated that modifications would be limited to "very special circumstances" -- those existing in about fifty California municipalities, a few other West Coast cities, San Juan, American Samoa, Honolulu and the Virgin Islands (BNA, 1977:8:1317-18). After passage of section 301(h) the agency received over 200 applications, many of them from East Coast cities (BNA, 1978:9:1174, 1203-04). Initial regulations proposed by the EPA for section 301(h) were criticized as too stringent, likely to yield few if any modifications (BNA, 1978:9:155-56). The agency, it appears, has come to agree to some extent; under the

new regulations "a significant number of the over 200 waiver applications submitted to EPA will be approved," according to agency officials (BNA, 1979: 10:120).

By and large, the new regulations represent a straightforward reading of section 301(h) and its legislative history. Sources must apply at least primary treatment (or secondary if they are already doing so or are required by state law to do so), may not discharge sludge, and must comply with any other applicable federal law that conflicts with granting of a modification. In addition, sources applying for modifications must "demonstrate" (the burden of proof is on the source) that each statutory requirement specified by Congress in section 301(h) is met (see above).

One important easing of originally proposed requirements reflected in the final regulations concerns the definition of "existing discharges" (to which section 301(h) is limited). Under the final regulations, sources may base their applications on future, improved discharges, providing the source was a discharger as of late December 1977. Unfortunately, the revised definition still precludes application of section 301(h) to new sources, a contrary limitation from the standpoint of minimizing costs by taking full account of the ocean's assimilative capacity.

One should not, of course, fault the EPA for the limitation of section 301(h) to existing sources; that is Congress' doing. Much the same can be said of other shortcomings in the modification provision. It is Congress, for example, which has insisted that modifications not result in additional treatment requirements for other sources -- an inefficient provision in cases where other sources could control less expensively than the POTW. It is also Congress which has insisted that water quality be an ultimate limiting factor in all cases, but without permitting examination of the costs and benefits involved in given water quality standards, especially the fishable-swimmable standards.

Applicants for section 301(h) modifications may face considerable difficulty in making the required demonstrations; that is largely a technical matter beyond the scope of this chapter (see comments of Ryther in BNA, 1978:9:155). It is clear that a high standard of proof will be applied (BNA, 1979: 10:228-29). On the other hand, comments, mentioned

above, that many applicants (presumably including East Coast applicants) will receive waivers suggests that reasonable standards will be applied. One hopes this will be the case, for an estimated $4.6 billion in secondary treatment construction costs is at stake (BNA, 1979:10:228-29).

Section 301(h) represents an important, but limited, step in the right direction. More could be done, especially in the case of conventional municipal wastes. In particular, a program of geographically varying "management standards" for POTWs could achieve savings in control costs, promote integrated protection of all environmental media, and provide the groundwork for a similar approach to waste disposal generally. We shall simply give a brief sketch of the management standards approach here, saving elaboration for Chapter 12.

In essence, management standards for POTWs would consider the assimilative capacity of all media (water, land, and air) in each area of concern, bring the controls for the various media into phase with one another, and apply timetables for improvement that were sensitive to local situations (all matters which section 301(h) neglects). Thus, in the case of POTWs discharging into ocean waters, the management standards approach would first of all acknowledge that the United States is bounded by a number of differentiated ocean regions, each with is own peculiar environmental characteristics (see Chapter 3 herein). It would also acknowledge that areas vary in terms of the availability and condition of land and air resources, and that some areas are more generally constrained than others. These factors in mind, it would not prohibit, say, ocean dumping without considering the impact of such a prohibition on other disposal practices, nor demand treatment levels that result in more sludge with few places to dispose of it, nor overlook the broad variations in assimilative capacity from area to area -- all as present law tends to do. Moreover, the management standards approach would not apply the same deadlines for improvement to each area -- again as present law tends to do. Rather, the approach would take the total condition of each area reasonably into account in determining that area's requirements.

A program along these lines would hardly be entirely novel. It is essentially the approach used in some countries with conditions and problems not unlike those in the United States (e.g., Canada -- see Buxton, 1977); it would represent the sort of

coordinated, regional program that has been called for in careful studies (see, e.g., Ackerman, 1974). Management standards would depend on close monitoring, and research and development efforts along these lines would be necessary. Yet the approach itself would tend to spur efforts in this regard, just as, for example, rapid increases in waste disposal and legal controls on waste disposal are spurring development of industrial waste exchanges (on the latter, see USEPA, 1978; Ricci, 1976; BNA, 1979:10:128-29). Management standards would forego some of the administrative advantages of uniform treatment requirements, but would avoid the high costs of overly demanding controls that are the price of uniformity (Luken and Pechan, 1977). Given the relatively small number of POTWs discharging into ocean waters (a fact that would ease the administrative burden), coupled with the expense of advanced treatment requirements and their compounding of the sludge disposal problem, the management standards approach would almost surely yield net savings. Applied to non-toxic pollutants, it would take advantage of assimilative capacity, ease the sludge problem, reduce total control costs, and provide, at little risk, better insight into integrated, regional programs of waste disposal control.

Postscript. Research on this chapter was completed in early 1981. Subsequent developments have not been taken into account.

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12

Evaluation of Key Issues and Alternative Strategies

Norman H. Brooks

INTRODUCTION

In preceding chapters, we have seen how the ocean is being used for disposal of municipal wastewater and sludge and what is known about the effects of these discharges. Use of the coastal ocean for waste assimilation may or may not conflict with other beneficial uses of the ocean, depending on the methods of treatment and disposal, and on the local characteristics of the ocean waters receiving the wastewater and sludge discharges. From a scientific and engineering point of view, we seek to enlarge our understanding of the processes that affect the ultimate fates of these wastes in order to make appropriate decisions among various alternatives for wastewater disposal, while keeping any impacts on other ocean uses or resources to acceptably low levels.

The reader is reminded that the discussion in this book deals primarily with discharges into open coastal waters, as opposed to enclosed bays and estuaries. The latter present special problems and can be badly damaged by improper wastewater discharges. When at all feasible, coastal cities generally are well-advised to discharge into the open ocean rather than into estuaries or poorly flushed bays and lagoons.

Federal policy for ocean discharge of sewage and sludge has changed significantly during the 1970s, as described in Chapter 11; further changes, in response to better understanding of environmental processes and effects (as described in many of the previous chapters), are anticipated. The principal issues for future federal policy are: what levels of sewage treatment to require for sewage effluents (primary, secondary, or more) discharged through ocean outfalls; whether to allow ocean disposal of digested sludge either by barge or outfall; and how to control toxic substances in these discharges. Subsidiary issues are

procedures for decision-making, setting of regulations and conditions, and monitoring and reviewing the effects of the allowed discharges.

This chapter summarizes and evaluates the problems of ocean disposal of sewage and sludge, and alternative strategies for controlling them. Chapter divisions cover current practices and risks; discuss the key technical issues affecting management alternatives; present those alternatives; discuss basic policy and administrative questions, and present conclusions.

This chapter will explain why present policies for ocean discharge of wastewater and sludges need to be revised - they are not properly based on current scientific and engineering knowledge of ocean disposal. In general, excessive resources are being spent on treating conventional pollutants (like BOD), and insufficient effort is being directed at controlling the discharge of toxic pollutants (e.g., certain heavy metals and persistent organics). Furthermore, there is practically no effective integration of ocean discharge management with air and land pollution control, with the ultimate objective of achieving the best integrated systems of environmental control for reducing overall environmental risk in all media.

ASSESSMENT OF CURRENT OCEAN DISPOSAL PRACTICES

Effluent and Sludge Discharges

Coastal cities all over the world discharge sewage effluents and sludges into the ocean for convenience and, in many instances, as a matter of necessity. The resulting ocean water quality varies from extremely polluted to excellent, depending on: 1. the constituents of the raw sewage (especially toxic substances); 2. the degree of treatment provided; 3. the nature of the outfall or barging systems; and 4. the characteristics of the receiving water. Using the ocean for receiving wastewater, while protecting water quality at the same time, requires proper system design in light of the elements above.

The engineering technology of treatment plant and ocean outfall design is sufficiently advanced that systems to meet specific water quality standards can be designed and the costs estimated. There is, however, uncertainty about what the requirements

should be (especially toxic substances), and as more knowledge is acquired they should be periodically reviewed and revised. Decisions on upgrading standards should balance the marginal environment benefits (including reduction of risks) against the marginal costs of constructing and operating additional or improved facilities for treatment and disposal. In spite of uncertainties in assessing the effects of improved systems, it is still essential to identify, as best one can, what benefits will result from public expenditures for treatment plants and outfalls; otherwise, over- (or under-) control is almost sure to follow. Unfortunately, though, such a balancing approach is not usually employed in the current system of legislation and regulations governing ocean water quality in the United States.

Although we are faced with different disposal problems and requirements for sewage effluents and sludge, it is essential that we deal with the treatment and disposal of municipal wastewater as a complete system. For instance, higher degrees of wastewater treatment mean more sludge to dispose of; and the transfer of trace contaminants from effluents to sludge does not necessarily reduce overall environmental risk.

Before discussing the regulatory structure further, it will be useful to describe the various types of pollutants in municipal sewer systems.

Types of Pollutants

In Chapter 10, we saw that it is useful to separate pollutants into four general classes:

I: Ecosystem materials which naturally cycle through ecosystems in large quantities -- suspended solids (natural organic material), ammonia, other nutrients, and other natural oxygen-demanding substances.

II: Pathogens, bacteria and viruses causing disease in humans.

III: Trace metals, such as cadmium and lead, in concentrations far greater than natural systems.

IV: Toxic or exotic materials such as persistent manufactured organic chemicals and radioactive wastes.

During the last 50 years, most of the efforts for sewage treatment and disposal have been directed almost entirely at Class I and II pollutants -- the classical ones to the environmental engineer. Given a proper combination of treatment plants and outfalls for a given ocean site, these pollutants represent little problem. There is some debate over the water quality objectives for these substances, but overall there is little evidence that long-range coastal pollution problems from Class I and II cannot be easily remedied. Adverse effects of inadequate outfalls have been reversible when the outfalls are shut off, with complete ecosystem recovery taking only a few years. In the future, if changes in requirements for Class I pollutants are needed at any particular disposal site, disposal practices can be modified with improvements appearing quickly in the ocean without long-term, lingering effects.

For pollutants in Classes III and IV (other than radioactive substances) there has been relatively little attention until the last decade, and the problems may not be considered well in hand. Trace organic contaminants discharged by sewer systems may be bioaccumulated to levels which are damaging to marine ecosystems, especially the higher trophic levels. The effects of persistent organics can last for decades. Because release of these hazardous substances into the environment is a high priority matter (not only for the ocean, but also for the air, land, and fresh waters), effective control over their discharge at the initial source is a better strategy than trying to remove substances once they arrive at sewage treatment plants. Such removal is not really removal at all, but rather a displacement, via the sludge, from one environmental medium to another; the problem of safe ultimate disposal may still be very difficult to solve. However, the volumes are much reduced, and the different kinds of toxic wastes can be managed appropriately on a case-by-case basis.

Although the distinction between the four classes of pollutants discussed above is not always sharp, it nonetheless provides focus for management efforts. Wide dispersion into the marine environment is a viable strategy for pollutants of Classes I and II, but definitely not for those of Classes III and IV. For the latter, we must contain or prevent releases except in the small amounts which can be safely dispersed. Conversely, in our efforts to control Class III or IV pollutants, it would be a mistake to try to control the vast amounts of

relatively innocuous Class I pollutants by preventing their release back into the natural environment for natural recycling.

Present Disposal Regulations and Practices

As explained in Chapter 11, current US policy differentiates between sewage effluent and sludge. Effluent must have received secondary treatment unless a waiver for ocean discharge of less-than-secondary effluent is granted under the provisions of Section 301(h) of the Federal Water Pollution Control Act Amendments of 1977. In general terms, the basis for granting the waiver is compliance by the discharger with conditions regarding water quality, monitoring, pretreatment, toxic pollutants, and waste volume. The waiver provision is a significant improvement over the 1972 legislation (P.L. 92-500), which required secondary treatment regardless of whether or not it produced significant water quality benefits in the receiving waters.

The discharge of sewage sludge into the ocean is covered by two different federal laws. First, the Federal Water Pollution Control Act (as amended in 1977) prohibits discharge by outfall because sludge itself obviously neither meets the secondary treatment requirement nor is eligible for any exceptions under Section 301(h). Secondly, the Ocean Dumping Act effectively prohibits dumping after 1981 by any means (i.e., barges, since pipelines are already out).* A presumption was made by Congress and the Environmental Protection Agency that disposal of digested sludge in the ocean is unacceptable, regardless of how or where it is done, and other methods of disposal must, in each case, be preferable. This policy does not have an adequate scientific basis, or does it consider the impacts and costs of other alternatives for disposal, which may produce additional air or land pollution. Furthermore, the mandate for secondary treatment of effluent in the 1972 act did not consider the negative impacts of disposal of greatly increased volumes of sludge from the higher degree of treatment.

The legal and institutional aspects of ocean discharges of municipal wastewater and sludge have been analyzed and evaluated by Krier in Chapter 11. In the federal policy, he points out unnecessary uniformity nationally which has been costly and inefficient; poorly integrated regulatory efforts for air, land and water; a presumption that land and air can receive residuals with less impact than the ocean;

* See note at end of the chapter.

and a lack of adequate attention to the disposal of greatly increased volumes of sewage sludge from universal secondary treatment. Although recent changes in the law (e.g., secondary treatment waivers) represent important improvements, there are additional reforms needed as described at the end of Chapter 11, and summarized at the end of this chapter.

Federal law governing ocean discharge of municipal wastes gives EPA the authority to delegate a good deal of regulatory responsibility, including issuance of permits, to the states -- subject to EPA oversight and veto. General discharge requirements may be established by the states in addition to those established under federal law, and discharges must conform to both state and federal regulations. For example, although secondary treatment is not required by the State of California, federal law does require it unless a 301(h) waiver is granted. On the other hand, California has other regulations which are more stringent than the federal rules (e.g., generally true for trace metals), and sources must comply with these. In addition, some states (including California) have regional boards which may adopt additional requirements more stringent than the state and federal regulations.

The involvement of multiple layers of government often leads to confusion and controversy. A municipal discharge agency represents yet a fourth level of government. Each of these four levels might well have different attitudes about the benefits of water-pollution control (beyond basic primary treatment and a good outfall) because they pay different fractions of the cost and operate under different legal structures, sources of funds, and priorities. Consequently, good cost-effective decisions on public works projects practically never occur. For example, when there are alternatives, local sewerage agencies prefer capital expenditures (their share is only 12.5 to 25 percent of eligible capital costs) versus operation and maintenance costs (for which they pay 100 percent); or capital expenditures now are chosen even when it might be wiser to proceed more gradually because cities fear an end of the federal grants program for sewerage facilities!

The State of California adopted its comprehensive set of requirements for ocean discharges, called the Ocean Plan, in 1972, and revised the Plan in 1978. The 1978 version is reprinted as Appendix E. It is a well-conceived set

of regulations which is based primarily on receiving water quality objectives. For instance, the effluent concentration limit for any trace metal is calculated from the outfall dilution factor, the ambient water quality objective, and the natural background concentration. Further refinements are being considered for adoption in 1983.

In October 1980, the EPA issued final Ocean Discharge Criteria (EPA, 1980, 45 FR 65942) as required by Section 403(c) of P.L. 92-500. This criteria document is reproduced in Appendix F. In the future, these criteria must be met as conditions necessary for issuance of a National Pollution Discharge Elimination System (NPDES) permit by the EPA or the state. All other rules and regulations still apply. In contrast to the California Ocean Plan, the requirements of these Criteria are broad and subjective, rather than based on specific numerical values of water quality standards.* However, many more individual toxic pollutants are identified for measurement and control (see EPA list in Chapter 9). Wastewater effluents must be analyzed for all of these substances, and all data that equal or exceed the detection limits (5 times noise level) must be reported in the permit application (Sec. 125.126). Dischargers are also required to indicate how they might reduce the amounts of these identified toxic pollutants. Judgments about what pollutant levels are satisfactory are entirely up to the EPA. The issuance of a permit depends, among other things, on whether the EPA is satisfied that the pollutants will not interfere unreasonably with fish and wildlife, or other uses of the ocean. However, there is no general consensus on what concentration levels of synthetic organics might be damaging in the marine environment, and what concentrations might be expected in sewage. For the pollutants on the list the required chemical analyses are estimated to cost several thousand dollars for each sample.

Effects and Risks

There are three types of effects of wastewater discharges on coastal ocean waters: public health, ecological, and aesthetic.

1. Public health effects. Health effects may occur either by transmission of infectious disease (see Chapter 8) or through accumulation of trace contaminants in the marine food chain up to fish

* See note at end of the chapter.

harvested for human consumption (Chapter 9). Infectious diseases may be transmitted to swimmers through entry of small amounts of seawater into the nose, throat and/or stomach of bathers, or through skin cuts and sores; another principal route is via shellfish which accumulate bacteria and viruses in their digestive tracts. Not much is known about airborne pathogens transferred from polluted seawater to air, such as when waves break or wind driven spray is formed.

The transmission of infectious disease is well under control in the United States, with practically no evidence of any diseases carried by seawater. This favorable result is due to generally good submarine outfalls which extend considerable distances from shore with high dilutions, and to at least primary treatment and disinfection where and when it is necessary to maintain acceptable beach counts. In other countries, such as Great Britain, where outfalls are generally shorter and treatment is less, there is still little documentation of water-borne disease (see Appendix D). In some countries with grossly inadequate facilities (shoreline discharge of raw sewage, for example), epidemics of typhoid fever and cholera have occurred and will continue until improvements are made. Thus, the risk of infectious disease is not to be neglected, but the present US practice has essentially eliminated public health risks of marine discharges. (In fact, it is desirable to cut back on chlorination of effluents wherever possible to protect marine organisms from damaging chlorination by-products.)

Severe cases of poisoning by trace contaminants accumulated in the food chain have been documented (e.g., Minamata disease in Japan). However, such acute problems are now relatively easy to avoid with better monitoring and detection. On the other hand, chronic problems are still difficult to detect and many of them may not yet be fully identified. Thus, considerable caution, conservatism, and further research are warranted when dealing with discharges of trace contaminants into the marine environment.

2. Ecological effects. Much of the drive started in the 1970s to clean up water pollution in the United States was motivated by the desire to reduce adverse ecological effects and to improve the aesthetics of the water environment. Public health effects of marine discharges were already fairly well under control before 1970.

The impact of wastewater upon individual marine organisms and on the health of communities has been described in Chapter 7. Poorly diluted sewage, with less than secondary treatment, can have devastating local effects on marine communities by reducing dissolved oxygen levels to near zero and increasing nutrients and turbidity to damaging levels. However, high dilution and wide dispersal by a good outfall diffuser in a well flushed ocean can greatly diminish or sometimes practically eliminate adverse ecological effects. Basically, if the added doses of natural organic materials and nutrients are made small enough in a given volume of water, they can be assimilated into the ocean's ecosystem as an easily tolerated perturbation. In fact, the added nutrients, when well dispersed, can have a mild biostimulatory effect in waters which are nutrient poor, which is desirable for support of fisheries.

If an ocean discharge is well diluted into a body of water with good circulation, there is little difference between the ecological impacts of primary versus secondary effluent, except in some cases for benthic impacts. Although secondary effluent has slightly fewer nutrients than primary, the chemical form for the nutrients (e.g., nitrate) allows them to be more rapidly taken up than the undecomposed forms in primary effluents.

Ecological effects have been proved, in many instances, to be subtle and very difficult to assess or quantify. Continued research is needed to assure that important effects are not overlooked. At the same time, caution should be used to make sure that expensive wastewater disposal systems are not required to correct negligible or nonexistent ecological problems. For marine ecosystems, which are generally quite resilient, sewage effects are often readily reversible (toxic substances excluded) so that large margins of safety are not necessary and options for future changes in disposal practices can be exercised if undesirable ecological trends are observed.

3. Aesthetic effects. Even if public health and ecological effects are well under control, nobody likes to see discolored water and floating debris. Primary treatment (including good removal of floatables) and well engineered ocean outfalls usually control aesthetic problems very effectively. Therefore, secondary treatment cannot be justified for aesthetic improvement when discharge is to the open coast.

Other sources of shoreline pollution often contribute significantly to aesthetic nuisances, such as debris in storm water flows, refuse thrown overboard from commercial ships or pleasure craft, and beach litter. Refuse problems, which are highly visible at the shoreline, are often blamed erroneously on sewage or sludge disposal procedures.

In summary, the health, ecological, and aesthetic effects of ocean discharges can be kept to reasonable levels by appropriate regulations and engineering measures. The problem is really to devise the best management system. In the next section we shall examine the major technical points which bear on management choices.

KEY TECHNICAL ISSUES AFFECTING MANAGEMENT ALTERNATIVES

In the previous section we discussed the current practice of coastal wastewater disposal, and noted difficulties involved in achieving good regulations and appropriate engineering works. Some of those difficulties have arisen from a lack of understanding of the underlying science and engineering of wastewater disposal in the ocean. In this section we will recapitulate the key technical knowledge and uncertainties which bear strongly on decisions for wastewater management of ocean discharges.

Basis for Regional Differentiation of Policy

Waters that receive sewage effluents vary greatly in their hydrographic, chemical, and biological characteristics -- ranging from small rivers in the Midwest to the Pacific Ocean around the Hawaiian Islands. Nonetheless, the federal Water Pollution Control Act Amendments of 1972 set the same high minimum treatment standards for all dischargers, large and small, regardless of where and how their effluent is discharged. This was clearly inefficient as a national policy.

One obvious large sub-class is coastal waters, the subject of this book. In general they can assimilate wastewater far better than inland fresh waters because of their much larger volumes, and because salt water is a hostile environment for most freshwater organisms, including pathogens. Congress has now recognized the ocean sub-class of receiving waters in Section 301(h) of the 1977 Amendments, which allows for case-by-case waivers of the secondary

treatment requirement for five years for existing ocean dischargers.* Further differentiation in ocean discharge policy is also feasible to reduce the burden of case-by-case analysis. The scientific basis for differentiation was presented in Chapter 3.

The coastal waters of the United States can be broadly classified in five categories, arranged in order of increasingly favorable environmental factors, and decreasing need for secondary treatment:

	Type	Geographic Region
1.	Coastal plain estuaries (shallow)	Delaware and Hudson River Estuaries; Chesapeake Bay; San Francisco Bay
2.	Broad continental shelf	Atlantic Coast and Gulf Coast
3.	Fjord-like estuaries (deep)	Puget Sound
4.	Narrow continental shelf	Pacific Coast
5.	Islands (surrounded by deep water)	Hawaii and other Pacific Islands

The most significant distinguishing features separating the classes are hydrographic, but other important variables are coupled with hydrography. Generally, the Pacific Coast, with its narrow continental shelf, is especially favorable for waste discharge compared to the Atlantic and Gulf Coast, with its broad shelf because:

- coastal currents are stronger;
- flushing (or exchange) of shelf water with ocean water is faster;
- outfalls can reach deeper depths at practical distances from shore and get higher dilution
- density stratification persists almost all year;
- outfall diffusers are deep enough to cause the diluted sewage plume to be trapped below the pycnocline for most of the year;
- dissolved oxygen concentrations are closer to saturation levels at all times;

* See note at end of the chapter.

storms do not stir up the bottom as much;

effluents are less likely to be biostimulatory in an adverse way because of greater volumes of ocean water (i.e., higher dilutions).

The ocean around Hawaii is also favorable hydrographically because of deep water and good flushing. In Puget Sound the waters are deep and exchanges with the open ocean are strong enough that the Sound enjoys favorable disposal conditions except in its innermost channels. But the normal shallow water estuaries are the least favorable of the five groups listed above and effluents will definitely require higher degrees of treatment (secondary or more).

The Section 301(h) waiver regulations (for waiving the secondary treatment requirement) require applicants to submit substantial information on the marine environment near each proposed discharge and to demonstrate that the proposed discharge will not interfere with maintaining a healthy indigenous environment. (For complete statement of the regulations, see EPA, 1979.) The waiver process implicitly recognizes regional variations in the marine environment and allows the decision-making process to take these variations into account. This change will increase overall economic efficiency by allowing for systems analysis including: control of industrial sources, sewage treatment processes, design of outfalls, and characteristics of the ocean receiving waters.

Particles

In the discussion of chemistry in Chapter 6, we saw that particles in municipal sewage effluents are far more important than dissolved constituents in their potential for adverse impact on ocean waters. Particles range in size down to sub-micron, and carry many of the nutrients, most of the trace metals, and significant portions of the persistent trace organics. They also provide a substrate for bacteria and viruses. The particles themselves are mainly organic carbon which readily decays in a few months in the presence of oxygen. The oxygen demand of the particles (as well as the BOD of dissolved compounds) is easily satisfied by the natural supply of dissolved oxygen in the ocean, provided high initial dilutions (>100) are achieved by an outfall diffuser. In-plant reduction of BOD by secondary treatment is simply not

necessary if good dilution and flushing are possible at the ocean site. However, it is highly desirable to provide primary treatment (essentially sedimentation tanks for gravity separation of settleable solids and floating material).

What are the environmental problems associated with particles?

1. Even the fine particles which escape the sedimentation tanks tend to flocculate rapidly and settle as soon as the effluent is highly diluted in oxic salt water (Chapter 6); significant zones of organic enrichment of the bottom sediment around an outfall diffuser may occur. The sedimentation of sewage particles disturbs the benthic community, and the bottom sediments may become anoxic if the sedimentation rate is greater than the rate at which the organic settled material is oxidized in place on the bottom.

2. Trace metals in the sewage are primarily attached to the surfaces of particles, and dissolve slowly from the particles as long as they are suspended in the oxic water column. However, the dissolved metal fractions in the effluent, plus the additional amounts dissolved from the particles, are generally such small additions to ocean background values as to be a negligible problem (for high dilution outfalls, even with significant industrial waste inputs). When particles reach the bottom, the metals will still be released very slowly if the sediments are oxic, or not at all if the sediments are anoxic. In the latter case, the "ultimate fate" of the metals is to be locked into the sedimentary profile.

3. Trace organics are also attached to particles and may thus be accumulated on the bottom sediments over long periods of time, with gradual release to the environment.

4. The particles reduce light transmission in surface waters interfering with growth of a wide range of organisms, ranging from plankton to giant kelp. However, if growth is not limited by a shortage of nutrients, typically nitrogen, then too much light may overstimulate growth in the presence of sewage nutrients (this is more likely to be a consideration in estuaries and enclosed bays).

As indicated above for the four kinds of impact, the particles per se may not be bad, but only as they

interact with other factors. Problem 1 (organic enrichment of benthos) can be mitigated through more dilution and wider dispersal by appropriate outfall design and location. Problem 4 (light reduction) is reduced in the same ways and also by trapping the sewage field below the thermocline (or pycnocline). Problems 2 and 3 (trace contaminants carried on particles) disappear if the input of metals and trace organics is sharply reduced by source control. But as a backup measure, the treatment can be improved over conventional primary sedimentation (which removes only about 50 to 60 percent of suspended solids, leaving 40 to 50 percent in the effluent). "Advanced primary" -- such as adding chemicals to enhance flocculation -- can achieve nearly 70 to 80 percent removal, cutting the remainder down to about 20 to 30 percent. Because no new tanks are required, the cost of upgrading to advanced primary is quite small (mostly operational) compared to going the next step -- secondary treatment, which boosts the plant removal of suspended solids up to 85 to 90 percent. Still the finest particles will remain, and with their large surface areas, they can still transport significant amounts of toxic pollutants.

For resolving problems involving sewage particles, decision-makers should seize low-cost opportunities to enhance primary sedimentation, to improve source control of toxic substances, and to improve outfall dilutions and/or distance from shore. On the other hand, except for very small systems, secondary treatment provides, for the money spent, the least benefits for protection of the marine environment -- BOD reduction is simply unnecessary in most cases. Enhanced particle removal by advanced primary treatment also provides a benefit in reducing chlorine demand in case disinfection is done.

Finally, more research on particle behavior is needed for such questions as: how to improve primary sedimentation; how to prevent flocculation and rapid settling after sewage is mixed with seawater; and how metals and trace organics interact with particles.

Effluent vs Sludge

Federal law and its administration often treat effluent and sludge as if they were completely separate types of waste, whereas in fact, of course, they come from the same source. For some systems, both sewage effluent and digested sludge are discharged to the ocean (as has been the practice of

the Hyperion Treatment Plant of the City of Los Angeles for the past 20 years), one may ask why the solids are separated from the raw sewage in the first place. There are several good reasons:

1. primary sedimentation removes floatable particles and oil, as well as settleable solids (the floating material -- a small fraction of the total -- is burned or sent to landfill);

2. the settled sludge is digested anaerobically, producing useful methane gas;

3. the size of sludge particles is greatly reduced during digestion, so that wide dispersion is facilitated in the ocean after discharge;

4. the organic material is biologically more stable (less BOD) after digestion;

5. for environmental reasons, the location and means of disposal for sludge solids in the ocean may be different from that for effluent; and

6. keeping the sludge solids separate from the effluent allows more flexibility for future management choices, especially for control of toxic substances which are preferentially attached to the sludge particles.

However, when secondary treatment is considered for ocean discharges, the decision-making process should give full consideration to the problem of disposing of the additional amounts of sludge produced (National Academy of Sciences, 1978). In the absence of secondary treatment, the fine particles which would have become secondary sludge would instead be discharged to the ocean as a component of the primary effluent. Such particles may represent only about 100 mg/l in the effluent, or about 1 mg/l after dilution by a good outfall. Since sludge coming from the digester is likely to have a solids concentration of 30,000 mg/l or more, for ocean discharge we would need to achieve a dilution of at least 30,000:1 (which is impossible) to get to the low level of 1 mg/l, as if the particles had been left in the effluent in the first place. One can question the benefit obtained by concentrating the particles in the secondary treatment process when high dispersal facilitates their assimilation in the ocean. Also, at the low incremental level of 1 mg/l there is sufficient

dissolved oxygen to meet the particle BOD, and any dissolved trace contaminants will be at very low concentrations.

The policies for ocean disposal of effluents and sludges should be based on comprehensive analyses of all the effects of alternate engineering systems for managing sewage and sludge. Furthermore, in view of the current terrestrial problems caused by past waste disposal practices (Love Canal, etc.), the ban on ocean discharge of sludge should be reconsidered in the light of current scientific knowledge. An evaluation of risks and costs of various alternatives for discharging sludge into the ocean (by barge and pipeline) should be pursued. Some of the alternatives could be improved in ways which would reduce effects below those experienced in current practice (i.e., sludge discharges at 300 m depth; see Jackson et al., 1979).

Fates and Effects of Dissolved Pollutants

Dissolved pollutants have much less effect than pollutants carried by particles, for basically two reasons. First, the biological uptake rate of dissolved species appears to be much less than for particles upon which trace contaminants may be concentrated. Second, dissolved materials in the water column are transported great distances away by the overall coastal flushing process, thereby spreading the effects widely at a low level.

With proper source control and/or industrial waste pretreatment, followed by primary treatment, and an outfall giving at least 100:1 dilution, the concentrations of dissolved contaminants can readily be kept to safe levels in the water column, avoiding both acute and chronic effects (such as those defined by the California Ocean Plan, Appendix A).

Source Control of Toxic Substances

Source control of toxic substances in now considered essential for all wastewater systems regardless of the treatment methods or disposal techniques; for example, if secondary or other advanced treatment is applied for removal of toxicants from municipal effluents, then the same toxicants are being added to another waste stream -- the sludge -- and are still part of the disposal problem.

Until the 1970s, sewerage agencies did not attempt to exercise much control over industrial contributions to sewers so long as such discharges did not damage the sewers, upset the treatment processes (like poisoning the sludge digesters), or threaten the health and safety of treatment plant personnel. Various state plans (e.g., California's Ocean Plan, first adopted in 1972) and the Water Pollution Control Act of 1972 (leading finally to the 1980 EPA proposed pretreatment standards, see Chapter 2) have required discharge agencies to find out where their toxics are coming from and regulate them. Large agencies have literally thousands of industrial waste dischargers under permit, so that the administrative problems are large. Agencies can observe the total concentration of each toxic substance, but when concentrations must be reduced to meet standards, it is hardly clear how the reductions should be achieved. If the waste is produced uniquely by only certain industries (like silver in the photographic processing industry), then tracking down the sources and enforcing limits is easy because they are few in number. Or, if the toxic discharge is exceptionally large, it may be tracked to a large manufacturer (like the large DDT discharge from Montrose Chemical Company to the Los Angeles County system in the late 1960s). But other substances such as oil, PCBs, and TCE (trichloroethylene) are used widely by many industries in moderate amounts with hundreds of significant sources in a large system. Even individual householders using products from shelves of any supermarket can contribute significantly to the problem. For more discussion of progress in source control, see Chapter 2.

As explained in Chapter 2, diligent agency efforts at source control may not increase environmental protection because a plant which is denied the use of the sewer for disposal of hazardous wastes may simply dump them elsewhere (e.g., on the ground), or hire someone to carry away the wastes for landfill disposal or even surreptitious dumping in a nearby water source. These alternatives may be even more harmful to the environment than ocean outfall discharge. Therefore, a good program of source control (or pretreatment) requires that safe alternative means of hazardous waste disposal be made available. In addition, industrial plants should be encouraged to use alternate, less hazardous, chemicals.

Source control of toxic substances is a technical and administrative problem for sewerage agencies which has not yet been generally solved, in

spite of notable progress by some agencies with some substances. Management approaches are discussed in more detail later in this chapter. The experience record is short, but it is absolutely essential that adequate source control be made a reality in the next decade. There simply are no technically feasible alternatives.

Disinfection of Effluents

Routine chlorination of effluents before marine disposal has been found to produce chlorine compounds which can be damaging to marine ecosystems. Furthermore, the transport and application of large amounts of chlorine represent some human health hazards in case of leaks or spill; and the cost of chlorination is high because of the large amount of energy needed to produce chlorine. Consequently, some regulatory agencies have dropped mandatory disinfection and encourage the use of outfalls which are long enough and deep enough so that disinfection is not necessary to maintain acceptably low bacteria counts on beaches and at shellfish beds. (This is generally the case for West Coast outfalls.)

If disinfection is unavoidable, then agencies may be required to install de-chlorination facilities after chlorination. Although this procedure eliminates free chlorine residuals, it does not eliminate the organic chlorine compounds which may be harmful to marine life. Other disinfection techniques are being researched, but one alternative, ozonation, still produces biocides and is much more costly. For small systems without a deepwater outfall, a reduction in disinfection costs may be a reason in favor of installing secondary treatment (because less chlorine is needed for higher quality effluent).

Predictability of Outfall Effects

Effective management of wastewater and sludge discharges to the ocean depends critically on the ability to measure and predict the effects of treatment and outfall systems. The design of new or revised outfall systems is based on certain water quality objectives to be satisfied by the system under design. Fifty years ago outfalls were designed simply as hydraulic pipelines into the sea with the location and length decided either by guessing or by pure judgment, without any scientific analysis of the resulting water quality in the ocean.

In the intervening years, the state of the art has advanced considerably, especially in the physical aspects of design. In the design of outfalls we can now do the following predictive analyses quite well:

-- initial dilution for a multiport diffuser, taking account of environmental factors such as ambient density stratification and currents

-- maximum height of rise of the initial plume in a stratified ocean, i.e., the level at which neutral buoyancy is achieved between the diluted plume and the ambient ocean and further spreading is done as an internal flow

-- frequency of advection toward shore and probable travel times

-- coliform die-off and expected coliform counts along the shore

-- maximum dissolved oxygen depletion based on dilution

-- order of magnitude of regional flushing based on oceanographic variables

-- rate of dissolution of particulate forms of metals on sewage particles

-- substances likely to be bioaccumulated in the food chain and to require special attention and source control

We have good empirical observations, but less ability to predict, with regard to things like the following:

-- settling velocity of particles as a result of flocculation in the ocean

-- size and extent of resulting enriched patches on the bottom near outfalls

-- detailed ecological effects of wastewater, other than by empirical comparison to existing discharges

-- biochemical conversion of trace contaminants to more toxic forms by marine organisms

-- rate of degradation of potentially toxic persistent organics

None of the predictions are exact; there is a degree of uncertainty. However, the performance of an outfall is not fixed but follows frequency distributions driven by variations in the ocean environment (the time of the year, storms, tides, etc.). Consequently, when all the factors are considered and conservative assumptions are made, recent design experience has been successful in meeting the prescribed sets of water quality requirements.

The reliability of the predictive modeling techniques is described in Chapters 4 and 5 for physical processes, Chapter 6 for chemical processes, Chapter 7 for biological and ecological effects, and Chapters 8 and 9 for health hazards.

In summary, the development of outfall structures to achieve very high dilutions (over 100:1) in locations of good coastal water circulation has been instrumental in achieving good disposal for the Class I and Class II pollutants -- the normal ecosystem materials and pathogens. In many locations, secondary treatment is not necessary as part of the system and primary treatment is sufficient.

In recent decades, however, we have failed to predict the effect of trace contaminants such as DDT and PCBs because of their persistence in the environment and bioaccumulation in the food chain. It is now generally agreed that management of this part of the ocean-disposal problem depends not on better outfall designs or more advanced treatment, but rather on controlling the release of hazardous substances into the sewer system in the first place. By measurements of the effluent for trace contaminants it is easy to establish which substances need special attention and source control, and it is easy also to measure an agency's progress toward that goal. What is not within the state of the art, however, is deciding what values of various trace organics can safely be discharged to the ocean. As instrumentation gets better, we expect to find that, for many substances undetected until now, the concentrations are small yet measurable quantities. Since we do not know how small is small enough, we have to take some risks, but these risks are no different and perhaps less than similar risks we face constantly in all parts of our environment, vis-a-vis trace contaminants.

While we have a good understanding of normal variability of ocean characteristics such as temperatures and currents, we know less about major

infrequent events like the anoxia that occurred in subsurface waters of the New York Bight in the summer of 1976. Some such events probably occur as a consequence of natural incidents like large floods which bring big doses of nutrients to coastal waters, or they may be the results of instability in zones which continually receive large amounts of sewage and sludge. If the cause is the latter by itself, then there is a question as to why large-scale events occur so infrequently when wastewater and sludge have been discharged -- to the New York Bight, for instance -- on a continuous basis year in and year out.

Reversibility of Effects

When a sewage discharge in a particular location is stopped or in some way modified, we have a unique opportunity to observe the recovery of the ocean waters or the bottom from effects that previously existed. For conventional pollutants (Classes I and II), recovery of benthos is rather quick, and in a few years, the effects of the previous sewage discharge may be completely gone. By comparison, in terrestrial ecosystems, an action like cutting a forest may produce disturbances which take many decades to heal. However, if the ocean is used for marine disposal (with proper control of hazardous substances), the risk is minimal in a long-term sense; in some cases it may be feasible then to develop the management plan for a discharge by increments based on observed impacts of each step. For example, a new dump site for sludge could be established (if not prohibited by law) with careful monitoring of effects; if the impacts are judged to be reasonable, the dumping could continue, but if not, the dumping could be terminated.

However, the reversibility of effects regarding trace organics and heavy metals is very much slower, and may at some times be irreversible if the doses are large enough to cause failure of certain species in the marine ecosystem. The effects of DDT in the Southern California Bight are still there although the industrial source was shut off in 1970. Large reservoirs were created in the sediments and in the marine organisms which do not disappear because the substance itself has such a long life. Therefore, caution is necessary to prevent the release of persistent materials which may be damaging, because we cannot take the "experimental" approach that is possible with conventional pollutants.

Summary of Key Technical Issues

In this section, we have enumerated some of the most important technical factors that affect management decisions for discharge of sewage effluents and sludge to the coastal waters. Included among these are:

-- There are sufficient differences in hydrographic characteristics of different parts of the United States coastline that a uniform ocean discharge policy is not appropriate.

-- Particles are important in their effects in two ways -- as carriers of toxic substances and because they can flocculate, settle, and accumulate on the bottom around outfalls.

-- The physical, chemical, and biological analyses of sludge disposal in the ocean are similar to that for effluent, except for the greatly increased number of particles; the separation of particles from effluent (as in secondary treatment) may or may not be useful if both sludge and effluent are to be discharged in the ocean.

-- Dissolved constituents in wastewater are a secondary problem compared to particulate phases because they stay in the water column for better flushing and are taken up less readily by marine organisms.

-- Source control of toxic substances is technically the only feasible means of control, but success on a broad scale is not yet part of our operating experience.

-- Disinfection is a tool for control of pathogens which can be used sparingly as needed, but with recognition that it is a process which has its own adverse effects on the marine environment.

-- Models for predicting the physical, chemical, and biological effects of outfalls are sufficiently advanced that we may satisfactorily design projects to meet prescribed ocean requirements.

-- The predictability of the fate of toxics in the ocean, however, needs improvement, although we are able to set approximate upper limits on physical and chemical processes.

-- Effects of traditional contaminants are readily reversible in a few years, whereas the effects of toxics may be permanent or reversible only over a period of decades.

In the next two sections we shall discuss further the implications of these results for ocean discharge management and policy.

MANAGEMENT ALTERNATIVES

In the preceding sections we summarized the current practices and identified the key technical issues affecting management alternatives. Our knowledge of transport, fates and effects has increased considerably in the last decade and further advances of knowledge are expected. Therefore, management of ocean discharges should be directly related to knowledge of ocean processes, while being flexible enough to be adjusted in the light of new research results; in addition, alternate disposal processes to air or land should be evaluated. The ocean disposal option should have technical parity with land and air disposal options. In retrospect, the Ocean Dumping Act (in effect banning sludge dumping by 1981) lacked a sound scientific basis and precluded an evaluation of alternate disposal options.

The technical problems of discharge to the ocean are different from discharges into rivers or inland waters or estuaries. The ocean has great value for assimilation of wastes and the policy governing it should be separate from discharges into other bodies of water. For example, biochemical oxygen demand (BOD) is rarely a problem for ocean discharge, but for discharges to certain inland waters it may be an overwhelming problem that necessitates secondary treatment.

This section discusses management alternatives in the context of our current understanding of the needs for controlling ocean pollution from municipal sources.

Toxic Substances

A conclusion clearly following from the discussion in foregoing chapters of this book is that the ocean can safely be used to receive wastewaters, provided we improve our ability to control the entry of toxic substances into municipal sewer systems. The

overriding future risk of ocean discharge of municipal waste is the intentional (or unintentional) introduction of long-lived toxic substances into the marine environment in concentrations which may be damaging to marine ecosystems, or a threat to human health by consumption of seafood. Since similar threats of toxic material exist also in air and on land, the ocean problem is not likely to be solved by a displacement of such substances to another environmental medium. Rather, there is a consensus that the best approach to control of toxic substances is at their sources. There must be a change in the general attitude that sewers are for dumping anything that you want to get rid of.

The concept of source control for municipal sewer systems is to reduce or control the amount of a trace contaminant entering the sewer system from industrial plants or other sources. This is probably the most cost-effective approach -- or possibly the only viable approach. To allow hazardous substances to enter the sewer system in excessive quantities creates what appears to be intractable problems of disposal. If the hazardous wastes are captured in sludge by the sewage treatment process, then there is a problem of safe disposal of the sludge. On the other hand, if they pass through to the ocean there may be risks to the ecosystem or human health.

Although we noted earlier in a section on "Fates and Effects of Dissolved Pollutants" that the trace contaminants in solution probably pose less marine impact than those attached to particles, it does not mean that industrial waste discharges of dissolved contaminants can necessarily be better tolerated. The reason is that dissolved constituents, upon being introduced into the sewer system, may rapidly become adsorbed onto the many particles already in the wastewater, depending on the overall chemistry of the mixture, and cannot be expected to stay in solution.

Source control involves a range of possible actions by industrial dischargers:

-- reduction or elimination of use of a chemical by substitution or change of process

-- recovery of waste products from industrial wastewater before discharging to sewers (e.g., recovery of silver by photographic laboratories or metals by plating firms)

-- chemical treatment of wastes to convert them into less toxic substances

-- removal of hazardous wastes to landfill

-- removal of hazardous wastes by burning or discharge to the atmosphere

Chapter 2 gives examples of progress in source control as measured by the constituents in raw sewage.

These control activities all have their impacts on other media and may represent significant costs to the dischargers. However, for economic efficiency, source control is probably a good way to impose otherwise external costs of environmental control on the industries which are potentially damaging, and thereby stimulate the development of control technology or alternative products.

Sewerage agencies generally have the authority by local ordinance to regulate the discharge of industrial wastes to the municipal sewer system, and to impose costs on such dischargers for the waste loads added. Sewerage agencies do not have to take wastes which are damaging to the sewer system, the treatment plants, or the receiving water environment (for example, see the Industrial Waste Ordinance of the Los Angeles County Sanitation Districts). A sewer system of a community may be considered to exist primarily to receive human and domestic wastes; and accommodation of industrial wastes is a matter of convenience and is reached by mutual agreement.

The establishment of source control depends on setting quantitative limits and having means of monitoring and enforcement. There are two ways to establish limits for a given pollutant: 1. by setting maximum allowable concentrations in a plant's industrial wastewater; or 2. by setting limits on mass emission rates. The latter is the more logical approach because the total material flux is what matters to the treatment plant or the marine environment.

Setting concentration limits for industrial wastewater may be convenient for monitoring, but it only encourages wasting water (in order to dilute the waste products below the concentration limits). Furthermore, concentration limits are a strong disincentive to active water conservation; i.e., a firm which meets the concentration requirements may

not be able to save water because their effluent concentrations might increase above the limit even though the mass fluxes would remain the same!

For individual dischargers, the setting of source control limits (or pretreatment standards, as they are sometimes called by regulatory agencies) may be approached in two ways:

1. If a sewerage agency has an overall maximum allowable mass emission rate through an outfall, it may then allocate that limit among various industrial dischargers in a way which is most efficient.

2. Uniform pretreatment standards may be set by a regulatory agency, such as the national standards being developed by EPA for various industrial categories (see Chapter 2).

The first approach is appealing because it essentially represents a back-calculation from the environmental objectives, so that excessive control costs are avoided. The EPA approach (pretreatment standards) may be administratively attractive, but fails to provide flexibility for regional differentiation or consideration of the characteristics of a particular sewerage system. For example, a certain quantity of acid wastes might be devastating when introduced into a small sewer system, but easily buffered when combined with flow in a large sewer system.

So far there has not been extensive experience with source control. When there are only a few major dischargers of a certain substance into a particular sewer system, it is of course administratively easier to get significant source reduction. On the other hand, when there are hundreds of dischargers of a more commonly used substance, all in moderate amounts, it may be much more difficult to achieve an overall reduction in the total wastewater stream. Another very difficult problem is the ever-increasing number of toxic substances to be controlled (e.g., see list in Chapter 9). The costs of detailed chemical analysis for monitoring large numbers of sources in major cities is becoming a significant cost factor in municipal waste disposal. Experiments with the administration of source control are needed to find good cost-effective procedures.

The most difficult remaining technical and economic issue is how much source control is enough.

Too little source control increases the risks in the marine environment (and wherever else sludge may be disposed of). On the other hand, too much source control may divert too much of society's resources to reduction of emissions to rates which are lower than necessary for satisfactory environmental protection. To do too much may not necessarily be wise or prudent because of the costs and risks of alternative approaches. Chapter 10 discusses risk management and the difficulties of making decisions including the risks of false negatives and false positives in environmental tests and environmental impact assessments.

In summary, much is to be gained from good programs of source control, but progress to date has been uneven. Much remains to be learned, both about administrative approaches and risk managment in making decisions about effluent limits.

Treatment Processes for Ocean Discharge

When conventional secondary treatment (activated sludge) was invented, its main objective was to reduce the BOD of sewage before discharge into fresh water bodies with limited oxygen resources. However, in the preceding sections we have seen that BOD is not generally a problem in the ocean, and that rather the priority for treatment for traditional (Class I) pollutants is particle removal (i.e., suspended solids removal). In Chapter 2, it is indicated that conventional primary sedimentation may remove about 50 to 60 percent of suspended solids, whereas secondary treatment can be expected to remove 85 percent. However, there are opportunities for intermediate levels of particle removal, called "advanced primary", which are basically enhanced sedimentation processes. The addition of flocculating agents such as polymers, alum, ferric chloride or lime tend to increase suspended solids removal up to the range of 70 to 80 percent (see Chapter 2). The cost of advanced primary varies considerably with the coagulating agent used, with polymers being probably the most promising, because of the small amount necessary. If too much flocculant is added there may be a large cost for the chemical (e.g., lime) and substantially increased cost of sludge disposal. For ocean discharge it is recommended that additional research be done on developing cost-effective improved sedimentation processes.

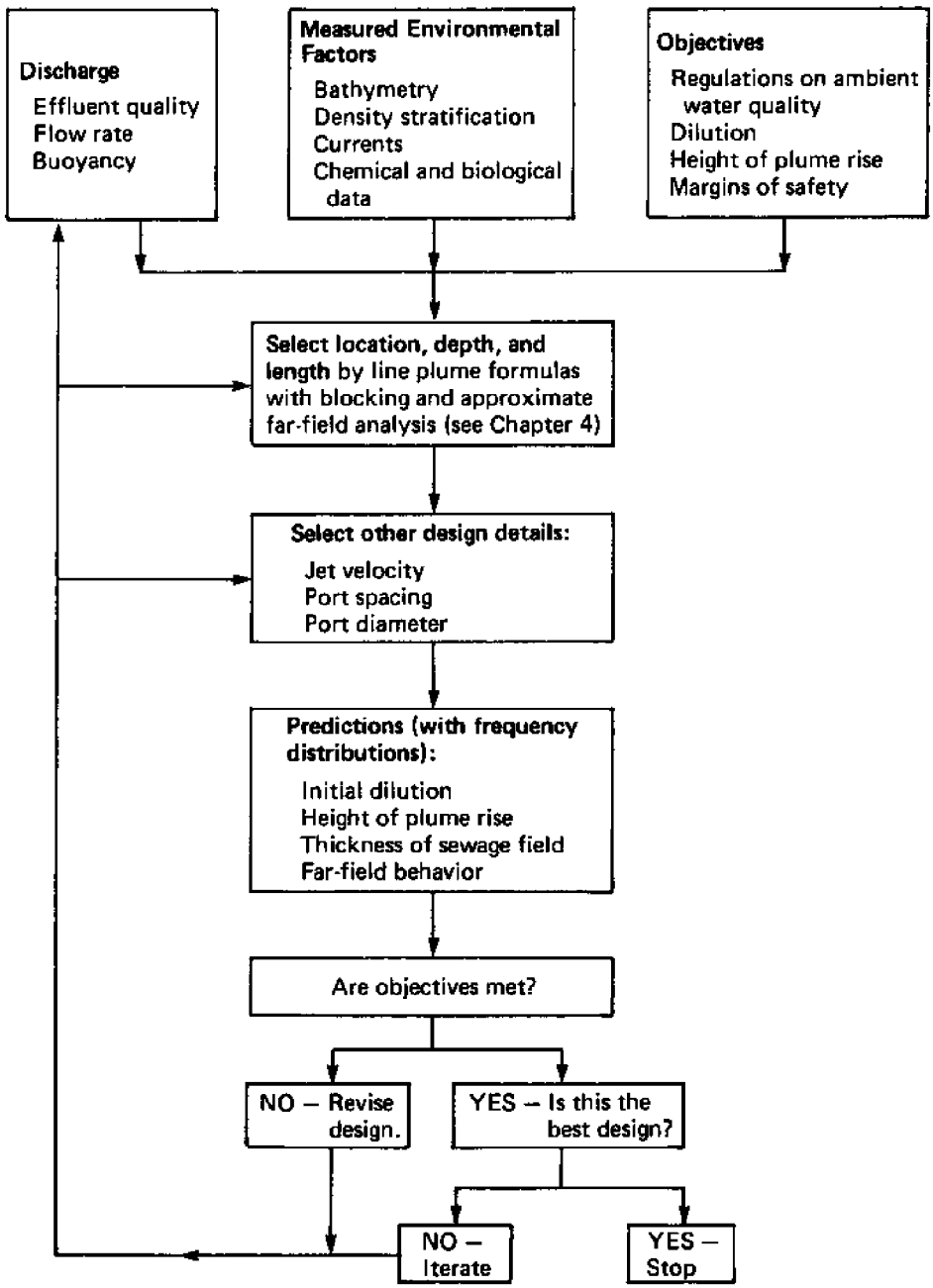
One important advantage of advanced primary over full secondary treatment is the fact that polymers (or other chemicals) can be added with minor modifications to existing primary sedimentation tanks at relatively low cost in structures and land. On the other hand, to upgrade a plant to full secondary treatment requires large new batteries of tanks for in-plant aeration and final clarification in addition to the primary sedimentation tanks; the area of tanks is typically doubled.

Another example of how treatment processes and outfall design have been made specific to ocean discharges is disinfection. It has been found with long ocean outfalls, such as off southern California, that there is a trade-off between outfall length and disinfection, while maintaining the same shoreline bacterial count. For example, in Orange County, when the Orange County Sanitation Districts stopped using the old 7,000 ft outfall and started operation of the new 27,400 ft outfall in 1971, effluent chlorination was completely stopped. Previously heavy chlorination was necessary over 50 percent of the time to meet shoreline bacteria standards, which are now easily met by natural processes in the ocean. Not only is the saving in chlorine cost very significant, but also the risk of damage to marine ecosystems and in-plant chlorine hazards to personnel have been avoided.

Outfall vs Treatment Plants -- Trade-offs

For effluent discharge into the ocean the basic choices and trade-offs involve level of treatment and the design of the outfall. For the following discussion we will presume that a good program of source control is being implemented, and thus we focus on Class I and Class II pollutants (organic material, nutrients, and pathogens). The performance of an outfall in diluting and dispersing effluent is characterized by the initial plume dilution over the diffuser, possible plume submergence beneath the pycnocline, and current patterns (see Chapters 4, 5, and 8). Figure 12.1 shows schematically the procedure followed in designing an outfall, starting with: 1. data on effluent quantity and quality; 2. environmental data for the ocean; and 3. ambient water quality requirements, and dilution and submergence objectives. The optimum location of the discharge, the diffuser length and depth, and various port details are then determined so that the requirements are met with the desired margin of safety. (For details, see Chapter 4.)

Figure 12.1 Design procedure for sewage outfall with diffuser.



By this process the treatment plant and the outfall are considered as a system with trade-offs. When the treatment is more extensive (e.g., secondary with disinfection), the outfall can discharge closer to shore with less dilution; but on the other hand, when high outfall performance is easily and economically achieved, then less treatment (e.g., primary without disinfection) may be sufficient.

The trade-offs are not simply a matter of cost. When secondary treatment is used (with an implied shorter outfall) there is an additional disposal problem for the increased amounts of digested sludge which may have intangible as well as tangible costs. Another factor is that for secondary treatment a long high-dilution outfall may still be desired for ecological reasons, even if it is not essential for bacterial, dissolved oxygen, or turbidity requirements. Since a large fraction of the original nutrients remain in the secondary effluent, high dilutions are helpful to avoid undue biostimulation.

It is practically impossible to make a generalized economic analysis of the trade-off between outfalls and degree of treatment. It should and must be done on a case-by-case basis. Under the 1972 law, this was not permissible, because secondary treatment was mandated regardless of the character of the receiving water or the outfall design; now with the 1977 amendments, including Section 301(h), such a systems approach is possible because it is recognized that the resulting ambient water quality depends on the outfall design and receiving water characteristics as well as the level of treatment.

In the author's experience, it has been found that the cost trade-off is sensitive to the mean flow of the system. For small flows, perhaps on the order of 1 to 10 MGD*, the addition of secondary treatment may be cheaper than building a sufficiently long outfall; on the other hand, for very large systems (several hundred MGD) the cost advantage of a long outfall in deep water over secondary treatment may be very large. The reason for this is that the treatment costs tend to be nearly proportional to the flow rate, whereas outfalls exhibit a much larger economy of scale.

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* 1 MGD = 1.55 ft^3/s = 0.044 m^3/s.

The cost of an ocean outfall depends not only on water quality requirements, but also on the physical hazards and foundation conditions at the site. Factors which will increase outfall costs significantly are: large ocean waves, shifting bottom profiles, poor foundation conditions, earthquake faults, and hard rock and coral which must be excavated. The cost then is very site specific. The depth profile also enters into the cost significantly in the sense that an outfall is most difficult to build through the shallower surf-zone (typically up to 30 ft depth) and easiest to build in the offshore sections, where burial may not even be required (i.e., the pipe may be simply laid on the bottom with a rock berm on each side to hold it in place).

To get some idea of cost for purposes of comparison with the treatment costs developed in Chapter 2, we may consider the example of the Sand Island Outfall at Honolulu, Hawaii, which was finished in 1975 (see Table 4.1 in Chapter 4). The design average flow of that outfall is 164 ft^3/s or 106 MGD, approximately the same as for the nominal 100 MGD plant alternatives studied in Chapter 2. The length of this outfall (7 ft inside diameter) was 13,500 ft including a 3,384 ft diffuser with a terminal depth of 235 ft. The construction cost was $13.6 million (contract awarded in October 1973); in 1980 prices this would probably have been about twice as much, or $27 million. The construction conditions would be considered somewhat more difficult than average because of about 6,000 ft of coral excavation through the surf-zone. Although the <u>average</u> construction cost would be about $2,000 per foot, the marginal cost of additional length would be much less. A rough cost formula for an outfall for 100 MGD flow might be

$$C = 5{,}000{,}000 + 2{,}500\ L_S + 1{,}000\ (x - L_S)$$

where C = cost in dollars
L_S = length of the surf zone in feet (requiring trestle construction)
x = outfall length including diffuser (in feet)

The $5 million constant is the basic fixed cost of mobilization and demobilization. The marginal cost for length is thus estimated to be $1,000 per foot for a 7 ft outfall for 100 MGD for original construction, but not for retrofitting which requires remobilization of heavy equipment.

For a hypothetical example, the required outfall length for primary effluent might be 18,000 ft, but

only 12,000 ft for secondary; the surf zone is taken as 4,000 ft. The cost comparisons for two equivalent treatment-outfall combinations are given in Table 12.1. Since the capital costs shown are original project costs, the differences understate the cost of projects to upgrade; i.e., if an outfall to 12,000 ft is already built, then the cost to extend it to 18,000 ft would be much more than $6 million because of the mobilization cost; and if a primary plant already exists, the addition of secondary processes would cost more than the $28 million differential shown.

Although the analysis presented in Table 12.1 is very generalized and ignores site-specific factors, it does show the nature of the cost trade-off between a longer outfall and more treatment (presuming that the ocean currents are favorable for flushing highly diluted primary effluent). The savings in annual outfall costs for secondary effluent versus primary effluent would be only approximately $0.7 million for this case; on the other hand, costs of secondary treatment would represent an increment of $6.1 million for a net increase in cost of $5.4 million annually. This represents about a 50-percent increase over the option of primary plus a longer outfall.

There may be instances where the comparison may be more striking or less striking. The design of the Barber's Point outfall in Honolulu (a new system) considered the effect of the treatment level on outfall design (see R.M. Towill Corporation, 1974). Although the outfall was officially designed for secondary effluent, the design report indicated that the additional length needed for primary effluent was only 600 ft out of a total length of 10,500 ft, and represented only 2 to 3 percent additional cost. Because of the possibility that secondary treatment might not be required in the future, the designers recommended building the slightly longer outfall (i.e., 10,500 ft) to take advantage of the possiblity of not being required to go to full secondary. (The mean design flow for this outfall was 59 MGD; the peak design flow 112 MGD; the depth of discharge 195-200 ft; the length of diffuser 1,750 ft, which could have been 1,150 ft for secondary effluent; and the inside outfall diameter was 78 inches.) One of the reasons the marginal length (and cost) is so low here is the bottom topography; the outfall crosses a long coral shelf and then follows a rather steep decline to a depth of about 195 ft where the diffuser turns and follows the 200 ft contour. In order to meet the water quality requirements (especially for nutrients), and take advantage of the favorable density structure

Table 12.1 Estimated Typical Cost Differentials of Outfalls and Treatment Needed for Primary Versus Secondary Options (Costs in $ Million in 1980)

Treatment: / Outfall Length:	PRIMARY 18,000 ft		SECONDARY 12,000 ft		Difference (Sec. - Prim.)	
	Capital	Annual	Capital	Annual	Capital	Annual
OUTFALL						
Capital Cost	29		23		-6	
Annual Capital Cost (10.25%, 75 yrs)		3.0		2.4		-0.6
Annual O & M		0.3		0.2		-0.1
Subtotal	29	3.3	23	2.6	-6	-0.7
TREATMENT PLANT*						
Capital Cost	41		69		+28	
Annual Capital Cost		5.2		8.8		+3.6
Annual O & M		1.8		4.3		+2.5
Subtotal	41	7.0	69	13.1	+28	+6.1
TOTAL OUTFALL & TREATMENT PLANT	70	10.3	92	15.7	+22	+5.4

*Based on alternatives numbers 1 and 5 in Chapter 2, Table 2.24. Average design flow = 100 MGD.

to achieve plume submergence, it is necessary to go to approximately 200 ft depth for either primary or secondary levels of treatment.

There are other trade-off possibilities. For instance, consider an outfall which is already built and the choice is between lengthening it and adding secondary treatment. In this case, the cost of the incremental length must include the full cost of mobilization of the construction effort; and if the mean flow is small (of the order of 1 to 10 MGD), then installing secondary treatment might be cheaper. On the other hand, if an agency already has an outfall that performs very well with primary treatment, then the secondary treatment requirement may have no trade-off in outfall length (because the outfall is already designed to perform for primary effluent).

In the case of combined sewers, the storage, treatment and outfall disposal are all relatively expensive because of the large flows, especially considering the fact that combined sewer overflows may be a relatively infrequent occurrence (e.g., only 4 percent of the time at San Francisco). Secondary treatment is not the issue here, but rather the question is whether to provide primary treatment for the mixed storm water and sewage; or whether to gradually build a system of separate sewers; or whether to do nothing if it is judged that the damage of occasional overflows to coastal waters is less than the cost of correcting the problem. The federal law at present does not clearly specify how storm water overflows are to be regulated, although the present procedure appears to permit it only when a 301(h) waiver is granted (for coastal discharge of less than secondary treated effluent).

Sludge Disposal -- The Ocean Option

For disposal of digested sludge the ocean option is precluded by current law, but if it were modified the ocean disposal option would be found in some cases to be much cheaper than land disposal or incineration. As an example, the Orange County Sanitation Districts is faced with a program for dewatering sludge and hauling it by truck inland 30 km to landfills in the foothills of the Santa Ana Mountains on the opposite side of the county from the coastal treatment plant. The regional sludge study (LA/OMA, 1980) has estimated the cost of this disposal method as $75 per ton (raw sludge basis) or $9.7 million per year (1977 dollars), if partial secondary treatment is required (planning

Phase II). However, a deep-water sludge outfall to 300-400 m was suggested as a possible alternative by Jackson et al. (1979), who estimated the cost of a 33,000 ft (10 km) outfall (18 to 24 inches inside diameter) to be between $5 and $10 million.* The environmental impacts of such a system were predicted to be low, although additional field research and collection of environmental data were recommended before any project is built. A comprehensive research plan for this possible project has been prepared by Brooks et al. (1982).

A comparison between landfill and ocean disposal costs for Orange County Sanitation Districts is presented in Table 12.2, which includes about one million dollars per year for environmental research and monitoring for the ocean option. Although these numbers must be regarded as preliminary (not official estimates) they do indicate the wide difference in annual costs between land and ocean disposal (about $11.9 million vs $2.75 million). The basis is for 150 tons of sludge solids (dry weight) per day, consisting of mostly primary sludge anaerobically digested and screened to remove any large particles before discharge. It is assumed that under a Sec. 301(h) waiver, full secondary treatment will not be required.

Even if an experimental discharge were permitted for only five years, the savings over landfill disposal could still more than pay for an extensive program of ocean monitoring and fully amortize the outfall pipe in 5 years instead of 30 years. This is an example of a project where an experimental step, if allowed, would be useful, not only for that agency, but as an opportunity to gain valuable information for further developing the methodology for predicting the transport, fates, and effects of sludge introduced into deep water, as described by Brooks et al. (1982). Since the risk of such an experiment appears to be very low, such research and demonstration projects are recommended.

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* LA/OMA, 1980, estimated that for the year 2000 with full secondary treatment (planning Phase III) the cost of deep-ocean disposal would be $36 per ton (raw sludge basis, 1977 dollars), compared to the landfill disposal cost of $86 per ton for Orange County Sanitation Districts. However, the writer believes the capital cost estimates used for a marine sludge outfall were much too high.

Table 12.2 Estimated Alternates Costs of Digested Sludge Disposal for Orange County Sanitation Districts* (150 tons/day)***

	Landfill disposal (by truck)	Ocean disposal (by sludge outfall to 300m depth)
	$ million	$ million
Capital		
Land	23-36	
Dewatering, screening	6.0	1.0
Storage	4.5	
Trucking	2.0	
Pipeline		5-10
TOTAL CAPITAL COST	36-49	6-10
Annual costs		
Capital**	4.0-5.3	0.7-1.2
Annual operation and maintenance	6.6	0.25
SUBTOTAL, Capital and O & M	10.6-11.9	1.0-1.5
Special research and monitoring program		1.75-1.25
TOTAL, ALL ANNUAL COSTS ($ million)	10.6-11.9	1.75-2.75
COST PER TON*** (dollars)	82-92	13-21

* Based on preliminary data for Phase II from Orange County Sanitation Districts, February 1981.

** Based on 10.25% interest, amortized as follows: Land, interest only; storage tanks and outfall pipe, 30 yrs; pumping, dewatering and screening equipment, 10 yrs; trucks, 7 yrs.

*** Actual digested sludge discharge is 150 tons/day which is derived from 350 tons/day of raw sludge. For consistency, unit costs are given in costs per ton of original raw sludge.

The ocean disposal option for sewage sludge, of course, includes barging as well as special sludge outfalls. If barging were permitted in the future, there are various disposal strategies that could be analyzed systematically to predict environmental impacts and to estimate the capital and annual costs. Our ability to predict transport, fates, and effects of barge discharges can also be improved through additional research and monitoring.

The present methods of predicting dilution for barge discharges is described in Chapter 4. The practices of barge dumping off the English Coast and on the New York Bight are reviewed in the two case study papers on these regions (see Appendices C and D). For an evaluation of deep-water Dumpsite 106 off the Atlantic, see Goldberg (1979).

The practice of barging is significantly different from sludge outfalls in two respects (Jackson et al., 1979): first, the discharge is near the water surface so that it can have more impact on the photic zone (by reducing light transmission) than a bottom discharge; and secondly, the barges can be programmed to dump sludge at various places, as desired, rather than at a fixed point as for an outfall. Environmentally, the first point is a disadvantage, while the latter is an advantage.

The cost of barge disposal may be more or less than sludge disposal by pipeline depending on circumstances. Larger discharges tend to favor pipelines because continuous pipeline transport is cheaper than batch transport. Higher dilutions and submerged plumes can be more readily achieved. On the other hand, long distances and/or difficult outfall construction conditions favor barge disposal, especially for small or moderate sludge volumes.

Monitoring

An important part of the ocean option for disposal of sewage or sludge (when permitted) is an adequate monitoring program to identify any trends in environmental effects or to discover any new problems. Monitoring is typically required of the discharge agency by the regulatory agency. The discharger either does its own monitoring or else contracts the work out, and reports to the regulatory body.

There are basically two kinds of monitoring: 1. measurements of the chemical quality of the effluent and the flow rate, and 2. measurements of ambient water quality and benthic conditions in the vicinity of the discharge and along the nearby shoreline. For large discharges the required monitoring stations may be scattered over 10 kilometers in each direction.

The main purpose of monitoring efforts, it appears, is to determine compliance with the regulations rather than for research. For example, measurement of numerous water quality parameters may be required once a week (or more often), but no current data, density profiles, or other information is required even though it might be useful to understanding synoptic conditions at the time the water quality samples are taken. If a discharge is found to be out of compliance at a particular time of sampling, there is nothing that can be done in "real time" (because the offending plumes have already been discharged into the ocean). The fastest response is to start or stop chlorination, or adjust the dose of chlorine, or any other chemical that may be added as part of the treatment process (such as polymers to enhance sedimentation).

Unpermitted doses of toxic pollutants cannot be quickly detected in the effluent nor can the source of them be readily found without extensive investigations. Therefore, it appears that too much money and effort is spent on routine monitoring which could be better spent on research-oriented tasks and looking for new problems. This point is well illustrated by the industrial discharge of waste DDT (until 1970) through the sewer system of the Los Angeles County Sanitation Districts and the outfalls off Palos Verdes. The extensive adverse effects of this discharge were not uncovered by routine monitoring (DDT was not on the list of regulated substances then), but rather it was found through a special series of measurements and studies stimulated by the publication of research findings linking nesting problems of marine birds to bioaccumulation of DDT in the marine food chain.

The discharges of toxic pollutants from municipal sewer systems are very unlikely to accumulate to acutely toxic levels. The question is whether existing levels may be chronically toxic. Thus, the best strategy for monitoring the environment is to use tests which are integrative in nature, such as measuring at regular intervals (like a year) the accumulations of trace contaminants in bottom

sediments, or in particular indicator shellfish at designated locations (Goldberg's "mussel watch"). Such long-term monitoring will be required of any discharger who gets a Section 301(h) waiver.

However, the control of short-term problems, such as infection by pathogens, must be measured by instantaneous point samples of receiving water at some regular short interval depending on the size of the discharge, the intensity of beach use, and the rapidity with which prevailing currents or stratification may change. Measurement intervals are typically in the range of daily to weekly.

The cost of monitoring programs is difficult to determine. However, a good indication is given by an EPA analysis of the cost of providing the necessary information and monitoring in the discussion of the proposed ocean discharge criteria (EPA, 1980, p. 9552). For a large POTW discharging 360 MGD, the incremental user charge is estimated to be only $0.06 per month; however, that would rise to $0.80 per month for POTWs of 5 MGD capacity and to just over three dollars per month for plants of 0.5 to 1 MGD. The sharply rising unit costs per user as the plant size decreases indicates that the monitoring cost is a variable which decreases much more slowly than the discharge. For small discharges especially, it is highly doubtful that the value of the information obtained is commensurate with the costs incurred, simply because small discharges to the ocean represent very small threats in most cases.

In the past, the design of monitoring programs has included an excess of routine measurements, but insufficient research aimed at identifying new relationships or problems. A notable example of a forward-looking monitoring and research effort was the establishment of the Southern California Coastal Water Research Project (SCCWRP) in 1969, by the County Sanitation Districts of Los Angeles and Orange Counties, the Cities of San Diego and Los Angeles, and Ventura County. Organizationally it is separate from the sewerage agencies and under an independent board. The scientific work is guided by a director and a special board of consultants. Although the funding has come primarily from the discharge agencies, additional support has come in recent years from various other research funding agencies. SCCWRP issues comprehensive annual reports summarizing results of numerous special studies (see SCCWRP, 1980),

and occasional special publications, as well as publishing papers in the technical literature.

The agencies which founded SCCWRP discharge about 1 billion gallons per day of effluent to the Southern California Bight. Although they maintain their own routine monitoring effort involving frequent sampling of the shoreline and offshore waters for compliance with the requirements, they wisely established the special organization to conduct ongoing and long-term special studies of the effects. Some of the best results in the literature come from this effort and have been cited extensively in this book. The cost of such research efforts, in the total context of wastewater disposal costs, are small and very cost-effective.

Summary of Management Alternatives

A well-designed program of disposal of municipal wastewater and sludge to the ocean, as we have seen, includes a mixture of source control, sewage treatment and sludge processing, appropriate outfalls (or barging operations), and monitoring and research efforts commensurate with the scale of the discharge and the risks. The level of activity in each category is clearly related to each of the others, and the engineer-manager should seek an optimum combination of efforts. For example, good monitoring measurements of the accumulation of metals in sediments or in the food chain will provide the basis for rational discussions on how much source control effort is needed for particular substances. With monitoring and research information we can focus the attention on substances which are believed to be most important in the environment and avoid excessive efforts on nonproblems.

In Chapter 11, Krier recommends a program of geographically varying "management standards"; i.e., requirements for waste management which are set on a regional or case-by-case basis in sensible time steps, with the flexibility to adjust each control program in response to the ocean observations of the effects of a particular discharge. The compliance schedule would allow enough time and flexibility to study the effects of different actions as the basis for any further tightening of the requirements. It is not necessary, nor economically efficient, to tighten the requirements in one step to the tightest level that might possibly be necessary, because in most instances we are not dealing with a water quality crisis but

rather with some long-range concerns. The ability to make wise and cost-effective decisions is greatly improved by allowing time for a feedback loop to operate.

Another reason for using management standards is to better integrate marine, air and land disposal options. For example, time could be provided for incremental implementation of alternative disposal systems to land or air. For example, if it is believed that it is better to dispose of sludge on land rather than to the ocean, it might be wise to establish such an operation for a minor fraction of the sludge (such as 10 percent) to fully investigate the feasibility and environmental effects of the alternate disposal procedure before making a full-blown commitment to switch.

Heretofore, the federal legislation and regulations have not permitted enough flexibility in management alternatives, as pointed out in Chapter 11. The rigidity of the regulations does not encourage innovation in developing new cost-effective measures. Government policy so far has been relatively unresponsive to new technological approaches or to new environmental research results because of the lack of flexibility or regionalization of the regulations.

POLICY ALTERNATIVES AND ADMINISTRATION*

In the previous section we discussed the key technical components of management alternatives, which could improve the ocean disposal practices for municipal wastewater. This section treats general policy elements of a sensible strategy.

Balancing Ocean Discharge Effects and Risks Against the Costs of Avoidance

It is an obvious proposition that discharge of pollutants into the nation's waters imposes costs -- whether through adverse effects on humans and their environment, or through the threat of such effects. It is an equally obvious proposition that avoiding these costs can itself be very expensive. The need for trade-offs, for balancing one category of costs against the other with an eye to achieving a reasonable balance, is quite clearly posed. Yet, by and large, present federal policy disdains such balancing approach. Put differently, present policy

* See note at the end of the chapter.

reflects a very conservative (in the sense of risk-averse) attitude: high (and costly) levels of control are required, without much regard to resulting benefits. Put still differently, the benefits of demanding controls are presumed to be very large.

A conservative approach is not by any means a necessarily bad approach. Quite to the contrary, for example, it seems clearly justified in the case of toxic pollutants. Whether it is justified with regard to disposal of more conventional pollutants into ocean waters is, insofar as this volume is concerned, an open question. It is clear to us, however, that in the limited instance of ocean disposal of conventional municipal wastes, the conservatism of present policy -- a general (minimum) requirement of secondary treatment for POTW effluent -- is, on balance, unwarranted.

Several considerations, developed at length in earlier chapters, support this conclusion. First, a requirement of secondary treatment for POTWs discharging into ocean waters entails very substantial costs -- not just direct capital, operating, and maintenance costs, but also, for example, increased costs of sludge disposal. Second, alternative technologies -- much less expensive than secondary treatment, but satisfactory because they take advantage of the oceans' large assimilative capacities -- exist and are in fact in use. Third, and assuming effective source-control programs, secondary treatment, for all its costs, may yield negligible benefits. Conventional POTW pollutants -- e.g., BOD and suspended solids -- simply do not pose large risks for ocean waters when subjected to primary treatment and discharged by well-designed means. Even cautious calculations, then, suggest the wisdom of a more balanced approach in the case of ocean waters.

Congress, it appears, endorsed this conclusion in the Clean Water Act of 1977. As discussed in Chapter 11, section 301(h) of that legislation provides for modifications of secondary treatment requirements in the case of some ocean discharges. While the modification provisions are welcome, they are not without shortcomings. They apply only to existing discharges, only for a limited time, and (probably) only to limited coastal regions; they cannot result in imposition of additional controls on other sources; they are rigidly constrained by water quality standards; they entail difficult, expensive, and time-consuming application procedures (the possible adverse effects of secondary treatment have

never received the scrutiny to which proposed modifications are subjected). All in all, it is too early to tell whether section 301(h) will reflect the balanced attention to costs and benefits so important to sound ocean discharge policy. It is clear, though, that the section does neglect a second important policy component, considered next.

Relating Ocean Discharge Policy to Other Disposal Options

1. Alternatives to ocean discharge. Federal law for ocean discharge is written with the underlying premise that land disposal of effluent sludge may often be environmentally superior to ocean discharge. For example, the Ocean Dumping Act, by virtue of the ban on ocean dumping after 1981, presumes a priori that land disposal must be better than ocean disposal for digested sludge and other solid wastes. The multi-media choices and trade-offs for sewage sludge disposal have been discussed in detail in National Academy of Sciences (1978).

Since the basic laws were formulated in the early 1970s, there has been greatly increased awareness and measurement of groundwater contamination by toxic pollutants. There are apparently greater risks than heretofore realized for contamination of groundwater by surface disposal of both solid and liquid wastes, followed by rainfall percolation and leaching of contaminants to the groundwater supplies. Because groundwater is a major source of drinking water, the direct public health risks of land disposal for a given effluent with trace contaminants (and nitrates) are generally much greater than for marine disposal. Not only does the ocean provide much greater dilution (on the order of several hundred to one for a good outfall), but also the ocean is not a source of drinking water.

This book could not include an analysis of all the scientific and institutional factors for alternate disposal to land or to the atmosphere (e.g., in the case of burning sludge). Obviously, the scope of the book would then have become too great, but the policy-maker is nonetheless warned that, in fact, every rational case-by-case decision involving waste discharge into the ocean must compare the costs and impacts of ocean disposal with those of other methods of disposal. If we had only to minimize the impact on the ocean without regard to any other environmental media, then the answer would be very simple: zero

discharge to the ocean. Do anything else with wastewater at any cost but do not put any of it into the ocean!

As long as people choose to live in urban areas along the seacoast, there is no way to avoid some impacts on the quality of coastal waters not only from sewage, but also from storm water overflows, street runoff, aerial fallout, ship pollution, and so on. Only in rare instances will land disposal be a viable alternative to ocean disposal. Factors favoring land disposal might include an arid climate with a high evaporation rate, a big demand for reclaimed water, abundant land, the location of land disposal sites at places where there is no groundwater threat, small wastewater flows, and unfavorable oceanographic conditions for an outfall. In addition, if the source of the wastewater is a significant distance inland from the coast, then land or stream disposal becomes relatively more attractive because of the conveyance cost to the ocean. At the present, one of the longest transit distances for municipal sewerage systems from origin to ocean discharge is about 100 kilometers (e.g., Los Angeles County Sanitation Districts system).

2. Policy. Just as a sound policy on ocean discharge of municipal waste should take some account of the costs and benefits of various control options, so also should it be alert to trade-offs among environmental media; indeed, this second component of good policy is implicit in the first. Efforts to protect ocean waters without regard to effects on other resources -- land and air -- could well result in net social losses. Constraints limiting waste disposal on land or in the air, without regard to their implications for ocean disposal policy, could quite easily yield a situation where all disposal options have been made infeasible. Even with programs to limit the production of wastes and to promote recycling, there will still be residuals which must go somewhere, and the only alternatives are air, land, and water, or some mix of these receiving media.

In light of these considerations, it is evident that a sound approach would: 1. promote feasible source control, including reduction in waste production, pretreatment, conservation, and reuse; 2. direct disposal of remaining residuals, under proper controls, to those media that can best tolerate them. The effort, in short, should be to minimize the sum of all relevant costs (including environmental), taking all media into account.

As sensible as such an approach is, it does not represent present policy. Quite to the contrary, the present approach constrains disposal from all sides, and without much regard to the interrelated effects of doing so. In our particular case, ocean disposal of municipal waste, matters are in fact worse than this. Chief reliance on secondary treatment results in increased sludge production, yet opportunities for sludge disposal are severely limited. Section 301(h) modifications can have a positive effect here, but an unduly narrow one. Modification procedures should include a searching analysis of alternative disposal options, but it appears they do not. The impact on ocean waters of granting a modification is considered with great care; the impact on land and air of denying a modification is virtually ignored. Thus, for example, the limited assimilative capacity of some ocean waters of the East Coast is a matter of major concern, while similarly limited air and land capacity are more or less overlooked.

In short, federal controls are far less integrated than they should be and could be; a more systematic approach is both necessary and possible. This is not to say that trade-offs must be made with fine-tuned precision -- more rough-and-ready calculations would be a satisfactory beginning. Nor should our comments suggest an instant need for a completely integrated approach to environmental management. Rather we recommend a more modest program, limited to ocean disposal of municipal waste and sensitive simply to the most obvious trade-offs among air, land, and water as disposal sites. Given a relatively small number of sources and a relatively good understanding of the options and constraints, a more integrated approach to the municipal waste/ocean disposal problem would appear to be manageable.

Approaching Policy on a Regional Basis

If the interrelated effects -- costs and benefits -- of alternative management strategies are to be considered in formulating controls on ocean discharge of municipal waste, then this means almost necessarily that policy must be developed on a regional not a national basis. Land, air, and water resources differ significantly from place to place, and controls that make sense for one area will not suit another. Ocean waters are regionally variable, for example -- some provide greater assimilative capacity than others, some have more sensitive ecosystems than others, some are subjected to more effluent input than others.

Variations like these should be taken into account; standards and controls should be regional. Section 301(h) adopts such an approach in part by taking careful account of differences among ocean waters. As mentioned above, however, the modification provisions tend to ignore real differences among air and land resources; the tendency is a product of the unsystematic approach of present environmental controls. In other words, the need for the ocean disposal option (as for sludge) depends heavily on the impacts of not using the ocean, which may vary considerably among regions.

Developing policy on a regional basis implies not only different controls for different areas, but different timetables as well. For example, a phase-out of ocean dumping by a specified date may well prove to be appropriate for one set of cases in one region, but a different set may need a different schedule.

Present environmental policy in general displays undue amounts of national uniformity, and the job of revamping existing controls in order to take account of regional variability will be a big one. But that is not the concern here. Ocean disposal of municipal waste presents a discrete set of problems and can be dealt with accordingly, without having to address larger reforms.

Adjusting Policy in Response to New Information

Another important aspect of ocean discharge policy should be flexibility to respond to new information of all kinds: research on transport, fates, and effects of pollutants in coastal waters; monitoring results near outfalls; technological advances (in all aspects of source control, treatment plants, and outfall construction); impacts on other media related to ocean discharge policies; changing costs (both capital and operation and maintenance). Many present requirements are so rigidly prescribed (in law or regulations) that changes may take years to get through the "system." Individual POTWs can respond with more innovation and pollution control for the money if regulatory agencies have enough flexibility to adjust standards on a timely case-by-case basis. (See also discussion of management standards above.)

Sewerage agencies should also be encouraged and supported to undertake research and demonstration projects to improve overall environmental management.

A good example is the use of deepwater special outfalls for sludge disposal off southern California as suggested by Jackson et al. (1979) and Brooks et al. (1982). However, current policy prohibits such a project, probably even as a research and development activity. With good source control of contaminants such a project would not be risky on a trial basis to observe the impacts (see section on reversibility of effects).

Without flexibility, we deny ourselves the opportunity to profit by our experience -- both good and bad.

Summary

In this section we have discussed four general elements for a good policy for ocean discharge:

1. Costs of required pollution control measures should be balanced against the expected benefits (including reduction of environmental risks).

2. Ocean discharge policies should be integrated with policies for pollution control in other media.

3. Policies should not be uniform nationally, but should be developed on a regional (or case-by-case) basis to take account of the wide variations in the nature of the coastal waters.

4. Policies should be adjusted in response to new scientific and technological information.

The present policies controlling ocean discharge of effluent and digested sludge fall short of fulfilling any of these four.

SUMMARY AND CONCLUSIONS

Much has been learned from our experiences with disposal of municipal sewage effluents and sludges in the ocean, as seen from the various chapters in this book. Ocean discharge policies can be improved to make the best use of this scientific and technological knowledge, as discussed in this chapter. In brief, the main conclusions are as follows:

1. The coastal ocean has significant capacity for assimilation of municipal wastewater (both treatment plant effluents and digested sludge).

2. The effects of an ocean discharge vary widely depending on the design of the outfall, the quality of the effluent, and the characteristics of the receiving waters.

3. For sewage effluent, the effects and risks (public health, ecological and aesthetic) may be controlled to reasonably low limits by a suitable combination of pretreatment or source control measures (for industrial discharges to the sewers), sewage treatment processes, and outfalls with diffusion structures.

4. Present law does not allow enough flexibility to apply the most cost-effective approaches to pollution control on a case-by-case or regional basis. However, a recent reversal of this trend is Sec. 301(h) of the Federal Water Pollution Control Act Amendments of 1977, which allows applications for exceptions from mandatory secondary treatment.

5. Present policy does not provide for integrated regulation of oceans, fresh waters, land, and air, in spite of the fact that stricter ocean standards generally lead to increased impacts on other media.

6. The prohibition of sludge dumping in the ocean is a policy which is not based on scientific, engineering, and economic evaluations of trade-offs, considering alternative disposal methods impacting the land, fresh waters, and/or the atmosphere.

7. Specific technical issues affecting management are discussed in this Chapter under the heading, "Key Technical Issues Affecting Management Alternatives." Topics included are:

- basis for regional differentiation of policy
- particles
- effluent vs sludge
- fates and effects of dissolved pollutants
- source control of toxic substances
- disinfection of effluents
- predictability of outfall effects
- reversibility of effects

8. In general, the highest priority is for better control over the releases of trace toxic chemicals and heavy metals to the ocean (as well as to other environmental media). The only viable strategy is control at the industrial sources to prevent

release into the sewers. Improvement is needed in the procedures for setting standards for trace contaminants with potential long-term chronic effects (i.e., how stringent do they need to be to balance environmental risk against costs of control).

9. Sewage treatment processes required for ocean discharge should be more specifically directed at ocean water quality problems. When effluents are well diluted by well-situated, high-dilution outfalls, there is little value in removal of BOD by secondary treatment (or decrease in nutrients by tertiary treatment). However, improved removal of particles by advanced primary sedimentation is directly beneficial to ocean water quality in many cases, and is much cheaper than full secondary treatment.

10. Outfall technology has advanced considerably in recent decades with the use of longer outfalls with multiport diffusers. The initial dilutions and subsequent dispersion of the plumes can be predicted with good reliability. Long, high-dilution outfalls reduce the need for secondary treatment and disinfection, and may be economically attractive alternatives to secondary treatment when the ocean conditions are favorable.

11. Sludge disposal methods in the ocean can be improved. Instead of being banned outright, the ocean sludge disposal option should be kept open for analysis and evaluation on scientific, technical and economic grounds just as for other options. In some regions where the ocean has good assimilative characteristics, but air and land resources are already overstressed, the ocean disposal option may be the most desirable (such a region may be the southern California coast).

12. Research and demonstration projects should be encouraged to gain more information on effective techniques for management of ocean disposal. An example would be the deep-water (300 m) disposal of sewage sludge off Orange County for a trial period of five years to observe transport, fates, and effects of the sludge particles, and to improve the methodology for analysis of such discharge in general (Brooks et al., 1982).

13. Monitoring programs for ocean water near outfalls should be designed for early detection of any new problems as well as for routine enforcement of regulations. Because of the high cost of many monitoring programs, more research is needed on optimal monitoring strategies.

14. The basic elements of good policy for ocean discharge are:

- costs of control which are commensurate with environmental benefits
- integration with policies for other media
- flexibility to account for wide variations in the nature of coastal waters
- flexibility to adjust control programs on an incremental basis in response to environmental monitoring results or new scientific information.

NOTE

As was stated for Chapter 11, there have been changes in federal policy between the time this chapter was written (early 1981) and final publication; and some issues are being actively considered at press time. In 1981, further amendments of the Clear Water Act extended the eligibility of dischargers to apply for 301(h) waivers; in 1981, a decision of the US District Court in New York City vs EPA essentially stopped EPA from enforcing its total ban on ocean dumping of sludge under the Ocean Dumping Act; and several Congressional committees and EPA are studying the sewage sludge disposal issue in the context of a more integrated approach considering all media. In addition, more water quality criteria for marine waters have been issued by EPA. The National Advisory Committee on Oceans and Atmosphere ("The Role of the Ocean in a Waste Management Strategy," 1981, US Govt. Printing Office) has also critically discussed various aspects of federal policy along lines similar to this chapter.

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Appendix A

Characteristics and Effects of Municipal Wastewater Discharges to the Southern California Bight, A Case Study

Alan J. Mearns
David R. Young

SUMMARY AND CONCLUSIONS

At present, municipal wastewater discharges are the dominant source of most constituents of concern to the Southern California Bight. Effluents from these discharges are effectively diluted so that dissolved concentrations of trace materials, such as metals and chlorinated hydrocarbons, are well below levels known to cause toxic responses in marine organisms. However, because most constituents are bound to particulates, suspended matter and sediments near the outfall diffusers contain above normal concentrations of trace contaminants and organic matter.

The most serious known pollutants are the higher molecular weight chlorinated hydrocarbons, namely DDTs and PCBs. These synthetic organics have been shown to be biomagnified through the coastal marine food web leading to birds and mammals.

In contrast, trace metals do accumulate in some benthic and intertidal invertebrates, particularly molluscs, but do not undergo further food web biomagnification. Although mercury naturally undergoes bioamplification in marine food webs, available evidence indicates that there is no additional mercury biomagnification due to these municipal discharges. Preliminary data indicate that other volatile and extractable chlorinated hydrocarbons, while present in relatively high concentration in sewage, do not appear to bioaccumulate through nearby marine food webs. Finally, low concentrations of benzo(a) pyrene in coastal mussels indicate that this carcinogen is not contaminating nearshore biota as a result of municipal wastewater discharges.

Thus, the higher molecular weight chlorinated hydrocarbons, mainly DDTs and PCBs appear to be the

most serious pollutants with respect to food web amplification and seafood contamination.

The response of marine communities to the municipal wastewater discharges is strongest and most obvious in benthic macroinvertebrate populations, and weakest and least obvious in plankton, fish, and epibenthic invertebrate populations. The exception is at Palos Verdes, where the diversity of shallow rocky subtidal kelp bed communities has been damaged by offshore discharge. This site has partially recovered possibly as a result of reduced solids and DDT emissions.

At all major sites, the main benthos response is in a shift from communities dominated by suspension feeding infauna to those dominated by an abnormally high biomass of deposit-feeding invertebrates. The size of bottom area affected by this change is directly proportional to mass emission rates of suspended solids. If future emission rates are known, this relationship can be used to forecast changes in the size of bottom area affected by changed communities.

Some fish and larger macroinvertebrates are more abundant and others less abundant near all discharge sites. For the smaller discharges (e.g., Point Loma and Orange County) more species are increased than are decreased. At Palos Verdes, more species are decreased than are increased.

Fin erosion is the only disease conclusively linked to wastewater discharges, and it is initiated at only two of six outfall sites studied. The precise cause of the disease is not known, but high sediment and tissue concentrations of PCBs and contact with highly contaminated bottom sediments, such as occur at Palos Verdes, appear to be required.

Although we did not specifically review in this report the effects of southern California municipal discharges on dissolved oxygen or pH, numerous monitoring measurements indicate that these parameters are not significantly depressed at most sites; however, at Palos Verdes, DO depression approaching 10 percent below background has occasionally been reported.

Within the next five years, most of the treatment plants cited in this report will convert to partial secondary treatment. Thus, sources other than municipal waste discharges will soon dominate some

chemical inputs. Research and monitoring programs should be reviewed and carefully adjusted to test forecasts and maximize our understanding of ecological recovery rates and processes.

INTRODUCTION

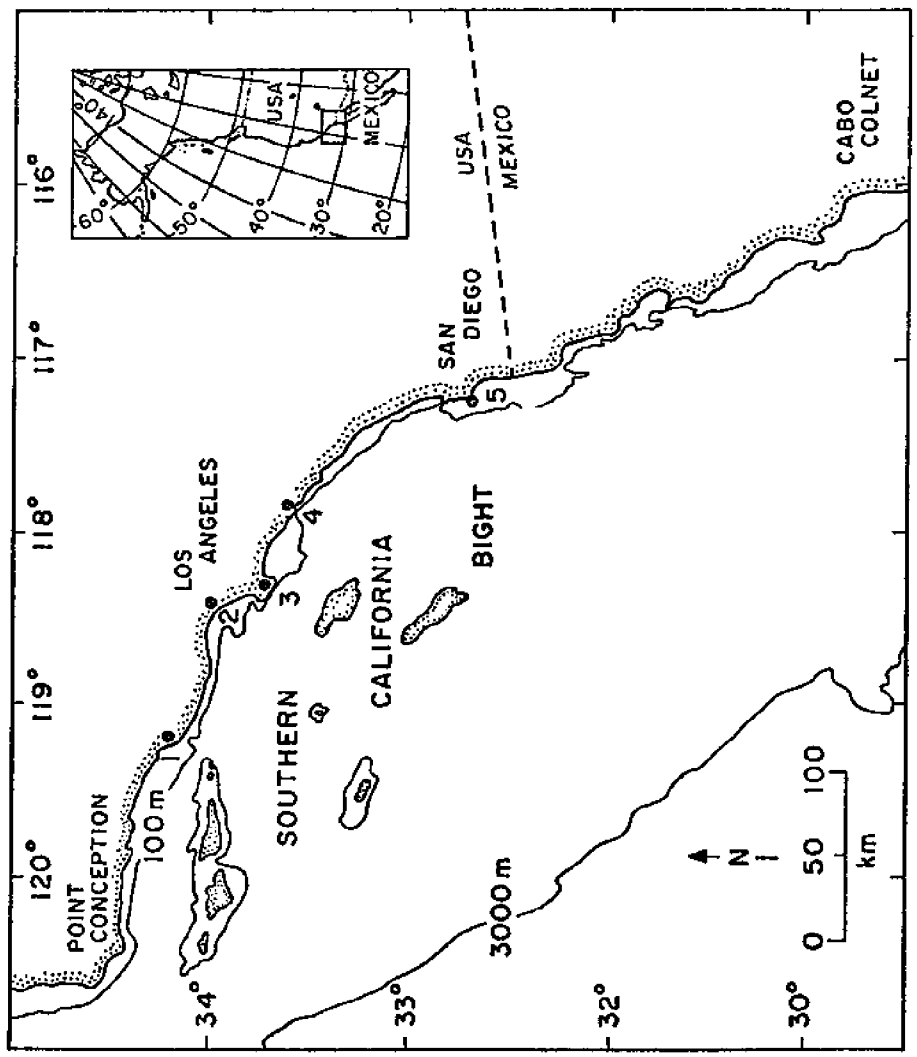
In this case study we summarize major findings obtained since 1971 by the Southern California Coastal Research Project (SCCWRP) and others on the fate and effects of municipal wastewater discharges to the Southern California Bight. The investigations conducted to date off the major deep ocean outfalls of the region have rather convincingly separated certain wastes of "clear concern" from others of "apparent non-concern" in well-flushed areas. However, a number of important questions remain to be answered, as described in detail in the following sections.

GEOGRAPHY, CIRCULATION, AND LAND USE

The Southern California Bight (Figure A.1) is the open embayment of the Pacific Ocean extending from Point Conception on the north to Cabo Colnett, Baja California, on the south. The California Current roughly defines the Bight on the west. As the Bight's current system usually extends westward to the area with water depth in the order of 3,000 m, a more precise western boundary is the Continental Slope off southern California and northern Baja California. Within these boundaries, the Bight is approximately 500 km long and 200 km wide, covering a surface area of 100,000 km^2.

The California Current generally follows the coastline southward until it reaches the northern limit of the Southern California Bight -- Point Conception. Here, the coastline turns abruptly eastward, and the flow of water departs from the coast, generally continuing in a southeastward direction. Further south, off the coast of northern Baja California, the main portion of the Current turns toward the land, and the flow divides into two branches. One branch, known as the Southern California Countercurrent, turns northward and flows through the Channel Islands, forming the inshore side of the Southern California Eddy. During the spring, however, the Countercurrent weakens and may even be absent. During such times, the surface flow within the Bight is to the south (Jones, 1971).

Figure A.1 The Southern California Bight and the adjacent coastal basin. Major municipal wastewater dischargers are (1) City of Oxnard (2) City of Los Angeles (Hyperion Treatment Plant) in Santa Monica (3) Joint Water Pollution Control Plant (JWPCP) off Palos Verdes (4) Orange County Sanitation Districts in southern San Pedro Bay, and (5) City of San Diego, Point Loma Treatment Plant. Note 100-m isobath occurs within 5-10 km from shore.



Hendricks (1977a) has shown that nearshore surface currents in the Bight have speeds on the order of 25 cm/s, and generally flow downcoast. In contrast, the subthermocline currents (which most affect submerged wastewater plumes), although highly variable, appear to move upcoast with a mean speed of about 3 cm/s. If this mean flow (which was measured off San Diego) continues along the entire coast between the US-Mexico border and Point Conception, it would correspond to a transit time of about 150 days.

Waters within the Bight are characterized by a warm (monthly mean temperatures between 12° and 20°C), well-oxygenated mixed layer (20 to 30 m deep most of the year) and cooler, less-oxygenated subthermocline waters (9° to 12°C and 4 to 6 milligrams per liter (mg/l) DO between depths of 50 and 200 m). The onset of strong, steady northerly winds blowing in spring moves the warmer surface water to the west and permits colder (denser) water to upwell near the coast. This water brings up nutrients which support heavy plankton blooms and cause the area to be one of high organic production.

The coastal plain adjacent to the Bight has an area of about 33,000 km^2 and is inhabited by approximately 12 million persons, or about 5 percent of the nation's population. Land use characteristics for 1970 are summarized below (SCCWRP, 1973):

Public	41.6 percent
Agriculture	26.6 percent
Residential	22.9 percent
Industrial	4.9 percent
Commercial	4.0 percent

Most of this population is concentrated in the Los Angeles-Orange County Basin (roughly bounded on the coast by Santa Monica and Newport Beach), with a second major urban-suburban area in San Diego County (Figure A.1). The resultant municipal wastewater (1.5×10^{12} liters per year (l/yr)) generally undergoes primary treatment only and then is discharged to the coastal waters through submarine outfall systems, as described in greater detail below.

Surface runoff flows to the Bight from 18 hydrologic units (SCCWRP, 1973) via more than 150 streams (excluding urban storm drains). However, most of this discharge is carried by approximately 20 major channels; the long-term near annual flow is 0.56 x 10^{12} l/yr, or less than half of the municipal wastewater inputs (SCCWRP, 1973). Most of this runoff occurs as pulse inputs carried by only a few storms per year.

This Los Angeles-Orange County coast also contains the major vessel activity in the Bight. In addition to the commercial and naval vessels served by Los Angeles-Long Beach Harbor, the largest southern California anchorage, approximately 70 percent of the 35,000 recreational vessels in the Bight are sheltered in marinas of this region. A second major harbor is located in San Diego. (Most of the direct industrial discharges are situated in these two harbors.) We concluded from a 1973 Bight-wide survey that approximately 300,000 liters of antifouling paints, containing 180 tonnes of copper, are applied annually to vessel bottoms in southern California (Young et al., 1979a).

Thermal effluents from electrical generating stations spread along the coast represent another potential source of contamination. The 1976 discharge rate for the eight stations of Southern California Edison company, situated between Ventura and San Diego Counties, was 6.8 x 10^{12} l/yr roughly 70 percent of the total cooling water discharged to the Bight (Young et al., 1977a). Finally, the severe atmospheric pollution which exists in the Los Angeles-Orange County and San Diego Basins constitutes another route of pollutant inputs to the adjacent marine ecosystem.

MUNICIPAL WASTEWATER TREATMENT AND DISCHARGE SYSTEMS

The five largest southern California discharges account for over 95 percent of the 1.0 billion gallons of municipal sewage discharged daily (1.4 x 10^{12} l/yr) into the Southern California Bight. The wastes are discharged through outfalls which range in depth from 20 to 100 m and terminate 2 to 11 km from shore. The largest are those of Los Angeles County (JWPCP) and the City of Los Angeles (Hyperion Treatment Plant), which each discharge approximately 330 mgd. Approximately one-third of the Hyperion flow (100 mgd) is given secondary treatment, biological oxidation and sludge removal, but the resulting sludge is discharged via a second outfall located 11 km from shore and 3 to

4 km north of the effluent outfall. The Orange County outfall (160 mgd primary and 20 mgd secondary) discharges at a depth of approximately 60 m and about 8 km from shore in southern San Pedro Bay. The City of San Digeo's Point Loma plant discharges approximately 120 mgd of primary treated effluent through a 3 km-long outfall located at a depth of 60 m. The next largest outfall, from the City of Oxnard, discharges about 11 mgd of primary treated effluent through a 2 km-long outfall at a depth of about 20 m. Finally, there are approximately 20 other outfalls in southern California which discharge a total of about 40 mgd through outfalls ranging in depth from 0 to 30 m. Most of the small plants chlorinate regularly, the JWPCP plant chlorinates during periods of shoreward transport, and the other three major plants rarely chlorinate.

The ocean outfalls through which the wastes are discharged are fitted with long (1000 to 2000 m) multiport diffusers which achieve initial dilutions of at least 100:1 (Hendricks, 1977b; Young and Jan, 1975). During most of the year, when a thermocline is present, plumes remain submerged, reaching an equilibrium depth just below the thermocline.

Following initial dilution, the southern California sewage plumes tend to move along-shore and up-coast (to the northwest), along the isobath of discharge (generally 60 m) and usually opposite to the surface currents. During transit, components of the diluted effluent begin to separate; about 10 to 20 percent of the heavier particulates fall out of the plumes and onto the bottom near the discharge (Herring and Abati, 1979). Thus, bottom sediments in the fallout area around the discharges contain above-normal concentrations of trace metals, chlorinated hydrocarbons, and other contaminants attached to wastewater particulates. The fate of the remaining mass of fine particulates is largely unknown, but a portion of it appears to remain suspended in the discharge area (Peterson, 1974) and perhaps many kilometers beyond. The fate of dissolved constituents is largely unknown, but ammonia is rapidly taken up by phytoplankton (Hendricks and Harding, 1974).

CHARACTERISTICS OF MUNICIPAL WASTEWATERS

Wastewater monitoring programs have been conducted for many years at the five major municipal treatment plants situated along the southern California coast.

With the adoption of sensitive analytical procedures for trace analysis (such as atomic absorption spectroscopy and electron-capture gas chromatography) around 1970, reasonably reliable data on elemental and chlorinated hydrocarbon concentrations in final effluents began to be available. Tables A.1 and A.2 present 1977 average concentrations and associated mass emission rates for wastewater constituents of concern produced by these monitoring programs.* It is important to note that most of the target trace metals and higher molecular weight chlorinated hydrocarbons (such as the DDTs and PCB 1254) have been found to be associated with the filterable (<0.4μ) particulates in municipal wastewaters (Young et al., 1973a; Luthy, 1972). This characteristic appears to be of major importance regarding the processes that control the dispersion and ultimate fate of such contaminants.

COMPARISON OF INPUTS

In 1971 to 1972, SCCWRP surveyed inputs of various chemicals to the Bight via municipal wastewater, surface runoff, and direct industrial discharges (SCCWRP, 1973). The result for "general" constituents of interest are summarized in Table A.3. Subsequently, several additional studies of trace metal and chlorinated hydrocarbon inputs were conducted (Tables A.4 and A.5). These data indicate that, for most substances investigated, inputs via municipal wastewater discharge far exceeded those via the other routes. As seen in Table A.2, the JWPCP discharge off Palos Verdes Peninsula has been the largest single point source input for most contaminants of interest.

RESULTANT DISTRIBUTIONS

Seawater

Limited studies around the largest outfall system (JWPCP) indicated that concentrations of dissolved metals (<0.4 μ) in the wastewater plume were no more than a few times baseline levels (Young and Jan, 1975). For example, there were no significant elevations of dissolved cadmium or chromium values above median control concentrations (0.05 and 0.2 μg/l respectively). In addition, median values of dissolved copper and nickel at three plume stations ranged from 0.3 to 0.6 and 0.7 to 1.2 μg/l, respectively, compared with corresponding control values of 0.1 and 0.3 μg/l. In contrast, the

* Data available at the time of publication indicate many constituents except flow have been significantly reduced. See Schafer, H.A., 1982. pp. 11-15.

Table A.1 Average Concentrations of General Constituents, Trace Metals, and Chlorinated Hydrocarbons in the Final Effluent of Municipal Waste Dischargers, 1977 (Schafer, 1979)*

	JWPCP	Hyperion 5-mile	Hyperion 7-mile	Orange County	Point Loma	Oxnard
Flow						
mgd	335	319	4.6	180	116	11.2
liters/day x 10^6	1,268	1,207	17.4	681	439	42.39
General constituents (mg/liter)						
Total suspended solids	220	62	8,100[a]	132	128	98.0
5-day BOD	220	145	ND	197	167	258
Oil and grease	46.1	19	584	42	33.4	23.3
Ammonia nitrogen	39.0	17.2	259	36	23.8	20.5
Total phosphate-P	13.0	7.6	189	ND	ND	ND
Detergent (MBAS)	6.3	4.2	ND	ND	4.0	ND
Cyanide (CN)	0.24	0.14	0.70	0.11	0.05	0.03
Phenols	3.3	0.06	0.56	0.12	0.19	0.15
Trace metals (mg/liter)						
Silver	0.008	0.03	1.89	0.01	0.017	0.009
Arsenic	0.009	0.01	0.20	0.06	0.013	0.006
Cadmium	0.025	0.02	1.30	0.04	0.017	0.009
Chromium	0.38	0.13	12.8	0.18	0.055	0.025
Copper	0.25	0.20	15.5	0.32	0.125	0.125
Mercury	0.001	0.021	0.132	0.0005	0.001	0.0015
Nickel	0.24	0.18	4.1	0.13	0.08	0.210
Lead	0.19	0.03	2.15	0.09	0.08	0.050
Selenium	0.016	0.01	1.68	ND	ND	ND
Zinc	0.84	0.32	28.0	0.40	0.17	0.18
Chlorinated hydrocarbons (µg/liter)						
Discharger values[b]						
Total DDT	1.58	0.20	3.65	0.05	ND	ND
Total PCB	1.81	2.13	35.4	0.73	ND	ND
Total identifiable chlorinated hydrocarbons					2.3	
Project values[c]						
Total DDT	1.58	0.02	1.1	0.02	0.13	0.05
Total PCB	0.68	0.15	16	4.0	0.52	0.05

[a] Total solids, suspended and dissolved.

[b] Based on analysis of one grab sample per month, except for JWPCP total DDT value, which was based on analysis of 52 weekly composite sample.

[c] Based on analysis of two 1-week composite samples.

(ND means "no data")

* Data available at the time of publication indicate many constituents except flow have been significantly reduced. See Schafer, 1982.

Table A.2 Mass Emission Rates of General Constituents, Trace Metals, and Chlorinated Hydrocarbons in Final Effluent of Municipal Wastewater Dischargers, 1977 (Schafer, 1979)*

	JWPCP	Hyperion 5-mile	Hyperion 7-mile	Orange County	Point Loma	Oxnard
Flow						
gallons/yr x 10^9	335	319	4.6	180	116	11.2
liters/yr x 10^9	462.8	441	6.36	247	160	15.47
General constituents (metric tons/year)						
Total SS	102,000	27,300	51,500[a]	33,300	20,500	1,500
5-day BOD	102,000	63,900	ND	47,900	26,700	3,900
Oil and grease	21,300	8,380	3,700	10,100	5,340	360
Ammonia nitrogen	18,000	7,590	1,650	9,830	3,810	317
Total phosphate-P	5,500	3,350	1,200	ND	ND	ND
Detergent (MBAS)	2,900	1,850	ND	ND	640	ND
Cyanide (CN)	111	61.7	4.4	29	8	0.5
Phenols	1,530	26.5	3.56	24.7	31.5	2.40
Trace metals (metric tons/yr)						
Silver	3.70	13.25	12.0	2.47	2.7	0.14
Arsenic	4.16	4.41	1.27	ND	2.1	0.09
Cadmium	11.6	8.82	8.27	10.9	2.7	0.14
Chromium	176	57.3	81.4	42.0	8.8	0.39
Copper	116	88	98.6	81.5	20	1.95
Mercury	0.462	0.926	0.84	0.37	0.16	0.023
Nickel	111	79.4	26.1	32.1	12.8	3.25
Lead	87.8	13.2	13.7	23.4	12.8	0.77
Selenium	7.39	4.4	10.7	ND	ND	ND
Zinc	388	141	178	101	27	1.7
Chlorinated hydrocarbons (kg/yr)						
Discharger values[b]						
Total DDT	730	88	23	14.1		ND
Total PCB	838	939	226	180		ND
Total identifiable chlorinated hydrocarbons					364	
Project values[c]						
Total DDT	730	8.8	7	6	21	0.8
Total PCB	314	67	108	986	83	0.8

[a] Total solids, suspended and dissolved.

[b] Based on analysis of one grab sample per month, except for JWPCP total DDT value, which was based on analysis of 52 weekly composite sample.

[c] Based on analysis of two 1-week composite samples.

ND = "no data"
SS = "suspended solids"
* Data available at the time of publication indicate many constituents except flow have been significantly reduced. See Schafer, 1982.

Table A.3 Estimated Fluvial Inputs (metric tons/year) of General Wastewater Constituents to the Bight, 1971-2 (SCCWRP, 1973)

General Constituent	Municipal Wastewater	Industrial Discharge	Surface Runoff*
Flow (10^{12} l/yr)	1.38	0.25	0.24
Total Susp. Solids	278,000	16,000	274,000
Volatile Susp. Solids	179,000	---	---
BOD (5-day)	291,000	6,000	---
COD	675,000	78,000	29,000
Oil and Grease (HEM)	65,000	2,200	4,400
Dissolved Silica (S_iO_2)	33,000	---	2,800
Nitrate Nitrogen	530	---	980
Ammonia Nitrogen	59,400	9,500	440
Organic Nitrogen	24,800	---	1,090
Total Nitrogen	84,500	10,000	2,510
Phosphate Phosphorus	13,300	---	410
Detergent (MBAS)	7,600	---	66
Cyanide	210	---	11
Phenols	1,730	43	269

*Includes both storm and dry-weather flows.

Table A.4 Estimated Annual Inputs (metric tons/year) of Trace Metals to the Bight[d] (From Young et al., 1973; 1978a)

	Municipal wastewater 1976	Dry fallout[a] 1975	Storm runoff 1971-72	Storm runoff 1972-73[b]	Thermal discharge[c] 1977
Ag	20	0.06	1.1	2.6	---
Cd	45	0.84	1.2	2.8	0.3
Cr	593	6.6	25	60	0.6
Cu	507	31	18	42	2.1
Hg	2.6	---	---	0.43	---
Ni	307	12	17	41	0.7
Pb	190	240	90	210	0.8
Zn	1,060	150	101	240	1.8

[a] 100 km x 100 km off L.A. - Orange Co. Basin

[b] Extrapolating flow-weighted results from 1971-72 storm survey (abnormally dry year)

[c] Southern California Edison cooling water (6.8×10^{12} l/yr)

[d] Approx. 180 tonnes/yr Cu used in vessel antifouling paints.

Table A.5 Estimated Annual Inputs (kg/yr) of Chlorinated Hydrocarbons to the Bight (Young and Heesen, 1978; Young, unpubl.)

Route	Year	Total DDT	Dieldrin	1242 PCB	1254 PCB
Muni. wastewat.[a]	1972	6,490	100	≥19,200	≥260
Muni. wastewat.	1973	3,920	≤280	≥ 1,900	1,510
Muni. wastewat.	1974	1,580	95	4,270	1,020
Muni. wastewat.	1975	1,270	---	2,400	680
Muni. wastewat.	1976	940	---	2,220	590
Muni. wastewat.	1977	770	---	1,260	300
Harbor indust.	1973-74	40	10	≤70	30
Antifoul. Paint	1973	<1	---	<1	<1
Surf. Runoff	1971-72	100	20	100-170	90-110
Surf. Runoff	1972-73	320	65	0-550	250-280
Aerial Fallout[b]	1973-74	1,400	---	---	1,100
Ocean Currents	1973	≤7,000	---	---	≤4,000

[a] 1971 JWPCP discharge of total DDT: 21,600 kg

[b] Inner coastal zone: 400 km by 50 km.

particulate fractions (>0.4μ) of these metals were 1 to 2 orders of magnitude above control levels (0.1 to 0.3 μg/l), again suggesting the importance of wastewater particulates regarding the dispersion and fate of trace contaminants following discharge to the marine environment.

Although not separated into dissolved and particulate phases, DDT and PCB residues also have been measured in seawater collected in the JWPCP discharge zone. In May 1973, De Lappe and Risebrough measured concentrations of p, p'-DDE and trichlorobiphenyls up to 26 and 27 ng/l, respectively, in seawater collected from the immediate vicinity of the outfall. In comparison, later that year they obtained concentrations that were lower by 1 to 2 orders of magnitude in seawater collected at the edge of the Bight (Risebrough et al., 1976).

Sediments

To obtain a comprehensive picture of the extent of sediment contamination caused by five major outfall systems, in 1977 SCCWRP conducted a systematic sediment survey along the 60 m contour, the approximate depth of discharge (Word and Mearns, 1979a; Word and Mearns, 1979b). Using a modified 0.1 m^2 Van Veen grab, samples of bottom sediment were collected every 10 km between Point Conception and the US-Mexico border (Figure A.1). The upper layer (0 to 5 cm) was analyzed for numerous wastewater constituents; examples of the resulting distributions are presented in Figure A.2. In an effort to summarize this extensive body of data for the four largest outfall systems, we have selected stations within the clear influence of the wastewater discharge; corresponding median and range values for various parameters were then calculated for each outfall zone, as well as for a large suite of control stations. The results are listed in Table A.6.

These data indicate that, at least along the 60 m contour, the JWPCP discharge off Palos Verdes Peninsula has had by far the greatest impact on sediment concentrations of oxygen-demanding substances, toxic trace metals, and higher molecular weight chlorinated hydrocarbons. However, comparable levels of most of these contaminants also occur in somewhat deeper water around the terminus of the Hyperion 7-mile outfall, which discharges diluted sludge at about 100 m in Santa Monica Bay (Schafer and Bascom, 1976). Detailed summaries of sediment

Figure A.2 Variations in concentrations of six materials in surface sediments from 77 stations along the 60m isobath, spring and summer 1977. Large peak is centered about the Palos Verdes discharge area. Some secondary peaks in 5-day BOD, cadmium, zinc, and total PCB are centered about the other major discharges but there are also secondary peaks elsewhere. Major DDT source is from Palos Verdes outfalls. (Word and Mearns, 1979b)

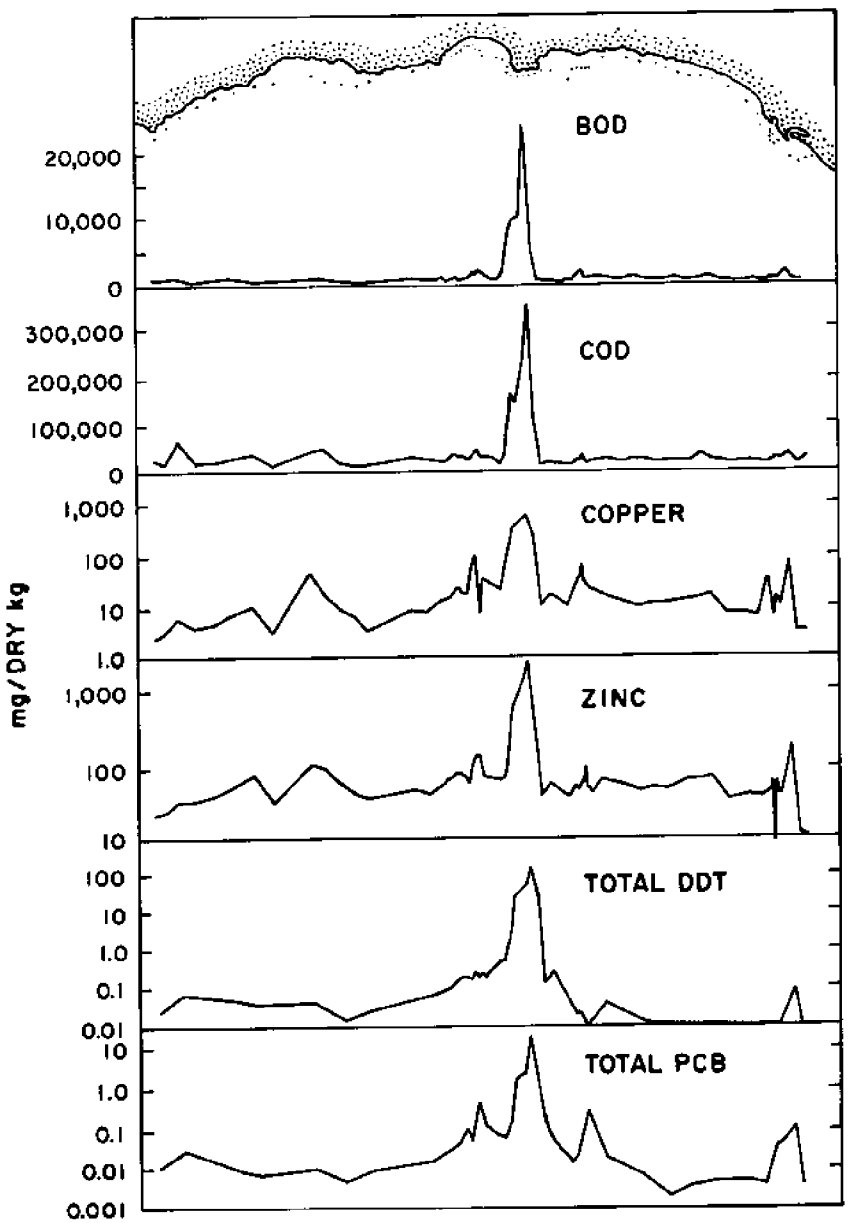


Table A.6 Concentrations (medians and ranges) of Selected General Constituents, Trace Metals, and Chlorinated Hydrocarbons in Sediments from 29 "Control" Sites and Four Outfall Sites on the Southern California Mainland Shelf. All Data from the 60-m Isobath Sampled between Point Conception and the US-Mexico Border, 1977 (Word and Mearns, 1979, except as noted). Concentrations based on dry weight.

General Constituents	Control Sites 29 Stations m̄	Control Sites 29 Stations Range	Point Loma 10 Stations m̄	Point Loma 10 Stations Range	Orange County 10 Stations m̄	Orange County 10 Stations Range	Santa Monica Bay 9 Stations m̄	Santa Monica Bay 9 Stations Range	Palos Verdes 7 Stations m̄	Palos Verdes 7 Stations Range
5-day BOD -25,050 (mg/kg)	636	266 -1020	754	407 -2180	927	499 -1930	817	486 -1850	7780	943
Volatile solids (%)	2.8	1.8-3.8	2.8	1.9-4.7	2.5	1.9-3.0	3.2	2.4-3.9	17	5.0-27
Acid volatile sulfides (mg/kg)	0.01	<0.007-0.07	0.03	0.005-0.32	0.04	0.01-0.11	0.04	0.01-0.20	0.67	0.03-1.6
Trace Metals (mg/kg)										
Chromium	22	6.5-43	23	4.4-28	37	33 -63	74	54 -146	680	86 -1320
Copper	8.3	2.8-31	10	2.8-62	27	18 -56	22	17 -192	289	33 -782
Lead	6.1	2.7-12	6.2	3.1-43	22	15 -36	16	11 -40	162	25 -537
Zinc	42	9.8-62	41	9.8-155	47	39 -88	68	58 -143	526	57 -2100
Chlorinated Hydrocarbons (mg/kg)										
Phenols[a]	0.007			ND	0.076	0.015-0.440		ND		ND
Chlorinated Benzenes[b]	ND			ND		ND		ND		0.36
Total DDTs	0.007	<0.001-0.09	0.001	0.001-0.08	0.018	0.003-0.019	0.17	0.15-0.50	25	1.3-180
Total PCBs	<0.002	<0.002-0.04	0.037	0.004-0.14	0.063	0.020-0.25	0.12	0.060-0.51	2.1	0.11-11

[a] Data taken by Orange County Sanitation Districts, October 1974-June 1975, pers. comm.
[b] One station near outfall diffusers; from Young and Heesen, 1978.
ND = No Data

contamination around the major outfalls, and in control regions of the Bight, have been presented elsewhere (Galloway, 1972; SCCWRP, 1973; Young et al., 1973b, 1975a, 1976a, 1976b, 1977b; Bruland et al., 1974; Hom et al., 1974; Eganhouse et al., 1976; Hershelman et al., 1977; Young et Heesen, 1978).

Biota

Elevated tissue concentrations of trace metals and higher molecular weight chlorinated hydrocarbons have been observed in a variety of marine organisms collected around (or inshore of) the major submarine outfall systems in southern California. The most dramatic example is the widespread contamination of the biota by DDT residues, caused by the massive discharge of this pesticide from the JWPCP outfalls up until the early 1970s (Anderson, 1973). Resultant levels of total DDT (mostly p, p'-DDE) in intertidal sand crabs (Emerita analoga) and mussels (Mytilus californianus) from Palos Verdes Peninsula were 2 to 3 orders of magnitude above baseline values at the edge of the Bight (Burnett, 1971; Young and Heesen, 1974; Risebrough et al., 1976; Young et al., 1978a). Corresponding gradients (or contamination levels) have been reported for midwater fishes (Risebrough, 1969; Duke and Wilson, 1971; MacGregor, 1974; Young et al., 1975b), benthic fish and crabs (Young et al., 1976a, 1977b), and marine birds (Anderson et al., 1975; Risebrough et al., 1976) and mammals (DeLong et al., 1973).

The severe eggshell thinning and resultant reproductive decline of Brown pelicans (Pelicanus occidentalis californicus) in southern California in the early 1970s has been attributed to this DDE pollution (Risebrough et al., 1976). Young et al. (1979b) also have attributed the deaths of marine birds in the Los Angeles Zoo to high DDT residues (approximately 4 milligrams per wet kilogram (mg/wet kg)) in queenfish (Seriphus politus) caught off Los Angeles and used as the exclusive diet of the birds affected. Muscle tissue concentrations of total DDT in black perch (Embiotoca jacksoni -- a popular sportfish) caught off Palos Verdes Peninsula in 1973 ranged up to 97 mg/wet kg (Young et al., 1975b), and approximately 30 percent of the seafood species collected from this discharge zone during 1975 to 1977 exceeded the US Food and Drug Administration limit of 5 mg/wet kg (Young et al., unpub.).

Residues of the industrially important PCBs are another important class of trace synthetic organics contaminating the biota in the Bight. McDermott et al. (1978) reported median liver tissue concentrations of total PCB for Dover sole (Microstomus pacificus) collected during 1975 from the JWPCP and OCSD discharge zones of 15 and 8 mg/wet kg, respectively; corresponding muscle tissue concentrations were 0.7 and 1.1 mg/wet kg. The highest mean value obtained in the 1975 to 1977 JWPCP seafood survey (about 2 mg/wet kg) was measured in the muscle of white croaker (Genyonemus lineatus), a water column and benthic feeder whose mean muscle tissue concentration of total DDT was 39 mg/wet kg (Young et al., unpubl.).

Extensive time-series studies off the JWPCP outfall system have indicated that contamination of bottom sediments by the higher molecular-weight chlorinated hydrocarbons such as DDTs and PCBs can result in persistent contamination of benthic organisms in the region. For example, despite order of magnitude reductions in JWPCP annual mass emission rates of these two contaminants between 1972 and 1975, median levels in both bottom sediments and flatfish from the JWPCP monitoring zone decreased by only about a factor of 1.5 or less (Young et al., 1977b). Subsequent surveys suggest that the present ecological half-time of these chlorinated hydrocarbons in benthic sediments and organisms is on the order of 7 years (US Congress, 1978).

In contrast to the case for DDT and PCB residues, where inputs from the JWPCP system are clearly reflected by local gradients in biological concentrations of these contaminants, levels of the known carcinogen benzo(a)pyrene are relatively low in mussels living inshore of the discharge zone (Dunn and Young, 1976). The concentration for two rocky substrate stations on Palos Verdes Peninsula known to concentrate chlorinated hydrocarbons from the outfall system (Young and Heesen, 1978) were 0.1 and 0.5 µg/wet kg, compared with values of 0.6, 1.4, and 2.3 µg/wet kg for specimens collected from rocks near the three entrances to nearby San Pedro Harbor. (Specimens from nearby pilings ranged up to 8.2 µg/wet kg, resumably relecting the effect of creosote applied as a wood preservative.)

Numerous surveys of trace metals in various species from around major southern California municipal outfalls have shown that these submarine discharges can cause distinct elevations in metals tissue burdens of certain organisms. For example,

Alexander and Young (1976) concluded that coastal intertidal mussels (Mytilus californianus) collected off the Los Angeles-Orange County Basin reflected distinct contamination of the water column by chromium, copper and silver. This same observation was made for a subtidal bivalve mollusc, the purple-hinged rock scallop Hinnites giganteus (= H. multirugosus; Young and Jan, 1979). The three municipal outfall systems located off Los Angeles-Orange counties accounted for the large majority of known anthropogenic inputs of these three trace metals (Young et al., 1973a). The fact that these few organisms feed by filtering particulate material from the water column, and the previously mentioned fact that most of the metals contamination found around municipal diffusers is associated with filterable material, strengthens the hypothesis that suspended wastewater particulates are the vector for this contamination of the biota.

In addition, various invertebrate deposit feeders collected from the highly impacted sediments around the JWPCP outfalls, also reflect the trace metals elevations, with measurable contamination factors (outfall-to-control) ranging from 2 to 10 (Young et al., 1978a, 1978b). Unfortunately, we do not yet know to what degree such elevations of metals concentrations cause physiological, ecological, or public health problems. It should be noted that although levels of total mercury in some invertebrates collected near discharge zones are elevated a few times above normal, for neither invertebrates nor fishes have we observed a median concentration in edible tissue above the past FDA seafood standard of 0.5 mg/wet kg (deGoeij et al., 1974; Eganhouse and Young, 1976, 1978a, 1978b; Eganhouse et al., 1976; Young and Jan, 1979).

In contrast to the situation for invertebrates, fishes living and feeding in or near the highly impacted bottom sediments in major discharge zones have not reflected this dramatic metals contamination in their tissues (deGoeij et al., 1974; Young et al., 1978b). Although we have occasionally observed 2 to 3 fold elevations in outfall zone specimens, depressions of the same order are about as common (McDermott et al., 1976a; Sherwood, 1979). With the exception of mercury, we have concluded that the common toxic trace metals studied to date (Ag, Cd, Cr, Cu, Ni, Pb, Zn) are not biomagnified up marine food webs in areas of trace metals contamination (Young and Mearns, 1979).

ECOLOGICAL EFFECTS

During the past two decades, the coastal discharge sites cited in this report have been surveyed to document the abundance, diversity, and health of marine life. Much of the work has been done by the dischargers themselves or through private and university contracts. Independent researchers, including those at the Southern California Coastal Water Research Project, have also conducted a number of regional surveys of benthic and water column life and contrasted conditions with those at distant coastal sites.

The Benthic Environment

Visual conditions around discharge sites. Using submersibles, remote television, and 35-mm cameras, the Coastal Water Research Project found that the Hyperion outfalls in Santa Monica Bay were colonized by a variety of large attached invertebrates including several species of gorgonians inshore of 30 m and large anemones (Metridium senile) and aggregate anemones (Corynactis sp.) in deeper water (Allen et al., 1976). Large schools of a variety of water column feeding rockfish were common over both outfalls, including diffusers, in deeper water, and the rock ballast appeared to provide refuge for other fishes and invertebrates, such as crabs and lobsters (Allen et al., 1976). Similar results have been obtained from photographic and television surveys at outfalls at Point Loma, Orange County, and Palos Verdes (SCCWRP, unpubl.). In most areas, the bottom adjacent to the pipes had a normal color and was occupied by fishes common in bottom trawls. However, television surveys near the Hyperion sludge outfall in the fall of 1976 revealed a bottom containing loosely aggregated sludge-like material with large numbers of white croaker (Genyonemus lineatus), northern anchovy (Engraulis mordax), and numerous large Pacific electric rays (Torpedo californicus). A grab survey revealed the area of bottom containing the loose sludge-like material was about 2 km^2 in water about 100 m deep at the head of a submarine canyon (Schafer and Bascom, 1976).

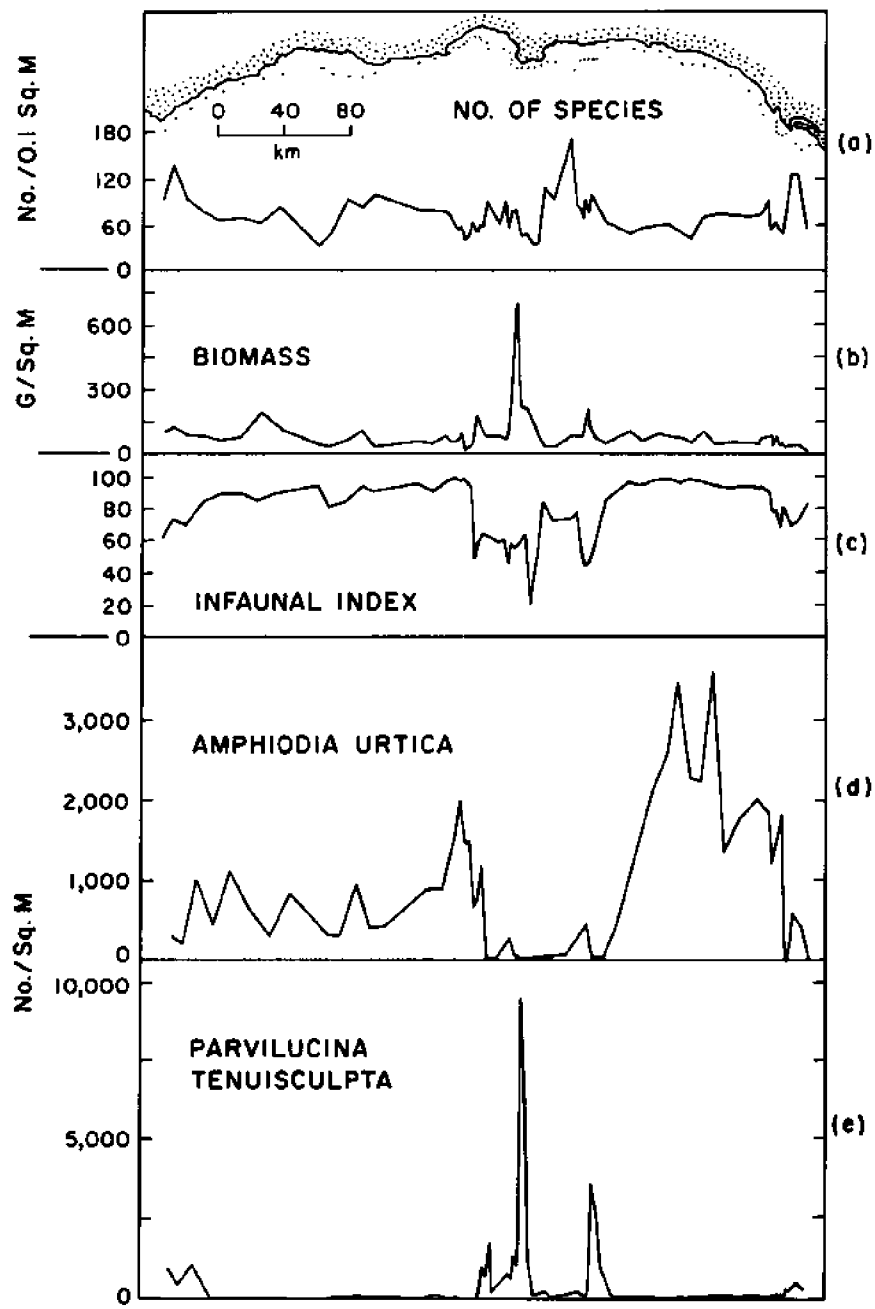
Response of the benthic infauna. The benthic macrofauna that normally colonizes sediments of the coastal shelf is one of the most diverse and stable communities of the open coastal zone. During 1977 and 1978, the Coastal Water Research Project conducted

several large-scale, region-wide, surveys to document variations in the abundance, variety and trophic structure of benthic communities at one depth (60 m) along the entire 200 km southern California mainland shelf (Word and Mearns, 1979a) and at many depths in the 100 km zone offshore of Los Angeles, Orange, and southern San Diego counties (Bascom, 1979). A key element in this study was the Infaunal Trophic Index developed by biologist Jack Word (Word, 1979). The index, which ranges in values from 0 to 100, is a measure of the trophic structure of infuanal communities and uses numerical data from 26 abundant taxa of polychaetes, echinoderms, pelecypods, gastropods, and small crustaceans including species both sensitive to, and tolerant of, conditions near waste discharge sites. High Index values (e.g., 60 to 90) are indicative of a fauna dominated by suspension-feeding organisms which are abundant in rural coastal areas uninfluenced by the major discharges or other sources of organic matter; lower values (0 to 60) are indicative of communities dominated by surface deposit feeding animals (30 to 60) or subsurface deposit feeding animals (0 to 30). When used together with data or number of species, biomass (g/m^2), and some measure of organic content of surface sediments (e.g., percent volatile solids), the Infaunal Trophic Index provides a useful quantitative guide to areas biologically affected by the discharges.

Background conditions. As shown in Figure A.3, much of the southern California mainland shelf at a depth of 60 m was characterized by benthic communities with moderate biomass, high Infaunal Index values, and high diversity (number of species). However, near urban areas of Los Angeles, Orange, and San Diego counties, volatile solids (Figure A.2) and benthic biomass increase, and diversity and thus the Infaunal Trophic Index decreases. In fact, major depressions in the Index actually coincide with the four major discharges (marked on Figure A.3). In addition, however, low Index values indicate that the rural coastal shelf near Point Conception is inhabited by a benthic infuana with an unusually large number of deposit-feeding animals.

In 1956-58, the Allan Hancock Foundation, University of Southern California, conducted a similar survey of the coast. Using their data, Word (1979) found that trophic structure of benthic communities at 60 m was substantially unchanged along most of the coast during the past 20 years. However, deep discharges in Santa Monica Bay, off Orange County and

Figure A.3 Variation in benthic infaunal community structure along the 60-meter contour (Word and Mearns, 1979a).



off Point Loma did not exist, and the early data show that infaunal trophic indicies at these sites were higher and at background levels (Word, 1979).

Using the 1977 60-m data, Word and Mearns (1979b) identified 29 reference or "control" stations with which to compare the scope and magnitude of outfall effects. Overall, these data indicate that a 0.1 m^2 grab captures an average of 71 species and 423 individuals with a Shannon-Weaver Index of 3.05, an Infuanal Trophic Index of 93.5, and a wet weight of 7.05 g (Word and Mearns, 1979b). Echinoderms (particularly the brittle star, Amphiodia urtica) and polychaetes dominated abundance in these "control" regions (Figure A.3), but polychaetes, arthropods, and molluscs dominated species richness (Word and Mearns, 1979b).

Responses to the discharges. It is evident in Figure A.3 that as one approaches the epicenter of a deep water (60 m) outfall impact zone, infaunal abundance and biomass increase while the number of species and diversity generally decrease. Specific changes include a major reduction in the brittle star, Amphiodia (Figure A.3), increases, and then decreases, in several polychaetes (Tharyx sp. and Mediomastus californiensis) and ostracods (Euphilomedes sp.), and increases in several pelycypods (notably Parvilucina tenuisculpta, Figure A.3); in the areas of severe impacts (mainly at the two largest outfalls in Santa Monica Bay and off Palos Verdes) there are increases in another pelycypod (Solemya panamensis) and three polychaetes (Shistomeringus longicornis, Capitella capitata, and Armandia bioculata; Word et al., 1977). Associated with this shift, total abundance and biomass of the community can increase by as much as an order of magnitude to over 30,000 animals, weighing over 100 g/0.1 m^2 (at the Palos Verdes site). Increased biomass is partly caused by increased size of some organisms such as Parvilucina (Word et al., 1977). The total number of species drops from about 70 to less than 50 in a 0.1 m^2 sample.

The magnitude of changes in abundance, biomass, diversity, and number of species from selected stations along the 60-m contour at each discharge site is summarized in Table A.7. As indicated in this table, most biological variables around the Point Loma, Orange County, and Santa Monica Bay outfalls are only slightly changed from control conditions while Palos Verdes is markedly changed. An exception is the Infaunal Trophic Index, which decreases with

Table A.7 Summary of Conditions of the Benthic Infauna at a Series of Stations along the 60-m Isobath at Four Major Discharge Sites and at 29 Coastal Control Sites, All Sampled by 0.1 m^2 Van Veen Grab during 1977. Values are Medians and Ranges. No Stations Taken at Oxnard Discharge Zone.

	Controls	Point Loma	Orange County	Santa Monica	Palos Verdes
No. Samples	29	10	9	9	7
Abundance No/m^2	4,167 (910-12,130)	3,550 (1,860-8,330)	5,530 (3,960-10,780)	3,670 (1,980-12,590)	8,560 (2,810-30,570)
Biomass g/m^2	70.5 (28-112)	52 (43-94)	89 (76-218)	87 (15-135)	207 (77-707)
No. Species No. 0.1 m^2	73 (32-167)	67 (47-125)	80 (65-96)	59 (38-90)	47 (36-78)
Infaunal Trophic Index[a]	93 (83-98)	77 (69-94)	54 (44-79)	63 (48-99)	54.8 (21-64.4)
Shannon-Weaver Diversity	3.17 (2.19-4.16)	3.23 (3.09-3.94)	3.16 (2.52-4.16)	2.93 (1.54-3.31)	2.09 (1.34-3.30)
5-Day BOD mg/kg, wet	632 (266-1,017)	754 (407-2,175)	927 (499-1,931)	817 (486-1,850)	7,781 (943-25,048)

[a] Values > 60 dominated by suspension feeding fauna, <60 and >30 by surface deposit feeding fauna and < 30 by subsurface deposit feeding fauna such as Capitella capitata; see Word, 1979.

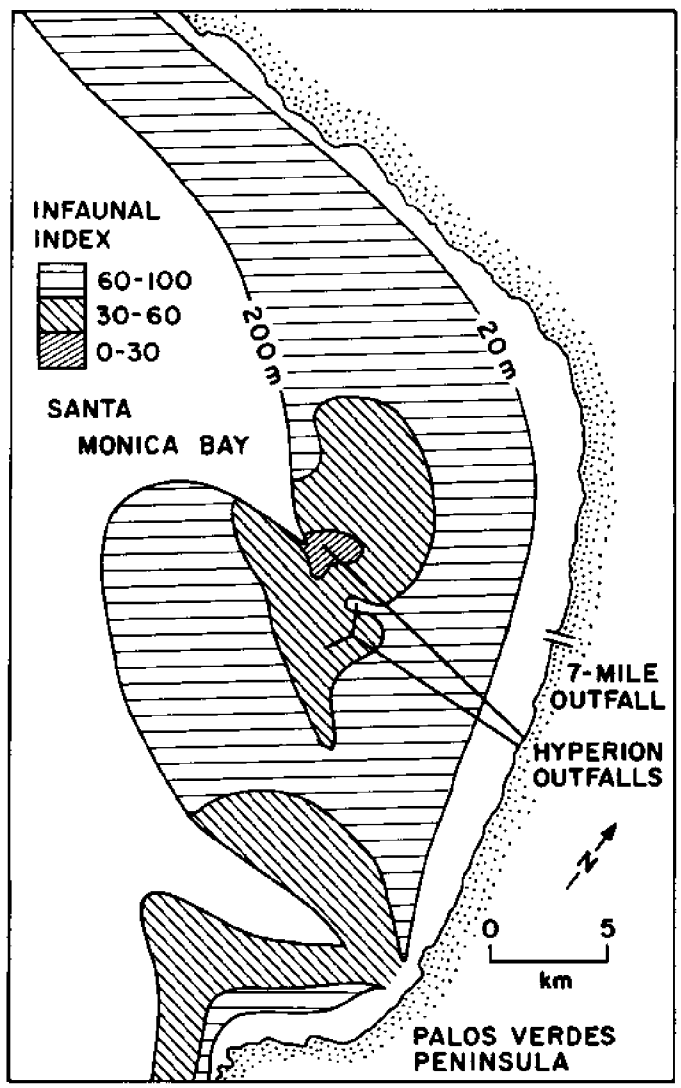
increasing size of discharge. Generally, however, the values indicate that the communities around the Point Loma outfall are still dominated by suspension-feeding fauna (the control condition), while both deposit-feeding and suspension-feeding communities occur around the Orange County and Hyperion outfalls (values both below and above 60 occur); at Palos Verdes, most of the samples were dominated by deposit-feeding organisms.

The other interesting data appear to be the Shannon-Weaver diversities which actually show a slight increase above background at Point Loma (3.23), no change from background at Orange County (3.16), slight decrease in Santa Monica Bay (2.93), but a major decrease off Palos Verdes (2.09) (see Word and Mearns, 1979b).

During 1978, the Coastal Water Research Project conducted surveys at over 300 stations between Pt. Dume and Dana Point and between La Jolla and the US-Mexico border, at depths ranging from 10 to over 300 m, in order to make detailed contours of biological conditions around the four major deep-water dicharge sites (Bascom, 1979). Figure A.4 shows results of contours of Infaunal Index values around the Hyperion discharge sites in Santa Monica Bay. As indicated in this figure, benthic communities dominated by surface or subsurface deposit-feeding infauna (Index values below 60) surround the entire Hyperion 5-mile and 7-mile outfalls. Similar plots for other discharge areas are given in Bascom et al. (1979). Calculations indicate bottom areas dominated by deposit-feeding benthic communities were about 4 km^2 at Point Loma, 11 km^2 at Orange County, at least 60 km^2 in Santa Monica Bay (Figure A.4) and over 94 km^2 associated with the Palos Verdes outfall; other studies indicate there was no area around the Oxnard outfall dominated by a deposit-feeding infauna. In total, then, at least 170 km^2 or 4.7 percent of the 3640 km^2 southern California mainland shelf (between the 20 and 200 m isobaths) is experiencing measurable change in the structure of benthic communities as a result of deep water waste discharge. As suggested by Bascom et al. (1979), about 12 km^2 (or 0.3 percent), principally off Palos Verdes, is considered "degraded," i.e., very dense benthic communities dominated by sub-surface deposit-feeding organisms such as the indicator polychaete, Capitella capitata.

These and earlier surveys also indicate that an area of 2 to 4 km^2 at 80 to 100 m at the end of the Hyperion sludge outfall is occupied by a very dense,

Figure A.4 Contours of infaunal trophic index values calculated for over 120 benthic grab samples taken in Santa Monica Bay during 1977. Values 30 and below indicate a benthic infauna dominated by sub-surface deposit-feeding organisms such as Capitella capitata and are centered about the "7-mile" sludge outfall, but not about the "5-mile" effluent. The region of low index values in the southern third of the Bay may be due to the effects from the Palos Verdes discharges. (Bascom, et al., 1979)



low-diversity, sub-surface, deposit-feeding infaunal community dominated by polychaetes (Figure A.4); for example, in July and December 1974, densities ranged from 3,300 to 14,500 animals/m^2 with 70 percent of the individuals represented by 1 to 3 species (Anon., 1977).

Fish and Macroinvertebrate Populations

Bottom fish and populations of larger macro-invertebrates (crabs, shrimp, sea urchins, etc.) respond to some of the waste discharges through changes in abundance, diversity and health, but in most cases the responses are not pronounced, highly influenced by year-to-year natural variability and not as easily documented as those of the infauna.

Background conditions. During the 1977 60 m survey, 28 coastal control sites, far removed from the major discharges, produced an average of 378 fish (188 of which were considered 1 year and older) of 14.5 species weighing 4.7 kg with a Shannon-Weaver diversity of 1.38, and 181 large invertebrates (prawns, crabs, urchins, starfish, etc.) of 10.5 species weighing 7.4 kg (Table A.8). The catches were dominated by Pacific sanddab (Citharichthys sordidus), ridgeback prawn (Sycionia ingentis), stripetail rockfish (Sebastes saxicola), plainfin midshipman (Porichthys notatus), pink sea perch (Zalembius rosaceus), an urchin (Lytechinus anamesus), and a sea cucumber (Parastichopus californicus). Earlier (Mearns and Greene, 1974), we had concluded that the fishes noted above were normal, prominent and recurring members of the epibenthic community of the mainland shelf at these depths.

Responses to discharges. Compared with the 1977 background data, two of the discharge sites (Point Loma and Orange County) produced catches that were statistically indistinguishable from background in terms of abundance, but were significantly higher in terms of diversity (Shannon-Weaver). The Orange County site also had significantly higher numbers of invertebrate species (22) than the control regions (10.5; Table A.8). In Santa Monica Bay, total catch was significantly lower than in control areas, but catch of fish 1 year and older was not. Also fish diversity and invertebrate abundance, number of species and biomass were significantly higher in Santa Monica Bay than at control sites.

Table A.8 Summary of Catch Statistics for Fishes and Invertebrates Collected in 10-minute On-bottom Time Otter Trawls Along the 60-m Isobath. 28 Coastal Control Sites and Four Waste Discharge Sites on the Southern California Mainland Shelf, Spring-Summer, 1977. (Word and Mearns, 1979)

	Controls[a]	Point Loma[b]	Orange County[b]	Santa Monica[b]	Palos Verdes[b]
N	28	5	3	7	7
FISH					
Abundance (No./tow)					
All Fish	378 (186-623)	232 (114-593)	523 (236-917)	151* (17-393)	56* (26-852)
Less Recruits[c]	188 (109-291)	190 (114-370)	172 (104-757)	100 (17-340)	34* (24-145)
No. Species/tow	14.5 (12-16)	17 (14-21)	16 (14-16)	16 (4-23)	10* (7-15)
Biomass (kg/tow)	4.7 (3.2-7.9)	6.3 (4-16.5)	6.2 (2.5-14.8)	3.6 (1.5-14.8)	2.5* (0.8-4.9)
Shannon-Weaver Diversity	1.38 (1.22-1.57)	1.73** (1.63-2.45)	1.87** (1.02-1.94)	1.83** (1.21-2.08)	1.66 (0.88-1.96)
INVERTEBRATES					
Abundance (No./tow)	181 (90-351)	313 (110-3,640)	167 (112-190)	488** (225-638)	286 (25-843)
No. Species/tow	10.5 (7-14)	11 (9-16)	22** (22-26)	15** (8-20)	11 (9-32)
Biomass (kg/tow)	7.4 (2.5-12.5)	1.17 (0.8-6.0)	7.4 (2.1-16.7)	7.9** (3.1-23.1)	18.9** (4.9-21.3)

[a] Median and 95% confidence limits.

[b] Medians and ranges

[c] Recently settled post-larval young of the year occurred in patches along the coast; this statistic excludes these.

*Value significantly lower than control ($p \leq 0.05$)
**Value significantly higher than control ($p \leq 0.05$)

The fourth site, Palos Verdes, produced fish catches that were significantly lower than the control sites in terms of abundance (56 fish per haul), number of species (10) and biomass in spite of apparently normal Shannon-Weaver diversity (1.66) and normal or above normal catches of invertebrates (Table A.8). Clearly the fish fauna at this site was less abundant than background at this depth and time. Other data indicate catches can be very high and very variable at this site.

Some of these sites also differed from each other and from controls in terms of dominance and community structure. For example, Pacific sanddabs (Citharichthys sordidus) were captured in unusually low abundance at Point Loma, Santa Monica Bay, and especially Palos Verdes; similar differences were noted for stripetail rockfish (Sebastes saxicola). Inspection of the data revealed the differences were due primarily to low numbers of young-of-the-year at these outfall sites and to a few patchy, but high catches of young-of-the-year juveniles at several control sites off Santa Barbara. Examination of Table A.9 indicates that pink sea perch (Zalembius rosaceus) and English sole (Parophrys vetulus) occurred in unusually high abundance in the Point Loma area, yellowchin sculpin (Icelinus quadriseriatus), California tonguefish (Symphurus atricauda) and speckled sanddab (Citharichthys stigmaeus) in the Orange County area and speckled sandabs in the Santa Monica Bay area. Dover sole (Microstomus pacificus) were particularly abundant at Palos Verdes.

Examination of invertebrates revealed a similar mix of possible enhancements and depressions. For example, the urchin, Lytechinus anamesus appeared to be unusually abundant in the Point Loma trawl area, but unusually reduced in Santa Monica Bay and off Palos Verdes. In contrast, the ridgeback prawn, Sycionia ingentis, appeared to occur in normal abundance at all sites except Palos Verdes where it was nearly an order of magnitude more abundant than in the control areas.

A summary of possible enhancements and depressions is shown at the bottom of Table A.9 which indicates there were twice as many enhancements (4) as depressions (2) at Point Loma (the smallest discharge), five times as many at Orange County (5 and 1, respectively), but only about one-third as many at Santa Monica Bay and Palos Verdes (the largest discharge).

Table A.9 Summary of Median Catch per 10 min. Otter Trawl Tow for 18 Species of Bottom Fish and Invertebrates at Control Sites (n=28) and Four Discharge Sites along the 60-m Isobath of the Southern California Mainland Shelf, 1977 (Word and Mearns, 1979).

	Controls		Point Loma	Orange County	Santa Monica	Palos Verdes
No. Samples	28		5	6	7	7
FISHES						
Pacific sanddab	124	(80-209)	57 **	89	36 **	1 **
Stripetail rockfish	27	(10-177)	1 *	6 *	20	7 *
Plainfin midshipman	21	(2-40)	3	18	7	3 *
Pink seaperch	9.5	(5-15)	21 *	7.5	1 **	0 **
Dover sole	4.5	(2-7)	8	10.5	7	13 *
Bigmouth sole	1	(0-3)	3	2	0 *	0 *
Yellowchin sculpin	5.5	(3-15)	1	65 **	2	0 **
Longspine combfish	2	(0-12)	0	7.5	3	0 **
California tonguefish	2	(1-4)	5	8 *	<1 *	1 **
English sole	2	(1-5)	12 +	8.5	1	0
Speckled sanddab	<1	(0-1)	0	130 **	19 **	0
INVERTEBRATES						
Lytechinus anamesus	9.5	(3-162)	218 *	26	0 *	0 **
Parastichopus californicus	6	(2-15)	2	22 *	12	0 **
Astropectin verrilli	<1	(0-9)	2	1	27 **	0
Luidia foliata	3	(1-6)	1 *	9	1 *	0 **
Sycionia ingentis	24	(16-98)	15	32	149	228 *
Mursia quadichaudi	<1	(0-<1)	0	2.5 **	0	1 *
Pleurobrachaea californica	<1	(0-1)	0	1.5 *	0	3 **
No. Enchancement			4	5	2	4
No. Depressions			2	1	6	11
Ratio Enhancements/ Depressions			2.0	5.0	0.3	0.4

+0.2 < p >0.1 Different from control value.
* 0.1 < p >0.05 Different from control value.
**p <.05 Different from control value.

These results generally confirm what has been concluded from numerous earlier surveys (e.g., Mearns and Greene, 1974; Allen, 1975; and Allen and Voglin, 1976) except that fish were much more abundant at Palos Verdes in the past. Special trawls and photographic surveys have also confirmed that bottom fish and macroinvertebrates have been abundant in the 2 to 4 km^2 area containing sludge or sludge-like material in Santa Monica Submarine Canyon. For example, a 20-minute trawl taken precisely through the affected area in December 1976, produced 1,279 fish, dominated by Dover sole (26 percent of the catch), white croaker (24 percent), pink sea perch (15 percent), longspine combfish (Zaniolepis latipinnis, 14 percent), speckled sanddab (7.2 percent), and 18 other species, as well as over 1,400 invertebrates of 24 species dominated by sandstars (Astropectin verrilli, 51 percent) and ridgeback prawn (24 percent, Anon., 1977). Routine surveys of this site have not been conducted in recent years, but comparison with low catches in past sampling (e.g., Mearns and Green, 1974) indicate that bottom fish population are extremely variable at this site.

Diseased fish populations. While abundance and diversity of bottom fish appear to be only moderately changed at major discharge sites, more profound changes occur in the health of some fishes at a few sites. For example, a fin erosion disease occurs or has occurred at three of the sites, but is absent at the remaining two and at control sites. In addition, bottom fish living at some of the sites have enlarged livers.

The fin erosion disease was discovered in Dover sole from the Palos Verdes Peninsula, when trawls were taken there for the first time in 1969. Since then, over 30 species of fish bearing eroded fins have been reported from Palos Verdes (Sherwood and Mearns, 1977). As described by Sindermann (1979) and Sherwood (1979), a similar disease has been observed in bottom fish from dumping sites in the New York Bight, an industrial waterway in Seattle, off Orange County, in deep water in Santa Monica Bay, and in Boston Harbor. Additional trawl surveys confirmed this disease was absent at Point Loma, Oxnard, and at control sites in southern California (Sherwood and Mearns, 1977, Table A.10).

Table A.10 Prevalence of Fin Erosion and Skin Tumors in Dover Sole Collected in Trawls Taken along the 60-m Isobath between Point Conception and the US-Mexico Border, Summer, 1977. Samples Grouped to Increase Regional Sample Sizes.

	Pt. Conception to Point Dume	Santa Monica Bay	Palos Verdes	Southern San Pedro Bay	Laguna Beach to Pt. La Jolla	Point Loma
No. Trawls	13	8	7	3	12	3
Dover Sole						
Total number	120	53	201	36	66	81
Catch/haul	9.2	6.6	28.7	12.0	5.5	27.0
No. with Fin Erosion (%)	0	0	63 (31%)	0	0	0
No. ≤ 120 mm SL*	90	16	147	15	18	58
No. with tumors (%)	3 (3.3%)	1 (6.2%)	3 (2.0%)	1 (6.7%)	0	1 (1.7%)

*Disease is initiated in young of the year and decreases in prevalence with age; size limitation (≤120 mm SL) includes mainly 1976-77 year class.

The diseases do not appear to be caused by infectious microorganisms, but may be related to high chlorinated hydrocarbon levels (especially PCBs) in tissues of affected fishes and to sediments highly contaminated with trace metals and chlorinated hydrocarbons (Sherwood and Mearns, 1977; Sindermann, 1979). The disease was experimentally induced in previously healthy Dover sole exposed to sediments from Palos Verdes (Sherwood and Mearns, 1977). It also affects post larval fish within one to two months after they settle out of the plankton into the Palos Verdes discharge area (Sherwood, 1979). Finally, another study suggested that affected Dover sole at the Orange County outfall were in fact migrants from Palos Verdes (McDermott-Ehrlich et al., 1977), and that the disease was probably affecting Dover sole in Santa Monica submarine canyon within a year after discharge of sludge was initiated in 1957 (Sherwood, 1979).

Together, these studies suggest that of the five discharge sites, the disease is clearly initiated at one (Palos Verdes), possibly at another (the Hyperion sludge discharge site), but not at any other site.

Liver size, color, and structure of flatfish also seem to respond to some of the waste discharges. For example, livers in Dover sole from Palos Verdes are over twice the size of control fish; fish from Orange County, Santa Monica Bay, and at a natural oil seep have livers of intermediate size (Sherwood, 1979). Several histological studies indicate livers of some fish have increased lipid vacuolation and compensatory changes in structure and organization of the tissue (Pierce et al., 1977; Sherwood and Mearns, 1977). Laboratory experiments indicated that both Palos Verdes sediments and sediments contaminated only by PCBs caused increased liver size in Dover sole (Sherwood, 1979). The significance of this change to the fish, or its reversibility, is unknown.

Skin tumors occur in pleuronectid flatfish throughout the north Pacific, but have also been implicated as responses to waste discharge (reviewed in Sindermann, 1979). However, all studies in Puget Sound and southern California have failed to show that tumor prevalence near discharge sites is any higher than at control sites (Sherwood and Mearns, 1977, and Table A.10).

The Inshore Environment

Although the major southern California ocean outfalls discharge in relatively deep water offshore, inshore waters and subtidal habitats (i.e., within 1 km of shore of the 15 m isobath) are not immune to effects.

During the 1950s and 1960s, beds of giant kelp, (Macrocystis pyrifera), decreased in size at Palos Verdes, La Jolla, Point Loma, and other coastal sites. By the late 1960s, beds were nearly extinct off Palos Verdes, although elsewhere they had partially recovered (Mearns et al., 1977). In addition, the diversity and abundance of other seaweeds, macroinvertebrates, and fishes along the 15 m contour at Palos Verdes were greatly reduced, and divers reported a large amount of flock or detritus on rocks in the area (Grigg and Kiwala, 1970). These conditions were attributed to a combination of factors including inshore accumulation of wastewater solids from the Palos Verdes outfalls, overgrazing by sea urchins, and abnormally warm sea surface temperatures during 1957-58.

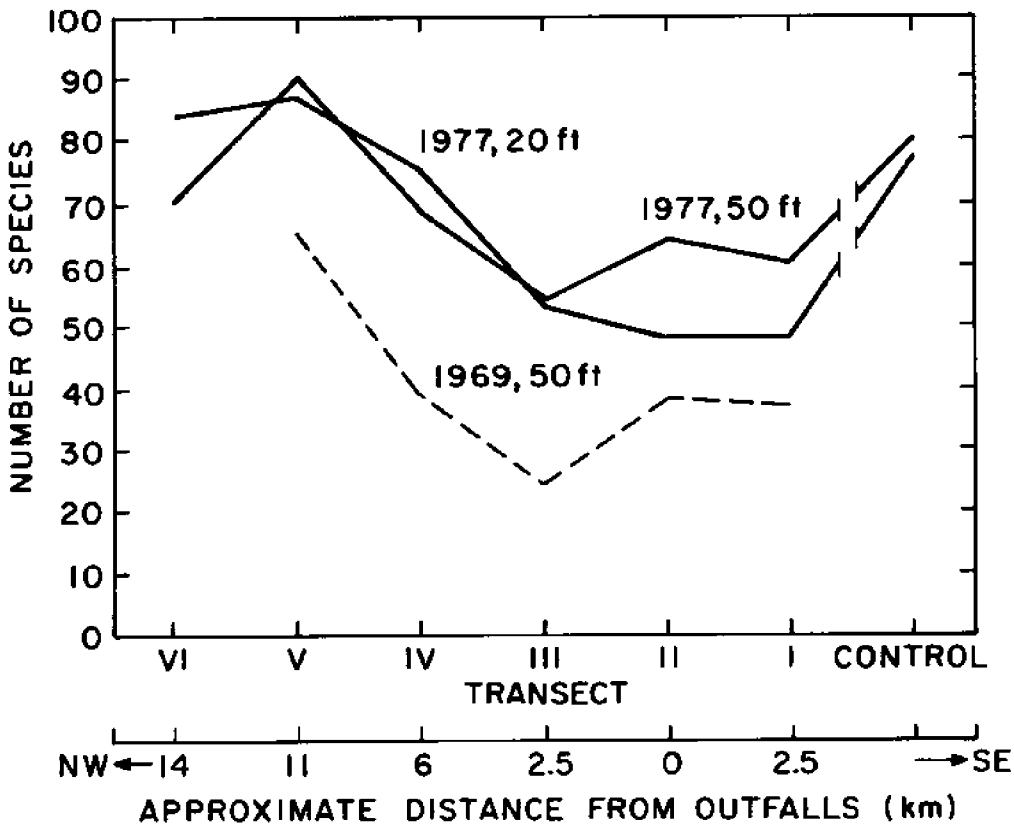
By 1973, these conditions began to reverse and by 1977-78, there were marked increases in diversity and abundance of fishes, macroinvertebrates, and

seaweeds at 15 m (Grigg, 1979, and Figure A.5), and kelp beds had increased from a few plants to several large beds covering 1.2 km^2. Most of the recovery took place along the coast northwest and most distant from the discharges. The cause, or causes, of the decline and recovery of biological conditions at Palos Verdes are still uncertain. Kelp transplant activity has certainly been a factor (Wilson et al., 1978). However, there has also been a long-term general increase in visibility in Santa Monica Bay during 1973 to 1978 (Mearns, 1979a). As noted above, some changes have also occurred in effluent quality such as a dramatic (96 percent) decrease in DDT mass emission rate (from 21.6 tonnes in 1971 to less than 0.8 MT in 1977) and moderate (38 percent) decrease in suspended solids mass emission rates (from 164,000 MT in 1971 to 104,000 MT in 1978; Schafer, SCCWRP, pers. comm.). Perhaps all these changes have led to improved conditions. At present, it appears that outfall-related factors are still involved in decreased biological diversity at 15 m isobath near the discharge site itself; 6 m depths appear to be little affected at this time (Grigg, 1979).

There is still some confusion about changes in condition of kelp beds at Point Loma. As pointed out above, the decline in the size of kelp canopy off Point Loma occurred prior to initiation of discharge offshore (1964), and recovery from 1964 onward coincided with both wastewater discharge and culture activity. At present, a kelp bed of substantial size (over 7.5 km^2) is centered about the base of an outfall discharging over 20,000 MT/yr (dry weight) of suspended solids. These data indicate that deepwater discharge was not a factor causing previous loss of these kelp beds. The situation also suggests that deepwater waste discharge itself at the present mass emission and effluent characteristics is not sufficient to degrade kelp beds in inshore rocky subtidal areas.

Outfalls off Orange County and Ventura and in Santa Monica Bay discharge mainly offshore of sandy beaches. With the exception of infrequent abnormal beach coliform counts (some of which are due to sources other than the offshore outfalls), the nearshore benthic biota show no responses that can be attributed to the offshore outfalls. For example, there is an area of depressed infaunal indices inshore of the Orange County deep-water outfall, but it is isolated from the low values around the outfall itself (Bascom, 1979); our present opinion is that this inshore area has been affected by runoff from the Santa Ana River.

Figure A.5 Total number of algae, invertebrate and fish species observed at stations off Palos Verdes and La Jolla (control) in 1969 and 1977.



The Pelagic Environment

The occurrence of DDE, originating from the Palos Verdes discharge, in intertidal biota (Young et al., 1978a) and at a distant pelican breeding site (Anderson et al., 1977) confirms that materials from deep water discharge can enter the upper pelagic zone. However, other than DDE, the effects of potentially toxic materials on euphotic zone organisms is largely unknown.

Several past studies confirm that the plankton community has responded to some of the larger discharges. The Southern California Coastal Water Research Project (1973) reported that in Santa Monica Bay over a 14 year period from 1957 to 1970, annual variations in net-haul plankton volumes varied according to changes in the plankton abundance in the entire California Current region. However, the study also found evidence that total plankton volume in the bay was higher when the Hyperion Treatment Plant was operating a shallow water discharge before discharge was diverted to the deep water sites offshore. Presumably, in past years, much more of the wastes were entering the mixed layer and contributing to plankton growth. However, recent evidence for stimulation of phytoplankton by nutrients (especially ammonia) from the discharges has been gathered by several workers. Throughout several seasons, Eppley et al. (1978) found consistently higher phytoplankton standing crops and productivity in Santa Monica Bay than at other sites of equivalent depth and distance from shore. In an earlier study, MacIssac et al. (in press) found that ammonia levels in the mixed layer near both the Santa Monica Bay and Palos Verdes discharges were sufficient to enhance phytoplankton growth yet inhibit phytoplankton nitrate uptake.

Attempts to determine responses of fishery populations have so far been inconclusive. Using catch as well as catch and effort data compiled by the California Department of Fish and Game from 16 by 16 km^2 statistical blocks, Mearns (1977) and Allen and Voglin (1977) were unable to identify geographical trends that could be conclusively attributed to proximity to the discharges. For example, in 1973 in 37 statistical areas ranging from Point Conception to the US-Mexico border, the average commercial party boat coastal catch was 1.47 fish per angler hour with a cumulative average of 24.3 species per area. Seven statistical areas occupied by the five major discharge sites produced an average of 1.67 fish per angler hour (ranging from 1.25 off Point Loma to 3.84 in Santa

Monica Bay) and a cumulative average of 34 species per block (Mearns, 1977). Lack of an observable adverse response may simply be due to the large geographical areas (10 by 10 mi^2) over which catch and effort data are summarized. However, Allen et al. (1975), using standardized fishing procedures with set lines and rod and reel, demonstrated a slightly higher catch per unit effort and variety near the Hyperion sludge outfall than at more remote sites in Santa Monica Bay. Commercial landings data did produce one important observation; seventeen statistical blocks within a 50 km radius of the three largest outfalls produced 73 percent of the 1973 southern California catch (178,000 MT; Allen and Voglin, 1977). Thus, most of the commercial fish landed are taken from those waters closest to the urban areas and outfalls.

Recovery and Resiliency

Changes in the health, diversity, and abundance of marine life caused by southern California coastal discharges are not necessarily permanent. From August 1970 through August 1972, Smith (1974) recorded biological and physical changes accompanying termination of a 15-year-long discharge of primary effluent from an inshore (20 m deep) diffuser off Orange County. He also monitored changes in the benthic infauna associated with initiation of discharge of these wastes through the deeper (60 m) diffuser. Later surveys of both sites were done by the Coastal Water Research Project and the Orange County Sanitation Districts.

Following termination of shallow discharge, Smith (1974) reported rapid recovery; sediment organic carbon and total sulfide concentrations dropped to background within three months, while the infauna returned to normal diversity, abundance, and community structure within one year. In addition, trawl surveys indicated that fish abundances decreased by about one-half several years after termination. In contrast, changes were much slower at the new deeper site following initiation of discharge. Fish catches and the abundance and biomass of infauna increased slowly from 1969 through 1976 at stations nearest the new diffuser, but the most dramatic responses occurred two to four years after discharge began; during this period, *Capitella capitata* density increased from less than 50/m^2 to over 1,000/m^2, while the brittlestar *Amphiodia urtica* decreased from 500 to 1,000/m^2 down to less than 50/m^2 (Mearns, 1979b). Also, during the second through fourth years, sediment copper

concentrations increased from an average background value of about 16 mg/dry kg to slightly over 30 mg/kg. Subsequent sampling (1975 to 1978) suggests no further progressive increase in intensity of effects or size of affected area after 1975 to 1976; thus, the benthic environment at this site may be in equilibrium with the input (Mearns, 1979b).

Unfortunately, no experience exists which would allow observation of the rate of recovery from termination of deep-water (e.g., 60 m) discharge. However, several periods of sludge discharge at Point Loma (October 1964 through December 1966 and July through December 1968) were accompanied by a major increase in sediment BOD and benthic biomass, and these periods were then followed by a return to background sediment BOD in 3 to 4 years and return to background biomass in less than 2 years (Orlob, 1977). As noted earlier, there were apparently no effects of these added discharges on inshore kelp beds. In contrast, as noted above, a nearly 40 percent reduction in solids emission from Palos Verdes during the 1970s may be a factor involved in the nearly parallel recovery of rocky subtidal communities and kelp beds. These kinds of observations therefore suggest that biological responses to control of solids at deep water sites may occur within a period of several years.

As noted in the previous section of this report, such is not necessarily the case for specific toxic chemicals such as chlorinated hydrocarbons, however. Despite source control of DDT in 1970, sediments off Palos Verdes continue to contain extremely high concentrations which are, in turn, leading to continued contamination of the benthic and water column biota at this site (Young et al., 1977b; Young and Heesen, 1978). Fortunately, DDT concentrations in the Brown pelicans roosting at an offshore island (100 km distant from Palos Verdes) have decreased substantially, and through 1975 reproductive success has increased (Anderson et al., 1975); reproductive success has decreased again since 1975 and low availability of prey fish is thought to be a factor (Hunt, pers. comm.)

PUBLIC HEALTH

Pathogens and Their Indicators

Enteric bacteria and viruses from southern California effluents occur at measurable concentrations in seawater near the discharge sites. Total coliform bacteria are most frequently monitored in effluents, surface and subsurface samples above sewage plumes, at "nearshore" stations (within 1,000 ft from shore or over the 30 ft isobath, whichever is closest to shore) and at numerous "shore" stations (sampled from the beaches). Sampling is conducted daily at many beach stations, and weekly, monthly, and quarterly at offshore surface and subthermocline stations. All of the data are available to the public, but very little have been formally published.

Inspection of numerous data sets by us indicates that total coliform concentrations in final effluents are generally in the range of 10^7 to 10^8/100 milliliters (ml) (confirmed in Morris et al., 1976). Thus at the edge of initial dilution zones and below the thermocline, concentrations can be expected to be on the order of 10^5 to 10^6/100 ml. Subsequent dilution and settling reduces these subsurface concentrations to 10^2 to 10^4/100 ml within a few km from the outfalls (confirmed in Morris et al., 1976), while the thermocline generally keeps surface water concentrations well below 10/100 ml. Inshore of all major discharge sites, surface "nearshore" and "shore" samples (defined above) are generally below 10/100 ml (e.g., 75 percent of the time at Point Loma, Orlob, 1977) and generally meet all state bathing and shellfish standards. However, there are exceptions, particularly in winter months, when runoff causes increased concentration at shore stations near drains and rivers, and when loss of the thermocline and onshore winds occasionally combine to transport diluted surfacing plumes inshore. This situation is rare in Santa Monica Bay, at Point Loma, and off Orange County, but more common in winter at Oxnard and Palos Verdes; these dischargers do chlorinate during such episodes.

An earlier study (Rittenberg, 1956) indicated high coliform concentrations (up to 12,000 MPN/cm^2) in surface sediments around the large southern California outfalls during the years when the discharges were nearshore. Although few sediment samples have been analyzed for bacteria offshore and adjacent to the newer deep water discharges, it can be

assumed that such sediments also contain high concentrations of coliform bacteria that have settled out of the plumes.

In 1974 and 1975, SCCWRP sampled pathogens in water and sediments near the Hyperion 5-mile and 7-mile outfalls. Of the two pathogens studied, only salmonella was isolated in water and sediment samples along with coliform and fecal streptococcus bacteria. Salmonella were isolated in less than 8 percent of the samples containing less than 1,000 indicator bacteria (coliform or fecal streptococcus) per 100 ml. However, salmonella could be detected in 25 to 75 percent of the samples containing in excess of 1,000 fecal coliforma/100 ml, and in 38 to 100 percent of the samples containing in excess of 1,000 fecal streptococci/100 ml. Using these data, we estimate that the maximum concentration of salmonella near the plumes was about one organism in 400 ml of water (0.25/100 ml) and about one salmonella per 20,000 indicator bacteria.

In a pilot study, Morris et al. (1976) detected enteric viruses in caged mussels (Mytilus californiensis) attached to buoys and deployed several miles offshore and adjacent to each of the three Los Angeles and Orange County outfalls. Like the coliform bacteria, virus concentrations (plaque forming units, PFU) were low or undetectable at the surface (average 0.01 PFU/g of mussel tissue), but increased with depth (0.19 PFU/g of tissue at 8 to 20 m, and 0.5 PFU/g at 25 to 35 m). Since mussels concentrate bacteria and other particulates about 1,000-fold, we judge virus concentration in water adjacent to the outfalls to range from 0.00001 to 0.0005 viruses/ml or from 0.01 to 0.5/l, with the lowest concentrations occurring near the surface and away from the discharges. Additional calculations suggest that enteric virus at these sites have a deactivation rate about one-sixth that of coliform mortality (Morris et al., 1976). Measurements of coliform bacteria die-off rates in the field range from 0.1 to 2 days.

Of nine shore stations sampled, Morris et al. (1976) detected enteric viruses in only two sites - near a marina and an adjacent flood control channel. Coupled with data on coliform concentration, these observations suggested runoff to be the most likely source.

Trace Chemicals

As discussed above, the principal, known public health problem from chemical pollutants in municipal wastewater effluents is the contamination of organisms by DDT residues in the JWPCP discharge zone off Palos Verdes Peninsula. Five of seventeen seafood species (approximately 30 percent) collected there during 1975 to 1977 exceeded the US Food and Drug Administration standard of 5 mg/wet kg (Young et al., unpubl.), and we have repeatedly informed public regulatory agencies of these findings. Although PCB residues are easily measured in organisms collected near several large municipal outfalls, typical levels in edible tissues seldom exceed the proposed FDA standard of 2 mg/wet kg (Young et al., 1975a and unpubl.; McDermott et al., 1976b, 1978; Young and Heesen, 1978). Similarly, typical concentrations of total mercury in edible tissues have not been observed above the past FDA standard of 0.5 mg/wet kg for organisms collected anywhere in the Bight (Eganhouse and Young, 1976, 1978a, 1978b; Young and Jan, 1979).

PROJECTIONS

Mass emissions of materials from the major southern California discharges are not necessarily constant; in fact, during the past several years there have been slight but significant reductions in inputs of many metals, PCBs and suspended solids due both to source control programs and to major changes at several treatment plants. For example, mass emission rates of suspended solids from JWPCP into coastal waters off Palos Verdes have decreased about 38 percent between 1971 and 1978 (from 167,000 to about 102,000 MT/yr). This and other plants are now constructing facilities to further reduce emissions (including partial secondary treatment).

Recently, the Coastal Water Research Project was asked to forecast coastal conditions that might exist during and after initiation of the new treatment schemes (US Congress, 1978). To begin the forecast effort, data on projected wastewater quality were obtained from the three largest dischargers and converted to mass emission rates. The forecast period is to begin in 1983 and extend to 1985 (or longer, if required for a specific forecaster). Time series data on sediment chemistry and on variation in benthic infauna, fish populations, and diseased populations were acquired and reviewed, and a new mathematical model incorporating sedimentation and sediment

resuspension processes was constructed and tested (Hendricks, 1979). Next, similarities and differences among the outfall sites were carefully examined to reveal possible relationships between input variables (such as suspended solids) and size or magnitude of coastal responses (e.g., benthic biomass, see Figure A.6, relative sediment contamination, etc.). Finally, based on available information from field and laboratory tests, assumptions were made about the behavior of pollutants (e.g., there is no significant desorption of metals from particulates, coliform die-off, and sedimentation rates will remain unchanged, etc.) and about cause-effect relationships (e.g., the excess benthic biomass and depressed infaunal indicies around outfalls are biostimulatory and due to the excess organic matter in solids, part of which settle around the discharge sites; fin erosion disease is a function of PCB contamination in affected fish; etc.). The net result was a collection of 10 forecasts covering a variety of subjects and each with various degrees of uncertainty and specificity.

Several kinds of conclusions were reached. First, we determined that pollutant inputs for the 3 largest dischargers would be reduced by more than half of 1976 levels (e.g., 50 to 70 percent depending on the material; some examples in Table A.11). Next, we concluded that some environmental conditions will not change substantially either because: 1. they are not caused by the discharges (e.g., incidence of skin tumors in young-of-the-year pleuronectid flatfish); 2. they are already too close to background to discern a measurable change (e.g., surface and subsurface DO concentrations; phytoplankton and zooplankton densities); or 3. they are naturally too variable to discern a measurable change (e.g., sport and commercial fish landings, and again, phytoplankton and zooplankton densities). We then identified some changes that would occur in general and at all three sites. For example, coliform bacterial concentration at nearshore stations (defined above) would be reduced by one to two orders of magnitude, so that present occasional excursions above state standards will occur much less frequently in the future (unless the uncertainty that they are caused by other sources is verified); total excess benthic infaunal standing crop (biomass in tonnes), will be reduced by about 90 percent (with most of the change occurring off Palos Verdes, Table A.11). The size (in square kilometers) decreases by about 70 percent (from 73 to 22 km^2, Table A.11), and the length of coastline along

Figure A.6 Relationship between suspended solids, mass emission rates (J_e) and excess standing crop of benthic infauna (S_e) at six southern California ocean discharge sites. Open circle and bar represents median and range of estimates of standing crop; x represents standing crop near the Hyperion 7-mile sludge outfall (excluded from regression line).

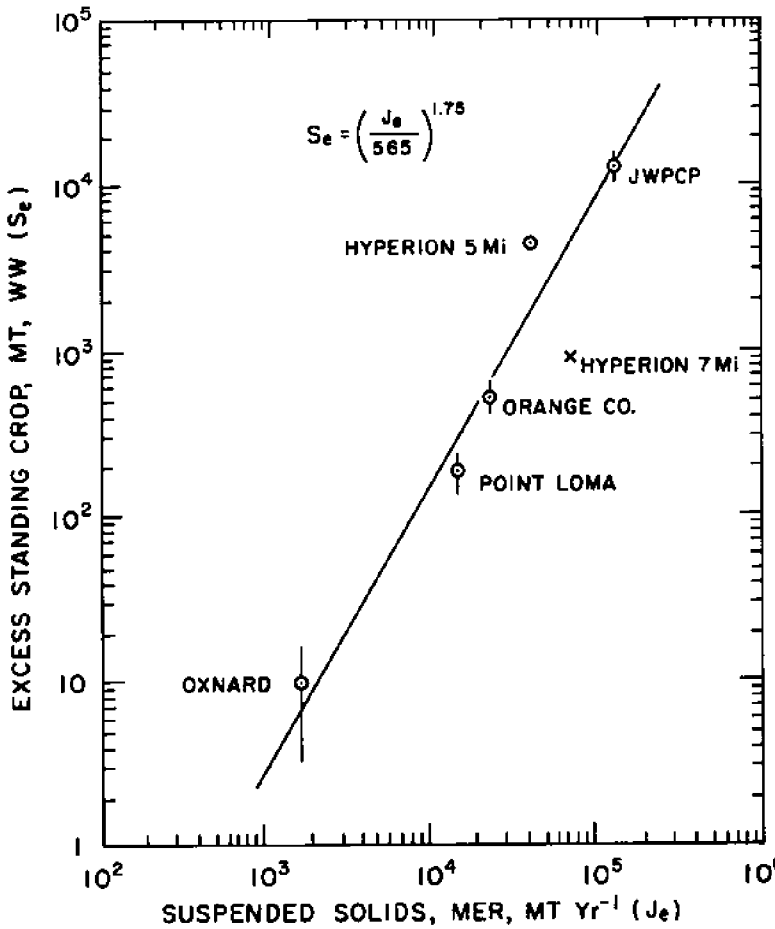


Table A.11 Summary of Some Existing (1976) and Predicted Conditions in Effluent Mass Emission Rates, Benthic Infauna and Sediment Chemistry Associated with Expected Changes in Effluent Quality from the Three Largest Southern California Discharges (in US Congress, 1978).

	Hyperion						OCSD		Total	
			5-mile		7-mile					
	1976	1983-88	1976	1983-88	1976	1983-88	1976	1983-88	1976	1983-88
Flow (mgd)	353	275	359	365	4.1	4.0	182	215	898.1	859
SS (mt/yr x 10^3)	138.9	19.9	38.3	15.1	56.2	58.0	32.3	9.94	265.7	102.9
Chromium (mt/yr)	370	50	95	23	66	96	48	8.5	579	177.5
PCB (kg/yr)	1,300	170	110	120	120	120	990	660	2,520	1,070
Excess Standing Crop. (mt)	13,390	600	4,660	1,000	NC	NC	550	110	18,600	1,710
Area affected by ESC (km^2)	41	8	23	10	NC	NC	9.1	4	73.1	22
Distance to Background Infaunal Index (km)	48	5 *	24	1	NC	NC	8	<0.5	80	<6.5
Chromium Affected area (ppm)	1,050	220	100	25 *	570	NC	17	8.2	NA	NA

*Greater than 20 yrs, or post 1983 equilibrium
SS = Suspended solids
NA = Not applicable
NC = Not computed
ESC = Excess Standing Crop

the 60 m isobath affected by below-background infaunal trophic indicies will decrease by 92 percent (from 80 to less than 6.5 km, Table A.11). Also sediment contamination by specific pollutants (e.g., chromium, Table A.11) would decrease by 50 to 80 percent (depending on the site and contaminant).

Finally, we arrived at some conclusion regarding site specific change. Among the most important was the conclusion that rate of recovery off Palos Verdes would be extremely slow compared with other sites; this we believe is due mainly to very heavy past loadings of bottom sediments and to the apparently long ecological half-life of DDT and PCB contamination in sediments and in benthic or nearshore populations (estimated to be approximately 7 years). We also concluded that the prevalence of fin erosion disease would decline, but would still be affecting about 9 percent of the Palos Verdes Dover sole population in 1988; however, there remain uncertainties that total PCB is indeed the only cause of the disease and that the ecological half-life is indeed on the order of 7 years at this site. (See Table A.12)

Metals contamination levels in the benthic environment, as reflected by surficial sediment concentrations, do not appear to be rapidly decreasing around the major outfall systems. Young (US Congress, 1978) concluded from 1974 to 1976 monitoring data for the JWPCP discharge that the median half-time for six metals (Cd, Cr, Cu, Ni, Pb, Zn) in sediments a few kilometers downcurrent of the outfall diffusers was 6 years (range 5 to 13 years). Further, it appears that sediment concentrations of metals around the Hyperion (City of Los Angeles) 7-mile sludge line remained fairly stable between 1970 to 1971 and 1976, and that metals levels had reached a high equilibrium in sediments around the new (172) 5-mile outfall of Orange County Sanitation District (US Congress, 1978). This is consistent with the fact that, in general, only second-order reductions in metals emissions have been obtained since 1971 by these major municipal wastewater systems (Schafer, 1977).

In addition to the previously published forecasts (US Congress, 1978), new data (Grigg, 1979) suggest that nearshore rocky subtidal communities may have already responded and partially recovered from reduction in pollutant emissions off Palos Verdes. A recent computation (Mearns) suggests that further reduction in solids to the level reported in Table A.1 (e.g., 20,000 MT/yr) could be sufficient to allow full

Table A.12 Summary of Ecological and Public Health Effects at Each of Five Major Southern California Discharge Sites.

	Oxnard	Point Loma	Orange County	Santa Monica		Palos Verdes
				5-mile	7-mile	
Flow (mgd)	11.2	116	180	319	4.6	335
SS* (mt/yr x 10^3)	1.52	20.5	32.8	59.7	26.9	102.0
Sediment Contam.						
Metals	1	2	2	2 -3	10 -100	10 -100
CHCs	1	5	10	50	100	100 -1,000
Ecosystem Response						
Area (km^2) of excess biomass	1	8	9.1		23	41
Area (km^2)						
Infaunal Index 60	0	4	11		60	94
Infaunal Index 30	0	0	0		2-4	11
Bottom fish & shellfish (ratio enhanced/ depressed species)	1 (est.)	2	5		0.3	0.4
Fin rot initiated?	No	No	No	No	Poss.	Yes
Nearshore diversity affected?	No	No	No	No	No	Yes (8 km)
Public Health Response						
Nearshore coliform standards exceeded?[a]	Occ.	No	Occ.	Occ.	--	Occ.
Seafood contamination?						
Metals, fish?[b]	No	No	No	No	No	No
Metals, shellfish?[b]	ND	Yes	Yes	Yes	Yes	Yes
DDT[c]	No	No	No	No	No	Yes
PCBs[d]	No	No	No	No	Occ.	Occ.
Hg[e]	No	No	No	No	No	No

* SS = Suspended Solids
[a] May not be due to discharge [b] No standards [c] >5ppm [d] >2ppm [e] >0.5ppm

recovery of kelp beds and subtidal community structure at 15 m along the peninsula. In summary, then, it is not unlikely that some major ecological recovery will occur as a result of planned and projected treatment strategies, that the recovery will involve mainly subtidal and benthic ecosystems.

ACKNOWLEDGEMENTS

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Appendix B

Puget Sound Case Study

Lawrence E. Birke, Jr.
L. Edwin Coate
Richard C. Bain, Jr.

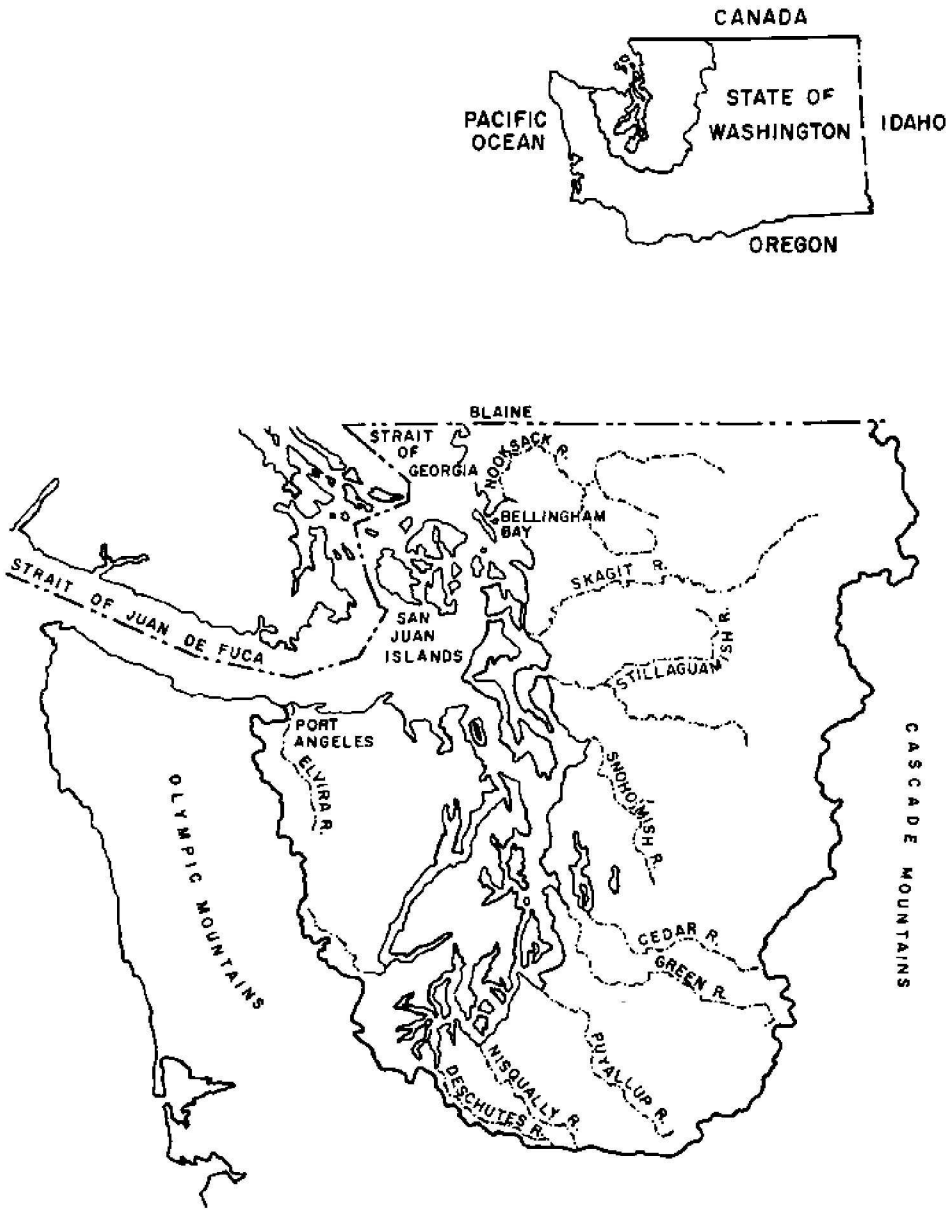
INTRODUCTION

Puget Sound, a large inland sea, is located in the northwest corner of Washington state. For the purposes of this case study, Puget Sound will be defined as that body of marine/estuarine water lying east and south of a line extending from Port Angeles to approximately Blaine, Washington. As Figure B.1 shows, this definition places Puget Sound entirely within US boundaries. This case study will attempt to review the past and present impacts of municipal point source discharges on Puget Sound, with specific reference to transport and mixing of discharged constituents, influences on biota, economic growth and aesthetics. Finally, some attempt will be made to assess future directions resulting from existing regulatory requirements and continuing planning efforts.

General Description of Receiving Waters

Puget Sound is the third largest sound in the United States. Consisting of approximately 6300 km^2 (2433 sq mi) of marine and estuarine waters, it is composed of a large number of distinct and semi-distinct bays and inlets and interspersed with more than 300 islands. The Sound extends approximately 200 km (124 mi) from Olympia in the south to the Canadian border in the north, and approximately 80 km (50 mi) at its widest point from the Strait of Juan de Fuca in the west to Everett, Washington in the east. The average width of Puget Sound, including Hood Canal however, is more nearly 50 km (30 mi). Puget Sound lies in a series of glacial basins bounded to the east by the Cascade Mountains and to the west by the Olympic Mountains. While only four of the Cascade Mountain peaks exceed 3000 m (9842 ft), both mountain ranges serve to protect the region from climatic extremes typical of its latitude at other locations. Winter air temperatures rarely fall below -10°C (14°F) at sea

Figure B.1 Puget Sound and adjacent waters



level or exceed 30°C (95°F) in the summer. Water temperatures, even in shallow embayments, exhibit similar moderate fluctuations ranging between 7.5°C (45.5°F) and 12°C (53.6°F) winter to summer. The temperature of the majority of the open Sound is even more constant ranging only from 8.4°C (47°F) to 10.5°C (51°F). As a consequence of these mountain barriers, annual precipitation varies from 50 to 125 cm (20 to 50 in) at sea level to over 500 cm (200 in) in the surrounding higher elevations. Approximately 75 percent of the precipitation typically falls between October and April (Puget Sound Task Force, 1971).

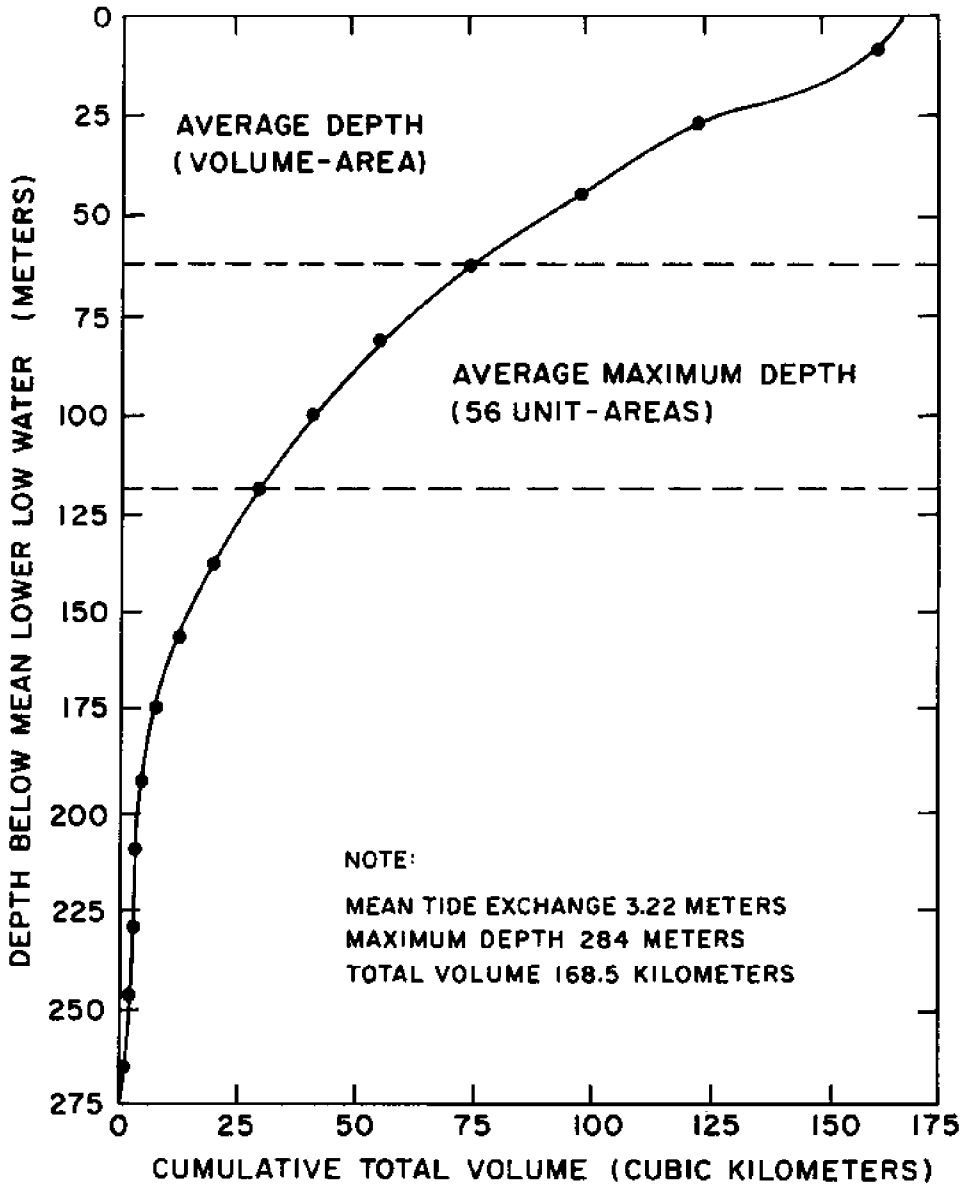
Puget Sound is a fjord-like basin with an average depth in excess of 100 m (330 ft). Depths of 150 to 250 m (492 to 820 ft) are common in the northern Sound while south of Port Gardner Bay, approximately mid-Sound, depths are more typically 50 to 100 m (164 to 329 ft) (Figure B.2). Occasional deep holes are found throughout the Sound, analogous to the topography of adjacent land areas. The maximum depth of Puget Sound, found in one such hole off Point Jefferson, is 284 m (932 ft) (University of Washington for METRO, 1953). The total volume of Puget Sound is 168.5 km^3 (40.5 mi^3). From both a depth and volume standpoint, Puget Sound is nearly three times greater than any estuarine system in the continental United States (Domenowske, 1974).

The tidal pattern in Puget Sound is one of the mixed type characterized by two highs and two lows daily. Port Angeles, at the western entrance to Puget Sound, has a mean tide range of 1.5 m (4.9 ft) and a diurnal tide range of 2.5 m (8.2 ft).

At the southern reaches of the Sound, mean tide range and diurnal range are more closely 3.5 and 4.5 m (11.5 and 14.8 ft) respectively. While tidal currents over sills and through narrow passes can be strong (measured up to 13 km/hr (8.1 mph) in Deception Pass) the majority of Puget Sound usually exhibits tidal currents of less than 2 km/hr (1.3 mph) (Crutchfield, 1969). The volume of water within the tidal prism (Mean High Water to Mean Lower Low Water) is approximately 8.1 km^3 (1.94 mi^3) or 5 percent of the total volume of the Sound (Stevens, et al., 1975).

Puget Sound is a positive estuary. The input of fresh water from all sources is greater than evaporation losses, resulting in a net outflow of water seaward. Stratified to varying degrees and depths depending on season and temperature, excess fresh water is primarily drained seaward at the surface, while a

Figure B.2 Volume of water by depth for Puget Sound



net influx of salt water occurs at lower depths. The primary source of salt water to Puget Sound is the Strait of Juan de Fuca with some also coming from the Canadian Strait of Georgia. Freshwater inputs include direct precipitation, runoff, and some groundwater sources (Collias, et al., 1974).

The combined drainage basin of Puget Sound is approximately 37,550 km^2 (14,500 mi^2) (Crutchfield, 1969). Figure B.1 outlines the drainage pattern for the Sound and delineates the boundaries for major river basins. Streams and rivers rising in the mountainous areas of the Puget Basin generally may be divided into two types: streams draining relatively low elevations, being wholly rain fed, which show one period of peak runoff annually; streams rising to higher elevations which show two periods of peak runoff as a consequence of both rain and melt water from snow fall and glaciers (University of Washington for METRO, 1953). More than a dozen major rivers, with a total average discharge of approximately 1,275 cubic meters per second (cms) (45,000 cubic feet per second (cfs)) flow to Puget Sound. Two of these, the Skagit and Snohomish Rivers, each separately carry approximately one-third of the total stream runoff to the Sound. Both enter the northern portion of Puget Sound through the Possession Sound system. The remaining one-third runoff is fairly well distributed between the remaining streams and rivers. In addition to the direct Sound-discharging streams, significant fresh water is received via the Strait of Georgia from the Fraser River in Canada whose mouth is approximately 16 km north of the US-Canadian border. The Fraser River is one of the largest rivers on the West Coast of North America, draining over 221,500 km^2 (85,500 mi^2) or six times the total drainage of Puget Sound (Stevens, et al., 1975).

Description and History of Population Growth

Puget Sound forms a unique haven from the severe winter storms which are born in the Northwest Pacific Ocean and Gulf of Alaska. Unlike the East Coast of North America, the West Coast has few major harbors and protected embayments. It is not surprising then that prior to the arrival of the European explorer, many marine-oriented Indian tribes were based around the Sound or on one of its many islands. The common names of many rivers, lakes, and counties today reflect this Indian heritage.

When Captain George Vancouver sailed into central and southern Puget Sound in 1792, naming the area after

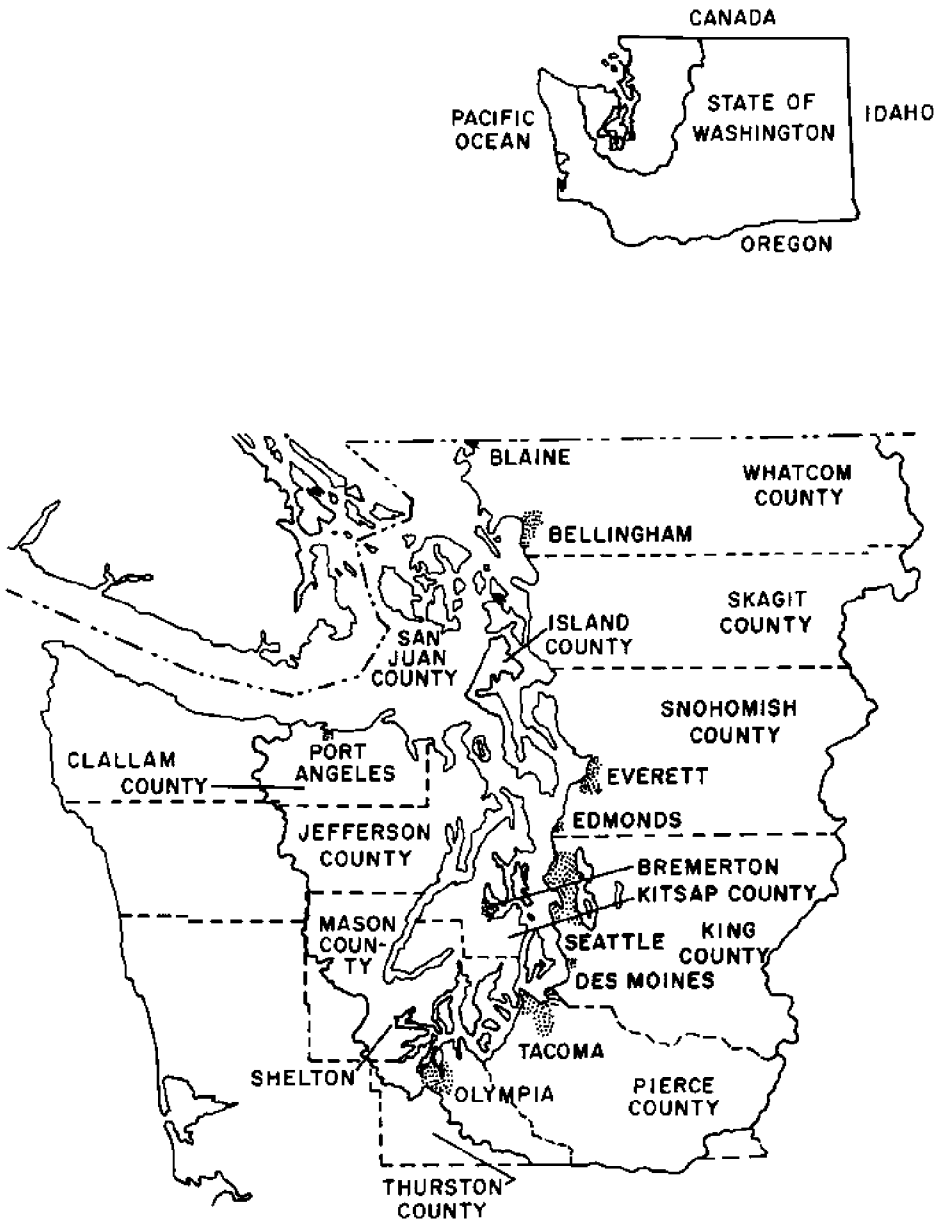
one of his explorer-lieutenants, the Indian population of the area was approximately ten thousand. Among the many tribes that inhabited the area, the largest and most influential were the Nisqually and the linguistically related Puyallup and Duwamish tribes in the southern Sound area and the Skagit and Lummi tribes of the north Sound area. Most of these tribes subsisted by harvesting the biota-rich waters of the Sound and hunting the many mammals of the nearby Olympic and Cascade Mountains (Swanton, 1952).

In 1832 the first permanent settlement by Europeans on Puget Sound was established by the Hudson Bay Company at Fort Nisqually on Nisqually Bay. Serving as a focal point of the trapping, fishing and later logging industries, Fort Nisqually attracted a primarily British population to the area. During the period following the establishment of Fort Nisqually (1838-42), the majority of Puget Sound was explored and mapped by the United States Exploring Expedition under the command of the oceanographer/geographer Lt. Charles Wilkes. In 1846, a treaty signed by Great Britain and the United States established a northern boundary (49th parallel) of the Old Oregon Territory, later to become the northern boundary of Washington State. In 1853, the Territory of Washington was established, followed by statehood in 1889 (Crutchfield, 1969).

With the establishment of the Washington Territory in 1853, the first population survey was taken. The local census reported 2,063 European-descent inhabitants in the area, most of whom lived in the southern reaches of the Sound. The population of the area grew rapidly thereafter, particularly during the periods of the California and Alaska gold rushes. The population of the Puget Sound area jumped from 25,000 in 1880 to over 600,000 in 1910, reaching 1,000,000 in 1949 (Power, 1979b). The populations of the area in 1979 was estimated to be in excess of 2,600,000, representing 65 percent of the total population of the State of Washington (Power, 1979a and 1979b).

Twelve counties border Puget Sound (Figure B.3). These counties are frequently divided into Northern (Whatcom, Skagit, San Juan and Island counties); Central (Kitsap, King, Pierce and Snohomish counties); and Western regions (Thurston, Mason, Clallam and Jefferson counties) (Puget Sound Governmental Conference, 1970; University of Washington for METRO, 1953; Water Resources Engineers for the EPA, 1974). Almost 80 percent of the population in the three regions is located in the three central counties of

Figure B.3 Puget Sound region: counties and major population centers



King, Pierce and Snohomish. Including the cities of Seattle, Tacoma, Bellevue and Everett, this area is the largest metropolitan area on the West Coast north of San Francisco.

Any study of the impact of municipal waste on Puget Sound should consider the population distribution around the Sound. Figure B.3 gives a rough indication of the distribution of population within the region, showing most of this population concentrated in a few cities predominantly on the east side of the Sound. All of the larger cities are located at the mouth of a major river, while a few smaller cities are positioned on rivers not far from the Sound.

More than 95 percent of the municipal discharges to Puget Sound come from ten cities (Table B.1). The impact of these cities constitutes the major domestic point source of water pollution in the area. Most of the cities in the region operate sewage collection and treatment systems within their municipal boundaries, although a few regional sewerage agencies or unincorporated districts have been formed.

Table B.1 Major Cities on Puget Sound (1979 pop. est.)

1.	Metropolitan Seattle	875,000
2.	Tacoma	156,500
3.	Everett	56,000
4.	Bellingham	43,760
5.	Bremerton	40,000
6.	Edmonds	27,000
7.	Olympia	26,500
8.	Port Angeles	17,025
9.	Anacortes	9,500
10.	Des Moines	7,000

The largest of these sewerage agencies, (METRO, The Municipality of Metropolitan Seattle) is the major water quality management agency for King County. METRO, formed in 1958 primarily to facilitate the cleanup of Lake Washington, Seattle's large urban lake, has developed a national reputation for its progressive wastewater management programs. METRO services nearly 500,000 customers, including significant commercial and industrial dischargers. Operating five waste treatment plants, METRO constitutes the largest single municipal discharger in the region. Further, one of these plants, the West Point Sewage Treatment Plant (STP) is

the largest single discharger to Puget Sound with average effluent flow in excess of 120 mgd. (Average dry weather flow 90-95 mgd). The five METRO waste treatment plants replaced 28 older facilities thereby ending the discharge of treated effluent into Lakes Washington and Sammamish and the discharge of untreated effluent into the Duwamish River, Elliott Bay and adjoining Puget Sound waters (METRO, 1968, 1974a. 1974b, 1977b and 1978a) (Figure B.4).

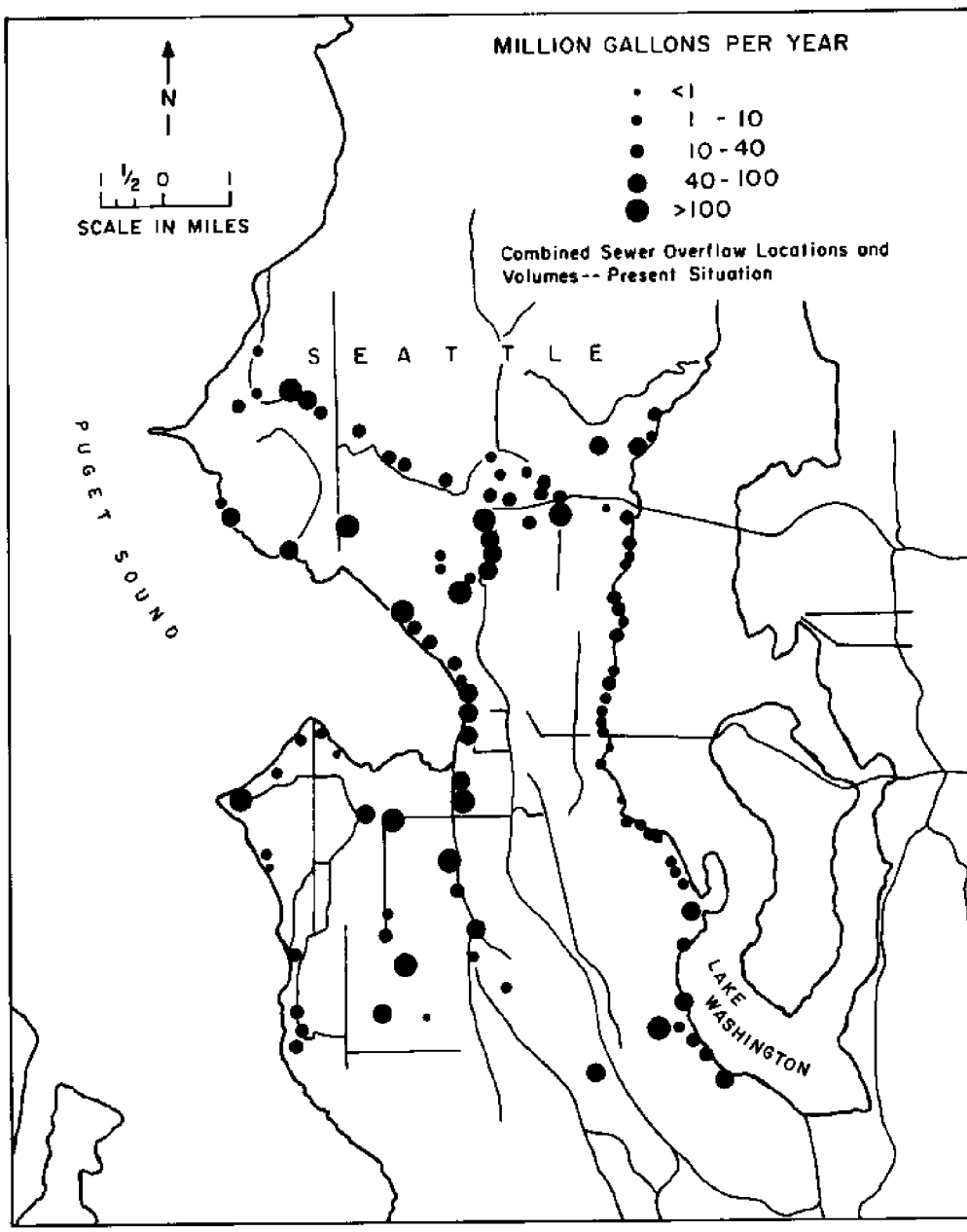
If there seems to be an emphasis on METRO data in this case history, it stems from the limited data available to the authors on the impacts of other municipal dischargers to Puget Sound. Because of its size and local government commitments, METRO has collected through consultants and its own resources a majority of the contemporary water quality data in the central Puget Sound region. The smaller municipalities/sewer districts on the Sound simply have not had the resources to conduct impact studies of the depth or duration of those done by METRO. In addition to METRO, the University of Washington, the Washington Department of Ecology (WDOE) and the NOAA have conducted significant studies in Puget Sound which have been utilized as data sources for this case history (Oceanographic Institute of Washington, 1974).

Economy of the Region

Puget Sound is now one of the faster growing areas in the nation. Its present population of over 2,600,000 is predicted to reach 3,000,000 by the year 2000 (Power, 1979a). This rapid growth is due, in part, to the area's strong economic base and its "liveability." The migration patterns of people moving out of the northeastern United States' industrial centers, in the mid-sixties also accounts for the rapid increases. The Boeing mini-recession in the early 1970s briefly interrupted that trend. Since that time, however, the high levels of employment and personal income common to the Puget Sound area have continued to attract new firms and individuals.

The economy of the region is largely tied to the land and marine resources of the area. Much of the originally forested lowland surrounding Puget Sound, especially on the eastern side has been cleared for urban, industrial or agricultural uses. While this land is generally fertile for a wide variety of crops, competition between commercial and agricultural uses is tending to push farming out of the flat river valleys immediately adjacent to the Sound. Some land has been

Figure B.4 Central Puget Sound and METRO discharges



cleared in the surrounding benchlands and foothills for agriculture which, like the lowlands, is largely used for intensive production of grass for dairy cattle, or for vegetable or berry crops. Approximately 75 percent of the land used for agricultural purposes is in rotation grass for livestock (Crutchfield, 1969).

The uplands and mountains surrounding Puget Sound primarily remain forested with largely coniferous growth, characterized by climax species of cedar and hemlock. These forests, most of which have been harvested at least once and are now in various stages of regrowth, support the large forest-product industries of western Washington. Forest products particularly lumber, plywood and pulp, and paper are major impacts on the region's economy. In 1978, over two billion dollars of forest products were shipped from the regions.

The economic base of Puget Sound rests firmly on wood product and transportation equipment manufacturing (Table B.2). The Seattle metropolitan area has one of the highest, if not the highest, ratios of employment in manufacturing to total employment among the nation's major metropolitan areas. Tacoma, Everett, Bellingham and Bremerton are also essentially industrial cities. Total non-agricultural employment for the Seattle/Everett/Tacoma metropolitan area is estimated to be more than 725,000 in 1979 (Washington State Employment Security Department, 1979). Further, with its many protected harbors in the Sound and its proximity to the Orient and Alaska, the Seattle metropolitan area has become a major seaport for both domestic and international shipping. This has promoted the growth of secondary industries such as international banking and manufacturing (Seattle Chamber of Commerce, 1980).

In addition to the forest products industry, transportation equipment is the largest, fastest-growing regional industry. Although the industry is dominated by the Boeing Company (aircraft, spacecraft production) and PACCAR (railroad, heavy equipment production) a number of smaller companies manufacture boats, trucks and other transportation-related equipment. Transportation equipment production also includes Puget Sound's four major shipyards and a large number of smaller shipyards and marine-related operations that serve both commercial fisheries and pleasure boaters.

Table B.2 Employment Groups: Seattle-Everett Metropolitan Area

	December, 1978
Total Non-Agricultural Employment	725,800
Manufacturing	152,900
Durable goods	123,800
Lumber and wood products	10,900
Furniture & fixtures	1,700
Stone, clay & glass products	3,500
Primary & fabricated metal products	9,700
Machinery (including electrical)	13,900
Transportation equipment	78,800
Other durable goods	5,300
Nondurable goods	28,900
Food & kindred products	11,100
Apparel	4,500
Paper & allied products	2,000
Printing & publishing	7,500
Chemicals & allied products	1,300
Other nondurable goods	2,500
Agricultural services & mining	2,800
Construction	43,100
Transportation, communications & utilities	48,500
Wholesale & retail trade	181,200
Finance, insurance & real estate	49,800
Services	132,300
Government	115,400

December 1978 statistics
Adapted from: Labor Area Summary, Washington State Employment Security Department

Puget Sound also contains the far west's largest and most diverse concentration of primary metals production. A copper smelter at Tacoma, an aluminum reduction plant near Bellingham, and a steel mill and lead refinery at Seattle (both fed by reclaimed metals) provide about 70 percent as much direct employment as the pulp and paper industry.

Fisheries and seafood processing, a major resource-based industry of Puget Sound, has commercial significance extending well beyond the limits of the local resource. Puget Sound is the traditional outfitting and financial focus of the Alaskan fishery; and boats based on Puget Sound work waters from the Gulf of Alaska to the Gulf of California. After an extended period of over-competition and declining catches, several factors have rejuvenated the fishery, including the two hundred-mile limit, and the emergence of markets for species other than salmon and halibut.

Military and other federal activities are also significant components of Puget Sound's economy. Together, Fort Lewis, McCord Field and Bremerton area naval facilities rival or exceed any private firm in providing employment. Furthermore, Seattle is the focus of federal government programs for the Pacific Northwest States and Alaska (Federal Region X).

The growth of industry within the region has had a major impact on water quality of the Sound, particularly in localized areas. Table B.3 depicts the 20 largest dischargers to the Sound in terms of flow and major pollutant loadings. As illustrated, 13 of the 20 largest dischargers are industrial: pulp and paper production, chemical manufacture, aluminum reduction, and oil refining. Present municipal and industrial water use in the Puget Sound area is approaching 2840 million liters per day (750 mgd). Of this amount, nearly two thirds of the supply use occurs in the Cedar-Green, Snohomish and Puyallup basins. Water used primarily for industrial applications is about 1,890 million liters per day (500 mgd), or about two thirds of the total usage. Most of the industrial water use is by the ten pulp and paper plants on the Sound who collectively generate a total wastewater discharge exceeding the average flow of all municipal plants in the region (Stevens et al., 1975).

Other significant sources of wastewater discharges include fish hatcheries and military bases. Collectively fish hatcheries and military bases discharge over 720 million liters per day (190 mgd) of municipal and/or process wastewater. While this case

Table B.3 Puget Sound Dischargers with Largest Effluent Volume*

		Flow mgd	BOD lb/D	TSS lb/D	N lb/D	P lb/D
1	West Point METRO STP Seattle	122.1	91,600	61,100	34,600	5,100
2	Scott Paper 2611 Everett	67.2	54,500	25,000	46,000	21,000
3	Georgia Pacific Bellingham 2611	56.2	54,000	24,000	46,000	21,000
4	Weyerhaeuser Everett 2421	2.0	1,800	2,900	300	200
	Weyerhaeuser Snohomish R. 2611	31.9	3,900	8,600	600	300
5	ITT-Rayonier Port Angeles 2621	32.3	77,000	34,958	55,000	24,970
6	Renton METRO STP	29.8	1,100	1,800	1,300	900
7	Weyerhaeuser Everett (Port Gardner Bay)	6.	10,500	5,000	7,000	3,200
8	St. Regis Paper Tacoma 2621	38	27,000	12,270	9,400	4,272
9	Hooker Chemical & Plastic Tacoma 2812	18.5	544	248	---	---
10	Hooker Chemical & Plastic 2621 Port Angeles	27.1	16,100	13,400	2,200	1,100
11	Tacoma STP	23.4	42,900	8,500	7,800	1,900
12	Bellingham STP	17.5	34,300	12,500	5,900	1,500
13	Crown Zellerbach Port Townsend 2611	16.7	10,000	4,540	4,800	2,180
14	Intalco Alum. Ferndale 3334	7.0	4,400	2,000	---	---
15	Alki METRO STP Seattle	11.8	4,400	2,800	3,900	1,000
16	Scott Paper Anacortes 2611	7.0	9,000	4,100	7,600	3,450
17	NW Steel Seattle 3312	7.9	4,400	2,000	---	---
18	Everett STP	7.8	1,800	3,600	1,300	700
19	Edmonds STP	4.5	4,900	7,500	1,500	400
20	Atlantic Richfield Ferndale 2911	2.5	1,000	450	1,500	(100) (72)

*Adapted from Stevens, Thompson & Runyan, Inc., 1975 and current NPDES permits and operating records.

study is designed to document the impact of municipal discharges to Puget Sound, the significance of industries, fish hatcheries or military bases discharging to the same local receiving waters as the municipalities, is often greater.

Not all industries in the area constitute a polluting use of Puget Sound water. Several industries and activities, important in the region's economy, require high quality, non-contaminated estuarine water and are therefore theoretically in competition with the wastewater discharging industries and municipalities for use of the available resource. Commercial and recreational fishing, swimming, boating and shellfish harvesting are extremely valuable to the region's economy (Aaron et al., 1979). The salmon, oyster and Dungeness crab harvesting industries of Puget Sound are world famous and depend to a large extent on the maintenance of high quality marine and estuarine waters.

It is in this setting of varied and valuable natural resources that a rapidly expanding population has attempted to minimize and/or balance the impacts of municipal and industrial waste disposal.

WATER RESOURCES AND WASTEWATER SYSTEMS

Most of the municipal and industrial waste effluents in the Puget Sound region discharge to estuarine or marine waters. In all, over 28 cms (1,000 cfs) of waste effluents are discharged into a saltwater system with an enormous range of physical dispersal properties. Thus, potential effects of discharges are highly site specific.

River systems discharging to Puget Sound waters contribute approximately 1400 cms (50,000 cfs) as an annual average; through the year flows vary widely depending on seasonal rainfall and melting snow. Table B.4 summarizes flow characteristics of the region's twelve major river systems. These twelve river systems, identified in Figure B.1, account for over 75 percent of the total runoff to Puget Sound.

Seasonal variations in river flow and water temperature are significant to water quality considerations because warm water has a lower capacity for absorbing waste loadings. Discharges to river estuaries such as the Duwamish and Puyallup are of greatest concern in low flow periods. During warm periods in late summer and early autumn, the colder saltwater entering at depth with the tide fails to mix

Table B.4 Principal Rivers of the Puget Sound Region (Puget Sound Task Force, 1971)

Basin	River	Drainage Area (sq. miles)	Discharge (cfs) Minimum Daily	Maximum Daily	Average Annual
Nooksack-Sumas	Nooksack River near Lynden	648	595	46,200	3,699
Skagit-Samish	Skagit River near Mt. Vernon	3,093	2,740	144,000	16,490
Stillaguamish	South Fork Stillaguamish River near Granite Falls	119	55	32,400	1,064
Snohomish	Snohomish River near Snohomish	1,714	--[a]	136,000	9,500[b]
Cedar-Green	Cedar River at Renton	186	56	6,640	711
Cedar-Green	Green River at Tukwila	440	195	12,100	1,462
Puyallup	Puyallup River at Puyallup	948	306	57,000	3,364
Nisqually-Deschutes	Nisqually at McKenna	517	20	25,700	1,415
Nisqually-Deschutes	Deschutes River near Rainier	90	16	5,620	263
West Sound	Skokomish River near Potlatch	227	125	27,000	1,183
Elwha-Dungeness	Elwha River near Port Angeles	269	10	41,600	1,487
Elwha-Dungeness	Dungeness River near Sequim	156	77	8,400	373

[a]Discharge below 10,000 cfs not generally computed due to large tidal fluctuations.

[b]Accurate continuous streamflow records in the Snohomish River have not been possible because river stages are affected by tidal fluctuations. Projection of upstream records, however, suggests that the mean annual discharge of the entire Snohomish River system is probably about 9,500 cfs.

with surface warm, fresh waters. Resultant stratified conditions can contribute to less favorable water quality which in turn may impact migratory salmon and other indigenous fish species, particularly at river mouths and restricted embayments.

The region's rivers are the primary source of water supply. More than two million people and numerous industries depend directly on these rivers; less than 15 percent of the area's water supplies are from groundwater sources. Because the major municipal watersheds are well protected, most municipally-supplied water in the area receives only chlorination and fluoridation treatment. Filtration is under study in several areas because of new turbidity standards set by the EPA and the significant amount of natural asbestos fibers in the watershed areas. Overall municipal water quality, however, is very good. Several of the area's major water supplies are extremely soft and therefore quite corrosive. In the Seattle area the soft waters of the Cedar and Tolt systems have been linked with extensive damage to plumbing systems; corrosion releases copper and zinc into the water and eventually into the wastewater discharges (METRO, 1975). Thus, the water supplies themselves are a source of at least two heavy metals as well as asbestos fibers. Copper, zinc and asbestos are each listed as a priority pollutant by EPA because of their potential toxicity.

River flows include substantial amounts of runoff from urban areas. Urban drainage carries a range of pollutants including pesticides from agricultural and horticultural activities, and various contaminants associated with erosion, animal excreta, industrial yard drainage, as well as oils and other accumulations from paved surfaces. Other pollutants enter Puget Sound directly from ships and as atmospheric fallout; examples from this latter category include arsenic and other metals from a large smelter in Tacoma as well as lead from vehicle emissions throughout the region.

Puget Sound receives effluent from over 480 municipal and industrial dischargers, as well as several fish hatcheries and federal installations (Seattle Water Department, 1968). Treatment and effluent disposal practices at the industrial plants vary; the pulp mills rely on both inplant controls and end-of-the-pipe treatment. Various forms of secondary treatment, e.g., aerated lagoons and activated sludge systems, are common among the pulp mills. In contrast, most municipal treatment plants employ primary treatment with disinfection. Table B.5 provides a list

Table B.5 Major Municipal Discharges to Puget Sound Area Waters[a]

Agency	Type of treatment	Discharge location	Flow (mgd) NPDES limit	Flow (mgd) Average	Pollutant loading pounds/day BOD	Pollutant loading pounds/day SS
Anacortes	primary		2.2	0.6	620	550
Bellingham	primary	Bellingham Bay	18	7.5	7,000	4,250
Bremerton						
Charleston	primary	Sinclair Inlet	3.5	2.8	1,730	1,430
Manette	primary	Port Washington Narrows	5	3.1	2,940	1,680
Des Moines	primary	Puget Sound	6	3.1	--	--
Edmonds	primary	Puget Sound	10	4.8	5,510	3,420
Everett	secondary	Snohomish River	31	9.7	1,170	2,170
METRO						
Alki	primary	Puget Sound	15	8.6	7,600	8,200
Carkeek Park	primary	Puget Sound	7.2	3.5	3,700	4,800
Renton	secondary	Duwamish River	50	35.5	--	--
Richmond Beach	primary	Puget Sound	3.2	1.5	2,500	2,500
West Point	primary	Puget Sound	195	118	122,000	185,000
Olympia[b]	secondary	Budd Inlet	20	7.3	2,950	2,950
Port Angeles	primary	Straits of Juan de Fuca	4.8	1.8	3,040	2,720
Shelton	primary	Oakland Bay	3.3	2.1	480	750
Southwest Suburban						
Salmon Creek	primary	Puget Sound	7	2.7	--	--
Miller Creek	primary	Puget Sound	8	2.1	--	--
Tacoma						
Commencement Bay	primary	Commencement Bay	10	6.5	--	--
Puyallup	primary	Puyallup River	78	20.8	--	--
Tacoma Narrows	primary	Tacoma Narrows	3	1.4	--	--

a. Based on Stevens, Thompson & Runyan, Inc., 1975
b. Revised NPDES permit conditions 1982

of 20 of the largest municipal dischargers together with their NPDES permit flow limits, average flow data and estimated BOD and TSS loads. Nearly all of these facilities provide primary treatment prior to discharge to saltwater.

Public Law 92-500 passed in 1972 required all municipal treatment plants be upgraded to a minimum of secondary treatment. While some cities complied, considerable resistance developed against this aspect of PL 92-500 by many West Coast marine dischargers. Efforts of these discharge agencies, including the Municipality of Metropolitan Seattle, led to amendment of the Federal Water Pollution Control Act to allow a process for obtaining secondary treatment waivers. This amendment passed the Congress in December 1977; waiver regulations (Section 301 (h)) were finalized on June 15, 1979. A number of agencies in the Puget Sound region, including Seattle METRO and Tacoma, filed formal applications for secondary treatment waivers. (As of October 1983, when this book was in final stages of preparation for publication, Seattle METRO's Richmond Beach and West Point plants had been tentatively approved, and the Duwamish plant denied; Tacoma's Central and North End plants had been tentatively denied and the West Slope plant tentatively approved.)

For purposes of this case study, the six largest municipal dischargers are described. These include the Municipality of Metropolitan Seattle, and the cities of Tacoma, Everett, Bellingham, Olympia and Bremerton. These six cities operate a total of thirteen treatment plants, only two of which provide secondary treatment. These plants contribute over 80 percent of the total municipal discharge to Puget Sound.

Municipality of Metropolitan Seattle

Seattle METRO operates the largest wastewater collection, treatment and disposal system on Puget Sound. Serving roughly 1140 km^2 (710 mi^2) wastewater is treated at plants at Richmond Beach, Carkeek Park, West Point, Alki, and Renton (Figure B.4). The first four provide primary treatment and discharge their effluent into the marine waters of Puget Sound. Renton, however, provides secondary treatment and discharges its effluent into the Duwamish-Green River system. Service areas of these METRO facilities vary in character. The West Point and Alki sewerage systems, include combined sewers within the the city of Seattle which receive substantial

volumes of urban runoff during wet weather; as a consequence, overflows of untreated sewage occur at approximately 110 points around the city of Seattle. These localized nearshore discharges vary depending on storm and local sewer characteristics; however, overflow frequencies of 40 per year are common for some overflow points. The geographic distribution and relative magnitude of these occurrences are illustrated in Figure B.5. Past efforts to reduce overflows include use of computer-regulated sewer system storage by METRO and sewer separation by the City of Seattle.

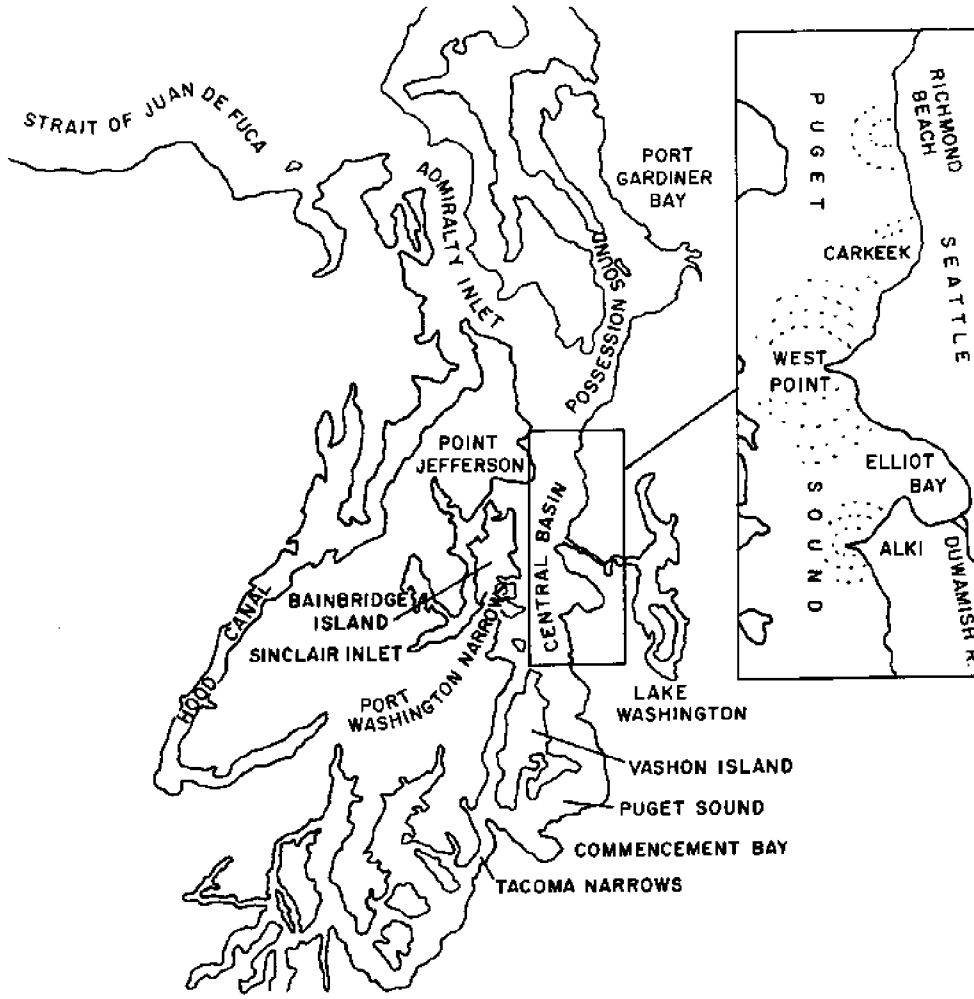
METRO's West Point treatment plant, the region's largest, has a dry weather capacity of 125 mgd; however, the plant can process up to 375 mgd during wet weather periods. Accordingly, much of the wet weather flow from the combined system can be treated if it can be transported to the plant. Following primary treatment, the effluent is disinfected with chlorine and discharged through a 230 cm (96 in) diameter submarine outfall with a multiport diffuser system which reaches a maximum depth of 73 m (240 ft) at a distance of 1112 m (3,650 ft) from shore. Digested sludge was discharged through this outfall until December 1972.

The Alki primary treatment plant has hydraulic limitations and cannot effectively treat flows over about 10 mgd although peak wet weather flows occasionally exceed 70 mgd in that area; as a result overflows occur at several points along the west Seattle shoreline, see Figure B.5. The Alki 104 cm (42 in) diameter outfall discharges at a depth of 25 m (80 ft), 420 m (1400 ft) from shore.

The Richmond Beach and Carkeek Park service areas are generally residential in character. The Richmond Beach system is relatively free of wet weather influences; the 3.2 mgd capacity primary plant performs well. The average effluent flow of 1.2 mgd is discharged through a 61 cm (24 in) diameter outfall at a depth of 30 m (100 ft). The end of the outfall is a 12 head diffuser 4877 m (16,000 ft) from shore. The Carkeek Park system is in need of sewer system rehabilitation. Peak flows hydraulically overload the 3.4 mgd facility during storm periods. The Carkeek Park outfall is an 85 cm (33 in) diameter pipe with multiport diffusers which discharge 640 m (2,100 ft) from shore at a maximum depth of 60 m (200 ft).

Renton area sewers are relatively new and in good condition so that infiltration problems are minimal. Industrial wastes in the Renton system include diverse manufacturing plants and a major food processor, Safeway. The Renton area is experiencing rapid growth;

Figure B.5 Seattle combined overflow sewers



as a result, this 36 mgd activated sludge plant is virtually at capacity. Discharge is directly to the Green River after disinfection by chlorination and dechlorination with sulfur dioxide. Effluent dilution exceeds 10:1 except during unusually dry periods, when minimum river flows fall below 200 cfs. Sludges from the Renton plant are currently transferred to the West Point plant for processing (digestion).

Following an extensive review of alternatives, METRO called for the abandonment of the Alki and Carkeek STPs with these wastewaters to be transported to West Point. METRO applied for secondary treatment waivers for its remaining Puget Sound discharge sewage treatment plants (West Point and Richmond Beach) (METRO, 1975 and 1977a).

City of Tacoma

Community sewers in Tacoma were begun in 1880. Between 1928 and 1946, local sewer construction consisted of a combined system but since 1946 the city has used separate sewer collection systems. Tacoma, a major city with a population slightly in excess of 100,000, maintains over 1,150 km (720 mi) of sewer lines ranging in size from 20 to 140 cm (8 to 54 in) in diameter, 20 pump stations and 3 primary treatment plants. The largest of these plants (average flow 20.8 mgd) is the central treatment plant, which discharges through a 122 cm (48 in) diameter outfall to a tidal reach of the Puyallup River. This facility is hydraulically overloaded during heavy rainfall periods, and there are numerous in-system overflows discharging raw sewage. The city has proposed relocating this discharge into Commencement Bay. Tacoma's two smaller treatment plants discharge primary effluent directly to saltwater. One of these plants (average flow 6.5 mgd) discharges through a 96 cm (36 in) diameter outfall 210 m (100 ft) long in 25 m (80 ft) of water at the north end of Commencement Bay. The other plant (average flow 1.4 mgd) discharges through a 96 cm (36 in) diameter outfall 86 m (280 ft) long into 24 m (75 ft) of water to the Tacoma Narrows.

The port area of Tacoma includes reaches of Commencement Bay known as the Hylebos Waterway. While it is known that this waterway is extensively impacted by industrial and commercial shipping discharges and groundwater runoff, few studies have been made to assess the magnitude of water quality problems. Most researchers, however, believe the Hylebos Waterway to be similarly contaminated as the Duwamish Waterway of Seattle (Consoer, et al., 1975; Tacoma Department of Public Works, 1975).

City of Everett

The city of Everett (population 52,000) provides sewer service to an area of approximately 10,500 acres. The city's sewer collection system consists of 290 km (180 mi) of sewer mains and laterals. The first sewers were built around 1900; most of the system was built prior to 1920 and after 1960. Some of the older system is severely damaged. Thirty pump stations and 32 major regulators are used to maintain and regulate flows. Eighteen overflow outfalls are used in wet weather as about one-half of the system has combined sewers. Five of these outfalls discharge raw sewage directly into the Snohomish River.

Influent wastewater is composed mostly of domestic sewage and wastes from light industry. Flows to the plant seldom exceed 10 mgd during dry weather but exceed 50 mgd during storms. Treatment is provided also by a large lagoon complex, including an aerated lagoon and two polishing ponds which discharge directly to a tidal reach of the lower Snohomish River (CH_2M Hill, 1979); US Public Health Service, 1967).

City of Bellingham

Sewer systems within the city of Bellingham (population 45,000) are very old. Many Bellingham sewers have excessive groundwater infiltration, blockages by roots, and sedimentation of solids during low flow. Excessive flows during wet weather are due to combined sewers and direct infiltration of surface runoff into the sanitary sewer system. While separate storm sewers have been constructed to serve Bellingham, some of the service area is still served by combined sewers (CH_2M Hill, 1977; Cornell, et al., 1970).

Solids which deposit in the sewers during low flow periods are periodically flushed to the treatment plant during wet weather periods. These slug loadings cause operational difficulties at the treatment plant. The plant was constructed in two stages, the initial plant sized for 4.5 mgd was built in 1947, expanded to 11 mgd in 1960 and expanded again in 1961. The present modernized plant has a design capacity of 18 mgd. Bellingham's existing treatment complex povides primary treatment prior to discharge through a 150 cm (60 in) deepwater outfall into Bellingham Bay. The outfall line extends approximately 750 m (2500 ft) offshore; the last 150 m (500 ft) of which is a multiport diffuser.

City of Olympia

The city of Olympia has a population of approximately 26,500. The larger metropolitan area population, however, is approximately 40,000. Served by a single sewage treatment plant discharging treated waste to Budd Inlet, the sewage system for the area is a mixture of combined and separate sewers. A recent sewer rehabilitation and storm water sewer construction program has eliminated many combined sewers such that now storm water inputs occur mainly in the downtown, central district. Anticipated disruption to this core area due to sewer construction, however, has caused the city, for at least the time being, to postpone any further sewer separation projects.

The 9.1 mgd design capacity primary treatment plant has been rebuilt several times; the last rebuild occurring in 1974-75 with the addition of increased primary clarification capacity. Dry weather flows average 6 to 6.5 mgd and wet weather inputs average 11-12 mgd. Above the design capacity of the plant, storm water inflows receive only chlorine disinfection before discharge. Olympia has committed to secondary treatment and construction is presently underway for a cryogenic oxygen activated sludge plant of approximately 15 mgd capacity, employing ozone disinfection and anerobic solid digestion. Sludge from the plant will be removed by centrifugation and land filled.

The present discharge employs two outfall/ diffuser lines depending upon the tidal stage of the receiving waters. The 78 cm (30 in) primary outfall used mainly during low tide extends 1460 m (4800 ft) into 5.7 m (19 ft) depth at Mean Lower Low Water (MLLW). The auxiliary outfall is 120 cm (48 in) in depth diameter and discharges 760 m (2500 ft) from shore into 2.7 m (9 ft) of water. Both outfalls are fitted with multiport diffusers (Kramer, et al., 1976a).

City of Bremerton

The city of Bremerton is served by 182 km (113 mi) of sewers, of which approximately 25 percent are combined. Combined sewers are largely in the older downtown area. Overflows occur at numerous pump stations due to inflow from combined sewers and now illegal residential drain connections.

The city is served by two primary treatment facilities, the Manette plant and the Charleston

plant. The Manette plant is the oldest and serves combined sewer areas in established residential and downtown areas and industrial sources including a portion of the Puget Sound Naval Shipyard. The plant provides primary treatment with anaerobic digestion of sludges; disinfected effluent is discharged at a depth of 7.6 m (25 ft) through a 51 cm (20 in) diameter outfall extending 137 m (450 ft) into Port Washington Narrows. Sludges from the Manette plant are hauled to the Charleston plant for digestion, drying and disposal. Major sewage flows to the Charleston plant originate from the Puget Sound Naval Shipyard, septic tank pumps and combined sewers. Following primary treatment, the disinfected effluent is discharged to Sinclair Inlet through a 96 cm (36 in) diameter outfall terminating in a diffuser in about 9 m (30 ft) of water (CH2M Hill, 1978a).

RECEIVING ENVIRONS

Physical Description

Puget Sound is a small portion of a large marine complex including the Canadian Strait of Georgia and the Strait of Juan de Fuca. The entire marine system is composed of many interconnected bays, inlets, passages and channels which can be divided into nine distinct and semi-distinct bodies of water: Georgia Strait, San Juan Archipelago, Bellingham Bay, Possession Sound, Hood Canal, Southern Puget Sound, the Central Basin, Admiralty Inlet and the Strait of Juan de Fuca (Figure B.1) (Friebertshauser et al., 1971).

Only four regions, Central Puget Sound Basin, Bellingham Bay, Possession Sound, and South Puget Sound contain significant population centers. Of these, the Central Basin, extending from the south end of Admiralty Inlet to the north end of the Tacoma Narrows, approximately 80 km (50 mi) long by 8 km (5 mi) wide, alone receives major municipal wastewater discharges. The Central Basin is divided into several suboceanographic regions (Figure B.4). Four of these regions, Commencement Bay, Sinclair Inlet, Elliot Bay and the waters west of Bainbridge Island receive significant volumes of treated municipal wastewaters either directly or indirectly through adjacent stream discharge. Major municipalities on the eastern side of the Central Basin include Seattle, Tacoma and Edmonds. Bremerton and Port Orchard are located on the western

side of the Sound (Pacific Northwest River Basins Commission, 1971a and 1971b; Puget Sound Task Force, 1971).

Southern Puget Sound consists of all waters south of the Tacoma Narrows. The South Sound is composed of many finger-like inlets which produce approximately half of the yearly commercial harvest of shellfish in Washington State. Bordered by few towns and small cities, the major municipal dischargers to the South Sound are Olympia, Fort Lewis and Shelton.

Bellingham Bay and an arm of Possession Sound called Pt. Gardner Bay each receive municipal wastewater from one significant population center, Bellingham and Everett, Washington respectively. Several other small cities such as Port Angeles and Anacortes are located on the south shore and near the northeast end of the Straits of Juan de Fuca respectively.

Chemical Description

Water quality conditions in the majority of Puget Sound are generally excellent. Only in localized, typically nearshore areas with limited mixing, are waters measurably degraded from industrial, municipal or nonpoint pollutants (USEPA, 1979).

Surface salinities vary from near zero at the confluence with major rivers to over 32 parts per thousand in the Straits of Juan de Fuca (Duxbury, 1978). Puget Sound typically has high concentrations of dissolved oxygen in the near-surface waters, frequently exceeding saturation (Collias, 1974). These high values result from reaeration at the surface and from photosynthesis by phytoplankton in the upper layers. There is, however, typically an oxygen minimum at the bottom of the Central Basin of Puget Sound and a seasonal anoxic layer in Hood Canal. Both of these conditions are maintained by water column stratification and a resulting lack of vertical mixing. The South Sound may also have seasonal anoxic/hypoxic areas, particularly at depth.

The waters of Puget Sound are high in phosphate, particularly in the deeper more saline waters as a result of upwelled ocean water which is carried into the Sound at depth (Collias, 1977). Typically, the phosphate concentration of these deep waters may exceed 25 microgram-atoms/liter.

Occasionally nitrogen may be a limiting factor in determining algal growth, particularly in the summer months, as it has been shown to be temporarily depleted in the surface waters, during bloom conditions. Water column stability is also an important factor and during the winter, available light is most likely the limiting factor in phytoplankton production (Stevens et al., 1975).

Biological Description

Puget Sound supports a wide variety of marine fish and shellfish species. Commercial and sport fisheries occur in the saltwaters of the Sound and in the estimated 4,500 km (2,800 mi) of streams throughout the region which are utilized by anadromous fish for spawning and rearing. These migratory fish include a number of salmon species (chinook, coho, sockeye, pink and chum) as well as several species of sea run trout (Puget Sound Task Force, 1970; Washington Department of Fisheries, 1978).

The most significant production of salmon and trout comes from the Green, Puyallup, Skagit, Nisqually, and Snohomish Rivers. Several of these rivers run through highly populated areas emphasizing the potential conflict between high quality water use for fish runs and alternative commercial/industrial and municipal uses. Table B.6 depicts the estimated commercial catch of major fish and shellfish species in Puget Sound for 1977 (Washington Department of Fisheries, 1979).

Puget Sound currently supports a thriving fishery for shellfish, principally oysters, hardshell clams, soft-shell clams, geoducks, crab and shrimp. During 1966, more than 15 million kg (33 million lbs) of these species were taken commercially with crab and shrimp each contributing roughly 4.6 million kg (10 million lbs). Sport fishing accounts for an estimated 1.8 million kg (4 million lbs) of additional shellfish catch annually.

Like the fin fisheries, conflicts of potential water use and maintenance of high quality waters exist between the shellfish industry and other industries/agriculture/municipalities. These conflicts have been most evident in basins with the least mixing such as in the South Sound between oyster-rearing operations and adjacent industries and municipalities which discharge wastewater. Shallow embayments of the South Sound have also been the location of the most

Table B.6 1977 Commercial Fish Catch in Puget Sound and Adjacent Waters (Washington Department of Fisheries, 1979)

Species	Approximate Pounds
Chinook Salmon	4,823,636
Chum Salmon	4,839,746
Pink Salmon	12,848,927
Coho Salmon	8,801,192
Sockeye Salmon	11,227,831
Dover Sole	1,751,668
English Sole	2,487,392
Petrale Sole	971,519
Flounder	1,232,473
Black Cod	1,279,235
Ling Cod	1,927,630
True Cod	8,433,110
Rockfish	13,598,258
Pacific Ocean Perch	4,437,575
Slope Rockfish	830,981
Other bottomfish	985,870
Clams & Mussels	3,126,599
Geoduck Clams	8,643,948
Oysters	2,365,632
Crab	2,386,486
Shrimp	1,229,214
Octopus & Squid	16,656
Other shellfish	967,179
Aquaculture Fish	228,017
Albacore Tuna	461,759
Halibut	1,006,644
Herring	6,665,303
Hake	3,620,575
Dogfish Shark	5,428,282
Other fish	916,193

extensive "red tide" episodes. While there has been no defined relationship between the blooms of these toxic planktonic organisms and industrial or municipal discharges, the "tide" has on occasion significantly impacted the shellfish industry by closing areas to harvest and/or causing the mortality of shellfish larvae.

In addition to the commercial and sport fish and shellfish, a diverse population of invertebrates, fish and marine mammals inhabits Puget Sound. This diversity along with significant biomass of phyto and zooplankton, attests to the high quality waters of the Sound and its variety of unique habitats.

Water Quality Standards

In compliance with state law and EPA regulation, the Washington Department of Ecology has established Water Quality Standards and assigned classifications for the marine and surface water of the state (Washington Department of Ecology, 1977). Quality criteria have been set up in conformance with present and potential water uses and in consideration of the existing natural water quality potential. As such, they reflect to a reasonable measure the high quality of existing surface waters in the Sound. While water characteristics in most regions of Puget Sound routinely better their assigned classification, there are occasions when industrial and municipal discharges as well as naturally occurring phenomena, such as stratification, upwelling, storm flows, etc., produce conditions which cause one or more of the established criteria to be violated (US Department of Commerce, 1978). An outline of the Water Quality Standards for marine waters is found in Table B.7. Tables B.8 and B.9 list the uses to be protected for each classification and assign classifications to the marine/estuarine waters of Puget Sound respectively.

IMPACTS ON RECEIVING WATERS

Pollutant Properties

Effects of pollutants discharged to Puget Sound are determined by the properties of the pollutant and the way the pollutant enters and traverses the system. Nearshore discharges of persistant toxicants having bioaccumulation properites are far more likely to be a problem than biodegradable wastes discharged at depth

Table B.7 Washington State Water Quality Standards (Washington Department of Ecology, 1977)

Class for Surface Marine/Estuarine Waters

Parameter	AA (Extraordinary)	A (Excellent)	B (Good)	C (Fair)
Fecal Coliform	median less than 14 organisms/100ml	median less than 14 org./100ml	median less than 100 org./100ml	median less than 200 org./100ml
Dissolved Oxygen	greater than 7.0 mg/l	greater than 6.0 mg/l	greater than 5.0 mg/l or 70% saturation	greater than 4.0 mg/l or 50% saturation
Temperature	less than 13.0°C	less than 16.0°C	less than 19.0°C	less than 22.0°C
pH	7.0 - 8.5	7.0 - 8.5	7.0 - 8.5	6.5 - 9.0
Turbidity	shall not exceed 5 NTU over background*	shall not exceed 5 NTU over background*	shall not exceed 10 NTU over background*	shall not exceed 10 NTU over background*

*when background is 50 NTU or less

Table B.8 Characteristic Uses to be Protected (Washington Department of Ecology, 1977)

The following is a noninclusive list of uses to be protected by the various classifications for marine surface waters:

	Watercourse Classification			
	AA	A	B	C
Fisheries				
Salmonid				
Migration	M	M	M	M
Rearing	M	M	M	
Spawning				
Warm Water Game Fish				
Rearing				
Spawning				
Other Food Fish	M	M	M	
Commercial Fishing	M	M	M	
Shellfish	M	M	M	
Wildlife	M	M	M	
Recreation				
Water Contact	M	M		
Boating and Fishing	M	M	M	M
Environmental Aesthetics	M	M	M	M
Water Supply				
Domestic				
Industrial	M	M	M	M
Agricultural				
Navigation	M	M	M	M
Log Storage & Rafting	M	M	M	M
Hydro-power				

Table B.9 Classification of Marine/Estuarine Waters of Puget Sound (Washington Department of Ecology, 1977)

	Class
1. Bellingham Bay east of a line bearing 185° true from entrance of boat basin (light No. 2), except as otherwise noted.	B
2. Bellingham Bay, inner, easterly of a line bearing 142° true through fixed green navigation light at southeast end of dock (approximately 300 yards northeast of bell buoy "2") to the east boat basin jetty.	B
3. Budd Inlet south of latitude 47° 04' N. (south of Priest Point Park).	B
4. Commencement Bay from south and east of a line bearing 258° true from "Brown's point" and north and west of line bearing 225° true through the Hylebos waterway light.	A
5. Commencement Bay, inner, from south and east of a line bearing 225° true through Hylebos Waterway light except the city waterway south and east of south 11th Street.	B
6. Commencement Bay, city waterway south and east of south 11th Street	C
7. Drayton Harbor, south of entrance.	A
8. Dyes and Sinclair Inlets west of longitude 122° 37'W	A
9. Elliott Bay east of a line between Pier 91 and Duwamish Head	A
10. Everett Harbor east of longitude 122° 13' 40" W and southwest of a line bearing 121° true from light"4" (Snohomish River mouth).	A
11. Everett Harbor, inner, north and east of a line bearing 121° true from light "4" (Snohomish River mouth).	B
12. Guemes Channel, Padilla, Samish and Bellingham Bays east of longitude 122° 39' W and north of latitude 48° 27' 20" N, except as otherwise noted.	A
13. Hood Canal	AA
14. Mulilteo and all North Puget Sound west of longitude 122° 39' W (Whidbey, Fidalgo, Guemes and Lummi Island), except as otherwise noted.	AA
15. Oakland Bay west of longitude 123° 05' W (inner Shelton Harbor).	B
16. Port Angeles south and west of a line bearing 152° true from buoy "2" at the tip of Ediz Hook.	A
17. Port Gamble south of latitude 47° 51' 20" N	A
18. Port Townsend west of a line between Point Hudson and Kala Point	A
19. Possession Sound, south of latitude 47° 57' N	AA
20. Possession Sound, Port Susan, Saratoga Passage, and Skagit Bay east of Whidbey Island and longitude 122° 38' 35" W (bridge) between latitude 47° 57' N (Mukilteo) and latitude 48° 27' 20" N (Similk Bay), except as otherwise noted.	A
21. Puget Sound through Admiralty Inlet and South Puget Sound, south and west to longitude 122° 52' 30" W (Brisco Point) and longitude 122° 51' W (northern tip of Hartstene Island).	AA
22. Sequim Bay southward of entrance	AA
23. South Puget Sound west of longitude 122° 52' 30" W (Brisco Point) and longitude 122° 51' W (northern tip of Hartstene Island, except as otherwise noted).	A
24. Strait of Juan de Fuca	AA

through multiport outfall systems designed to optimize dispersal in strong offshore currents.

For example, a transformer spill of PCBs into the Duwamish River estuary in the early 1970's has been blamed for contaminating river sediments such that the Duwamish PCB levels are reportedly among the highest in the nation. There is concern that observed fish tissue (Starry flounder and English sole) abnormalities may be a long-term legacy of this incident.

On the other hand, discharge of BOD in primary treated wastewater from METRO's West Point plant has been shown to have minimal effect on dissolved oxygen resources of the receiving system. Maximum decreases of 0.5 mg/l DO have been measured in the vicinity of the outfall on infrequent occasions (USEPA Region X, 1978). These results obtained from several independent research studies, attest to the high degree of mixing in the vicinity of the West Point outfall (Collias, 1977; Environmental Quality Analysts, 1975).

Frequently, pollutant impact in nearshore metropolitan areas has been inadequately researched or considered. Yet much of the pollutant load to Puget Sound enters this zone as urban drainage either through direct runoff or as combined sewer overflows from many of the municipal sewerage systems. Combined sewer overflows are a particular concern in the Seattle area as well as in many of the other larger Puget Sound cities (Metropolitan Engineers for the Municipality of Metropolitan Seattle, 1969; METRO, 1978; US Public Health Service, 1958.)

Studies of municipal storm water discharges reveal significant localized pollution effects, as shown in the following example from the METRO system: Example 1. The largest single storm water overflow in the METRO system entered the East Waterway of the Duwamish estuary (Miller, 1977). Following a storm, coliform bacteria levels measured in this area ranged up to 324,000/100 ml. Suspended solids in the overflow varied from about 100 mg/l to approximately 500 mg/l. These values are comparable to raw sewage. Additionally, the discharge was found to contain black, tarry substances and relatively high concentrations of PCB. The PCB species found in the outfall were traced in the receiving water and differentiated from the PCB species involved in the past transformer spill. No microscopic organisms were found in sediments immediately adjacent to the discharge although myriads of syphonid worms were found outside the prevalent path of the discharge plume. Highest metal concentrations

were found in nearby black, oily sediments which appeared to be a direct result of the overflow (Brown and Caldwell, 1958 and 1979).

Studies of dry weather discharges through deep water outfalls exhibit minimal environmental impact as shown in the following example also from the METRO system: Example 2. Studies in 1975 and 1976 of the West Point outfall traced the effluent plume using light transmittance as an indicator (Bendiner, 1976; Evan-Hamilton, Inc., 1975). Simultaneously, DO values were measured in the plume. The resulting analysis showed that during these studies DO values in effluent plume were not significantly different from background values (Environmental Quality Analysts, 1975). Coliform bacteria were also measured. Typically, chlorine disinfection reduces the fecal coliform bacteria levels to less than 1000 organisms per 100 ml in the raw discharge and subsequent dilution (140:1) from the multiport diffuser at the end of the outfall reduces fecal coliform concentration below the Washington State Water Quality Standard of 14 fecal coliform organisms per 100 ml (METRO, 1978c).

Water, marine life and bottom sediments sampled near the outfall and in control areas within the central basin contain significant concentrations of heavy metals, notably lead, zinc, copper and mercury; however, metal enrichment is almost uniform through the area sampled (Schell et al., 1977). The researchers concluded from these results that the West Point discharge has minimal influence upon the quantities of metals in Puget Sound. Studies of benthic macrofauna and foraminifera were conducted during 1976-77 for the purpose of determining effects of the discharge; however, local dredge disposal sites and other influences confounded the data analysis preventing statistical determinations of effect (Harman et al., 1977). More recent macrofaunal studies (1978-79) have indicated some organic enrichment of sediments in the path of effluent dispersion from the outfall (Thom and Chew, 1979).

Mixing and Transport

Mixing and transport characteristics also affect impacts. The Sound consists of a series of interconnected basins often separated from each other by shallow regions called sills (Cannon, 1979). Sills tend to isolate the deeper water of one basin from that

of adjoining basins. Thus, deeper basin waters tend to age, decreasing in dissolved oxygen content and increasing in nutrients.

The circulation of sea water at intermediate and shallow depths is more complete than the marine waters of the deep basins. Intermediate and shallow waters of the Sound are driven by the constant addition of fresh water from rivers and surface runoff and by the influx of sea water at depth. Mixing occurs in the zone between seaward moving surface waters and incoming deep saline waters. It is this mechanism by which the freshwater influx to the Sound picks up salt and nutrients from the sea water. Although Puget Sound is an estuary, its separate components function as distinct estuaries with their own sets of characteristics and responses.

Recently, the NOAA-sponsored Puget Sound MESA project has provided new insights into the circulation of waters in the Central Puget Sound Basin (US Department of Commerce, 1978). Continuous measurements of currents, temperature and salinity suggest that on the average water below a depth of approximately 50 meters (165 ft) is exchanged by new oceanic waters in about one to two weeks.

The main basin of the Puget Sound system exhibits a large advective transport that forces exchange of oceanic waters through the Straits of Juan de Fuca. The main basin itself is influenced by a tidal action at the Tacoma Narrows which induces a net clockwise flow around Vashon Island. Flood tide currents travel southward toward Commencement Bay where circulation becomes weaker. On ebb tides, water from south Puget Sound moves rapidly out through the Tacoma Narrows pumping water up from depths east of Vashon Island and forcing it north along the west side of the island. This pumping action apparently increases the transport at depths in the Central Basin. However, studies also suggest that much of the water is recycled to depth again as a result of vigorous mixing in Admiralty Inlet. This flow pattern may then result in a lengthening of residence times for municipal and industrial discharges in the Central Basin.

Water Quality Related Studies

A great deal of work has been done on specific aspects of Puget Sound water quality; some of the earlier studies which date into the 1930's allow some speculation as to human influence on the system

(Pacific Northwest River Basins Commission, 1971a; University of Washington for METRO, 1953). Studies recently sponsored by METRO have drawn on the early data and recently collected information to focus on the impact of primary effluent discharges from METRO's four Puget Sound treatment plants (USEPA Region X, 1977). These independent research efforts are summarized in Table B.10 and include physical, chemical and biological studies conducted from 1975-1977 at a cost of approximately $1.1 million (Brown and Caldwell, 1979).

Physical and chemical studies have demonstrated the good dispersal characteristics of the Central Basin. Dye trace studies showed a minimum dilution of 100:1 in the discharge zones of the METRO outfalls. Water quality and nutrient investigations have been conducted with the hopes of identifying the effect of METRO's West Point treatment plant effluent on certain water quality parameters: temperature, salinity, density, dissolved oxygen, light transmittance reduction (turbidity) and pH. While all parameters but pH were found to be relatable to the effluent plume, only turbidity was more than barely detectable in its influence.

Potential toxicants are receiving increased study. METRO's Toxicant Pretreatment Planning Study is assessing toxicant sources, their fate in treatment processes and the presence and impact of toxicants in receiving water systems. Industrial waters, residential/commercial discharges, and area wide non-point sources such as urban runoff and atmospheric fallout are being investigated.

METRO's Pretreatment Planning Study is being developed in direct response to the 1977 amendments to the Federal Water Pollution Control Act, which included major emphasis on toxicant control (Metropolitan Engineers, 1978; METRO, 1974b and 1978b). Toxicants identified by EPA as priority pollutants have been detected in municipal dischargers and receiving waters within the Puget Sound area. METRO identified at least 38 of the EPA designated priority pollutants in a preliminary evaluation of treatment plant discharges. Although most of these toxicants were found in low concentrations using sensitive gas chromotography/mass spectrophotometry techniques, their presence suggests more research is warranted.

Studies dealing with toxicants in Puget Sound waters have been conducted by NOAA and the Corps of Engineers (US Department of Commerce, 1978).

Table B.10 METRO Sponsored Puget Sound Research Studies: 1975-77 - Summary
(Brown & Caldwell, 1979)

Study Contractor	Topic	Major Findings
Applied Physics Laboratory University of Washington	Field measurement of effluent dispersion from West Point outfall using dye tracers	West Point dye tracer studies demonstrated effluent diluted by factor of 100 at outfall, subsequent dilution factor is 2 to 15 fold; diluted effluent was detected on eastern shoreline in summer.
Evans-Hamilton	Field measurements of currents and mixing properties near West Point using drogues.	Tidal currents stir effluent in large easterly eddies which formed patches that persisted for up to 15 hours; small scale eddies at slack tides are most effective in breaking up large patches of effluent.
Environmental Quality Analysts	Effluent plume location and water quality monitoring at each of the four Puget Sound outfalls using light transmittance to locate plans.	Effluent related turbidity was detected within 1 1/2 miles from West Point outfall. Measureable oxygen fluctuations were detected in initial effluent plume but not the subsequent effluent field. Lower oxygen values noted in effluent plume at Alki Point.
College of Fisheries Departments of Oceanography Chemistry & Civil Engineering. University of Washington	Measurement of heavy metals in water, biota and sediments from West Point vicinity and control areas and a review of heavy metal sources.	Net increase in heavy metals near outfall not significantly greater than control areas; small increases measured in biota and sediments without indication of harmful effect. Effluent adds minor heavy metals contribution when compared with other sources.
Department of Oceanography University of Washington	Comparison of present and historical dissolved oxygen and nutrient data to determine if significant changes have occurred subsequent to METRO discharge.	Effluent discharge has had no measurable effect on dissolved oxygen or nutrients in the main basin of Puget Sound. Ammonia noted in effluent plume; however, this "near field" effect is very localized and has no impact on main Puget Sound system.

Shoreline Community College & Fisheries Research Institute, University of Washington	Assessment on subtidal benthic organisms, sediments and habitats near West Point outfall.	The present effluent discharge from the West Point outfall does not significantly influence the bottom sediment or benthic macro fauna; however, there may be differences in species distribution and abundance which further data may demonstrate.
College of Fisheries University of Washington	Assessment of effects of effluent discharge on intertidal biota at several central Puget Sound beaches.	No measurable effects of wastewater discharge were observed on intertidal fauna; minor differences were detected between algae found at beaches surveyed.
Fisheries Research Institute, College of Fisheries, University of Washington	Survey of demersal fishes near West Point and Alki outfalls in the Duwamish River and at a control site to determine effects of discharge on fish community and to examine relationships between discharges and disease and parasite infections.	No evidence that discharge had adversely affected nearshore, shallow water fishes, possible seasonal impact on juvenile salmonids, reduction in species and diversity and abundance at deeper waters may occur near West Point. There was no evidence linking tumor incidence in flatfish with METRO discharges. High PCB levels in the Duwamish were linked to fin erosion and high incidence of liver disease.
Department of Oceanography and Fisheries Research Institute, University of Washington	Midwater trawl and acoustic surveys of fishes and zooplankton were conducted to determine effects of outfall discharges.	Fewer organisms observed in effluent plume although there was no basis to suggest organisms in the plume differed from those in the control site. No major or widespread effects of METROs effluent plumes were observed.
Fisheries Research Institute, University of Washington	Bioassays of West Point primary effluent were conducted to determine toxicity to five species of marine organisms.	Chlorine used for disinfection was most toxic substance found in the effluent; toxicity can be reduced by controlling residual chlorine, sludge discharges of heavy metals and other industrially related toxic substances.

Much of the emphasis in NOAA studies is to determine the amount of synthetic organic chemicals in sediments of Elliott Bay along Seattle's waterfront and in commencement Bay near Tacoma. The Corps study deals with effects of dredge spoil disposal as related to PCB-contaminated sediments from the Duwamish River.

BIOLOGICAL AND PUBLIC HEALTH IMPACTS

While more than 150 major industries, municipalities and/or sewerage agencies discharge to Puget Sound, relatively few areas are impacted sufficiently to document widespread ecological damage (Water Resources Engineers for the EPA, 1974). However, localized, nearshore water quality degradation associated with large volume discharges from the pulp and paper industry and inadequately treated wastes, upset conditions, and storm water overflows from municipal sewage treatment facilities, has occurred. Localized biological and/or public health impacts from municipal plants primarily take the form of irregular fish kills, closure of water contact beaches, and prohibition of shellfish harvesting.

During 1978, eleven fish kills involving 526,000 fish were reported to the Washington Department of Ecology (Seattle Times, 1979). Of these kills, five involving the vast majority of fish (over 500,000) were attributed to die-off of natural plankton blooms in restricted embayments off the Central Basin of the Sound. Of the remaining six incidents, construction runoff, dynamite blasts, industrial dumps, and wastewater from cement trucks were identified as the probable causative factors. In only one incident, involving the failure of a sewage pumping station, was municipal waste believed to have caused the fish kill. In 1977, ten fish kills, two of which may have been caused by municipal upsets, were reported in Puget Sound.

Beach closures, caused by improperly treated municipal waste or storm water overflows have occurred occasionally in metropolitan areas of Puget Sound. Although localized high bacterial levels still occur during wet weather, in general health-related bacterial contamination has been virtually eliminated due to comprehensive wastewater management programs of METRO and other marine discharging municipalities. By 1978, no beaches were closed in the Puget Sound area as a result of municipal bacterial contamination.

Shellfish Harvesting

Population growth and associated sewage wastes appear to pose a significant threat to approved shellfish growing areas in Puget Sound. METRO water quality monitoring data indicate that while Central Basin beaches routinely meet coliform bacterial standards for bathing, standards for commercial shellfish waters are frequently not met (USEPA Region X, 1978). These beaches are then classified as "closed growing areas." "Closed growing areas" are so classified largely because of fecal contamination or the potential for such contamination from nearby municipal sewage discharge locations. The condition of shellfish and the extent of "closed growing areas" have been used as indicators of pollution intensity and as assessment tools for measuring the impact of pollution control efforts (Stevens, et al., 1975; Woelke, 1972). "Conditionally approved areas" are generally characterized by excessive fecal coliform contamination from intermittent freshwater runoff of agricultural or silvicultural activities, occasional storm water overflows and treatment plant bypasses.

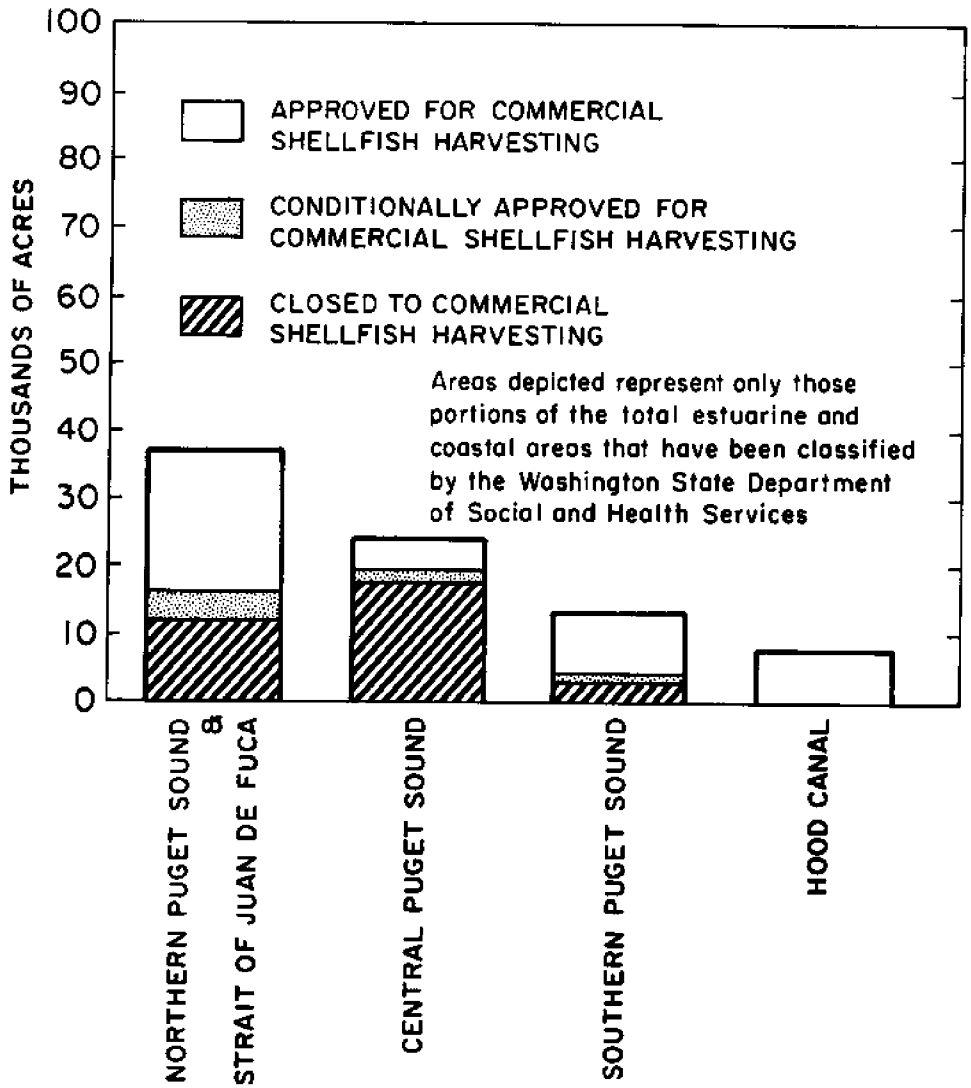
Using criteria established by the US Food and Drug Administration for the National Shellfish Sanitation Program, 68 percent of the 228,900 acres of classified commercial shellfish growing waters in Washington state are currently classified as "approved for commercial harvesting." Another 11 percent are "conditionally approved", and depending on specific conditions, are monitored throughout the year. The remaining 21 percent are classified as "closed" and cannot be used to produce shellfish for human consumption (USEPA, 1979). Figure B.6 shows the status of classified shellfish areas in the Puget Sound region. As illustrated, the greatest acreage of "closed shellfish growing areas" is found in central Puget Sound primarily due to actual or potential bacterial contamination from metropolitan Seattle and Tacoma.

The presence of a biotoxin in shellfish caused by a naturally occurring planktonic organism, Gonyaulax, has also resulted in the closure of specific growing areas during bloom conditions.

METRO Studies

A number of ecological investigations have been conducted to assess the impact of specific constituents from municipal waste on the biotic communities of Puget

Figure B.6 Status of classified shellfish areas in Washington



Sound. Historical and ongoing assessment studies sponsored by METRO provide the greatest opportunity for evaluation of the impacts of municipal discharges on marine/estuarine organisms (Oceanographic Institute of Washington, 1974; USEPA Region X, 1976; Washington Department of Ecology, 1976). While the West Point Sewage Treatment Plant and associated marine discharge has been the subject of the majority of these assessment studies, there is some evidence to indicate that the Elliott Bay-Duwamish system has been the most stressed of the several greater-Seattle discharge areas. Recently, studies have been initiated by METRO, the Corps of Engineers, and NOAA which will provide more information on this system. Much of the earlier research was undertaken due to the size of West Point plant and METRO's intent to apply for a secondary treatment waiver under Section 301(h) of the Federal Clean Water Act. Most aspects of this discharge, ranging from the effects of its chlorination processes to the impacts of the discharge on benthic, planktonic, and free-swimming populations, have been evaluated in one or more studies.

Chlorination. Several of these METRO studies dealt with the significance of residual chlorine following effluent chlorination (Buckley, et al., 1976; Strober et al., 1974). Safe effluent chlorine concentrations, as measured by average total residual chlorine ($TRCl_2$), have been estimated to be between 0.3 and 1.1 percent. Above this level, reduced hemoglobin/hematocrit numbers in salmon, equivalent to anemia, have been documented. Because of the natural processes of chlorine decay in marine waters (chlorine demand, ultraviolet radiation, and dilution), as well as the avoidance behavior of free-swimming species, it appears that chlorine residuals from the West Point discharge have little of no impact on the indigenous biota.

The potential for chlorine toxicity at relatively low concentrations does exist, however. Impacts may occur on some biological communities in Puget Sound where municipal discharges are overly-chlorinated and/or discharged into low volume, poorly circulated receiving waters. As a consequence, METRO has installed dechlorination facilities at its Renton Sewage Treatment Plant to protect fish populations of the Duwamish-Green River system.

Plankton impacts. Numerous studies have looked at the possible impacts of the West Point Sewage

Treatment Plant discharge on species diversity, cyclic concentrations, and productivity of planktonic communities (Collias, 1977; Environmental Quality Analysts, 1975; Evans-Hamilton, Inc., 1975; University of Washington for METRO, 1977a). These studies, dating from the mid 60s through 1975, show no evidence that nutrient additions by West Point have altered the pattern or seasonal cycles or species diversity of phytoplankton in the main Puget Sound basin. In fact, it has been identified that stability of the water column and solar radiation are the major factors in promoting phytoplankton blooms and subsequent growth of grazing zooplankton (University of Washington for METRO, 1977a).

In Puget Sound, light penetration of typical rapidly mixed waters is insufficient to promote blooms. During late April and early May, a combination of high river drainage, the approach of neap tides and/or increasing solar insolation promotes stratification. This condition allows the surface waters to generate significant phytoplankton blooms. The die-off of some of these blooms especially in poorly flushed nearshore areas ultimately produce localized water quality problems such as low dissolved oxygen and toxicity. During heavy bloom conditions, decreased nutrients have been observed but only rarely has widespread depletion been observed. Nutrient discharges from METRO have, therefore, not correlated with bloom development. Breakdown of stratification in the fall, along with net water outflow at the surface, typically combine to terminate bloom conditions. Carbon-14 studies performed by METRO during the period of 1969-75 attempted to correlate the high production rates of the central Sound with ammonia levels from West Point (Evans-Hamilton, Inc., 1977). While natural production averaging 3.18 metric tons (3.5 tons) of organic material/acre per year, was measured, most of the required nutrient contributions were traced to upwelling of deep saline waters. As a result of this study, the West Point discharge has been postulated to contribute up to 20 percent of the localized phytoplankton production. However, as the equilibrium depth of the discharge was found to be typically between 20 and 50 m (66 and 130 ft) and dilution of effluent ammonia to background levels occurs within 6 to 12 hours, it was further postulated that nutrients from the discharge rarely get into the waters where the blooms occur. Further, if discharges do reach bloom areas, they are present at significant concentrations for less time than required normal phytoplankton division.

Benthic and intertidal impacts. As the West Point outfall is located at approximately 80 m (260 ft) depth approximately one km (0.6 mi) from shore, high concentrations of effluent seldom reach the intertidal zone. Dye studies of the effluent plume have shown that during the summer, effluent contact with the shore may occur at maximum estimated concentrations of 3.5 ppt (University of Washington for METRO, 1976c). Little or no effluent appears to reach the beach in winter. Intertidal organism studies since 1971, however, of nearshore areas of periodic effluent contact have been unable to document any correlation between the West Point discharge and changes in intertidal populations (Harman, et al., 1977; University of Washington for METRO, 1976a; USEPA Region X, 1978).

Studies of the benthos in the central basin of the Sound, however, have shown several distribution patterns correlative to plume distributions of municipal outfalls. High density benthic populations are prevalent around 50 m (180 ft) depth as a result of little new water movement. (This is the approximate depth of the boundary layer between incoming deep saline waters and surface waters with net seaward movement). As the West Point discharge typically reaches equilibrium at this depth, the plume is believed to be related to several benthic distribution patterns, including those found in several species of diatoms and arenaceaous foraminifera. One of the foram species, Buliminella elegantissima, which appears to be associated with the West Point discharge, has been shown to be associated with a southern California municipal outfall (University of Washington for METRO, 1976a). Relatively high concentrations of organic material have been measured in the sediments under the West Point plume. It has been postulated that this accumulation of organics is primarily responsible for the benthic distribution patterns.

Impacts of fish populations. Studies to measure the possible impact of METRO discharges on the fish populations have primarily taken two forms: catch statistics of morphological and physiological abnormalities, and laboratory toxicity measurements. Studies of fish distribution patterns, incidence of tumor-bearing species, and fin rot disease have been conducted throughout the Central Basin. These studies have produced varied and inconclusive results: (University of Washington for METRO, 1976a and 1976b). English sole appears to have the highest incidence of tumors among the indigenous demersal (bottom dwelling)

fishes. However, English sole from non-municipal impacted areas (Gulf of Alaska, Bering Sea, etc.), frequently have a higher incidence of tumors than the local species associated with municipal discharges. The incidence of fin rot, reportedly associated with several municipal outfalls in southern California is very low in all areas of Puget Sound, including the West Point outfall area. It is postulated that the levels of bottom sediment sludge contamination, which does not occur at West Point, may make a difference in the incidence of fin rot. A higher incidence of tumors, fin rot, and liver disease was found in demersal fish of the Duwamish River. This river, however, receives significant municipal and industrial effluent discharge and urban runoff, presumably a contributing factor.

Beach, purse seine and trawl studies in the West Point area have shown no disturbance of the distribution of pelagic fish. To the contrary, the area has been identified as having high numbers of juvenile coho, chinook, and chum salmon, along with an abundance of juvenile and adult demersal fish. It has been suggested that the area in the vicinity of the outfall may serve as a nursing ground for several species (University of Washington for METRO, 1976b and 1977). Thus, it appears that no significant avoidance of fish to the discharge plume takes place in the receiving waters of the West Point outfall.

Acute toxicity studies in the laboratory have been conducted on representative West Point effluent and Puget Sound indigenous species (University of Washington for METRO, 1976a and 1977b; University of Washington, 1977). The results of these studies have shown that juvenile English sole and shiner perch are the most sensitive of the species tested with 96 hr - LC 50's* in the range of 15 to 16 percent effluent concentrations. Local shrimp, sculpins, crab and other species have also been tested.

In addition, pilot treatment systems (filtration, SO_2, dechlorination, ion exchange) of the West Point waste have been studied with bioassays. From these studies it has been shown that the solid fraction of the effluent contributes 15 to 20 percent of the potential toxicity, chlorine residual approximately 50 percent and ammonia toxicity approximately 20 percent (University of Washington for METRO, 1976a).

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*The concentration of test media which at the end of four-day bioassays yields 50 percent mortality (50 percent survival) of the test species.

The subject of toxicity is receiving even more attention in ongoing studies conducted by METRO with EPA grant support.

FUTURE DIRECTIONS

Regional population growth will create a need for rapid expansion of water and sewer services. Recent four county studies of the Seattle area by the Puget Sound Council of Governments (See Table B.11) (Power, 1979b) indicate a need to provide roughly 30 percent more water and sewer services in the next 21 years. Other areas adjacent to Puget Sound will experience similar or greater growth, also requiring additional water and sewer services. Projected funding needs will probably be larger than that required simply to meet expansion of services, however, due to existing system deficiencies.

Upgrade Existing Treatment Plants

The Federal Water Pollution Control Act has probably been the single largest driving force to construct and upgrade municipal and industrial treatment plants discharging to the waters of Puget Sound. The 1972 Amendments to the Act (PL 92-500) significantly expanded the federal and state governments' roles in pollution control. The amendments raised the level of funding for municipal sewage facilities construction and created a national permit system requiring uniform technology-based effluent limitations for both municipal and industrial discharges.

Coupled with the natural ability of many receiving waters to absorb, disperse, degrade wastes, the Act has functioned to protect and enhance the waters of Puget Sound. Today, these high quality waters now offer some margin of safety before significant water use losses occur. However, this margin of safety is unknown or non-existent in some areas, and could be threatened by the increased growth projected for the area. In fact, recent studies have shown degradation in certain shallow bays, such as Bellingham and Port Gardner bays. To guard against further water use losses, the State of Washington has adopted, with EPA's assistance, an anti-degradation policy that calls for maintaining, whenever possible, existing water quality in high quality areas, which includes the waters of Puget Sound.

Table B.11 Central Puget Sound Regional Population Forecasts (Puget Sound Governmental Conference, 1970)

County	1976	1978	1980	1990
King	1,155,700	1,186,900	1,255,289	1,428,491
Kitsap	88,600	129,400	138,200	176,508
Pierce	420,700	442,600	453,379	562,041
Snohomish	270,100	292,300	340,163	432,604
Totals	1,935,100	2,051,200	2,187,031	2,599,644

Sewer Service Area (in acres)
Central Puget Sound Area (4 counties)

County	1976	1980	1990
King	141,500	145,585	192,557
Pierce	44,236	72,185	101,337
Snohomish	34,500	39,620	60,270
Kitsap	11,345	14,850	23,500
Total	231,581	272,240	377,664

Many of the region's larger cities, including Seattle, Tacoma, and Bellingham, provide only primary treatment. Although the requirement for secondary treatment for municipalities in Puget Sound was clearly mandated by Congress in the Federal Water Pollution Control Act, Amendments of 1972, the 1977 Amendments now provide for waivers if marine-discharging municipalities can show conclusively that water quality and other parameters will not be impaired. Applications for these 301(h) waivers were filed by many Puget Sound municipalities.

Even if waivers are granted to some municipalities, additional control or treatment may be needed to reduce or eliminate specific pollutants. For example, water quality could be jeopardized if oil and grease, nutrients, bacterial and viral organisms and certain other substances aren't controlled to safe levels in all waste discharges. Future treatment systems must be designed to remove, or significantly reduce, these substances which are damaging to a marine environment. As many metals, pesticides and other pollutants adsorb to solids in wastewater, enhancing removal of settleable and suspended solids may be required to protect water quality. Additionally, studies have shown that pathogens adsorbed on particles which settle rapidly remain infectious for long periods of time; a potential problem if the sediments in which they reside are resuspended by natural water forces (USEPA Region X, 1978).

When the Water Pollution Control Act was amended in 1977, the concept of Best Available Control Technology (BAT) was redefined creating greater attention to the treatment and removal of toxic substances. As now stated in the Act, if economic analysis and water quality improvement analysis show a need, municipalities may be required to remove pollutants to levels beyond what is required by secondary treatment.

The need to control toxic pollutants requires special consideration due to the damage which can result from their high concentration discharge and/or accumulating properties in receiving waters. For example, high concentration of PCBs have been found in the Duwamish estuary sediments which may have adversely affected the finfish populations. One of the potential sources of toxics in municipal waste occurs through chemical reaction of chlorine used in disinfection with organics present in treated effluent. Other toxics may enter the municipal system from industrial hookups. Therefore, a partial solution to prevent the discharge

of toxics prior to discharge into receiving waters or treatment works may be found in the industrial pretreatment requirements of PL 92-500 (Section 301), which all municipalities and sewer districts are currently developing. However, statewide pretreatment programs in the future could be more effective. A major difficulty arises in assessing the existence and effects of toxics. An EPA grant to Seattle METRO for developing a pretreatment program for toxicant control is attempting to answer some of these questions and solve some of these problems.

Combined Sewer Overflows

Many of the larger cities bordering the Sound such as Seattle, Tacoma, Everett and Bellingham have combined sewers in at least part of their service areas. These combined systems are now the main source of untreated waste in the area. Several municipalities are now focusing their efforts on correcting these storm overflows. Significant amounts of money are being sought to correct the problem. This will be a long and costly program that will compete for funds with the need to upgrade primary treatment systems. Jointly funded waste interception and combined sewer overflow projects will probably receive higher priority in the future than will upgrading treatment plants for some cities discharging into the Sound.

Non-Point Sources of Pollution

Congress and those concerned with protecting the water quality of Puget Sound recognized that achieving desirable water quality goals may require more than control of point source discharges. Non-point sources of pollution, such as urban drainage, are now thought to contribute over 92 percent of the suspended solids that enter our nation's waterways each year and 98 percent of the fecal coliform contamination (University of Washington, 1977). A moderately sized city like Seattle typically washes from its streets and parking lots between a hundred and two hundred thousand pounds of lead and between six and thirty pounds of mercury each year into receiving waters (Puget Sound Governmental Conference, 1970).

To begin to deal with these problems in the Puget Sound area, EPA awarded planning and management grants under Section 208 (Areawide Planning) of PL 92-500 to Seattle METRO, the Snohomish County Metropolitan Municipal Corporation, and the Washington Department of

Ecology. The water quality management programs developed by these agencies will deal with pollution from urban runoff and agricultural and forestry activities. With the catalyst provided by Section 208, state and local agencies are now committed to a full-scale effort to dovetail water control planning with other environmental control activities and regional planning such as the Coastal Zone Management Act and the State of Washington's Shoreline Management Act.

SUMMARY AND CONCLUSIONS

1. Puget Sound, a large multi-branched fjord-like estuary in northwest Washington State, is the center of a rapidly growing population center whose economic base rests on manufacturing, forest products and marine resources. The population of the area adjacent to the Sound is estimated to be 2,600,000 (1979).

2. The waters of Puget Sound are generally of high quality, routinely exceeding their assigned state water quality standard classification. Most waters within the Sound are classified Excellent (A) or Extraordinary (AA).

3. Water quality problems that do exist within the Sound are normally associated with municipal and/or industrial discharges into poorly flushed, localized embayments (bays, inlets, etc.). Almost all water quality problems that have been identified are from the highly populated Central Basin of the Sound or poorly exchanged inlets of the South Sound.

4. In localized areas, particularly the Central and South Sound, conflicts exist between consumptive/assimilative uses (industrial process water, waste discharge) and nonconsumptive uses (shellfish, finfish rearing and harvesting).

5. In addition to industrial wastewater discharges, particularly pulp and paper and food processing, inadequately treated waste upsets conditions. And storm water overflows from municipal sewage treatment facilities are the major point source influences on Puget Sound. Nonpoint pollutant sources, urban runoff and agricultural losses also contribute significantly to localized impacts.

6. A major impact of municipal wastewater discharge to Puget Sound is to shellfish rearing and harvesting. At present, 21 percent of Puget Sound's waters are "closed" to commercial shellfish harvesting; another 11 percent are "conditionally approved." Most of these closed or marginal areas are in the Sound and Central Basins. Secondary threats in the form of beach closures to swimming and localized fish kills have also been documented in the past.

7. Most major municipal dischargers to Puget Sound employ primary treatment; at present secondary treatment dischargers to this marine system are the exception. Several of the marine dischargers have applied to obtain secondary treatment waivers under Section 301(h) of the Clean Water Act. If waivers are granted, additional treatment such as physical-chemical treatment may be needed to reduce specific pollutants.

8. The only untreated municipal sewage now reaching the Sound and/or its tributary streams is combined sewer discharges which release raw sewage during periods of high rainfall. During these periods influents exceed collection and/or treatment facility capacities. Seattle METRO and other agencies are now giving a high priority to control, through special treatment or separation of sewers, of this pollution source.

9. Water quality studies sponsored by Seattle METRO, NOAA, EPA, WDOE, and the University of Washington provide a comprehensive record of water quality, marine life and physical mixing properties of Puget Sound, particularly in the Central Basin. Only recently have the smaller sewage agencies discharging to the Sound begun to collect data on the ecological significance of their discharges.

10. Studies sponsored by METRO and conducted by a wide variety of academic and consultant organizations have failed to document significant degradation by municipal discharges of plankton blooms, intertidal diversity and fish populations within the Sound. Minor changes to benthic distributional patterns and localized water column productivity have been observed or strongly suggested.

11. Toxicity studies of municipal effluents have produced varied results. Laboratory data have documented acute toxicity at high concentrations

of waste. Chlorine residuals and chlorinated hydrocarbons from the disinfection process are the apparent major toxicants. Toxicant-related studies, especially chronic toxicity impacts, are currently receiving much support with major investigations being conducted by Seattle METRO, NOAA and the State of Washington. The METRO work, partially funded by the EPA and the WDOE, is designed to produce a toxicant pretreatment program which will control both point and non-point sources.

12. Projected growth in the Puget Sound area will put additional pressure on existing treatment and collection systems. System upgrading and expansion of facilities will be required. It is estimated that a 30 percent increase in facilities will be required by the year 2000. Availability of federal funds and the existing political and regulatory climate will play a major role in the timing and extent of these upgrades and expansions.

13. The current status (1983) of treatment plans for the major municipalities discharging to Puget Sound is in a state of flux. Several efforts presently underway such as METRO's pretreatment study, the outcome of secondary treatment waiver applications, etc. will play major roles in determining the direction of treatment programs for all marine discharging municipalities.

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Appendix C

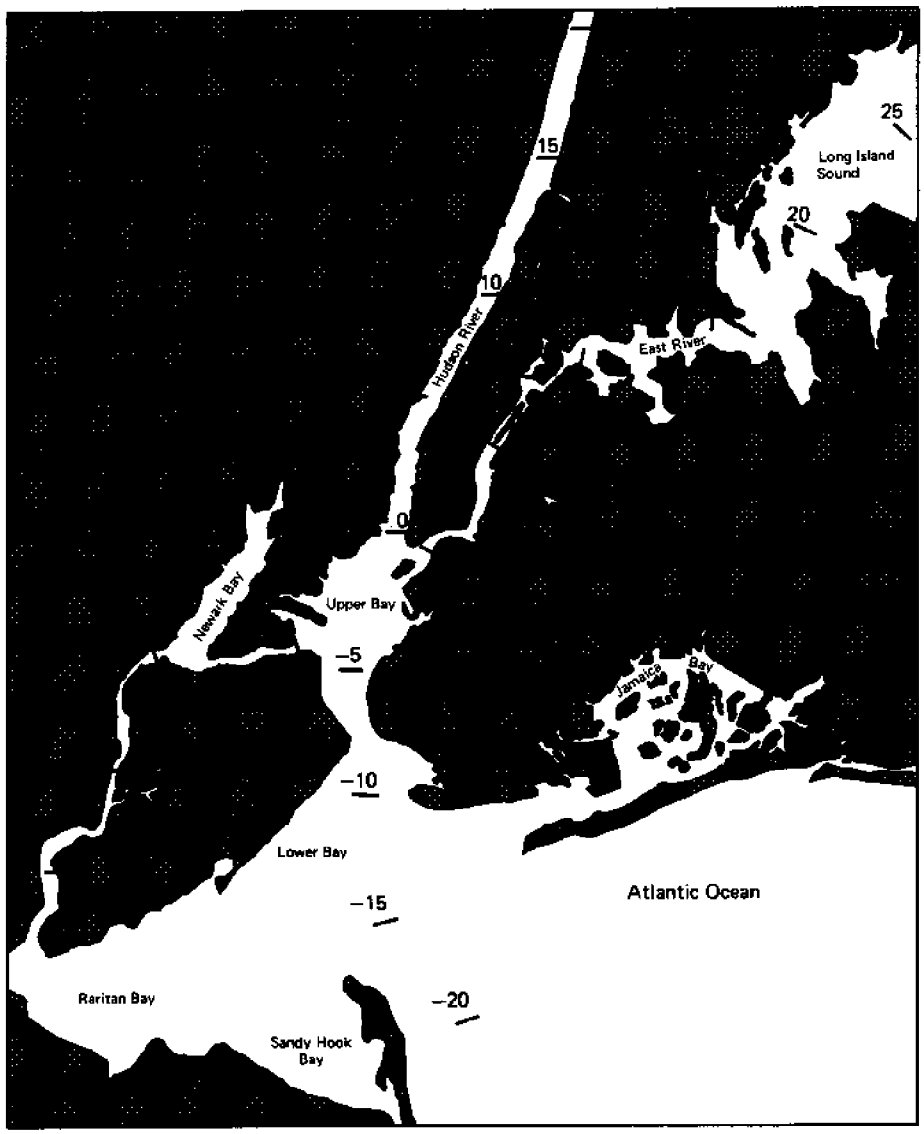
Waste Disposal in the New York Metropolitan Area, A Case Study

Charles G. Gunnerson

INTRODUCTION

Waste disposal in the New York metropolitan area developed along conventional ancient lines (Tarr, 1977; Gunnerson, 1978). Periodic flooding has always been a problem in low-lying metropolitan areas so that stream channels were gradually improved to handle the runoff. As population densities increased several things happened: 1. local water supplies became inadequate and had to be supplemented by development of increasingly distant supplies, beginning with the Croton aqueduct in 1842; 2. manual conservancy systems for utilizing night soil from buckets or pit latrines became overloaded and people began using drainage channels for disposal; 3. ordinances were passed forbidding but not preventing placing of human wastes in drains; 4. increased availability of water permitted introduction of flush toilets and cesspools; 5. pollution of local groundwater supplies from increasing amounts of water discharged to overloaded cesspools which "constituted a means of distributing fecal pollution over immense areas" (Lee, 1894); 6. loss of local water supplies leading to even greater dependence on imported water; 7. piping of dilute household wastes to storm drains and inadvertently creating combined sewers discharging into the Hudson, Harlem, and East Rivers, Raritan River and the Kills (Figure C.1); 8. gradual construction of separate and combined sewers serving 18,032 hectares (ha) (44,557 acres (ac)) and 63,543 ha (157,015 ac), respectively by 1977 (Anderson et al., 1979); 9. local nuisances and public health risks, losses of fisheries, and other ecosystem damage first in the smaller rivers and embayments and, continuing to the present, throughout the estuary; 10. beginning in 1924 treatment and removal from dry weather flows of putrescible materials which resulted in increasing amounts of sludge being disposed of at a convenient site about 12 miles seaward of New York Harbor; 11. enactment of Federal and State legislation mandating increased removals of solids

Figure C.1 New York Harbor area. Numbers show miles from the Battery (Hazen and Sawyer, 1978).



(sludge) which would then have to be disposed of; 12. increasing competition for space so that people increasingly objected to other people's wastes being dumped on or near them (their own wastes were bad enough); 13. increasingly restrictive environmental protection mandates; and 14. increasingly expensive and energy intensive remedial technological responses and proposals.

Similar technological development has occurred in industrial waste treatment. The common characteristic of both the municipal and industrial approaches is that water supplies have been developed and rate structures set with no regard for the costs of getting rid of the water; sewerage and industrial waste systems are designed and optimized with no regard for the cost of flushing water (note that the costs of sewage treatment are not those getting solids out of the water, but getting water out of the solids - the greater the dilution, the greater the cost). The consequences of these conventional approaches are summarized in this chapter.

Geography

The location of the New York Bight is shown on Figure C.2. It lies seaward of most of the people in the Boston-Washington corridor whose predicted future by the year 2000 includes a tripling of the urban area (Koebel and Krueckenberg, 1975). This future is the continuation of five centuries of immigration, growth, and development of natural resources such as climate, harbors, and soils, and is clear evidence of the increasing need for effective land and water resource management. Seventy-three percent of the fresh water entering the New York Bight is from the 35,000 km^2 of the Hudson River drainage basin. It strongly affects water quality and circulation in the western portion of the Bight, especially in the Bight Apex (Figure C.3). Table C.1 presents average annual inputs. Estimated average outflows vary from 210 m^3/sec (7,500 cfs) in summer to 1,300 m^3/sec (47,400 cfs) in April (O'Connor et al., 1977).

Oceanography

Findings of recent oceanographic research have been summarized by Gunnerson (1981). The following has been abstracted from that review:

Figure C.2 New York Bight showing ocean dumping locations.

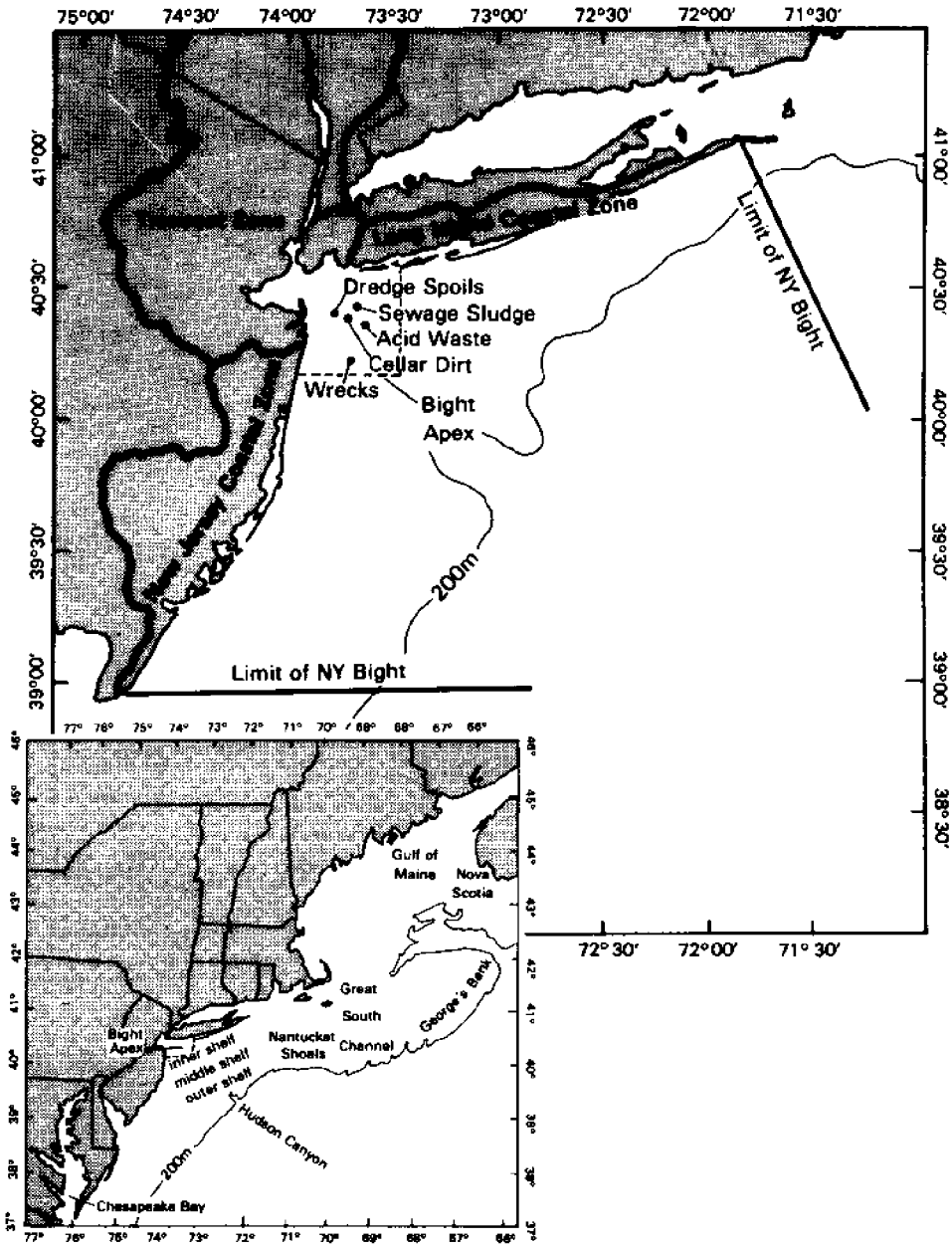


Figure C.3 New York Bight Apex showing depths in meters (Freeland et al., 1976).

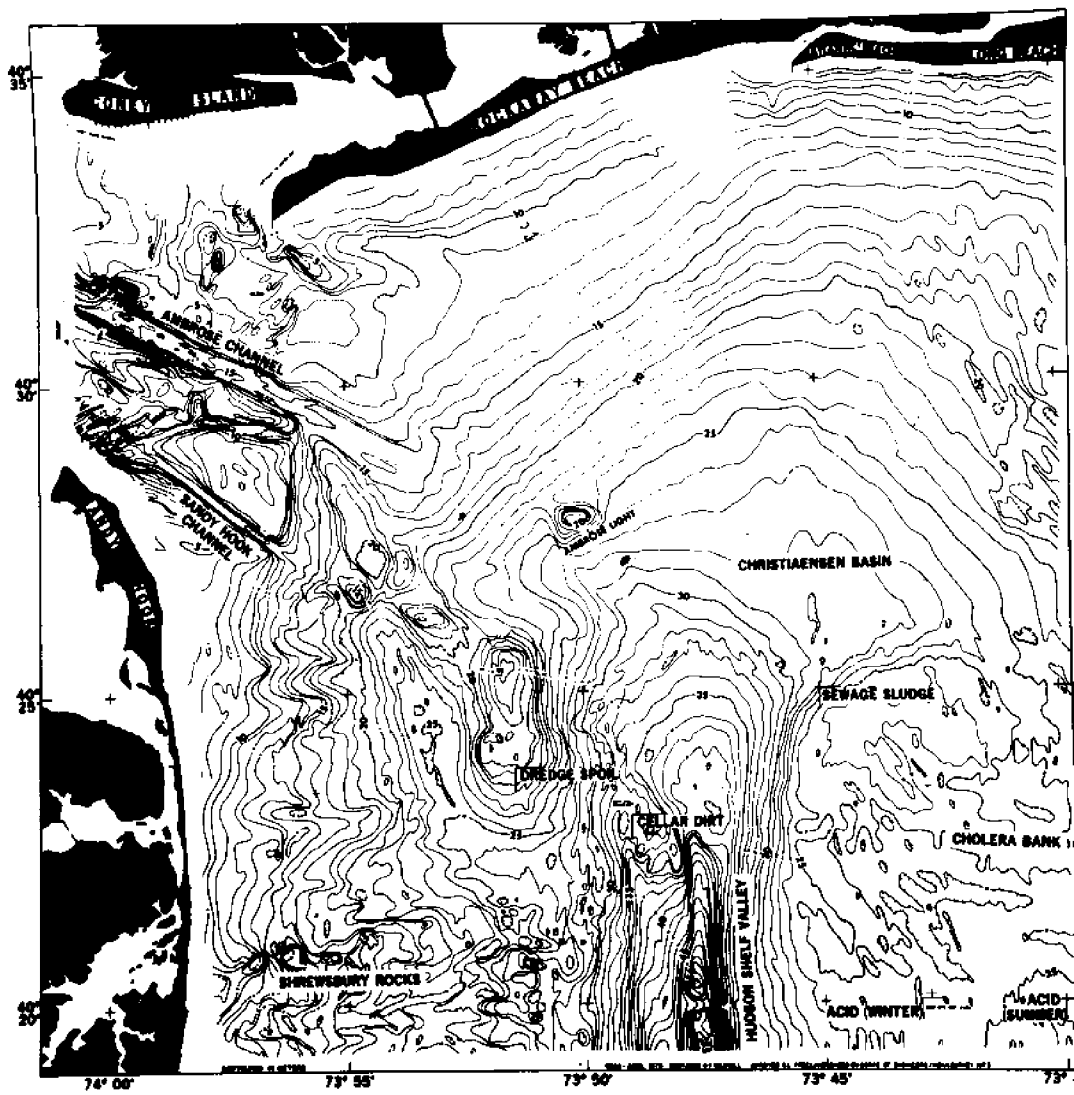


Table C.1 Freshwater Inflows to the New York Bight (O'Connor, et al., 1977)

	Average Annual Runoff		
	m^3/sec	(cfs)	km^3/yr
Hudson River Basin	580	(20,300)	183
New York City and Long Island	80	(2,900)	26
Northern New Jersey (into NY harbor)	110	(4,000)	35
New Jersey Coastal	20	(700)	6
Total	790	(27,900)	250

Physical oceanography

Tides. Mean semidiurnal tidal ranges are from 0.8 to 1.4 m (2.8 to 4.6 ft) along the coast and up to 2.2 m (7.2 ft) in Long Island Sound (Swanson, 1979). These are superimposed upon the annual cyclical variations of approximately 1/4 m (8 in) and a secular increase in mean sea level of about 35 cm (1.1 ft) per century. Storm tides occur with heights up to 4 m and periods of 3 hours to 6 days (Pore and Barrientos, 1977).

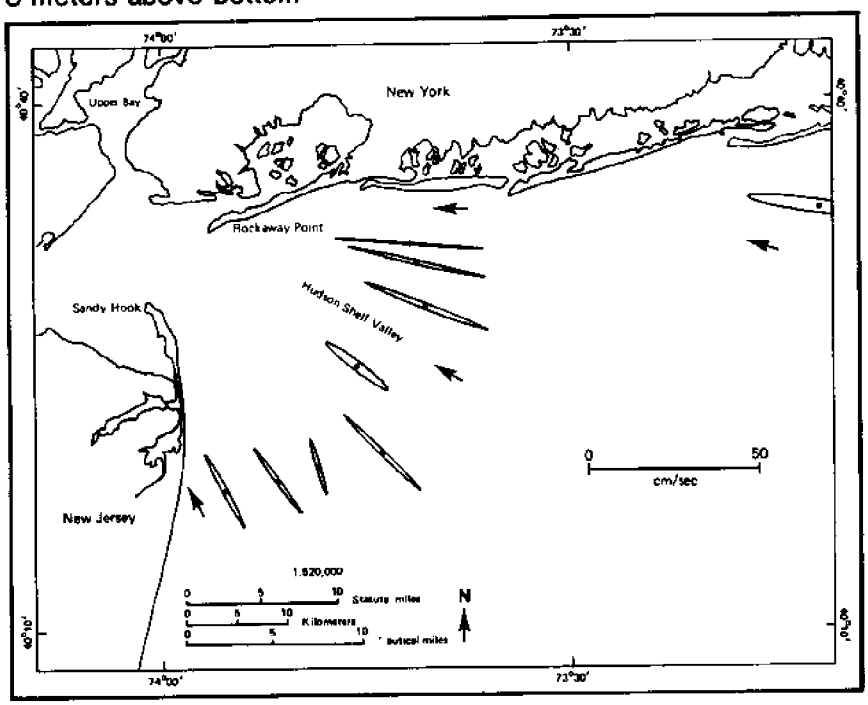
Currents. Tidal currents reported by Hansen (1977) increase near the entrance to the harbor as shown on Figure C.4. Net surface outflows average 16 cm/sec near the center of the Sandy Hook-Rockaway Point transect while average inflows reach 12 cm/sec near Rockaway Point and 4 to 6 cm/sec within the deeper channels off Sandy Hook.

Within the New York Bight, surface flows are generally southwesterly but highly variable. Along the bottom near the river mouths estuarine circulation results in a net shoreward and upstream movement. In the Hudson Canyon and Hudson Shelf Valley, aperiodic up- or down-slope movements result from regional meteorological conditions; monthly average velocities in the Valley are as much as 5 cm/sec (0.1 knots) up-valley (Hansen, 1977).

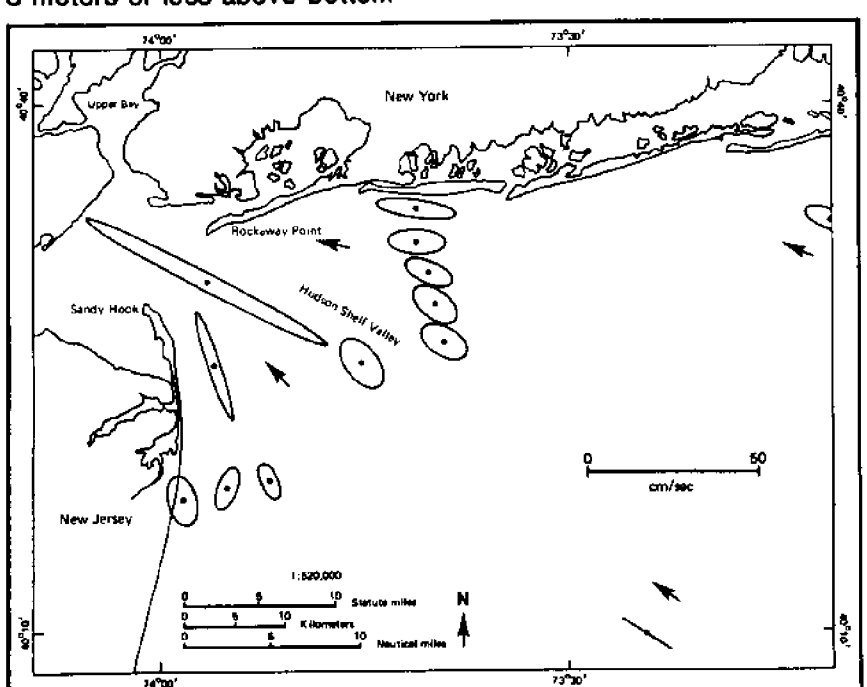
Mean velocities calculated by Beardsley et al. (1976) over the 180 km (100 nautical miles (n mi)) wide continental shelf lead to estimated transport of 2 x 10^5 m^3/sec within the 100 m isobath and residence times of 3/4 year. Residence times in the Apex vary

Figure C.4 Tidal currents in New York Bight Apex (Hansen, 1977).

8 meters above bottom



3 meters or less above bottom



Rotation is predominantly clockwise at 8 m (26 ft) above the bottom, counterclockwise at 3 m (10 ft) or less above the bottom.
The centers of ellipses are the station locations.

Arrows indicate direction of progress of the maximum M_2 flood current velocity.
Current velocity vectors rotate 360° in 12 to 42 hours.

from about 4 to 12 days (Swanson 1979) with slower flushing during the September - October minimum Hudson River outflow period.

Currents over the shelf are strongly affected by local and regional meteorological factors which have been summarized by Lettau, Brower, and Quayle (1976). Surface wind speed and directions over the Bight have pronounced seasonal variations but generally little spatial variation, suggesting that observed air movements are governed by large-scale meteorological systems. Storm fronts which are approximately the scale of the Bight result in significant spatial variability. Wind persistence reaches maximum in winter with westerly to northwesterly winds generally averaging 16 to 25 kilometers per hour (km/hr) (9 to 13 knots). A secondary persistence maximum occurs in July with southwesterly to southerly winds 10 to 16 km/hr (7 to 10 knots). Strong winds occur mostly in winter (December - February when they exceed 50 km/hr (28 knots)) more than 10 percent of the time. The maximum recorded wind occurred at the Battery in 1954 during hurricane Hazel when it reached 180 km/hr (98 knots).

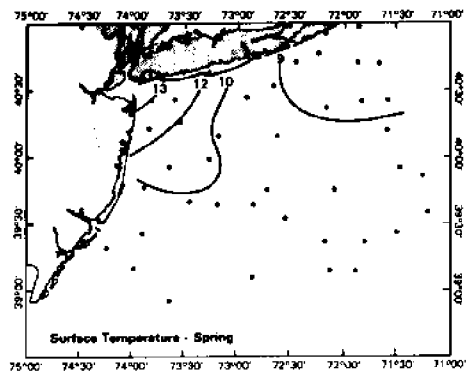
January and February mean air temperatures vary from 0°C nearshore to 4°C offshore along the continental margin. Nearshore mean surface water temperatures range from 2° in February to 23°C in July and August. Offshore water temperatures range from 8°C in February through April to 24°C in August.

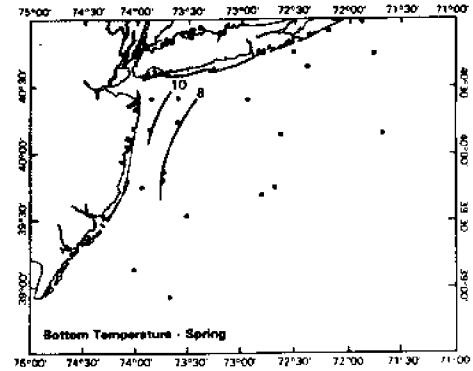
Mean water temperatures are generally 1° to 4°C cooler than air temperatures in spring and early summer and up to 6°C warmer than air in winter (Lettau et al., 1976). The springtime air-sea temperature differentials, local wind mixing, and outflows from the Hudson River influence the rates of development and the depth and strength of the summer pycnocline over the inner shelf.

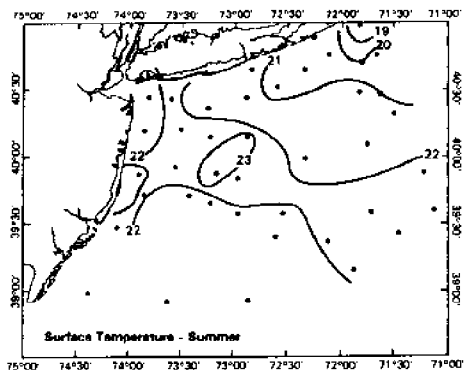
Chemical Oceanography

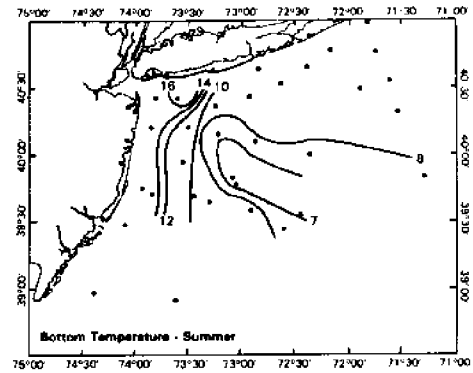
Seasonal distributions of surface and bottom temperature, salinity, and dissolved oxygen are shown on Figures C.5 through C.7 (after O'Connor et al., 1977). Average values for other properties are listed in Table C.2. Distributions of trace elements are discussed below in connection with waste discharges.

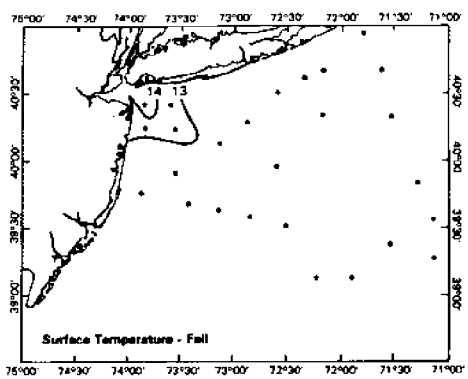
Figure C.5 Average surface and bottom temperature in New York Bight (O'Connor et al., 1977).

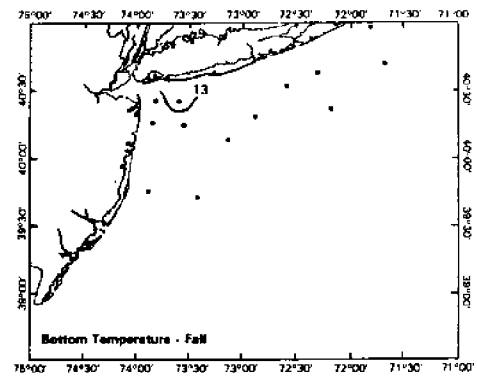


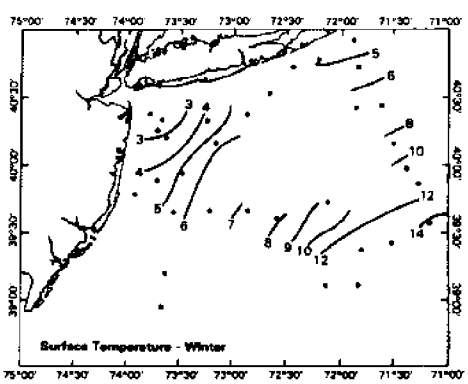












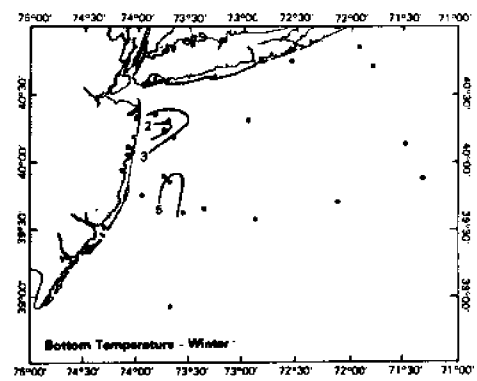


Figure C.6 Average surface and bottom salinity in New York Bight (O'Connor et al., 1977).

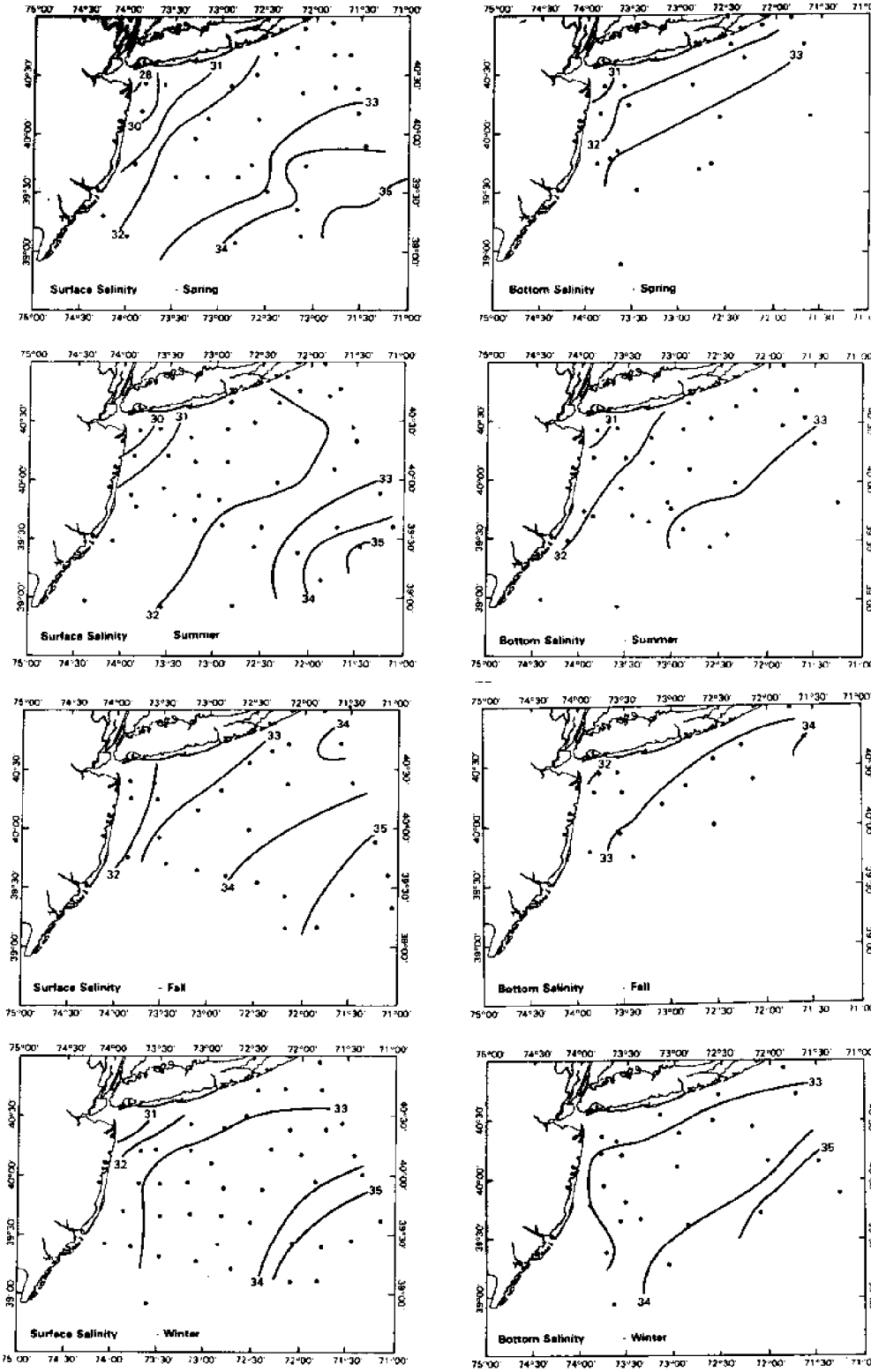


Figure C.7 Average surface and bottom dissolved oxygen in New York Bight (O'Connor et al., 1977).

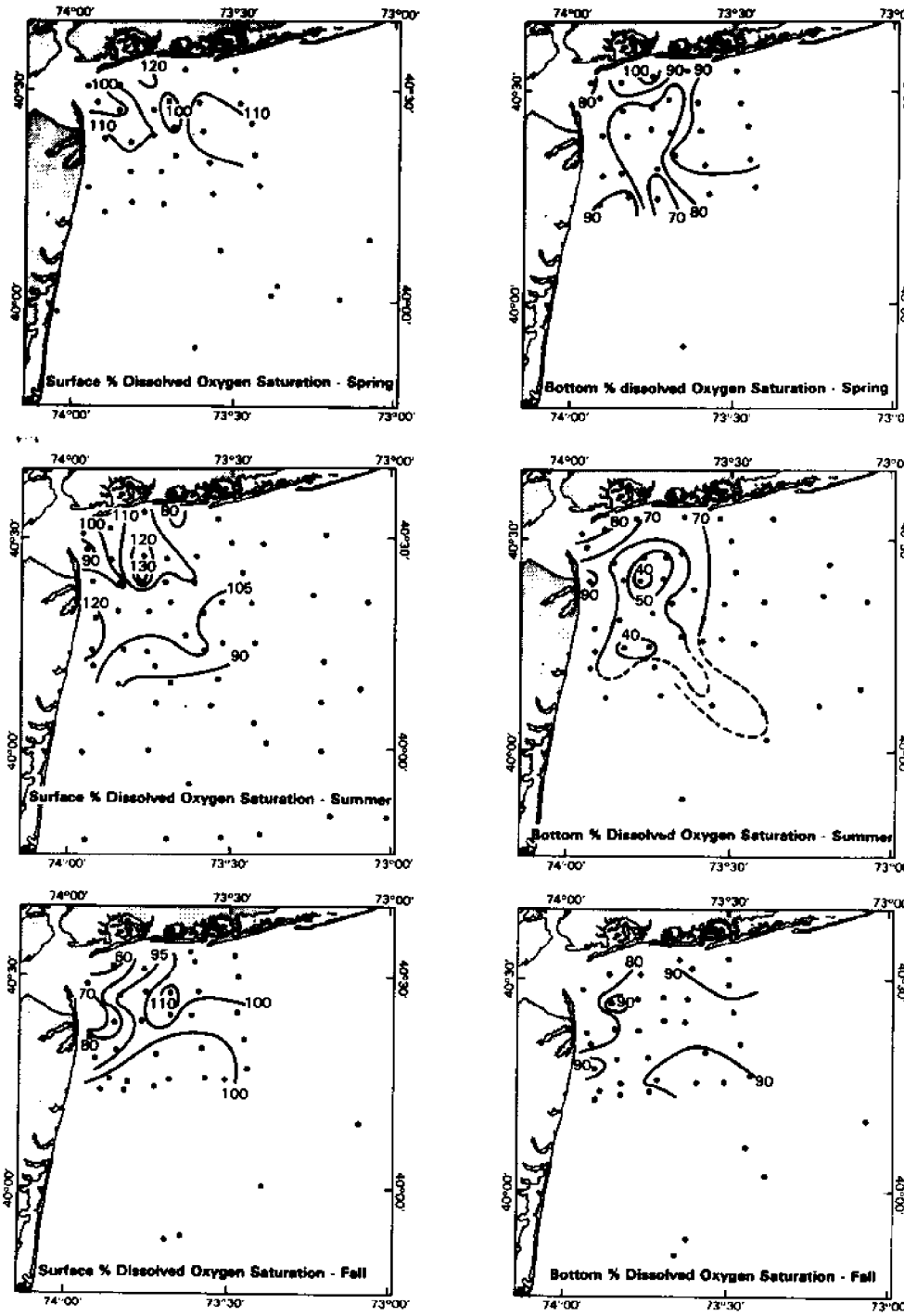


Table C.2 Average Concentrations of Materials in New York Bight (O'Connor et al. 1977)

		Spring		Summer		Fall		Winter	
		Surface	Bottom	Surface	Bottom	Surface	Bottom	Surface	Bottom
NO_3 - N	apex	.01-.05	.01-.04	.001-.08	.01-.04	.01-.05	.04-.08	.01-.1	.01-.01
(mg/l)	outer bight	.000-.004	.002-.1	.000-.007	.02-.2	.01-.04	.05-.07	.01-.1	0.01-.2
Inorg-P	apex	.005-.03	.01-.02	.01-.06	.03-.06	.02-.06	.02-.04	.02-.03	.01-.02
(mg/l)	outer bight	.002-.01	---	.01-.004	---	.007-.01	---	.007-.02	---
Chlorophyll-a (mg/l)	apex	1.-10	---	.4-8	---	3.-5	---	1.-10	---
	outer bight	.8-1.6	---	.1-.3	---	.5-2.9	---	.5-1.2	---
Iron (µg/l)	apex	20.-150	30.-100	20.-70	20.-100	10.-100	10.-200	10.-100	10.-80
Copper (µg/l)	apex	2.-9	2.-20	2.-18	1.-5	1.-7	2.-6	3.-9	3.-8

Biological Oceanography

Impacts of pollution upon living resources and their support systems in the New York Bight are a major concern. Current knowledge of the plankton in the Bight has been summarized by Yentsch (1977) and Malone (1979). These and other sources provide benchmark data for evaluating fish kills and other episodes.

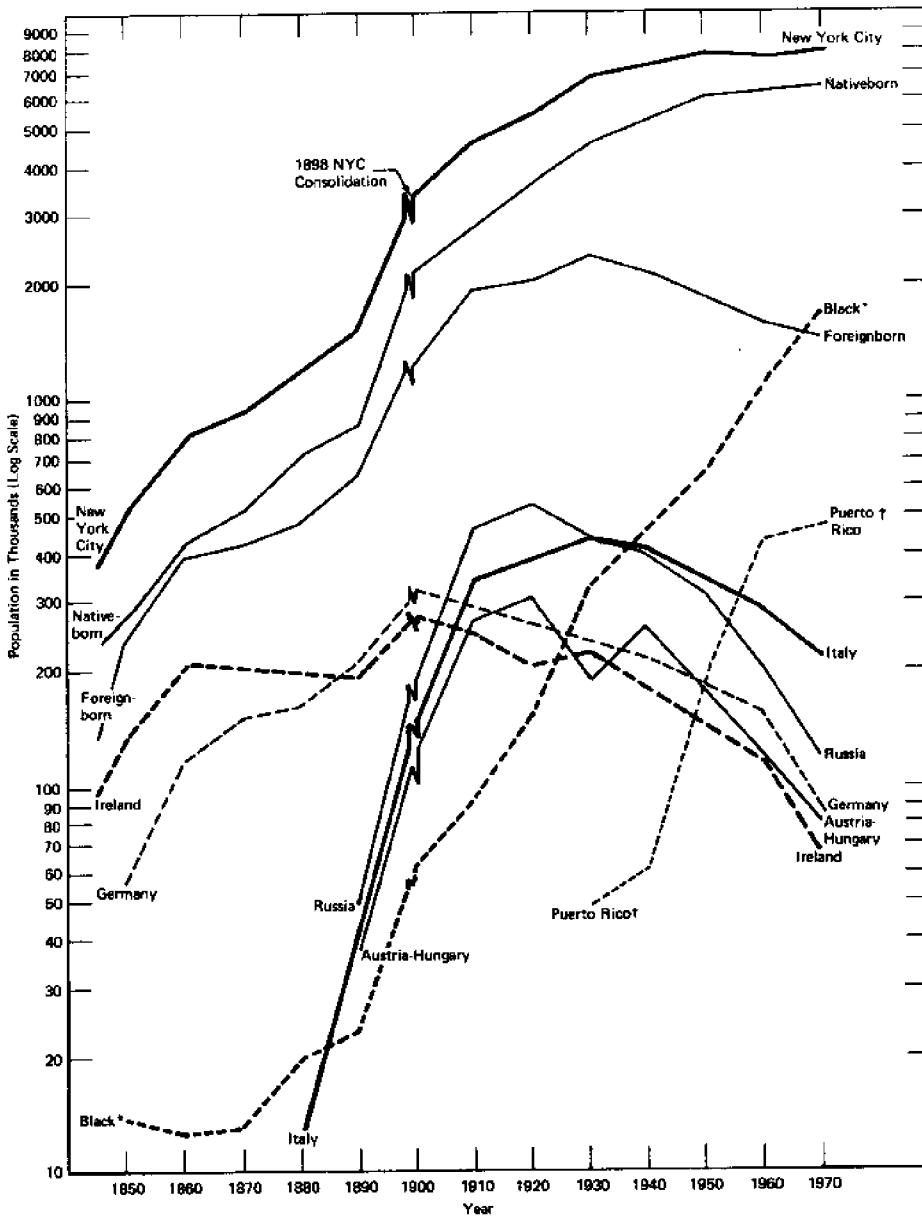
Fisheries are sensitive to both pollution and to fishing pressures. Edwards (1976) has shown recent reductions between 1963 and 1974 due in large part to overfishing of standing crops of commercial species off the northeastern United States. Depletions in stocks of haddock, herring, and other commercially important species have been noticeable. Mackerel, some of which spawn in the New York Bight, increased. One result of fishing pressures has been the imposition of a 200-mile limit within which fishing by foreign vessels is controlled, although the timing of the decrease in landings makes it clear that foreign fishing was not the cause (McHugh, 1977). Some of the decline in commercial finfishing has been offset by sport-fishing (McHugh and Ginter, 1978). In this regard, Jensen (1979) notes that increasing competition for particular species such as striped bass is leading to habitat destruction by fishing gear and disputes over areas, access, traditional rights, and legal prerogatives.

Shellfisheries also change with time. Commercial landings of the American oyster in the New York Bight have decreased from about 7,000 metric (or long) tons per year in 1950 to a low of 46 tons/year in 1967, followed by an increase to 960 tons in 1975. Hard clam landings changed from 6,000 tons in 1950 to 2,500 in 1960 to 4,700 tons in 1975 with a value of $16 million. The surf clam fishery has grown from about 3,000 tons in 1950 to about 16,000 tons from 1968 through 1975 (McHugh, 1977) but has been over-exploited in the past few years.

ECONOMIC DEVELOPMENT

Rapid population, commercial, and industrial growth of the New York metropolitan area began with 24,000 people in 1786 in lower Manhattan. Within New York City the most rapid growth rate was between 1900 and 1930; since then, population change have been mostly internal ethnic ones (See Figure C.8). By 1970 the metropolitan area included Long Island, coastal and northeastern New Jersey, Dutchess County in New York, Fairfield and New Haven counties in Connecticut, and 19.3 million

Figure C.8 Population growth and composition in New York City (Koebel and Krueckenberg, 1975).



people. This figure may reach 25.8 million by the year 2000 (Koebel and Krueckenberg, 1975). 1977 population densities range approximately from 1/km^2 (3/mi^2) in the lower Hudson Valley to 21,000/km^2 (67,000/mi^2) in New York City.

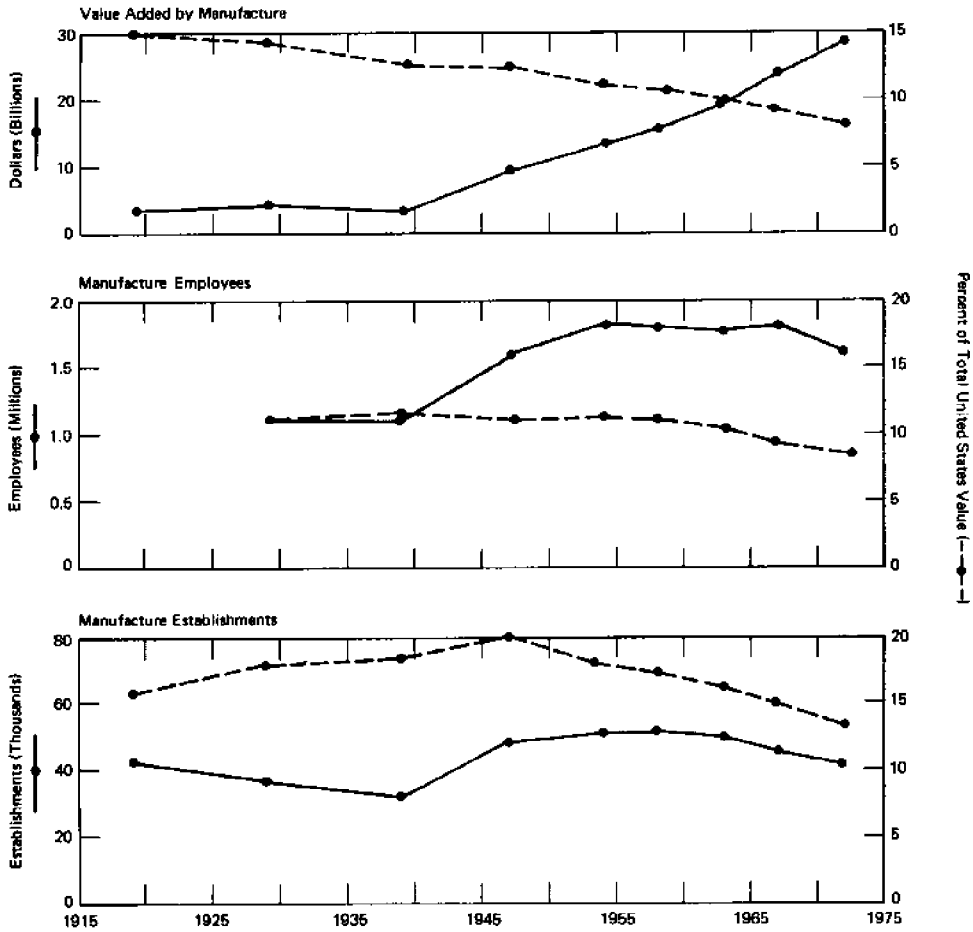
While much of the economic activity in the New York Metropolitan area is in the commercial, transportation and service sectors, manufacturing has increased steadily since 1939, with a shift toward fewer numbers of establishments and employees since about 1950 (see Figure C.9). By 1972, 1.8 million manufacturing employees in the metropolitan area contributed over $28 billion worth of value added, approximately half from the chemical, publishing, and textile industries (Mueller and Anderson, 1978).

WATER SUPPLY

Municipal water supplies for the New York Metropolitan area have increasingly relied upon imported water. The original 1842 Croton aqueduct extended about 60 km (37 mi) from Manhattan to Westchester and Putnam counties. This was paralleled by the New Croton aqueduct in 1890 and extended by 150 km (90 mi) with the Catskill aqueduct in 1915 to west of Kingston. In 1944, the Delaware aqueduct was placed in service, bringing water 135 km (85 mi) from the Delaware basin to the Hill View terminal reservoir. The safe yield of the system was assumed to be 790 m^3/s (1800 mgd) until the 1961-1967 drought revealed a yield of about 565 m^3/s (1290 mgd) (Corps of Engineers, 1977b). During this period, water stored in the system's eleven reservoirs dropped to 29 percent of the 149 km^3 (565 billion gallons) storage capacity.

During the period of system expansion, demand for water increased to match the assumed supply. Most of the water is not metered (New York City is unique among major cities of the world in this regard) and per capita consumption has increased from 380 lcd (100 gcd) in 1900 to 660 lcd (175 gcd) in 1977 (Corps of Engineers, 1977a). During 1965 and 1966, consumption was reduced from the 1960 level of 570 lcd (150 gcd) to 480 lcd (127 gcd) due to drought restrictions (Corps of Engineers, 1977b). System losses are unknown because of unmetered service; some is due to leakage which has increased by an undetermined amount since leakage detection and control activities were suspended in 1955 (Corps of Engineers, 1977c).

Figure C.9 Manufacturing in New York metropolitan area (Mueller and Anderson, 1978).



Per capita consumption in New York City is relatively high in comparison with other US cities. Of 30 cities with populations of 250,000 or more, the 1960 domestic consumption in New York of 337 lcd (89 gcd) was exceeded only by Cincinnati with 348 lcd (92 gcd) from the local Ohio River sources and by Los Angeles (365 lcd or 96 gcd) and Denver (550 lcd or 145 gcd). Like New York City, Los Angeles and Denver rely largely on imported supplies but receive less than 40 cm (16 in) per year precipitation compared to about 120 cm (48 in) in the New York metropolitan area. Other cities in the comparison used from 138 to 291 lcd (36 to 77 gcd) with mean and median values of 212 lcd or 56 gcd (Corps of Engineers, 1977c).

Potential water conservation measures are estimated to yield 23 lcd (6 gcd) from metering, 11 lcd (3 gcd) from leakage control, 15 lcd (4 gcd) from domestic conservation devices, and 68 lcd (18 gcd) from drought contingency measures for a total of 117 lcd (31 gcd) by the year 2000 for New York City. An additional reduction of 265 lcd (70 gcd) is predicted to result from industrial recirculation mandated by PL 92-500. However, the Corps of Engineers (1977b) ignored these potential savings and, following conventional engineering practice, predicted an increase from a 1970 level of 670 lcd (177 gcd) to 685 lcd (181 gcd) in the year 2000. These figures yield average water uses of 61 m^3/s (1400 mgd) in 1970 and 67 m^3/s (1530 mgd) in the year 2000.

WASTE DISPOSAL

Waste Loadings

Major municipal and industrial discharges leading directly or indirectly to the New York Bight are shown on Figures C.10 and C.11, respectively, and listed in Tables C.3 and C.4.

Mueller et al. (1976a,b, 1978), Gross (1976b), and Hazen and Sawyer (1978) have compiled information on waste loadings (which include both water supply and storm flows) to the New York Bight region. In 1972, 127 major municipal discharges contributed 114 m^3/sec (2,600 mgd) of sewage of which 21 m^3/sec (480 mgd) were untreated. By 1978, the untreated sewage had been reduced to 9 m^3/sec (295 mgd); 1972 industrial discharges process amounted to 27 m^3/sec (610 mgd) of which 47 percent went through municipal systems. Another 210 m^3/sec (4800 mgd) of cooling waters were discharged.

Figure C.10 Municipal discharges.



Figure C.11 Industrial discharges



Table C.3 Major Municipal Discharges[a]

Location Figure 10	Source	Year	Flow[b] m^3/s	mgd	Est. Percent industrial	Treatment[c]	Tributary population, 1975
1	Orangetown SD #2	--	0.66	15	--	S	--
2	Rockland Co. Dist. #1	--	0.96	22	0.7	S	--
3	Yonkers Joint Treatment	--	4.1	92	6.6	S	--
4	West New York	1953	0.44	10	--	P	--
5	Hoboken	1958	0.74	17	40	P	--
6	Jersey City - East	--	1.6	37	--	P	--
7	New York City - Owls Head	1952	4.4	100	1.3	MA	720,800
8	-Newtown Creek	1967	13	300	6.8	MA	968,300
9	-Wards Island	1936	11	245	1.0	S	1,278,400
10	-Bowery Bay	1942	5.6	128	6.8	S	698,500
11	-Hunts Point	1952	7.0	160	0.9	S	717,200
12	-Tallmans Island	1938	3.1	70	2.6	S	405,000
13	-Jamaica	1943	4.4	100	2.1	S	573,500
14	-26th Ward	1945	3.5	80	1.6	S	338,200
15	-Coney Island	1935	4.4	100	0.3	MA	692,900
16	-Rockaway	1952	0.88	20	0.1	S	118,700
17	-Oakwood Beach	1954	0.88	20	0.01	S	156,600
18	Bay Park (Pearsall's Hassock)	---	3.0	68	9.8	S	---
19	Middlesex Co. Sanitary Auth.	1958	5.3	120	50	S	---
20	Perth Amboy	---	0.31	7	46	P	---
21	NYC - Port Richmond	1953	1.6	36	14	P	171,600
22	Linden - Roselle Joint Meeting	---	1.1	26	30	S	---
23	Rahway Valley	---	1.5	35	26	S	---
24	Essex - Union Joint Meeting	---	3.3	75	17	S	---
25	Bayonne	1954	0.74	17	22	P	---
26	Passaic Valley	1924	13	300	39	P	---
27	Jersey City - West	---	1.1	24	24	P	---
28	Bergen Co. Sanitary Auth.	---	2.8	64	20	S	---
29	Atlantic City	---	0.70	16	1.2	P	---

[a] Mueller and Anderson (1978); Anderson et al. (1979); Interstate Sanitation Commission (1979)

[b] Not including 6.0 m^3/s (137 mgd) primary and secondary effluent from 138 smaller communities or 8.8 m^3/s (200 mgd) raw sewage from Manhattan and Brooklyn

[c] 1980 baseline treatment for New York City and most New Jersey discharges

P = primary; MA = modified aeration; S = secondary

Table C.4 Direct Industrial Discharge Locations and Wastewater Flows, 1972 (From Mueller and Anderson, 1978)

ID No.[a]	Waste Source	Flow (mgd)
TRANSECT ZONE		
	Hudson River	
1	Little Falls Paper	0.5
2	Cornwall Paper	1.6
3	Standard Brands	19.3
4	Kay Fries Chemical	1.2
5	Penn Central	0.2
6	Continental Can	2.6
7	Anaconda Wire	0.7
8	CPC International	8.6
9	Spencer Kellog	1.2
10	Lever Bros.	7.7
11	Wilson-Martin Div.	0.6
12	Colgate Palmolive	11.3
	East River	
13	Consolidated Edison Co**	1,100.0
14	Amstar Corporation	9.3
15	Consolidated Edison Co**	202.0
16	Phelps-Dodge Ref. Co.	0.5
17	Sun Oil co.	0.0*
18	Rendering Co. of America	0.6
19	Pinkas Fischer Co.	0.1
20	Van Iderstine Co.	3.1
21	Pepsico Inc.	5.0
22	Consolidated Edison Co**	1,390.0
23	Consolidated Edison Co**	490.0
24	Port of NY Auth.-La Guardia	0.5
	Upper Bay	
25	Bush Terminal Assoc.	1.0
	Jamaica Bay	
26	Port of NY Authority	0.8
27	Port of NY Authority	1.0
	Lower, Raritan, and Sandy Hook bays	
28	Lanvin Charles of the Ritz	0.1
29	IFF Inc.	0.2
30	Owens-Illinois Inc.	0.3
	Arthur Kill and tributaries	
31	Nassau Smelting & Ref.	0.6
32	AM Smelting & Ref.	7.3
33	Chevron Oil Co.**	63.0
34	American Cyanamid Co.	0.0*
35	Amerada Hess Corp.	1.0
36	US Metals Ref. Co.	32.0
37	FMC Corp.	4.9
38	GAF Corp.	20.0
39	El DuPont De Nemours Co.	18.4
40	Humble Oil & Ref. Co.**	202.0
41	Phelps Dodge Copper Products	2.2
83	Scherring Corp.	0.8
	Kill Van Kull	
42	Exxon Co.	2.8
43	Humble Oil & Ref. Co.	3.6
44	United States Gypsum Co.	1.0
45	General Cable Corp.	0.2
	Newark Bay	
46	Singer Co.	1.2
47	Allied Chemical Corp.	0.6
48	A. Gross Co.	2.6
49	Anheuser-Busch Co.	0.0*
50	Vulcan Materials Co.	15.0
50a	El DuPont De Nemours Co.	0.9
	Raritan River and tributaries	
51	Jersey Central P & L Co.**	198.0
52	Anaconda Co.	2.2
53	Panacon Corp.	0.5
54	Tenneco Chemicals Inc.	1.6
55	Jersey Central P&L Co.**	302.0
56	Tenneco Chemicals Inc.	0.4
57	Anheuser-Busch Inc.	0.7
58	Triangle Conduit & Cable Co.	0.0*
59	Johnson & Johnson	0.0*
59a	Ford Motor Co.	0.3
84	American Cyanamid	23.8
85	General Amer. Transport	0.0*
86	Johns Manville	13.1
87	Ortho Diagnostics	0.5
88	Hercules Inc.	2.1
	Rahway River	
60	American Cyanamid Co.	2.5
61	Middlesex Water Co.	23.2
62	Rheem Manufacturing Co.	0.0*
63	Merck & Co. Inc.	0.4
	Passaic River and tributaries	
64	Western Electric Co.	8.3
65	Essex Chemical Corp.	5.8
89	Marcal Paper Mills Inc.	1.2
90	Curtiss-Wright Corp.	0.3
91	Whippany Paper Board	7.9
92	SB Penick	0.4
93	Ford Motor Co.	1.4
94	Tenneco Chemical Co.	5.5
95	Curtiss-Wright Corp.	1.4
	Hackensack River	
69	Union Textile Printers Inc.	0.6
70	Koppers Company Inc.	0.8
71	Tanatex Chemical Co.	0.5
72	Diamond Shamrock Corp.	4.1
73	Spinnerin Yarn Co., Inc.	0.4
74	Universal Oil Products	0.9
75	Aeroil Products Co., Inc.	1.8
	NEW JERSEY COASTAL ZONE	
	Toms River	
76	Toms River Chemical Corp.	11.2
	Direct Bight	
77	Toms River Chemical Corp., Dover Township	5.2
	Manasquan River	
78	Borden, Inc., Pt. Pleasant	0.0*
79	Nestle	0.9
	Southern bays	
81	Atlantic City Elect. Co.**	831.0
82	Atlantic City Elect. Co.**	14.3

Notes:
* Less than 0.1 mgd
[a] See Figure 11
** Cooling water discharge

Ocean dumping at locations shown on Figure C.2 is the most visible discharge to the Bight; it even can be seen from satellites. Mueller et al., (1976a, b) report an annual total of 27.6 x 10^6 m^3 (21.7 x 10^6 yd^3) dumped of which 53 percent is dredge spoils, 26 percent is sewage sludge with about 5 percent solids, 15 percent is acid waste, 3 percent is chemical waste and 3 percent is building and construction debris. Dredge spoils account for most of the sediments entering the Bight; of these, about half derive from sands moved by littoral drift and estuarine bottom flow into the harbor (Gross, 1976b). Because of sediment contamination by waste discharges to upstream dredged channels, at least half of the cadmium, chromium, copper, and iron entering the Bight is associated with dredge spoils (Mueller and Anderson, 1978).

Tables C.5 and C.6 list sources and loadings of materials currently entering the New York Bight. Estimates of future quantities of wastes depend upon assumptions of growth, funding, effectiveness, and timing of remedial projects and programs and upon technological and institutional developments in resource conservation and recovery. Sewage treatment is being mandated to remove 85 percent of the suspended solids and BOD. This results in a three-fold increase in the volume of sludge to be handled. Legislation is being studied that would stop the present practice of sludge dumping and alternative landbased methods are being considered. Meanwhile, additional areas are being sewered; sewage treatment plants are being built; effluent outfalls are being constructed from drainage basins tributary to the Bight; consideration is being given to removal of floatables from storm water overflows for aesthetic reasons (there is no evidence of a health hazard from this source); and 6.6 m^3/s (150 mgd) of raw sewage continues to be discharged from Manhattan's West Side (one major hotel there even invites its guests to flush razor blades down the toilets).

Anderson and Mueller (1978) have estimated future loadings of some waste constituents assuming secondary (biological) treatment and effluent chlorination of municipal sewage. They further assume that mass loadings of industrial, atmospheric, ocean dumping, and urban runoff loadings will remain the same as at present. While it can be argued that implementation of regulations which limit discharge of toxic or hazardous substances will reduce the amounts of materials entering the Bight, the large reservoir of these materials in estuarine and river sediments and on land

Table C.5 Total Mass Loads into the New York Bight (Mueller et al., 1976a,b)

Parameter	Mass Load metric tons/day	Percentage contribution by location: Direct Bight	Hudson and Passaic River Basins	New Jersey Coast	Long Island Coast
Flow, cfs(m^3/s)	82,700 (2340)	59	36	4	1
SS	24,000	68	31	0.6	0.1
ALK	5,100				
BOD_5	2,100	30	67	3	0.5
COD	10,000	42	48	9	0.5
TOC	2,600	37	58	4	0.6
MBAS	59		95	4	0.5
O & G	870	38	53	9	0.6
NH_3-N	210	28	67	3	2
ORG-N	190	27	68	3	2
TKN	400	27	68	3	2
NO_2+NO_3-N	120	33	55	10	2
Total-N	520	29	65	4	2
ORTHO-P	51	1	91	3	5
Total-P	138	51	45	2	2
Cd	2.4	84	15	0.5	0.07
Cr	5.0	51	44	0.6	4
Cu	13.8	54	45	0.9	0.2
Fe	230	82	16	2	0.2
Hg	0.30	9	85	6	0.6
Pb	12.7	53	46	0.5	0.2
Zn	33	47	52	0.2	0.3
Fecal Coli -winter	5.6×10^7	< 0.01	100	0.2	<0.001
-summer	4.9×10^7	< 0.01	100	0.2	<0.001
Total Coli -winter	21×10^7	< 0.01	100	0.1	<0.001
-summer	11×10^7	< 0.01	100	0.2	<0.001

Table C.6 Total New York Bight Loads by Source (Mueller et al., 1976a,b)

Percentage Contribution

	Direct Bight		Coastal Zone				
			Wastewater		Runoff		
Parameter	Barge	Atmospheric	Municipal	Industrial	Gaged	Urban	Groundwater
Flow	0.02	59	5	0.4	33	2	0.4
SS	63	5	4	0.2	16	12	Nil
ALK	1	Nil	35	0.3	59	5	0.03
BOD_5	21	9	48	2	11	9	0.01
COD	32	10	35	1	13	9	0.01
TOC	25	12	29	1	18	15	0.02
MBAS			86		5	9	0.05
O & G	38		22	0.7	16	23	
NH_3-N	24	4	55	3	10	4	0.04
ORG-N	19	9	45	2	21	5	0.02
TKN	21	6	51	2	15	5	0.02
NO_2+NO_3-N	0.07	33	6	0.3	60	0.6	0.7
Total-N	16	13	40	2	25	4	0.2
ORTHO-P		1	72		18	9	Nil
Total-P	50	0.7	35	1	9	4	Nil
Cd	82	2	5	0.6	5	5	0.001
Cr	50	1	22	0.8	10	16	Nil
Cu	51	3	11	9	10	16	0.006
Fe	79	3	5	0.5	6	6	0.01
Hg	9		71	2	13	5	
Pb	44	9	19	3	6	19	0.004
Zn	29	18	8	2	21	22	0.009
F.Coli							
-winter	<0.01	Nil	87	0.2	0.01	13	Nil
-summer	<0.01	Nil	85	0.2	0.01	15	Nil
T.Coli							
-winter	<0.01	Nil	91	0.1	0.05	9	Nil
-summer	<0.01	Nil	84	0.2	0.1	16	Nil

surfaces continues to be a source of Bight pollution. The Anderson and Mueller results are summarized on Tables C.7 and C.8 which, by comparison with Tables C.5 and C.6, predict little overall change in loadings to the Bight. These results are consistent with studies made for the Section 208 Areawide Waste Treatment Management Planning Program (Hazen and Sawyer, 1978), as well as others made for major US inland streams (see, e.g., Mackenthum, 1961). Clearly, if stresses of waste discharges on the Bight are to be reduced, quite different waste management strategies will be required.

EFFECTS OF WASTE DISPOSAL

Estuarine Pollution

Figure C.1 shows locations of estuarine areas and transects for which present water quality values are listed on Table C.9. Mean values for DO concentrations are as low as 0.7 mg/l in the Kills and 1.0 mg/l in the Harlem River. High Kjeldahl nitrogen and phosphorus levels are found in the Kills, Jamaica Bay, and Hackensack River. Mean high concentrations of heavy metals occur throughout the estuary, with 140 mg/l Zn in the Hudson River, 8.2 mg/l Cd and 10 mg/l Cu at Rockaway Beach, 37 mg/l Pb in the Hackensack River, and 50 mg/l Ni in the Kills.

Bacteriological pollution is greatest in the Harlem River with 100,000 fecal coli/100 ml. Steady-state levels of coliform pollution are explained by times of 40 to 70 hours for 90 percent reductions due to sedimentation and mortality. Coliforms ordinarily disappear from marine surface waters much more quickly except for areas where they are resuspended by waves, currents, and ship propellors from contaminated bottom sediments (Rittenberg et al., 1958; Gunnerson, 1959 and 1975a).

Shoreline Pollution

Coastal surf zone and beach pollution includes enteric bacteria in the water and by floating and stranding of anthropogenic flotsam. It does not come about by the resuspension and shoreward transportation of sewage sludge envisaged as the legendary Sludge Monster (MESA, 1975). 1973-1977 data on coliform bacteria concentrations in the surf zone are summarized in Table C.10. Numbers of samples per year varied from 6 to 48, mostly in summer. Differences between stations reflect proximity to waste discharges (Coney Island is within

Table C.7 Present and Future Municipal Influent, Effluent and Sludge Loads (Anderson and Mueller, 1978).

Parameter	Present (1972-73) Loads (metric tons/day) Influent	Effluent[a]	Sludge[a]	Future (2000) Loads (metric tons/day) Influent Load	%Change	Effluent Load	%Change	Sludge Load	%Change
Flow, mgd	2,600	2,600	3	3,200	+23%	3,200	+23%	8	+170%
Suspended Solids	1,700	854	450	2,200	+29%	200	-74%	1,220	+171%
Organic Carbon	1,070	735	110	1,340	+25%	400	-46%	290[3]	+164%
Total Nitrogen	230	208	17	290	+26%	220	+ 6%	72	+324%
Total Phosphorus	52	48	4.7	70	+35%	49	+ 2%	21	+347%
Lead	3.4	2.7	0.72	4.07	+20%	3.26	+21%	0.81	+12%
Fecal Coliform without municipal chlorination[b]	--	4.9×10^7	3,600	--	--	1.3×10^7	-73%	9,800	+172%
Fecal Coliform with municipal chlorination[b]	--	4.2×10^7	3,600	--	--	1,900	-100%	9,800	+172%

[a]Mueller et al., 1975

[b]Coliform load (=) 10^{10} organisms/day.

[c]Based on TOC:SS ration (0.24) in present sludge (Mueller et al., 1975).

Table C.8 Total Future Waste Loadings by Source (Anderson and Mueller, 1978)

Parameter	Mass Load metric tons/day	Percentage Contribution						
		Barge	Atmospheric	Wastewater		Runoff		
				Municipal	Industrial	Gaged	Urban	Groundwater
Flow, cfs	83,800	0.03	58	5.9	0.5	33	2.5	0.3
SS	24,000	67	4.9	0.9	0.4	14	13	Nil
TOC	2,400	34	13	16	2.1	18	17	Nil
Total-N	590	24	11	37	2.0	22	3.6	Nil
Total-P	155	55	0.6	32	1.5	7.7	4.1	Nil
Pb	13.6	42	8.8	24	2.2	5.7	18	Nil
F.Coli without chlorination*	2.0×10^7	0.05	Nil	65	0.5	0.04	36	Nil
F.Coli with chlorination*	7.4×10^6	0.1	Nil	0.03	1.3	0.09	99	Nil

*Coliform load (=) 10^{10} organisms/day.

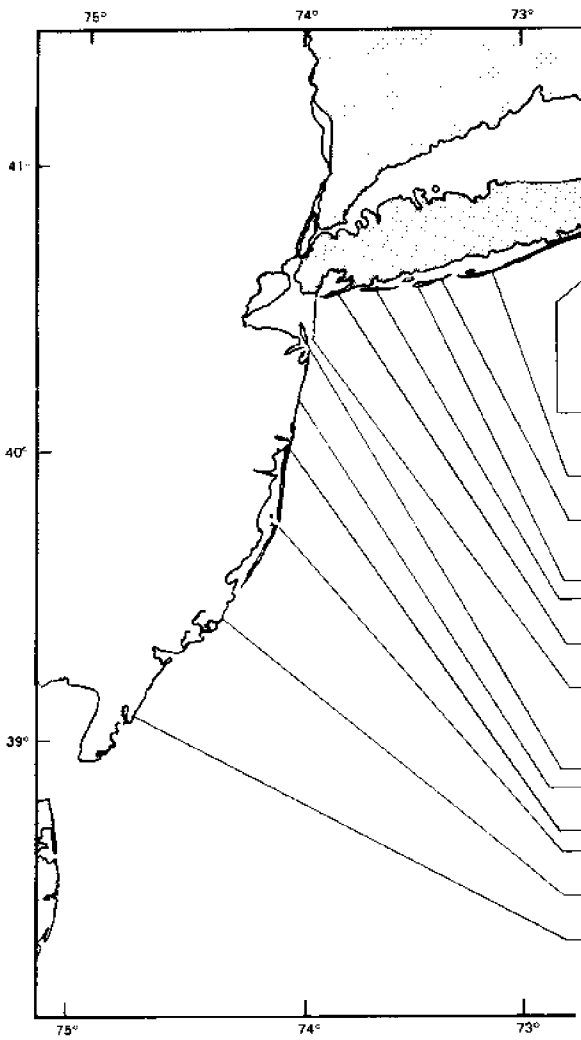
Table C.9 Average and Worst Mean Values of Pollutant Concentrations in NYC 208 Study Area Transects (Hydroscience, 1978a)

Transect	DO (mg/l) Avg	(Max)	BOD_5 (mg/l) Avg	(Max)	E.Coli/100 ml Avg	(Max)	F.Coli/100ml Avg	(Max)	TKN (mg/l) Avg	(Max)	NO_3-N (mg/l) Avg	(Max)	TSS (mg/l) Avg	(Max)	T-PO_4-P (mg/l) Avg	(Max)
Hudson River	5.0	(2.0)	1.5	(2.0)	1500	(70,000)	800	(10,000)	0.8	(1.0)	0.4	(0.7)	10	(28)	0.3	(0.4)
Kills	2.5	(0.7)	2.3	(4.6	50,000	(100,000)	8,000	(50,000)	1.9	(2.3)	0.1	(0.2)	8	(10)	0.6	(0.7)
Harlem River	2.0	(1.0)	2.2	(2.8)	100,000	(500,000)	10,000	(100,000)	1.3	(1.3)	0.2	(0.2)	10	(14)	0.8	(1.3)
East River	4.0	(2.0)	1.8	(2.2)	5,000	(40,000)	900	(7000)	0.8	(1.0)	0.1	(0.2)	9	(14)	0.6	(0.9)
Hackensack River	3.0	(2.5)	2.5	(2.6)	20,000	(40,000)	7,000	(10,000)	1.8	(1.9)	0.2	(0.2)	6	(7)	0.5	(0.6)
Raritan Bay	6.0	(4.0)	2.0	(7.2)	800	(7,000)	100	(500)	1.1	(1.5)	0.1	(0.2)	7	(13)	0.4	(0.4)
Jamaica Bay	6.0	(4.0)	2.0	(7.2)	2,000	(10,000)	300	(1,100)	1.0	(2.0)	0.1	(0.1)	5	(7)	0.3	(0.4)
Rockaway Beach	7.0	(7.0)	1.0	(1.0)	60	(100)	30	(70)	0.3	(0.3)	No Data		4	(5)	0.3	(0.3)

Transect (µg/l)	Zn Avg	(Min)	Cd Avg	(Max)	Cr Avg	(Max)	Cu Avg	(Max)	Pb Avg	(Max)	Ni Avg	(Max)	Hg Avg	(Max)	PCB Avg	(Max)
Hudson River	40	(140)	2.0	(5.5)	<25	(<25)	15	(27)	5	(10)	5	(10)	<1	(<1)	0.1	(0.2)
Kills	80	(110)	1.8	(4.0)	<25	(<25)	30	(54)	15	(22)	30	(50)	<1	(<1)	0.02	(<0.02)
Harlem River	5	(7)	<1	(<1.0)	<25	(<25)	12	(14)	10	(11)	10	(10)	<1	(<1)	No Data	
East River	30	(60)	<1	(<1.0)	<25	(<25)	10	(12)	<5	(10)	6	(18)	<1	(<1)	0.15	(0.30)
Hackensack River	75	(90)	1.8	(2.5)	<25	(<25)	30	(47)	25	(37)	25	(25)	<1	(<1)	No Data	
Raritan Bay	50	(130)	<1	(<1.0)	<25	(<25)	12	(20)	5	(9)	15	(29)	<1	(<1)	No Data	
Jamaica Bay	35	(50)	1.0	(2.0)	<25	(<25)	9	(18)	<5	(5)	5	(10)	<1	(<1)	No Data	
Rockaway Beach	20	(20)	4.0	(8.2)	<25	(<25)	19	(140)	5	(15)	3	(3)	<1	(<1)	No Data	

Notes:
1. DO, BOD_5, T.Coli, F.Coli, NO_3-N, PCB Data obtained during Summer, 1977
2. TSS, T-PO_4-P Data obtained during September, 1975
3. Heavy metals data obtained during Winter, 1976
4. Any "<" implies that the data obtained was below the limit of detectability

Table C.10 Annual Geometric Mean Coliform Concentrations in Waters along Long Island South Shore.



	1973	1974	1975	1976	1977
New York					
Suffolk County					
East Hampton - Atlantic Ave.	1.8	- -	- -	b/60	b 10
Main	1.8	- -	- -	b/120	b 170
Georgica	1.8	- -	- -	b/20	b 370
Islip - Saltaire Village	234	- -	- -	b/20	b/120
Babylon - Cedar	10	- -	- -	b/150	b/70
Overlook	270	- -	- -	b/10	b/110
West Gilgo	20	- -	- -	b/240	b/100
Nassau County					
Ocean Beach - East	7	5	6	7	10
Ocean Beach - West	18	15	26	22	25
New York City					
Rockaway - Rus Beach	54	62	- -	9	22
116th St	69	79	45	16	15
Coney Island - W 23rd	4500	1200	300	900	1000
W 24th	2500	1000	1000	1900	9000
New Jersey					
Monmouth County					
Sandy Hook	-	-	25	38	c/20
Asbury Park	-	-	39	30	c/170
Ocean County					
Pt. Pleasant	-	-	c/330	65	29
Barnegat Light	-	-	c/20	c/70	c/20
Atlantic County					
Atlantic City	-	-	c/130	c/20	34
Cape May County					
North Wildwood	-	-	33	17	28

(a) Total coliforms for New York; fecal coliforms for New Jersey. Data compiled from Report of Bathing Beach Sampling Analyses 1970 to 1973, and 1976 and 1977 Bathing Beach Quality Reports, Dept. of Health Services, Suffolk County, NY; Nassau County Water Quality Assessment Reports for 1976 and 1977 Report Years, Dept. of Health, Nassau County, NY; Beach and Harbor Water Sampling Program, Reports for 1973 through 1977, Bureau of Public Health Engineers, City of New York, NY; and annual reports of counties to State of New Jersey Cooperative Coastal Monitoring Program 1975 through 1977.

(b) Geometric means for all stations <10. Maximum value is included for year to year comparisons only.

(c) Geometric mean fecal coliform concentration <20. Maximum value is included for year to year comparisons only.

the estuary) and differences in sampling, analytical procedures, and methods of data analysis and reporting. Differences between years at individual stations are real and reflect changes in waste discharges and treatment and in runoff.

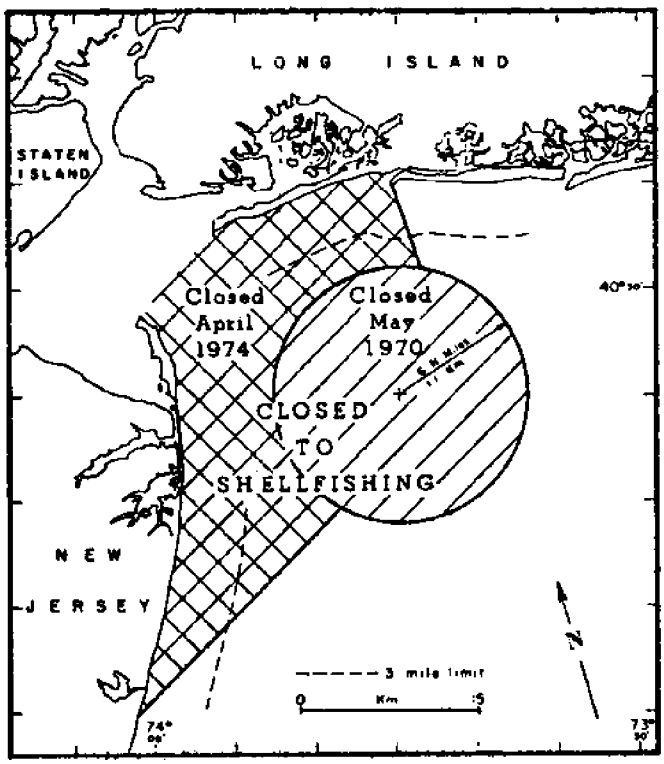
Public health benefits of ocean bathing water protection are elusive. Nevertheless bacteriological standards have been applied, beginning with the California bathing water standard of 1000 total coliforms per 100 ml for at least 80 percent of the time (Gillespie, 1943). This is comparable to the current EPA guideline of a mean fecal coliform density of 200/100 ml. The California standard has been copied or adapted throughout the world although it is strictly applicable to populations with low incidence of disease. Recent work by Cabelli and his associates (see Chapter 8 of this volume) in the New York Bight and elsewhere provides a measure of the risk involved in swimming in polluted water. Subclinical respiratory and gastrointestinal symptoms were directly correlated with beach pollution at Coney Island and at the Rockaway Beaches. Enterococci (Streptococcus fecalis, S. faecium, and some S. avium) are generally more reliable than total or fecal coliforms in assessing subclinical health effects of pollution. This has been confirmed by more recent work. Levin et al. (1975) have described the enumeration technique for enterococci; for given levels of accuracy and precision, it is simpler and correspondingly cheaper than that for coliforms. While enterococci survival in laboratory experiments exceeds that for coliforms, comparative in situ studies of their survival in the sea are needed before enterococci data can be substituted for coliform data now used for system design and monitoring (Gunnerson, 1975a).

Nearshore Effects

The most obvious economic impact of waste discharges into the Bight has been the bacteriological contamination of shellfisheries resulting in a quarantine being placed upon the area shown in Figure C.12. The configuration of the area shows the effects of ocean dumping of sewage sludge and of Hudson River discharges. Within New York Harbor, shellfisheries and some finfisheries have long been stressed, abandoned, or quarantined because of pollution problems.

Other nearshore effects are evident in the sediments. Comparison of bathymetric surveys in 1936 and 1973 reveal that the most significant change has

Figure C.12 Areas closed to shellfishing because of bacteriological pollution.



occurred at the dredged material dump site where 93 x 10^6 m^3 of sediment have accumulated resulting in up to 10 m of shoaling (see Figure C.13). Elsewhere over the 718 km^2 Bight Apex, there has been general erosion so that the net change is 1 x 10^6 m^3 of deposition (Freeland and Swift, 1978).

However, most of the 15 x 10^6 m^3 of dredge spoils dumped annually is dispersed and moves down slope. Figure C.13 shows concentrations of mercury, cadmium, copper and zinc in surficial sediments. Higher concentrations in sediments are found in and near topographic lows. Movement of sediments down the Hudson Canyon is shown by copper and zinc concentration; lead moves similarly (Gunnerson, 1981). The mobility of mercury is qualitatively different from that of the other metals, presumably because it is bound to finer sediments are influenced more by near-surface coastal currents while they settle than by subsequent movement.

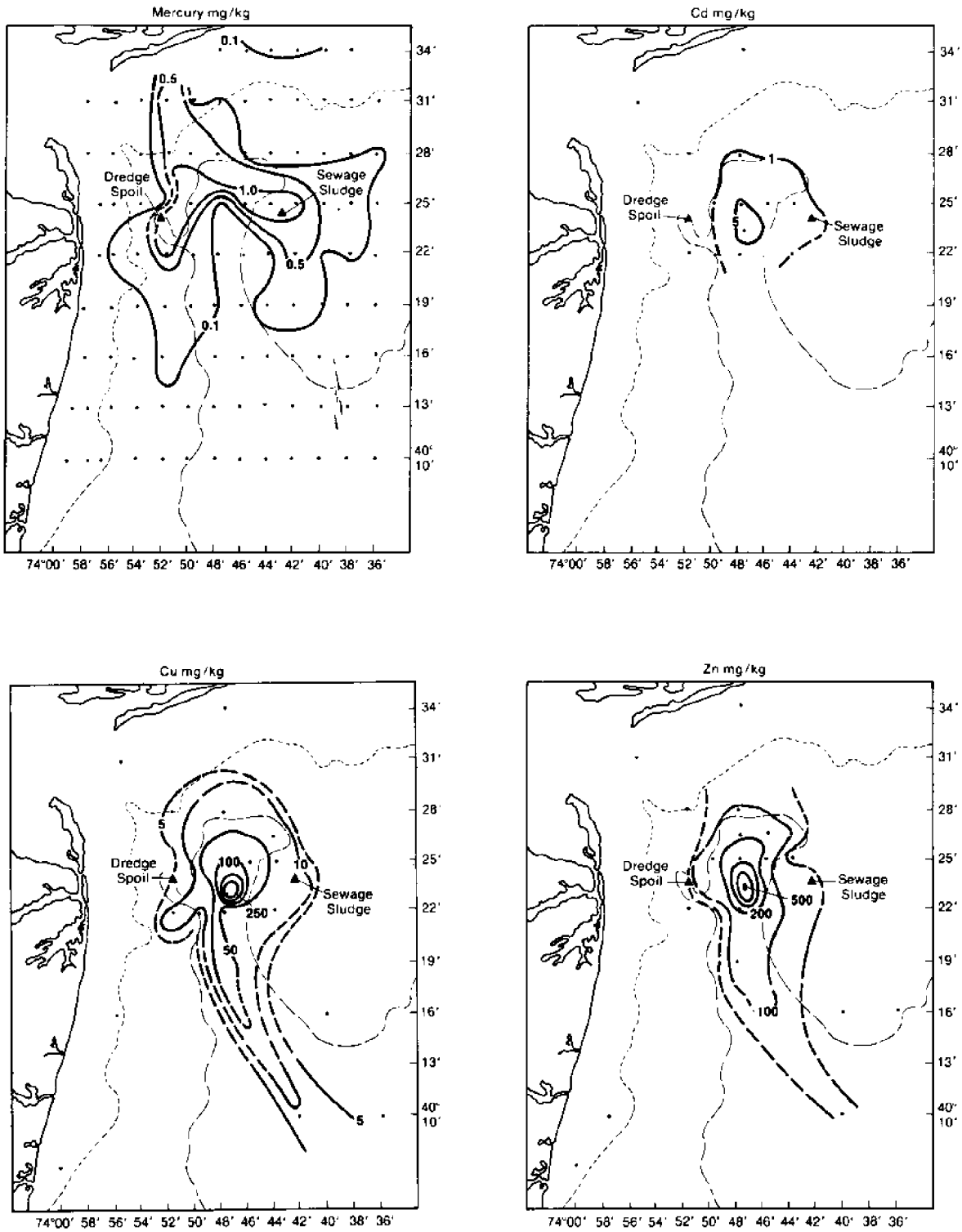
PCBs are also found in sediments at the head of Hudson Shelf Valley between the sewage sludge and dredge spoil dumping sites where concentrations of 1500 ppb are found. Up to 6200 ppb total PCB has been found in analyses of sewage sludge samples directly from the treatment plant (MESA, 1978).

Biological impacts of waste disposal are more diffuse (Swanson, 1979). Benthic invertebrates are sedentary and indicate coastal contamination. Most species found in the Apex in the late 1960s were again found in 1973 and 1974. Density of individuals is expected to vary considerably over the area because of contamination from several sources.

On the fringes, the generally enriched sediment probably supports greater than normal populations of many species (Hume et al., 1959). Nearer the center of contamination, most populations fall below the normally expected levels, except within the topographic depression at the head of the Shelf Valley. A similar relationship between diversity and the contaminated sediments occurs. Opportunistic species predominate in the most highly contaminated regions. Commercial-size surf clams (Spisula solidissima) are unusually rare in the area surrounding the Apex dump sites; adult clams are found along the shallow margins of the Apex (Swanson, 1979).

Recent events lead to the question as to whether portions of the Bight are becoming eutrophic. If so, this could be the first such case in open coastal

Figure C.13 Heavy metals in New York Bight sediments.



waters (Swanson, 1979). In 1974 oxygen concentrations were as low as 30 percent saturation in bottom waters at the head of the Hudson Shelf Valley (Segar and Berberian, 1976). In 1976 oxygen deficient (< 2.0 ml/l bottom waters, sometimes with zero dissolved oxygen levels, occurred over an extensive area off the coast of New Jersey resulting in mass fish and shellfish mortalities. Smaller mortalities also were noted in 1968, 1971 and 1974. In 1977 oxygen depletion again occurred but was limited to a band 6-9 km off the New Jersey coast (See Figure C.14).

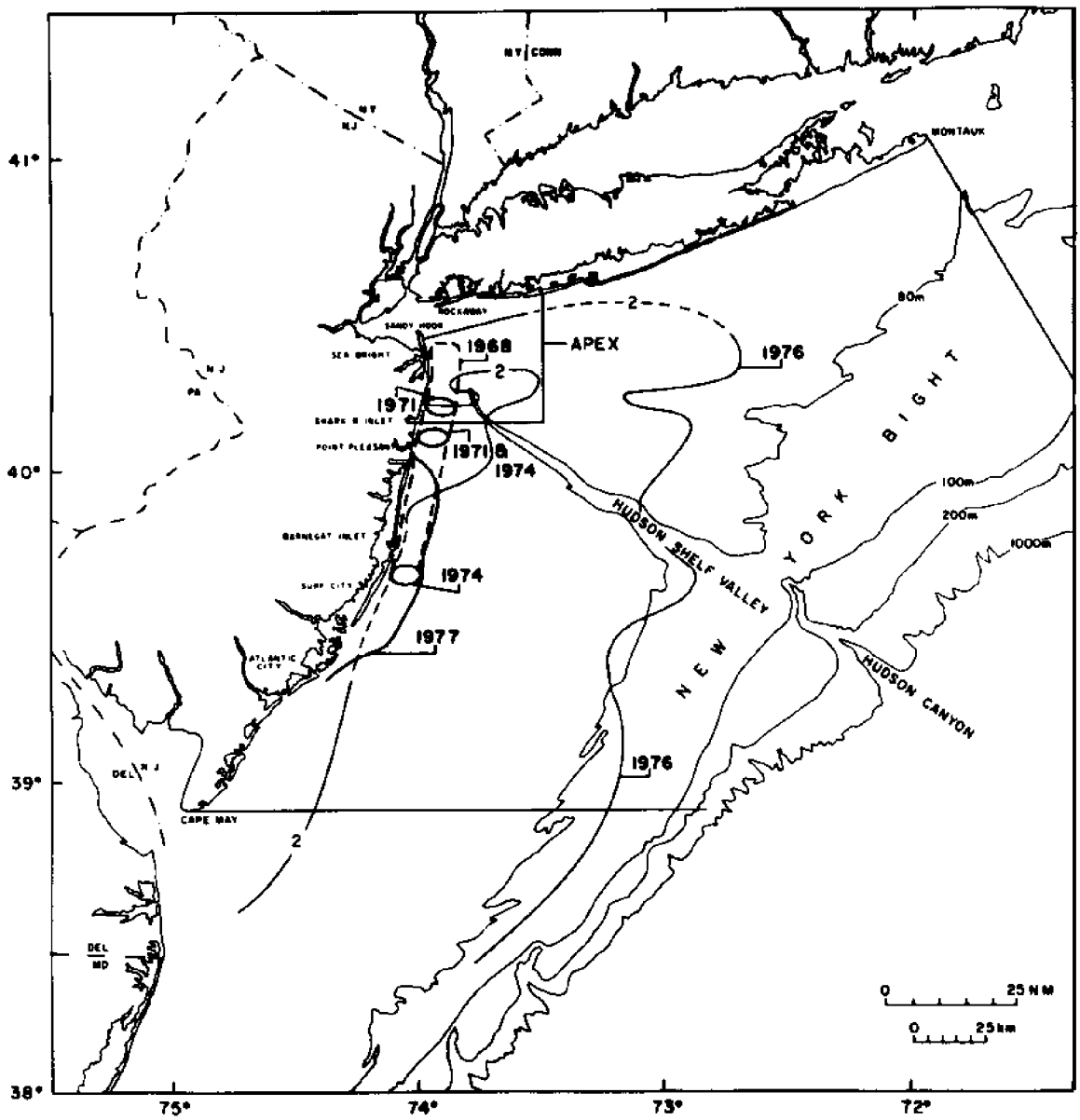
Offshore Effects

Sources on offshore contaminants include ocean dumping of chemical wastes at "Deepwater Dumpsite 106" located 106 n mi offshore, in depths ≥ 2000 m (see Figure C.2). The bulk of this material is 2 million m^3/yr of waste acids, pigments, and iron hydroxides (Gross, 1976b). Research begun in 1974 included photographic and visual reconnaisance of the slope using a small research submarine; assessment of benthic and midwater fisheries resources, plankton and micronutrients in the water column, physical and geological oceanography, and an investigation of the effects of previous radioactive waste disposal operations in a 10 n mi square centered on $38^{\circ}30'$N, $72^{\circ}06'$W. Initial findings were reassuring (NOAA, 1975): except for previously reported waste diffusion, discoloration of the water, and attraction of bluefish to the area of the 106-mile site (Ketchum and Ford, 1948 and Redfield and Walford, 1951), no impacts of the dumping were revealed. The findings were also consistent with the reporting of no evidence of impacts on the plankton (Grice et al., 1973; Vaccaro et al., 1972). Nevertheless, the research effort is continuing.

Data have been summarized on trace elements in fish and shellfish from Cape Charles, Virginia to Cape Cod, Massachusetts. The data bases were those of the National Marine Fisheries Service, College Park, MD, and Milford, Connecticut Laboratories including concentrations of 15 elements in 2309 samples from 50 species of fishes and shellfishes (MESA, 1978). Most of the species were commercial food fish.

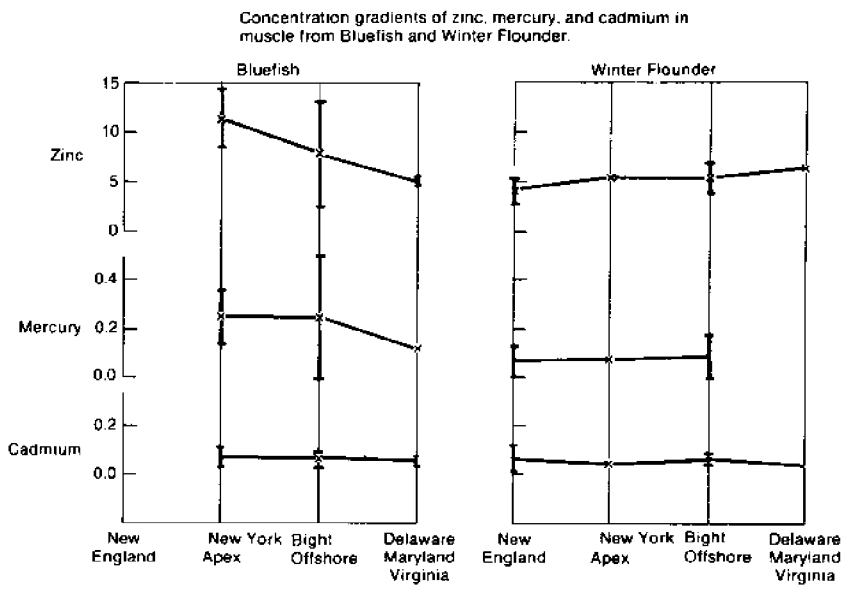
Mixed geographical trends were found; in some cases, highest metal concentrations were found in fish from Long Island Sound and the inner New York Bight, with levels decreasing with increasing distance from the inner Bight. The species with highest metal concentrations include crabs, lobsters, eels, clams,

Figure C.14 Oxygen depletion in New York Bight, 1968-1976. Isopleths show areas where bottom waters contained less than 2 ml/l dissolved oxygen.



butterfish, herrings, and sharks. The highest values in fish were found in the livers of 7 sandbar sharks where the mean and high values were 6.6 and 30 ppm, respectively (MESA, 1978). Regional trends are well revealed by data listed in Table C.11 for lobster. Concentrations were low along the Maine coast, higher in harbors, and highest in Hudson Canyon. The latter is consistent with movement of contaminated sediments shown in Figure C.13. In contrast, regional trends for the low concentrations of trace metals in fish are mixed, as indicated by Figure C.15.

Figure C.15 Concentration gradients of zinc, mercury, and cadmium in muscle from bluefish and winter flounder.



Winter flounder and other fishes, rock crabs, lobsters, and other shellfish stressed by contamination of the Bight also are diseased. The problem of fin rot in fishes has gained widespread public attention; fortunately, fin rot has declined recently (Murchelano and Ziskowski, 1976). A less widely known group of shellfish diseases include "black gill" and

Table C-11 Concentrations of Mercury in Lobster (Homarus Americanus)

	Number of Observations	Mercury Concentration, ppm		
		Mean Whole Body	Range Whole Body	Tail
Nearshore				
Jonesport, Maine	50	.13	.05 to .22	.05 to .27
Portland Harbor, Maine	50	.27	.13 to .60	.13 to 1.24
Boston Harbor, MA	25	.52	.28 to .87	.41 to 1.38
Green Harbor, MA	37	.44	.27 to .79	.34 to 1.23
Offshore				
Veatch Canyon (a)	82	.30	.16 to 1.21	.21 to 1.55
Block Canyon (b)	93	.44	.16 to 1.21	.25 to 1.90
Hudson Canyon (c)	89	.56	.16 to 1.51	.22 to 2.31
Norfolk Canyon (d)	78	.53	.16 to 1.07	.20 to 1.49

(a) approximately 200 km (110 n mi) south of Cape Cod
(b) approximately 150 km (80 n mi) south of Newport, RI
(c) approximately 125 km (79 n mi) south of Long Island
(d) approximately 110 km (60 n mi) east of Chesapeake Bay

"exoskeletal erosion." These are diseases of lobsters, shrimp, rock crabs, horseshoe crabs, and other organisms with hard outer skeletons. For example, "black gill" disease is linked to the organic content of underlying sediment (Sawyer et al., in press). The incidence of diseases in these commercial and potentially commercial organisms may limit fishing and marketing (MESA, 1978).

Koditschek (1974, 1976), Timoney et al. (1978) and Litchfield (1980) have reported abnormal tolerances of bacteria from sediments for unusually high concentrations of some metals and antibiotics. Large numbers of Bacillus from near the dump sites of the inner Bight were resistant to mercury and the antibiotic ampicillin in concentrations which would kill normal Bacillus strains. Bacillus from inshore stations over four miles from the dump sites were less resistent, and still less resistance was found in samples from near the outer edge of the continental shelf. Because the concentrations of several heavy metals are high in the more contaminated sediments, resistant strains have a selective advantage over strains more susceptible to metal toxicity. An additional finding was that one-third of the Bacillus strains from the dump site area metabolized mercury and released it as elemental mercury. This could be a significant mechanism for promoting a flux of mercury from sediments to the water column (Timoney, 1978).

Acute Effects of Waste Discharges

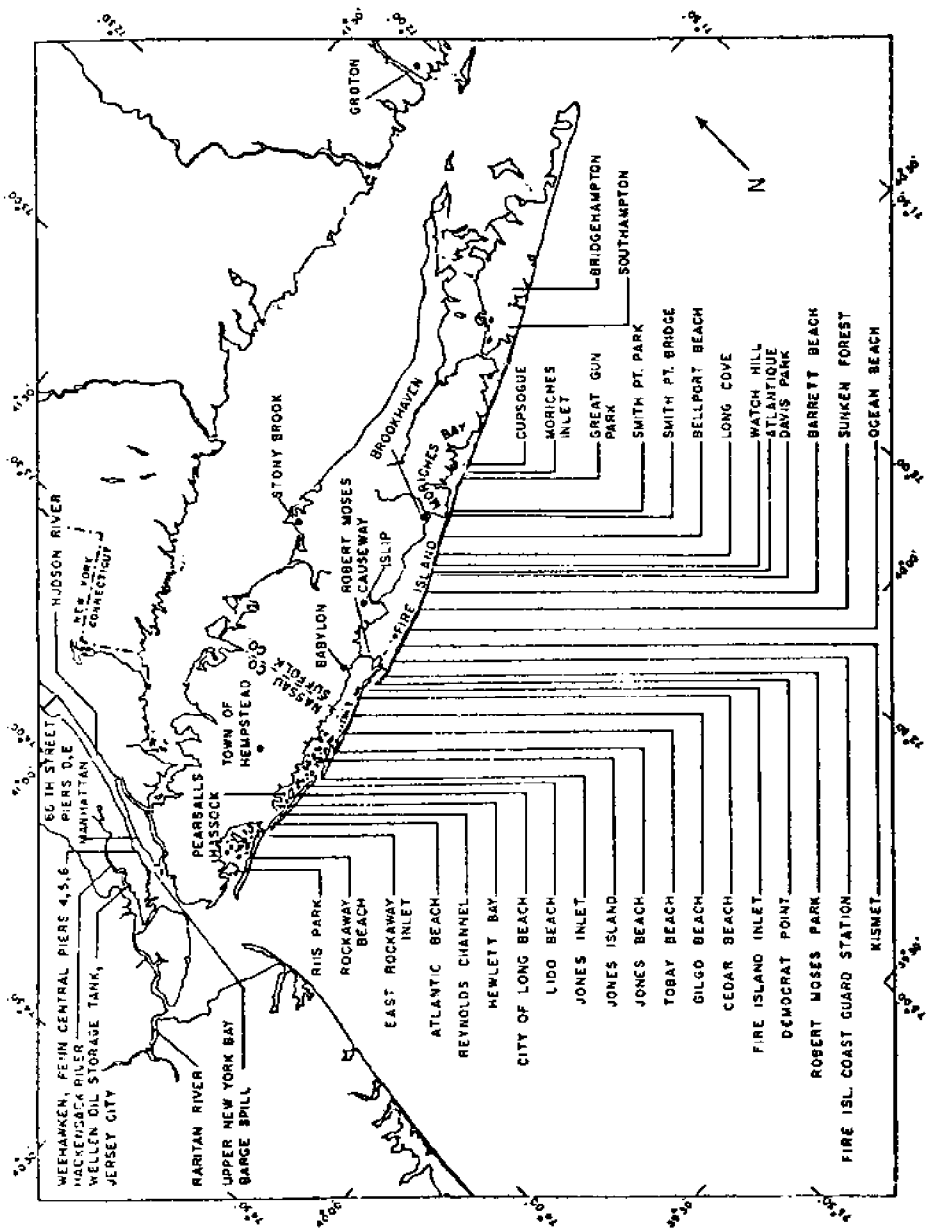
Although the chronic effects of pollution of the New York Bight are more important, the occasional severe episodes attract public attention and demand scientific and engineering response. Two recent examples include the sudden stranding of grease balls and plastics from sewage on Long Island beaches; and the slow development of anoxic bottom waters and an attending fish and shellfish kill of the New Jersey shore, initially thought to be due solely to nutrients in waste discharges.

June 1976 Long Island Beach Pollution (MESA, 1977). In June 1976 almost all of Long Island's major public ocean beaches were closed to swimmers for varying periods because of the stranding of floating trash and pollutants. These closings began with 20 miles of Fire Island beaches on June 15, followed by most of Long Island's south shore beaches during the next week. By July 1 these beaches were open. Additional strandings recurred, but on a lesser scale,

and resulting closings have been brief. Chronology of the event is as follows: 1. Throughout May, the Hudson River discharge was far above normal. 2. In early May an oil spill in Upper New York Bay resulted in large quantities of black oil balls (tar balls) washing up on beaches from Jacob Riis Park to Fire Island. On May 26, a storage tank ruptured at Jersey City, New Jersey, and more oil was spilled into the Hackensack River and into the wetlands of the Hackensack Meadows. 3. On June 2, two sewage sludge tanks on Pearsalls Hassock exploded; 1 million gallons of sewage sludge flowed into the water, and 1.1 million gallons spilled onto the land. The tanks were used to store sludge prior to barging it to sea. The tanks were routinely emptied from the bottom and an unknown amount of floatable material had accumulated in the upper portion of the tanks during the 12 years of their operation. 4. On June 3 and 11, pier fires broke out at Weehawken, New Jersey, and Manhattan, New York, dumping tons of wreckage and debris into the water. Large amounts of this debris could not be recovered in cleanup operations because of the small size. 5. Finally, winds became southerly in early June and continued throughout most of the month. 6. Meanwhile, the usual sources of wastes continued unabated. These included raw and treated wastewater discharges, occasional spills from refuse disposal operations, commercial shipping, recreational and commercial boating, and ocean dumping.

About noon, June 14, there was a report of unusual amounts of materials washing up on the Davis Park Beach. Tar balls up to 10 cm (4 in) and grease balls from 3 to 5 cm (1 to 2 in), mixed with a variety of other debris, were stranded between Sunken Forest and Smith Point (Figure C.16). On the afternoon of June 15 the Suffolk County Department of Health closed all county beaches to swimmers. On June 16, the New York State Department of Environmental Conservation closed waters to shellfishing for seven days; state beaches were also closed. During the next week most beaches on the south shore of Long Island were closed. On June 23, the Governor of New York declared Nassau and Suffolk Counties disaster areas. Two days later, the President of the United States assigned the Job Corps to clean up the Long Island south shore beaches under Coast Guard supervision. Cleanup efforts were completed by the end of the month, and by July 1, the last beach was reopened. Estimated costs of beach cleanup exceeded $100,000. Estimated losses to the Long Island recreational industry were $15,000,000 to $25,000,000.

Figure C.16 Long Island beaches and sources of beach pollution.



Initial reports indicated that raw human feces were observed on the beaches. Later examination showed these materials to be tar balls and grease balls, with high concentrations of both total coliform and fecal coliform bacteria. The grease balls were generally of lighter outside colors and of smaller size than the tar balls. Most grease balls contained materials that looked like congealed kitchen grease and oil. About 75 percent of the waste balls were formed around foreign substances, generally pieces of plastic, including tampon applicators. A swath about 0.3 m to 3 m (1 ft to 9 ft) wide along the swash line of the preceding high tide contained seaweed, driftwood, small pieces of burned wood (about 3 to 20 cm (1 to 8 in) long), considerable trash, garbage, and waste balls. From 60 to 95 percent of the material was charred wood, driftwood and dunnage. Plastics ranged from 1 to 9 percent and included: plastic tampon applicators (about one per 3 m (10 ft) of beach), decomposed condoms, thin pieces of plastic sheeting from sanitary napkins, and disposable diapers. Other materials were plastic straws, pieces of styrofoam cups, plastic bottle caps, pieces of manmade sponges, corks and softballs, small plastic toys and household wares, and plastic cigar and cigarette tips.

There was no correlation between quantities of stranded materials and bacteriological pollution. Water samples collected along the beaches during the peak of the pollution problem showed total coliform levels well within the New York State standard for swimming (2,400 Most Probable Number [(MPN)/100 ml]). Fecal coliform levels were less than 100 MPN/100 ml, and usually less than 10 MPN/100 ml. Where concentrations in individual water samples were high, known local sources were responsible. Beaches were closed to swimming as a precautionary measure, pending results of bacterial analyses, because of the suspected origin of the waste materials. Although grease and tar balls contained up to 1 million total and fecal coliform bacteria per 100 grams of sample, pathogens such as Salmonella from grease balls were not found.

Within Hempstead Bay, the explosion of the sludge storage tanks had little impact upon bay water quality. Daily sampling at 26 stations from June 3 to 7, revealed that coliform bacteria level were generally normal or below, except on June 3, when coliform levels were 3 to 5 times above normal. DO levels were even less affected. A similar sludge accident occurred on June 18, 1976, when 10,000 gallons of sludge was spilled from a storage trough at the Lawrence Sewage Treatment Plant and eventually entered the western end

of Reynolds Channel. This accident also produced a temporary increase in coliform bacteria levels in the bay, with a quick return to normal levels soon after the incident (Nassau County Health Department, 1977).

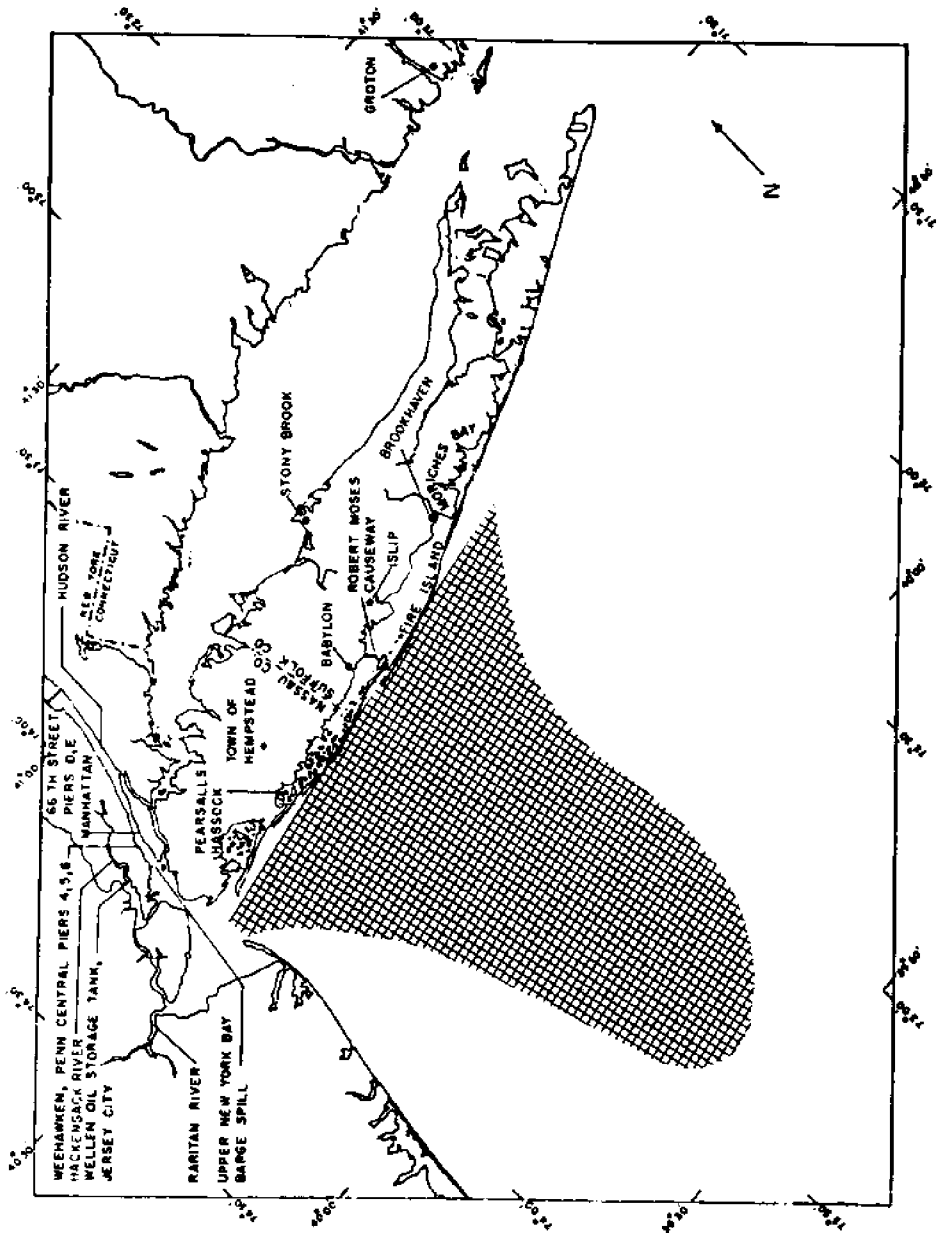
Possible sources of oil and grease loadings to the Bight and estuary include 191 metric tons/day (t/d) from municipal discharges, some of the 570 m^3 (150,000 gal) of No. 6 oil spilled into the Hackensack River on May 26, 23 t/d from sludge dumping, 300 t/d from dredge spoil disposal, an undetermined amount from urban runoff and from combined sewer overflows from rainfalls in excess of 1 mm (0.04 in) that occurred on 19 days during May and June, and a portion of the sludge from the ruptured tanks on Pearsall's Hassock.

Floatable transport mechanisms include wind-driven surface currents and surface currents driven by outflows from the Hudson River. Both were unusual during May and June. Mathematical models of surface transport have been developed by the Coast Guard and the Brookhaven National Laboratory. The latter assumes that the water surface (and floating material) moves as a vector sum of the vertically integrated current and 3 percent of the wind velocity. Figure C.17 presents the results. Anything floating within the shaded area from June 1 to 24, regardless of original source, would have been stranded on Long Island (Swanson et al., 1978).

Summarizing the June 1976 floatables episode - there were multiple sources for the grease balls and for the plastic sanitary materials found on the beaches and there were wind and surface current mechanisms to transport them there. Although the absolute contributions from individual sources are unknown, the timing of the sludge tank explosions followed by reporting of unprecedented amounts of floatables, such as tampon applicators and condoms whose sewage origin was incontestable, brought to public and official attention the narrow limits within which present waste disposal systems operate satisfactorily. Minimizing the future probability of such incidents will require a combination of materials selection, (such as readily decomposable tampon applicators), segregation, and removal at the sources of such materials as grease discharged through garbage grinders.

1976 anoxia and fish kill. The 1976 anoxia and fish kill event was the most severe on local record and is first incident of its kind along an open coastline where waste discharges were implicated. An area of

Figure C.17 June 1976 beach pollution transport. Anything floating in the shaded area would have been stranded on Long Island beaches.



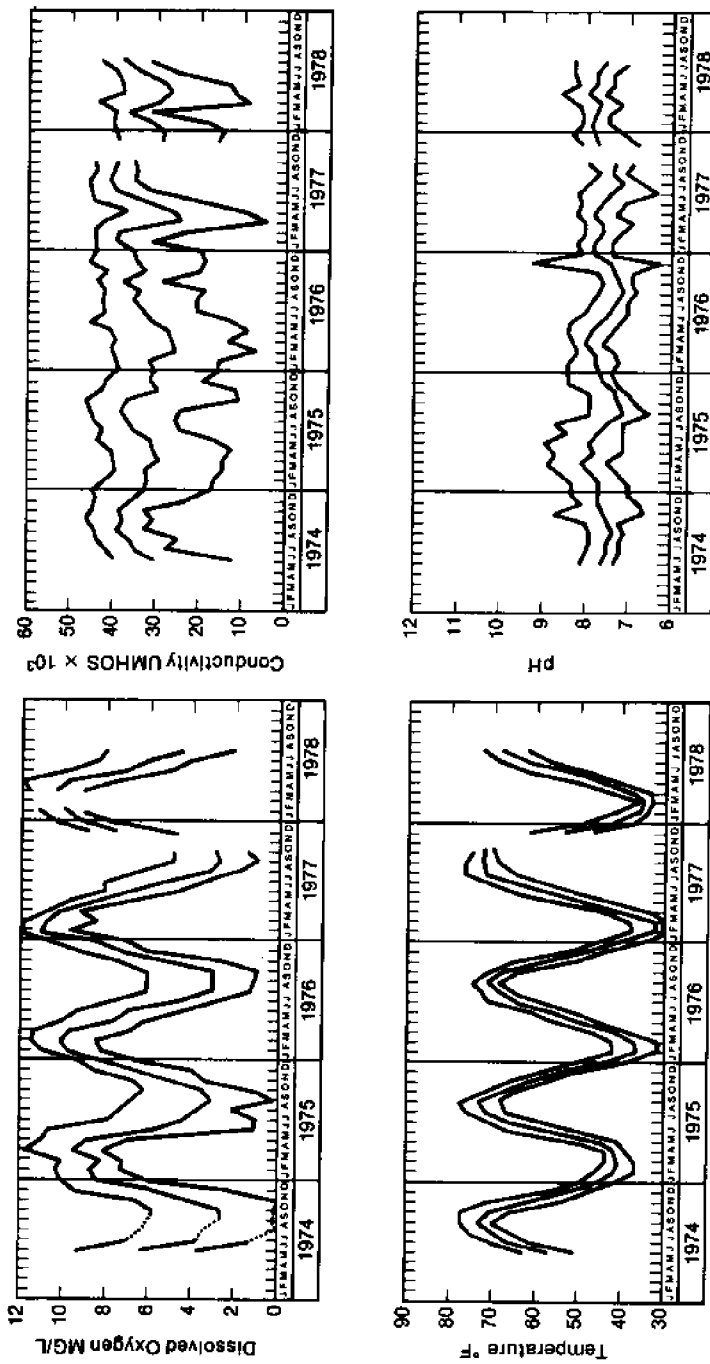
approximately 8,600 km^2 (Figure C.14) along the continental shelf off the New Jersey coast experienced mass mortalities of benthic organisms from July through October, 1976 (Swanson and Sindermann, 1979). The observations suggest a north to south progression of stressed or dead organisms from the vicinity of Monmouth Beach in early July to Atlantic City by late July. Surf clams, ocean quahogs, finfishes, lobsters and sea scallops (in decreasing order) were the commercial species most affected. Figley et al. (1979) estimated losses to the fishing, processing, and marketing industries of $7.9 million during the year and that the estimated losses of the resource were about $62.5 million. Until the stocks are replenished through recruitment, which may take up to seven years for surf clams, potential losses could exceed $550 million. The immediate economic loss to the sport fishery was estimated to be $3.7 million. Summer flounder fishing along the New Jersey coast was excellent, because their normal movement to offshore areas was prevented by the anoxic water.

Monitoring based on 1976 and subsequent events (see Figure C.18) can be useful for assessing the likelihood, on a time scale of about two months, of future mass benthic mortalities. These events were according to Swanson, Sindermann, and Han (1979) and Swanson (1979) as follows:

1. In January, there was a large bloom of the dinoflagellate Ceratium tripos over the New York Bight and much of the northeastern continental shelf beginning in January and peaking in April and June (Malone et al., 1979); this geographic scale indicates that waste inputs were not influential.

2. There was an early and warm northeast spring initiated by atmospheric circulation patterns that inhibited the transport of arctic air into the region (Diaz, 1979). During February through May, there was a large, continuous flow of fresh water from the Hudson River and surrounding drainage basins, and an early warming of the sea surface waters. The resulting lens of relatively warm fresh water caused density stratification in coastal waters some six weeks earlier than usual. This in turn contributed to isolation of bottom waters from reaeration by turbulent mixing at the air-sea surface (Starr and Steimle, 1979 and Armstrong, 1979).

Figure C.18 Results from monitoring dissolved oxygen, temperature, conductivity and pH and the Narrows (Interstate Sanitation Commission, 1978).



3. The wind field generated by the anomalous atmospheric pressure patterns of early spring was such as to establish a potential for upwelling of bottom waters along the New Jersey coast. Evidence of upwelling off the New Jersey coast was also observed in the bottom current meter data analyzed by Mayer et al. (1979).

4. Mayer et al. also noted that the offshore (100 km) southward flow was considerably less than in the previous year, whereas the alongshore component on the inner shelf (50 km) actually reversed, flowing northward from mid-May through July. In early August the net bottom water displacement was small.

5. During most of June the net circulation near the bottom was northwestward along the New Jersey coast, into the Apex region, along with strong onshore flow and probable associated upwelling (Han et al., 1979). Intensification of the flow occurred along the Hudson Shelf Valley. This flow transported oxygen, and in all likelihood C. tripos, into the impacted area. The net utilization rate of oxygen was calculated by Han et al. to be three to ten times larger in the Apex and the New Jersey coastal segment than in other segments of the Bight.

6. C. tripos concentrated below the photic zone at the base of the pycnocline (1 to 3 percent incident light), where the water is subject to shoreward transport. At this depth phytosynthetic processes were certainly reduced and the organisms were possibly living heterotrophically (Malone et al., 1979), thereby using oxygen through respiration. Subsequent stratification, possible reduction of particulate organic material, and reduction in Hudson River runoff may have limited the nutrient supply to C. tripos below the pycnocline during late June. In any event, the bloom declined rapidly during July, creating an organic floc at the bottom in or near the area of depressed bottom DO.

7. The respiration of the C. tripos can be calculated from the data of Malone et al. and then compared to net utilization rate. The average C. tripos cell counts below the pycnocline were 0.64×10^8 cells/m^3 off Long Island and 1.1×10^8 cells/m^3 off New Jersey. The respiration rates for these cell concentrations are 0.37 ml/l/d and 0.22 ml/l/d.

If the mean benthic respiration rate of 11 ml/m/hr (Thomas et al., 1976) is distributed over the 16 meter thickness of the lower layer - the benthic respiration rates are 0.03 to 0.22 ml/l/d. The total respiration of the C. tripos and benthic communities is then 0.24 to 0.40 ml/l/d. The net utilization rates used to produce the observed oxygen decline, as calculated by Han et al., were 0.05 to 0.17 with 0.15 ml/l/d in the Apex.

In mid-May the bottom DO concentrations were nearly the same in all segments. The time rates of change in concentration (supply) and of net utilization (demand) in the Apex and along the New Jersey coast were similar. The Long Island coast had lower estimated rates of respiration during this period of time and was much less affected. With no advective inputs of oxygen into the system, respiration alone could have caused anoxic conditions along the New Jersey coast in approximately 10 days. Thus, the respiration of the large population of C. tripos was sufficient to account for the observed oxygen decline through June.

Garside and Malone (1978) conclude that the average carbon loading to the Apex, including anthropogenic inputs, is insufficient to cause anoxic conditions. However, the normally large carbon load, in addition to the respiration of C. tripos (which to some degree may have been concentrated more in the southern Apex by flow up in the Hudson Shelf Valley), likely created a greater total load in the southern Apex.

8. Thomas et al. (1979) found that C. tripos were not present in the water column in August and September, though Mahoney (1979) found a decaying floc of C. tripos near the bottom in July. Malone et al. estimated the biological demand from the oxidation (BOD) of the C. tripos over 60 days (July and August) was equivalent to 0.16 ml/l/d, which would have maintained anoxic conditions throughout the remainder of the summer.

9. Anaerobic conditions and sulfide generation in the anoxic area were observed by Thomas et al. (1979), along with high rates of seabed oxygen consumption at the periphery of the area. The decaying floc increased stress on benthic organisms and the continuous anthropogenic loadings and oxidation of dissolved and

particulate matter and carbon could only intensify and maintain the already depressed conditions.

10. Although density stratification may not have been the primary cause in oxygen depletion, the anoxic conditions were not relieved until the breakdown of the pycnocline in autumn. Even the disturbing effects of hurricane Belle were only transitory. During autumn and winter, bottom oxygen values continued to increase, once again reaching saturation.

Existing evidence does not show that waste inputs triggered the anoxic event. However, there is slight evidence of a gradual decrease over the last 10 years in bottom DO and,

> ...the sensitivity of the system might therefore be changing such that a slight imbalance (either due to natural causes or increases in waste loadings) in the "normal" cycle of environmental conditions is sufficient to drive the system towards anoxia with increasing frequency. If this is true and if the slight decreasing trend in bottom DO over the last three decades is real and monotonic, then there is an increasing probability of low DO events in any year...(O'Connor, 1979).

MONITORING AND REAPPRAISAL

O'Connor's (1979) warning is sobering. It comes at a time of increasing competition for resources and a decreasing margin for error in allocating them. These include construction of conventional sewers and treatment plants which are variously estimated to have cost some $2 billion since 1950 and to require at least $3 billion (1980 dollars) more by the year 2000 (Hazen and Sawyer, 1978; ISC, 1979; USEPA Reg II, 1979). Assuming a 10 percent annual capital recovery factor, 1950-2000 capital cost of $500 million per year, annual operation and maintenance costs may be estimated at about 30 percent of capital costs (Hydroscience, 1979; Kalbermatten et al., 1980). The marginal cost of 65 to 95 lcd (17 to 25 gcd) of flushing water may be estimated from the cost of supplemental water described in the Corps of Engineers (1977b) NEWS study. This additional supply would provide 102 lcd (27 gcd) in the year 2000 at an estimated annual cost in 1980 dollars of $640 million for capital recovery, operation,

maintenance and replacement. Thus, marginal costs of flushing water are $640 x 17/27 = $400 million per year. Total estimated present costs of sewerage (not including plumbing costs) are $500 million--capital recovery, $150 million--O&M, and $400 million--flushing water, totaling $1,050 million per year.

Receiving water benefits from these costs are elusive. The Mueller and Anderson data previously cited reveal that future waste loadings to the New York Bight will remain at essentially present levels. These are not much different from 1950 levels except for increased toxic elements and compounds whose removal costs are not evaluated; the Hydroscience (1978b) and Hazen and Sawyer (1978) studies considered primarily the costs and effectiveness of suspended solids, BOD, nutrients, and coliform removals. Within the harbor, coliform and DO levels have generally improved since 1974, although anoxia is still a common summer event in Arthur Kill (Interstate Sanitation Commission 1978, 1979). Meanwhile, fisheries have been lost due to wastes for some 120 years and more recently by high body concentrations of PCBs and other chemicals (Goode, 1887 and Pearce, 1979).

In the Bight, the main improvement came with the elimination of garbage dumping in 1943. There has been a gradual increase in the amounts of nutrients which is expected to continue (see Table C.7).

Monitoring

Successful monitoring and prediction depend upon previous research and observation. Remedial measures such as regulation or construction and operation of treatment facilities can be designed from baseline surveys of waste discharges and effects. Optimal timing for investing in and implementing these measures is best determined by monitoring.

An operational definition of ocean pollution monitoring is that which consists of a series of observations of limiting or damaging environmental characteristics which can be managed by present technologies or institutions. It includes interaction with technological and economic monitoring and is effective only when the users and uses of monitoring information are known, so that optimal resolutions in time, space and parameters are also known. Note that the resolution in time (sampling frequency) is fixed by institutional or technological response schedules (Gunnerson, 1975b). These criteria may generally be

met in source and effects monitoring but generally not in ambient monitoring of water column or sediment pathways. An exception to the latter is DO monitoring in streams where oxygen sources, sinks and rate functions are reasonably established, and where remedial projects can be constructed.

Conventional oxygen monitoring (Interstate Sanitation Commission 1978) is of limited value. Figure C.18 shows the results of 1974 to 1978 continuous monitoring of DO of Hudson River outflow on the west bank at the Narrows. DO values here are poorly related to those in the Bight; certainly they provided no hint that the 1976 anoxia event was imminent. In contrast, Figure C.19 shows that biweekly water column monitoring in the Bight Apex may permit such prediction. Such information could be used for advising surf clam fishermen in advance of a kill of the sort that happened in 1976. The point is that a response to oxygen monitoring in the Bight is possible, while a response to oxygen monitoring of the estuarine outflow is not.

Reappraisal of Water Supply and Waste Disposal Systems

The predicted urbanization of the northeast coastal region shown on Figure C.20 will, with conventional practices in water supply, utilization and disposal, result in increasing severity and frequency of marine environmental crises. These crises can be minimized by developing analytical tools for predicting environmental impacts of alternative strategies for waste management. However, waste management is only one element of resource management. Other elements, with examples of policy options to deal with them, are as follows.

Scale and coordination. Economies of scale are usually achieved by loss of redundancy and system resilience (see Lovins, 1977 and Hollings, 1978) so that small perturbations cause greater total damage as system size, regionalization, and interdependency increase. This increase is accompanied by (and possibly causes) increasing compartmentalization of planning effort. For example, the 1977 Northeast Water Supply Study (Corps of Engineers 1977a,b,c,d,e) makes no reference to the costs of disposing of additional water for New York City. Similarly the 1978 City of New York Section 208 (PL 92-500) Area Waste Treatment Management Planning Program (Hazen and Sawyer, 1978) makes no reference to disposal costs for the planned increase in water supply or to the costs of flushing

Figure C.19 Annual cycle of bottom dissolved oxygen on the New Jersey shelf during 1976, showing corresponding values for proposed seasonal and biweekly monitoring stations (Armstrong, 1979).

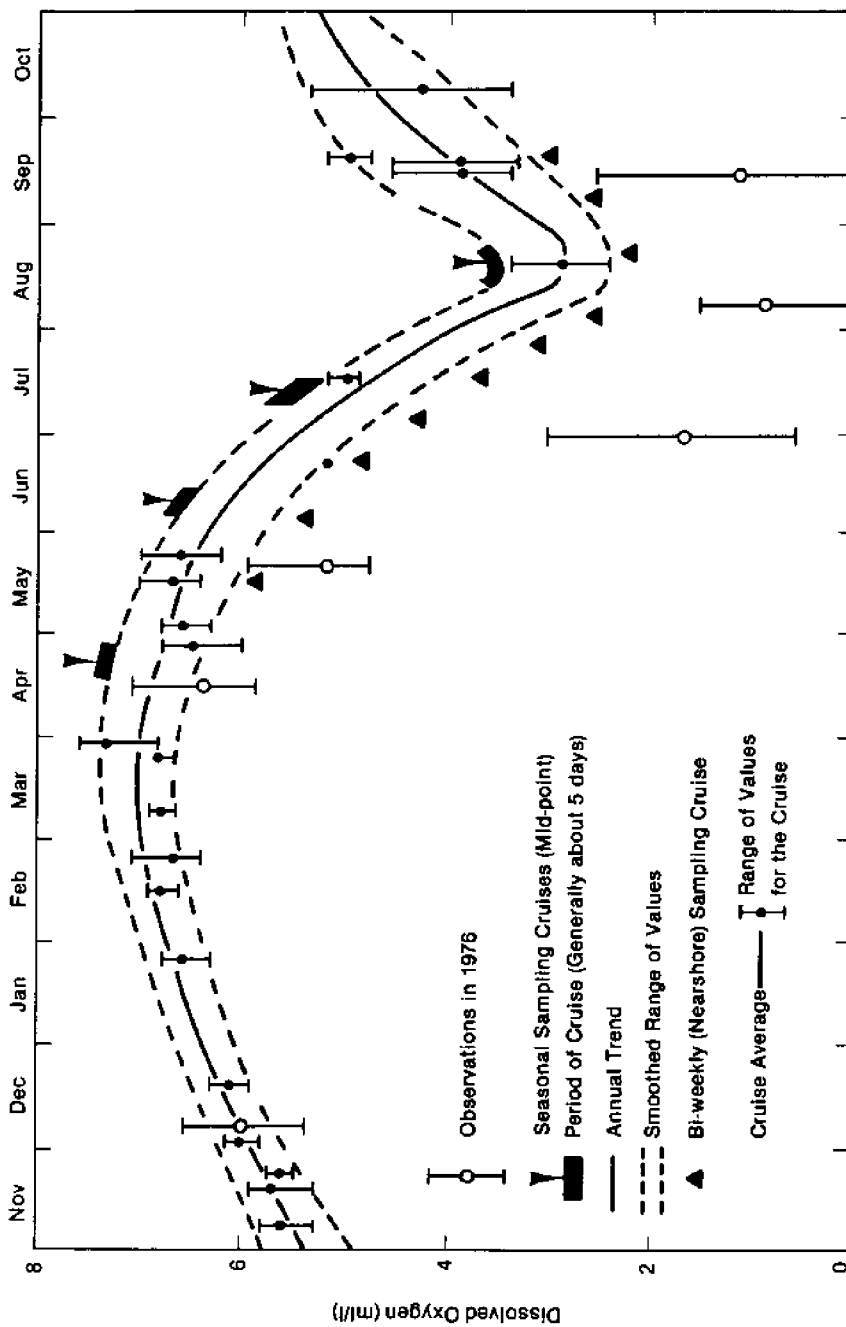
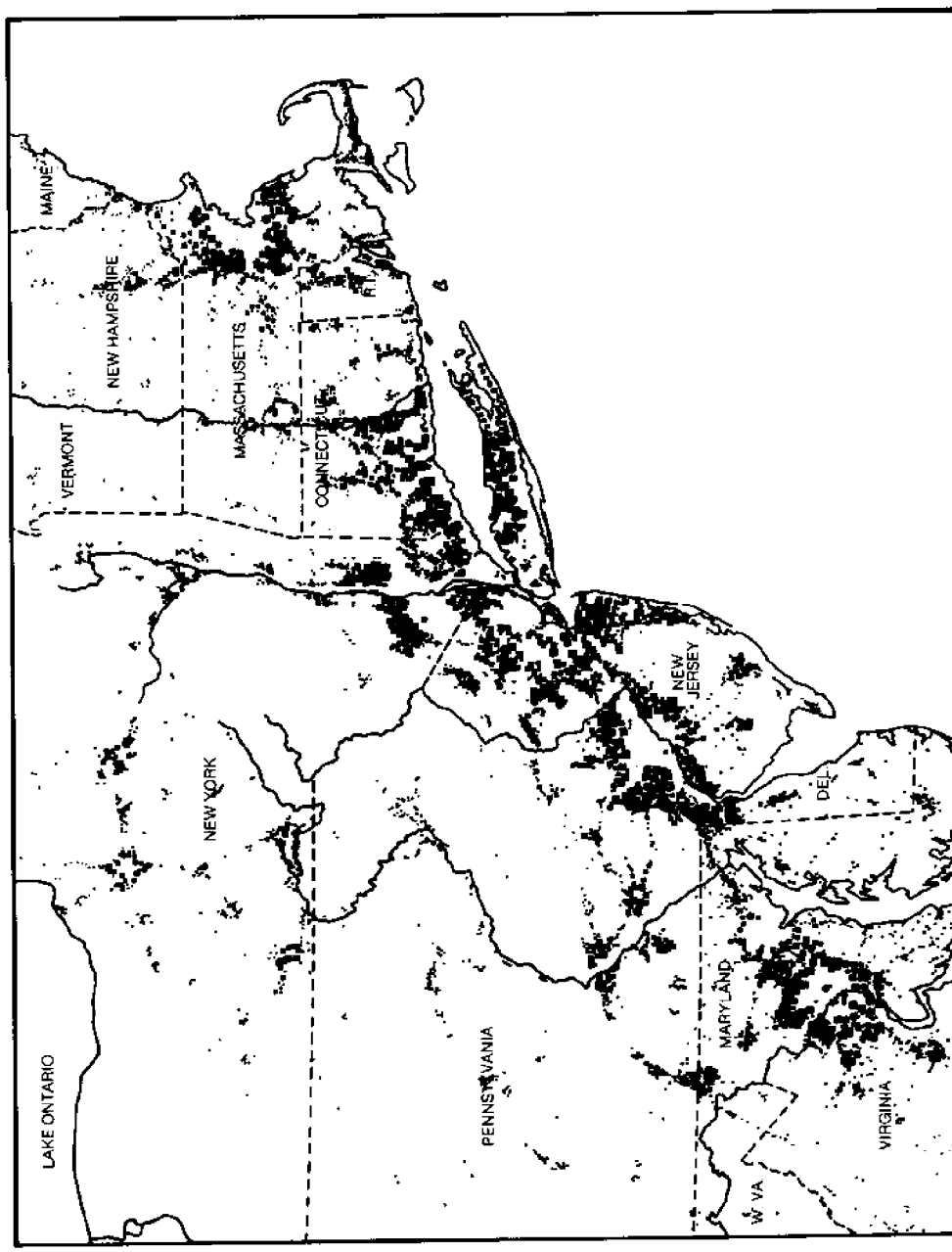


Figure C.20 Urbanization of the northeast Atlantic seaboard in 1965 (shaded) and additional urbanization possible by the year 2000 (darker areas) (Koebel and Krueckenberg, 1975).



water. It is probable that institutions exist at all levels which could develop needed coordination; that they are not used suggests problem of local jurisdictional boundaries. These may be resolved with coordinated guidance from the 14 Federal agencies involved with the 418 local, State, and interstate agencies concerned with the New York Bight (Marr, 1979). However, local participation and direction is essential to avoid the problems of scale described above.

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AFTERWORD

On November 2, 1981 Judge Abraham D. Sofaer of the US District Court, Southern District of New York issued his Final Judgment in The City of New York vs the United States Environmental Protection Agency (80 Civ. 1677). In May 1982 similar judgments on six sewerage authorities were issued by Judge H. Lee Sorokin of the US District Court of New Jersey. The decisions stated that the 1981 deadline shall remain in efect only for ocean dumping that may unreasonably degrade the marine environment in view of environmental

risks associated with land based alternatives. EPA was ordered to: (1) revise its regulations to allow review of "all relevant statutory criteria" (e.g., balancing). EPA may use existing regulations in evaluating applications for dumping permits, as long as such reviews are consistent with applicable law. EPA may not establish a conclusion presumption of "unreasonable degradation" with regard to a waste solely upon the waste's non-compliance with environmental impact criteria. (2) Evaluate discharger's applications in accordance with item 1 above. (3) Take no enforcement action against the dischargers with respect to its ocean dumping activities, as long as such dumping is in accordance with the conditions of the last permit. (4) Allow dumping to continue at the present site, pending action on petitions to redesignate the sites. If EPA declines to redesignate the site, the City (unless a court intervenes) will take all steps necessary for a rapid and orderly removal of dumping to a location designated by EPA.

Of particular interest is that intervention in these cases was sought but denied to two Congressmen, the National Wildlife Federation, and other environmental groups. These groups had raised the issue that Judge Sofaer's decision did not take into account provisions of the London Ocean Dumping Convention that prohibit the dumping of wastes (including sewage sludges) containing mercury and cadmium (as elements or compounds), organohalogens, and oil as other than trace contaminants. More recently, the City of New York petitioned EPA, under Section 401(h) of the 1977 amendments to the Clean Water Act, to be relieved of the requirement for secondary sewage treatment. The petition is based on observations (from New York and elsewhere) that the marginal benefits of secondary treatment discharges to marine waters are not commensurate with the costs. Predictions of future sludge production and waste loadings presented in this paper are thus provisional. Costs and benefits of current waste management practices are valid (CGG, 1/15/83).

*Information on Court decisions was abstracted from a summary prepared by Peter W. Anderson, US Environmental Protection Agency. Region II, New York, in G.F. Mayer (ed.) Ecological Stress and the New York Bight: Science and Management. Estuarine Research Federation, Columbia, SC (1982) pp. 108-111.

Appendix D

Experiences in the United Kingdom on the Control of Discharges of Sewage Sludge to Estuarine and Coastal Waters, A Case Study

Michael G. Norton

INTRODUCTION

In the text of this book, the physical, chemical and biological processes which take place in estuaries and coastal waters have been described in detail so far as current knowledge allows. General descriptions have also been provided on how our understanding of these processes may be used to predict the environmental impact of waste disposal on the marine environment and the assimilative capacity of the receiving waters. The conclusion has been reached that the ocean does have a capacity to receive certain wastes without damage, but that it is essential that this assimilative capacity be quantified and managed effectively through adequate controls, in order to ensure that disposal does not give rise to adverse effects on the receiving waters, the biota present or humans as the consumer of marine resources.

Due largely to its extensive coastline and the proximity of much of its population to estuaries or coastal waters, the United Kingdom (UK) has a long history of discharging sewage effluent (after varying degrees of treatment) directly to estuaries and coastal waters. In addition, approximately 28 percent of the sewage sludges produced by the treatment of sewage within the UK is disposed of at sea by dumping from vessels.

These disposal practices are controlled under national and, in some cases, international regulations. Dumping of wastes at sea is regulated by the Dumping at Sea (DAS) Act 1974, and by the provisions of the Oslo and London Conventions on the prevention of marine pollution. Discharge to estuaries and coastal waters is regulated under the Rivers (Prevention of Pollution) Acts of 1951 and 1961 and by the Control of Pollution Act 1974, which, when fully implemented, will extend controls to all discharges to estuarine and coastal waters. The Paris Convention on

the Prevention of Marine Pollution from Land-based Sources provides an international framework for these national controls on discharges to coastal waters.

All these regulations (both national and international) place their emphasis on the prevention of pollution from the disposal of waste and not the prevention of discharges per se. Pollution is defined in slightly different ways in the respective laws and regulations, but all are generally consistent with the definition given by GESAMP (The UN's Group of Experts on the Scientific Aspects of Marine Pollution.): "Pollution is the introduction by man, directly or indirectly, of substances or energy into the marine environment (including estuaries) resulting in such deleterious effects as harm to living resources, hazards to human health, hindrance to marine activities including fishing, impairing of quality of use of seawater and reduction of amenity."

In regulating the discharge of wastes to estuaries and coastal waters to avoid pollution, UK authorities* have carried out assessments of the environmental impact of disposal practices for some years which have necessitated the application of the techniques and knowledge set out in this book.

The presentation of detailed case histories for some of the most important areas off the UK is intended to assist in the interpretation and application of these predictive and monitoring techniques. In particular, the use of pre-discharge studies of the waste and the characteristics of the receiving area to predict the dispersion pathways, the ultimate fate of the waste and its constituents, together with its effects on biota will be described. The results of monitoring after commencement of discharge to confirm the validity of the predictions made and to measure biological, chemical and physical effects in the water column and at the seabed will also be presented.

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* Regulatory Authorities are: Dumping at Sea Act, 1974; Ministry of Agriculture, Fisheries and Food (MAFF) for England, Department of Agriculture and Fisheries for Scotland, Secretary of State for Wales and the Department of the Environment for Northern Ireland. Control of Pollution Act, 1974: Regional Water Authorities in England and Wales and River Purification Boards in Scotland.

The three areas selected as examples in this case study are those where the largest quantities of sewage sludge are dumped (the Outer Thames Estuary and Liverpool Bay for England, and the Firth of Clyde for Scotland). Although the emphasis will be on the application of scientific techniques to regulate dumping, reference will also be made to the control of sewage discharges to the Thames Estuary and to the importance of dumping relative to other pollutant inputs in all three areas.

THE THAMES ESTUARY

Sewage Sludge Dumping

Background. The Outer Thames Estuary has been used for the dumping of sewage sludge from London since 1887. Prior to 1967 the sewage sludge was deposited in the Black Deep, shown in Figure D.1, which had also been used for the deposit of dredged spoil up to 1964. Since 1967, the sewage sludge has been dumped in the adjacent Barrow Deep; no other wastes have been dumped at this location (Figure D.1).

The sewage sludge arises at two large sewage treatment plants operated by the Thames Water Authority and located in eastern London-Beckton (serving a population of 3.13 million) and Crossness (serving a population of 1.6 million) - which provide primary and secondary biological (activated sludge) treatment for 60 percent of London's sewage flow. Approximately 90 percent of the sludge is digested in heated anaerobic digesters (to reduce the quantity of sludge, control odour, and generate gas for power generation on site) before loading into dumping vessels of 2,000 to 2,600 metric ton (tonne) capacity for transportation to the dumping area 74 km downstream.

The composition and quantity of sludge dumped each year has varied slightly. Since statutory control under the DAS Act started in 1974, the most noticeable change has been a reduction in the concentration of mercury through tighter controls on industrial discharges to the sewage works by the Thames Water Authority. The quantity of sludge dumped, its average composition, and the total mass loads of its constituents are presented in Table D.1 for 1974 to 1977.

Figure D.1 The Outer Thames Estuary

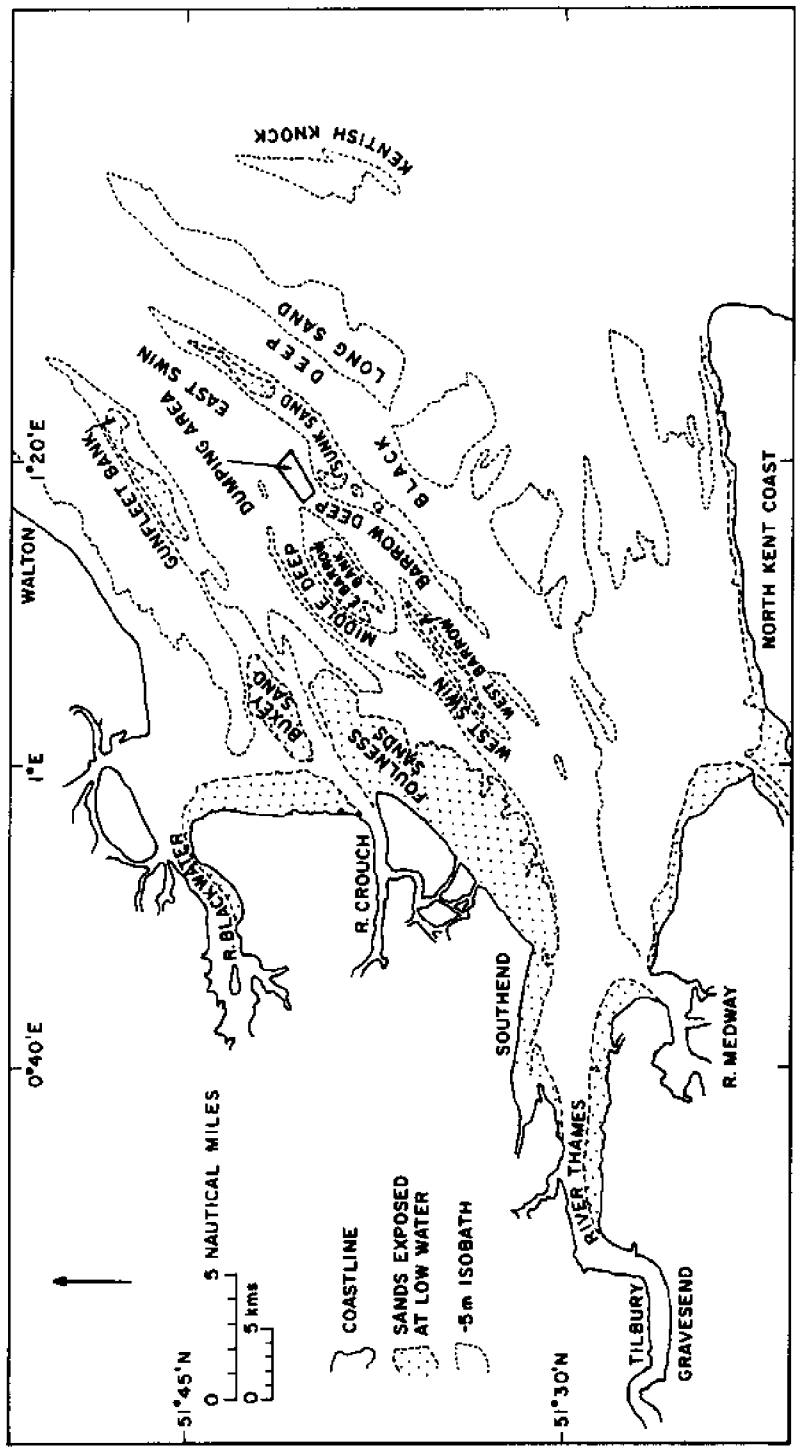


Table D.1 Quantity, Composition and Total Inputs of Sewage Sludge Dumped in Years 1974 - 1977 in the Barrow Deep

	1974		1975		1976		1977	
	Composition	Quantity (tonnes)	Composition	Quantity (tonnes)	Composition	Quantity (tonnes)	Composition	Quantity (tonnes)
Wet sludge		4.36×10^6		4.19×10^6		4.45×10^6		4.62×10^6
Particulate organics (%)	1.7	74,120	1.7	71,230	1.7	76,650	1.8	83,160
Particulate inorganics (%)	0.8	38,880	0.9	37,710	0.9	40,050	0.9	41,580
mg/kg - dry solids								
Cadmium	50	5.4	60	6.5	60	6.9	45.2	5.6
Mercury	56	6.1	14	1.5	12	1.4	13.2	1.6
Chromium	560	61.0	470	51.2	470	54.4	570	71.1
Copper	820	89.4	810	88.2	810	93.7	770	96.0
Lead	790	86.1	1,020	111.1	1020	118.0	760	94.8
Zinc	3200	348.8	3,420	372.6	3420	395.7	3630	452.8
% dry solids								
Nitrogen	7.6+	8,284	7.6+	8,279	7.6	8,793	7.6+	9,830
Phosporus	1.6+	1,743	1.6+	1,743	1.2	1,851	1.6+	2,069
Carbon	32 *	34,880	32 *	34,860	32 *	37,020	32 *	39,920

+ Assumed from 1976 figure
* Assumed from 1978 analyses

Estimates have been made of the relative inputs of organics, nutrients and trace metals to the outer estuary from dumping, discharges of effluent, river discharges and atmospheric deposition (see Table D.2). Even from the point of view of the whole outer estuary, it is seen that sludge dumping is a very significant input for several constituents and, within the immediate vicinity of the dumping area which is relatively remote from other point source discharges, dumping is clearly the dominant anthropogenic source of the substances listed.

Table D.2 Estimate of Relative Inputs to the Outer Thames Estuary, 1977 (National Water Council, 1979).

	Percent of Total Contribution										
Source	Suspended Solids	BOD	Total N	Total P	Cd	Cr	Cu	Ni	Pb	Zn	Hg
River Inputs	40	12	47	22	69	35	34	43	20	35	33
Sewage & Industrial Discharge	30	46	43	66	16	28	19	34	18	22	
Dumping	30	42	9	12	12	36	42	21	54	40	67
Atmospheric Deposition	-	-	-	-	3	1	5	2	8	3	-

The use of the outer estuary for waste disposal must be compatible with a number of other activities in the region. These include navigation for vessels discharging or loading in the Port of London, extraction of sand and gravel from licensed areas to the northeast of the dumping ground, and commercial exploitation of fish and shellfish stocks (value in 1978 of about one million pounds per annum). The area is of limited amenity use offshore, but important bathing beaches are located at Southend and from Walton northwards (Figure D.1).

This section will present the results of work carried out by Ministry of Agriculture, Fisheries and Food (MAFF) from 1972 to 1978 to determine the initial dispersion paths of the sludge, subsequent transport processes in the water column and sediments, the effects of disposal on the physical and chemical

characteristics of the receiving area and effects on the biota. References for this section are Shelton (1971), Talbot et al. (1982), and Norton et al. (1981). Details of the policy, objectives and methodology of dumping ground monitoring have been published by Norton and Rolfe (1978) and Eagle et al. (1978), respectively.

Initial Dispersion of the Sludge

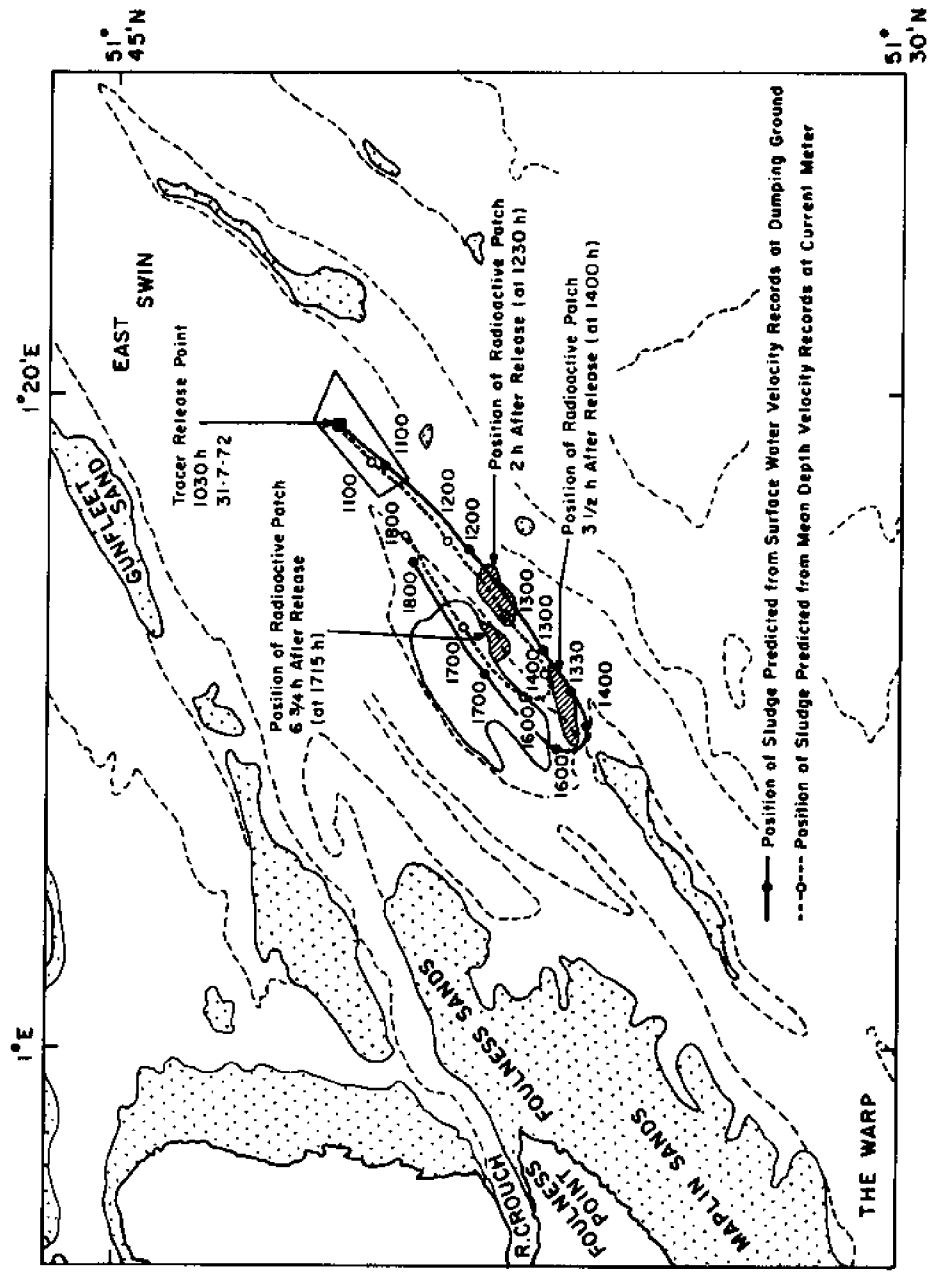
The initial dispersion paths of the sludge may either be predicted from a knowledge of tidal currents and bathymetry of the area, or be determined by measurements at the dumping area. Both approaches have been employed.

Predictive: Experiments in the laboratory have determined the extent of flocculation of particles, the range of settling velocities and the nature of chemical changes (such as the solubilization of metals from the particulate phase) which follow dilution with seawater. When these data are combined with the calculated rate of dilution of the sludge by initial turbulent mixing in the wake of the dumping vessel and subsequent advection and diffusion, it is possible to predict at least the short-term behavior of the sludge after dumping.

Experiments using Beckton sludge in seawater demonstrated that the dilutions expected to be achieved by discharging the sludge load into the vessel's wake over 10 to 15 minutes while steaming at 6 knots (> 100-fold in first 100s) were sufficiently high to avoid flocculation, the separate particles settling only slowly at 0.01 to 0.001 millimeters per second (mm/s). Since peak tidal velocities exceed 100 cm/s in the area and water depths are around 20 m, it was concluded that the sludge particles would be distributed within the water column and reach the bottom waters within the first few hours after discharge by eddy diffusion rather than by settlement.

The sludge is dumped at low water; thus initial movement will be southwest down the Barrow Deep and the track of the dispersing sludge field can be estimated from tidal current data and is shown in Figure D.2. Thus the bulk of the sludge particles can be expected to reach the seabed initially within the path shown in this figure. Laboratory experiments indicated that, with the exception of cadmium and possibly nickel, the

Figure D.2 Movement of the dumped sludge predicted from measurement of tidal currents at the dumping ground and actual positions of radioactively labeled sludge in the water column during the tidal cycle after discharge (Talbot et al., 1982).



majority of the metals originally present in the particulate phase remain there and thus reach the sediment.

Field measurements: In a study carried out in 1972 by MAFF, a load of sludge was labeled with radioactive silver and its behavior after discharge monitored. The location of the sludge 2, 3.5 and 7 hours after discharge is also shown in Figure D.2 and is consistent with the predicted movement of the sludge. Determination of the sludge location in the water column showed that the bulk of the sludge particles settled out of the upper half of the water column within 5 to 6 hours after discharge, again consistent with prediction.

Regular measurements of dissolved oxygen concentrations at the dumping ground by the Thames Water Authority have not revealed any reductions due to dumping, consistent with the good dispersion of the waste caused by the strong tidal currents.

Medium and Long-term Dispersal of the Sludge

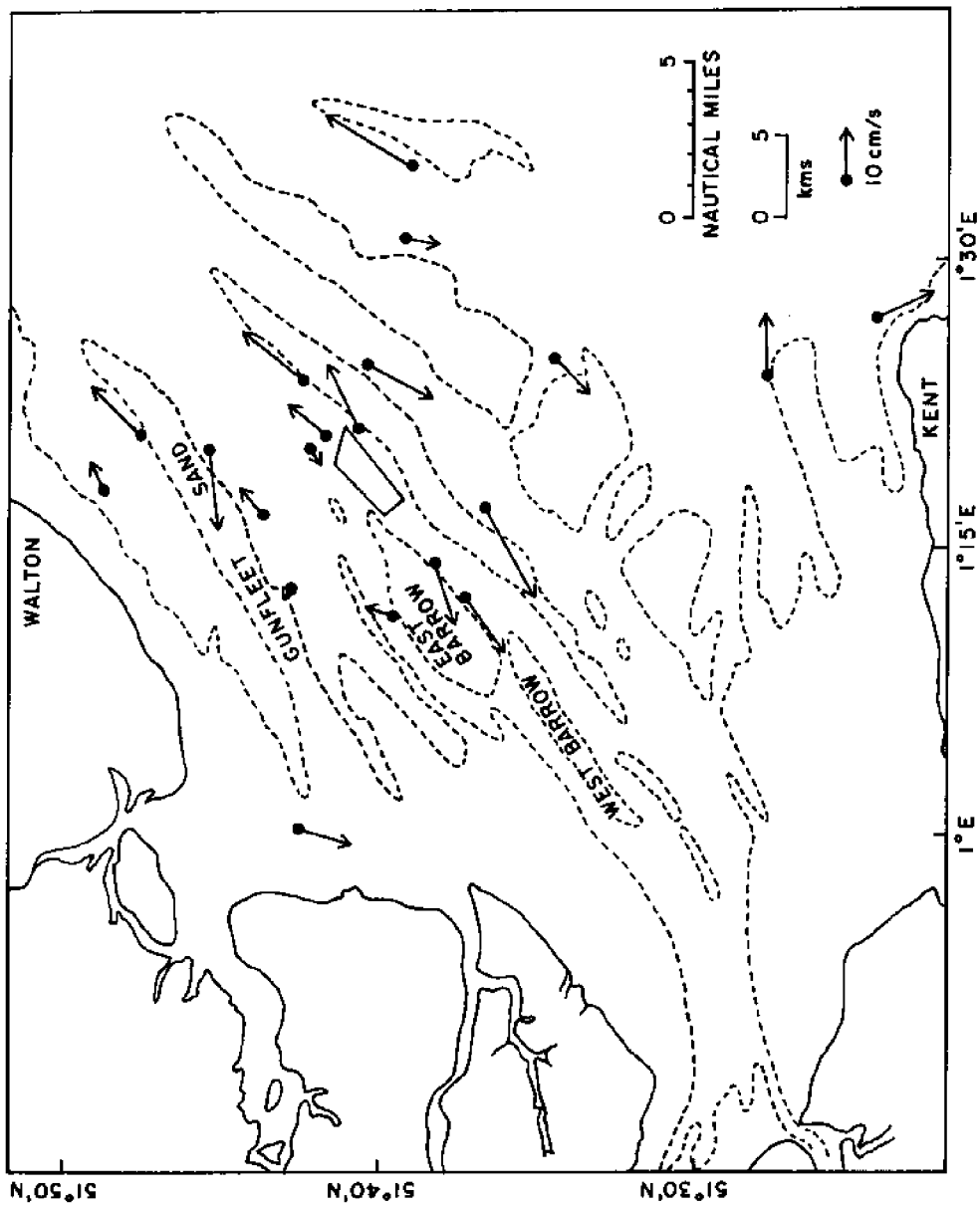
Dispersion in the water column: The movement of soluble components of the sludge and fine particles carried in suspension can be predicted from a knowledge of the residual currents of the area. These were determined by measurements in 1972 at the current meter stations shown in Figure D.3. Seabed drifters were also released from several points; 44 percent of the drifters released were returned enabling a good picture to be assembled of residual bottom water movements over extended periods.

The meter results showed currents to be highly variable and subject to the influence of winds. If the effect of winds were discounted, however, the residual pattern in Figure D.3 was observed, consistent with a general southerly residual drift in the region of the dumping area leading to a persistent loss of water from the southern part of the estuary. Under certain wind conditions, however, this residual flow may be reversed, saline water flowing into the southern part of the estuary. These findings were consistent with the results of the seabed drifter releases, which showed that the general movement of drifters was to the south, very few entering the Thames above Southend and none penetrating farther upstream than Tilbury (Figure D.1). Thus there was no evidence of a significant bottom water movement upstream into the inner estuary,

Position of moored current meter stations and direction and strength of mean residual currents for relatively calm conditions (10 knots wind) (Talbot et al., 1982).

● position of current meter

----- 5m isobath



supporting the suggestion from current meter data that dispersion in the water column would ultimately be out of the estuary.

The components of the sludge most likely to be affected by this dispersal process are those components already soluble or those which may be solubilized by prolonged suspension in, or exposure to, seawater. Soluble organic substances and nutrients are among these, but some metals may also enter the water column despite the majority being associated with particulate matter at the time of dumping. Experiments have shown that, over a period of up to 14 days, substantial solubilization of cadmium and, to a lesser extent, nickel occurred from Beckton sludge; thus the removal pathway for a significant fraction of these metals could be through dispersion and movement in the water column.

Dispersion at the seabed: The dispersion of sludge particles after they have reached the seabed and are capable of being incorporated into the natural sediments of the area will be governed by, and may be inferred from, the behavior of the natural sediments. Thus an understanding of natural sediment distribution and dynamics can assist in the interpretation of observed distributions of sludge components and may also enable the pattern and extent of sludge accumulation to be predicted.

Natural sediment movements can be induced by tidal or wave-induced water movements at the seabed. The near-bed current velocities generated by tidal flows at the dumping ground can be computed from current meter data and appear unlikely to exceed 5 to 6 cm/s, capable of moving cohesionless sediment grains of up to about 5 mm diameter. Wave-induced currents near the dumping site appear unlikely to have the strength of near-bed tidal flows, except during the strongest storms of the year and then only in the shallower waters (e.g., on and at the edge of the sand banks in the estuary). Storm surges which sweep down the east coast may, however, upset the tidal regime within the estuary, altering the strength and direction of tidal currents substantially for periods of many hours, thereby, influencing the movement of sediment in the estuary.

The present day distribution of natural sediments is a result of the above forces acting on the sediments available in the estuary. This distribution was determined by grab sampling at the stations shown in

Figure D.4 in 1976-77, by side-scan sonar and from published data on the geology of the estuary (e.g., D'Olier and Maddrell, 1970).

Superficial deposits in the Outer Thames Estuary are comprised mainly of sand, shell or gravel in depths of a few centimeters to as much as 50 m. Much of the sediment is contained in large linear sandbanks, oriented southwest-northeast. In the central region of the estuary (Barrow Deep area), the sandbanks are composed almost entirely of very mobile and well-sorted fine sands with extensive areas of sandwaves and megaripples. The channels between the banks are usually floored by a heterogeneous mixture of gravel, shell, mud and sand, with localized incursions of sand from adjacent banks. This variation in sediment types is seen along a section of the Barrow Deep surveyed by side-scan sonar (Figure D.4), where gravels and outcrops of underlying clay in the Central Deeps area pass into megarippled sand as the shallow banks are approached. A complete description of the distribution of sediment types would be too lengthy for this appendix, but certain features revealing the sediment dynamics of the area will be described which enable the fate of dumped particulates to be predicted.

First, it can be shown (from asymmetry in the tidal currents) that the mobile sands in the banks tend to move in a clockwise transport pathway around the banks, forming circulation cells. These cells are largely self-maintaining entities, but do provide a mechanism whereby material can be buried in the mobile sediment. Second, muds occur in the estuary from a variety of sources (local erosion of outcrops at the seabed and intertidal marsh deposits, runoff, etc.) and the survey of sediments revealed areas where particles of this size range may settle out and accumulate (e.g., as interstitial deposits in gravelly or sandy substrates). The distribution of mud (<63 μm) in the region of the dumping area is shown in Figure D.5. Since the particle size of much of the sewage sludge dumped is in the mud range, this distribution may identify areas where sewage particles may accumulate in the sediment.

The insights into the possible behavior of sewage sludge at the seabed, gained from the study of the natural sediments above can be compared with the results of the radio-tracer study already described, which followed the fate of the labeled sludge particles for up to six months. This survey identified one initial zone of settlement of the sludge within the

Figure D.4 Sampling stations during 1976-1977 surveys and a general description of sediment types along a longitudinal transect of the Barrow Deep (from side-scan sonar survey).

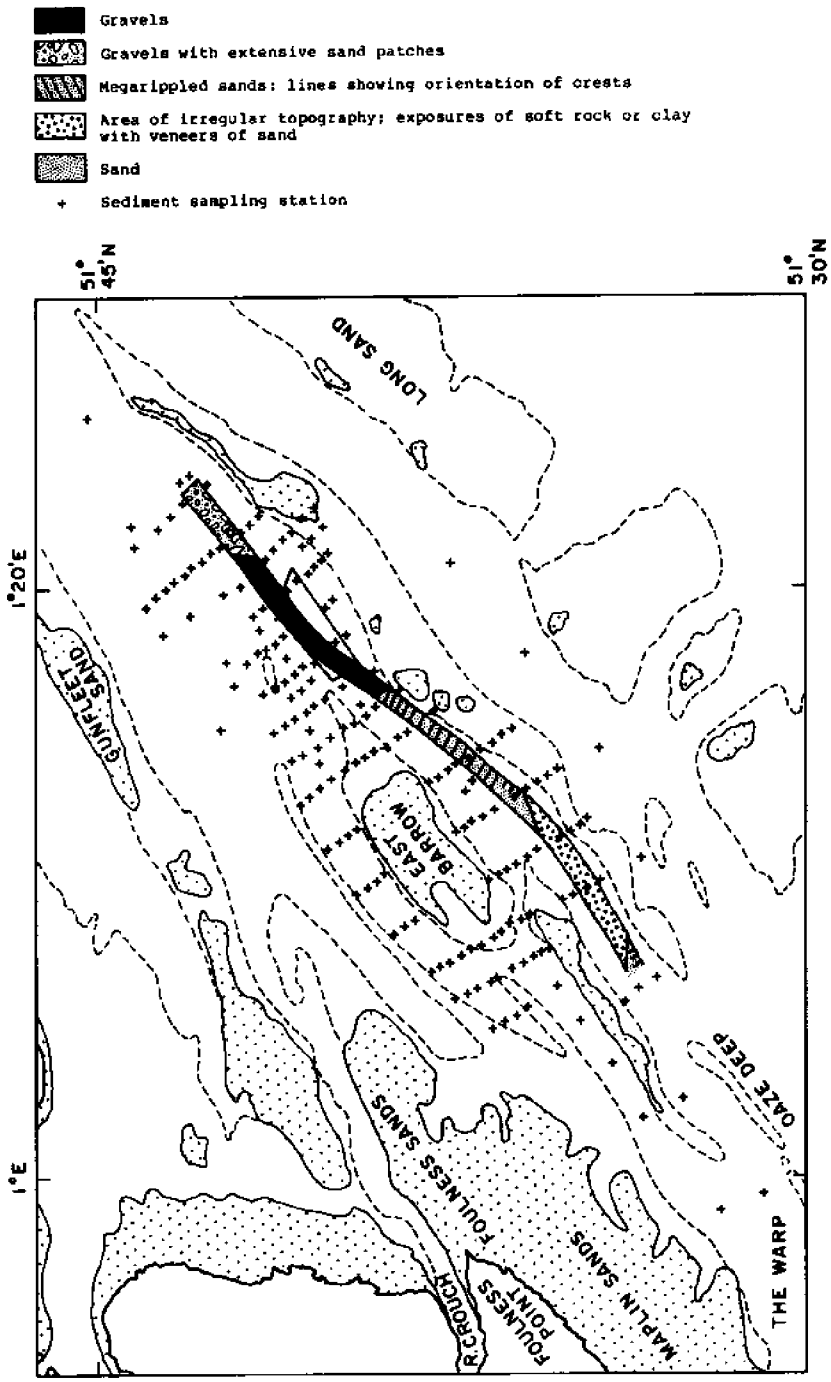
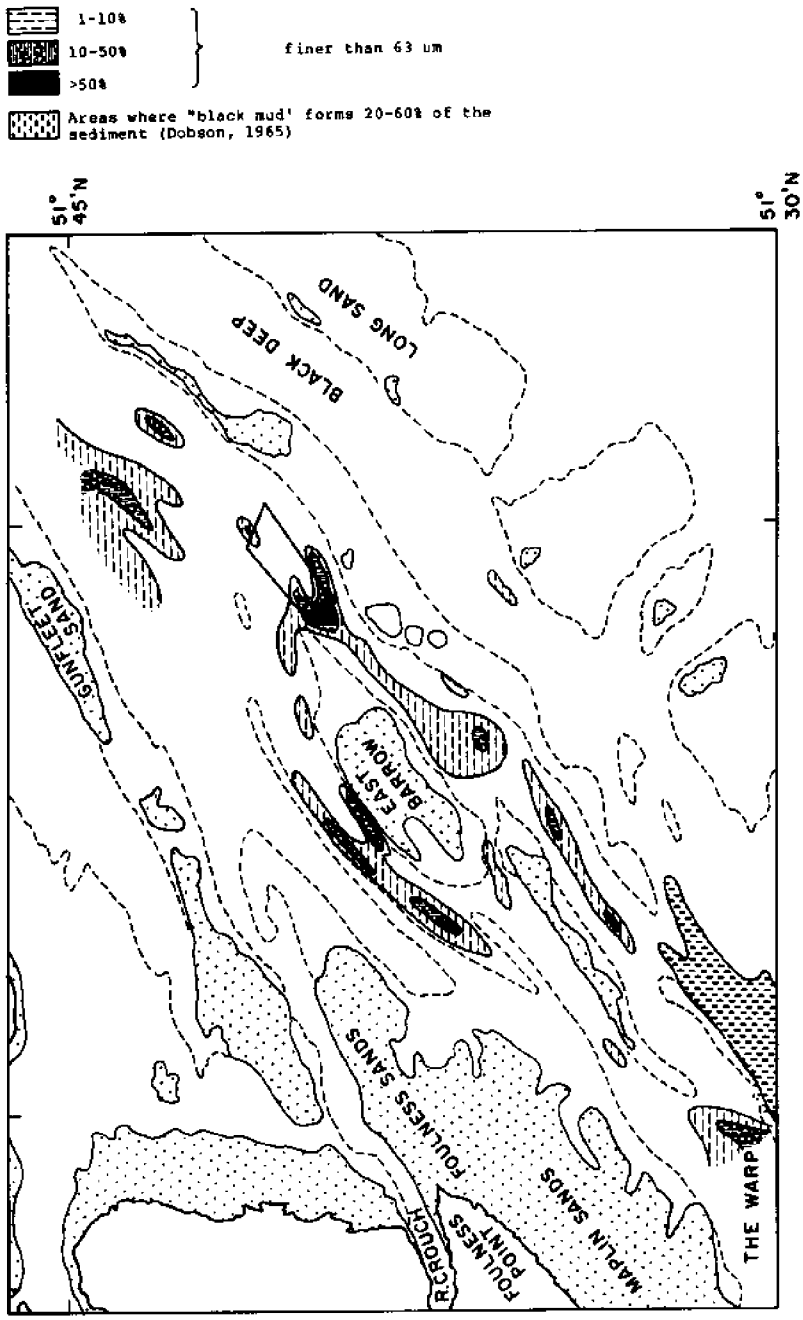


Figure D.5 The distribution of mud (<63 μm fraction of the surface sediment) in 1976-77.



first tidal cycle close under the East Barrow Sand (Figure D.6). This area of radioactivity was shown to move slightly during the period of observation toward the southeasterly side of the Barrow Deep Channel (Figure D.6). The distribution of radioactivity showed the sludge to have been buried up to 20 cm in the sandy sediment, supporting the burial mechanism postulated from the study of natural sediments. Further areas of settlement of radioactively labeled sludge coincided with areas of mud in the Barrow and Middle deeps shown earlier in Figure D.5. Thus the tracer release tends to confirm the predictions of sludge movement made from the study of natural sediments.

Initial settlement of the sludge particles may thus result in their incorporation and dilution with the naturally mobile sand and in the interstices of gravels or sand. The potential therefore exists for the accumulation of sludge in some parts of the Barrow and Middle deeps and along the flanks of the adjacent banks. The mobility of the sediment over most of the area suggests that accumulation would be short-lived. The only areas of potentially longer-term accumulation identified by the study of natural sediments were the gravels close to the dumping area, and the southwestern end of the Middle Deep, where natural sediment is accumulating and sludge particles incorporated in the sediment would not be extensively reworked and dispersed.

Contamination of Sediments by Sludge Particulate Matter

The distribution of sludge particulate matter in the sediments can be determined by the measurement of several parameters including fecal coliforms, organic carbon and several metals, the concentrations of which are shown in Figures D.7, D.8 and D.9(a-f) respectively.

It can be seen from these figures and from the table of correlation coefficients for these variables (Table D.3) that there is a highly significant correlation between the fecal coliform content, the organic carbon content and the metals. The contamination of the estuary in the vicinity of the sludge dumping ground can thus be clearly attributed to the sewage sludge, although the influence of sewage discharges may be felt at the inshore area of the Warp Channel (Figure D.8). The extent of the contamination observed in 1976-77 is given in the figures, and can be seen to include those areas where sludge was expected to accumulate. Maximum concentrations of organic carbon in the sediment were 1.6 percent (equivalent to

Figure D. 6 Settlement zones of radio-activity labelled sludge and subsequent direction of movement (Talbot et al., 1979).

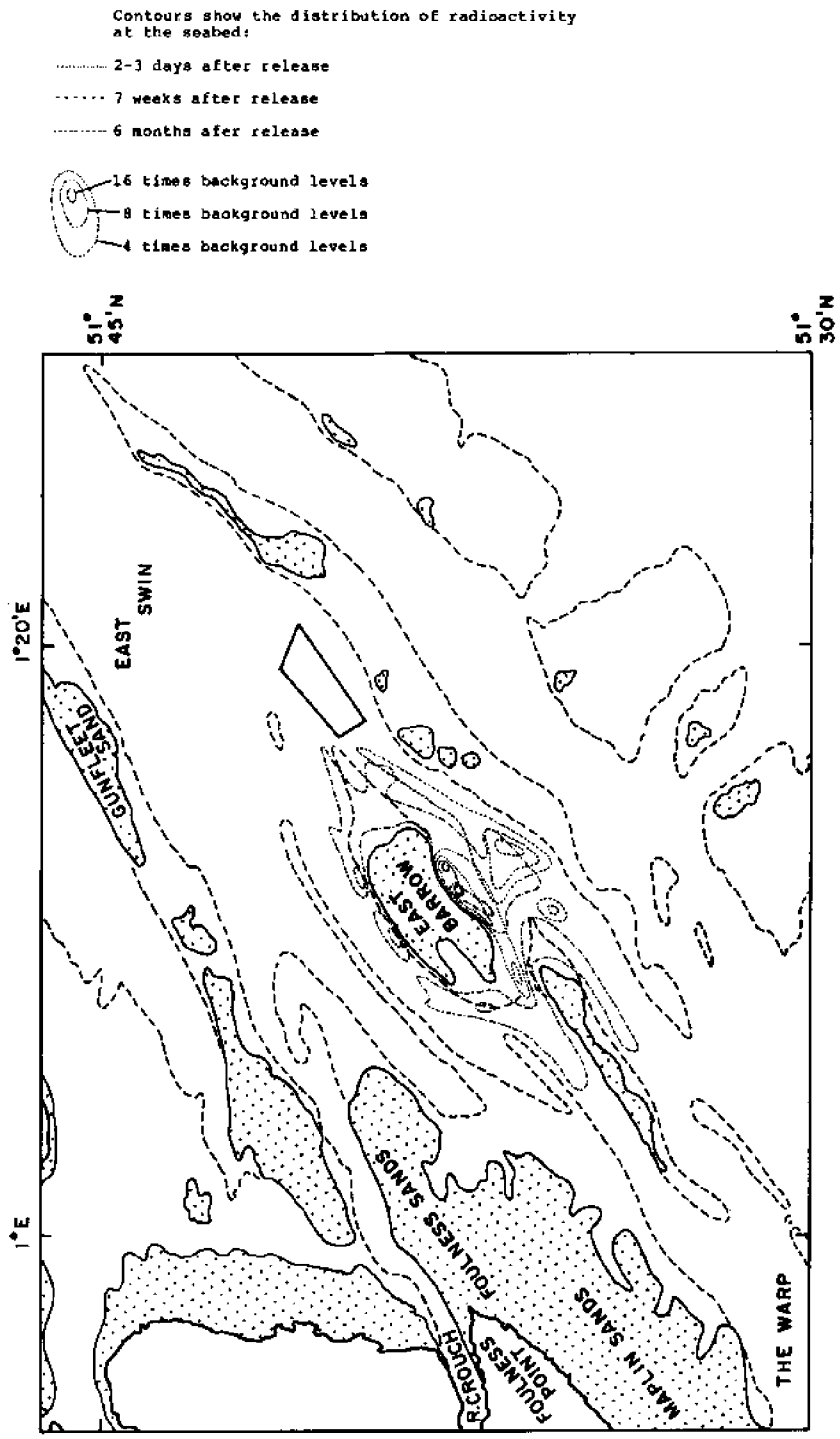


Figure D.7 The distribution of fecal coliforms in the surface sediment.

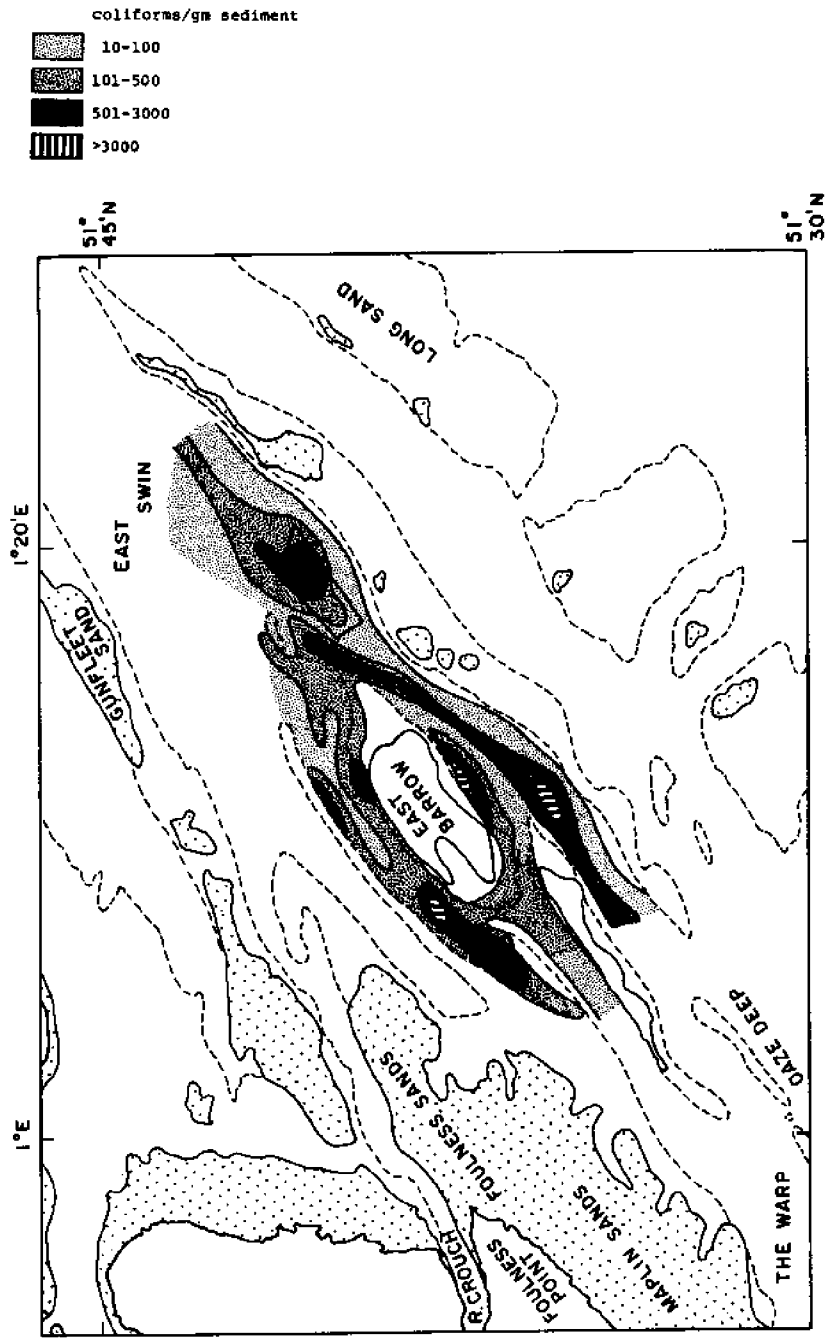


Figure D.8 The distribution of organic carbon in the surface sediment

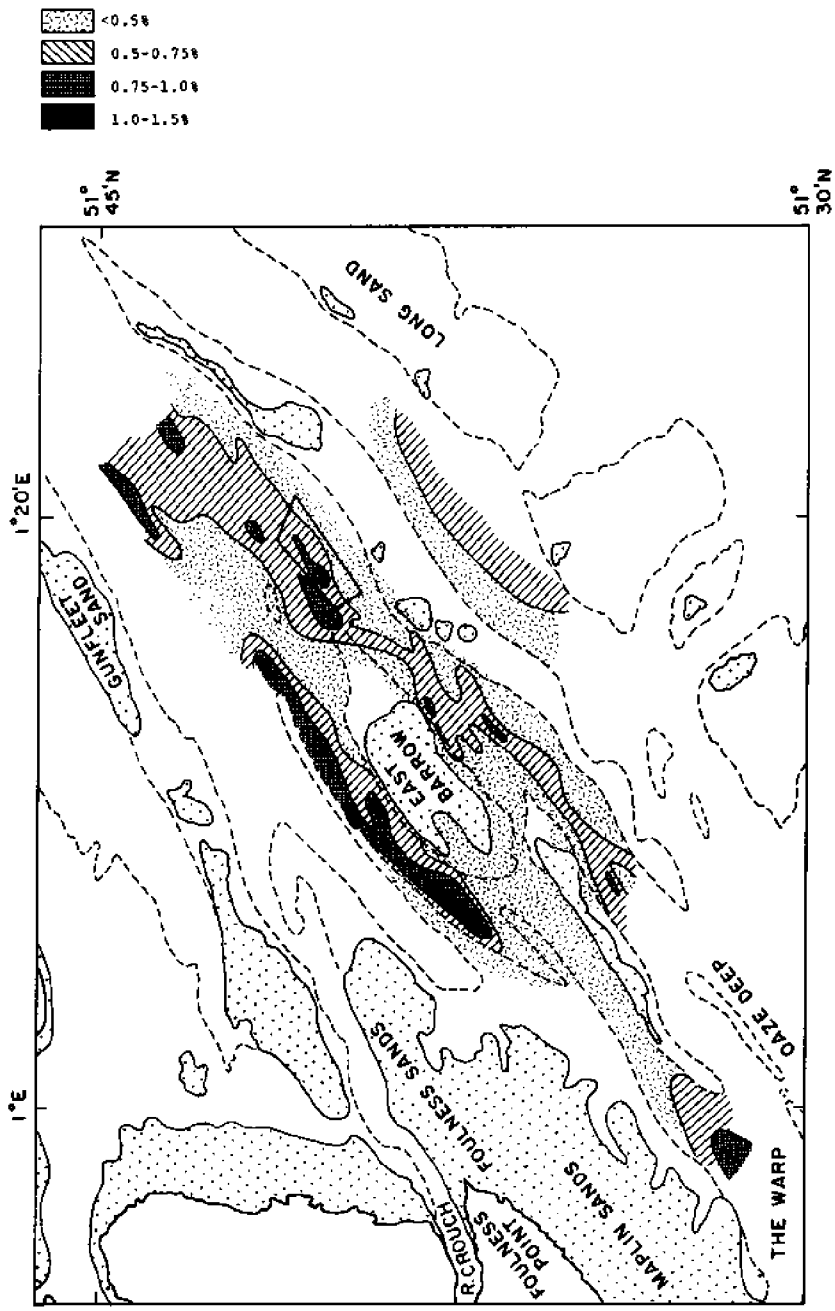


Figure D.9a Concentration of mercury in the <90 µm fraction of the sediments. Isolines in mg/kg x 10^2.

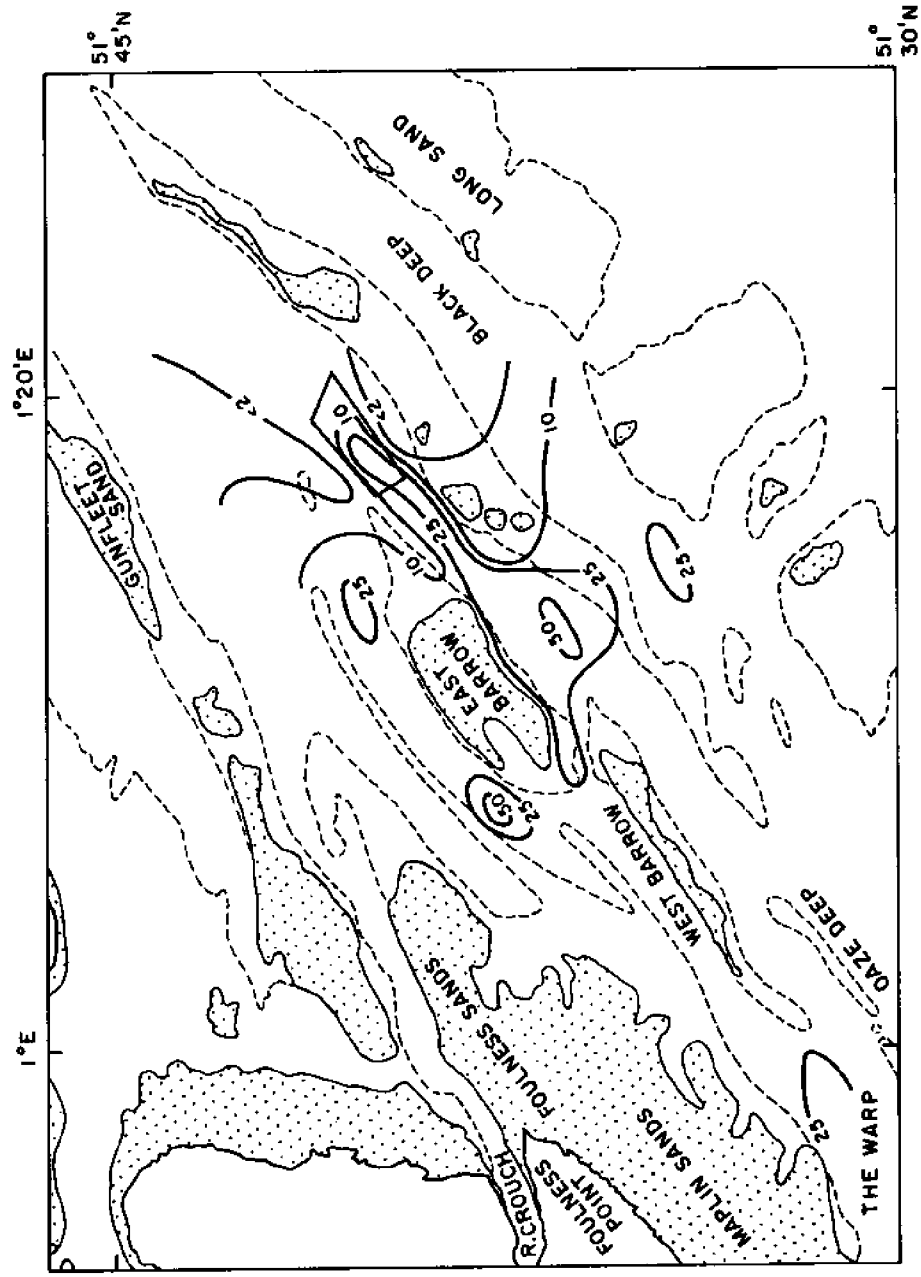


Figure D.9b Concentration of zinc in the <90 µm fraction of the sediments. Isolines in mg/kg.

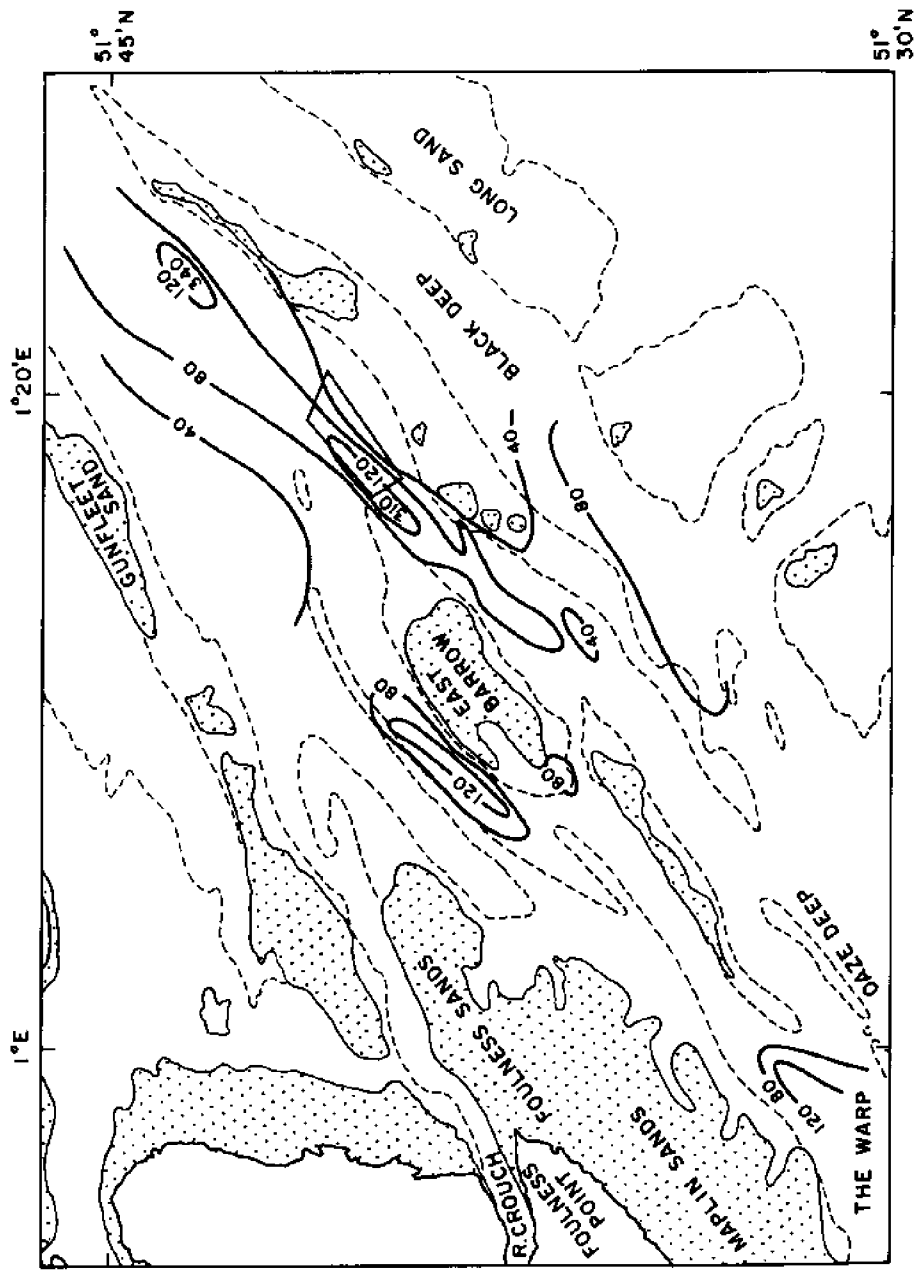


Figure D.9c Concentration of copper in the <90 µm fraction of the sediments. Isolines in mg/kg.

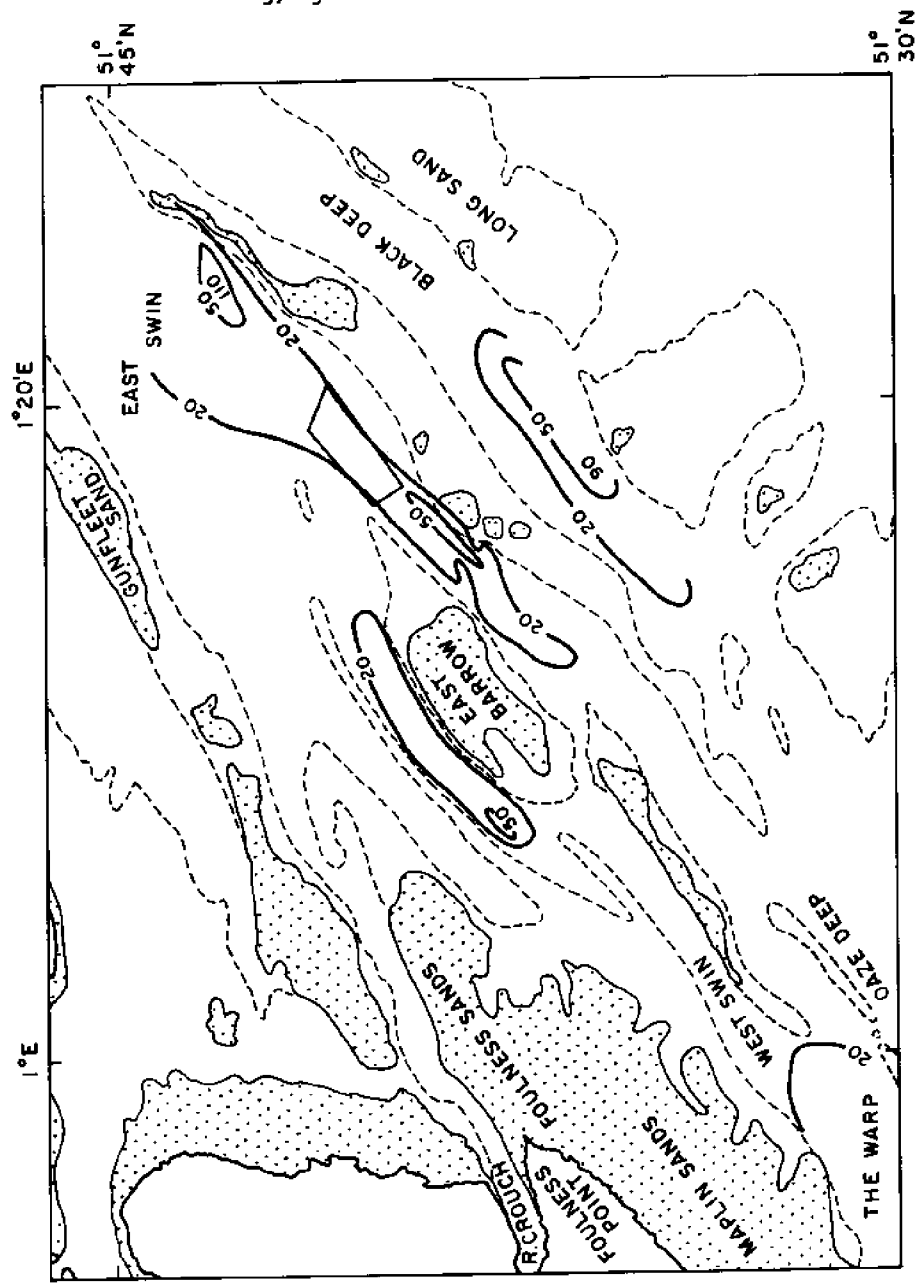


Figure D.9d Concentration of lead in the <90 µm fraction of the sediments. Isolines in mg/kg.

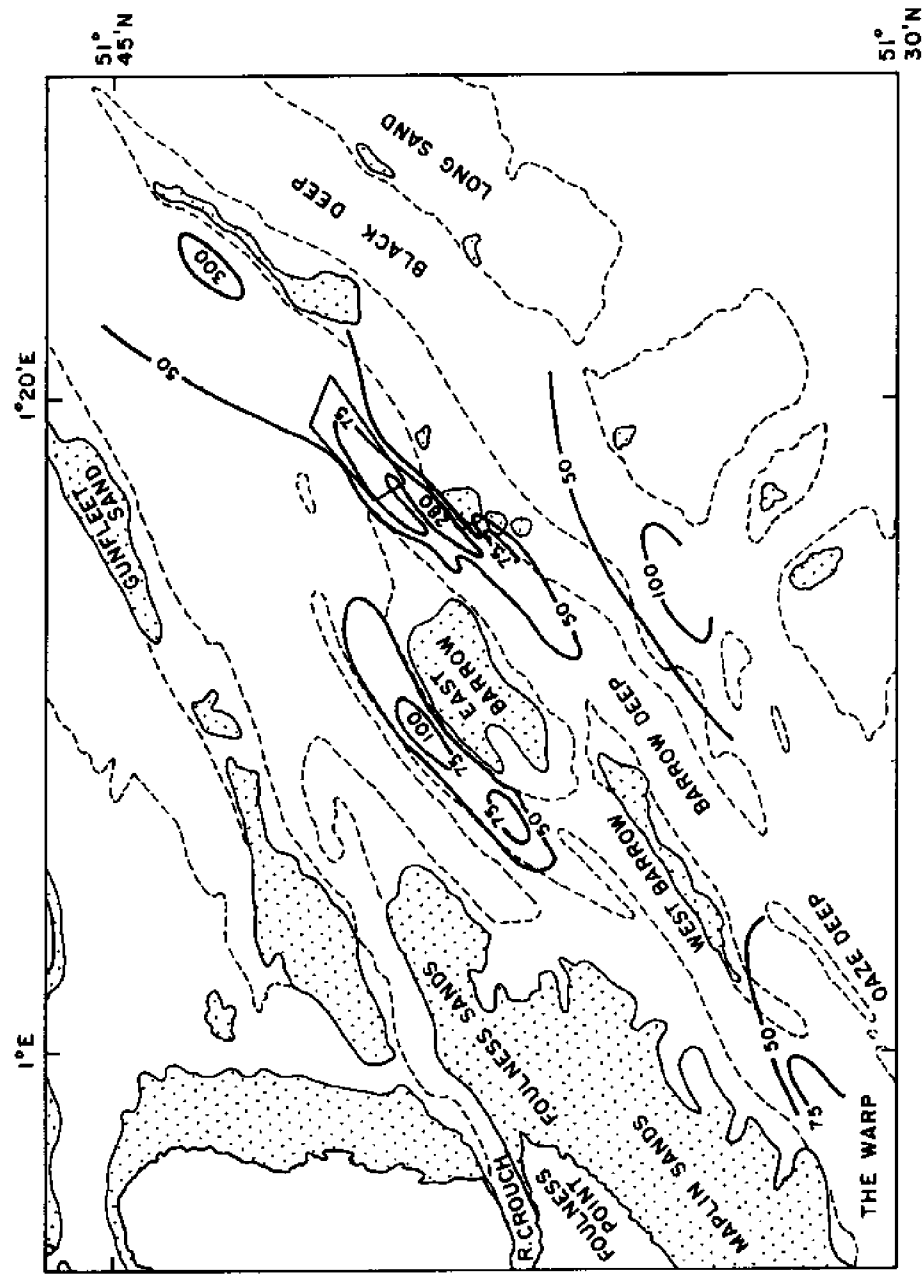


Figure D.9e Concentration of chromium in the <90 µm fraction of the sediments. Isolines in mg/kg.

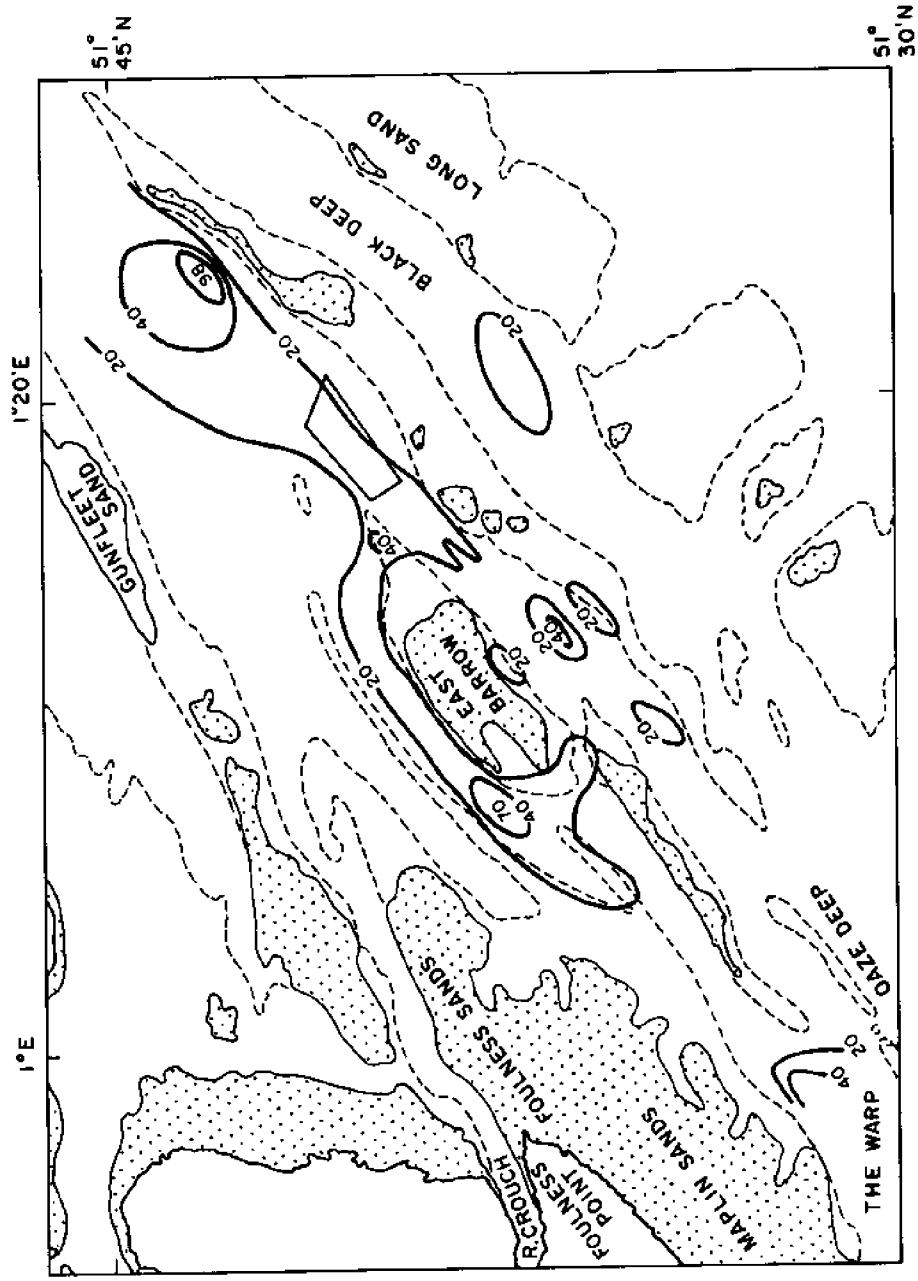


Figure D.9f Concentration of nickel in the <90 µm fraction of the sediments. Isolines in mg/kg.

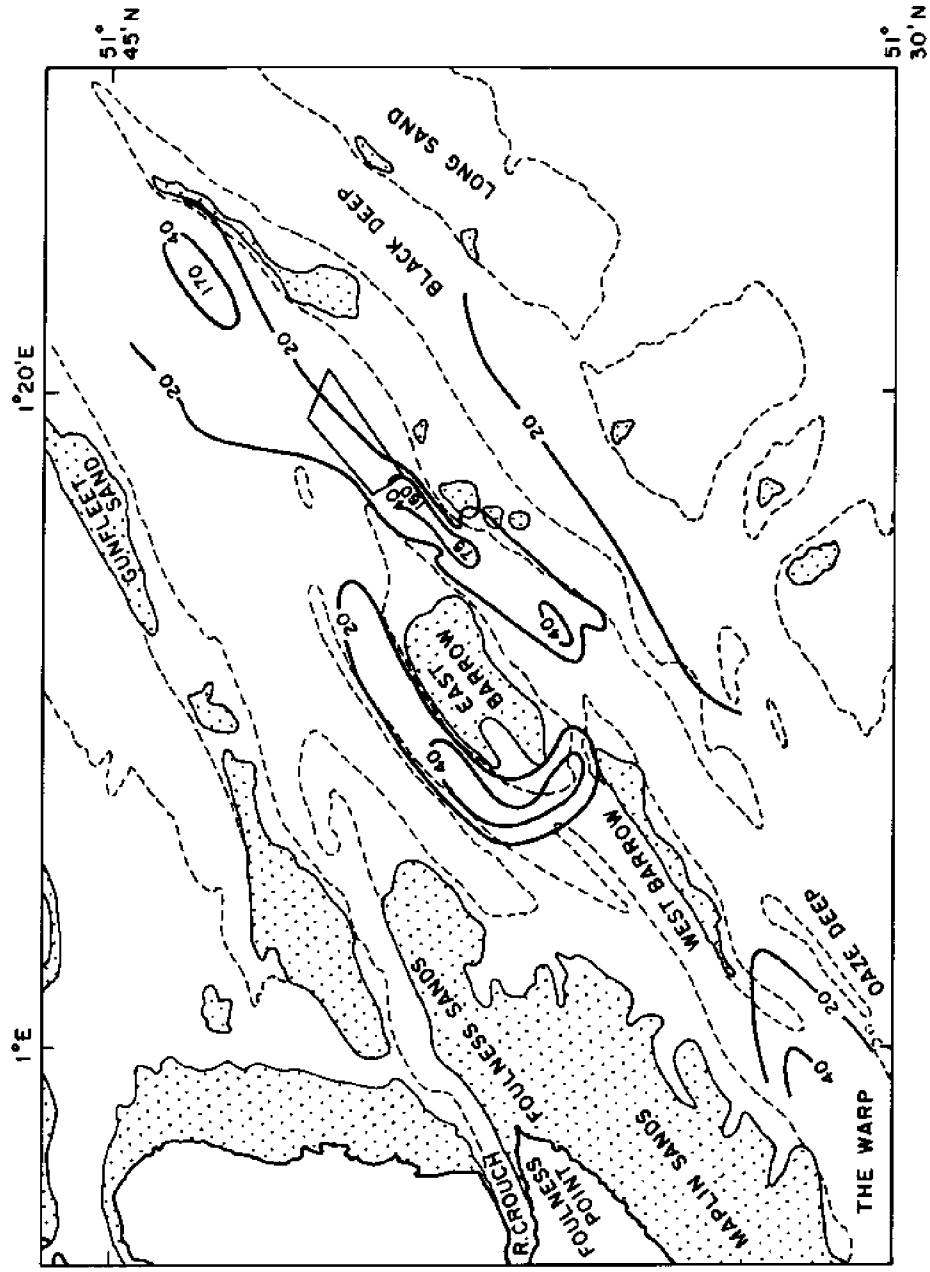


Table D.3 Correlation Between Various Organic and Trace Metal Variables

Variables correlated		Correlation coefficients	N	P(0.001)
% clay in the <90 µm fraction	% organic carbon in <90 µm fraction	-0.0520	69	0.380
E.coli/gm sediment	% organic carbon in <90 µm fraction	+0.4521	110	0.321
Zn (<90 µm)	% C (<90 µm)	+0.7200	81	0.357
Cu "	"	+0.6129	"	"
Pb "	"	+0.6921	"	"
Ni "	"	+0.4052	"	"
Cr "	"	+0.3940	"	"
Hg "	"	+0.5170	46	0.465

N = number of degrees of freedom, P (0.001) = value above which correlations are significant at the 99.9 percent level.

approx. 4 percent organic matter). The concentration of mercury in the 90 m fraction of the sediment was as high as 0.7 mg/kg in the Middle Deep. This metal, together with zinc, copper and lead, was strongly correlated with the organic carbon content of the sediment, confirming the dominance of the sewage sludge source in the area. Chromium and nickel were not so highly correlated; in the case of chromium this was found to be due to the greater importance of minerologically bound metal in the area, but for nickel it was believed to be due to loss of the metal from the particulate phase to solution after dumping. This latter effect was more apparent for cadmium, where the metal was only detected in areas of initial sludge settlement between the Barrow and Middle deeps - other areas of organic enrichment having presumably lost cadmium to solution.

Biological Effects

Effects in the water column: The detection of any adverse biological effects in the water column is difficult due to the high degree of spatial and temporal variability of the pelagic ecosystem. The likelihood of toxic effects arising from the dumping of sewage sludge can, however, be predicted from laboratory tests and from the knowledge of the rate of dilution of the dumped sludge.

In the case of the sewage sludge dumped in the Thames Estuary, acute toxicity tests on a number of species of marine organisms have been conducted, and concentrations in excess of 20,000 mg/l over exposure periods of 4 days or more were found to be required to cause mortality to common marine organisms.

As has been stated in a previous section, the initial dilution is likely to be in excess of 100-fold in the first minute after discharge and further dilution to around 1,000-fold will occur during or soon after the turbulent phase of mixing (i.e., first 10 to 20 minutes). Subsequent dilution will be slower and dominated by eddy diffusion from the tidal currents. Thus concentrations shown to be lethal over the period of the bioassay (4 days) are not likely to be encountered in the wake of the dumping vessel for more than a few minutes, and the probability of acute toxic effects is low. Other possible effects on the water column could arise from the introduction of large quantities of nutrients. In this area of the North Sea, however, it appears that high turbidity is the

factor restraining primary production, so local promotion of primary production appears unlikely. Over a wider area, increased nutrient inputs may have led to some changes in plankton community structure, but the contribution of sewage sludge relative to river runoff is small (Table D.2).

Effects on the benthos: The extent of any biological effects arising from the contamination of the sediments described earlier can be determined by sampling the biota quantitatively, preferably at the same time that a sediment survey is conducted. Such a joint survey was carried out by MAFF in 1977, when 175 stations in the vicinity of the dumping area were sampled for both benthos and sediments. The fauna retained by a one mm sieve were identified to species level where possible, and the faunal data were processed by computer and analysed by classification and ordination to identify groupings and trends within the fauna which could be related to the environmental variables.

The classification of the faunal data allowed eleven associations to be identified at the 65 percent level of similarity which are shown in Figure D.10. The faunal abundances and species richness are given in Figures D.11 and D.12 respectively. A detailed description of the associations identified and their characterizing fauna would be too lengthy for this appendix; however, some of the associations in Figure D.10 may be influenced by sludge dumping and will be discussed.

Animal association 10 was confined to the vicinity of the sludge dumping around the area of muddy gravels, and the number of species in this association was higher than in similar sediment types elsewhere. Whether the apparent increase in species richness was due to dumping or to other differences in the characteristics of the sediment has not, at the time of writing, been established statistically.

Some areas of the Barrow and Middle deeps exhibited blackening of the sediments. Animal association 6 was virtually confined to sediments which showed some degree of blackening, but none of the accepted "pollution indicator species" was common. Other associations showed no clear preference for, or avoidance of, blackened sediments. Statistical treatment of the faunal data indicates that, within sandy sediments (associations 1, 2, 4, 10 and 13), those sampling stations that were blackened had a

Figure D.10 Distribution of animal associations (1-11) identified by classification of the faunal data.

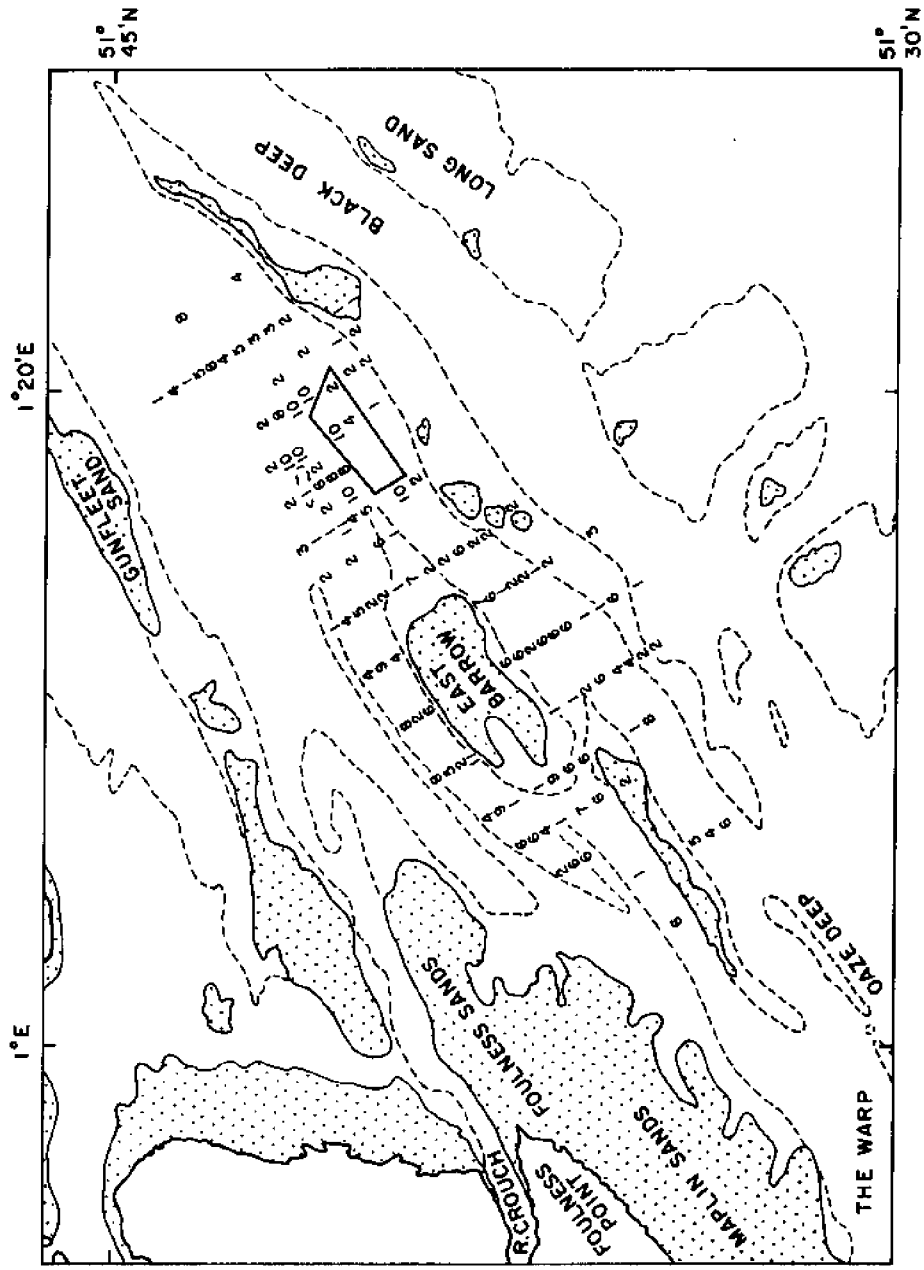


Figure D.11 Faunal density. Numbers show the number of animals per m^2.

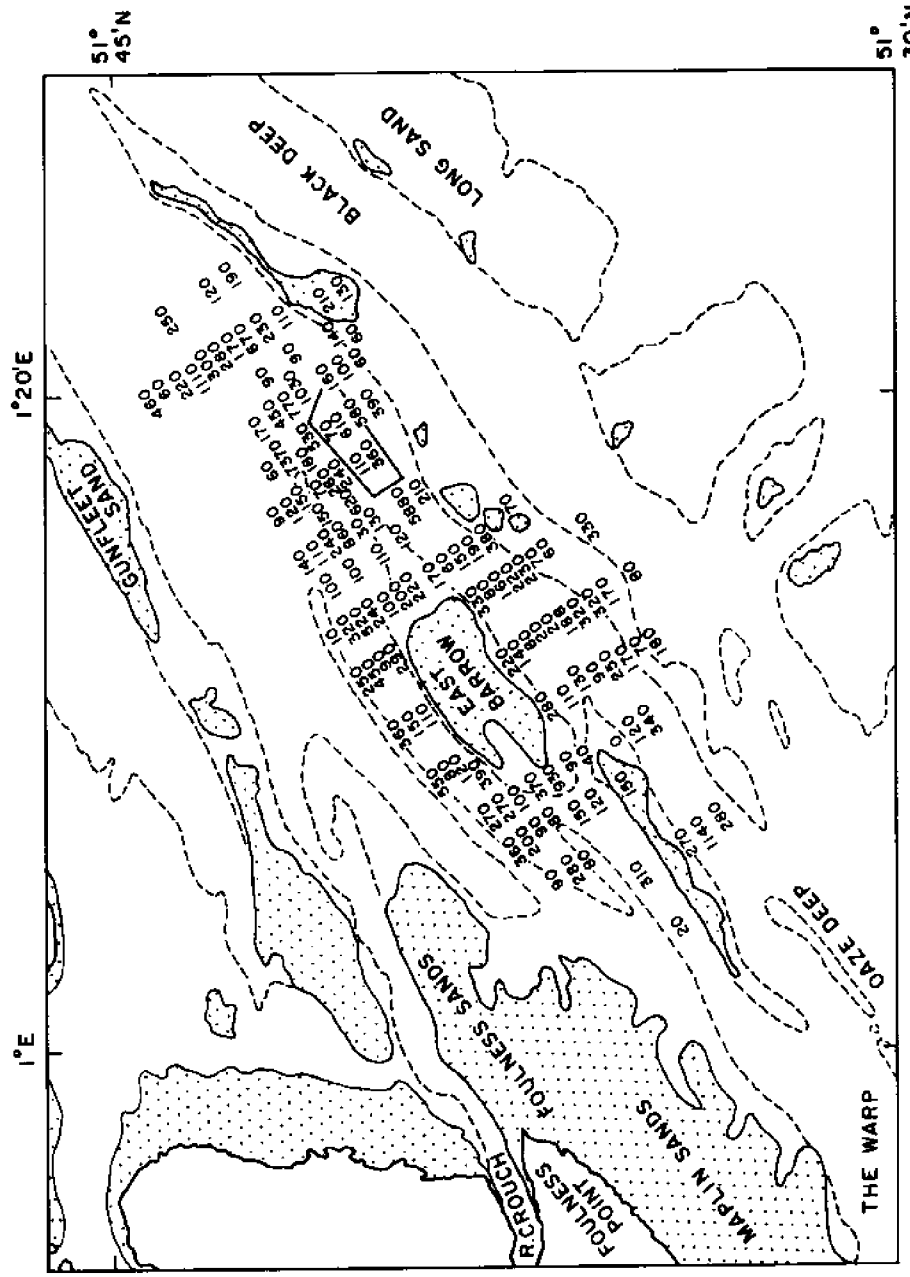
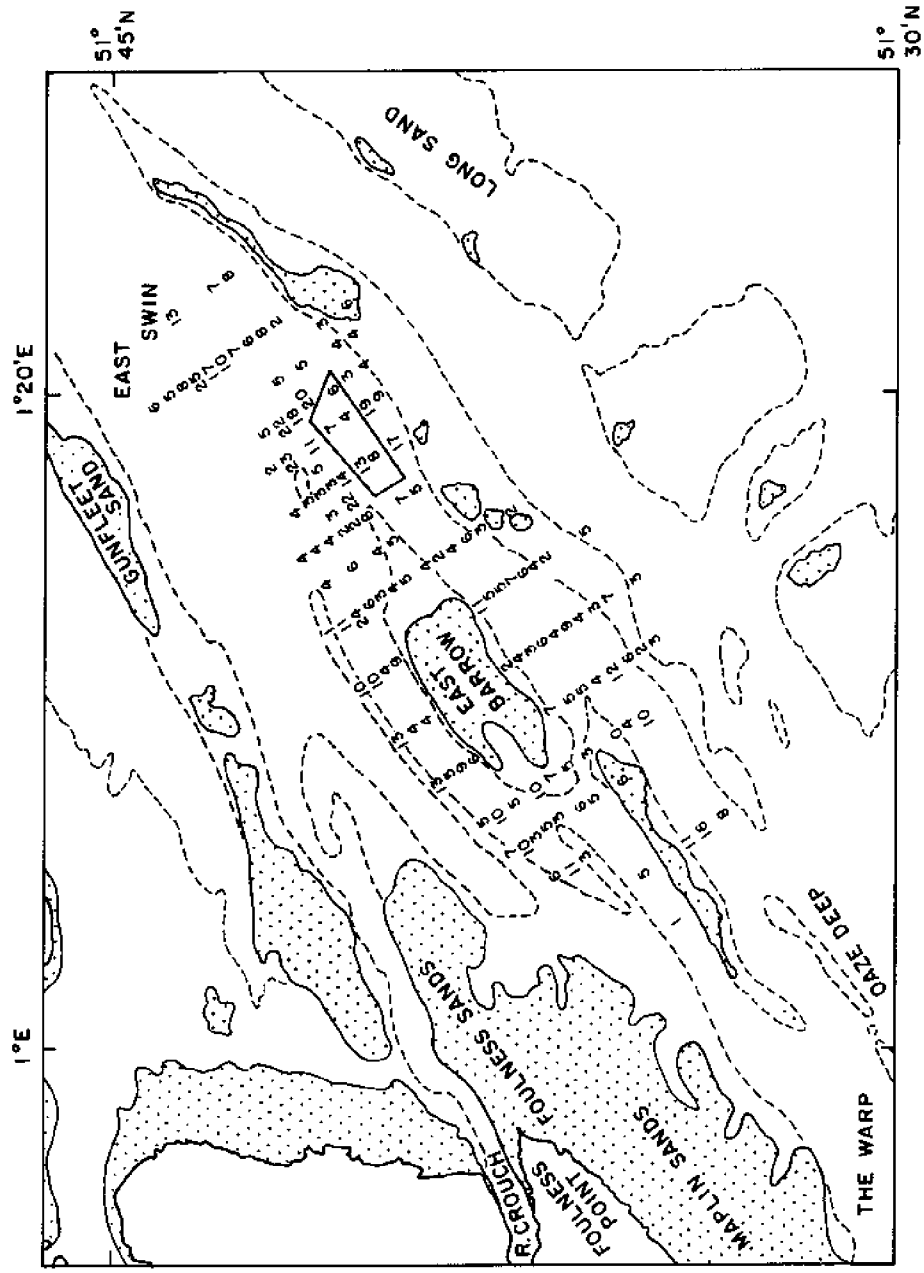


Figure D.12 Species richness. Numbers denote number of species identified per 0.1 m^2.



significantly lower density of animals. The number of species present did not, however, appear to be affected by sediment blackening. Since the areas of blackening coincide with areas of organic enrichment caused by sludge dumping, a causative link is suggested with the reduction in faunal density.

A full statistical analysis of the relationships between faunal abundance/diversity and the physical/chemical characteristics of the sediment was also carried out. This showed that in some associations (1, 4, 10), sediments with increased carbon content were inhabited by fewer animals, fewer species or both. Only in association 6 was a positive correlation suggested between the faunal abundance and diversity and carbon. This latter association contained a number of species that would be expected to be tolerant of increased concentrations of fine organic material.

The general picture that emerges from the statistical analysis is therefore that the fauna was largely determined by the natural substrate and hydrographic conditions. These faunal associations did not appear to respond by utilizing the organic input; thus in most of the associations increased carbon concentrations in the sediment appeared to slightly reduce the number of species present, their abundance or both. On the other hand, in a few limited areas it appeared that the faunal community may have adapted to exploit higher concentrations of organic carbon. The overall effects of dumping on the benthic ecology of the area were thus small.

Chemical Quality of Fish and Shellfish

Although direct effects on the biota themselves do not appear to arise from dumping, there is a potential impact on the commercial fishery through uptake of metals or other persistent substances by fish and shellfish. Thus, the chemical quality of commercial species taken from the Outer Thames Estuary has been monitored by MAFF since 1970.

With the exception of mercury, the concentrations of metals in commercially exploited fish and shellfish from the estuary have been found to be similar to those in other UK coastal waters; thus the contamination of the area's sediments detected analytically does not appear to have led to significant accumulations in marine organisms. In the case of mercury, however, concentrations in demersal fish were found to be

elevated in 1970-71 relative to other areas, and subsequent investigations of the possible sources indicated that the largest input to the Outer Thames Estuary was via sewage sludge (6.5 out of a total of 8.5 tonnes/year). Subsequently, the concentration of mercury in the sludge was reduced by 75 percent through pretreatment at an industrial works in late 1974. Since then, mercury levels in fish have fallen: in the case of the flounder, which is resident in the estuary, from 0.74 mg/kg in 1970-71 through 0.5 mg/kg in 1976 to 0.3 mg/kg in 1977.

Management Considerations

The dumping of 5 x 10^6 tonnes per year of sewage sludge in the Outer Thames Estuary has been shown to cause local effects on the composition of the sediments, as indicated by elevated levels of bacteria, organic substances, and persistent components of the sludge such as metals. The area is, however, one of mobile sediments, so that extensive accumulations of dumped material and major dumping-induced changes appear unlikely.

Minor and local effects on the benthos have been suggested but, since they include a possible enrichment effect, are not believed to be ecologically harmful. The only deleterious effect to have been attributed to sludge dumping was the accumulation of mercury in fish caught in the estuary. Following controls in industrial discharges to the sewage works, however, mercury levels in fish have returned to acceptably low levels.

At present levels of dumping and under the present controls in sludge quality, it appears therefore that disposal at sea is without significant environmental effects on the receiving area, and dumping in this location is expected to continue to provide the most acceptable disposal method for the sewage sludge from eastern London for the forseeable future.

Sewage Discharges to the Thames Estuary

History. Prior to the industrial revolution, the River Thames supported a large fishery based on whitebait, shad, smelts, salmon and sea-trout with its outlet at the present-day market of Billingsgate. As the population of London grew and increasing amounts of

sewage entered the river following the advent of the water closet during the nineteenth century, the quality of the Thames declined. By the 1840s and 50s, the pollution of the Thames had eliminated much of the fishery and was a source of cholera due to the continued extraction of drinking water from the river. In summer, the river gave off sulphidic odors, even interfering with the work of the Houses of Parliament.

These factors led to the creation by Parliament in 1856 of the Metropolitan Board of Works, having a duty, inter alia, of taking charge of main sewers and constructing a comprehensive system of main drainage. They appointed a Chief Engineer, Sir Joseph Bazalgette, whose sewerage scheme was formally opened in 1865 and involved the diversion of the sewage flow to Beckton on the north bank and Crossness on the south bank to be discharged on the ebb tide. This scheme improved the condition of the river in the metropolis, but led to deterioration in the locality of the outfalls. As a result of public demands, the situation was improved in 1882 by the provision of sedimentation channels at the two outfalls to reduce the solids content by treatment with coagulants, and transport of the resulting sludge to sea. By the end of the century, the river was returned almost to its condition a century earlier.

After 1909, responsibility for pollution control passed to the Port of London Authority (PLA). Further improvements in the quality of treatment followed with the completion of the large Mogden works in 1936 and the installation of partial biological treatment at Beckton. Despite this, the increasing population resulted in a further decline in river quality to 1940. This decline was accelerated by bomb damage to sewers and sewage works during the 1939-1945 war, by the lack of funds for reconstruction afterward, and by the introduction of non-degradable detergents which reduced the transfer rate of oxygen from the atmosphere to water, resulting in less efficient sewage treatment and reducing the self-purification capacity of the river. These problems led to the formation in 1951 of an expert committee to examine the mechanisms by which discharges exerted their polluting effect and to suggest the most effective remedial action. This committee concluded that the depletion of oxygen through the degradation of organic pollutants was the major deleterious influence on the estuary and considered that maintenance of minimum levels of DO, which would prevent nuisance throughout the estuary, should be the basic objective of improvements in water quality. Sewage works were found to contribute 79 percent of the total BOD inputs to the estuary, and it

was thus recommended, inter alia, that efforts be directed to improving sewage treatment at Crossness and Beckton.

Subsequently, improvements comprising primary settlement, biological treatment, and sludge digestion were completed in 1964 at Crossness and in stages until 1974 at Beckton. Major industrial effluents were also controlled by pretreatment at source or, in one case where this was not possible, by the installation of aerators in the river to put in more oxygen than the effluent removed. The changeover to degradable detergents, which took place in the 1960s, also assisted in raising treatment standards and the degree of river self-purification.

The results of this program are best shown by the improved DO levels over the years 1950 to 1977. Figure D.13 shows levels of DO are now always high enough to avoid putrescence and exceed those necessary (30 percent) to allow the passage of migratory fish under normal conditions. This improvement has also been achieved despite the depleting effect of an increased river temperature of 3°C due to its use as coolant by power stations. However, the combined storm water/sewage flow in many of the sewers can still lead to pollution of the river following heavy rainfall.

The effect of improvements in water quality on fisheries has been dramatic. In 1957 a length of over 60 km of the river from Gravesend to Kew (Figure D.15) was devoid of fish (except for a few eels which could breathe at the surface). The first fish returning to the cleaner river were caught on the intake screens of power stations in 1965, and subsequent monitoring of the fish caught by this method showed a steady increase in the number of species reaching the inner estuary. Currently, over 80 species of fish are found in the estuary including the juvenile stages of several species of commercial importance (such as herring, flounder, plaice and dab). The recovery of the estuary's fish is shown in Figure D.14 where the increasing number of fish species caught on the intake screens of West Thurrock power station are shown for 1963 to 1978. Finally, the water quality has been considered good enough by the Thames Water Authority (who took over responsibility for the Thames in 1974) to start restocking the river with salmon.

Figure D.13 Third quarter oxygen sag curves for periods between 1895 and 1977 (Wood, 1971, 1973, and 1978).

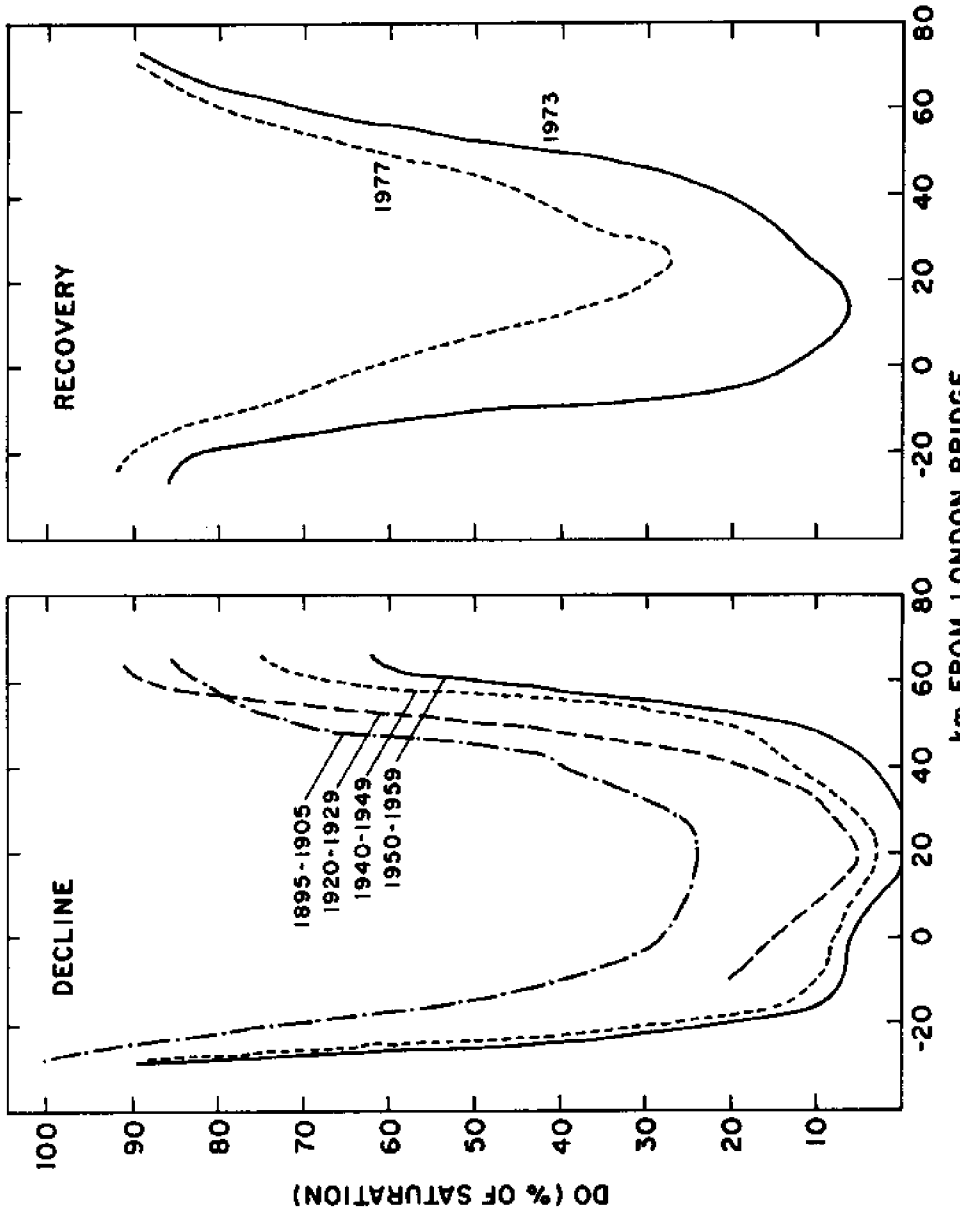


Figure D.14 Number of species of fish caught on intake screens of West Thurrock power station (M. Andrews, Thames Water Authority, personal communication). (Total number of fish species identified 1963-1978 is 66).

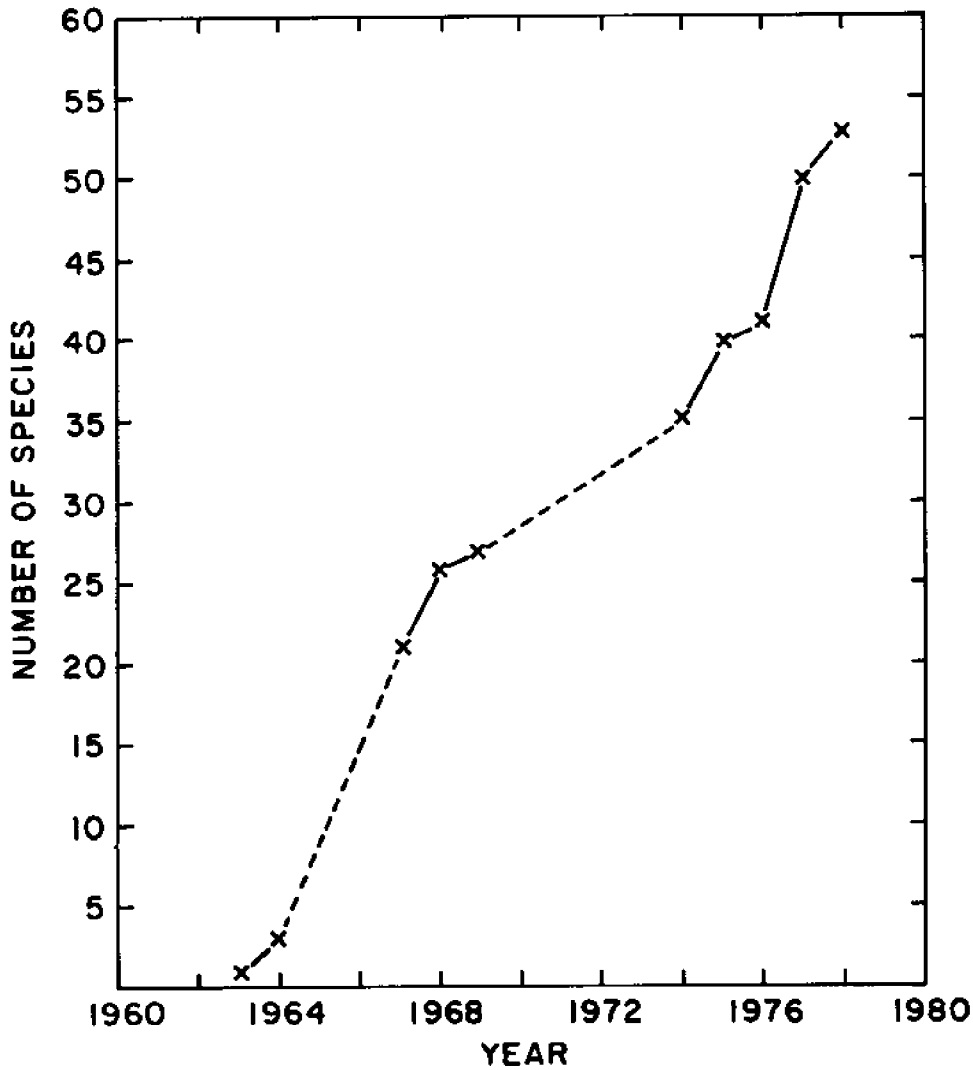


Figure D.15 The tidal reaches of the River Thames and the major sewage discharges (Wood, 1978).

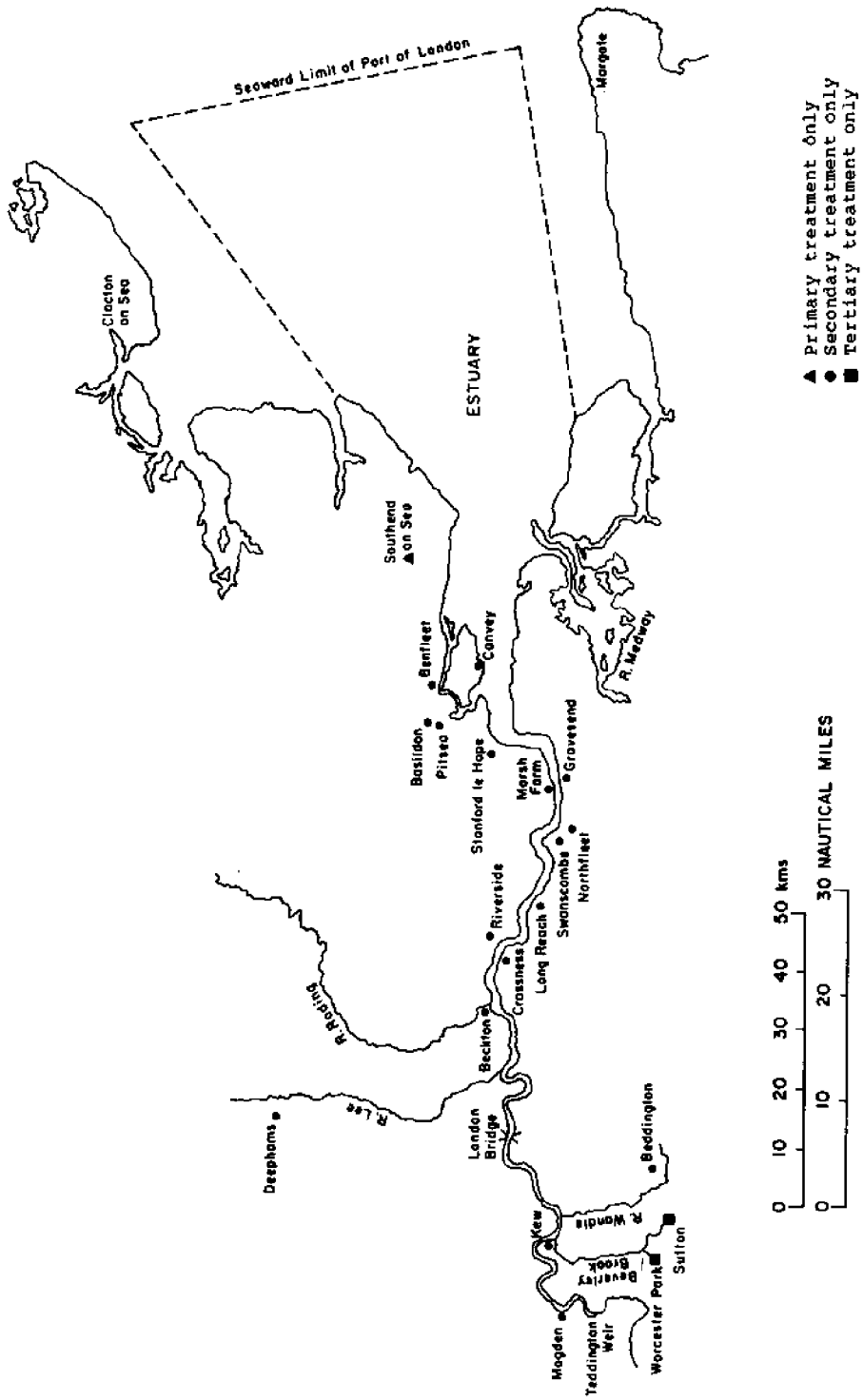


Figure D.15 (continued)

Sewage Treatment Works	Flow ($m^3\ d^{-1}$)	EOL (tonnes d^{-1})	Type of Treatment
Mogden	446,000	18.8	Secondary
Worcester Park	16,400	0.8	Tertiary
Sutton	7,950	0.4	Tertiary
Beddington	86,400	3.8	Secondary
Deephams	185,000	5.2	Secondary
Beckton	818,000	23.5	Secondary
Crossness	500,000	32.8	Secondary
Kew	34,400	0.8	Secondary
Riverside	86,800	14.8	Secondary
Long Reach	186,000	96.8	Secondary
Basildon	18,200	2.1	Secondary
Benfleet	6,140	0.5	Secondary
Canvey	4,500	1.1	Secondary
Marsh Farm	31,500	16.0	Primary + some Secondary
Pitsea	5,750	0.3	Secondary
Southend	32,700	15.0	Primary
Stanford-le-Hope	4,660	0.6	Secondary
Gravesend	7,200	3.1	80% Primary, 20% Secondary
Northfleet	7,450	1.1	Secondary
Swanscombe	1,000	0.2	Secondary

The Role of Predictive Models in Estuary Management

The drastic improvement in the Thames Estuary and in the life (both of fish and birds) it now supports has not been achieved by any uniform application of standards of treatment before discharge. Even now when the water quality is sufficiently good to allow the passage of migratory fish, there are wide variations in the treatment given to sewage and industrial effluents throughout the estuary. Since the clean-up of the estuary began, the main objective has been to gain the maximum environmental benefit from the commitment of available resources; thus, attempts have been made to understand the causative factors in the pollution of the estuary and to identify the sources where remedial action would be most cost-effective. The pursuit of this policy has led to the present situation where the primary water quality objective is met by a mixture of primary, secondary and in two cases, tertiary treatment at sewage plants. The extent of treatment of the major sewage inputs to the estuary is shown in Figure D.15, which illustrates that lower levels of treatment can be accepted seaward, where dilution and dispersion are faster, without affecting the primary index of water quality.

To be able to predict the payback in water quality improvement for a given investment in capital, a model has been derived and used for some years by the pollution control authorities. The model is now described to illustrate how the scientific knowledge of processes described in this book can be applied to a specific estuarine situation.

The mathematical model constructed by the Water Pollution Research Laboratory considers the estuary as a barrier-free tidal river whose boundaries are the tidal limit (at Teddington Weir) to the seaward limit of the Port of London Authority (PLA) - about 150 km. This distance is divided into 3.3 km (2 miles) segments, each assumed to be of uniform chemical composition throughout. The concentration of each segment is determined by:

1. The inputs of pollutants from all sources;
2. their subsequent movement by diffusion and advection from the freshwater flow from rivers, groundwater drainage, outfalls and sewers discharging directly to the river and tidal movement and exchange with seawater; and
3. the rate of decay or removal of the constituents by chemical, physical or biological means.

The model can predict the dissolved oxygen, ammonia, oxidized nitrogen concentrations and temperature of the river along its length, based on established decay characteristics for these components of water quality.

Despite the relative simplicity of the model (it is based on the assumption of an average exchange coefficient for the whole estuary, does not take account of changes in oxygen transfer rate from air to water in different wind/wave conditions and does not allow for the increased oxygen uptake of benthos suspended by high tides), agreement with observed concentrations is good, as can be seen in Figure D.16 which shows predicted and actual levels of DO in 1971. Having used an understanding of physical, chemical and biological processes to construct and calibrate this model, it may be used as an effective management tool. For instance, the effect of committing resources at two works (Beckton and Long Reach) may be assessed. Figure D.17 (a and b) shows the predicted DO of the river for varying effluent standards at these two works. It is seen that although significant improvements will be achieved by high levels of treatment at Beckton (to achieve a BOD of 10 mg/l and NH_3 of 2.5 mg/l in the final effluent) the water quality is not so sensitive to the degree of treatment at the Long Reach works; thus the additional expenditure required to raise the effluent quality from BOD 20 mg/l, NH_3 30 mg/l to a BOD of 10 mg/l and NH_3 of 10 mg/l or below would not be cost-effective.

Similar predictions can be made to assess the effect of further expansions in directly cooled power stations on the Thames. The effect of those presently operating is shown in Figure D.18, indicating that the river waters are up to 3°C warmer through its use as coolant, thereby reducing DO levels through an increase in the rate of degradation and a reduction in the rate of oxygen uptake from the atmosphere. In Figure D.18, the effect of installing three additional directly cooled power stations is also shown. Clearly, the use of direct cooling would negate much of the improvement in the river achieved by improved effluent treatment and would not allow present water quality standards to be maintained.

Conclusions on Effectiveness of Estuary Management

In conclusion, it is hoped that this section has illustrated that environmental improvements may be achieved on a significant scale by the deployment of

Figure D.16 Observed and predicted concentrations of dissolved oxygen (expressed as percent of saturation) in 1971 (Wood, 1973).

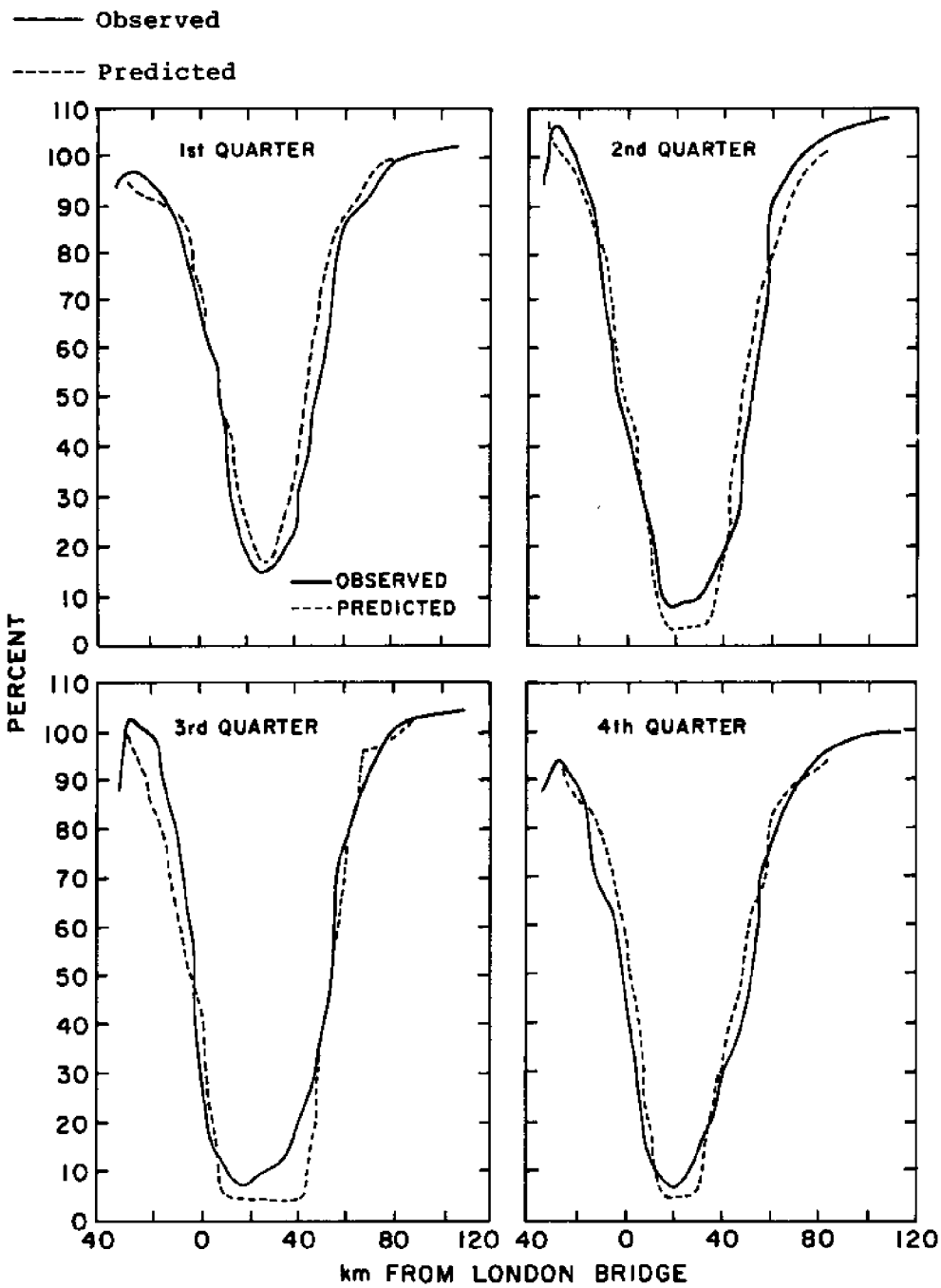


Figure D.17a The effect of varying Beckton sewage treatment works effluent quality, maintaining Long Reach sewage treatment works at 1974 quality of 150 mg/l BOD, 32 mg/l NH_3 (Wood, 1977).

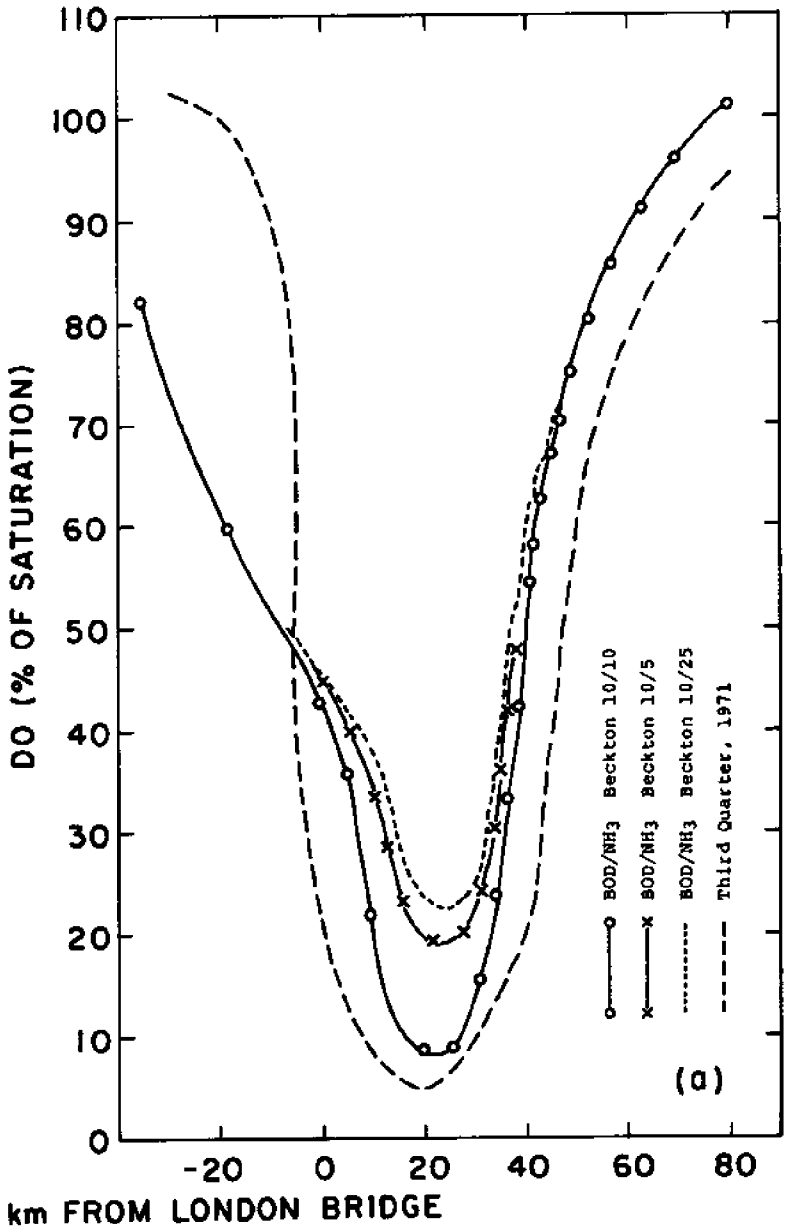


Figure D.17b The effect of varying Long Reach sewage treatment works effluent quality, maintaining Beckton sewage treatment works at BOD/NH_3 10/2.5 mg/l (Wood, 1977).

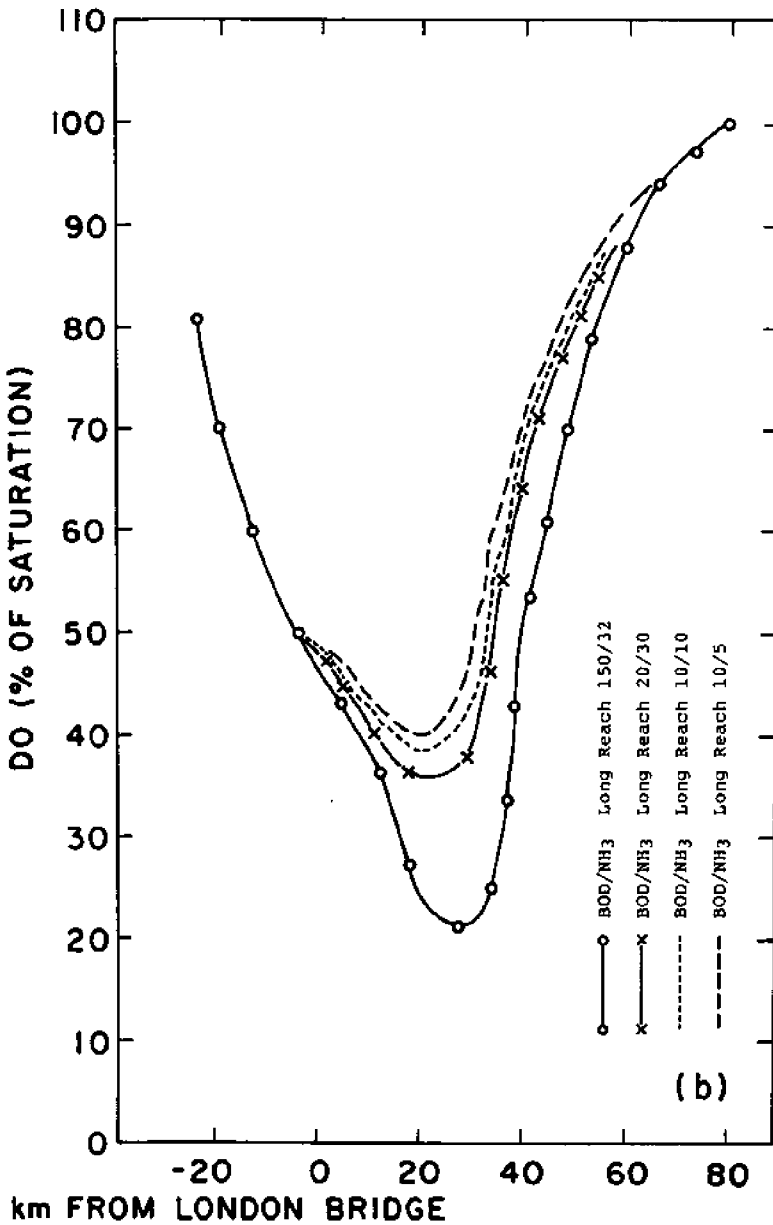
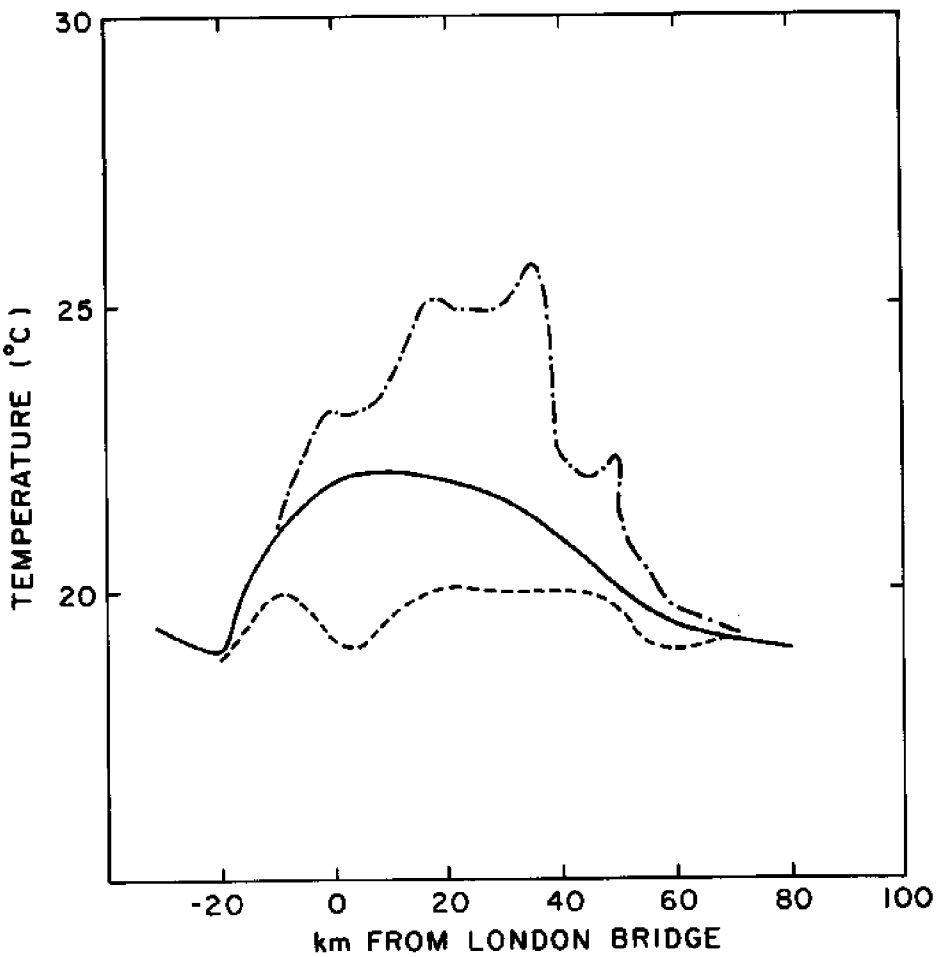


Figure D.18 Temperature profile of Thames Estuary, showing the effect of established directly cooled power stations, under third quarter, low flow conditions, 1980 (Wood, 1977).

——— Established power stations 1980
------- The effect of power stations removed
.—.—. The effect of three additional large directly-cooled power stations.



resources in a flexible and cost-effective manner without the necessity of applying uniform and inflexible standards throughout the water body. Account is taken through a model of the different dispersive capacities of an estuary at different points and the required water quality achieved by the deployment of resources at the most critical points.

The final test of the efficiency of the methods used to set standards for controlling an estuary is clearly measured by the results achieved. Figure D.19 shows the reduction in pollution load from sewage works over the past 25 years. The Thames Tideway in the early 1950s was an offensive, evil-smelling river over stretches of many kilometers during the summer months. No sulphide has been found for nine years now and there are 83 different species of fish; a live salmon was caught in November 1974, the first in almost 150 years.

SEWAGE SLUDGE DUMPING IN LIVERPOOL BAY

Introduction and Relative Inputs

Sewage sludge from several large sewage works is loaded at four points along the River Mersey and Manchester Ship Canal, taken out ito Liverpool Bay and dumped at the site shown in Figure D.20. Dumping started in the late 19th century and has increased to its current level of 1.8×10^6 tonnes per annum of sludge, whose composition is given in Table D.4. Comprehensive investigations of the effects of disposal started in 1970. Since that date, the area has been the most intensively studied off the United Kingdom and results have been published in a series of volumes (Department of the Environment, 1972, 1973, 1976 and 1979).

In addition to inputs from sewage sludge dumping, industrial wastes are dumped at the same locality and dredge spoils at another site, also shown in Figure D.20. Additionally, discharges from the rivers Mersey, Dee, and others give rise to large inputs of organic substances and metals to the same sea area, together with local coastal discharges. The inputs of a number of substances to the bay from all these sources have been estimated and presented in Table D.5 for 1976.

Figure D.19 Effective oxygen load (EOL tonnes/day) from the four major treatment works on the Thames Estuary.

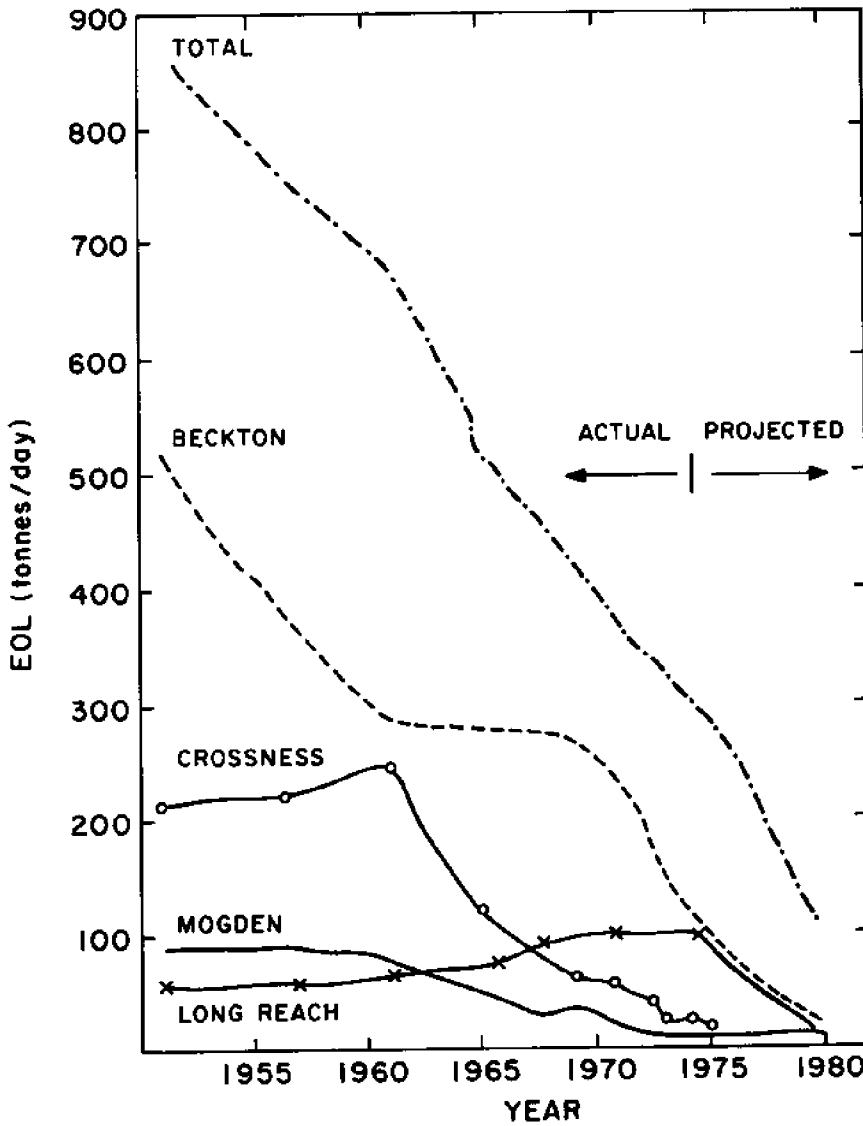


Table D.4 Composition of Sewage Sludge Dumped in Liverpool Bay (1977)

Component	Concentration		
	Range	Mean	
Total solids	4.1 - 10	4.4	% wet wt
Nitrogen	1.8 - 5.9	4.8	% dry solids
Phosphorus	0.8 - 3.6	1.6	% dry solids
Mercury	0.1 - 47	36	mg/kg dry solids
Cadmium	4 - 70	25	mg/kg dry solids
Chromium	34 - 1738	1300	mg/kg dry solids
Copper	93 - 1612	1500	mg/kg dry solids
Nickel	60 - 350	210	mg/kg dry solids
Lead	84 - 1400	300	mg/kg dry solids
Zinc	390 - 8000	3700	mg/kg dry solids
PCB and Organochlorine Pesticides	<1.6 - 5.8	4.0	mg/kg dry solids

Figure D.20 Liverpool Bay and the dumping grounds used for sewage sludge and dredged spoil.

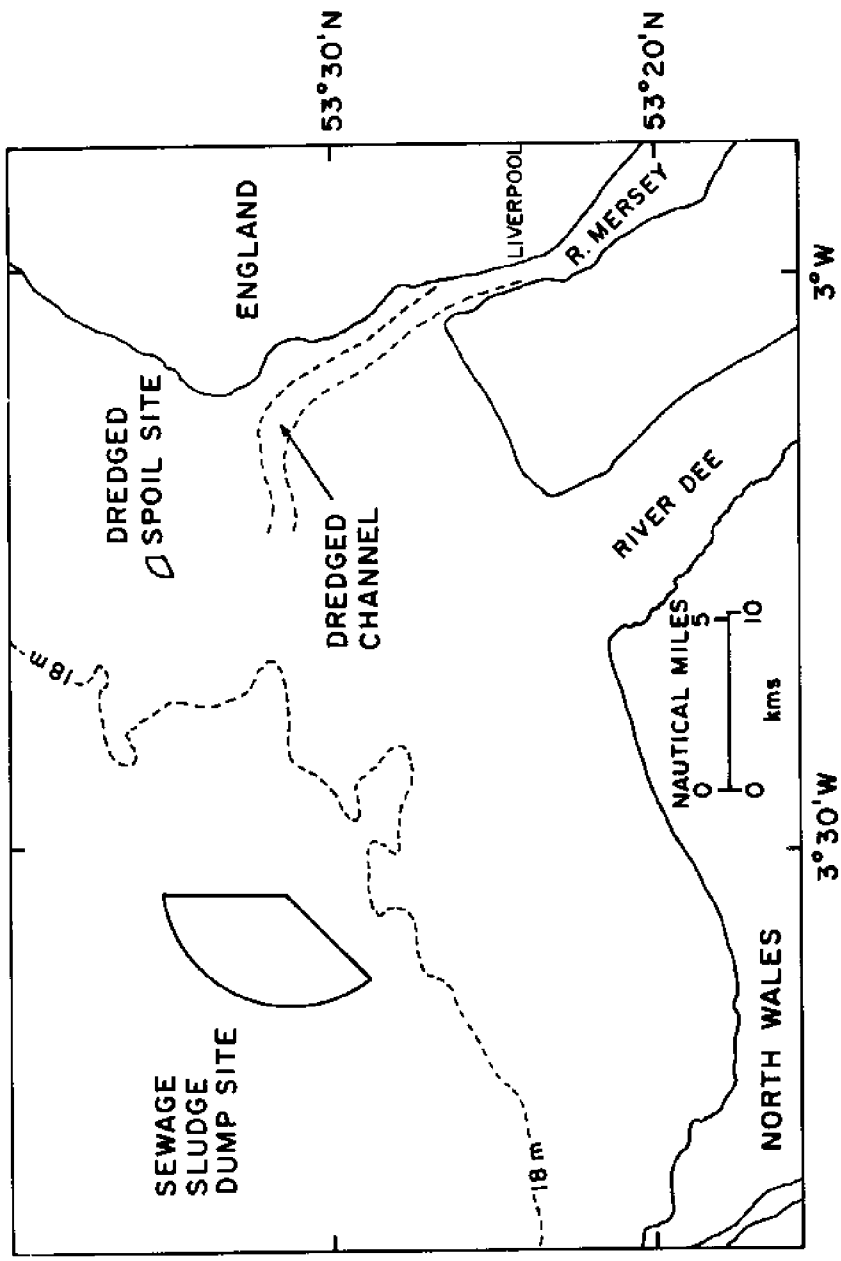


Table D.5 Inputs (kg/d) of Substances into Liverpool Bay, 1976 (MAFF and National Water Council, 1979)

Source	Cd	Cr	Cu	Ni	Pb	Zn	Hg	PCBs	BOD	Total N	Total P
River discharge	95	220	205	270	ND	425	19	ND	55,000	77,800	5,695
Direct sewage/ industrial discharge	45	418	146	203	129	4,089	ND	ND	270,000	17,000	7,800
Dumping of sewage sludge/industrial wastes	7	250	300	43	60	710	7	0.8	55,000	12,000	3,120
Dumping of dredged spoils	3.6	409	310	290	930	1,940	12	ND	ND	ND	ND

ND - not determined

The relative magnitude of point source inputs may give a misleading impression of the relative biological importance of these sources. This is likely to be especially the case for dredged spoils which, despite the large quantities dumped, may have little impact on the environment in the vicinity of the sludge dumping area due to the location and small area of the dredged spoil site and the likelihood that a major part of the metals in the spoil will be unavailable due to burial by further spoil dumping. Additionally, there is a tendency for metals in spoils to remain in particulate phase after dumping due to association with the mineral lattice, or presence as non-labile complexes with organic material, or as sulphides or other inorganic species of low solubility.

Of the other input sources to Liverpool Bay (rivers, discharges and dumping of wastes), the dominant source of each substance listed in Table D.5 varies. For instance, sewage sludge dumping represents the largest input of copper to the bay; river discharges the highest input of cadmium, mercury and nickel; and direct discharges of sewage and industrial effluents the largest quantities of chromium, lead and zinc.

Fate and Behavior of Sludge

The short-term behavior of dumped sludge has been determined by a series of dye and radio-tracer releases from the sludge dumping vessels (1,000 to 1,500 tonne capacity). Dilutions of 80:1 or more were encountered immediately after dumping, increasing to around 1,000:1 after an hour and 10,000:1 after 10 hours in relatively calm conditions.

During the flood tide, rapid vertical mixing was found to occur causing the sludge particles to reach the seabed within a few hours of dumping; indeed the radioactive tracer study showed that over half the sludge had settled initially at the seabed after one day.

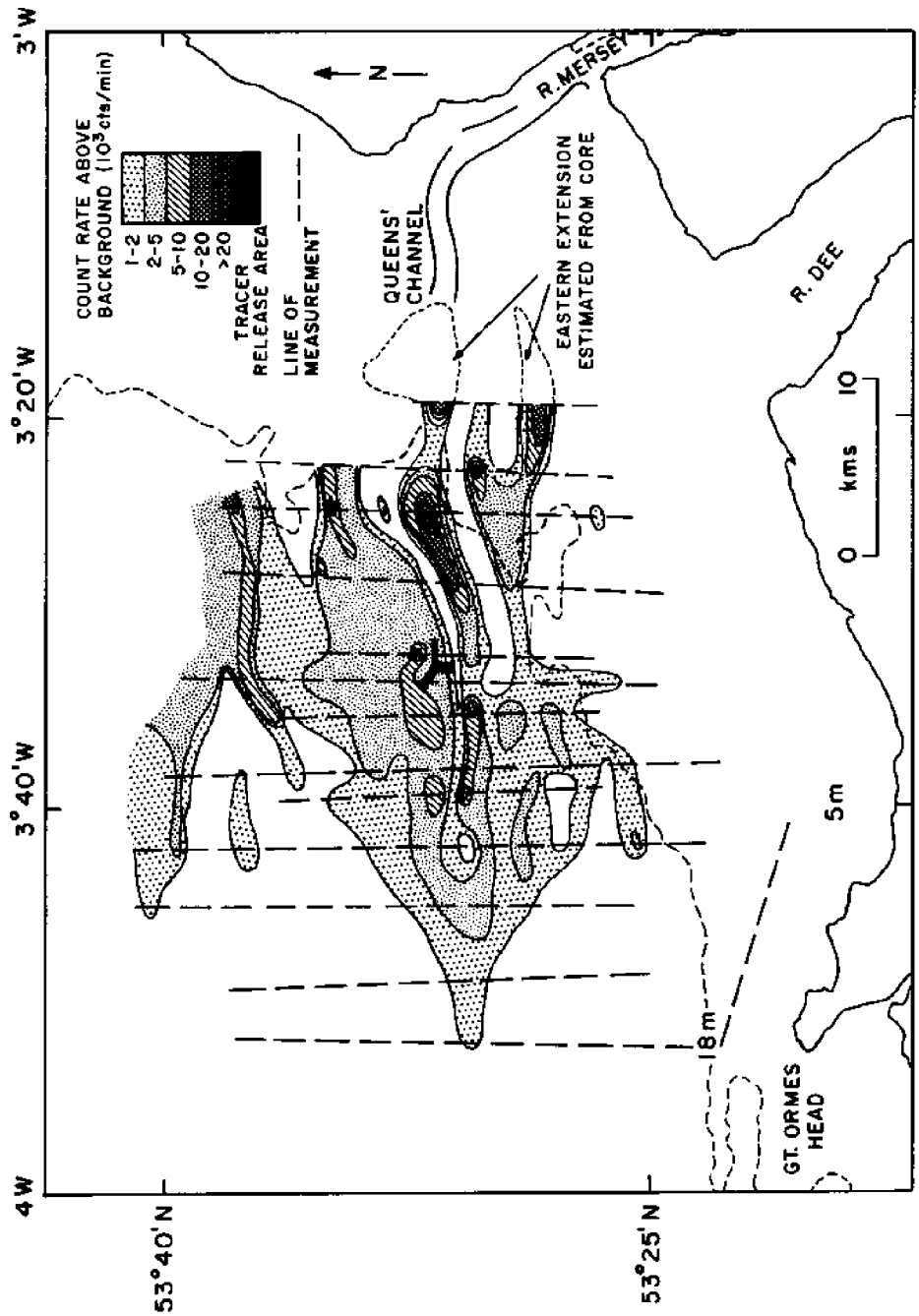
Subsequent movement of the sludge is determined by the tide- and wave-generated currents, the long-term distribution being similar to that of naturally occurring muds and silts in the area. Thus, although the tracer studies showed that initial settlement occurred over all the sediment types present near the dumping ground, elsewhere the maximum sludge concentrations coincided with muddy areas. Studies of the long-term water movements near the seabed indicated a strong inshore residual movement to the muddy areas near the mouth of the Mersey and the adjacent coastlines. The distribution of the radio-tracer in the sediment 2 1/2 months after the last release of labeled sludge is shown in Figure D.21, showing muddy areas to be zones of settlement of sludge particles.

Components of the sludge which are either soluble or capable of being transported in suspension in surface waters tend to move with the residual surface water drifts, which are north to northeast from the dumping site and north through the eastern Irish Sea.

Results of Monitoring Studies

Water quality: The effects of dumping on water quality at the dumping ground have been monitored for several years, both routinely from dumping vessels and in special surveys. The levels of DO have been found to be near saturation at all times of the year in surface waters. In periods of calm summer weather, however, where water is thermally stratified, some local reduction in DO levels below saturation has been observed in bottom waters, leading to differences between surface and bottom values of up to 20 percent.

Figure D.21 Distribution of radioactive tracer in sediments 2 1/2 months after date of last tracer release (DOE London, 1972).



The distribution of nutrients in the bay is dominated by the outflow of the River Mersey (see Table D.5), and effects due to dumping are difficult to detect. Nevertheless, there is some limited enrichment of the offshore area from dumping, but the concentrations of nutrients encountered are low relative to the inshore zone.

A limited number of measurements have also been made on metals in suspension and solution in the vicinity of the dumping area. These revealed slight elevation of metals (Ni, Cu, Zn) in the suspended particulate phase in the region of the dumping area, but the increase was less than two times background level and restricted to an area smaller than the dumping area.

Effects on sediment: Several studies of the physical and chemical characteristics of the sediment have been carried out. These have demonstrated that, in spite of the generally good dispersive characteristics of the bay, some localized accumulation of dumped material in the sediments occurs. Concentrations are also elevated in the zone between the dumping site and the Mersey estuary to which both dumping and estuarine discharges probably contribute.

The distribution of dumped material is best shown in the fine fraction of the sediments: organic carbon and some metal concentrations in 1978 are shown in Figures D.22 and D.23, respectively. These figures demonstrate the local elevations in fine sediment concentrations at the dumping site, high concentrations near the mouth of the Mersey estuary and an elevated zone connecting the two. Another influence apparent in the case of some metals is that of dredged spoil dumping (Figure D.23).

Other surveys have shown different degrees of contamination by sludge-derived material and this, together with the presence of mobile sand bodies in the general region of the dumping ground suggests that accumulation near the dump site may only be temporary. Further inshore, however, the stability of the muddy sands over longer periods makes significant accumulation of dumped material more likely.

Effects on benthos: Intensive studies of the benthos since 1971 have allowed the annual changes to be monitored. Until 1977 there was no evidence of any anomalous characteristics in the dumping ground fauna. From 1977-9, however, an area roughly corresponding to the organically enriched area of sediments near the

Figure D.22 Concentration of organic carbon (%) in 1978 in superficial sediments (<90 µm fraction)

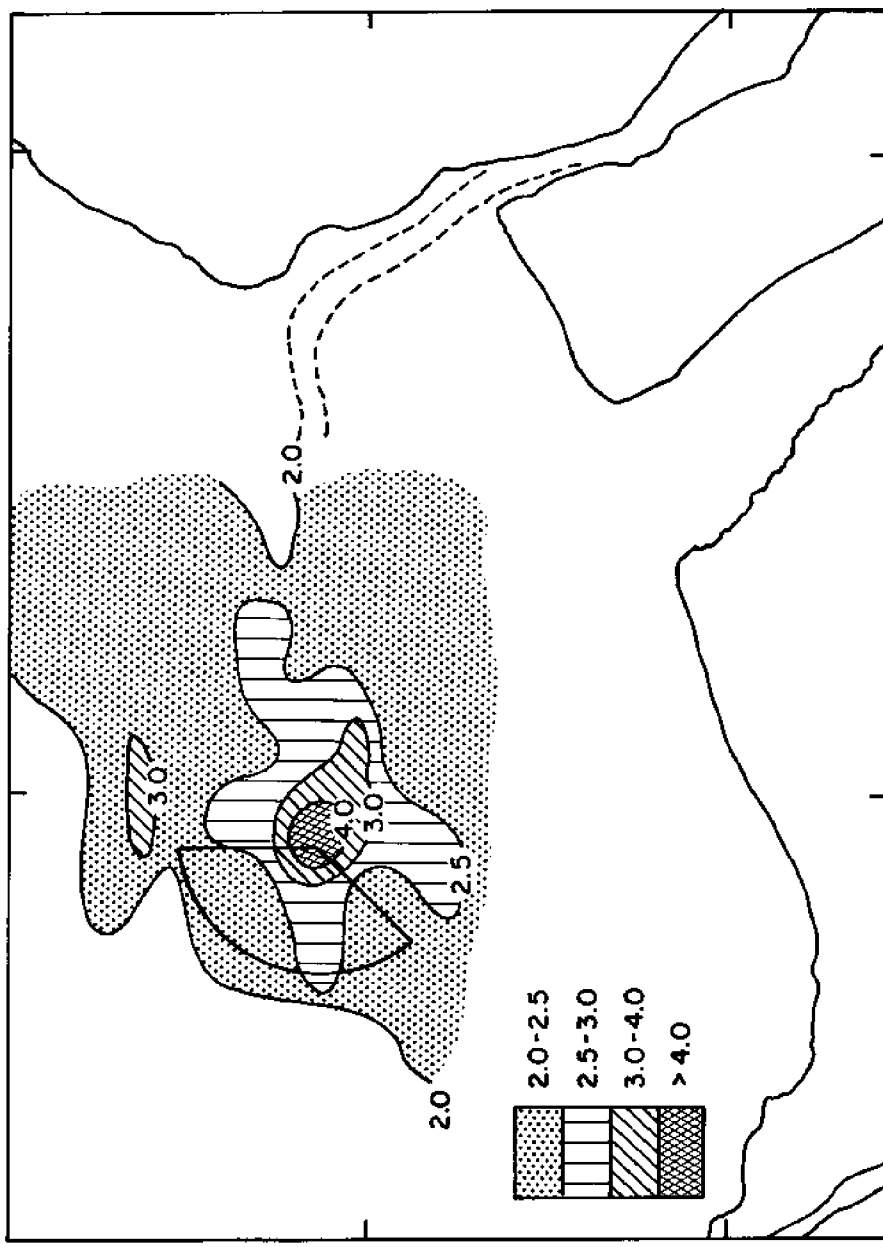


Figure D.23a Concentrations of mercury (mg/kg) in superficial sediments in 1978 (<90 µm fraction)

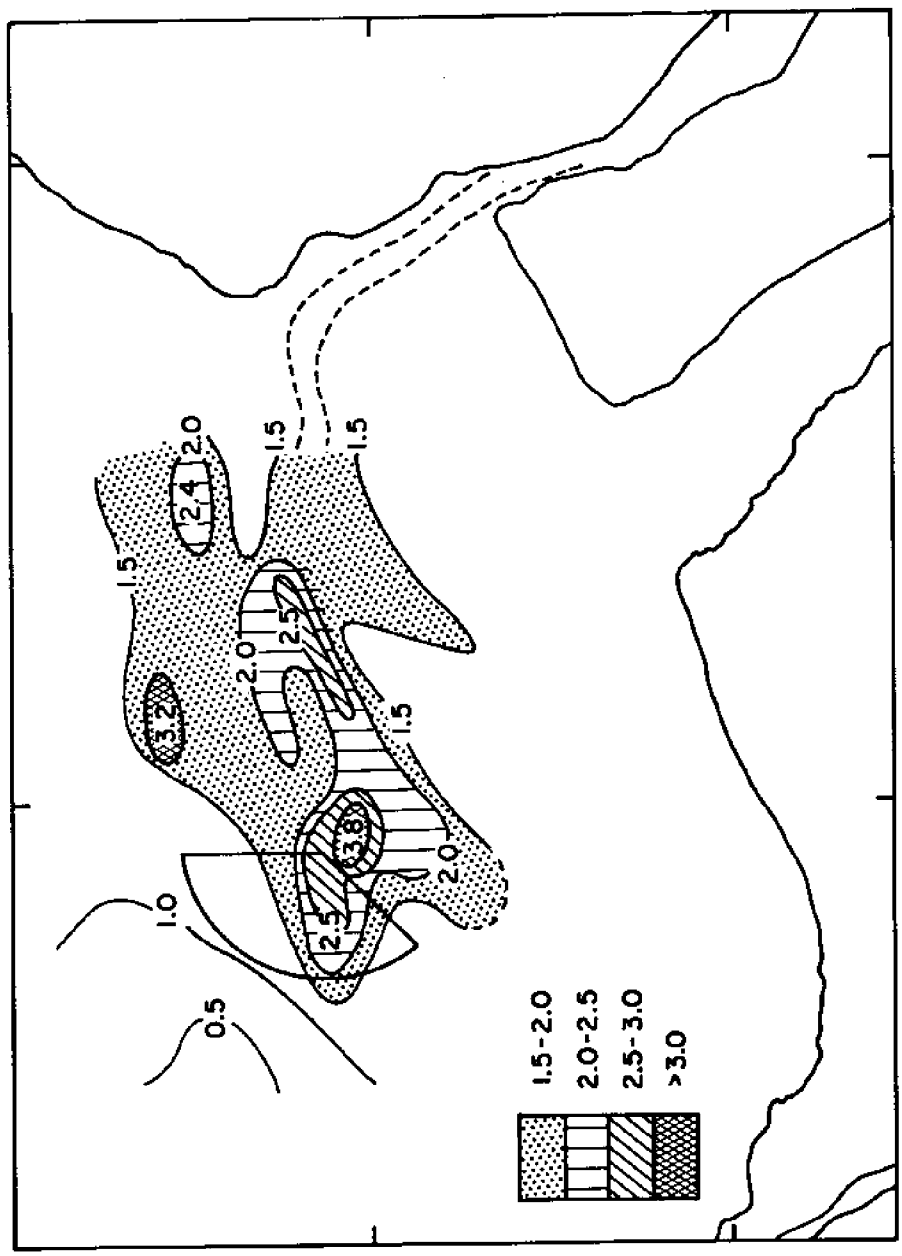


Figure D.23b Concentrations of zinc (mg/kg) in superficial sediments in 1978 (<90 µm fraction)

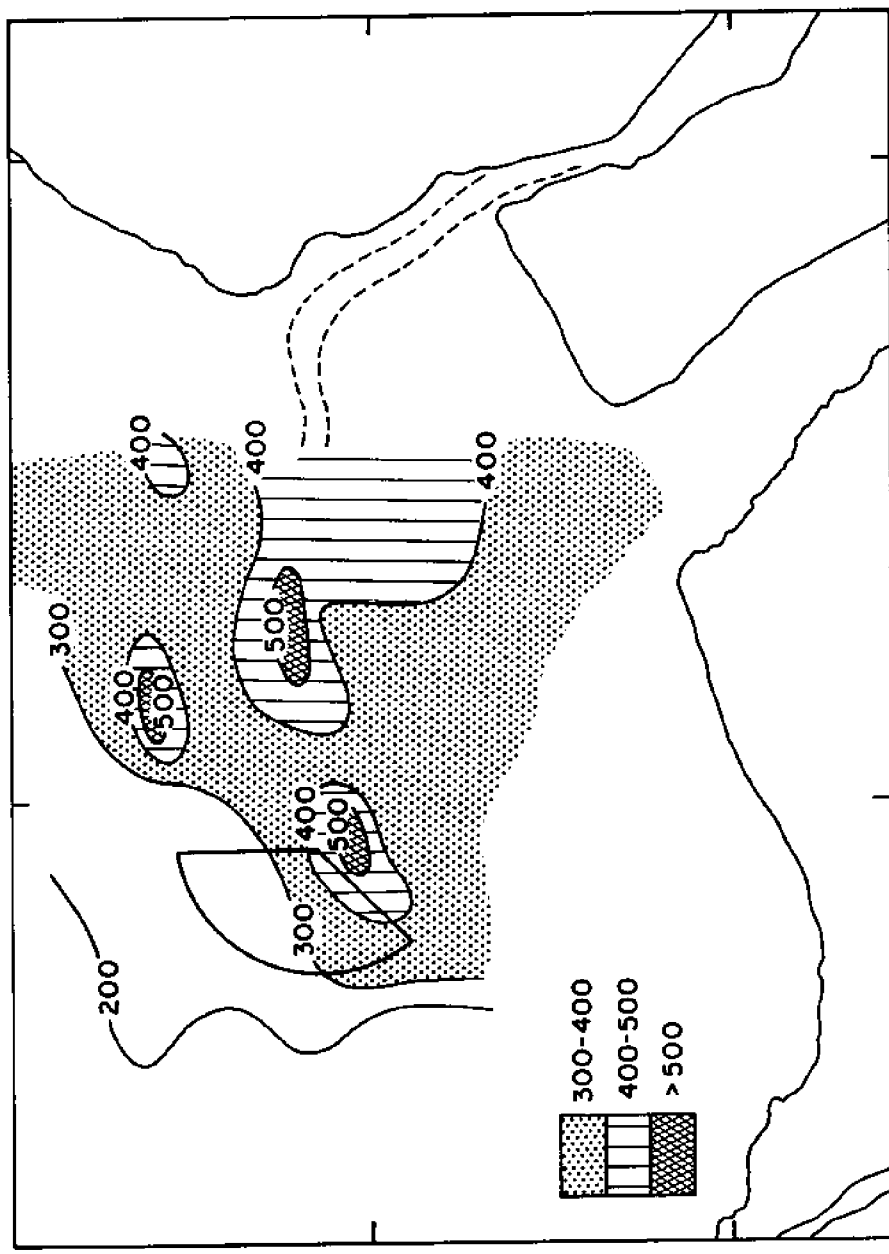


Figure D.23c Concentrations of copper (mg/kg) in superficial sediments in 1978 (<90 µm fraction).

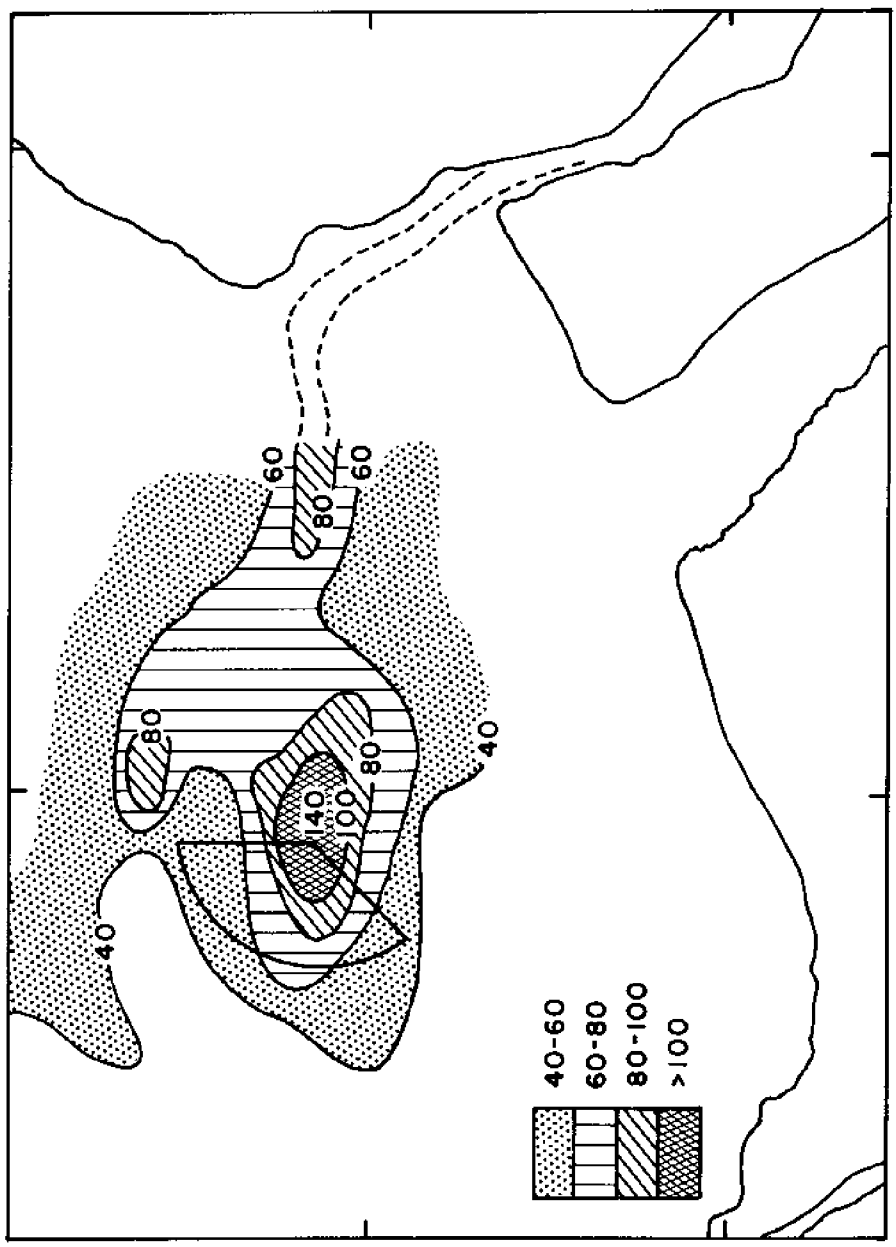
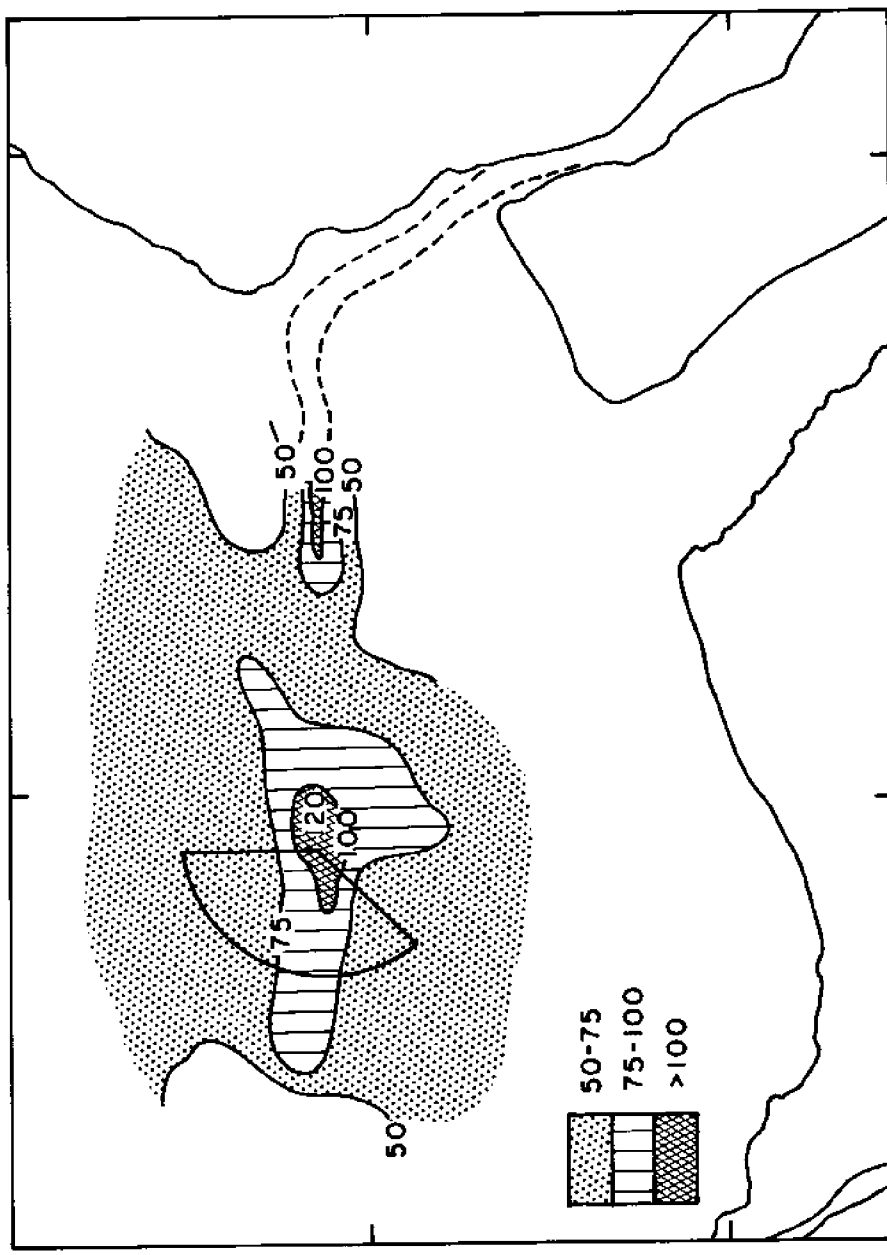


Figure D.23d Concentrations of chromium (mg/kg) in superficial sediments in 1978 (<90 µm fraction)



dumping site exhibited a large population of the tubificid worm Pectinaria koreni. The success of this organism led to increased faunal density but lowered diversity in the affected zone (about 20 km^2), and it was postulated that this represented a response to the organic input of the sludge. In 1980, however, the trend was reversed and the faunal diversity returned to the level of earlier years. Present evidence thus suggests that the sludge dumping may have a minor effect on the benthos, but at present rates of disposal such effects are quite capable of being reversed or overshadowed by changes induced by natural events.

Effects on fish and shellfish quality: The chemical quality of all commercial species of fish and shellfish taken from Liverpool Bay has been extensively monitored since 1970 for metals (cadmium, zinc, mercury, lead and copper), organochlorine pesticides and PCBs. Of the chlorinated organic substances, PCBs and DDT metabolites were present in significantly higher concentrations in several species of fish and shellfish, particularly in fish liver tissue.

Of the metals, mercury concentrations in fish were significantly elevated during the early 1970s but have since fallen to concentrations more typical of inshore waters generally. The reduction in mercury concentrations (from an average concentration of 0.55 mg/kg in 1970 to 0.26 mg/kg in 1978) has been achieved largely as a result of increased control of industrial discharges to the River Mersey, since sewage sludge inputs increased slightly over the same period (See Table D.6 for estimate of relative inputs in 1976). Since 1977, however, sewage sludge concentrations of mercury have been considerably reduced which is likely to lead to further reductions in fish mercury concentrations.

There were no general elevations in the concentration of metals other than mercury although some species of shellfish and fish did exhibit higher than expected concentrations in some samples.

Management Considerations

A large deployment of resources in monitoring the effects of sludge disposal in Liverpool Bay has not detected any major adverse effects attributable to sludge dumping at the present rate of disposal of 1.8×10^6 tonnes per annum. There are, however, indications that the assimilative capacity of the area

Table D.6 Composition of Sewage Sludge and Relative Inputs from Several Sources to the Firth of Clyde (1976)

Component	Sludge composition+		Annual input from dumping (tonnes)	% Contribution to total inputs to Firth of Clyde*: Sludge dumping	Rivers	Sewage & direct industrial discharges	Atmospheric Deposition
Wet sludge	100%		1,400,000				
Solids	5%		70,000	33	55	12	ND
Nitrogen	4.3	% dry	3,000	11	33	56	ND
Phosphorus	0.8	weight	560	37	41	22	ND
Mercury	3.8		0.3	ND	ND	ND	ND
Cadmium	13.1		1	16	45	16	22
Chromium	1,720	all	120	59	13.5	25.5	2
Copper	680	mg/kg	48	30	26	23	20
Nickel	87	dry	6	13	52	17	19
Lead	610	weight	43	23	41	12.6	23
Zinc	1,800		126	18	34	7.4	42
PCBs	4.2		0.3	ND	ND	ND	ND

ND = no data

* Estimates from National Water Council, 1979

+ Department of the Environment, 1977

may be fully utilized under special conditions. Occasional depletion of DO in bottom waters has been observed under special hydrographic conditions, concentrations of metals and organic substances are elevated near the dumping site, and effects on benthos are suggested. Additionally, the concentrations of some contaminants (particularly mercury) have been elevated in commercially exploited species of fish and shellfish from the Bay. Inputs of mercury have been reduced both from the industrial discharges to the estuary and (since 1977) from dumping, and the effectiveness of this control at source has been demonstrated by falling mercury concentrations in fish. It is thus expected that dumping in Liverpool Bay will continue to provide a substantial disposal outlet for sewage sludge in that area of the United Kingdom. The sludge will, however, be subject to control of its quality, and monitoring will continue to determine if the nature of effects changes with time. Caution would have to be exercised, however, in considering possible increases in dumping at the present site.

DUMPING OF SEWAGE SLUDGE IN THE FIRTH OF CLYDE

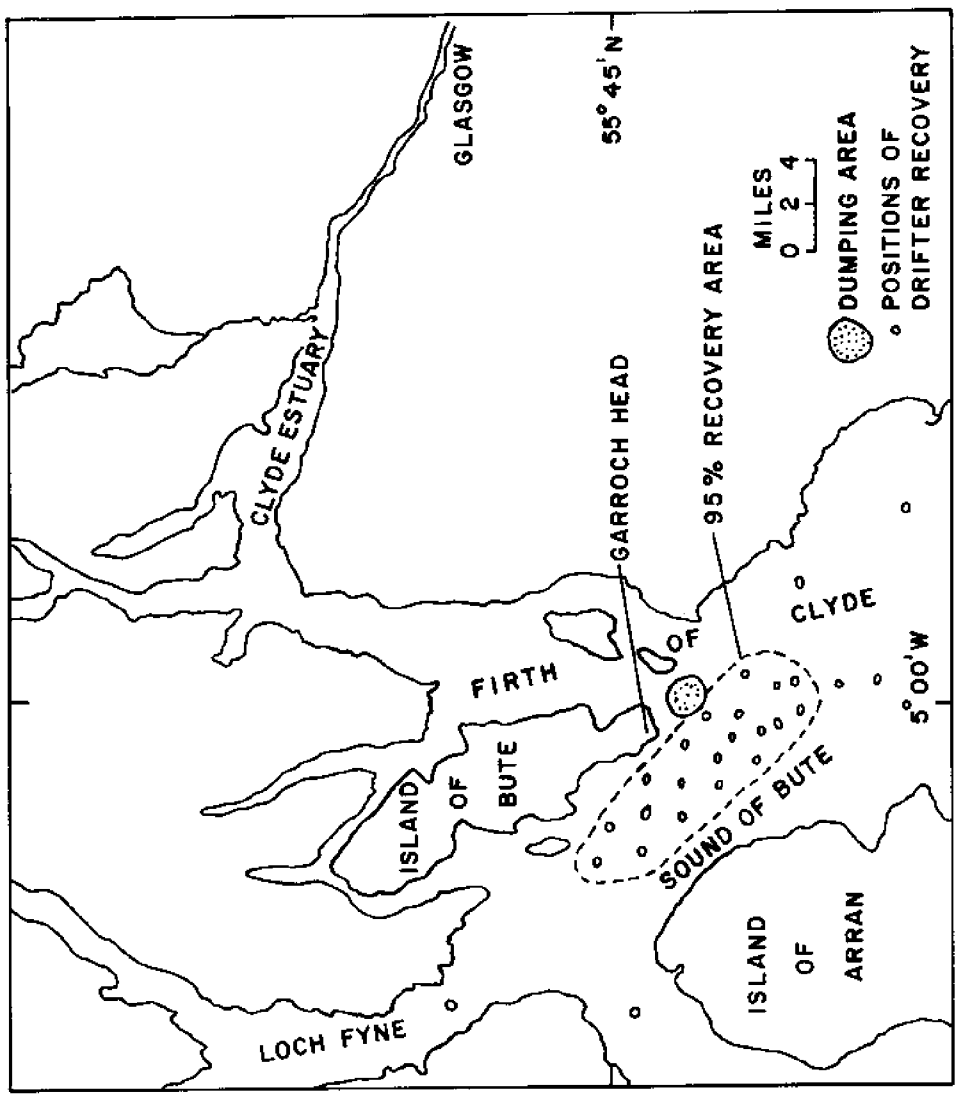
Introduction

Sewage sludge has been dumped in the Firth of Clyde from Glasgow sewage treatment plants since 1904 (Figure D.24). Since that year, the quantity of sludge dumped has increased, and at the present time approximately 1,400,000 tonnes of sludge are dumped each year, most of it primary or biological sludge without digestion.

The average composition of the sludge in 1976 is given in Table D.6, together with the total amounts of metals and nutrients dumped. Estimates have also been made of the inputs of these substances to the Clyde Estuary from other sources (river and effluent discharges, atmospheric deposition).

Viewed from the position of the whole Clyde sea area, sludge dumping can be seen to provide up to 59 percent of the total inputs of some metals and 33 percent of the total solids input. In local terms however, the relative importance of sludge dumping may be higher still, due to the confined nature of the receiving waters and the resulting limited dispersion of particulate material, as will be seen in subsequent sections.

Figure D.24 Garroch Head dumping area, and release and recovery points of seabed drifters (Mackay and Topping, 1970).



Distribution and Fate of Dumped Sludge

The sewage sludge is dumped in depths of 100 m through the bottom of vessels of 1,800 to 3,000 tonnes carrying capacity while stationary or moving slowly in the dumping area. Thus initial dilutions are low and subsequent dilution and dispersion is achieved through tidal currents, diffusion, and differential settling rates through the water column.

The rate of settlement of sludge particulates has been measured (MacKay and Topping, 1970) and 75 to 80 percent of the sludge found to settle at rates of 0.3 cm/s, allowing the bulk of a load of dumped sludge to settle to the bottom in around 9 hours. Maximum tidal strengths are below 0.5 knot (<25 cm/s), so little lateral movement of the sludge from the dumping site is to be expected. Residual movements of near-bed currents were estimated using seabed drifters which indicated some movement along a northeast-southwest axis into the Sound of Bute (Figure D.24); movement toward adjacent tourist resorts was not found to be an important dispersal route.

Despite the relatively slow rate of dispersion, studies have not shown DO concentrations in the water volume to fall below about 95 percent of saturation.

With such a relatively simple hydrography and uniformity of the natural sediments around the dumping area, it is to be expected that the physical and chemical effects of dumping at the seabed will be readily detected. This has proved to be the case in surveys carried out from 1969 to 1972 and reported by MacKay et al. (1972) and Halcrow et al. (1973, 1974), which have demonstrated that the natural sediments (mainly silts and reworked boulder clays) which are smooth grey muds away from the dumping area are modified within about 2 miles (4 km) of the dumping area to a black, fibrous, smelly substrate.

Analysis of the sediments showed that the central zone was characterized by high concentrations of organic carbon, many metals, and PCBs. Distributions of organic carbon, copper, mercury, zinc, lead and PCBs in the sediment are shown in Figure D.25 which shows that the main area of enrichment is restricted to an area of about 15 km^2 centered on the dumping ground. The range of concentrations found in the disposal area for those and other metals, compared with those in sediments from a control area in the Clyde with sediments of similar type, are shown in Table D.7 and confirms the degree of enrichment by dumping. Although

concentrations of metals were significantly increased at the dumping ground, the greatest increase relative to unpolluted sediments were found with PCBs. These exhibited an increase in concentration of about 300-fold from the outer part of the survey area to the center of the dumping area (Figure D.25). The copper, zinc, chromium, lead and PCB concentrations in the sediment were all strongly correlated with the organic carbon content while nickel and cadmium were not, suggesting that a higher proportion of the former metals remains with the sludge particles after dumping.

Table D.7 Comparison of the Composition of Sediments from the Disposal Area with a Control Area in the Clyde (Halcrow et al., 1973)

	Organic carbon (%)	Cd	Hg	Cr	Cu	Ni	Pb	Zn	PCB* (ng/g)
					(mg/kg)				
Disposal area	6.66-7.93	4-8	2	87-175	250-300	58-87	269-403	432-681	80-2,890
Control (Clyde)	0.4-1.7	1-2	0.04-0.15	10-65	9-20	14-50	24-67	60-130	10

Data are based on 10 samples except for Hg with 1 sample from disposal area, 2 from control area.

*from Halcrow et al., 1974.

The variation in metal concentrations in the sediment at different depths was also determined by coring. At the center of the dumping area, significant differences were found between surface levels and those at depth. Since the elevated levels of metal persisted to a depth of 40 cm, it has been suggested that this may represent the extent of accumulation since dumping started. PCB concentrations were elevated to a depth of less than 20 cm, suggesting that contamination of the sludge with PCBs had started about 20 years ago.

Effects on the Biota

The most noticeable biological effects of dumping are to be found in the sediments of the dumping area

Figure D.25a Distribution of organic carbon in superficial sediments

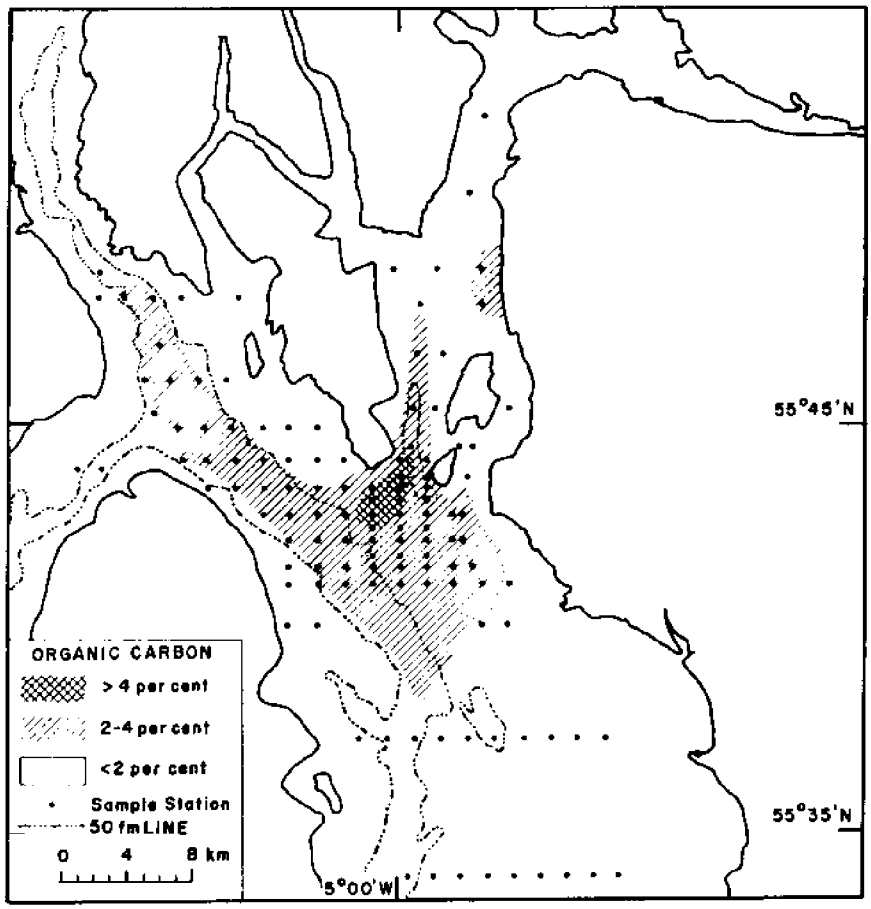


Figure D.25b Distribution of mercury in superficial sediments

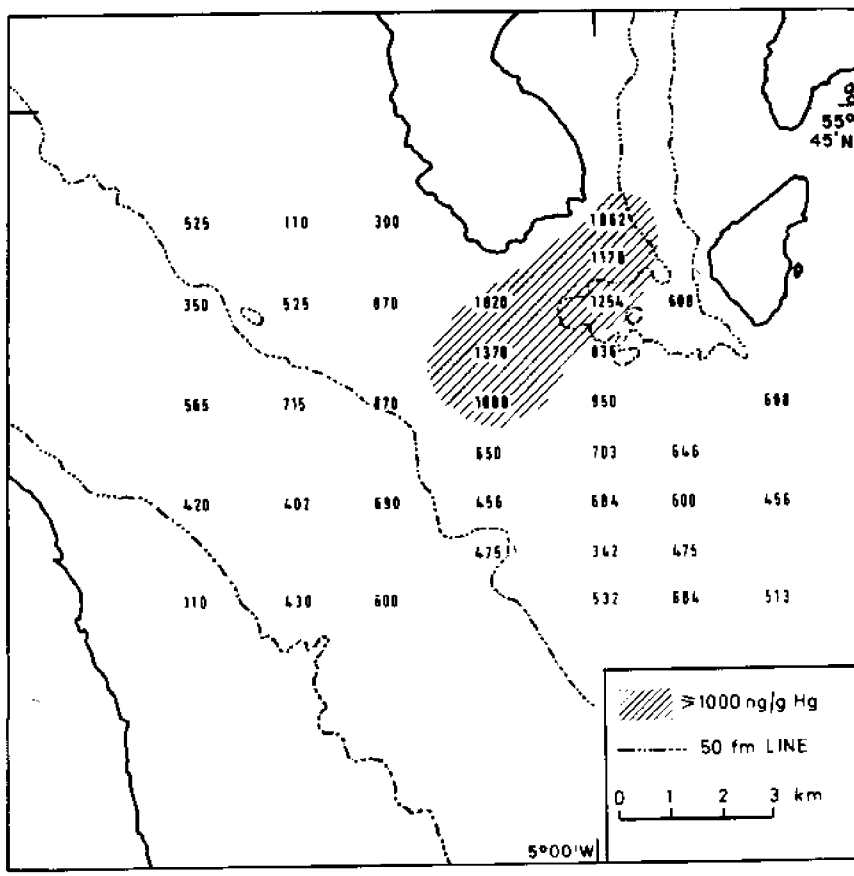


Figure D.25c Distribution of zinc in superficial sediments

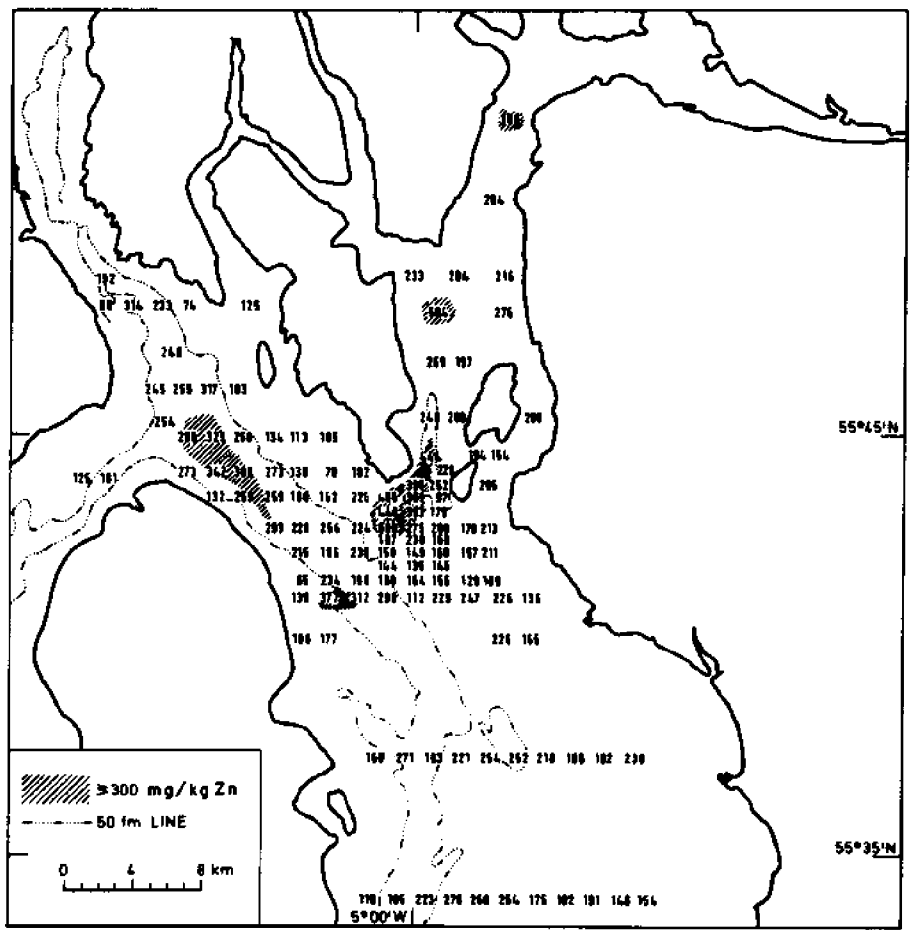


Figure D.25d Distribution of copper in superficial sediments

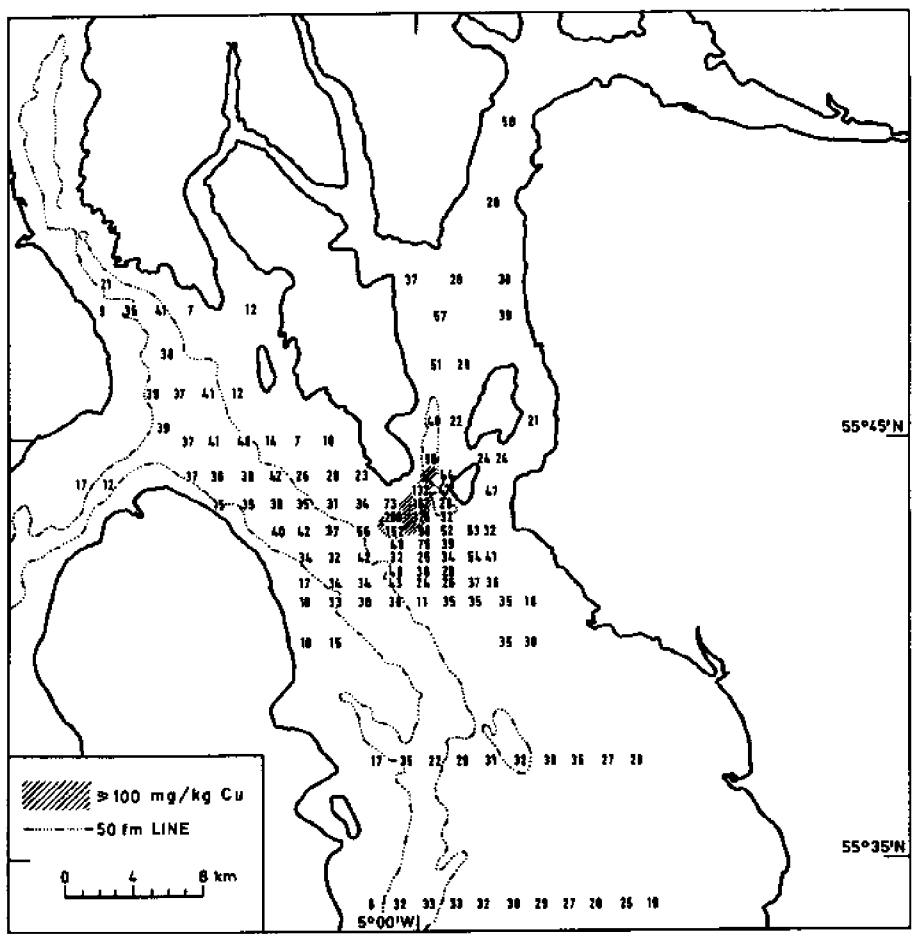


Figure D.25e Distribution of lead in superficial sediments

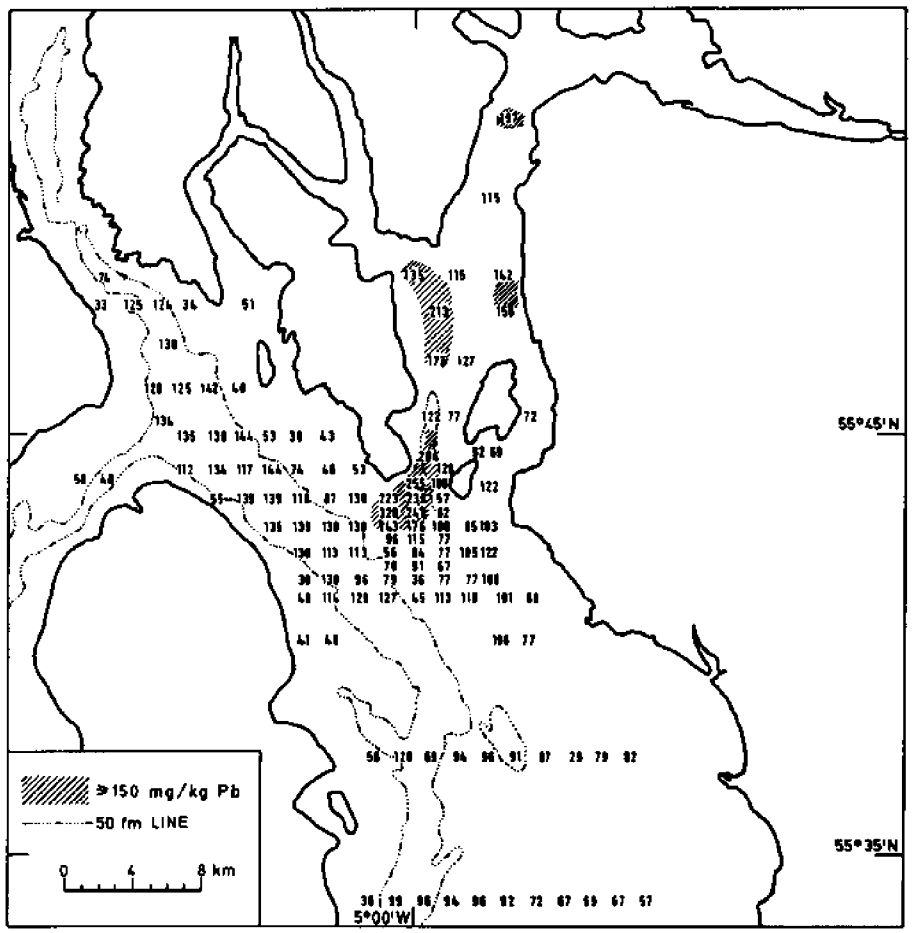
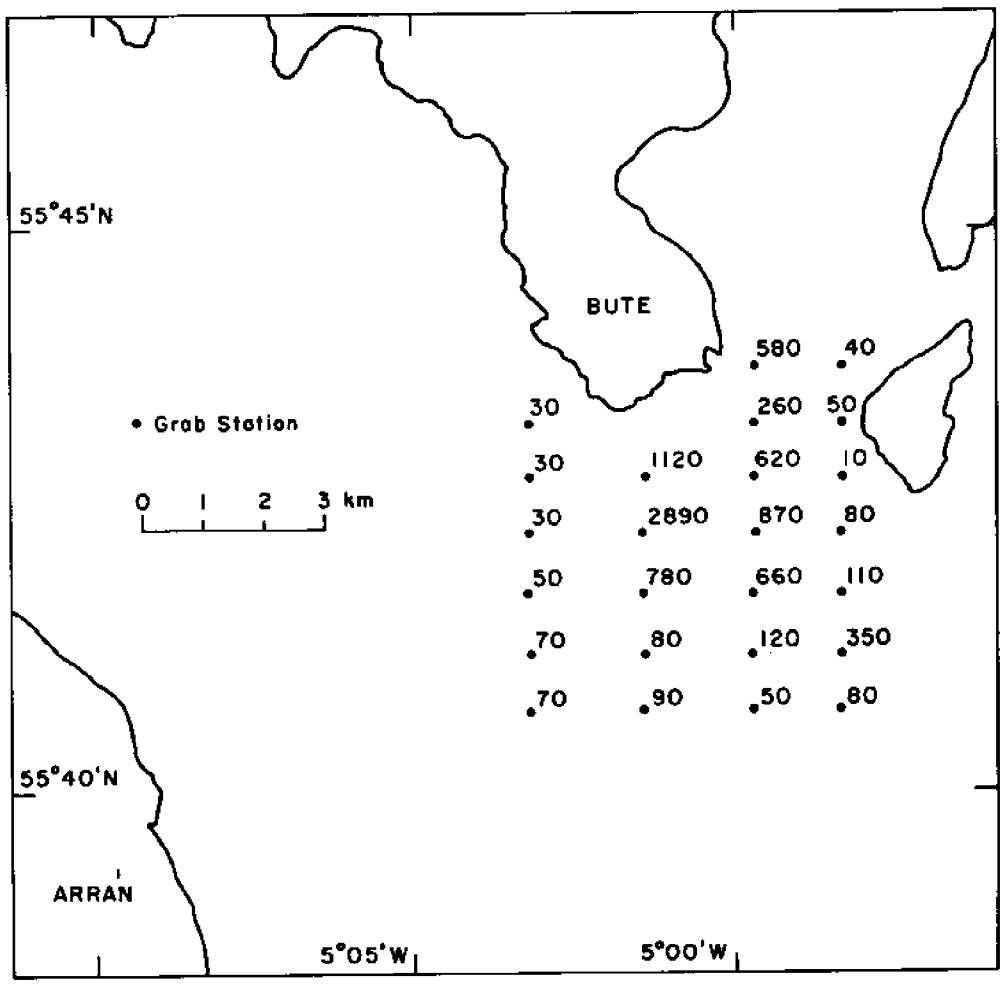


Figure D.25f Distribution of PCBs in superficial sediments (Halcrow et al., 1973 and 1974).



(Halcrow et al., 1973; MacKay et al., 1970 and 1972). The changes in sediment composition described in the preceding paragraphs have led to major changes in the assemblage of animals living in or on the sediment. The benthic infauna at the center of the disposal area are dominated by polychaetes typical of an opportunistic fauna, capable of utilizing the organic enrichment in the presence of substances such as PCBs at concentrations which may be toxic to many species of crustaceans and molluscs. This simple faunal community at the center of the dumping ground gives way to the "natural" fauna (comprised of a molluscan/echinoderm community) about 4 km from the center of the disposal area. This main area of organic enrichment (about 15 km^2) does not support the commercially exploited burrowing prawn Nephrops norvegicus found away from the dump site nor other infauna such as crustaceans, molluscs, or echinoderms. However, some species of epibenthos (e.g., Pandalus and Crangon) were quite abundant on the outer parts of the dumping area. Although there is a marked increase in the benthic biomass in the organic-rich sediments, this is of questionable value, since the typical infaunal species of the dumping area have not been found in the gut of fish taken from the area, whereas Abra and Nucula, which are common away from the dump site, are consumed by plaice and whiting. Large gadoids have however been observed to feed on the large pelagic crustaceans in the water column at the center of the dumping ground (McIntyre and Johnston, 1975). Thus it is only the central 15 km^2 or so of the dumping area that, although quite productive in terms of the benthic fauna biomass, is unlikely to contribute significantly to the commercial fishery - either in terms of supporting directly exploited species or in providing a source of food for demersal fish. The extent of loss to the commercial Nephrops fisheries has been estimated at around 3 tonnes of Nephrops each year, or about 0.15 percent of the landings from the Firth of Clyde fishery.

The benthic macrofauna at the dumping ground have been compared with that in less polluted environments elsewhere and related to organic inputs by McIntyre (1976). It was estimated that the input of carbon to the sediment at the dumping ground was about 55 times the natural rate from phytoplankton, and that this large input was at least partly responsible for the elimination of the meiofauna and the gross simplification of the macrofauna community. By contrast, an adjoining bay receiving sewage and industrial discharges was found to receive about six times the natural input of carbon, and to have a more abundant macrofauna, without the species reductions

associated with the higher carbon inputs of the sludge dumping area. Some reductions in the number of species of the meiofauna were noted, however, and McIntyre suggested that the input level of six times "natural" was just beyond the optimum for overall community health.

In addition to the effects on the biota themselves, the effects on the chemical quality of the biota have been investigated. It has been found that benthic animals near the center of the deposit area, including species known to be eaten by commercial species of fish, contain higher levels of metals and PCBs than in adjoining areas not affected by sewage sludge. The most noticeable increase was found in the whelk Buccinum undatum, which exhibited increased tissue levels of zinc, cadmium, nickel, lead, and PCBs. Elevated levels of metals were not, however, found in commercially exploited fish taken from the area, although accumulation of PCBs was found to have occurred in a species of fish closely related to species of commercial importance.

Management Considerations

Surveys have shown that the effects on the receiving area of depositing 1.4×10^6 tonnes per annum of sewage sludge are localized, and there are no indications of significant effects being located outside of an area of approximately 15 km^2. There are no amenity nor public health concerns over disposal in this area, and the only effect that is quantifiable is the local reduction in the productivity of a commercial fishery due to the unsuitability of the dumping - modified substrate for Nephrops and a possible reduction in benthic species of interest to commercial fish. The evidence for substantial accumulations of PCBs in the sediments and biota in 1972 was of potential concern, and control measures were taken by the Clyde River Purification Board (Waddington et al., 1973) to reduce substantially the input of PCBs via the sludge -- over and above the reduction that was already occurring due to restrictions on the use of PCBs introduced by the manufacturers.

To offset the environmental "costs" identified is the fact that the economic cost of alternative methods of sewage disposal in Glasgow and the surrounding areas would be very large. To the economic cost can also be added an environmental cost, since areas of land in the congested city are not readily available, necessitating

higher energy cost from having to use alternatives involving transport or incineration. Thus until an alternative method of disposal is identified that has fewer environmental costs and is of similar economic cost, ocean disposal is likely to remain the preferred option.

SUMMARY AND CONCLUSIONS

The studies of areas used for the disposal of sewage sludge by dumping from vessels reveal that the assimilative capacity of different areas varies widely, depending particularly on the dispersive potential from tide- or wave-generated currents and the water depth in the dumping area.

Of the three areas described, the Firth of Clyde dumping area receives the smallest quantity of sludge, yet the effects on the area are the most marked, because dumping takes place in an area of low tidal currents.

The Thames Estuary dumping area, however, receives between three and four times the Clyde input but with less measurable effect, due to rapid dispersion associated with fast tidal currents and shallow depths.

It can be shown from the experience described in this appendix that the general behavior and effects of sludge dumping can be predicted from a knowledge of the characteristics of the receiving area, particularly the strength and direction of tide- and wave-generated currents, residual water movements, sediment distribution, and sediment transport processes. In most cases, however, it is also essential to monitor the effects of disposal to increase understanding of the dispersive processes at work, to determine the actual extent of contamination resulting from dumping, and to guard against the possibility of unforeseen effects.

Monitoring should establish the extent of chemical and physical effects in the water column and in the sediments, establish whether the biota in the receiving area are affected, and investigate the possible contamination of the area's fisheries. In interpreting the results of monitoring, it is desirable to know the relative importance of all sources of potential pollutants in the area.

The nature and degree of effects of dumping vary with the amount of waste and area used. No adverse

direct effects outside the immediate dumping locality have been observed on marine organisms. More widespread contamination of commercially exploited species (particularly by mercury) has presented the most significant problem and has necessitated control of sludge quality through increased pretreatment of effluents.

These controls have been shown by subsequent monitoring to be effective, and it is concluded that the present environmental impact of sludge dumping is restricted to chemical contamination in the immediate vicinity of the disposal area and minor ecological changes which may not be detrimental to fisheries. It is also concluded that disposal to sea represents the best environmental as well as economic option in each case studied.

The history of the control of sewage discharges to the Thames Estuary demonstrates that a scientific understanding of the system allows the effect of improving treatment methods at different points to be predicted to gain the maximum benefit from the available resources. The Thames is now cleaner than at any time this century through this approach, despite wide variations in the degree of treatment throughout the estuary.

The overall conclusion of United Kingdom experience is that an understanding of the processes operating in the disposal of waste, either through discharge or dumping, enables the assimilative capacity of an estuary or ocean to be utilized while still protecting the marine environment and meeting environmental quality criteria for the receiving area. In such a way it is believed resources for environmental protection are used more effectively than through the application of uniform standards to different water bodies which may have very different characteristics.

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Appendix E

The Water Quality Control Plan of California

WATER QUALITY CONTROL PLAN

OCEAN WATERS OF CALIFORNIA



REPRINTED FEBRUARY 1981

State Water Resources Control Board



STATE OF CALIFORNIA
Edmund G. Brown Jr., Governor

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CONTROL BOARD

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State of California
The Resources Agency

STATE WATER RESOURCES CONTROL BOARD

1978

WATER QUALITY CONTROL PLAN

FOR

OCEAN WATERS OF CALIFORNIA

Adopted and Effective
1978

Reprinted February 1981

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RESOLUTION NO. 78-2

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STATE WATER RESOURCES CONTROL BOARD
RESOLUTION NO. 78-2

WATER QUALITY CONTROL PLAN
FOR
OCEAN WATERS OF CALIFORNIA

WHEREAS:

1. The Board adopted the "Water Quality Control Plan for Ocean Waters of California" on July 6, 1972;

2. In the period between July 1972 and December 1977, the Board has received additional information regarding ocean water quality and wastewater management needs.

3. The Board conducted an extensive review of the 1972 Plan, prepared a draft set of amendments, prepared and circulated a draft initial study and negative declaration in accordance with the provisions of the California Environmental Quality Act, and on October 19, 1977, held a public hearing regarding the draft amendments.

THEREFORE BE IT RESOLVED THAT:

1. The Board adopts the "Water Quality Control Plan for Ocean Waters of California 1978" and the Negative Declaration and Initial Study pertaining thereto.

2. The Board hereby declares its intent to determine from time to time the need for revising the Plan to assure that it reflects current knowledge of water quality objectives necessary to protect beneficial uses of ocean waters and that it is based on latest technological improvements.

3. The Board hereby declares its intent to require continual monitoring of the marine environment to assure that the Plan reflects the latest available data and that the water quality objectives fully protect the marine environment.

CERTIFICATION

The undersigned, Executive Director of the State Water Resources Control Board, does hereby certify that the foregoing is a full, true, and correct copy of a resolution duly and regularly adopted at a meeting of the State Water Resources Control Board held on January 19, 1978.

Larry F. Walker
Executive Director
Water Quality

CALIFORNIA STATE WATER RESOURCES CONTROL BOARD

WATER QUALITY CONTROL PLAN FOR OCEAN WATERS OF CALIFORNIA
(As amended, 1978)

INTRODUCTION

In furtherance of legislative policy set forth in Section 13000 of Division 7 of the California Water Code (Stats. 1969, Chap. 482) and pursuant to the authority contained in Section 13170 (Stats. 1971, Chap. 1288) the State Water Resources Control Board hereby finds and declares that protection of the quality of the ocean waters for use and enjoyment by the people of the State requires control of the discharge of waste[1/] to ocean water[2/] in accordance with the provisions contained herein.

CHAPTER I.
BENEFICIAL USES

The beneficial uses of the ocean waters of the State that shall be protected include industrial water supply, recreation, esthetic enjoyment, navigation, and preservation and enhancement of fish, wildlife, and other marine resources or preserves.

CHAPTER II.
WATER QUALITY OBJECTIVES

This chapter sets forth limits or levels of water quality characteristics for ocean waters to ensure the reasonable protection of beneficial uses and the prevention of nuisance. The discharge of waste shall not cause violation of these objectives.[3/]

A. Bacteriological Characteristics

1. Within a zone bounded by the shoreline and a distance of 1,000 feet from the shoreline or the 30-foot depth contour, whichever is further from the shoreline, and in areas[4/] outside this zone used for body-contact sports, the following bacteriological objectives shall be maintained throughout the water column:

   (a.) Samples of water from each sampling station shall have a concentration of coliform organisms less than 1,000 per 100 ml (10 per ml); provided that not more than 20 percent of the samples at any sampling station, in any 30-day period, may

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Originally adopted by the State Water Resources Control Board by Resolution No. 72-45 on July 6, 1972, and amended in 1973 and 1978.

exceed 1,000 per 100 ml (10 per ml), and provided further that no single sample when verified by a repeat sample taken within 48 hours shall exceed 10,000 per 100 ml (100 per ml).

(b.) The fecal coliform concentration based on a minimum of not less than five samples for any 30-day period, shall not exceed a log mean of 200 per 100 ml nor shall more than 10 percent of the total samples during any 30-day period exceed 400 per 100 ml.

2. At all areas[4] where shellfish may be harvested for human consumption, the following bacteriological objectives shall be maintained throughout the water column:

The median total coliform concentration shall not exceed 70 per 100 ml, and not more than 10 percent of the samples shall exceed 230 per 100 ml.

B. Physical Characteristics

1. Floating particulates and grease and oil shall not be visible.

2. The discharge of waste shall not cause esthetically undesirable discoloration of the ocean surface.

3. The transmittance of natural light shall not be significantly[5] reduced at any point outside the initial dilution zone.[6]

4. The rate of deposition of inert solids and the characteristics of inert solids in ocean sediments shall not be changed such that benthic communities are degraded.[7]

C. Chemical Characteristics

1. The dissolved oxygen concentration[8] shall not at any time be depressed more than 10 percent from that which occurs naturally, as the result of the discharge of oxygen demanding waste materials.

2. The pH[8] shall not be changed at any time more than 0.2 units from that which occurs naturally.

Chapter II. C.

3. The dissolved sulfide concentration of waters in and near sediments shall not be significantly[5/] increased above that present under natural conditions.

4. The concentration of substances set forth in Chapter IV, Table B, in marine sediments shall not be increased to levels which would degrade[7/] indigenous biota.

5. The concentration of organic materials in marine sediments shall not be increased above that which would degrade[7/] marine life.

6. Nutrient materials shall not cause objectionable aquatic growths or degrade[7/] indigenous biota.

D. Biological Characteristics

1. Marine communities, including vertebrate, invertebrate, and plant species, shall not be degraded.[7/]

2. The natural taste, odor, and color of fish, shellfish, or other marine resources used for human consumption shall not be altered.

E. Radioactivity

1. Radioactivity shall not exceed the limits specified in Title 17, Chapter 5, Subchapter 4, Group 3, Article 3, Section 30269 of the California Administrative Code.

CHAPTER III.
GENERAL REQUIREMENTS FOR MANAGEMENT OF WASTE DISCHARGE TO THE OCEAN

A. Waste management systems that discharge to the ocean must be designed and operated in a manner that will maintain the indigenous marine life and a healthy and diverse marine community.

B. Waste discharged to the ocean must be essentially free of:

1. material that is floatable or will become floatable upon discharge,

2. settleable material or substances that form sediments which degrade[7/] benthic communities or other aquatic life,

3. substances toxic to marine life due to increases in concentrations in marine waters or sediments,

4. substances that significantly decrease the natural light to benthic communities and other marine life, and

5. materials that result in esthetically undesirable discoloration of the ocean surface.

C. Waste effluents shall be discharged in a manner which provides sufficient initial dilution to minimize the concentrations of substances not removed in treatment.

D. Location of waste discharges must be determined after a detailed assessment of the oceanographic characteristics and current patterns to assure that:

1. pathogenic organisms and viruses are not present in areas where shellfish are harvested for human consumption or in areas used for swimming or other body-contact sports.9/

2. natural water quality conditions are not altered in areas designated as being of special biological significance, and

3. maximum protection is provided to the marine environment.

CHAPTER IV.
QUALITY REQUIREMENTS
FOR WASTE DISCHARGES
(EFFLUENT QUALITY REQUIREMENTS)

This chapter sets forth the quality requirements for waste discharge to the ocean.3/

Table A limitations apply only to publicly owned treatment works and industrial discharges for which Effluent Guideline Limitations have not been established pursuant to Sections 301, 302, 304, or 306 of the Federal Water Pollution Control Act of 1972.

Table B limitations apply to all discharges within the jurisdiction of this Plan.

The State Board is authorized to administer and enforce effluent requirements established pursuant to the Federal Water Pollution Control Act of 1972. Effluent limitations established

under Sections 301, 302, 306, 307, 316, 403, and 405 of the aforementioned Federal Act and administrative procedures pertaining thereto, are included in this Plan by reference.

Compliance with Table A limitations, or Environmental Protection Agency Effluent Guideline Limitations for industrial discharges, shall be the minimum level of treatment acceptable under this Plan, and define reasonable treatment and waste control technology.

TABLE A

MAJOR WASTEWATER CONSTITUENTS AND PROPERTIES

	Unit of measurement	Limiting Concentrations Monthly (30 day Average)	Weekly (7 day Average)	Maximum at any time
Grease and Oil	mg/l	25	40	75
Suspended Solids	mg/l	75 Percent Removal		
Settleable Solids	ml/l	1.0	1.5	3.0
Turbidity	JTU	75	100	225
pH	units	within limits of 6.0 to 9.0 at all times		
Toxicity Concentration[12/]	tu	1.5	2.0	2.5

TABLE B

TOXIC MATERIALS LIMITATIONS

Effluent limitations shall be imposed in a manner prescribed by the State Board[10/] such that the concentrations set forth below as water quality objectives, shall not be exceeded in the receiving water upon the completion of initial dilution, except that limitations indicated for total chlorinated pesticides and PCB's and Radioactivity shall apply driectly to the undiluted waste effluent.

Chapter IV.
Table B (continued)

	Unit of Measurement	Limiting Concentrations 6-Month Median	Daily Maximum	Instantaneous Maximum
Arsenic	mg/l	0.008	0.032	0.08
Cadmium	mg/l	0.003	0.012	0.03
Total Chromium	mg/l	0.002	0.008	0.02
Copper	mg/l	0.005	0.020	0.05
Lead	mg/l	0.008	0.032	0.08
Mercury	mg/l	0.00014	0.00056	0.0014
Nickel	mg/l	0.02	0.08	0.2
Silver	mg/l	0.00045	0.0018	0.0045
Zinc	mg/l	0.020	0.08	0.2
Cyanide	mg/l	0.005	0.02	0.05
Phenolic Compounds	mg/l	0.03	0.12	0.3
Total Chlorine Residual	mg/l	0.002	(See Footnote 11/)	
Ammonia (expressed as nitrogen)	mg/l	0.6	2.4	6.0
Toxicity Concentration 12/	tu	0.05	-	-
Total Chlorinated Pesticides and PCB's 13/	mg/l	0.002	0.004	0.006
Radioactivity	Not to exceed limits specified in Section 30269 of the California Administrative Code.			

CHAPTER V.
DISCHARGE PROHIBITIONS

A. Hazardous Substances

The discharge of any radiological, chemical, or biological warfare agent or high-level radioactive waste into the ocean is prohibited.

B. Areas of Special Biological Significance

Waste shall be discharged a sufficient distance from areas designated as being of special biological significance to assure maintenance of natural water quality conditions in these areas.

C. Sludge

The discharge of municipal and industrial waste sludge directly to the ocean, or into a waste stream that discharges to the ocean, shall be prohibited. The discharge of sludge digester supernatant directly to the ocean, or into a waste stream that discharges to the ocean without further treatment shall be prohibited.

D. By-Passing

The by-passing of untreated wastes containing concentrations of pollutants in excess of those of Table A or Table B to the ocean is prohibited.

CHAPTER VI.
GENERAL PROVISIONS

A. Effective Date

This Plan is in effect as of the date of adoption by the State Water Resources Control Board.

B. Waste Discharge Requirements

The Regional Boards may establish more restrictive water quality objectives and effluent quality requirements than those set forth in this Plan as necessary for the protection of beneficial uses of the ocean.

Regional Boards may impose alternative less restrictive provisions to those contained within Table B of the Plan, provided an applicant can demonstrate that:

Reasonable control technologies (including source control, material substitution, treatment and dispersion) will not provide for complete compliance or;

Any less stringent provisions would encourage water reclamation;

Provided further than;

a) Any alternative water quality objectives shall be below the conservative estimate of chronic toxicity, upon which this Plan is based, and such alternative will provide for adequate protection of the marine environment;

b) A receiving water toxicity objective of 0.05 tu is not exceeded; and

c) The State Board and the Environmental Protection Agency concur in the Regional Board findings and alternative limits.

C. Revision of Waste Discharge Requirements

The Regional Board shall revise the waste discharge requirements for existing discharges as necessary to achieve compliance with this Plan and shall also establish a time schedule for compliance in accordance with State Board Resolution 74-5.

D. State Board Review of Time Schedules

The State Board shall review proposed time schedules for all municipal discharges throughout the State and shall recommend to the Regional Boards specific schedules to assure the maximum benefit from, and equitable distribution of, available state and federal grant funds.

E. Monitoring Program

The Regional Board shall require dischargers to conduct self-monitoring programs and submit reports necessary to determine compliance with the waste discharge requirements, and may require dischargers to contract with agencies or persons acceptable to the Regional Board to provide monitoring reports. Such monitoring programs shall comply with Guidelines for Monitoring the Effects of Waste Discharges on the Ocean which shall be issued by the Executive Director of the State Board.

F. Areas of Special Biological Significance

Areas of special biological significance shall be designated by the State Board after a public hearing by the Regional Board and review of its recommendations.

G. State Board Exceptions to Plan Requirements

The State Board may, subsequent to a public hearing, and with the concurrence of the Environmental Protection Agency, grant exceptions to any provision of this Plan where the Board determines:

1) The existence of unusual circumstances not anticipated at the time of the Plan's adoption;

2) The exception will not compromise protection of ocean waters for beneficial uses, and

3) The public interest will be served.

FOOTNOTES

1/ This Plan is applicable, in its entirety, to point source discharges to the ocean. Nonpoint sources of waste discharges to the ocean are subject to Chapter I - Beneficial Uses, Chapter II - Water Quality Objectives, Chapter III - General Requirements, Chapter IV - Table B (wherein compliance with water quality objectives shall, in all cases, be determined by direct measurements in the receiving waters), and Chapter V- Discharge Prohibitions.

This Plan is not applicable to discharges to enclosed bays and estuaries or inland waters nor is it applicable to vessel wastes, and the control of dredging spoil.

Provisions regulating the thermal aspects of waste discharged to the ocean are set forth in the Water Quality Control Plan for the Control of Temperature in the Coastal and Interstate Waters and Enclosed Bays and Estuaries of California dated May 18, 1972.

2/ Ocean waters are waters of the Pacific Ocean adjacent to the California coast outside of enclosed bays, estuaries, and coastal lagoons.

Enclosed bays are indentations along the coast which enclose an area of oceanic water within distinct headlands or harbor works. Enclosed bays include all bays where the narrowest distance between headlands or outermost harbor works is less than 75 percent of the greatest dimension of the enclosed portion of the bay. This definition includes but is not limited to: Humboldt Bay, Bodega Harbor, Tomales Bay, Drakes Estero, San Francisco Bay, Morro Bay, Los Angeles Harbor, Upper and Lower Newport Bay, Mission Bay, and San Diego Bay.

Estuaries and coastal lagoons are waters at the mouths of streams which serve as mixing zones for fresh and ocean waters during a major portion of the year. Mouths of streams which are temporarily separated from the ocean by sandbars shall be considered as estuaries. Estuarine waters will generally be considered to extend from a bay or the open ocean to the upstream limit of tidal action but may be considered to extend seaward if significant mixing of fresh and salt water occurs in the open coastal waters. The waters described by this definition include but are not limited to the Sacramento-San Joaquin Delta as defined by Section 12220 of the California Water Code, Suisun Bay, Carquinez Strait downstream to Carquinez Bridge, and appropriate areas of the Smith, Klamath, Mad, Eel, Noyo, and Russian Rivers.

Footnotes (continued)

3/ The Water Quality Objectives and Effluent Quality Requirements are defined by a statistical distribution when appropriate. This method recognizes the normally occurring variations in treatment efficiency and sampling and analytical techniques and does not condone poor operating practices.

4/ Body-contact sports areas outside the shoreline zone set forth in Chapter II.A.1. and all shellfishing areas shall be determined by the Regional Board on an individual basis.

5/ A significant difference is defined as a statistically significant difference in the means of two distributions of sampling results at the 95 percent confidence level.

6/ Initial dilution is the process which results in the rapid and irreversible turbulent mixing of wastewater with ocean water around the point of discharge.

For a submerged buoyant discharge, characteristic of most municipal and industrial wastes that are released from the submarine outfalls, the momentum of the discharge and its initial buoyancy act together to produce turbulent mixing. Initial dilution in this case is completed when the diluting wastewater ceases to rise in the water column and first begins to spread horizontally.

For shallow water submerged discharges, surface discharges, and nonbuoyant discharges, characteristic of cooling water wastes and some individual discharges, turbulent mixing results primarily from the momentum of discharge. Initial dilution, in these cases, is considered to be completed when the momentum induced velocity of the discharge ceases to produce significant mixing of the waste, or the diluting plume reaches a fixed distance from the discharge to be specified by the Regional Board, whichever results in the lower estimate for initial dilution.

For the purpose of this Plan, minimum initial dilution is the lowest average initial dilution within any single month of the year. Dilution estimates shall be based on observed waste flow characteristics, observed receiving water density structure, and the assumption that no currents, of sufficient strength to influence the initial dilution process, flow across the discharge structure.

The Executive Director shall issue guidelines to be used by the State and Regional Boards for determining the initial dilution achieved by each ocean discharge.

Footnotes (continued)

7/ Degradation shall be determined by analysis of the effects of waste discharge on species diversity, population density, contamination, growth anomalies, debility, or supplanting of normal species by undersirable plant and animal species.

8/ Compliance with the water quality objectives of Chapter II shall be determined from samples collected at stations representative of the area within the waste field where initial dilution is completed.

9/ Waste that contains pathogenic organisms or viruses should be discharged a sufficient distance from shellfishing and body-contact sports areas to maintain applicable bacteriological standards without disinfection. Where conditions are such that an adequate distance cannot be attained, reliable disinfection in conjunction with a reasonable separation of the discharge point from the area of use must be provided. Consideration should be given to disinfection procedures that do not increase effluent toxicity and that constitute the least environmental and human hazard in their production, transport, and utilization.

10/ Effluent limitations for substances identified in Chapter IV, Table B, with the exceptions of Radioactivity and Total Identifiable Chlorinated Hydrocarbons, shall be determined through the use of the following equation:

$$Ce = Co + Dm\ (Co - Cs) \qquad (1)$$

where:

Ce = the effluent concentration limit,
Co = the concentration to be met at the completion of initial dilution,
Cs = background seawater concentration, and
Dm = minimum probable initial dilution expressed as parts seawater per part wastewater.

The State Board shall assist the Regional Boards in the evaluation of Dm, the minimum initial dilution for a specific waste discharge. Discharging agencies will be informed of the basis for the determination of minimum initial dilution.

Footnotes (continued)

Waste Constituent	Cs mg/l
Arsenic	0.003
Cadmium	0.000
Total Chromium	0.000
Copper	0.002
Lead	0.000
Mercury	0.00006
Nickel	0.00
Silver	0.00016
Zinc	0.008
Cyanide	0.000
Phenolic Compounds	0.0
Total Chlorine Residual	0.000
Ammonia (expressed as nitrogen)	0.0
Toxicity Concentration (in toxic units)	0.00

The six-month median effluent concentration limit shall apply as a moving median of daily values for any 180 day period in which daily values represent flow weighted average concentrations within a 24-hour period. For intermittent discharges, the daily value shall be considered to equal zero for days on which no discharge occurred.

The daily maximum effluent concentration limit shall apply to flow weighted concentrations within 24 hours.

The instantaneous maximum shall apply to grab sample determination.

Discharge requirements shall also specify effluent requirements in terms of mass emission rate limits utilizing the general formula:

$$\text{lbs/day} = 8.34 \times Ce \times Q \qquad (2)$$

The six-month median limit on daily mass emissions shall be determined using the six-month median effluent concentration as Ce and the observed flow rate Q in million of gallons per day. The daily maximum mass emission shall be determined using the daily maximum effluent concentration limit as Ce and the observed flow rate Q in millions of gallons per day.

Any significant change in waste flow shall be cause for reevaluating effluent quality requirements.

If a calculated Ce value falls below the limit of detection of the test method specified in the Code of the Federal Register, 40 CFR 136, the limit of detection shall serve as

Footnotes (continued)

the limiting effluent concentration. The State or Regional Board may, at their discretion, specify test methods which are more sensitive than those specified in 40 CFR 136. Total chlorine residual is likely to be a "limit of detection" effluent requirement in many cases. The limit of detection of total chlorine residual in standard test methods is less than, or equal to, 0.02 mg/l.

Due to the large total volume of powerplant and other heat exchange discharges, special procedures must be applied for determining compliance with Table B limitations on a routine basis. Effluent concentration values (Ce) shall be determined through the use of Equation 1 considering the minimal probable initial dilution of the combined effluent (in-plant waste streams plus cooling water flow). These concentration values shall then be converted to mass emission limitations as indicated in Equation 2. The mass emission limits will then serve as requirements applied to all in-plant waste streams taken together which discharge into the cooling water flow. The procedure described above shall apply to all Table B materials except limitations on total chlorine residual and radioactivity which shall apply to, and be measured in, the combined final effluent.

11 Water quality objectives for total chlorine residual applying to intermittent discharges not exceeding two hours, and 24 hours and instantaneous (1 minute) maximum objectives applying to continuous sources, shall be determined through the use of the following equation:

$$\log y = -0.328 \log x - 0.905$$

where: y = the water quality objective to apply when chlorine is being discharged;
x = the duration of uninterrupted chlorine discharge in minutes.

12/ This parameter shall be used to measure the acceptability of waters for supporting a healthy marine biota until improved methods are developed to evaluate biological response.

a. Toxicity Concentration (Tc)

Expressed in Toxicity Units (tu)

$$Tc\ (tu) = \frac{100}{96\text{-hr. } TLm\%}$$

Footnotes (continued)

b. Median Tolerance Limit (TLm%)

The TLm shall be determined by static or continuous flow bioassay techniques using standard test species. If specific identifiable substances in wastewater can be demonstrated by the discharger as being rapidly rendered harmless upon discharge to the marine environment, the TLm may be determined after the test samples are adjusted to remove the influence of those substances.

When it is not possible to measure the 96-hr. TLm due to greater than 50 percent survival of the test species in 100 percent waste, the toxicity concentration shall be calculated by the expression:

$$Tc\ (tu) = \frac{\log\ (100 - S)}{1.7}$$

S = percentage survival in 100% waste.

13/ Total chlorinated pesticides and PCB's shall be measured by summing the individual concentrations of DDT, DDD, DDE, aldrin, BHC, chlordane, endrin, heptachlor, lindane, dieldrin, and polychlorinated biphenyls.

The Executive Director shall undertake a preliminary investigation into the presence and hazards posed by the other halogenated hydrocarbons which may be present in wastewater discharges. Such review shall lead to a recommendation regarding the necessity for regulation, gathering of necessary data, or other appropriate actions which should be taken by the State Board.

As part of the preliminary investigation, the Executive Director may impose additional monitoring requirements on discharging agencies to assess the occurrence of halogenated hydrocarbons other than those specifically mentioned in this Plan.

STATE WATER RESOURCES CONTROL BOARD

P. O. Box 100, Sacramento, CA 95801

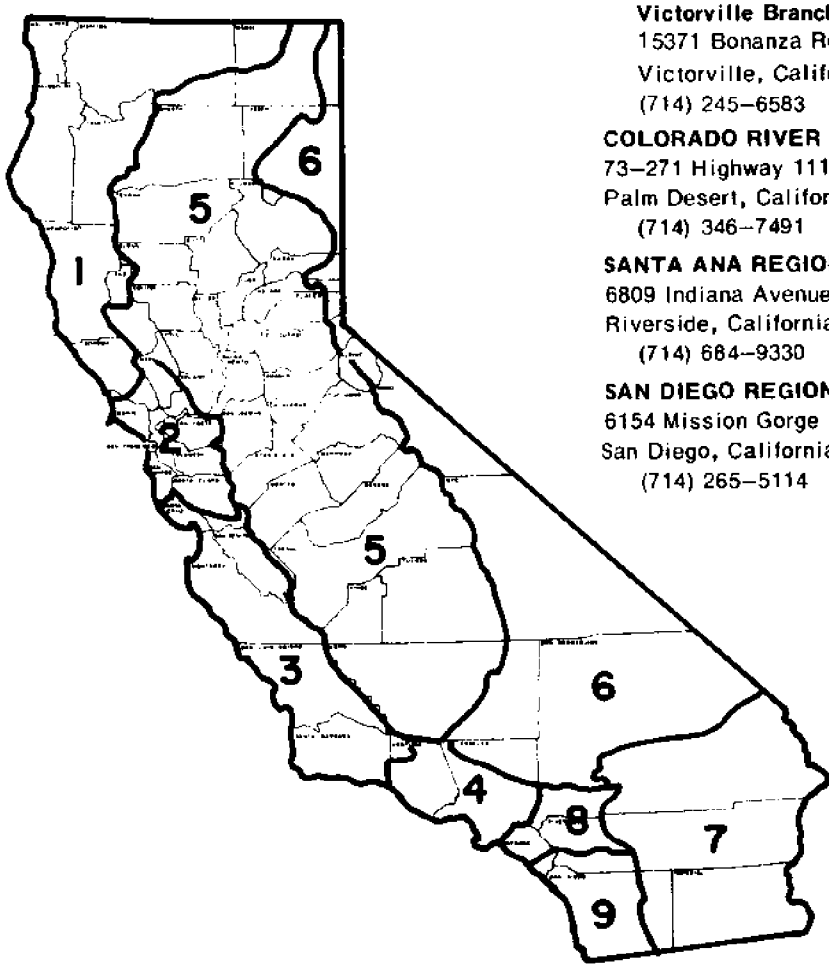
CALIFORNIA REGIONAL WATER QUALITY CONTROL BOARDS

NORTH COAST REGION (1)
1000 Coddingtown Center
Santa Rosa, California 95401
(707) 545–2620

SAN FRANCISCO BAY REGION (2)
1111 Jackson Street, Room 6040
Oakland, California 94607
(415) 464–1255

CENTRAL COAST REGION (3)
1102–A Laurel Lane
San Luis Obispo, California 93401
(805) 549–3147

LOS ANGELES REGION (4)
107 South Broadway, Room 4027
Los Angeles, California 90012
(213) 620–4460



CENTRAL VALLEY REGION (5)
3201 S Street
Sacramento, California 95816
(916) 445–0270

Fresno Branch Office
3374 East Shields Avenue, Room 18
Fresno, California 93726
(209) 445–5116

Redding Branch Office
100 East Cypress Avenue
Redding, California 96001
(916) 246–6376

LAHONTAN REGION (6)
2092 Lake Tahoe Boulevard
P. O. Box 14367
South Lake Tahoe, California 95702
(916) 544–3481

Victorville Branch Office
15371 Bonanza Road
Victorville, California 92392
(714) 245–6583

COLORADO RIVER BASIN REGION (7)
73–271 Highway 111, Suite 21
Palm Desert, California 92260
(714) 346–7491

SANTA ANA REGION (8)
6809 Indiana Avenue
Riverside, California 92506
(714) 684–9330

SAN DIEGO REGION (9)
6154 Mission Gorge Road, Suite 205
San Diego, California 92120
(714) 265–5114

Appendix F

EPA Ocean Discharge Criteria

federal register

Friday
October 3, 1980

Part VI

Environmental Protection Agency

Ocean Discharge Criteria

ENVIRONMENTAL PROTECTION AGENCY

40 CFR Part 125

[FRL 1609-1]

Ocean Discharge Criteria

AGENCY: Environmental Protection Agency.

ACTION: Final rule.

SUMMARY: EPA is promulgating final guidelines under section 403(c) of the Clean Water Act. These guidelines will be applied in issuing and revising National Pollutant Discharge Elimination System Permits for discharges into the territorial seas, the contiguous zone and the oceans.

DATES: These guidelines become effective on November 3, 1980.

FOR FURTHER INFORMATION CONTACT: Kenneth Farber, Office of Water Regulations and Standards (WH-586), Environmental Protection Agency, 401 M Street, SW, Washington, D.C. 20460, 202-472-5746.

SUPPLEMENTARY INFORMATION:

I. Background

EPA is today promulgating revised guidelines for determining the degradation of the territorial seas, the contiguous zone and the oceans. Pursuant to section 403(a) of the Clean Water Act, no National Pollutant Discharge Elimination System ("NPDES") permit for discharges into these marine waters may be issued when these guidelines are in effect except in compliance with the guidelines.

These guidelines are issued pursuant to section 403(c)(1) which provides that:

> The Administrator shall, within one hundred and eighty days after enactment of this Act (and from time to time thereafter), promulgate guidelines for determining the degradation of the waters of the territorial seas, the contiguous zone, and the ocean, which shall include:
>
> (A) the effect of disposal of pollutants on human health or welfare, including but not limited to plankton, fish, shellfish, wildlife, shorelines, and beaches.
>
> (B) the effect of disposal of pollutants on marine life, including the transfer, concentration, and dispersal of pollutants or their byproducts through biological, physical, and chemical processes; changes in marine ecosystem diversity, productivity, and stability; and species and community population changes;
>
> (C) the effect of disposal of pollutants on esthetic, recreation, and economic values;
>
> (D) the persistence and permanence of the effects of disposal of pollutants;
>
> (E) the effect of the disposal at varying rates, of particular volumes and concentrations of pollutants;
>
> (F) other possible locations and methods of disposal or recycling of pollutants including land-based alternatives; and
>
> (G) the effect on alternate uses of the oceans, such as mineral exploitation and scientific study.

On October 15, 1973, EPA promulgated combined regulations implementing section 102(a) of the Marine Protection, Research, and Sanctuaries Act and section 403(c) of the Clean Water Act. The primary focus of these regulations was on the ocean disposal of waste material, including sewage sludges, liquid and solid industrial wastes and dredged materials, by dumping from moving vessels.

In practice, these regulations proved unworkable in many respects as section 403 ocean discharge criteria. At the same time, operating experience demonstrated that the ocean dumping regulations themselves required revision. EPA therefore determined that both the ocean dumping regulations and the ocean discharged criteria should be revised and published as separate regulations. All reference to section 403(c) guidelines was deleted from the revised ocean dumping regulations which were promulgated on January 11, 1977 (42 FR 2468). However, the Agency encountered substantial difficulty in developing revised ocean discharge guidelines, and, until recently, there have been no published national guidelines in place. Since withdrawal of the original guidelines, permit writers have been implementing section 403 on a case-by-case basis.

On June 21, 1979, the Pacific Legal Foundation filed suit in United States District Court for the Eastern District of California, seeking, among other things, that EPA promulgate revised section 403(c) guidelines, *Pacific Legal Foundation* v. *Costle*, Civ. No. S-79-429-PCW. The National Wildlife Federation intervened in that lawsuit. On October 31, 1979, the Court ordered EPA both to promulgate these guidelines and to publish interim guidelines stating Agency policy in reviewing, issuing, or denying NPDES permits under section 403, pending promulgation of the final guidelines. The interim guidelines were published in the **Federal Register** on November 15, 1979, 44 FR 65751, and they will be superseded by the final guidelines published today.

The Agency then published proposed ocean discharge criteria in the **Federal Register** (45 FR 9548) on February 12, 1980, held an oral hearing on the proposal on March 21, 1980, and provided a comment period for submission of written comments which was to end on March 28, 1980. At the request of various interested groups, the comment period was extended for 30 days. Based on the extended comment period and on the large volume of comments received, the Agency moved the court to extend the final promulgation date by 120 days beyond the July 30 deadline. The court extended the final deadline until September 30, 1980.

II. Development of the 403 Guidelines

1. Synopsis of the Guidelines

Section 403 is intended to prevent unreasonable degradation of the marine environment and to authorize imposition of effluent limitations, including a prohibition of discharge, if necessary, to ensure this goal. These guidelines were developed to satisfy this intent. They provide flexibility to permit writers to tailor application requirements, effluent limitations, and reporting requirements to the specific circumstances of each discharger's situation, while ensuring consistency and certainty by imposing minimum requirements, in situations where the long-term impact of a discharge is not fully understood.

Under these guidelines, no NPDES permit may be issued which authorizes a discharge of pollutants that will cause unreasonable degradation of the marine environment. Prior to permit issuance, the director, defined as either the Regional Administrator or the State Director where there is an approved State program, or an authorized representative, is required to evaluate whether a proposed discharge will cause such degradation. In making this determination, the director is to consider the factors specified in § 125.122 (a) and (b).

In cases where sufficient information is available for the director to make a reasonable determination whether unreasonable degradation of the marine environment will occur, the director is governed by § 125.123 (a) and (b) of the regulations. Discharges which will cause unreasonable degradation will be prohibited; other discharges may be permitted under conditions necessary to ensure that such degradation will not occur.

In those cases where the director is unable to determine whether unreasonable degradation will occur, § 125.123(c) governs. No discharge in this situation is allowed unless the director can reasonably determine that: (1) the discharge will not cause irreparable harm to the marine environment while further evaluation is undertaken; (2) there are no reasonable

alternatives to the discharge; and (3) the discharge will comply with certain mandatory permit conditions, including a bioassay-based discharge limitation and monitoring requirements. These permit conditions will assist in determining whether and to what extent further limitations are necessary to ensure that the discharge does not cause unreasonable degradation. If further limitations are necessary, § 125.123(d)(4) provides that the permit must be then modified to include these additional limitations or else revoked.

These guidelines encourage the use of available information in addition to any supplied by the permit applicant. Thus, the director may make determination based on information such as that contained in any relevant environmental impact statement, section 301(h) or other variance applications, existing technical and environmental field studies, or EPA industrial and municipal waste surveys.

2. Relationship Between the Statute and the Guidelines

(a) *Section 403(c)(1)*—Section 403(c)(1) specifies seven factors which are to be included in guidelines for determining the degradation of marine waters. These factors form the basis for the determinations which must be made pursuant to these guidelines.

Most of the statutory factors, including 403(c)(1)(A), (B), (C), (D), (E), and a portion of (G), involve consideration of the biological effects of the discharge of pollutants. These factors, either directly or indirectly, must be evaluated by the director in determining whether a discharge will cause unreasonable degradation of the marine environment. Section 125.122(a) requires that the director assess such variables as the location of the discharge, including the composition of the biological community and existence of special aquatic sites, such as marine sanctuaries; the nature of the pollutants which are to be discharged, including their quantities, composition, potential for bioaccumulation, persistence and their transport in the environment, and the effect on human health. This assessment should adequately address the statutory factors relating to biological effects of the discharge.

Section 403(c)(1)(C) also involves consideration of economic and social impacts of the discharge, as does section 403(c)(1)(G). The guidelines address these factors in assessing whether a discharge will cause unreasonable degradation of the marine environment. Section 122.121(f) defines "unreasonable degradation of the marine environment" to include, among other things, "loss of esthetic, recreational or economic values which are unreasonable in relation to the benefit derived from the discharge." Thus, even where the director has determined that there are no significant changes in ecosystem diversity, productivity and stability, and there is no threat to human health, he may conclude that a discharge may not be authorized if the adverse impact on such activities as fishing, recreation, and/or other economic or social benefits is unreasonable in relation to benefits, such as oil and gas production, derived from the discharge.

Section 403(c)(1)(F) involves consideration of other possible locations and disposal methods for pollutants. Although EPA has considered this factor in developing these guidelines, the director is not required to assess alternatives in all cases. Under section 125.123(c)(2) the director must assess the availability of alternatives, including land-based alternatives, only in those cases where he cannot determine that a discharge will not cause unreasonable degradation of the marine environment.

Additionally, the guidelines establish a presumption that discharges in compliance with sections 301(h), 316(a), 301(g) and State water quality standards under section 303 will not cause unreasonable degradation. Although the director may, on the basis of the factors specified in § 125.122(a), conclude that additional permit limitations are in fact necessary even though the requirements of these other sections have been met, the similarity between the objectives and requirements of these provisions and those of section 403 warrants a presumption that discharges in compliance with these sections also satisfy section 403. Also, even though the regulations provide that a successful section 316(a) demonstration creates only a rebuttable presumption that section 403 has been satisfied, the provisions of section 316(c) may in some cases preclude the imposition of more stringent limitations under section 403.

POTWs obtaining section 301(h) variances are entitled to a presumption that their entire discharge is in compliance with section 403. However, the presumption applies only to the thermal component of a discharge subject to a 316(a) variance or to those specific non-conventional pollutants subject to a section 301(g) variance or to pollutants specifically limited by criteria in State water quality standards. Each of those provisions, like section 403, is geared toward assessing the environmental impact of a discharge. In order for a point source to receive a section 301(g), 301(h) or 316(a) variance, an evaluation of the biological and environmental effect of the discharge is required. Indeed, the statutory factors specified in these sections are similar to those contained in section 403(c). Similarly, State water quality standards established pursuant to section 303 of the Act are designed to preserve the quality of waters under State jurisdiction, including the territorial seas, and compliance with these standards should insure protection of the uses for which the waters are designated with respect to pollutants for which standards have been established.

(b) *Section 403(c)(2)*—Section 403(c)(2) states that:

> Where insufficient information exists on any proposed discharge to make a reasonable judgment on any of the guidelines established pursuant to this section no permit shall be issued under section 402 of this Act.

This section is the basis for two central elements of these regulations. First, the guidelines require that the director make potentially complex factual determinations on the basis of information which, in many cases, may be conflicting and in dispute. Section 403(c)(2) provides that the standard on which the director is to make these judgments is one of "reasonableness." In assessing the information in the administrative record, the director may authorize the discharge of pollutants if he is able to make a "reasonable judgment" about the determinations specified in the guidelines. Although these issues may involve scientific matters, the director is not bound by the same burden of proof which a scientist might require to reach a conclusion. The administrative process and the burden of proof in making these determinations are discussed below.

Second, the regulation provides, as required by section 403(c)(2), that the director may not authorize the discharge of pollutants if there is insufficient information to make these judgments. The regulation does not, however, require that there be complete knowledge of the impact of a discharge prior to permit issuance. Section 125.123(c) provides that a permit may be issued if the director has sufficient information to reasonably conclude, among other things, that the discharge will not cause irreparable harm to the environment while additional information is collected. The provision implements Congress' intent that "discharges permitted today will not irreversibly modify the oceans for future uses." S. Rep. No. 92-414, 92nd Cong., 1st Sess. at 75 (1971). It should insure adequate protection of the environment at all times while allowing the director to issue NPDES permits where existing

data may not be adequate to assess the long term impact of a discharge.

3. The Role of Section 403(c) Guidelines in NPDES Permit Issuance

These guidelines will be used to develop NPDES permits for the discharge of pollutants into the territorial seas, the contiguous zone and the oceans. Application of the guidelines will aid in protecting marine resources and their uses from the impact of pollution and in preventing unreasonable degradation of the marine environment.

Although sometimes described here as "guidelines" or "criteria", these promulgated regulations establish minimum requirements on discharges to protect the receiving waters. These guidelines will be used in evaluating applications for new, modified or renewed permits as they are submitted.

These guidelines apply in addition to other applicable provisions of the Clean Water Act. Permittees subject to section 403 must still comply with all other requirements of the Act, including applicable technology-based requirements specified by sections 301, 304 or 306 and water-quality based limitations specified by sections 303 or 307. Permittees may in certain circumstances be subject to the provisions of section 311 as well.

Section 403 applies to all discharges seaward of the inner boundary of the territorial seas. This boundary is defined by section 502(a) of the Act to be the—

> belt of the seas measured from the line of ordinary low water along that portion of the coast which is in direct contact with the open sea and the line marking the seaward limit of inland waters. . . .

This definition limits the number of land-based dischargers subject to section 403. For example, Chesapeake Bay, Boston Harbor, New York Harbor, San Francisco Bay and Puget Sound lie inside this inner boundary so that discharges into these waters are not subject to section 403 requirements. Of the approximately 62,400 existing NPDES permittees, 232 are land-based point sources discharging seaward of this inner boundary. These include 102 publicly-owned treatment works, 74 industrial plants, 25 steam electric plants and 31 federal facilities. These figures do not include the dischargers in Alaska whose location relative to the baseline defining the ocean boundary is not known.

In addition to these land-based dischargers, section 403 applies to all other point sources discharging into the marine waters covered by this regulation. By far the largest group of marine dischargers are oil and gas exploratory and production facilities. The Agency estimates that there are approximately 3,000 such facilities now operating.

III. Modifications to the Proposal

EPA provided both an oral hearing and a written comment period on the proposal, the latter extended by thirty days in response to the request of several interested groups. The preamble to the proposal specifically solicited comment on certain points, including mixing zone definition and determination, control of toxic pollutants; monitoring requirements and procedures; and effect of meeting requirements for a Section 301(h) variance.

Ten persons testified at the March 21, 1980 hearing on the proposal, and written comments were received from 81 parties, including many industrial groups, municipalities, conservation groups, federal agencies and several State governments. A listing of these commenters is presented in Appendix A. The commenters addressed the issues raised for comment in the preamble and raised a range of additional issues concerning EPA's approach in developing the proposed regulations. Detailed responses to the major public comments are presented in Appendix B.

In conjunction with the public comment review, EPA has reevaluated the proposed regulations and concluded that certain changes are appropriate. The final regulations retain the basic approach of the proposal. As in the proposal, this regulation provides that the director must determine whether a discharge will cause unreasonable degradation of the marine environment. Based on review of the numerous comments, however, EPA has made certain modifications which are intended to provide greater clarity to the permit writer, to ensure consistency in application of this regulation and to minimize burdens on the permit applicant.

Under the proposed regulation, applicants were required to submit a wide range of analyses and evaluations. Numerous commenters objected, stating that submission of this information was unnecessary, and in many cases redundant. In response to these comments, the final regulations now provide that the director may request information from the applicant but is encouraged to use other available sources, such as environmental impact statements, section 301(h) variance applications, consolidated permit applications, or EPA industrial and municipal waste surveys.

These final regulations also clarify the director's authority to issue permits where certain pre-issuance determinations relating to the effects of a discharge cannot be made. These regulations now provide that, in those cases, the discharge of pollutants may be authorized only where the director has sufficient information to make reasonable determinations regarding the potential for irreparable harm from the discharge and on the availability of alternatives. These regulations also establish certain minimum permit requirements in these cases. These requirements, identified as possible permit conditions in the proposed regulation, have been made mandatory in part in response to comments that the regulations needed to provide greater guidance to the director and applicant regarding permit conditions.

Finally, with respect to the relationship between the permit requirements of section 403(c) and variances issued under sections 301(g), 301(h), or 316(a), the final regulation provides that an applicant who has met the conditions necessary to receive such a variance is presumed to be in compliance with section 403(c) for those pollutants to which the variance applies.

IV. Procedures for Issuance of Permits Under Guidelines

1. Determination of Applicability of Section 403

The threshold determination for applicability of the section 403 guidelines is whether a proposed discharge will occur seaward of the inner boundary of the territorial seas. EPA's consolidated permit regulations (45 FR 33290, May 19, 1980) require that applicants list the latitude, longitude and name of the receiving waters for each outfall. Where the director is uncertain as to whether the outfall is within the waters covered by section 403(c), he should request guidance from EPA headquarters. Where the proposed discharge is in an area where the baseline defining the boundary of the territorial seas has not been determined, EPA will request a determination from the Department of State, which is responsible for defining the boundaries of the territorial seas.

2. Determination of Information Requirements Under Section 403(c)

Once a determination has been made that section 403(c) applies to a particular discharge, the director must determine what information is required to evaluate the discharge according to the section 403(c) criteria. The first thing that the director should do is survey the

currently available information about the discharge itself and about the area in which the discharge would occur. This information would include the data contained in the consolidated application form, as well as the data available from Agency reports and studies. The director should also notify the applicant of the existence of other currently available information.

After completing this survey of the available information, the director should determine what additional information would be required from the applicant for the evaluation of the impact of the ocean discharge as required by section 403(c). The applicant will have the responsibility of collecting this additional information and of submitting it to the director.

3. Determination of Unreasonable Degradation of the Marine Environment

Section 125.121(e) defines "unreasonable degradation of the marine environment" to include—

(1) significant adverse changes in ecosystem diversity, productivity and stability of the biological community within the area of discharge and surrounding biological communities; (2) threats to human health through direct exposure to pollutants or consumption of exposed aquatic organisms; or (3) loss of esthetic, recreational, scientific or economic values which are unreasonable in relation to the benefits derived from the discharge.

Sections 125.122 (a) and (b) specify an array of factors relevant in making these determinations.

In many cases the director will be able to reach conclusions based on data related to the nature of the proposed discharge. In areas which do not contain sensitive species or unusual biological communities or are not important for surrounding biological communities, the director may conclude that discharges containing primarily conventional pollutants will not cause unreasonable degradation. This is especially appropriate where the data indicate that there will be significant mixing with the receiving waters based on the flow of the discharge and the physical characteristics of the discharge site, such as water depth and turbulence. This determination may be appropriate for such dischargers as small publicly owned treatment works ("POTWs") and for industrial dischargers such as fruit canneries and fish processors.

For discharges into areas of biological concern or for complex or toxic discharges, additional evaluation may be necessary to determine whether a proposed discharge will cause unreasonable degradation. In assessing the need for further evaluation, the director should consider the vulnerability of the area of discharge and its role in the larger biological community. Significant or sensitive areas might include spawning sites, nursery or forage areas, migratory pathways or areas necessary for other functions or critical stages in the life cycle of organisms, areas of high productivity, or areas under stress due to biological or climatic conditions or discharges from other sources. Additionally, the director should consider whether a discharge will affect marine and wildlife species which are identified as endangered or threatened pursuant to the Endangered Species Act, 16 U.S.C. § 1531 *et seq.*, and those species critical to the structure or function of the ecosystem, such as in food chain relationships.

An assessment of the potential toxicity of a discharge should initially focus on the pollutants which are present in significant quantities relative to marine water quality criteria developed pursuant to section 304(a). The potential for bioaccumulation or persistence of pollutants in the environment is of particular importance.

The director must also consider the potential impacts of the discharge on human health either directly as through physical contact or indirectly through the food chain. These factors should be addressed when considering the location of the discharge and the type and volume of the discharger's effluent.

Determinations of the economic impact of the discharge should be based on the potential effect of the discharge on such activities as commercial fishing, recreation, mineral exploitation and scientific study. In considering whether a discharge will cause unreasonable economic impacts, the social as well as economic effects on a community should be considered.

Much, if not all, of the information necessary to make these evaluations already will be available to the director. Pursuant to section 122.53 of the consolidated permit regulations, applicants for NPDES permits must submit a range of significant information, including in many cases a detailed analysis of toxic pollutants in the waste stream. Additionally, any relevant environmental impact statements or section 301(h), 301(g) or 316(a) variance applications should provide significant data about the environmental impact of the proposed discharge. EPA industrial and municipal waste surveys and any data from relevant technical and environmental field studies that may have been conducted may serve as a source of information. Finally, Coastal Zone Management Plans or proposals for designation of an area as a marine sanctuary should also contain relevant information.

In cases where available information is insufficient for the director to make a determination, he may request additional information of the applicant pursuant to § 125.124. Where further analysis of the area of the proposed discharge is required, the director may require the applicant to perform assessments similar to those identified in the Technical Support Document prepared in conjunction with EPA's section 301(h) regulations (44 FR 34784, June 15, 1979). EPA headquarters will be available to provide assistance to the permit writer in developing information requirements where the discharge may be affecting an area of biological concern.

The director should work with the applicant to determine what types of assessment are necessary and to review or evaluate the assessment as it progresses. This level of participation is intended to determine the actual feasibility and the costs of such assessments for the applicant. Furthermore, it should avoid duplicative or inadequate assessments, thereby preventing delays in permit issuance. The Agency recognizes that some of the anticipated assessments required for permit issuance on the Outer Continental Shelf are beyond those which can reasonably be expected of the applicant and will require continued Agency research efforts.

The guidelines establish a presumption that discharges in compliance with sections 301(g), 301(h), 316(a) or State water quality standards will not cause unreasonable degradation with respect to the pollutants covered by those sections. Unless available data indicate that a discharge will cause unreasonable degradation, the director need not take additional steps, including the compilation of additional data, to support a conclusion that no further limitations on the discharge of these pollutants is necessary.

4. Determination of Irreparable Harm

Section 125.123(c)(1) requires that the director determine whether a discharge will cause irreparable harm to the marine environment in situations where he cannot determine whether the discharge will cause unreasonable degradation. Although the concepts of "irreparable harm" and "unreasonable degradation" involve similar considerations, the determination of "irreparable harm" is much narrower in scope.

In assessing the probability of "irreparable harm", the director need not focus his analysis on the overall impact of the discharge. Rather, he is only required to make a reasonable determination that the discharger, operating pursuant to the permit conditions established in § 125.123(c), will not cause permanent and significant harm to the environment during the period in which further data on the effects of the discharge are collected. In cases where such data, primarily that produced by monitoring, indicate that continued discharge will produce unreasonable degradation, the discharge must be halted or additional limitations established. Although evaluation of irreparable harm may in some cases involve difficult factual issues, determinations of this kind are currently a part of the NPDES permit issuance process. Pursuant to 40 CFR 124.60(a)(2)(ii), the presiding officer at an evidentiary hearing may authorize a facility to commence discharging prior to receipt of a final NPDES permit if the permit applicant demonstrates, among other things, that the discharge will not cause "irreparable harm to the environment . . ." This is essentially the same finding which the director must now make pursuant to these guidelines.

Certain factors are particularly significant in assessing the likelihood of "irreparable harm". Two such factors are the quantity of pollutants expected to be discharged and their potential for persistence in the marine environment. For example, a permit writer might authorize the operation of exploratory oil and gas facilities or a limited number of production facilities based on a conclusion that these limited operations will not cause irreparable harm to an area.

An additional factor is the sensitivity of the area into which the discharge is proposed. The director might conclude that a discharge could cause irreparable harm to unusual and interdependent communities, such as the coral reefs and associated communities of the Flower Garden Banks proposed marine sanctuary in the Gulf of Mexico. In such areas special conditions, including the prohibition of discharge, might be required.

Finally, data on the effect of similar discharges in similar areas is directly relevant to the determination of irreparable harm. Information demonstrating the recovery of the environment after the cessation of discharges from similar facilities would be a strong indication that irreparable harm is not likely to occur. For example, data indicate that even in areas of biological concern, biological communities reestablish themselves after the termination of discharges from publicly-owned treatment works. Thus, where the other provisions of § 125.123(c) are satisfied, the director might properly conclude that discharges from POTWs pursuant to this section may be authorized while further information is being collected.

5. Determination of Reasonable Alternatives

These guidelines establish two bases for determining whether reasonable alternatives to the proposed discharge exist. The first is the physical availability of alternative sites for disposal of pollutants. Such alternative sites might include either disposal facilities located on land, discharge point(s) within internal waters, or existing ocean dumping sites approved pursuant to the Marine Preservation, Research and Sanctuaries Act. In determining whether a site is a reasonable alternative to on-site disposal, the director should consider its distance from the site of the proposed discharge and whether its use would cause unwarranted economic impact on the discharger. For example, shipping wastes over long distances would likely impose such impact. This provision is intended to ensure some rule of reason in evaluating alternatives, it is not intended to impose a "cost/benefit" analysis of alternative sites.

In considering the availability of alternatives the director shall consider, based on available information or that requested from the applicant, the estimates of the amount of material requiring disposal. He should review the availability of existing land-based disposal sites and ocean dumping sites within a reasonable distance from the point of discharge and the estimated uncommitted capacity of such sites. The director should evaluate any reports of economic impact of discharge alternatives as may be supplied by the applicant.

The second basis for evaluating the feasibility of alternative sites relates to the relative environmental harm of disposal. Pursuant to section 121(e)(2), alternative disposal sites are not considered "reasonable alternatives" if on-site disposal is judged to be environmentally preferable. Thus, the discharge of pollutants might be authorized where disposal in alternative sites might produce equal or greater environmental harm than on-site discharge, or where transportation to alternative sites might produce a significant risk of greater environmental harm or a significant risk to human safety. For example, during certain seasons it may be undesirable to transport wastes off-site in areas of the North Atlantic or Alaska. Where the environmental or human health risks of transportation are significant, such transportation should not be considered a reasonable alternative.

6. Determination of Permit Conditions

Section 125.123(d) identifies specific permit conditions which are required for the issuance of a permit where a pre-permit issuance determination regarding degradation of the marine environment cannot be made. The director may also require any necessary permit conditions identified in section 125.123(d) to assure that unreasonable degradation of the marine environment will not occur under § 125.123(a).

(a) Limiting Permissible Concentration Requirements—Section 125.123(d)(1) requires, if a determination regarding unreasonable degradation cannot be made, that the discharge must pass certain bioassay-based requirements similar to those of EPA's ocean dumping regulations (40 CFR Part 227).

The applicant must demonstrate that his discharge will not exceed the limiting permissible concentration ("LPC") at the boundary of the mixing zone for a liquid phase and a suspended particular phase bioassay, in accordance with procedures for determining the LPC which are described in *Bioassay Procedures for the Ocean Disposal Permit Program*, U.S. EPA 600/9–78–010 March 1978 and in *Ecological Evaluation of Proposed Discharge of Dredge Material into the Ocean Waters*, EPA/Corps of Engineers, July 1977. If these manuals are revised in the future, bioassays shall be performed in accordance with any such revisions.

These regulations require an LPC which is derived from, but not identical to, the ocean dumping bioassay requirements. First, these regulations do not use section 304(a)(1) marine water quality criteria as a basis for determining an LPC. By use of a bioassay-based LPC, the ocean discharge criteria address the impact of the whole effluent and account for any synergistic or antagonistic effects. EPA recognizes that section 304(a)(1) criteria may in some cases require changes to reflect site-specific conditions, and the Agency is reevaluating the use of marine water quality criteria in the ocean dumping program.

The ocean discharge criteria also use a mixing zone extending laterally 100 meters in all directions from the discharge point(s) or to the boundary of the zone of initial dilution as calculated

by a plume model approved by the director, whichever is greater, unless the director determines that the more restrictive mixing zone or another definition of the mixing zone is more appropriate. In calculating the dilution at the boundary of the mixing zone, the discharger may use any of the various documented plume models and dispersion models appropriate for the discharge and approved by the director. Some of these models are referenced in the technical documents for EPA's ocean dumping regulations and the technical document for EPA's section 301(h) regulations.

Where the discharge contains a solid phase, the applicant will be required to perform the solid phase bioassay and bioaccumulation testing on the waste material in accordance with procedures described in *Ecological Evaluation of Proposed Discharge of Dredge Material into the Ocean Waters*, EPA/Corps of Engineers, July 1977. For example, if a bioassay analysis is required in the case of offshore oil and gas platforms, the solid phase bioassay would be conducted as a test on drilling muds and cuttings which are to be discharged.

Not all applicants may have to perform bioassay tests on their effluents. Applicants may submit bioassay analyses performed on other wastes if the applicant provides documentation to show that the composition of the waste analyzed typifies that which the applicant is discharging or intends to discharge.

(b) Monitoring Requirements—Where a pre-issuance determination regarding degradation of the marine environment cannot be made, § 125.123(d)(2) requires that a monitoring program be in place which is sufficient to assess the impact of the discharge on water, sediment, and biological quality including, where appropriate, analysis of the bioaccumulative and/or persistent impact on aquatic life of the discharge. This monitoring program may include effluent analysis, bioassay analysis and field studies. The technical document for EPA's section 301(h) regulations should provide support in developing such a monitoring program. It is not possible to make an *a priori* determination as to what constitutes an acceptable monitoring program. Site-specific conditions such as the size of the discharger's waste stream, the types of pollutants discharged, and the location of the discharge will play a role in determining what if any specific monitoring will be required under section 403(c) in addition to other NPDES monitoring requirements.

Section 125.123(d)(2) provides the director a flexible mechanism to develop such site-specific monitoring requirements. For example, a low volume discharger whose waste stream is unlikely to contain significant amounts of toxic pollutants will not be required in most cases to establish a monitoring program under these regulations. Similarly, a discharger of pollutants into an area of biological concern may be subject to more stringent monitoring requirements than one not discharging into such an area.

Monitoring programs, in some instances, may be coordinated for several dischargers. For example, with offshore oil and gas platforms, areawide monitoring programs for several dischargers may be the desirable monitoring approach. EPA headquarters has been active in assisting the regions in developing monitoring programs for offshore oil and gas exploration in areas of biological concern such as the Flower Garden Banks and Georges Bank. Those monitoring programs will serve as valuable guides for the development of additional monitoring programs for other areas of offshore oil and gas exploration and production. EPA headquarters will continue to play an active role in providing technical assistance in developing such monitoring programs.

(c) Other Permit Conditions—Under § 125.123(d)(3), the director may also require under other permit conditions on the discharge. For example, the director may require seasonal restrictions on the volume of wastes discharged where such restrictions are needed to assure protection of the marine environment. Seasonal restrictions may be necessary where the discharge is itself affected by seasonal conditions or where the biological community may become more sensitive to the impact of the discharge during certain seasonal conditions, such as during migration or spawning.

The director may require that the applicant perform bioaccumulation testing of the liquid and/or suspended particulate phase of the discharge where the director suspects such potential for bioaccumulation may exist, based on the nature of the pollutants discharged.

The director may also require process modifications, such as the substitution of less hazardous chemicals for those which are potentially harmful. He may also require process changes which would favor the recycling and reuse of potentially harmful pollutants. The Agency has recently established a task force to evaluate the discharges from offshore oil and gas exploration and production facilities and to evaluate alternate control strategies to mitigate the effects of such discharges, which include drilling muds and cuttings and produced water. Its recommendations may be used in drafting future requirements under section 403(c) authority.

The director may also require that diffuser systems for the discharger be sufficient to assure adequate dispersion of the waste stream.

7. The Administrative Process and Burden of Proof

Under the Act and this regulation, the director is responsible for making "reasonable judgments" on the preceding issues, and these judgments will be made on available information compiled in the administrative record of the permit issuance. As discussed above, this information may come from many sources including data submitted under the consolidated permit application form, environmental impact statements or section 301(h) variance applications. These guidelines do not require that all applicants submit specific information to support the section 403 determinations, and the director is encouraged to make use of existing information not prepared by applicants.

However, under the Clean Water Act, the Administrative Procedure Act, and EPA's consolidated permit regulations it is the applicant who is responsible for persuading the Agency that a permit should be issued. See 40 CFR 124.85(a)(1) and Opinion of the General Counsel No. 72. This obligation is particularly apparent with respect to applicants seeking permits to discharge into marine waters. Section 403(c)(2) requires that the director deny an NPDES permit application if there is insufficient information to make reasonable judgments under the guidelines. This means that the permit applicants should be prepared to submit sufficient information to support a determination to issue an NPDES permit.

Under the Agency's permit issuance procedures there is opportunity to submit information for the administrative record. An applicant or interested person who disputes any permit condition or tentative decision to deny an application must submit available information supporting their position during the public comment period. 40 CFR 124.13. In any subsequent evidentiary hearing on the permit, the Agency will have the burden of going forward to present its case supporting a challenged permit condition, but, at the conclusion of the Agency's presentation, the applicant or any other hearing participant has the burden of going forward to present its case. 40 CFR 124.85(a) (2) and (3). Moreover, the ultimate burden of persuading the

Agency to issue a permit remains at all times on the permit applicant.

V. Cost and Economic Impacts

Executive Order 12044, 43 FR 12661 (March 23, 1978), requires EPA and other agencies to perform Regulatory Analyses of certain regulations. EPA's plan for implementing Executive Order 12044, 44 FR 30988 (May 29, 1979), requires a Regulatory Analysis for major regulations involving annual compliance costs of $100 million or meeting other specified criteria. Where these criteria are met, EPA's implementation plan requires a formal Regulatory Analysis including an economic impact analysis and an evaluation of regulatory alternatives. The Agency has determined that none of the criteria for requiring a regulatory analysis has been exceeded and therefore, the promulgated regulations for ocean dischargers do not require a formal Regulatory Analysis. Nevertheless, EPA performed an analysis that does meet all the requirements of Executive Order 12044 and EPA's plan for its implementation.

In accordance with the requirements under section 2(d)(6) of Executive Order 12044, the Agency intends to review the effectiveness and need for continuation of the provisions contained in this action no more than five (5) years from the effective date of these regulations. In particular, we will solicit comments from affected parties with regard to actual costs incurred and other burdens associated with compliance and will also review environmental data to evaluate the effectiveness of the regulation after it has gone into effect.

EPA's economic analysis divides the affected dischargers under the proposed regulation into five subcategories: POTWs, industrial dischargers, electric utilities, federal facilities, and offshore oil and gas wells. This analysis has assessed unit price increases, production changes for industrial dischargers, and user cost increases at POTWs.

The total cost of compliance is expected to be $13 million in 1981 and to increase to $28 million by 1985, in 1980 dollars. Costs may increase in succeeding years. The Agency expects no significant economic impacts will result from this regulation.

1. POTWs

There are presently 102 POTWs discharging 2.1 billion gallons of effluent a day into the ocean. These POTWs are currently operating under EPA's NPDES regulations. Under these regulations POTWs may be required to monitor, perform benthic analyses, bioaccumulation tests and run further analyses of disposal alternatives in addition to those required under their present NPDES permit. The Agency performed an economic analysis to determine the potential costs and increases in user charges currently paid by households serviced by affected POTWs. EPA estimates that 47 POTWs will incur additional costs, due to their location and the size of their discharges, consisting of a first year cost of $1.2 million and an average annual cost of $.828 million thereafter.

For 46 of the 47 affected POTWs, user charges will increase between $.09 to $.83 per family per year. The impact analysis then compared these costs to median family incomes and found that no significant economic impacts would occur. However, the impact analysis has indicated that for one community user costs could increase by $33.00 per family per year.

Currently 36 POTWs subject to these regulations have applied for 301(h) variances. EPA has not yet begun to issue decisions on section 301(h) variance requests. However, much of the information generated for purposes of section 301(h) applications can be utilized in determining compliance with the requirements of these regulations. Furthermore, in recent years there has been a trend towards centralization of POTWs in many coastal areas. This continued centralization will reduce the number of ocean outfalls, thus lowering total monitoring and user costs.

2. Industrial Dischargers

Industrial dischargers will face the same type of compliance requirements as POTWs. Monitoring and compliance requirements for industrial dischargers are dependent on the particular geographic area as well as the composition and volume of the effluent. At the present time there are 74 industrial operations affected by this regulation, discharging approximately 212 million gallons per day of effluent. The Agency expects that small industrial plants discharging non-toxic pollutants will not be affected by this regulation. EPA estimates 46 dischargers will incur additional costs due to this regulation, with a first year cost of $4.72 million, and annual costs of $3.52 million in the following years.

An analysis was conducted for a sample of industrial dischargers on both the East and West Coasts to determine the potential price increases that could result due to this regulation. EPA estimates that average unit prices for products produced will generally increase less than .1 percent to comply with this regulation. No plant closures, unemployment or other significant economic impacts are expected due to these requirements.

3. Federal Facilities

At the present time there are 31 federal facilities affected by this regulation. These facilities are discharging approximately 124 million gallons per day of effluent into the ocean. The greater part of the total, 75 million gallons per day, originates from the strategic oil reserve construction site on the Texas coast. The remaining 49 million gallons a day are from a variety of small sources, e.g., Defense Department and Coast Guard installations. The Agency estimates that four federal facilities discharging 102 million gallons per day will actually incure additional costs from this regulation. The total annual costs for compliance under the proposed regulation is expected to be approximately $.476 million, with the largest proportion of this amount being related to the construction of the United States strategic oil reserve. EPA does not expect any significant economic impacts to occur due to expenditures by these facilities.

4. Electric Utilities

The Agency does not expect any significant costs to be incurred by electric utilities. Compliance with the present effluent limitation requirements and with regulations implementing section 316(a) of the Clean Water Act are expected to result in compliance with requirements in this regulation. The Agency expects that monitoring for chlorine discharges may be required at some facilities. However, the cost of such monitoring would not be significant, and no economic impacts are expected to occur.

5. Offshore Oil and Gas Operations

There are presently fewer than 30 oil and gas platforms which are expected to incur additional costs due to this regulation. The Agency estimates that 7,582 exploratory and production wells will be drilled between 1981–1985 with approximately 835 (11 percent) expected to incur additional costs resulting from compliance with this regulation. The Agency has based its assessment on the assumption that compliance with applicable NPDES permit requirements will generally result in compliance with these regulations for all oil and gas wells except those located in areas of biological concern. Wells that cannot meet the requirements of this regulation through compliance with their NPDES permit terms will be required to initiate

monitoring, testing, or changes in their discharge practices.

The Agency's evaluation of the economic effects of this regulation assumed the installation of "zero discharge" technologies in order to evaluate the maximum possible impact of these regulations. The cost of "zero discharge" varies according to geographic location, differences in weather conditions, water depths, biological communities, and other similar factors. The economic analysis groups wells into four regions—the Atlantic Ocean, Gulf Ocean, West Coast, and Alaskan waters.

The number of wells that would need to make expenditures beyond those required under existing NPDES permit requirements was estimated from Department of Interior data regarding current and future lease tracts in the ocean and from current trends in new drilling. The estimates here project activity from 1981–1985. Should the amount of new drilling increase or decrease beyond that time the annual cost of this regulation would increase or decrease proportionately.

It is estimated that the annual cost for offshore oil and gas operations locating in or near areas of biological concern will incur compliance costs ranging from seven to 23 million dollars per year between 1981 and 1985. Costs for various forms of monitoring are expected to be less. There are numerous alternatives which include process changes, mud substitution and shunting as potential compliance alternatives. A number of combinations are possible, depending on geographic location, water depth, temperature and specific biological life. For this reason only the worst case, "zero discharge" requirement is presented here.

The typical compliance cost per year for each geographical area for the no discharge alternative is presented below. Costs for operations in the Atlantic Ocean are expected to range between .6 and 6.9 million dollars per year, and Gulf of Mexico operations will face costs ranging between 1 and 3.6 million dollars per year. Operations located on the West Coast will face costs ranging between 4.3 and 12.4 million dollars per year. However, the California Ocean Plan requires outer continental shelf operations to conform to strict State requirements, which may reduce the incremental compliance costs under this regulation. Alaskan operations will face compliance costs ranging between 1.2 and 4.5 million dollars per year.

Oil and gas prices at the well head are not expected to be affected by this regulation, since compliance costs cannot be directly passed forward due to various price controls. However, the cost of this regulation may be manifested in reduced bids for new lease tracts. The net effect of this regulation would be a loss in future revenues to the federal government in the amount which this regulation costs the oil industry.

Dated: September 26, 1980.

Douglas M. Costle,

Administrator.

Appendix A—Public Comments

The following parties responded with comments regarding the February 12, 1980 Ocean Discharge Criteria postmarked on or before the April 28, 1980 close of the public comment period: Charles A. Lunsford, Commonwealth of Virginia, State Water Control Board; State of Hawaii, Dept. of Health; County of San Diego, Community Services Agency, Dept. of Sanitation & Flood Control; City of Los Angeles, California Dept. of Public Works; Menasha Corporation; National Manufacturing Company; Commonwealth of Virginia, State Water Control; County Sanitation Districts of Los Angeles County; Crown Zellerbach Environmental Services; Kaiser Aluminum & Chemical Corporation; Boise Cascade, Paper Group; Davies Hamakua Sugar Company; Dept. of Health, Education & Welfare, Public Health Service; Hawaiian Sugar Planters' Association; Hilo Coast Processing Company; Netarts-Oceanside Sanitary District; International Paper Company; State of Alaska, Dept. of Fish & Game; Commonwealth of Virginia, Hampton Roads Sanitation District; Marathon Oil Company, Production Operations; San Francisco Wastewater Program, City and County of San Francisco, California; U.S. Cape May County Municipal Utilities Authority, New Jersey; Dept. of the Army, South Atlantic Division, Corps of Engineers; State of California, Resources Agency, Dept. of Fish and Game; National Fisheries Institute, Inc., Natural Resources Defense Council, Inc., Sussex County Council, Georgetown, Delaware; American Paper Institute/ National Forest Products Association, Environmental Program, Houston Audubon Society; Texas Eastern Transmission Corporation; State of Delaware, Department of Natural Resources and Environmental Control, Division of Environmental Control; Department of the Air Force, Engineering and Serivce Center; Star-Kist Foods, Inc.; State of California, Resources Agency, State Water Resources Control Board; Office of the Assistant Secretary of Defense, Energy, Environment and Safety; the Ocean County Utilities Authority, New Jersey, National Wildlife Federation; Department of the Army, Office of the Chief of Engineers; Shell Oil Company; Texaco, Inc., American Cyanamid Company; Atlantic Richfield Company; E. I. du Pont de Nemours and Company, Inc.; Public Service Company of New Hampshire; Chevron U.S.A., Inc., Environmental Affairs; Commonwealth of Puerto Rico, Puerto Rico Aqueduct and Sewer Authority; Exxon Company, U.S.A.; Offshore Operators Committee, Southern California Edison Company; Virgin Islands Rum Industries, Ltd., Fried, Frank, Harris, Shriver & Kampelman; Chevron U.S.A. Inc., Pillsbury, Madison & Sutro; City of Watsonville, California; Conoco, Inc.; Conservation Law Foundation of New England, Inc.; Mobil Oil; Corporation; Western Oil & Gas Association; Alaska Lumber and Plup Company, Inc., Robertson, Monagle, Eastaugh & Bradley; American Petroleum Institute; Cody Biggs; Chemical Manufacturers Association, Covington & Burling; City of Skagway, Alaska, Robertson, Monagle, Eastaugh & Bradley; Columbia Gas System Service Corporation; Department of Energy; Gulf Oil Exploration & Production Company; National Food Processors Association; Pacific Legal Foundation; Phillips Petroleum Company; Tuna Research Foundation, Inc.; Union Oil Company of California; U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Environmental Research Laboratories; Natural Resources Defense Council, Inc.; U.S. Fish & Wildlife Service, Alaska Area Office; Utility Water Act Group, Hunton, & Williams; Department of Water & Power, the City of Los Angeles, California; U.S. Department of the Interior, Geological Survey.

The following parties responded with comments postmarked after the April 28, 1980 close of public comment period: U.S. Department of the Interior; Chevron, U.S.A., Pillsbury, Madison & Sutro; University of Southern Maine for State Planning Office, State of Maine; Monmouth County Board of Health, New Jersey; State of Maine, State Planning Office; Western Oil and Gas Association.

The following parties testified at the March 21, 1980 hearing: George P. Haley, Chevron USA; Frank Parker, Coordinator, Environmental and Government Affairs Chevron USA; Elizabeth F. Kroop, Counsel for the National Wildlife Federation; Walter J. Zizik, Project Coordinator, South Monmouth Regional Sewerage

Authority: Curt D. Rose, Manager, Aquatic Sciences Division, Energy Resources Company; Frank Melone, Southern California Edison Company; William A. Anderson, Attorney, Utility Water Act Group; Joseph F. Dietz, Coordinator of Environmental Affairs, San Diego Gas and Electric Company; Edward G. Gladbach, Civil Engineer, Department of Water and Power, City of Los Angeles; Peter Holmes, Research Assistant, Atlantic Coast Project, Natural Resources Defense Council.

Appendix B—Response to Public Comments

1. *Comment.* The Agency received several public comments questioning the accuracy of the inventory in the proposal and noting that there was uncertainty over the exact location of the baseline marking the landward boundary of the territorial seas, particularly in parts of Alaska, Florida, Puerto Rico, Oregon and Washington.

Response. Even where the baseline has not been plotted, there are available nautical charts for the various bays and harbors in question. The Office of the Geographer in the Department of state is responsible for the charts plotting closing lines across islands and shoreline markers depicting the baseline of the territorial sea. That office is assisted by the Interagency Baseline Committee, chaired by the Department of State, with other members coming from the National Oceanic and Atmospheric Administration, the Coast Guard, and the Departments of Justice and the Interior. The Committee meets several times a year to make baseline determinations.

To facilitate the ongoing implementation of section 403, EPA has identified dischargers whose coverage under that provision is in question, due to uncertainty concerning the location of the baseline. The Agency has submitted a written rquest to the Department of State and the Interagency Baseline Committee for a determination whether these dischargers are outside the baseline of the territorial seas and thus subject to section 403. EPA will continue to seek determinations when NPDES permits are issued, modified, or reissued, where there is doubt as to whether a discharger is within the purview of section 403.

2. *Comment.* One commenter stated that while it appeared that the Agency intended for coastal electric utilities to be subject to the regulations, these plants were not counted among the 71 land-based industrial dischargers.

Response. Electric utilities outside the baseline are covered by this regulation and EPA, in response to the comment, confirmed this with representatives of the affected industry during the public comment period. The Agency also extended the comment period by thirty days, at the request of this commenter and others, to allow additional time for submission of comments on the proposal.

The Agency has identified 25 covered plants, which are included in the cost and economic impact analysis for the regulation. In addition, EPA has updated the inventory of subject marine dischargers. As the preamble notes, the number of land-based dischargers subject to section 403 is limited. The updated inventory identifies 232 such dischargers, including 102 POTWs, 74 industrial facilities, 25 steam electric utilities, and 31 federal facilities. These figures do not include dischargers in Alaska whose location relative to the baseline defining the boundary of the territorial seas has not been established. However, the Agency believes that most of these dischargers are small and that any environmental or economic impacts would be minimal. The Agency also estimates that there are some 3,000 subject offshore oil and gas platforms.

3. *Comment.* A number of commenters stated that under the proposed regulation, ocean dischargers might be subject to more stringent controls and, accordingly, might incur higher costs than would dischargers into potentially more sensitive estuarine and freshwater systems where the assimilative capacity of the body of water may be less than in the oceans and the potential impact of pollution on the ecosystem greater.

Response. As the preamble to the regulation notes, the Clean Water Act limits the coverage of section 403 to dischargers into waters seaward of the baseline marking the territorial seas. This additional assurance of protection and its limitation to the waters of the territorial seas, the contiguous zone, and the oceans is, therefore, a matter of Congressional mandate. EPA has designed its regulations to provide this protection, as Congress has directed. As to the question of costs, the Agency anticipates that in most cases, technology-based effluent limitations required under other provisions of the Act will be adequate in themselves to afford the necessary protection for the marine environment. As to freshwater and estuarine systems, the Agency agrees that these waters must be protected also; the statutory authority to accomplish this, however, rests in other sections of the Act and in other environmental statutes.

4. *Comment.* A number of commenters suggested that the ocean discharge criteria should merely have the effect of guidelines, rather than regulatory requirements, and should provide flexibility and allow for discretion on the part of the director to apply appropriate portions of the guidelines to the situation of an individual discharger.

Response. As noted in the preamble, these regulations, although from time to time described as "guidelines" or "criteria" to avoid repetition, establish mandatory requirements authorized by section 403(c). Whatever the terminology, they have the effect of mandatory regulations because, at any time that promulgated guidelines are in effect, no NPDES permit may be issued "except in compliance with such guidelines." Nevertheless, the regulation provides the permit writer flexibility in tailoring information requests and permit conditions to the circumstances of individual dischargers, based on local conditions.

5. *Comment.* Several commenters expressed the concern that while the proposed regulation required a permit applicant to demonstrate to the director that its discharge would have no unreasonable adverse impact on the environment, or that adequate toxics control and monitoring programs were in place, the proposal failed to tell the applicant how to make such a demonstration.

Response. In order to ensure that applicants will receive adequate guidance, the final regulation has been clarified to require that the director inform an applicant of any specific information that must be supplied. In addition, in an attempt to minimize the information collection obligation of applicants, the final regulation provides that the director may consider information already available to him in making the determinations required under § 125.123(a), (b), or (c).

6. *Comment.* Several commenters suggested that, for the territorial seas, the purposes of section 403 were being served already by State water quality standards required under section 303 and by technology-based effluent limitations under sections 301 and 304 of the Clean Water Act. Another commenter stated that pretreatment programs required under section 307 should also ensure protection of the marine environment.

Response. The Agency agrees that State water quality standards, technology-based limitations and pretreatment programs are all necessary to protect the marine environment. While in most instances discharges in compliance with such standards, limits, and programs will also be determined to pass the "no unreasonable degradation" test in these regulations, there may be

instances where this will not be the case. For example, there may be instances where no State water quality standards have been established for specific pollutants being discharged. Further, State water quality standards do not generally apply beyond the limits of the territorial seas, while the section 403 criteria apply also to the contiguous zone and the oceans. In addition, there may be instances in which technology-based controls will not be sufficient to assure protection of a particular marine environment, necessitating more stringent controls to assure that the section 403 criteria are met. Such may also be true of pretreatment programs; while the director may consider the effectiveness of a given pretreatment program in making the "unreasonable degradation" determination, he should not assume that the existence of pretreatment ensures protection of the marine environment for purposes of section 403.

7. *Comment.* Several commenters stated that while the proposed regulation differentiated special "areas of biological sensitivity" to assure those areas were not adversely affected, the regulation failed to adequately define what constitutes such areas. Another commenter suggested that the term should be replaced by "areas of biological concern" because "biological sensitivity" connotes a narrow concern for unique or fragile ecosystems. The commenter suggested that the emphasis of the regulation should be on unwarranted ecological damage regardless of the biological sensitivity of the area.

Response. The scope of this regulation is broader than protecting only those areas that are termed "sensitive"; as in the proposed regulation, these guidelines seek to prevent unreasonable degradation of the marine environment regardless of where the discharge occurs. Although the regulation no longer attempts to classify areas as "sensitive" or "nonsensitive", the location of the discharge is an important element in determining the level of control necessary to prevent such degradation. Section 125.122 identifies for the director a number of factors relating to the biology of the local community which are important in assessing the impact of a discharge.

8. *Comment.* A substantial number of comments submitted on behalf of various dischargers suggested that the dischargers in question—including small POTWs, electric utilities, seafood processors, Alaskan logging operations, and offshore oil and gas exploration and production wells—should be exempt from the requirements of section 403. Several commenters made the argument, in some cases based on submissions of technical data and reports, that their discharges already were subject to controls adequate to protect the marine environment. Some stated that their discharges resulted in only *de minimus* effects on the environment. Some stated that compliance with various provisions of the proposal would result in economic hardship.

Response. EPA has concluded that there is no basis for categorically exempting classes of subject dischargers from the coverage of section 403. While the data submitted by some commenters may be useful in determining whether a particular discharge will meet the "unreasonable degradation" test, it does not provide a basis for such a blanket exemption. However, while a permit writer is not precluded from seeking additional site-specific information, the submission of large quantities of data for particular dischargers or classes of dischargers makes it unlikely that a permit writer will find it necessary to require these applicants to submit any substantial quantity of additional data. Similarly, in the cases of small POTWs and others where the discharge is expected to have only a minimal impact, the flexibility which the final regulation provides will allow the permit writer to take this situation into account, rather than mandating a rigid across-the-board application of all requirements, with their associated costs.

9. *Comment.* Several commenters suggested that POTWs granted section 301(h) variances from secondary treatment requirements should be exempt from section 403 because of significant similarities in the two provisions. Another commenter, however, stated that section 301(h) contains no analogue to section 403(c)(1)(F) or (G) and asserted that toxic pollutants are not adequately controlled under section 301(h).

Response. Despite differences in statutory language, sections 403 and 301(h) share similar objectives in seeking to assure protection of the marine environment, and the respective determinations whether those objectives have been met under each provision is based on similar information. Section 301(h)(2) requires that a successful applicant for a variance demonstrate, among other things, that "such modified requirements will not interfere with the attainment or maintenance of that water quality which assure protection of public water supplies and the protection of shellfish, fish, and wildlife, and allows recreational activities in and on the water." Section 125.61 of EPA's section 301(h) regulations requires full and detailed descriptions of the physical characteristics of the discharge, its biological impact on the marine environment, and its impact on public water supplies and recreation. Given therefore that a successful section 301(h) applicant will have collected and presented substantial amounts of data on the effect of its discharge on the marine environment, including its inhabitants and uses, the final ocean discharge regulations provide that a successful section 301(h) demonstration creates a rebuttable presumption that an applicant will satisfy the section 403(c) guidelines as well. While a permit writer is not precluded from placing additional requirements on such an applicant under these regulations, it is unlikely that this will be necessary in light of the through-going demonstration the applicant has made for purposes of section 301(h).

This approach is consistent with legislative history to the effect that section 301(h) applicants must comply also with section 403. This language indicates that Congress did not intend for section 403 to become a dead letter with the subsequent enactment of section 301(h). Unlike the approach of those commenters who sought to make compliance with section 403 automatic for an applicant who had obtained a section 301(h) variance, the "rebuttable presumption" approach does not treat section 403 as redundant. Nor, however, does it impose a redundant data-gathering task on successful section 301(h) applicants either, taking account as it does of the unmistakable similarities in the showings required under the two provisions.

The Agency disagrees with the comment that the toxic control provisions of the section 301(h) regulations are not adequate. Moreover, if a permit writer determines that toxic pollutants in the discharge of a successful section 301(h) applicant are not adequately controlled for purposes of section 403, he can require additional controls or, if necessary, require zero-discharge permit terms for those pollutants.

10. *Comment.* Several commenters suggested that coastal steam electric utility plants granted a variance under section 316(a) of the Act should be exempt from demonstrating compliance with section 403, on the grounds that the demonstration necessary for obtaining a section 316(a) variance provides the requisite assurance that the marine environment is protected for purposes of section 403(c).

Response. To obtain a section 316(a) variance, an applicant must demonstrate

that effluent controls on its thermal discharge will be sufficient to assure the protection and propagation of a balanced indigenous population of shellfish, fish and wildlife in and on the water. The Agency agrees that in most cases the demonstration required of a successful section 316(a) applicant will be sufficient to allow the permit writer to conclude that there will be no unreasonable degradation of the marine environment due to excess heat. While on the reasoning set out in the previous response successful section 316(a) applicants will not be exempt from section 403, the regulation provides that a successful section 316(a) application creates a rebuttable presumption of compliance with section 403(c) for the thermal component of the discharge.

11. *Comment.* Commenters also suggested that those publicly-owned treatment works which installed secondary treatment should be exempt from requirements under section 403.

Response. Limitations established pursuant to section 403 are a supplement to technology-based limitations such as secondary treatment for POTWs, and no class of a discharger is exempt from compliance with these regulations. However, it is likely that secondary treatment will generally be adequate to satisfy section 403 requirements where there is adequate pretreatment by industrial sources and where the POTW is not discharging into areas of biological concern.

12. *Comment.* As noted above, some commenters asserted that the ocean discharge criteria should merely be guidelines providing information to the permit writer. Other commenters, however, stated that the guidelines should require that a discharge pass a quantitative test, such as the bioassay requirements used in the ocean dumping regulations, and comply with State and EPA water quality criteria as a prerequisite to permit issuance.

Response. The Agency has revised the proposed regulation to allow necessary flexibility to the director in assessing both the impact of a discharge and permit limitations. However, the regulation does impose minimum permit limitations, including a bioassay-based limitations, in areas where the long range impact of a discharge is not fully understood. This approach should provide certainty and consistency in permit limitations in areas where the determinations by the permit writer would be the most difficult and complex.

Discharges into the territorial seas must comply with any applicable state water quality criteria. However, the Act generally does not provide for the application of these criteria to the contiguous zone and oceans. Although the Act establishes a complete water quality program for State waters based on designated uses and supporting criteria, no such scheme exists for marine waters beyond State jurisdiction. The 403(c) regulation is consistent with the Agency policy outlined in the section 301(h) regulations (44 FR 34810–34811), and will utilize water quality criteria published pursuant to section 304(a)(1), as they are developed, as a basis for assessing the environmental impact of such pollutants.

13. *Comment.* Several commenters asserted that, under the proposed regulation, no predischarge determination was required by the director to assure that the marine environment was protected. Instead, commenters stated, the proposed regulation relied on post discharge monitoring.

Response. Under the final regulations, no discharge of pollutants may be authorized unless, before permit issuance, the director has sufficient information to make a reasonable determination that there will be no irreparable harm to the environment while monitoring is undertaken to determine if there will be unreasonable degradation. In addition the permit must specify certain mandatory limitations.

14. *Comment.* The Agency received numerous comments regarding the monitoring requirements outlined in section 125.127 of the proposed regulation. The major issue raised was that the monitoring requirements should be as flexible as possible providing an applicant a clear description of the information he must provide. Commenters suggested that the rigor of the monitoring program should be tailored to site-specific conditions such as the nature and location of the discharge. In addition, a number of commenters stated that compliance with the proposed monitoring requirements would result in severe economic hardship for small dischargers. It was suggested that the latter, especially small POTWs, be exempted from the monitoring requirements.

Response. As discussed in the preamble, these regulations have been revised to allow the permit writer to request from the applicant only that information necessary to make judgments required by the guidelines. In some cases this will involve monitoring programs, and the director will work with the applicant in identifying specific information that must be supplied as part of the permit application process.

Since no discharge may be allowed which would result in unreasonable degradation of the marine environment, and since the permit writer must be afforded the means to make the necessary determinations under these regulations, EPA has concluded that it may not exempt categories of dischargers, even small dischargers, from monitoring requirements as an initial matter. Nevertheless, the final regulations do not require monitoring in all cases, and, where monitoring is necessary, provide for flexibility in fashioning site specific requirements. Although any monitoring that may be necessary will depend on the nature and location of the discharge in question, small dischargers generally are not expected to incur significant economic costs as a result of this regulation.

15. *Comment.* Several commenters suggested that, in light of the similarities between section 403(c) of the Clean Water Act and section 102(a) of the Marine Protection, Research and Sanctuaries Act, the ocean discharge criteria should be similar to the ocean dumping regulations.

Response. EPA recognizes that in section 403(c) of the Clean Water Act and section 102(a) of the Marine Protection, Research, and Sanctuaries Act, Congress adopted similar although not identical provisions. Hence, in the regulations implementing the respective statutes, similar criteria may be appropriate.

EPA first promulgated ocean dumping criteria in 1973; those criteria were amended in 1977. Initially, the regulations served as joint regulations for the CWA and the MPRSA. Since promulgation of the ocean dumping regulations, however, EPA has received a number of comments based on those regulations. In addition, increasing data has become available in respect to the environmental impact of disposing of material at various locations in the ocean, by various methods.

The ocean discharge regulations being promulgated today are based on the latest data and information available to EPA, and the Agency believes these regulations are consistent with the CWA and with current scientific and technical knowledge. Various factors, including the MPRSA comments and the new data, suggest that it may now be appropriate for EPA to review the ocean dumping regulations as well. Such a review may provide further insights on an appropriate overall approach for protecting the ocean; and inconsistencies which may exist between the current sets of regulations can be resolved in the context of that action. However, in addition to any statutory distinctions, differences in the manner of disposal and the types of pollutants discharged may warrant

different regulatory approaches under these two statutes.

16. *Comment.* Several commenters suggested that dischargers who were not causing unreasonable degradation of the marine environment should not be required to assess the availability of alternatives.

Response. The proposed regulation has been modified to require the assessment of reasonable alternatives only where the director cannot determine whether the discharge will cause unreasonable degradation of the marine environment. In cases where the director determines that the discharge will not cause such degradation, an NPDES permit may be issued notwithstanding the availability of an alternative to ocean disposal.

Although the Clean Water Act contains as an ultimate goal the complete elimination of the discharge of pollutants, the water quality provisions of the Act, including sections 303 and 403, do not require that discharges into either inland or offshore waters be prohibited in the absence of unreasonable water quality impacts.

17. *Comment.* Several commenters expressed concern that the requirements for general permits, specified in section 125.129 of the proposed regulation, were not consistent with the requirements for individual permits.

Response. Section 125.129 has been deleted from the final regulation, and the director is required to make the same determinations when issuing either general or individual permits.

18. *Comment.* A few commenters objected to language in the preamble of the proposal to the effect that the permitting authority would be free to draw on his own knowledge of conditions in the vicinity of an outfall in determining whether a discharge adversely affected the marine environment. These commenters expressed the "due process" concern that this language allowed the permitting authority to issue permits on the basis of information not made known to the permittee and not in the administrative record.

Response. The language in question does not appear in the final regulation or in its preamble. While the regulation provides that the director make the determinations under § 125.123(a), (b), or (c) on the basis of "available information," that language was added in response to the suggestion of numerous commenters that permit applicants should not be required for purposes of Section 403 to resubmit data which was already available to the permit writer.

19. *Comment.* EPA received comment to the effect that the reference in the proposed regulations to schedules allowing additional time for compliance with Section 403 requirements should be limited to existing dischargers, consistent with the provisions of the NPDES regulations.

Response. Section 125.123(d)(3) of the final regulation provides for "schedules of compliance for existing dischargers," as suggested above.

20. *Comment.* The Agency received several comments regarding the mixing zone analysis as described in § 125.123. Some commenters suggested that the models identified by the Agency in the proposal would not be appropriate for all types of discharge. Other commenters suggested that the Agency should use the ocean dumping mixing zone definition.

Response. The mixing zone analysis in the final regulations is intended for use in calculating whether the limiting permissible concentration is violated in instances where bioassay analysis is required. The proposed regulation required a mixing zone analysis for all dischargers to assure that, following initial dilution, the discharge was dispersed so as not to adversely affect areas of biological sensitivity. As noted previously, this requirement has been deleted from the final regulation. In the final regulation, a mixing zone analysis is required only in those instances where the director cannot determine that unreasonable degradation will not occur and where a bioassay analysis is required.

The mixing zone definition in these regulations is consistent with the ocean dumping mixing zone definition identified in the EPA/Corps of Engineers technical manual, with some modifications to account for the differences in the nature of discharged wastes versus those which are dumped. The ocean dumping mixing zone was devised primarily to facilitate analysis of impacts from intermittent discharges from moving vessels, whereas the 403(c) regulations are intended to facilitate analysis of continuous discharges from stationary sources. The final regulation also allows the discharger to use alternative methods for determining the mixing zone where scientific evidence demonstrates they are appropriate and where EPA concurs.

A new Subpart M is added to read as follows:

Subpart M—Ocean Discharge Criteria

Sec.

125.120 Scope and purpose.
125.121 Definitions.
125.122 Determination of unreasonable degradation of the marine environment.
125.123 Permit requirements.
125.124 Information required to be submitted by applicant.

§ 125.120 Scope and purpose.

This subpart establishes guidelines for issuance of National Pollutant Discharge Elimination System (NPDES) permits for the discharge of pollutants from a point source into the territorial seas, the contiguous zone, and the oceans.

§ 125.121 Definitions.

(a) "Irreparable harm" means significant undesirable effects occurring after the date of permit issuance which will not be reversed after cessation or modification of the discharge.

(b) "Marine environment" means that territorial seas, the contiguous zone and the oceans.

(c) "Mixing zone" means the zone extending from the sea's surface to seabed and extending laterally to a distance of 100 meters in all directions from the discharge point(s) or to the boundary of the zone of initial dilution as calculated by a plume model approved by the director, whichever is greater, unless the director determines that the more restrictive mixing zone or another definition of the mixing zone is more appropriate for a specific discharge.

(d) "No reasonable alternatives" means: (1) No land-based disposal sites, discharge point(s) within internal waters, or approved ocean dumping sites within a reasonable distance of the site of the proposed discharge the use of which would not cause unwarranted economic impacts on the discharger, or, notwithstanding the availability of such sites,

(2) On-site disposal is environmentally preferable to other alternative means of disposal after consideration of: (i) The relative environmental harm of disposal on-site, in disposal sites located on land, from discharge point(s) within internal waters, or in approved ocean dumping sites, and

(ii) The risk to the environment and human safety posed by the transportation of the pollutants.

(e) "Unreasonable degradation of the marine environment" means: (1) Significant adverse changes in ecosystem diversity, productivity and stability of the biological community within the area of discharge and surrounding biological communities.

(2) Threat to human health through direct exposure to pollutants or through consumption of exposed aquatic organisms, or

(3) Loss of esthetic, recreational, scientific or economic values which is unreasonable in relation to the benefit derived from the discharge.

§ 125.122 Determination of unreasonable degradation of the marine environment.

(a) The director shall determine whether a discharge will cause unreasonable degradation of the marine environment based on consideration of:

(1) The quantities, composition and potential for bioaccumulation or persistence of the pollutants to be discharged;

(2) The potential transport of such pollutants by biological, physical or chemical processes;

(3) The composition and vulnerability of the biological communities which may be exposed to such pollutants, including the presence of unique species or communities of species, the presence of species identified as endangered or threatened pursuant to the Endangered Species Act, or the presence of those species critical to the structure or function of the ecosystem, such as those important for the food chain;

(4) The importance of the receiving water area to the surrounding biological community, including the presence of spawning sites, nursery/forage areas, migratory pathways, or areas necessary for other functions or critical stages in the life cycle of an organism.

(5) The existence of special aquatic sites including, but not limited to marine sanctuaries and refuges, parks, national and historic monuments, national seashores, wilderness areas and coral reefs;

(6) The potential impacts on human health through direct and indirect pathways;

(7) Existing or potential recreational and commercial fishing, including finfishing and shellfishing;

(8) Any applicable requirements of an approved Coastal Zone Management plan;

(9) Such other factors relating to the effects of the discharge as may be appropriate;

(10) Marine water quality criteria developed pursuant to section 304(a)(1).

(b) Discharges in compliance with sections 301(g), 301(h), or 316(a) variance requirements or State water quality standards shall be presumed not to cause unreasonable degradation of the marine environment, for any specific pollutants or conditions specified in the variance or the standard.

§ 125.123 Permit requirements.

(a) If the director on the basis of available information including that supplied by the applicant pursuant to § 125.124 determines prior to permit issuance that the discharge will not cause unreasonable degradation of the marine environment after application of any necessary conditions specified in § 125.123(d), he may issue an NPDES permit containing such conditions.

(b) If the director, on the basis of available information including that supplied by the applicant pursuant to § 125.124 determines prior to permit issuance that the discharge will cause unreasonable degradation of the marine environment after application of all possible permit conditions specified in § 125.123(d), he may not issue an NPDES permit which authorizes the discharge of pollutants.

(c) If the director has insufficient information to determine prior to permit issuance that there will be no unreasonable degradation of the marine environment pursuant to § 125.122, there shall be no discharge of pollutants into the marine environment unless the director on the basis of available information, including that supplied by the applicant pursuant to § 125.124 determines that: (1) Such discharge will not cause irreparable harm to the marine environment during the period in which monitoring is undertaken, and

(2) There are no reasonable alternatives to the on-site disposal of these materials, and

(3) The discharge will be in compliance with all permit conditions established pursuant to paragraph (d) of this section.

(d) All permits which authorize the discharge of pollutants pursuant to paragraph (c) of this section shall: (1) Require that a discharge of pollutants will: (A) following dilution as measured at the boundary of the mixing zone not exceed the limiting permissible concentration for the liquid and suspended particulate phases of the waste material as described in section 227.27(a) (2) and (3), section 227.27(b), and section 227.27(c) of the Ocean Dumping Criteria; and (B) not exceed the limiting permissible concentration for the solid phase of the waste material or cause an accumulation of toxic materials in the human food chain as described in sections 227.27 (b) and (d) of the Ocean Dumping Criteria;

(2) Specify a monitoring program, which is sufficient to assess the impact of the discharge on water, sediment, and biological quality including, where appropriate, analysis of the bioaccumulative and/or persistent impact on aquatic life of the discharge;

(3) Contain any other conditions, such as performance of liquid or suspended particulate phase bioaccumulation tests, seasonal restrictions on discharge, process modifications, dispersion of pollutants, or schedule of compliance for existing discharges, which are determined to be necessary because of local environmental conditions, and

(4) Contain the following clause: In addition to any other grounds specified herein, this permit shall be modified or revoked at any time if, on the basis of any new data, the director determines that continued discharges may cause unreasonable degradation of the marine environment.

§ 125.124 Information required to be submitted by applicant.

The applicant is responsible for providing information which the director may request to make the determination required by this subpart. The director may require the following information as well as any other pertinent informaton:

(a) An analysis of the chemical constituents of any discharge;

(b) Appropriate bioassays necessary to determine the limiting permissible concentrations for the discharge;

(c) An analysis of initial dilution;

(d) Available process modifications which will reduce the quantities of pollutants which will be discharged;

(e) Analysis of the location where pollutants are sought to be discharged, including the biological community and the physical description of the discharge facility;

(f) Evaluation of available alternatives to the discharge of the pollutants including an evaluation of the possibility of land-based disposal or disposal in an approved ocean dumping site.

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Key to Abbreviations

AA	achieved availability; acrylic acid; adenylic acid; affected areas; amino acid; amino aceton
AC	acre
ac	acres
ADI	acceptable daily intake
APHA	American Pharmaceutical Assoc.; American Public Health Assoc.
ATP	adenosine triphosphatae
ATP ase	adenosine triphosphatase (an enzyme)
BaP	benzo-a-pyrene
BAT	best available technology
BCF	bioconcentration factor
BHC	benzene hexachloride (pesticide); Better Heating-Cooling Council
BNA	Bureau of National Affairs
BOD	biochemical oxygen demand
BOR	Bureau of Outdoor Recreation (Dept of Interior); Bureau of Reclamation (Dept of Interior)
BP	benzopyrene
BPCT	best practicable control technology
BPWTT	best practicable waste treatment technology
CB	chlorinated benzenes
cc	cubic centimeter
CEPEX	Controlled Ecosystem Pollution Experiment(s)
CFR	carbon-film resistor, Code of Fed. Regul.
cfs/ft	cubic feet per second
cfs	cubic feet per second
CFS	cubic feet per second
CGS	centimeter-gram-second
cm	centimeters
cm/s	centimeters/second
cm/second	centimeters per second
cms	cubic meters per second
CO_2	carbon dioxide

COD	chemical oxygen demand
CWA	Clean Water Act
cu mi	cubic miles
DAS	dumping at sea
DDE	dichlorodiphenyl dichloroethylene (pesticide residue)
DDT	dichlorodiphenyl trichloroethane (insecticide)
DIN	dissolved inorganic nitrogen
DLVO	Derjaguin, Landau, Verwey and Overbeek
DNA	deoxyribonucleic acid
DO	dissolved oxygen
DON	dissolved organic nitrate
E_H	electrochemical redox potential
EDTA	ethylenediamine-tetraacedic acid
ENR	Eng. Narrative Report
EPA	Environmental Protection Agency
EPN	ethyl para-nitrophenyl phenylphophonothioate (insecticide)
Eq.	equation
F	feet
FAO	Food and Agriculture Organization
FDA	Food and Drug Administration
ft	feet
FWPCA	Federal Water Pollution Control Administration (Dept of Interior)
g	grams
GAC	granular activated carbon
GAO	General Accounting Office
GCD	greatest common denominator
GESAMP	Group of Experts on the Scientific Aspects of Marine Pollution
GM	gram
gm/cc	grams per cubic centimeter
GPCD	gallons per capita per day
gpcd	gallons per capita per day
GPD	gallons per day
gpd	gallons per day
H	hour
HA	hectares
ha	hectares
HEM	hexane extractable materials
HR	hour
IARC	International Agency for Research on Cancer
IN	inch
in	inch
ISC	insoluble collagen (biochemistry)
JOS	Joint Outfall System
JWPCP	Joint Water Pollution Control Plant
K	knots
km	kilometer
km^2	square kilometer
km^3	cubic kilometers

km^3/yr	cubic kilometers per year
KT	knots
kW	kilowatt
l/yr	liters per year
LA/OMA	Los Angeles/Orange County metropolitan area
LACSD	Los Angeles County Sanitation Districts
lbs	pounds
LCD	least (or lowest) common denominator
lcd	least (or lowest) common denominator
LMHH	low molecular weight halogenated hydrocarbons
M	mile
M	Rhodamine WT dye
m	meter
m^2	square meters
m^2/s	square meters per second
m^2/sec^2	square meters per second squared
m^3/d	cubic meters per day
m^3/s	cubic meters per second
m tons	metric tons
m/sec	meters per second
MAFF	Ministry of Agriculture, Fisheries and Food (UK)
MATC	maximum acceptable toxicant concentration
MESA	Marine Ecosystems Analysis
METRO	Metropolitan West Point Treatment Plant -Seattle
METRO	The Municipality of Metropolitan Seattle
MGD	million gallons per day
mg	milligram
mg/l	milligrams per liter
mg/wet kg	milligrams per wet kilogram
mi	mile
mi^2	square miles
MLLW	mean lower low water
ml	milliliters
mm	millimeter
mm/s	millimeters per second
MPH	miles per hour
mph	miles per hour
MPN	most probable number
MPRSA	Marine Protection Research and Sanctuaries Act
MT	metric tons (1000 kg)
N	nitrogen
n mi	nautical miles
NACOA	National Advisory Committee on Oceans and Atmosphere
NAS	National Academy of Sciences
NCI	National Cancer Institute
NIOSH	National Institute for Occupational Safety and Health (HEW)
nm	nanometers

NMFS	National Marine Fisheries Service
NMI	nautical miles
NOAA	National Oceanic and Atmospheric Administration
NPDES	National Pollution Discharge Elimination System
NSF	National Science Foundation
NTAC	National Technical Advisory Committee
NTU	Naval Training/Toxicology Unit Nephelometric Turbidity Unit (Analyti. chemistry)
O & M	operation and maintenance; organization & management; organization & methods
ORCA	Office of Ocean Resources Coordination and Assessment
PBOD	phytoplankton biochemical oxygen demand
PCB	polychlorinated biphenyls
PDOD	phytoplankton dissolved oxygen deficit
PDOD	eutrophication oxidation potential
pE	electron ion activity
PFU	plaque forming units
pH	hydrogen ion activity
PLA	Port of London Authority (UK)
POTWs	Publically Owned Treatment Works
PPB	parts per billion
ppb	parts per billion
PPM	parts per million
ppm	parts per million
PPT	parts per trillion
PSP	paralytic shellfish poison
RCRA	Resource Conservation and Recovery Act
RMS	radian means per second
rms	radian means per second
RPD	redox potential discontinuity
s	second
SCCWRP	Southern California Coastal Waters Research Project
SQ	square
SS	suspended solids
STP	Sewage Treatment Plant
sec	second; secondary
sq	square
sq km	square kilometers (km^2)
sq mi	square miles
t	time (IUPAC)
t/d	tons per day
TCE	trichloroethylene
TDE	tetrachlorodiphenylethane (also DDD) (insecticide)
TOC	total organic carbon
tonnes	metric tons
$TRCL_2$	(average) total residual chlorine
TSCA	Toxic Substances Control Act

TSS	total suspended solids
UK	United Kingdom
US	United States
USDA	US Department of Agriculture
USDHEW	US Department of Health, Education and Welfare
USEPA	United States Environmental Protection Agency
UV	ultraviolet; ultravisible; underwater vehicle
WBOD	waste biochemical oxygen demand
WDOD	waste oxidation potential
WDOE	Washington Dept. of Ecology
WHO	World Health Organization
WRPs	water renovation and reclamation plants