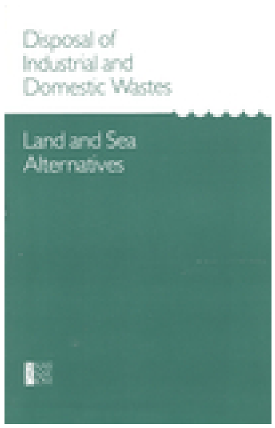

Disposal of Industrial and Domestic Wastes: Land and Sea Alternatives

ISBN: 0-309-56709-2, 210 pages, 6 x 9, (1984)

Disposal of Industrial and Domestic Wastes

Land and Sea Alternatives

Board on Ocean Science and Policy
Commission on Physical Sciences, Mathematics, and Resources
National Research Council

NATIONAL ACADEMY PRESS
Washington, D.C. 1984

NATIONAL ACADEMY PRESS 2101 Constitution Avenue, N.W. Washington, D.C. 20418

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Library of Congress Catalog Card Number 84-61446
International Standard Book Number 0-309-03484-1

Printed in the United States of America

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Introduction

As population increases, the burden of disposing of society's wastes increases with it. Along with the land and the atmosphere, the oceans present themselves as possible sites to receive some portion of these wastes, but a number of current U.S. laws impose constraints on ocean disposal. These laws include the Federal Water Pollution Control Act, (33 U.S.C. 1251 et seq.), the Marine Protection, Research and Sanctuaries Act of 1972, (33 U.S.C. 1401 et seq.), the Clean Air Act, (42 U.S.C. 6901 et seq.), and the Safe Drinking Water Act of 1974, (42 U.S.C. 300 et seq.). There are studies indicating that the coastal oceans are underused with respect to the accommodation of societal wastes (Assimilative Capacity of U.S. Coastal Waters for Pollutants. Crystal Mountain, Washington, Workshop Proceedings. NOAA Environmental Research Laboratories, 1979, 284 pp.; The Role of the Ocean in a Waste Management Strategy, National Advisory Committee on Oceans and Atmosphere, 1981). The concept that the marine environment can be effectively used for waste disposal, however, has not been validated in a comprehensive assessment with land and atmospheric options. Although there have been studies of multimedia management for waste disposal (Multimedium Management of Municipal Sludge, National Academy of Sciences, Washington, D.C., 1978, 187 pp.), the development of strategies for a multidisciplinary approach coupling both natural and social sciences remains inadequate.

It became clear to the Ocean Policy Committee (OPC) of the National Research Council that the time was right for a more thorough assessment of the current state of knowledge of waste management. Specifically, attention should focus on land versus sea options where (1) there are adequate data for rational assessment and (2) scientific, engineering, economic, social (including risk assessment), and political factors are all brought into perspective.

Therefore, in April 1982, OPC appointed a steering committee to organize a workshop devoted to consideration of the kinds of information needed in evaluating disposal options. The steering committee was cochaired by Edward D. Goldberg of the Scripps Institution of Oceanography and Stanley I. Auerbach of the Oak Ridge National Laboratory. The other members included Norman H. Brooks, California Institute of Technology; Judith M. Capuzzo, Woods Hole Oceanographic Institution; James A. Crutchfield, University of Washington; David A. Deese, Boston College; William F. Garber, Bureau of Sanitation, Los Angeles; George A. Jackson, Scripps Institution of Oceanography; and Richard F. Schwer, E.I. du Pont de Nemours & Company. The steering committee agreed that the workshop should be organized into eight panels: public policy, economics, risk assessment, marine sciences, biological effects, land disposal, sewage sludge, and industrial wastes. In addition to organizing the workshop, the steering committee identified information needs for assessing, for a given site and a given waste material, the options of land, sea, and air disposal and the environmental, economic, and regulatory criteria for selection among those options. The workshop, entitled “Land, Sea, and Air Options for the Disposal of Industrial and Domestic Wastes,” was held January 16-21, 1983, in Napa Valley, California. Financial support for the project was provided by the U.S. Environmental Protection Agency (EPA) and the National Oceanic and Atmospheric Administration (NOAA).

The steering committee agreed that it would be potentially most productive for the workshop to consider two cases: (1) the 106-Mile Ocean Waste Disposal Site (Dumpsite 106) off the New Jersey coast, the largest U.S. ocean site for disposal of industrial wastes, and (2) the sewage sludge disposal problem in Los Angeles and Orange counties, California.

In the first case, Dumpsite 106 has been a site for regulated disposal of industrial and some municipal wastes since 1972. Industries have assessed various land disposal schemes, as alternatives to current dumping at sea. The committee agreed that an evaluation of the data that these industries have assembled in their assessments of land versus sea disposal, complemented by dumpsite studies carried out by NOAA and EPA, would provide a springboard for the workshop deliberation.

The second case involves the disposal of sewage sludge in the Southern California Bight region. Impending changes in sludge generation and disposal have necessitated extensive studies of land disposal options. Legal restraints have limited complementary ocean disposal options.

Fifty-five individuals (see Appendix A) from the social and natural sciences participated in the workshop. Background papers addressing the goal of the workshop were prepared by some participants and were distributed prior to the workshop as a basis for discussions.

Six panels (the economics and risk assessment panels were combined with the public policy panel) considered the major relevant issues and presented their findings at plenary sessions. The panel session topics and their chairmen were public policy, Judith T. Kildow; marine sciences, George A. Jackson; land disposal, Stanley I. Auerbach; biological effects, Judith M. Capuzzo; industrial wastes, Richard F. Schwer; and sewage sludge, William F. Garber.

The participants were drawn from industry, government, academia, and public interest groups. Despite their different areas of professional activity, they were able to reach a consensus. All participants contributed to the written reports of their respective panels. The panel reports were presented to the workshop in plenary session on the last day, and revisions resulting from plenary session were incorporated into the panel reports by the chairmen of each panel. The revised panel reports were circulated to participants for review and comment. The panel chairmen, working with the steering committee, revised the panel reports in response to the extensive comments made by the participants at the workshop and outside reviewers. The result of this process is a report consisting of the following six chapters, which reflect the content and deliberations of the panel sessions at the workshop.

Executive Summary of Workshop Report

There is a necessity for multimedia and multidisciplinary assessments of waste disposal practices. The scientific, technological, political, and economic information needed to make an optimal choice among land, sea, and air sites is today generally available or attainable. Current analytical methods are embryonic, and with more sophisticated developments, additional information needs will become evident. Even after an optimal choice is made, however, constraints on policy implementation remain. They include (1) the statutory framework, (2) public administration processes, (3) public attitudes, (4) economic forces, and (5) information limits.

The principal objective of the decision-making process is to protect and enhance social welfare, a commitment that encompasses public health, environmental protection, and direct costs.

The steps in the multimedia selection process are (1) the identification of various air, land, and sea options and the subsequent selection of feasible alternatives; (2) an assessment involving the selected alternatives; and (3) the choice of the option. Currently, each of these steps toward reaching an option is poorly defined by the waste-management community. Better statutory and regulatory definitions of the criteria for comparing options are needed.

In this report, the information bases for the options were drawn from the natural and social sciences and from two examples of existing problems, viz., sewage sludge and industrial waste disposal. Disposal options for the entire range of industrial wastes were not examined, but site and waste specificities of all waste disposal problems were recognized. In addition, there is throughout the report an awareness that relevant information is often uncertain, controversial, or even lacking to such an extent that the selection process is obfuscated.

Three factors in the evaluation process are closely linked: waste properties, specific site characteristics, and design of the facility. Information required for an assessment can be costly, as, for example, in the case of hydrologic data; the cost of obtaining information must always be balanced against potential benefits of reducing uncertainties.

Criteria for the land disposal option were evaluated. The prevention of contamination of foodstuffs and of underground and surface waters was the primary focus. The most important long-term control on water movement on land is evapotranspiration, which includes evaporation from the soil surface and uptake and release of water by vegetation. Some assumptions in predicting the fate of wastes in groundwaters are questionable, especially those concerning homogeneous isotropic aquifers and average annual or seasonal flow rates for a given source. In future modeling of terrestrial systems, more consideration should be given to episodic and extreme events like earthquakes and to nonuniform properties of the disposal site.

The structural geology and stratigraphy of intended sites for disposal on land need to be evaluated in terms of such features as synclines, anticlines, faults, bedding planes, and burial stream valleys, which may either collect or divert water or waste and force it along pathways that cannot be predicted by simple analysis of pressure gradients.

Predictions of waste behavior in the oceans that will allow the estimation of impacts on public health, ecosystems, and aesthetic factors involve prediction of spatial and temporal concentrations of the substances and of the oceanic constituents that interact with the wastes. The modeling techniques respond to the site specificity of the wastes, of the system, and of the receiving waters. Existing capabilities for predictive modeling make possible first approximations of the fluxes and concentrations so that potential impacts can be estimated.

The predictive framework generally can be used for any combination of characteristics of waste, site, and disposal system design. The framework links predictions of concentrations of contaminants to the proposed disposal system through analyses of the physical, chemical, and biological processes that operate in the proposed disposal area.

The information needed for making predictions was identified in this report through the example of the proposed disposal of sewage sludge to the Pacific Ocean off Orange County, California. The substances of concern in the wastes are heavy metals, halogenated hydrocarbons, and organic substances. Predictive models have been formulated for estimating potential impact.

In the report, possible biological impacts on land, freshwater, and marine biospheres are considered simultaneously with focuses on ecosystem integrity and public health. Overriding concerns are threats to human health, possible species extinction, and alteration of the community structure of affected species. The last two can occur through the action of a toxic substance or through the destruction of habitat.

The assessment of biological impact can best be made through an understanding of how particular biotic communities respond to change. This understanding is already available in the case of some communities and can be developed for others. There is no simple index that identifies irremediable damage to populations, communities, or ecosystems.

The threats to public health involve the return to human habitats of undesirable concentrations of toxic substances or pathogens associated with the wastes. Sites of greatest concern are estuaries, groundwaters, and streams and pastoral and agricultural lands. The epidemiological information assessment of potential health effects related to disposal of sludge on land or in the sea must consider the potential transport route for pathogens and toxicants back to society.

Two examples of multimedia assessments for evaluating disposal options were developed and presented: (1) municipal sewage sludge and (2) industrial acid wastes.

The industrial example involved the disposition of acid iron wastes from titanium dioxide production. After the initial screening process was completed, two possible alternatives evolved: (a) ocean disposal and acid neutralization of the wastes using limestone followed by a disposal of the iron sludge to land and (b) the effluents to the stream. The types of data needed to assess these options were drawn from previous experiences of industry, those involving the 106-Mile Ocean Waste Disposal Site and the New York Bight Acid Waste Disposal Site. Environmental consideration of potential disposal sites required information about impacts on human health, property, and ecosystems, as well as social welfare, which involved aesthetics, recreation, noise, and odors. Institutional considerations associated with each possible option involved community attitudes, services, economy, and safety.

A matrix approach was used in evaluating the two alternatives in the titanium dioxide disposal. Environmental considerations decidedly favored ocean discharge, but institutional considerations were less biased toward that option. Cost analysis favored marine disposal.

Sewage sludge disposal site options were evaluated. The two primary factors governing the analysis, i.e., the composition of the sludge and the geography of the area, generally reduce the alternatives to a small number. The evaluation process involved public perceptions, regulatory considerations, available technology, environmental risks, and cost factors. Existing regulations or agency actions may bias the final decision toward an environmentally or economically unsound practice.

The available or obtainable information permits selecting a sewage sludge disposal option based on technically supported data from the different media. However, current studies on disposal techniques should be reviewed periodically, especially in cases involving new construction or replacement of existing facilities. System reliability is an important component of technological evaluation, and it strongly influences the economics of sludge disposal.

1

Report of the Panel on Sludge Management and Public Policy

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1.1 DEFINITION OF THE PROBLEM

The United States produces an estimated 3 billion tons of solid waste material annually. Some of this waste material is harmless to humans and the environment, some of it can be rendered harmless by natural or technological processes, and some of it must be carefully handled and stored to avoid known or potential harm.

The necessity of managing these waste materials--by recycling, treatment, storage, or dispersal--often produces clashes among conflicting interests.

Even though well-intentioned, disposal schemes, adopted under the pressure of urgency, often go awry because a disposal method is selected before sufficient information about it is available. This happened recently in South Essex, Massachusetts. Faced with a growing burden of municipal waste sludge, the town took what it thought was a careful look at the available options and chose to incinerate its sludge. The result, local officials and citizens believed, would be sterile ash that could be deposited in the town landfill without harm. The citizens of the town did not know that incineration would convert the chromium in the sludge to an extremely hazardous substance, hexavalent chromium, and that this hazardous ash would be difficult and expensive to dispose of. [1]

Other local governments have run into storms of citizen outrage that have thwarted well-intentioned plans to develop sludge disposal projects. In 1978, with the help of federal construction grants, Nassau County, New York, sought to solve its growing sludge disposal problem by building a community composting facility according to nationally tested methods. But the plant was never used. Local citizens opposed its operation, fearing groundwater contamination through chemical leaching if the compost was used on nearby lands as has been planned. Had the county not shut down the plant, its elected officials might have lost their jobs in the next election.

Of major concern in waste management are the toxic chemicals and pathogens in many wastes, even though these are usually only a small part of the waste. Often it is not feasible to destroy all pathogens or to remove toxic chemicals for disposal elsewhere. The amounts of toxic chemicals in wastes can be reduced in several ways, most notably through some types of regulatory action. Among the specific chemicals that have been banned from production and use because of scientific evidence or public suspicion of harm to the environment and public health are the following:

1. DDT for the effect on non-target organisms,
2. PCBs for the effect on public health, and
3. Fluorocarbons for the effect on the quality of the atmosphere (ozone layer).

Scientific evidence, economics, or regulatory constraints or combinations of the three have also been the stimuli for process changes such as the following:

--Recovery of chrome from tannery wastewater treatment facilities,
--Recycling of ferric chloride in sewage treatment for titanium dioxide extraction processes,
--Recycling of bark waste and gas by-products in the pulp and sugar cane industries for use as a fuel,
--Use of anaerobic digestion to produce methane from organic residues.

Wastes usually change the environment into which they are placed. The question is whether the change is harmful or beneficial. For example, waste disposal can result in increased or decreased biological productivity on land or in water.

1.2 CONSTRAINTS ON POLICY IMPLEMENTATION

Waste-management decisions take place in a world of political, bureaucratic, and financial forces that often impede the implementation of nationally existing regulations and standards. No matter how scientifically sound or economically rational a waste disposal decision may be, it will encounter resistance and inefficiency in the eventual administrative implementation of the decision and financial difficulties that affect the disposer's ability to comply with the original decision. In designing a method for assessing the appropriateness of using land, sea, or air through incineration or a combination thereof for disposal of waste (hereinafter referred to as “multimedia assessment”), therefore, it will be useful to know as much as possible about the constraints that may influence the practical utility of the process. Many municipalities have already developed rudimentary approaches to multimedia decision making--

deciding between land application and incineration, for instance--and some, such as Los Angeles and Chicago, have made considerable progress in rationalizing this process. We can learn from these case histories.

Constraints on waste management decision making can be put into four categories: (1) statutory limits, including federal laws and state and local ordinances, as well as their interpretation by the courts; (2) the public administration system in which waste management decisions are made, including the administrative regulations that interpret the laws, the people that make the decisions, and the public reaction to these decisions; (3) economic forces in the financial marketplace that affect the capacity of a producer of waste to raise capital or to generate operating revenues; and (4) information limits, including lack of information, as well as uncertainty and disagreement among experts about the validity and value of information. This section describes these constraints, and recommends strategies and tactics for overcoming some of the barriers.

1.2.1 The Statutory Framework

The medium-by-medium approach to environmental lawmaking over the past decade has resulted in a legal framework that creates barriers to a rational choice for waste disposal. Such constraints arise as a result of direct Congressional intent or come about through administrative or judicial interpretation of Congressional intent.

A number of studies have analyzed the complex statutory maze that Congress has created to manage ocean, land, and air resources and wastes. These include NAS's first report on multimedia decision making (1977) [2] and a more recent report by National Advisory Committee on Oceans and Atmosphere (NACOA) (1981). The authors of these studies as well as many observers have noted that the highly fragmented environmental statutory framework for managing wastes has made it extremely difficult for decision makers to implement these statutes effectively. Some of the most important federal laws that comprise this fragmented framework are the following:

1. The Marine Protection, Research and Sanctuaries Act of 1972, as amended;
2. The Federal Water Pollution Control Act of 1972 and amendments thereto;

3. The Clear Air Act of 1977;
4. The Resource Conservation and Recovery Act of 1976;
5. The Toxic Substances Control Act of 1976;
6. The Safe Water Drinking Act;
7. National Environmental Policy Act (NEPA) of 1970;
8. the Endangered Species Act of 1973;
9. The Marine Mammal Protection Act of 1972;
10. The Fish and Wildlife Coordination Act; and
11. The Federal Insecticide, Fungicide, and Rodenticide Act.

The proliferation of federal statutes relating to the environment reflects the inclination of our society to deal with wastes according to both source and disposal site. There is no uniform regulatory framework for governing the assessment of risks, costs, and disposal options associated with all wastes. The absence of such a framework poses a severe constraint at the federal level. There are statutory provisions, for example, that impose different restrictions on existing and new sources of waste material. If new source requirements are more stringent than those for existing sources, for instance, then the phase out of inefficient older sources or the retrofitting of these sources with more efficient technology will be severely constrained, because it will be easier and less costly to retain old sources.

State laws on regulating wastes are often modeled on federal waste regulations. The Federal Water Pollution Control Act of 1972, for instance, established for the first time nationally uniform water-quality standards and pollution-control requirements. The states were required to respond by changing their own laws and building new programs consistent with the federal initiative. The states, therefore, tend to replicate the same legal constraints that exist at the federal level.

Local waste ordinances tend to be much more diverse than do state and federal laws. They often prohibit certain waste disposal practices that place additional constraints on the implementation of federal and state laws. The proliferation of such local ordinances reflects the “first law of waste management”--that it is difficult to move wastes across political boundaries. Such local ordinances take numerous forms. They may be zoning regulations that prohibit landfills or siting of incinerators, laws that limit the transportation of waste, or health ordinances that regulate the application of sludge on agricultural land. A positive example follows: the

City of Los Angeles is prohibited from incinerating and can only use toxic waste landfill for sludge disposal. However, pyrolytic combustion of the gas produced is currently used to generate power. There was a change in the interpretation of incineration in order to allow this practice. Although state or federal officials do have the option of preempting local ordinances that are in conflict with a desired waste disposal decision, the practical utility of preemption as a normal instrument of public policy, given the politics and legal uncertainties involved, is quite limited.

The Federal Water Pollution Control Act of 1972 and other environmental laws add yet another dimension to this set of problems, since they include sanctions with criminal penalties attached. This can and often does result in an adversarial relationship between operating and enforcement agencies. The adversarial aspect of this relationship must be resolved, because much effort is spent at the local level, particularly protecting operating agencies from possible actions of state and federal regulatory agencies. To demonstrate the strength of this problem, the City of Burbank, California, responding to the enforcement level overseeing its activities, has contracted out all of its work because the burden of responsibility and liability was felt so strongly. This decision is more costly and probably less effective. An atmosphere must be sought in which regulatory agencies are supportive of operating agencies in their search for environmentally acceptable, practical solutions to sludge disposal questions.

The power to set public policy for waste disposal is divided among local, state, and federal governments. Thus, in some instances there may be no single entity in a position to accept full responsibility for the disposal of wastes. In addition, the deliberate diffusion of responsibility and frequent conflict within the decision-making structure can result in a suboptimal choice.

The absence of a legislative framework to encompass the various disposal media and to give evenhanded treatment to each sludge management case has a harmful effect on the decision-making process. Each of the three options--land, air, and sea disposal--falls under different legislation and different authority, making consistency and continuity in multimedia assessments difficult and resulting in the absence of a single focal point--governmental entity--for a decision.

There are several ways to ameliorate these problems. One would be to develop more effective ways to carry out multimedia analyses within the existing legal framework (and possibly by modifying existing regulations) and ultimately to force "organic" changes in the implementation of laws, e.g., more flexibility. A second would be to revise the laws. The former would be preferable, since legislation revision would leave open the possibility of creating a whole new--and equally unbalanced--set of laws owing to the nature of the legislative process and the need to serve pluralistic interests.

A third approach would be to enact a new federal statute that would require waste disposal decision making to be conducted that would give equal consideration to all media, allowing for the full use of (in an evenhanded manner) multimedia assessments. Such a statute might eliminate the problems caused by the present unbalanced statutory framework, disallowing some media and allowing others.

Fourth and finally, the use of the EPA Sludge Task Force could result in a movement toward changing legislation by developing a list of coherent recommendations for simultaneous, uniform changes in all the regulations through amendments.

1.2.2 The System of Public Administration

The second category of constraints on waste-management decision making is the public administration system in which such decisions are made. Included under this heading are the regulations, the regulator, and the public responses.

A variety of federal regulatory agencies have been set up to administer the waste-management statutes. This fragmentation of responsibility at the federal level is one of the primary bureaucratic constraints on multimedia assessment, which would require greater integration in designing criteria, implementing the criteria, and evaluating results scientifically. Should the National Oceanic and Atmospheric Administration, the U.S. Environmental Protection Agency (EPA) , the U.S. Coast Guard, the Food and Drug Administration, or the Corps of Engineers identify those characteristics of the ocean's response to waste that need to be measured for the purpose of comparing ocean disposal with land and air options? NACOA asserts that, depending on the agency of origin, current regulations may be overly protective of one or another medium. Waste disposal decisions therefore tend to follow the path of least regulatory resistance.

The selection of any medium or any specific site for disposal of waste will generate public reaction, both favorable and unfavorable, and it will be shaped by a number of factors. The most prominent is that such decisions are, by their nature, inequitable, since benefits and risks are distributed unequally. Indeed, effectively managing this distribution of benefits and risks by counterbalancing inequities with incentives is a necessary consideration in public decision making.

There is uncertainty in every scientific observation and measurement and in every prediction of the effects of a waste disposal plan. It would be helpful if these uncertainties were explicitly described as inescapable and communicated to the public.

More data and less uncertainty are always desirable in making environmental decisions. It should be recognized that while the possibility of certain effects may be the pivotal issue in making a decision, a decision may be much more affected by the uncertainties related to these effects.

At any given time, a decision to use one of the available options rather than to wait for more data will depend on the likelihood of obtaining more data on the cost and time required to get new data and on the likelihood that additional data will significantly reduce the uncertainties. The cost must include the transaction costs to the government of focusing on regulatory processes rather than moving on to other issues.

The high costs of resolving uncertainties in scientific data often deter decision makers from effectively addressing these questions. In addition to dealing with uncertainties resulting from insufficient data, decision makers must also address uncertainties with regard to time needed to evaluate environmental impacts. It took scientists about 10 years, for example, to ascertain that methyl mercury was the toxin in the Minimata Bay epidemic in Japan. It also took scientists many years to understand the relationship between DDT and eggshell thinning and the subsequent population decline of some marine birds.

Political reaction times on environmental problems also lag considerably. Effective political response only came years after the scientists had concluded their assessments of methyl mercury and DDT in the cases mentioned above. Such constraints are likely to be formidable barriers to the successful implementation of multimedia assessment.

Factors influencing public reaction to waste disposal plans include the quality of information to which the public has access, public perceptions of the risks associated with selection of a disposal medium, and the organization of affected parties into special interest groups. Indeed, unequal access to decision makers, by groups or individuals representing different and often competing interests in itself may constitute a barrier to a sound decision by limiting the decision-maker's understanding of the full impact of a decision. For example, in 1980 the City of Los Angeles considered pumping its water sludge to desert solar drying beds and then utilizing it with reclaimed water for agriculture or disposing of it in a captive hazardous-waste landfill near the drying beds. Although this option had considerable scientific-technical merit, it was never a realistic possibility because of concerted opposition by desert residents and environmental organizations. An educational program with some type of benefit to desert residents--for example, development of facilities such as a park and camping area or swimming pools--might have eventually allowed the environmental soundness of such a plan to be shown. However, court-mandated time schedules prevented this, so an incineration scheme in an air-quality “nonattainment” area was adopted, and construction funds now committed to the incineration alternative prevent further consideration of the desert disposal site or any other similar option. [3] The lesson is clear. Decision makers must consider not only the scientific-technical data but factors such as public perception, sufficiency of public education, time schedules mandated by laws or court action, and the extent of capital investment already made. Scientist cannot be truly effective until they realize that scientific facts are only a portion of the total information base from which policymakers will draw evidence for a choice.

1.2.3 Economic Factors

The third constraint on waste-management decision making is that the economic climate is an important but generally overlooked force in waste-management decisions. The ability to finance renovation or construction, or to pay operating costs, affects a waste generator's outlook on the selection of a particular waste-management method. Federal and state governments have paid as much as 75 percent of the construction costs of municipal sewage treatment facilities during the last decade. The total federal investment in these facilities since 1973 has been approximately $28 billion. Recent amendments to the Federal Water Pollution Control Act, however, will reduce federal funding for projects initiated after November 1984.

States and municipalities will therefore need to find innovative ways to decrease the costs of waste disposal projects as well as to discover new sources of financing. At least two things should be said here about cutting costs while still providing satisfactory facilities. First, construction engineers have claimed that complying with certain federal regulations regarding construction standards is costly and that those regulations often are not necessary. This suggests that there should be a thorough evaluation of the current criteria to eliminate costly and unnecessary design rules. Second, the delays that characterize every waste disposal project result in immense increases in cost that often make the projects far more expensive than necessary.

States and municipalities will increasingly look to the financial markets and private investors for funds to finance the capital costs of waste disposal projects. Economic trends, however, are likely to result in substantial disincentives to invest in waste-treatment facilities for several reasons:

- The market's movement away from general obligation bonds toward revenue bonds will put some states, such as New York (whose constitution prohibits municipalities from issuing revenue bonds), at a disadvantage;
- States and localities are increasingly seeking funds for other purposes, such as industrial development;
- General economic conditions, such as high interest rates and the 1981-1982 recession, have placed severe budgetary restraints on municipalities, in some cases leading to a lowering of credit ratings and greater difficulty in gaining access to financial markets.

These trends have several policy implications for waste disposal. First, policy on waste disposal may be driven by the crisis in municipal finance rather than by the health and environmental consequences of sludge disposal unless multimedia assessments are required by federal law. The number (more than 200) of recent applications by coastal communities for waivers from secondary treatment under Section 301(h) of the Federal Water Pollution Control Act is one indication of these financial pressures. It therefore appears that municipalities will develop sophisticated and innovative methods of raising private capital more easily if EPA simultaneously cuts back construction grants and firmly enforces standards and if accelerated depreciation provisions of the new federal tax laws are allowed to remain in effect. Municipalities can use these laws to facilitate private sector building of facilities and then lease or purchase the facility according to certain legal conditions. Business profits result and municipalities do not have to raise capital funds, which could have an impact on local taxes.

The synergistic relationship between municipal finances and waste disposal suggests that there is need for more study of the ways in which federal policy can help municipalities and states to obtain private capital. New Jersey, for instance, is seeking to establish an “infrastructure bank” in which federal construction grants, bond proceeds, and state revenues would be pooled, but changes in federal policy will be required if the state is to carry out this experiment.

The federal government has also affected the waste disposal site-selection process by the funding mechanism that it has used. In the past, the possibility of cost sharing among local, state, and federal agencies has often influenced the choice of a waste-management method. Some major cities, however, have found that smaller operating costs for certain disposal methods were large enough to offset construction grant incentives. EPA funding incentives, which emphasize pretreatment, have also influenced the quality of the waste streams.

Financial incentives could also be used to discourage undesirable waste disposal practices. From a human welfare perspective, the quantity of waste should be reduced as the cost of disposal per unit of waste increases. One possible, but not necessarily equitable, * way to foster this relationship would be to impose a fee

* Sometimes the waste is not entirely the responsiblity of the polluter (disposer) owing to inconsistencies in the regulatory process.

or tax on waste producers based on the quantity of their wastes. This would be an application of the "polluter pay" principle. [4]

Given the availability of federal construction grants and various tax benefits, however, the fees or taxes imposed on industries discharging into municipal waste disposal systems might be lower than the actual direct costs of disposing of such wastes. Public attempts to recover more fully the indirect costs imposed by externalities include legislation to redress public health impacts (such as industrial compensation laws in the United States and Japan), as well as efforts to recover the economic losses resulting from environmental degradation (Title III of the Outer Continental Shelf Lands Act Amendments). It is unlikely that such an approach will be taken soon. The quantification in monetary terms of the direct and indirect costs of waste disposal in a multimedia assessment would be useful as a basis for redistributing these costs, if the assessment were done on a national scale.

While many of the factors in waste management can be reduced to monetary terms in preparing a multimedia assessment, this is not true of social values--e.g., the value of a human life. While it is possible to quantify social values such as health risks, environmental impacts, and aesthetics in monetary terms, reducing them to these terms often becomes controversial. Effective inclusion of social values in multimedia assessments would minimize the undervaluing or overvaluing of many natural resources and the amenities they provide. Some approaches to this difficult task are described in the next section.

1.3 ANALYTICAL METHODS AND INFORMATION NEEDS FOR DEVELOPING AND ASSESSING PUBLIC POLICY ON WASTE MANAGEMENT

Certain analytical methods, in conjunction with the data that they require, provide the fundamental structure for formulating waste-management strategies. Determining the scope of the structure for formulating strategies involves performing the following tasks, which are elaborated in the succeeding section.

- Stating the assumption; defining the context in terms of a federal, state or local decision;

- Formulating criteria and a weighting system for measurement based on the context;
- Evaluating quality of information and the cost of obtaining it;
- Understanding the impacts or effects of concern of a decision.

Assessing the strengths, weaknesses, and information requirements of these methods is crucial to understanding the validity of the implications of alternative disposal policies.

1.3.1. The Assumptions

The Panel on Public Policy concluded that the appropriate analytical methods for assessing a particular waste-management strategy depend on the following four assumptions:

1. The first assumption is that the principal objective of public policy is to maximize social welfare (public health, environmental protection, and direct costs) subject to the constraints imposed by technology and scarcity of resources.
2. The second assumption is that equity does matter to political decision-makers, even though there is no single, agreed-on definition of equity for determining the best distribution of gains and costs. That is one of the fundamental problems of modern welfare economics. [1]

   If maximum social welfare is defined to be the sum of the welfare of all the individuals in a society, the implication is that there need not be any concern for the distribution of welfare among those individuals. But clearly, many policy choices can only be understood as efforts to distribute benefits and costs in certain ways. Thus, decision makers will often want information on the distribution of the costs and benefits of policy alternatives in addition to the aggregate figures of total benefits and total costs.
3. The third assumption is that there may be substantial uncertainties concerning physical, biological, economic, and social parameters of the problem of deciding on waste disposal media. The problem of uncertainty and how to deal with it is discussed later in this chapter.
4. The final assumption is the recognition that there is a finite supply of resources. Popular or desirable

activities will have priority over less popular or less desirable activities in the competition for these dwindling natural resources. Therefore, to improve opportunities for all to share in the use of these resources, a mechanism must be developed to provide for such allocation of resources.

Scarcity, real or potential, has implications both for the planning and management of waste disposal activities and for research and information gathering. Additional resources should be allocated for disposal and protection activities only as long as the additional benefits outweigh the opportunity costs. Additional resources should be committed to research and information gathering only if the value of the information (measured in terms of better decisions) exceeds the cost of the information gathering.

These assumptions determine the type of analytical methods and data that must be considered.

1.3.2 The Context for Waste-Management Decision Making

Seven basic tasks make up the inputs to decision making on waste management, whether at the national, state, regional, or local level:

1. Identification of policy objectives,
2. Identification of impacts,
3. Development of methods or models that measure or assess such impacts,
4. Identification of the parameters of the inputs used in such models,
5. Development of the information used in the models,
6. Estimation of the costs attributable to option, *
7. Determination of sociopolitical implication of decisions for specific options. *

Depending on whether the objective is to design waste management strategies at the national, state, regional, or local level, these tasks are often addressed quite differently. The environmental decision-making process at the various levels differs primarily in (1) the weights that are attached to certain concerns, (2) the scales of

*Considered in more detail in other sections.

resolution required to define information relevant to the problem at hand, and (3) the boundaries of the problem under analysis. These differences discussed below affect the scope and scale of information gathering. They also affect methods of assessing the overall costs and benefits of alternative waste-management strategies.

1.3.2.1 Weighting

Public opinion may cause local and federal decision makers to weight cumulative concerns differently. The local point of view, for example, may be that a sludge disposal method should produce <u>no</u> potential groundwater contamination.

Likewise, local decision makers may assign zero weights to factors that higher-level decision makers find extremely relevant. Statutory and institutional factors may also come into play. Typically, local and state standards for pollution control are stricter than federal standards. New York state standards for effluent discharges, for example, are more stringent than are federal standards. Pennsylvania standards for sewage sludge management are generally more stringent than federal standards. These are just two of the many states with stricter local controls than federal controls.

1.3.2.2 Scale of Resolution

Local redistributional effects of many kinds as a result of sludge management policies may be important in ways not apparent to state and federal decision makers. In an analogous situation, water-pollution-control strategies can have the following results: (1) cause the removal of freshwater from groundwater recharge, for example, increasing salt water intrusion into groundwater, or (2) the discharge of freshwater beyond the nearshore zone may have an unfavorable effect on local shellfish populations. Similar examples have been found in sludge management. [5]

Methods of analysis must be flexible enough to allow separate or complementary consideration of such effects, which may be missed by macroscale modeling. Policy prescriptions that require comprehensive analytical work for long time scales may not appropriately meet the requirements for decision makers. Local decisions, for example, are apt to be made on shorter time scales than those that models would prescribe. Attempts to define the proper social discount rate in terms of a single unitary measure would be hampered by consideration of these variable factors.

1.3.2.3 Boundaries of the Problem

Local decision makers are less likely than are their federal counterparts to view the imposition of externalities (effects on third parties) by sludge disposal projects as a problem. Therefore, they may see the analytical problems in calculating externalities as less of a concern.

1.3.3 Scientific Uncertainty and the Value of Information

Information gathering has a price. No single piece of information is worth acquiring "at any price," and the costs of reducing uncertainty by acquiring new information should be balanced against potential benefits before data acquisition starts. Analytical methods exist for calculating the expected value of information should one wish to reduce uncertainty.

The decision maker must understand the uncertainties in the information obtained. Every observation or prediction is accompanied by uncertainty. Recognizing the importance of particular types of uncertainty will result in better decisions. Scientists supplying the information for such decisions should be realistic about the quality of their information by explicitly stating the limits of their certainty and what would be necessary to establish a significant degree of more certainty. If, for example, a decision must be made on whether a toxic chemical compound should be applied on land or disposed of in an estuary, the fate of the compounds and their potential impact on society must be considered.

Environmental questions are often the most difficult to answer for the following reasons:

- The lack of rigorous mathematical underpinnings in ecology, compounded by the variability of ecosystems make predictions difficult.
- The numerous variables and their interactions, which make chemical fate (destination) modeling in ecological systems difficult.

- The lack of field verification, which makes modeling applications suspect in decision making.
- New discoveries in ocean dynamics that have added new dimensions to models of dispersion, dilution, and flow, making them more difficult to apply.

1.3.4 The Effects of Concern from Decision Making

Because every waste disposal decision relates to a specific situation, it is not possible to list all the effects of particular decisions that may cause concern. It is possible, however, to identify some of the effects that will influence the selection of a disposal option. Deciding which effects are matters of concern is a social value judgment. Not all effects arouse concern, and not all the effects that arouse concern are adverse effects.

Certain values are reflected in the various environmental laws that guide decision makers. The few explicitly environmental values addressed in law include the desire to avoid species extinction (irrespective of their commercial value), the desire to have waters suitable for fishing and swimming, and the desire to retain clean, not just healthful, air in the pristine areas of the United States.

Although not specifically addressed in federal statutes, the following are also effects that would probably be of concern to most decision makers: human deaths (from disease), incidence of chronic human disease, and significant negative effects on important ecosystems, e.g., commercial and sports fisheries, wetlands and estuaries, timberlands, and agriculture.

Other important effects include such things as dollar and resource costs. Any regulatory decision is based on a perceived need to alter the outcome that would otherwise be produced in the workings of a free market. Therefore, economic costs are always entailed in any decision. The size and nature of these costs can weigh heavily in the decision-making process.

Many of the cost effects of environmental decisions that concern the federal government are spelled out in Executive Order 12291 and in directives from the Office of Management and Budget. These include effects on the following:

U.S. balance of trade,

U.S. Gross National Product,

U.S. employment,
Employment, profiles, and costs to any particular sector of the U.S. economy,
Employment, profits, and costs to a particular firm,
Costs to consumers, and
Innovation.

Another major category of effects includes equity issues. While most environmental decisions yield a wide variety of positive and negative effects, they also redistribute costs, risks, and benefits. Thus, the decision-making process needs both to characterize the size and type of the effects of each option and to identify those who benefit and those who suffer from each option so that such inequities can be weighed with other trade-offs. In the case of inequities it may be necessary to make “transfer payments” in some way to compensate those who lose from a media decision. Effective mechanisms must be sought to smooth this process and means of determining types and amounts of compensation sought preferably outside of the court system.

The secondary effects of a given option--the signals that are sent to those affected by its selection--are often overlooked. A decision to require chemical neutralization of one waste stream, for example, might signal to corporate planners and others that similar action is likely to be taken with respect to similar wastes in the future. As a result, planners may build this expectation into their corporate plans to avoid trouble in the future. Thus, a decision in a single setting may have effects far beyond the scope of an action itself and may be an important reason for selecting one option over another.

The selection of any given option is more likely to result in certain effects than in others. While the decision maker initially needs to be concerned about the possibility that the option may cause certain effects, specific circumstances tend to make certain possible effects much more pivotal than others because of their magnitude. Thus, the decision-making process is most efficient if it is iterative.

For each possible effect, there needs to be an analytical method or model to estimate magnitude. The method or model selected will depend on the inherent uncertainties, the availability of input data, and cost. Analyses may be complex, such as an analysis of the impact on fisheries of a given discharge in a particular loca tion, or they may be simpler and more direct, such as extrapolating laboratory bioassays to determine impacts on individual species.

Analysis may include estimating human health effects from animal models, cell culture studies, or pharmaco-kinetic measurements, extrapolating the results of single-species toxicity tests to determine ecological effects and predicting costs to various individuals and institutions by means of econometric studies.

Depending on the models or methods used, effects can be defined and measured in a variety of terms. For example, the costs of constructing and operating a land depository are measured in dollars, while possible adverse effects on human health can be expressed in terms of an increase in the probability of cancer or increases in the incidence of chronic or acute illnesses.

The effects must be modeled or analyzed for their magnitude of disposal in the various media (ocean, land, or air) in terms of human health, environmental impact, and direct costs.

1.3.5 Matrix Approaches to Multimedia Assessment

Several approaches to multimedia assessment are in use. Industry has been using a matrix analysis method for several years (see Chapter 6 for a detailed discussion of this method). This tool allows a baseline consideration to be used in a pragmatic implementation where public response then also needs consideration before a final decision. Another approach has been used by SEAMOcean, a consulting firm under contract to New York City to recommend management strategies for sludge disposal. A description of this method follows.

Multimedia analysis of waste disposal options requires the use of a set of criteria for selecting the best of the available practices. These criteria must be a set of values that can be expressed in a language that is common to all media and whose magnitude of effects in each medium will be determined by the waste disposal practice. An example of one such value would be the need to protect human health. Identification of the values is a matter of public policy, which subsequently defines technical information needs. Once the technical information has been obtained, a comparison of the magnitudes of the values must be made and the best disposal practice selected. Before a decision can be made, however, ac ceptable methods must be developed to weigh the options, based on comparative values. This part of the process is obviously controversial. Three basic steps are needed to complete a multimedia waste management assessment:

- STEP A. Identification of Alternatives--review of potential air-, land-, and water-based options and selection of feasible alternatives for further assessment.
- STEP B. Assessment of Alternatives--detailed study and comparative evaluation of the alternatives.
- STEP C. Selection of an Alternative--selection of a preferred alternative, based on the results of the assessment.

These steps are poorly understood by those responsible for managing the disposal of wastes. Since the technical information needed at various points in the selection process will vary, depending on the step and on the administrative process used to implement the step, the most critical need at present is to define the steps better. Following is a description of each of the three steps.

STEP A. Identification of Alternatives

In determining the preferred alternative for disposing of a particular waste, the first step is to establish a range of potential alternatives. Of these alternatives, a small number must be chosen for detailed study.

The purpose of identifying alternatives is to keep the number of site-specific technical studies that must be conducted at affordable levels. The identification process is not well defined, either by statute (e.g., NEPA) or by regulations. Better statutory and regulatory definition of possible options is needed.

Many current statutory prohibitions or restrictions, such as those that ban ocean dumping or restrict the use of landfills, may preclude choosing the best option. It will be a difficult task to amend environmental statutes and regulations that now control disposal in a single medium so that they facilitate multimedia assessment while still precluding unacceptable impacts. Ideally, the criteria should ensure that all important factors in each medium are appropriately addressed. The amendment process will require considerable public debate, an understanding of the multimedia assessment process, and confidence in its effectiveness. Initially, it might be simpler and more effective to establish waiver provisions that would permit an otherwise prohibited waste-management alternative when a multimedia assessment demonstrates that it is the preferred alternative.

STEP B. Assessment of Alternatives

Once a number of disposal alternatives have been identified as feasible, a comprehensive multimedia or cross-media assessment should be performed. The criteria used in the assessment must reflect public attitudes. They should include the following: (1) human health risks, (2) environmental effects, and (3) direct costs.

The establishment of the criteria and the processes used to apply these criteria are matters of public policy. One possible approach is discussed in Step C.

STEP C. Selection of an Alternative

The third step in a multimedia assessment involves balancing information to select the best alternative or combination of alternatives. In certain cases, a mix of alternatives will be preferable to placing sole reliance on a single alternative. In the event that one alternative is clearly superior to all others with regard to one or more of the assessed factors and not significantly different with regard to other factors, the decision will be simple. Such an outcome is rare, however, and some means of weighing and balancing the factors must be developed.

Determining the relative importance of the criteria is a difficult and controversial process, since it is necessary to weigh health, environmental, political, and economic considerations. Various types of social cost-benefit procedures can be used with a range of criteria. The matrix elimination procedure is mentioned here simply to illustrate one possible method that has been used recently in the New York City appeal.

Economic analysis offers a complementary but not comprehensive method for weighing unequal criteria. Cost-benefit analysis reduces the various measures of human welfare to a common measure--money. Many of the things that matter to human welfare can be bought and sold in markets. If markets function reasonably well in a technical economic sense, the prices of economic goods can be taken as measures of their relative value. The price reflects an individual's marginal willingness to pay--that is, the willingness to trade the consumption of one thing that contributes to the individual's welfare in order to obtain one more unit of another thing.

If each individual's marginal willingness to pay for a beneficial effect can be measured, and if marginal willingness to pay to avoid negative effects can be measured, then all the effects that are matters of interest can be measured in the common measure of money.

The net effect on human welfare of each disposal alternative can be calculated by adding up an individual's willingness to pay for beneficial effects and deducting the individual's willingness to pay to avoid negative effects when given choices among options. The option selected should represent the highest positive net willingness to pay (net benefit) or smallest negative net willingness to pay (net cost).

Whether the options will generate net benefits or net costs (negative net benefits) depends on the reference point or starting point from which effects are measured. In the case of waste-management policy, it is often convenient to take as the starting point a situation in which no wastes are being disposed of. Disposal options generate negative or positive effects in the form of disposal costs, health effects, and environmental effects, for example. The criterion for choosing among them is to choose the option that minimizes the total costs of waste generation and disposal.

Assume that the decision criterion is to choose the alternative that minimizes the total costs of using the environment to dispose of wastes. The next steps are to identify the individual components of the total cost and to discuss how they might be measured. Broadly speaking, there are two categories of costs resulting from waste disposal activities:

1. The direct costs of disposal. These include the costs of, for example, processing the materials, transporting them to the site, constructing barriers, and monitoring. These costs are for labor, capital, materials, fuels, and related items, expended in disposal. Since these inputs are purchased through markets, there should be no difficulty in measuring their direct costs.

2. The indirect costs associated with the presence of the disposed materials in the environment. These indirect or external costs arise because of the common property aspects of the environmental media. Substances that are added to the water, land, or air can have adverse effects on third parties through a variety of channels. For example, individuals may find their health impaired because of air pollutants, contaminated food, or contaminated water. Or the discharge of substances to the ocean might adversely affect aquatic ecosystems and result in decreased productivity of commercial fisheries or reduced recreation opportunities. The monetary values of such adverse effects are measured by individuals' willingness to pay to avoid these effects.

One way to view the waste-management problem is to consider different options as representing trade-offs between direct and indirect costs. Consider two options, A and B. Suppose that option B has direct costs that are greater than those of option A by $10 million. This could be because wastes are transported a greater distance to avoid sensitive environments or because the need to contain the wastes results in higher engineering costs or because it is necessary to process the wastes to remove potentially harmful substances. Option B should therefore be chosen only if incurring the direct costs would lead to a reduction of indirect costs at least equal to $10 million. Otherwise, the costs of option B would be even higher.

The conclusion above, however, is subject to two qualifications. First, if equity is an objective of public policy, decision makers may reject the option that minimizes total cost on the grounds that it imposes additional costs on already disadvantaged groups; that is, it leads to an inequitable distribution of costs. If equity matters to decision makers, then the data and models used to estimate benefits and costs must allow the assignment of benefits or costs to specific groups identified by characteristics that concern the decision makers, for example, income level, race, or political affiliation. Since an ideal estimation model would be disaggregated to begin with, e.g., with data segregated according to groups, effects, or other related factors--the effort to present information on distribution means only that the step of aggregation to total benefits or costs is omitted.

Second, it is necessary to express all effects in common units, usually monetary units. Where monetary measures of certain impacts are not available, one of two approaches can be followed: (1) the decision makers can establish some target level for the effect in question and choose the alternative that comes closest to the target at the lowest cost; or (2) having received information on the net monetary cost and the level of the incommensurable effect (e.g., health effect and monetary cost for each option), the decision makers can choose an approach on the basis of their own personal values.

Suppose, for example, that the health effects of three different options can be expressed in terms of number of expected deaths. Further, assume that decision makers are unwilling to place a monetary value on avoiding human deaths. Finally, assume that all the other factors have been measured in dollars. Suppose the three options are described as follows:

	Option A	Option B	Option C
Expected deaths	20	10	5
Total monetary costs	$20 million	$10 million	$15 million

This information shows decision makers the trade-offs between the two incommensurable classes of effects--health effects and money costs. It may then be possible to exclude some options through the application of simple criteria. In this example, both B and C are clearly superior to A because they entail fewer deaths and lower costs. But no simple rule can be invoked to make a choice between B and C. The decision maker must then accept responsibility for the deciding between them. The choice of either alternative reveals something about the implicit value placed by the decision maker on avoiding deaths. If B is chosen, this reveals that the decision maker was unwilling to incur an extra $5 million in costs to avoid five expected deaths.

We turn now to the information needed to estimate indirect costs. The techniques for estimating indirect costs are a function of the environmental effects of disposal. We need to understand the whole system of transport and dispersion or containment of waste material in the environment, the effects on species of importance to humans for economic, aesthetic, or cultural reasons and the direct effects on human health and welfare. The measurement of these effects in economic terms requires first their quantification in the relevant units of measurement (e.g., reduced productivity of commercial fisheries, reduced opportunities for recreation or agriculture, or increased illness or risk of death) and then the imputation of money measures of willingness to pay to avoid these effects.

Consider, for example, the indirect costs of spreading sludge on land surfaces. If runoff from the land surface containing pollutants reaches surface waters and reduces the attractiveness or suitability of these waters for recreation, this is a cost. There are several economic models and techniques for predicting changes in recreation behavior as a function of (among other things) changes in the quality of water or changes in the probability of catching desirable species of fish. If the changes in water quality or fishing success are known, these models can be used to impute the willingness of recreationists to pay to avoid these adverse effects.

If certain pollutants would contaminate drinking water and thus have adverse effects on human health, and if the magnitude of the health effects can be predicted from dose-effect relationships, then economic methods exist for inferring individuals' willingness to pay to avoid these health effects. Models and methods also exist for estimating the value of groundwater, commercial fisheries, and other resources. If these resources were to be destroyed or impaired by pollution from waste disposal activities, these models and methods could be the basis for imputing the indirect costs of disposal.

There is one type of possible effect of waste disposal that lies outside the economic sphere: adverse effects on some species of no commercial, recreational, or aesthetic importance to man. Economic methods are relevant only to measuring effects on human welfare, that is, those effects that matter to individuals who are therefore willing to pay to avoid them. If some species is adversely affected by pollution but individuals have zero willingness to pay to avoid the effect, the economic cost of the effect is zero. If such effects matter to decision makers, they must be listed separately from economic effects. And the decision maker must make trade-offs within the process, since the equity issue is constantly present.

1.3.6 Methodological Problems of Integrating Various Types of Information

Regardless of the management context--that is, the geographical area involved, the time scales of concern, or the waste constituents of interest--there is a finite set of information that must always be considered in making public policy decisions about sludge disposal. This consists of information on the following:

- The economic activities generating the wastes or the constituents of waste to be disposed of;
- The dispersal and temporal patterns in the environment of the wastes to be disposed of;
- The ways in which specific economic activities would be affected by alternative waste-management strategies;
- The dispersal and temporal patterns of environmental quality resulting from disposal under alternative waste-management strategies, measured by whatever indices are relevant;
- The magnitude and distribution of the effects that would result from alternative levels of environmental quality, including effects on human health, effects on valued living resources, and effects on the use of other resources;
- The magnitude and distribution of the economic impacts (who pays and who benefits) associated with alternative waste-management strategies;
- The incentives and institutional arrangements required to implement each alternative strategy and their associated administrative costs; and
- Public perceptions regarding the nature of the problem and suitable solutions.

1.3.7 Methods for Resolving Conflict in the Public Policy Process

Even when the stated objective of public policy is to maximize social welfare, elements of the policy formulation process itself may effectively prevent this objective from being achieved. Such elements can include (1) protracted conflict among those with a stake in the policy decision under consideration, (2) uncertainty about the events that might follow various choices, (3) public disagreement among technical experts about technical facts, and (4) inability of the policy process to judge when maximum welfare has been achieved. Other features of decision-making systems that can lead to suboptimal choices are disaggregated and decentralized decision making, inappropriate use of technical information resulting in part from confusion about which statements express facts and which express value preferences, and judgments by inappropriate persons. (Policymakers, for example, may begin to make technical judgments, while technical experts may tend to render value judgments if a protracted debate over policy objectives develops.)

Over the past decade, conflict has been characteristic of the process for formulating ocean dumping policy both at the national level, where general dumping policies are set, and at the local and regional levels, where site-specific choices are made. An unfortunate consequence of policymaking in an adversarial atmosphere is that it can result in outcomes that leave one or more parties to the decision worse off while the general welfare has not been improved. [6] Such would be the case, for example, if sludge dumping activities in the New York Bight were to be moved to a more distant site at a greater cost to municipalities <u>without</u> a corresponding improvement in the quality of the nearshore marine environment. On the other hand, cleanup of a contaminated dump site such as the New York Bight must start somewhere if the desire for environmental improvement is ever to be satisfied.

The problem that confronts the decision maker is one of making a decision under uncertainty in the face of multiple and conflicting objectives. A number of analytic methods exist for determining relative values and identifying trade-offs when all the effects of a decision are not readily quantifiable in monetary terms. These “decision aids” are aimed at identifying and quantifying the important elements of decision problems so that an “optimal” solution can be developed. [7]

The results of applying the methods of decision analysis to problems as complex as multimedia waste management have been decidedly mixed, and replacing traditional institutional structures for dealing with such problems with new, formal decision-making structures is premature. Instead, the methods of decision analysis can be of great value as supplements to traditional decision-making studies if they are accompanied by major efforts at public education or by the creation of ad hoc groups to review waste-management decisions by means of analytic policy methods. [8]

Lessons gained from studies of public decision-making processes can be applied to developing a number of principles relevant to waste management. For example, decision-making bodies broadly representative of the interests involved may succeed where more narrowly constituted bodies fail. Likewise, an initial definition of the problem stated in terms broad enough to encompass nontechnical, hard-to-quantify factors may prove more successful in the long run than would trying to limit consideration to those technical features of the problem that most lend themselves to quantification and prediction. There may be advantages to making the initial list of alternatives as broad as possible. The alternative finally selected can in many cases be a creative recombination of initially rejected alternatives. Because technical questions are best addressed by the technically trained and decisions are best carried out by those who work in and understand federal procedures, combining these two areas in order to reach a viable decision regarding a technical option is not any easy task. The question of how political judgments are best rendered (i.e., by elected public officials versus public referenda or by judicial-like proceedings versus administrative action), particularly when complex and controversial technical issues are involved, can also be approached through studies and methods designed to deal directly with the decision process itself. [9]

The greatest utility of analytic policy methods may lie in their power to break down complex decision-making processes into their component parts so that strategies for problem resolution and the criteria by which they are to be evaluated can be openly and explicitly formulated. In this way the sources of conflict can be accurately identified, and value judgments can be separated from technical disagreements to the greatest possible extent. The development of management schemes for wastes that are currently candidates for ocean dumping, for example, will no doubt require the weighing of relative risks across media and trade-offs among conflicting values. These are precisely the kinds of problems for which decision methods have been developed.

1.4 CONCLUSIONS

1. A goal of analytical methods is to describe the best set of solutions, given the criteria for judging and constraints on resolving the problem.

2. There is no threshold for information, that is, no definition of how much is necessary. No minimum amount of information is worth an undefined, large amount of money. The costs of obtaining additional information must be weighed against the benefits of acquiring it.
3. National waste-management decisions and site-specific decisions require different information bases. The information developed in site-specific and regional studies is not necessarily sufficient for national waste-management decisions.
4. Consideration of multimedia waste-management strategies may increase the complexity of analysis and the amount of information needed for decision making.
5. The waste generator who bears the true total costs of waste disposal activities has strong incentives to optimize both the scale and methods of waste-management practices. It is therefore appropriate to consider all options in multimedia assessments, including reductions in the quantity of wastes generated or changes in their form.
6. Because all decisions on policy have equity impacts, compensation mechanisms must be considered part of the decision to rectify equity imbalances. Economic analysis of alternatives entails comparison of benefits and costs and selection of the alternative having the highest net benefits (total benefits minus total costs). This benefit-cost criterion ignores the distribution of benefits and costs across groups. Environmental policy options may impose costs on one group while generating benefits for another group. In some instances those who bear the costs (if they are aware of the situation) will oppose the use of benefit-cost analysis. The problem of equity impacts waste disposal policy when an option imposes risks on one group while the benefits accrue to some other group. One might argue that equity considerations call for compensation to those who bear the risks associated with waste disposal. Compensation could be the dollar amount that would make the individuals willing to incur the risk in question. The possibility of compensation is relevant to political processes, since if adequate compensation is paid, in some cases neighbors of dump sites may no longer oppose the location of the dump. In some cases, however, compensation may have limited or no utility.

A policy of appropriately compensating losers associated with waste disposal alternatives could avoid the impasses brought on by the “not-in-my-backyard” syndrome. The compensation should ultimately be paid by those responsible for generating the wastes, although it should be recognized that the entity responsible for generation of the wastes may not in all cases be the entity most directly responsible for the process generating the wastes, e.g., the regulatory process may account for the waste being generated in the first place.

7. Uncertainties stem from many sources. As a result, they need to be identified and, where possible, quantified to make comparisons and recommendations meaningful. Otherwise, they proliferate and may compromise the ability to discriminate among waste-management strategies. However, care must be taken to consider the limitations of quantitative data in such a complex decision where qualitative variables are important.
8. Little agreement exists on how assessments of risk to human health are to be used in the waste-management decision-making process. Risk assessments should not be used unless associated benefits are also taken into account.

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[6] See R. Dorfman, H. G. Jacoby, and H. A. Thomas, Jr., eds. Models for Managing Regional Water Quality, Harvard U. Press, Cambridge, Mass., 1972, Chapter 2, for a discussion of methods for determining so-called “Pareto admissible” decisions with analytical models.

[7] Analytical decision models are discussed generally in D. Bell, R. Keeney, and H. Raiffa, eds., Conflicting Objectives in Decisions, Wiley, New York, 1977. Chapter 12 discusses the application of a number of decision analytic methods in a coastal zone planning context. R. Keeney and H. Raiffa, Decisions with Multiple Objectives: Preferences and Value Tradeoffs, Wiley, New York, 1976, develop in detail multiattribute utility theory, from which most of these analytic methods are derived.

[8] Numerous novel decision-making or review processes have been suggested for social choices that involve complex technological issues or questions of risk. The science court proposed by A. Kantrowitz (Science 156:763--764, 1967) has been criticized for relying on traditional adversarial models of dispute settlement (A. R. Matheny and B. A. Williams, Law & Policy Quart. 3:341-364, 1981). D. Bazelon (Science 205:277-280, 1979) has suggested that more discretionary review power be granted to the judiciary for dealing with technologically complex questions. K. Lee (Science 208:679-684, 1980) proposed a siting jury for locating a repository for high-level radioactive wastes. S. Ramo (Science 213:837-842, 1980) proposed that specially convened boards make decisions on regulating technological activities after a separate agency review of the positive and negative characteristics of the technology involved.

[9] L. Susskind and L. Dunlap (Environ. Impact Assessment Rev. 2:335-395, 1982) discuss the effects of problem definition and composition of decision-making body on the decision outcomes in several waste-management case studies. See also D. Nelkin, ed., Controversy: Politics of Technical Decisions, Sage Publications, San Antonio, Tex., 1979. Keeney and Raiffa, op. cit., Chapter 10, discuss decision problem solution by creative recombination of alternatives. H. M. Sapolsky (Science 162:427-433, 1968) noted that technical controversy can lead to risk-adverse voter behavior. In a study of the drinking-water fluoridation issue he noted that fluoridation was accepted much more frequently by communities in which public officials made the decision than where it was put to a public referendum.

2

Report of the Panel on Marine Sciences

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2.1 INTRODUCTION

Current national policy (Public Law 95-532, as amended) prohibits ocean disposal of municipal sewage sludge and limits ocean disposal of industrial wastes. This chapter deals with the evaluation of ocean disposal in terms of existing marine science.

The chapter considers only waste disposal by direct injection using specific engineering devices. However, the ocean receives waste from other pathways, including surface-water runoff transported by rivers and deposition from the atmosphere. While the assessment framework developed here is conceptually applicable to the disposal of radioactive wastes, the Panel on Marine Sciences has not given explicit consideration to this problem.

The assessment of environmental impact for a marine discharge occurs many times during both the planning and the operation stages. Predictions of environmental impact under any given discharge strategy are important not only in deciding on the environmental costs associated with a project but also in choosing an area where there will be fewer effects and in tailoring the predischarge processes to minimize the environmental impacts. Thus, the ability to make environmental assessments is important to the understanding of impacts and to the proper engineering of the disposal system (Figure 2.1).

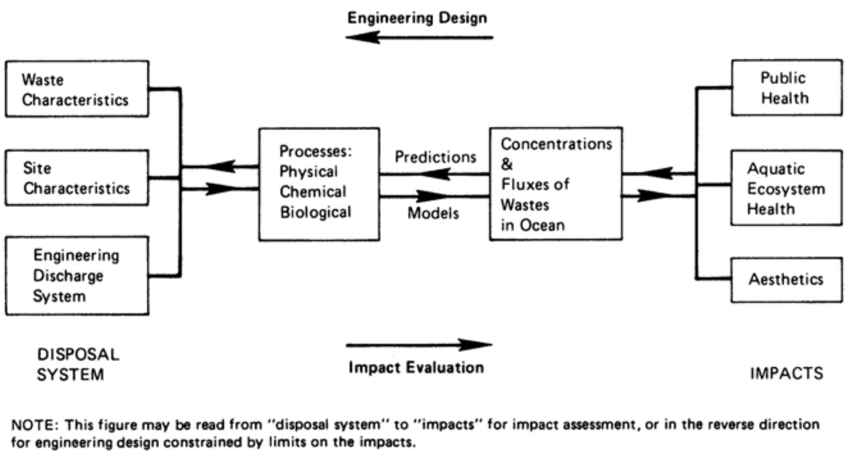

FIGURE 2.1 Engineering design for disposal system for impact assessment.

This chapter is concerned with the information needed to determine concentrations and fluxes of waste substances in the ocean environment. Concentrations and fluxes are the common currency between the physical transport/chemical effects community and the biological effects community that is used to estimate impacts on public health, aquatic ecosystem health, and aesthetics. The types of assessments that can be used to analyze those impacts, and the types of information needed for the assessments, are discussed in Chapter 3.

As indicated in Figure 2.1, a knowledge of physical, chemical, and biological processes is essential in order to proceed from an understanding of concentrations and fluxes of waste substances in the ocean to a description of an appropriate disposal system. Information about the processes that are critical in particular ocean disposal situations is linked together by means of predictive techniques or models. The focus of this chapter, then, is on the processes and the information needs associated with them.

The characteristics of wastes, sites, and engineering discharge systems can vary widely. Wastes may be municipal sewage sludge, industrial wastes, or material discharged from offshore drilling. Sites may include both nearshore and deep-water locations. Engineering discharge systems include outfalls at the ocean bottom and near-surface releases from moving sources (barges). The characteristics of wastes, sites, and engineering discharge systems are not treated separately in this chapter but are part of the information needed to describe a particular physical, chemical, or biological process.

The processes are not discussed in terms of a specific waste, site, or discharge system because different processes are important for different wastes and waste disposal systems. The selection of processes to predict concentrations and fluxes of critical substances will differ for each location, waste, and disposal system. As an example, the panel presents predictions on the proposed disposal of municipal sludge from an outfall in 300 to 400 m of water off the coast of Orange County, California.

2.2 OCEAN PROCESSES DETERMINING TRANSPORT, FATE, AND EFFECTS

2.2.2 The Process-Oriented Approach

To assess the overall effects of waste discharge in the ocean, we must understand the various physical, chemical, and biological processes that determine the effects. Some physical processes such as the turbulent mixing that occurs in a discharge jet happen rapidly and in a small space (in just a few minutes over a few tens of meters). Some biological processes like bioaccumulation in the food web may take years and affect many square kilometers. The rough classification of physical, chemical, and biological processes into near field versus far field and short term versus long term indicates that a range of space and time scales must be considered.

The phrase “transport, fate, and effects” is a shorthand reference to a process-oriented perspective on waste. “Transport” refers to those physical phenomena that move suspended or dissolved materials from one place to another. “Fate” refers to the group of chemical and physical processes that converts constituents from one form to another or causes them to be removed from the aquatic system. “Effects” covers the biological responses (including those that pertain to human health) that are in some way related to the discharge. Transport and fate are discussed in this chapter; effects are covered in Chapter 3.

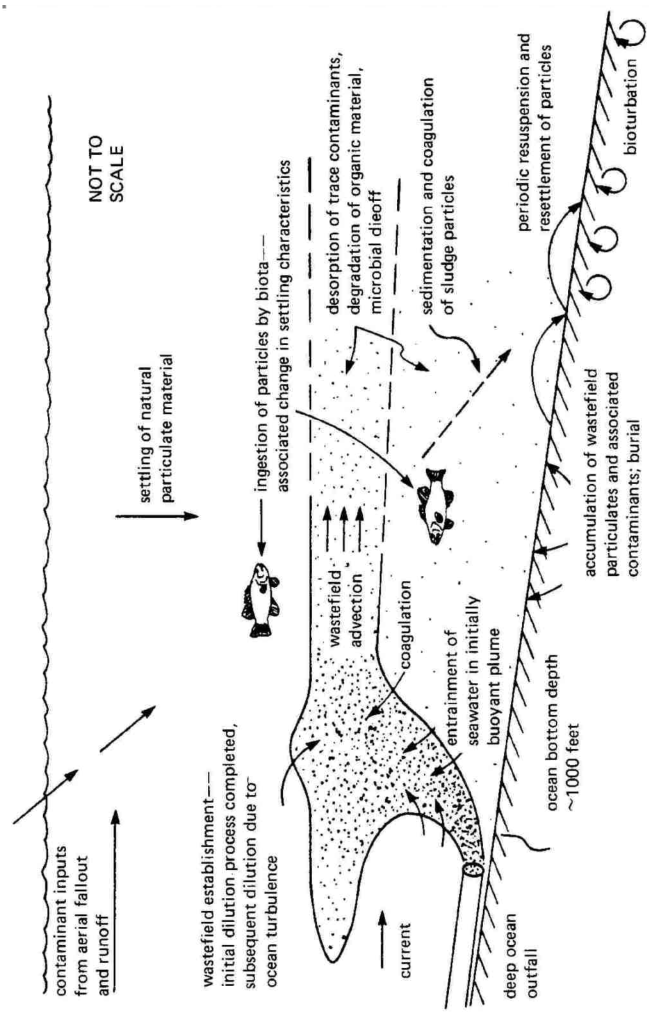

FIGURE 2.2 Factors affecting particulate fluxes and contaminant fates in marine waters. Note: Example shows an outfall discharging digested sewage sludge. Source: Brooks et al., 1982.

FIGURE 2.2 Factors affecting particulate fluxes and contaminant fates in marine waters. Note: Example shows an outfall discharging digested sewage sludge. Source: Brooks et al., 1982.

The following sections describe the processes involved in transport and fate and identify the information needed to allow predictions of transport and fate by conceptual or mathematical models. These predictions are for pollutant concentrations in the water column or their fluxes to the sea bottom. These derived data are the basic dosage parameters needed by biologists to estimate biological effects (see Figure 2.1). Figure 2.2 depicts an outfall discharge of digested sewage sludge. (It is not intended to be all-inclusive, nor is it drawn to scale.)

Out of this focus on processes comes the method of analysis called "critical pathways." As the various physical, chemical, and biological processes have become better understood, scientists have been able to determine which ones link a source with an effect, i.e., to determine the critical pathways from source to effect. This understanding has been used to set standards for exposure to radioactivity and is now being applied to other persistent contaminants (e.g., PCBs in fish intended for human consumption).

2.2.2.1 Physical Processes

Initial Dilution When a waste is discharged from a barge or an outfall pipe, the turbulent jet (usually buoyant) mixes with seawater, resulting in rapid initial dilution. Dilution factors, which may range from 10:1 to more than 1,000:1, can be predicted from mathematical models of the fluid dynamics of buoyant jets (Fischer et al., 1979). The process is normally considered complete when the buoyant jet ceases to rise in the water (that is, achieves neutral buoyancy) or the jet plume reaches the surface and the initial jet energy is dissipated. Plume submergence (or trapping below the thermocline) depends on ambient density stratification and can also be predicted by existing models. At the completion of the initial dilution process, the mixture of waste and seawater is a drifting cloud often referred to as a "wastefield." If it is at the surface, the wastefield may still be slightly buoyant, but if submerged, it will be neutrally buoyant (Fischer et al., 1979).

The following information is needed for predictive plume models and can be obtained by standard procedures:

- Volume flow rate and bulk density of the waste stream.

- Geometry of discharge pipes or nozzles.
- Ambient density profiles taken at various seasons of the year and under various current and weather conditions to describe the full range of density stratification (density versus depth). Density for each point is calculated from measured salinity and temperature.
- Local current speed and direction (frequency distribution). In strong currents the initial dilution is greater and the plume has a shorter rise if in a stratified ocean (for positively buoyant wastes released at depth).

Advection After initial dilution the wastefield is carried by ocean currents away from the discharge site. For a fixed-point discharge (i.e., an outfall), dispersal depends on there being a net current drift. Recording current meters placed at the wastefield depth provide the necessary data for calculating near-field transport. While near-field transport is important for flushing the discharge site, far-field transport by currents affects where the contaminants ultimately go. Along the coastlines, there may be very large-scale eddies (many kilometers across) as well as smaller motions over a wide range of scales. Depending on the length and time scales being considered, the large-scale eddies may appear to cause unsteady advective transport and to cause turbulent mixing, and the smaller motions may appear. Unless a waste is enclosed in containers or consists of fast-settling particles (like sand), wide dispersal by the currents must be expected. By analysis of the currents and other processes to be described below (e.g., sedimentation), we can estimate the extent of long-term dispersal and the flux of various substances to the sediments.

The determination of large-scale water circulation is difficult. Data from recording current meters will show spatial and temporal patterns, as well as means, variances, and coherences, but the motion of a wastefield cannot be predicted exactly because of the effects of irregular bathymetry and the impossibility of spacing current meters closely enough. Thus drogues, bottom drifters, and drift cards are often used in conjunction with current meters to define the statistics of possible waste trajectories. Satellite photographs are also useful because the colors and derived temperatures often show large-scale flow patterns of surface water, which also can move a wastefield.

Lateral Spreading and Mixing of the Wastefield Lateral spreading and additional wastefield dilution accompany advection. Spreading and diffusion of the wastefield are caused by (1) gravitational effects (the cloud gets thinner as it moves horizontally at the neutral buoyancy level); (2) shear dispersion (current speed and direction vary with depth, tending to pull the wastefield apart); and (3) diffusion due to turbulent eddies. These three effects are not easily separated, but all contribute to dilution and dispersal of the waste. Vertical mixing is much less than horizontal, except in the surface-mixed layer.

Mixing may be predicted by the advective-diffusion equation using published diffusion coefficients or the results of field experiments (for example, see Koh, 1982; or Brooks, 1960). In general, further dilution in this passive phase takes place slowly, and approximate diffusion coefficients are sufficient.

Sedimentation and Coagulation Sedimentation of waste particles acts against the dilution of a waste stream that is achieved upon discharge because sedimentation reconcentrates the wastes on the bottom. Because sedimentation of waste particles occurs simultaneously with passive wastefield advection and lateral spreading, the importance of sedimentation in decreasing wastefield concentrations and in increasing sediment fluxes varies. For digested primary sewage sludge, typically 50 percent (by weight) of discharged sludge solids will have fall velocities of less than 10^{-3} cm/s (~1 m/day). Therefore, the sedimentation process occurs over a long period, during which slowly settling particles can be transported tens of kilometers. Although waste-particle sedimentation is expected over a large area, experience from the discharge of municipal waste has shown that the flux (or sediment accumulation rate) is low outside a small region near the outfall where relatively fast-settling particles collect typically within about 1 km of the outfall (e.g., Hering and Abati, 1978).

Particles or oil droplets lighter than seawater rise toward the surface. There is a small (not yet determined) fraction of such material in sewage sludge that is removed to the maximum extent possible.

Coagulation (or flocculation) of sludge particles with each other or with natural marine particles is a process in which small particles coalesce into larger, faster-settling solids. Coagulation rates are increased by increased frequency of particle collisions. Thus, shear currents and small-scale turbulence enhance coagulation rates, and high dilution impedes them. Because increased salt concentrations favor coagulation, the dilution of a freshwater waste, such as sewage sludge, with saltwater probably favors coagulation. The distribution of particle sizes is important, with heterogeneous suspensions coagulating faster than homodisperse ones. How these factors interact in the ocean is poorly understood. Nevertheless, coagulation is important in determining the fate of particles because it can accelerate particle sedimentaion.

The prediction of sedimentation patterns by computer models (e.g., Hendricks, 1982; Koh, 1982) depends on information of three kinds: (1) the pattern of ocean currents and turbulent diffusion as described in the previous sections; (2) the mass emission rate of waste; and (3) the frequency distribution of fall velocities of waste particles in seawater. Fall-velocity distributions can readily be measured in settling columns by the pipette method. Waste material is diluted with seawater by a factor calculated for the prototype discharge (Brooks et al., 1982; Wang and Koh, 1982). This standard laboratory technique does not reproduce the proper flocculation effects, primarily because the turbulent shear of ocean water is not present. However, it is clear that settling velocities are increased by flocculation, which should cause the sedimentation pattern to exhibit higher fluxes near the discharge point.

<u>Sediment Resuspension and Bioturbation; Turbidity Currents</u> Sediment resuspension and bioturbation also play important roles as determinants of sediment characteristics in the vicinity of a discharge. Bottom currents at a discharge site can periodically reach speeds capable of resuspending surface sediments in the water. Such events will destroy the chronology of sediment deposition (by winnowing away or redepositing lighter material on the top), oxygenate surface sediments, release interstitial water, deplete dissolved oxygen in bottom water, enhance remobilization, and gradually transport sludge materials away from a discharge point. If a waste discharge point is on the continental slope or in (or near) a submarine canyon, sediments may be removed by occasional density or turbidity currents or even sediment slumps. Such removal can carry sediment deposits to deeper water (generally

considered desirable) but may cause sudden release of fine particles and entrapped trace contaminants to the bottom waters.

Bioturbation by burrowing animals (except in anoxic sediments) reduces vertical stratification among sediments, moves particles back to the sediment-water interface, and permits sediment oxygenation and other exchanges with the water column by increasing sediment porosity. Together, resuspension and bioturbation may greatly accelerate the release of contaminants from the sediments and the pore water to overlying waters.

Most models do not take account of resuspension or bioturbation, but if these factors could be included, the contaminants would be predicted to be generally more widely dispersed in sediments and water. The primary indicator of resuspension is the distribution of grain size in natural sediments at the sediment-water interface. The smallest sizes remaining at a given place are equal to the largest sizes (with their corresponding fall velocities) that are scoured and carried away by strong currents.

Light Scattering and Absorption In the case of near-surface discharge of turbid wastes (like sewage sludge), the transmission of light through the water may be significantly reduced by light scattering and absorption. This causes a decrease in the depth of the photic zone.

The reduction in light transmissions can be measured in the laboratory with mixtures of waste and seawater at appropriate dilutions.

Volatilization Volatilization represents the transfer of volatile chemicals from the water to the atmosphere. To be important in a waste discharge to the coastal sea, particulate and dissolved chemical species must be transformed into dissolved, nonassociated chemical species in which the concentration gradient between water and air favors water-to-air transport. Resistance to mass transfer occurs in both the liquid and gas films or on either side of the air-water interface. For most hydrophobic chemical species, resistance to transfer occurs in the liquid phase. Waste disposal that occurs below the thermocline may not allow chemical species to contact the upper mixed zone and to participate in air-water interactions. Waste discharges at depth do not result in losses of volatile components to the atmosphere unless there is significant vertical mixing.

2.2.2.2 Chemical Processes

The distribution of chemical constituents in a wastefield is determined by the method of discharge and the initial dilution described above in the section on Initial Dilution under Section 2.2.2.1. The initial ambient concentration (C_0) of any substance (after rapid dilution but before chemical changes are considered) is given by

$$C_0 = C_b + (C_w - C_b)/(S + 1),$$

where
C_b = background concentration;
C_w = concentration in waste stream;
S = initial dilution, as parts of seawater per part of waste.

The chemical and biological processes discussed below (as well as the physical processes already discussed) cause concentrations and fluxes to change over space and time.

Sorption/Desorption The sorption of inorganic and organic components on particles depends on the properties of sorbent, sorbate, and solution:

- Sorbent: concentration, surface area, organic carbon content, lipid content, surface properties, particle size, exchange capacity, composition.
- Sorbate: concentration, polarity, inorganic versus organic, charge, molecular (ionic) size.
- Solution: pH, temperature, ionic strength, total cation and anion concentrations of specific components.

For inorganic components (e.g., metals), sorption depends mostly on particle surface charge and area, on speciation in solution, and on concentrations of competing ions. Ion-exchange processes at the solid-solution interface also determine sorption of many inorganic species. For hydrophobic organic compounds, the surface area and organic carbon content of the particles, and the aqueous solubility of the organic component of interest, are probably the most important properties. Desorption of chemical species depends on the same factors.

For many organics, adsorption may be significantly faster than desorption. Chemical species sorbed on particles in a wastefield can be released as they equilibriate with the ambient aquatic environment. The results are that the concentrations of dissolved constituents increase and that the removal of contaminants by sedimentation is slowed.

Chemical Precipitation and Dissolution During the few minutes of the initial dilution of a buoyant plume discharged from an outfall pipe or a barge, chemical precipitation and dissolution are probably unimportant. They may be significant, however, as the plume is carried from the site by currents of oxygenated water and as particles are deposited in oxic or anoxic sediments. Morel et al. (1975) modeled the changes in metal speciation that occur when an anoxic municipal wastewater is discharged to an oxic ocean environment. The results are affected by precipitation/dissolution and by redox reactions. Metals such as manganese, which can be soluble in an anaerobic discharge, are precipitated when they are oxidized by dissolved oxygen in ocean waters. Metals such as zinc, present as sulfide precipitates in the anoxic wastewater, are dissolved as a result of sulfide oxidation by dissolved oxygen. Metals such as iron are transformed from one solid form [e.g., $FeS_2(s)$] to another [e.g., $FeOOH(s)$] by a combination of dissolution, oxidation, and precipitation reactions. Similar transformations can occur in sediments if the metals are deposited there.

Precipitation and dissolution control the transport and fate of several metals. They may also have some influence on the passage of phosphorus. Their direct effects on other pollutants are probably less important. Some species formed by precipitation reactions (e.g., oxidized precipitates of iron and manganese) probably participate in adsorption/desorption reactions with other pollutants.

Redox Processes Electron transfer (redox) reactions are significant in the transport and fate of several substances. Some important examples include iron, manganese, and sulfur. Changes in the redox states of these metals change their chemical speciation, which can, in turn, cause changes in the speciation and transport of other chemical constituents. For example, the oxidation of soluble iron (II) to insoluble iron (III) by oxygen results in a precipitate $Fe(OH)_3(s)$ that can scavenge metals, organics, and silica from the water.

Organisms have the ability to catalyze many chemical transformations. Microbiological catalysis of the oxygenation of manganese (II) to manganese (IV) leads to the precipitation of $MnO_2(s)$.

Both thermodynamic and kinetic effects govern redox processes in ocean waters. Significant factors include the concentrations of oxidized and reduced species in the waste and the ocean waters, pH, ionic strength, dissolved organic substances, particulate materials, temperature, and light. Thermodynamic calculations of the equilibrium state of the ocean system are feasible for some species (see, e.g., Morel et al., 1975). Reaction rates, which are difficult to predict, can vary widely depending on the substances involved and specific chemical conditions at a disposal site.

Photodegradation Photodecomposition of organic species in water depends on the sensitivity of a particular species to photolytic transformations and the availability of light having specific spectral properties at depth. Thus, controlling factors in photodegradation include the following:

- Light spectrum (especially in the ultraviolet)
- Light intensity
- Light attenuation at water depth by water absorption, particle reflectance
- Sensitivity of organic species to light of specific spectral properties
- Efficiency and rate of photon transfer
- Exposure time of specific light properties

In general, photoinduced transformations of organic species are important only in the upper reaches of the euphotic zone. The extent to which these reactions are important depends on the partitioning of photosensitive species between particle and dissolved phases and complexation in solution.

Complex Formation Complexes, also referred to as chemical species, are compounds consisting of a metal ion bonded to ligands. Speciation changes frequently result when solutions with different chemical compositions are mixed. Thus, the zone of initial mixing of a waste with seawater is a zone in which complex formation and changes in chemical speciation of trace metals and other constituents should occur. Because the reactivity or availability to organisms of a dissolved species is dependent on its speciation, the nature, extent, and rates of chemical transformations subsequent to input of a waste are important. For instance, copper is toxic to organisms as the free ion but is essentially unavailable when complexed with chloride, carbonate, or hydroxide, for example. Thermodynamic considerations indicate that only a small fraction of copper is uncomplexed in uncontaminated seawater. The complexes/speciation as well as the total amount of the input waste material should be known, but this information is frequently not available.

The second major zone in which speciation changes should occur is the sediment-water interface. Redox changes cause changes in ligand concentrations across the boundary and thereby cause speciation changes. For instance, a metal might change from being complexed with chloride to being complexed with bisulfide, polysulfides, or organic ligands. Such changes can result in increased mobility of a complex in sedimentary pore waters or a greater likelihood of its being deposited.

Because of the difficulty of analysis, few actual measurements have been made of metal speciation in the natural marine environment and almost none where a waste has been injected. There are thermodynamic models that predict how metal speciation may vary in response to changing ligand concentration. Unfortunately, these have not always agreed with each other.

<u>Other Chemical Transformations</u> Chemical transformations (other than those discussed above) include chemical alterations of organic species by hydrolysis reactions in solution. These chemically induced processes can alter organic pesticides having ester groups as part of their functional components. Hydrolysis rates depend on solution properties (pH; temperature; ionic strength; presence of catalysts, e.g., certain metals) and type of organic compound. General acid- or base-catalyzed hydrolysis of organic esters is common in aqueous environments and is somewhat understood.

2.2.2.3 Biological Processes

<u>Pathogen Mortality</u> Historically, one of the principal concerns in disposing of sewage wastes has been to make

sure that bacteria, viruses, and parasites present in the waste do not reinfect people. Control of human exposure to pathogens is accomplished either by disinfection of the waste stream or by discharge of the waste far enough from shore or deep enough to achieve sufficient bacterial (or viral) die-off or sedimentation before possible transport to shore. Pathogen mortality is discussed in Chapter 3.

Biodegradation/Biochemical Transformation Organic compounds in the aquatic environment may be biodegraded to carbon dioxide, water, and other compounds or transformed to other forms (e.g., DDT to DDE). The biochemical degradation/transformation of organics depends on the concentration of viable bacterial populations having the ability to perform the transformation. In addition, low ambient substrate concentrations, low environmental temperatures, and nonnatural organic structures greatly affect whether organics are transformed. Although microorganisms can be grown in culture to degrade/ transform all or most recalcitrant compounds, low substrate concentration and low temperatures in the aquatic environment make such transformations difficult to assess.

Bioconcentration/Bioaccumulation The partitioning of dissolved inorganic (e.g., trace metal) and organic species (e.g., tPCBs) into living organisms is referred to as bioconcentration. Trace elements exhibit a bioconcentration factor (BCF) of 10,000 or even higher, depending on mode of metal uptake, surface properties of the organism, and chemical speciation of the metal. The partitioning of organics into organisms depends on the lipid content of the organism, the water solubility of the organic, and the rate of organic transfer across the membrane surface. BCFs for typical nonionizable organics vary from 103 to 106. BCFs are proportional to the octanol-water partition coefficient (Kow) and inversely related to aqueous solubility. The relationship of Kow to BCF is:

$$\log BCF = 0.79 \log K_{ow} - 0.40$$

(Veith and Kosian, 1983). For many organisms, we can accurately estimate bioconcentration from knowledge of chemical concentrations (activity) in water and the BCF for organics that enter the organism primarily by par titioning across the membrane surface into the lipid pool. When ingestion of food is the dominant pathway, less success has been achieved in estimating bioconcentration.

Photosynthesis and Chemosynthesis Photosynthesis and chemosynthesis are the processes by which organisms take dissolved matter from solution to form particulate biological matter. Substances incorporated into organisms can become more concentrated than in solution. The energy to drive this process comes from light absorption (photosynthesis) or chemical transformations (chemosynthesis). Because biological material is composed of chemical elements that occur in relatively fixed ratios to each other, biomass production can be controlled by that element which is least available. Of the three major elements of phytoplankton biomass, nitrogen and phosphorus are relatively less abundant in oceanic surface waters than is carbon. As a result, their availability frequently determines the concentrations and growth rates of phytoplankton.

The processes of concentration into biological particles near the surface, the settling of these particles, and subsequent dissolving determine the oceanic distribution of many elements. These range from those depleted at the surface--including such nutrients as nitrogen and phosphorus and such trace metals as copper, cadmium, and zinc--to those substances depleted at depth, such as oxygen. Officer and Ryther (1977) noted that nutrient enrichment of surface waters in the New York Bight caused the production of plant matter at the surface and a subsequent settling of this organic matter to the bottom. The decay of this organic matter was responsible, they suggested, for the depletion of oxygen in near-bottom waters there.

The relatively fixed ratio of major chemical elements in organisms makes it possible to relate changes in oxygen, carbon dioxide, nitrogen, and phosphorus concentrations in solution to changes in phytoplankton biomass. The extent to which the approach involving fixed elemental ratios can be extended to estimate biomass concentrations of other elements and compounds is not clear. For some compounds, such as synthetic organics, there are chemical equilibrium relationships that make it possible to predict concentrations in organisms such as phytoplankton. Thus, our knowledge of the extent to which a substance is incorporated in new biological material depends on the material.

Information on rates of uptake is less available. For some major nutrients, adequate information exists. For most substances, it does not.

Respiration and Degradation of Biological Matter Biological respiration and degradation are major natural processes for the consumption of oxygen and release of substances from biological particles to solution. They are, thus, the opposite of the photosynthesis and chemosynthesis processes. A combination of biological respiration and degradation has been responsible for the decrease in oxygen and increase in trace-metal concentrations in deeper waters of the world's oceans and for the total depletion of oxygen in bottom waters of Lake Erie and at the New York Bight.

Oxygen depletion in rivers caused by organisms degrading sewage was an early focus of sanitary engineering, and adequate understanding of the depletion processes has been developed. An essential quantity in this description is the total amount of oxygen consumed in biological degradation of a material, or biological oxygen demand (BOD). Simple descriptions of oxygen/BOD consumption rates allow good predictions of oxygen and BOD distributions in rivers.

BOD in an ocean discharge is predominantly in particles. Significant degradation of particles can occur in the water column or in the sediments. Estimates from past studies and the results of experiments made with any proposed discharge material should provide adequate descriptions of water column degradation rates. Modeling oxygen/BOD consumption of material that has reached the sediments is more difficult because physical transport of oxygen can limit biological degradation rates. There have been in situ measurements of benthic oxygen consumption that can be used to estimate sedimentary oxygen demand under discharge conditions.

Fecal Material Organic matter composed of small phytoplankton growing near the ocean's surface reaches the deep ocean at rates far faster than predicted from settling velocities for the phytoplankton (days versus years). In part this happens because animals feeding on the phytoplankton drop fecal pellets that are much larger and settle several hundred times faster than do the phytoplankton. Such a mechanism explains the rapid appearance on the ocean bottom at 2,800 m of radionuclides entering the Pacific Ocean and in the Columbia River (Osterberg et al., 1963).

Incorporation of small particles into large fecal pellets can affect the fate of discharged particulate matter by increasing vertical transport rates. Decreased rate of dissolution may result from decreased area/unit weight rates of the particles.

Important questions still unanswered concern how significant this process is for the sedimentation of domestic or industrial waste components. Critical unknowns are the extent to which animal feeders can or will feed on discharged material and the importance of filter-feeding in subsurface waters.

2.2.3 Predictions of Concentrations and Fluxes

2.2.3.1 Basis for Prediction

The basis for predicting concentrations and fluxes is mass balance, or material accounting. The mass balance is usually represented by a model or group of models. (Model is used here in the broadest sense of the word and includes conceptual, mathematical, and physical models.) There are few problems in marine waste disposal for which a single model is adequate. No one model can be universally applicable to the variety of existing wastes, sites, and disposal systems that exist. Rather, the present practice of employing a hierarchy or suite of predictive models will continue. The essential ingredient in the development of the group of models to be used for concentration and flux predictions is identification of the processes that are important for the various predictions. Experience gained from other marine disposal activities and examinations of the chemical, physical, and biological processes may serve as an early screening tool in the identification of appropriate models. Order-of-magnitude analysis using waste, site, and system characteristics and knowledge of processes may further sharpen the focus on the important processes.

Numerical models that include components representing the important physical, chemical, and biological processes provide the opportunity to link (mathematically) these processes to yield an integrated response in terms of concentrations and fluxes. Unfortunately, the more complex the model the more difficult it is to gain insight from the model results. Consequently, numerical models may be developed to handle only portions of the predictions, and results from one model may be used as input or design guidance for another model. Time and space scales of physical, chemical, and biological processes often provide natural divisions in such modeling. Near-field and far-field models, and local and regional models, are examples of multimodel coupling.

Several models have been used in ocean waste disposal assessments to predict concentrations of substances. Transport trajectories and dilutions for the near-field and far-field behavior of buoyant waste discharges can be found from integral models described in Fischer et al. (1979). Koh (1982) has coupled sedimentation effects with the behavior of submerged sewage sludge plumes to predict waste-particle distribution on the ocean bottom in the vicinity of an outfall. Jackson (1982) employed a one-dimensional (vertical) model of transport processes to examine the impact on water-column chemical constituents of sludge injection at various depths in the Santa Monica-San Pedro Basin. He combined estimates of horizontally averaged physical transport processes in the basin with oxygen and nitrate demands of sediments and suspended sludge particles to predict vertical concentration distributions of oxygen, nitrate, particulate organic matter, and oxidized organic matter in the water column. Csanady et al. (1979) provided models of large gyre circulation interactions with waste dumping near the surface at the 106-Mile Ocean Waste Disposal Site (Dumpsite 106).

2.2.3.2 Information Needs for Prediction

The information needs for physical, chemical, and biological processes have been discussed in the preceding sections. There are, however, additional needs in the assembly of a prediction. The primary requirement is compatibility among the ingredients in the mass balance in terms of time and space scales. That is, chemical processes that have time scales of days should be matched with physical processes, such as transport by residual currents, having comparable scales. The determination of currents averaged over days may involve a different measurement strategy from that required for currents with shorter time scales. This emphasizes the importance of planning the prediction process before undertaking major measurement programs. Examination of the dominant processes may show that the region of interest can be modeled by assuming a one-dimensional geometry. Lumped or spatially integrated versions of connected processes and of substance concentrations are required for the predictions. For example, descriptions of "patchy" processes that are driven by local concentrations must be made compatible, if possible, with a model framework that uses spatially averaged concentrations. Special information needs regarding boundary conditions for several substances often result from the combination of various processes in a prediction of temporally or spatially averaged concentrations and fluxes. For instance, the prediction of the flux of a substance may require information on the concentration of the substance and the current at contemporary times.

The mix of information needs may change with each different kind of prediction for the same overall problem as a different process becomes the focus of the prediction. The prediction of the sedimentation of sludge in the local vicinity of a sludge outfall, for example, will require different kinds of information about physical transport than will a prediction of basinwide oxygen distribution in the water column as the result of the outfall. Formation of a prediction framework thus involves a reconsideration of physical, chemical, and biological processes, with an emphasis on the relationships among them in a context of common time and space scales.

2.3 CASE STUDY: PROPOSED DEEP-OCEAN DISCHARGE OF SEWAGE SLUDGE

The preceding sections indicate the existence of a wide range of waste and site characteristics, discharge systems, and physical, chemical, and biological processes. Figure 2.1 indicates that an analysis can be made for each pollutant of any given discharge at a particular site. To arrive at an engineering design, we must consider a range of sites, discharge techniques, and possibly predischarge treatment to modify the waste. The resulting matrix is huge.

In real engineering cases, however, it is quickly found that there are usually only a few critical requirements and analyses that determine the acceptable choices and rule out the unacceptable ones. In other words, when the critical design requirements are satisfied, other requirements may be met within a large margin. Thus, efforts in data gathering and research can be focused on the critical areas. Experience and engineering judgment are exercised at the beginning in reducing design options and planning oceanographic surveys.

This brief case study illustrates the application of the processes and modeling techniques described in Section 2.2. The case chosen is the proposed deep-ocean discharge of sewage sludge by the Orange County Sanitation District through a special outfall pipe terminating at 400-m depth about 12 km off Orange County in southern California. This project is not allowable under current state and federal laws and regulations and has been proposed as an experiment. A comprehensive research plan was prepared by the Environmental Quality Laboratory (EQL) of the California Institute of Technology to study the transport, fate, and effects of the discharge (Brooks et al., 1982).

The EQL report gives the basic information currently available on waste characteristics, site characteristics, and the engineering system (open-ended outfall pipe). Although the report describes extensive additional measurement and research needs, there is a sufficient base for the following discussion. This chapter carries the analysis only up to determination of concentrations and fluxes. These results can be used to analyze biological effects.

Development of an ocean disposal option for sewage sludge, as in this case, is iterative. If some effects are predicted or subsequently observed to be unacceptable, the plan will be adjusted or even abandoned. Adjustments can be made in the characteristics of the waste (more treatment or pretreatment), the site, or the method of discharge. The characteristics of the sludge stream are given in Table 2.1.

A 1:1 mixture of sludge and secondary effluent would be discharged from a pipe lying on the bottom 400 m below the ocean's surface. The warm, low-salinity mixture would rise in the denser seawater, mixing with seawater until it forms a sludge-seawater mixture with the density of the surrounding ocean. The height of rise and the initial dilution of this sewage plume vary with the currents. Rise height is expected to range between 50 and 110 m above the bottom and the initial dilution to vary between 450-700:1, seawater:sludge. The values used for this case study are a 50-m rise height and a 500:1 dilution. When the predilution of sludge with effluent is included as part of the dilution process, the initial dilution would be 1000:1.

TABLE 2.1 Characteristics of Projected Orange County Sanitation District Sludge Stream

Constituent	Concentration	Mass Emission Rate
Volume	-	3.0 MGD (11.4 x 10^{-6} L/day)
Total suspended solids	11,500 mg/L	1.33 x 10^3 kg/day
Volatile suspended solids	7,000 mg/L	73 x 10^3 kg/day
Biological oxygen demand	1,700 mg/L	17 x 10^3 kg/day
Silver	0.8 mg/L	9 x 10^3 kg/day
Cadmium	1.4 mg/L	16 x 10^3 kg/day
Chromium	5. mg/L	62 x 10^3 kg/day
Copper	16. mg/L	190 x 10^3 kg/day
Nickel	1.5 mg/L	17 x 10^3 kg/day
Lead	6. mg/L	70 x 10^3 kg/day
Zinc	20. mg/L	240 x 10^3 kg/day
PCB	50. µg/L	0.5 kg/day
CHC pesticides	4. µg/L	0.045 kg/day

SOURCE: Brooks et al., 1982.

Following plume rise, further mixing occurs as the plume drifts with the current. We expect the motion to be largely horizontal and parallel to local bottom contours. Turbulent diffusion is likely to be slower at those depths than near the surface. If we assume that the horizontal diffusion coefficient at depth is 10 percent of surface values, then, using results from Brooks (1960), turbulent mixing will provide additional dilution by a factor of 2 or 3 within 12 h.

2.3.1 Suspended Solids

Direct effects of sludge particles include turbidity in the water column and the formation of deposits on the ocean bed. Pollutants associated with sludge solids include pathogenic organisms and many toxic metals and synthetic organic substances.

A model for predicting the mass flux of solids to the ocean floor after discharge from an ocean outfall has been developed by Koh (1982) and applied to the Santa Monica Basin off southern California. The approach uses a mass balance to predict the effects of advection, diffusion, and sedimentation on the depositional flux of solids as a function of distance from the discharge site. Data requirements are the flow rate, solids concentration, and bulk density of the sludge to be discharged; the settling velocities of the sludge particles in seawater; and the ocean currents and density stratification in the region of the discharge. The measurement of ocean currents is difficult but feasible and necessary. Available laboratory measurements of settling velocities of sludge particles are sparse, and the range is very wide. Koh's conclusions are based primarily on horizontal turbulent diffusion coefficients of 2.5 km^2/day along the direction of the bottom contours and 0.1 km^2/day perpendicular to these contours, a vertical turbulent diffusion coefficient of 1 cm^2/s, and a particle settling velocity of 10^{-3} cm/s.

The results of Koh (1982) indicate that only a few percent of the mass of sludge particles discharged would settle within a kilometer of the discharge site; about one half of the solids would settle over an area of about 100 km^2. These results are comparable with actual observations at the Los Angeles County Sanitation District wastewater discharge at Palos Verdes (Myers, 1974).

These predictions of the transport and fate of sludge particles in ocean waters can be developed further. The model can be used to predict water-column concentrations. Predictions are limited, however, by uncertainties in the available data base and by the omission in the basic model of such important physical and biological reactions as flocculation and biodegradation.

2.3.1.1 Synthetic Organics

The synthetic organics discussed here are hydrophobic (i.e., with low water solubilities) and are preferentially bound to the organic sludge particles in the process stream. Typical median particle size for primary sludge particles is ~15 to 20 μm; particle size for secondary sludge will probably be less than 10 μm. The dilution processes will create a plume with a tPCB concentration of 50 ng/L. This compares with the background tPCB concentration of ~0.5 to 2 ng/L.

During plume advection and particle settling, the particles will be undergoing biodegradation according to first-order kinetics with a rate constant of ~0.05 to 0.1 day^{-1} (i.e., BOD). Also, synthetic organics will partition between sludge and natural particles and water to approach equilibrium. The partition coefficients between particle and water [Kp = (ng/g)p/(ng/g)w] need to be adjusted for organic carbon (OC) content (fraction of w^{+}) as OC of the particle:

$$\frac{K_p}{OC} = K_{oc}.$$

K_{oc} values for tPCBs and CHC pesticides are ~10^5 to 10^6. However, these may be approximated from octanol-water (K_{ow}) partition coefficients determined in the laboratory.

$$\log K_{oc} = \log K_{ow} - 0.21.$$

In essence, the lower the K_{oc} and particle concentration, the greater is the fraction of synthetic organic in the dissolved water phase.

The sludge particles containing high concentrations will desorb tPCBs and CHC, increasing the concentration in the dissolved phase. Ultimately, particles containing ~1 to 3 µg/g (smaller particle size, high OC content) will slowly settle out, resulting in a flux to the sediments. If the primary settling zone is ~100 km^2, the sludge discharge rate is 0.5 kg/day, PCBs and ~10 percent of tPCBs settle out in this region, and then the sediment flux of tPCBs will be ~5 $\mu g/m^2$ day.

The particle-bound and dissolved-phase synthetic organics may be degraded by bacteria, but little evidence exists that tPCBs and many CHC pesticides are degraded in the field under ambient conditions. Photodecomposition is not important below the photic zone (~60 m). Chemical transformations (e.g., hydrolysis) may be important for some compounds. Volatilization is not a factor with deep-water discharges. Therefore, the primary removal mechanisms for many organics (e.g., tPCBs) will be advection (dissolved and particulate) and sedimentation. Experience in the Southern California

Bight suggests that 85 percent of the discharged tPCBs cannot be accounted for by sedimentation in the region of the outfalls. It is likely that the water column is a major reservoir for them. Assuming a water residence time of 100 days and a plume thickness of ~40 m would give an average effluent dilution of 24,000:1. Thus, the initial sludge stream concentration of 50,000 ng/L will be diluted far field to a concentration of ~2 μg/L. Assuming sedimentation and other removal processes, far-field tPCB concentrations of several nanograms per liter are possible. Of this amount, ~70 to 80 percent will be in the dissolved phase because of the low particle concentrations of less than 0.5 mg/L.

2.3.1.2 Dissolved Oxygen/Biochemical Oxygen Demand

If we assume that secondary effluent mixed with the sludge stream has negligible biochemical oxygen demand (BOD), then the BOD in the layer due to the sludge discharge would be 1.7 mg/L--(~1 mL/L). Turbulent mixing would further reduce this to 0.5-0.7 mg/L within 12 h. Total oxygen consumption given unlimited time might be 50 percent greater, or 2.1 mg/L (1.5 mL/L). The dissolved oxygen (DO) content at those depths has the same approximate magnitude, indicating that local impacts might be significant.

Jackson et al. (1979) estimated sludge degradation rates under these temperature conditions as 1 percent/ day. If the ambient oxygen concentration is 0.5 mL/L, total BOD is 1.5 mL/L, and no mixing occurs subsequent to initial dilution, biological degradation would halve the oxygen concentration of the sludge-seawater mixture in 18 days. Particle removal by settling and mixing processes will decrease this impact on oxygen concentrations.

On the basis of these calculations, it seems possible that the naturally low ambient DO concentrations will be slightly depressed locally in the diluted sludge field near the discharge point. However, the sludge discharge system will be specifically designed to prevent significant oxygen depletion by various methods as more predischarge oceanographic information becomes available. Among methods of controlling DO depletion are (1) reducing the BOD mass emission rate by digesting the waste-activated sludge component, (2) reducing outfall depth to get the plume into water with more dissolved oxygen, and (3) employing larger predilutions of sludge with effluent.

2.3.1.3 Nutrients

Phosphorus (P), nitrogen (N), and carbon (C) are nutrients that stimulate biological production when they enter the photic zone. The nutrients potentially limiting growth are P (primarily in the form of phosphate) and N (primarily in the form of ammonium and nitrate). In the coastal zone, plant production is usually limited by nitrogen loading. In the case of the proposed sludge discharge, however, sludge nutrients will be confined to the 300- to 500-m depths, far below the photic zone depth of ~60 m. However, episodic upwelling can bring vertical nutrient gradients to the surface throughout the Southern California Bight, especially in the nearshore region.

The sludge discharge will result in the emission of ~136 t/day on a dry-weight basis. The composition of the sludge is as follows:

Total (T) P:	1 to 3 percent dry weight
TN:	2 to 5 percent dry weight
Organic (O) C:	25 to 30 percent dry weight

Thus, the proposed sludge output is ($t = 10^3$ kg = 1 metric ton).

Suspended Solids (SS) :	136 t/day
TP:	1.4 to 4.1 t/day
TN:	2.7 to 6.8 t/day
TOC:	34 to 41 t/day

These projected sludge discharge values can be compared with 1981 emissions from waste outfalls to the Southern California Bight as follows:

SS:	2.3×10^5 t/yr
NH_3:	4.1×10^4 t/yr
ON:	1.3×10^4 t/yr
TP:	9.5×10^3 t/yr
TOC:	5.8×10^4 t/yr

The proposed discharge represents <26 percent of the total nutrient discharge from 1981 wastewater outfalls into coastal waters.

The forms of phosphorus and nitrogen in sludge are not well understood. Chemical equilibrium models and laboratory measurements suggest that phosphorus exists in the form of solid $Ca_3(PO_4)_2$ and $Mg_3(PO_4)_2$, cellular inorganic phosphate, and smaller quantities of organic phosphates. Nitrogen will be in the form of NH_3 (or NH_4^+), NO_3^-, and organic nitrogen. Upon decomposition/mineralization of the sludge particles, P and N will be released in biologically available forms.

In the Southern California Bight, nutrient concentrations are lowest in the surface waters owing to biological utilization, peak at ~100- to 300-m depth owing to microbially mediated decomposition of settling particles (see table below), and generally decrease downward to the sediments.

Nutrient	Surface Water (0 to 60 m) µM	Middepth Water (100 to 300 m) µM
Phosphate	0.2	2-3
Nitrate	~0	30-40
Ammonia	0.3	0.1

Expected concentrations will be

Nutrient	Discharge Stream	1:1,000 Wastefield
TP	4-12 mM	4-12 µM
TN	17-43 mM	17-43 µM
TOC	250-300 mM	250-300 µM

Thus 1:1,000 of the discharge stream will perhaps double or triple the ambient concentrations at depths of 300 m. Far-field dilutions of an additional factor of 10 to 100 lead to additions much below ambient levels.

It is possible that >50 percent of the nutrients in the discharge stream will be in soluble form. Assuming that only 10 percent of the P, N, and OC in the initial sludge stream is deposited in the sediments within the ~100 km^2 predicted by Koh (1982), then P, N, and OC fluxes are as follows:

TP:	0.1-0.4 g/m^2 yr
TN:	0.2-0.6 g/m^2 yr
TOC:	2.5-3.0 g/m^2 yr

These data should be compared to natural nutrient fluxes to the sediments.

2.3.1.4 Metals

The characterization of sludge as far as trace metals are concerned is incomplete. Total concentrations (and ranges) of silver, arsenic, cadmium, chromium, copper, mercury, iron, manganese, nickel, lead, selenium, and zinc in the effluents have been measured. Distribution of trace metals in particles and in solution have also been measured for several sludges (Faisst, 1976). Morel et al. (1975) predicted from theoretical calculations that most metals form sulfide or oxide precipitates and thus are predominantly associated with the particulate matter of sludge. However, measurements in pore waters of reducing sediments have revealed concentrations of some metals and metalloids several orders of magnitude higher than their concentrations in oxygenated waters. Thus, the assumed association of metals with the particles may not be justified. In any case, total concentrations in the sludge effluents are many orders of magnitude higher than those in ambient seawater. Larger particles (faster settling; 10^{-1} cm/s) will collect within ~1 km of the outfall through sedimentation, whereas slower settling particles will fall over a larger area. Since metals are generally associated with the smallest particles, sedimentation will result in a preferential dispersal of small particles having generally higher metal concentrations. Following sedimentation, particles (with their associated trace-metal burdens) may be resuspended by bioturbation and strong currents, leading to further movement and dilution.

Chemical reactions between the particles and seawater can be as important as dilution is for metals. The injected sludge is organic-rich and reducing. Many metals would be in their lower oxidation states and could be complexed with the major ligands in the sludge. Seawater is oxidizing. Hence, on mixing, metals could become oxidized. This would affect their particulate-dissolved partition. There could be a substantial release of some metals to the seawater. For metals not subject to redox processes, some desorption could occur because of the different chemical compositions of sludge and seawater. Further, as organic-rich particulate matter containing metals reacts with oxygen in the water column, there would be a release of metals to the waters. Those metals such as cadmium and zinc that commonly have depth profiles in seawater similar to those of nutrients might be expected to show the largest releases to seawater. Jackson (1982) calculated the increased metal concentrations in seawater, assuming that the metals are released in proportion to the dissolution of organic matter. Such a calculation has been adapted to the proposed Orange County outfall (Table 2.2). Compared with the uncontaminated levels of cadmium, chromium, copper, lead, nickel, silver, and zinc, sewage disposal will not increase the ambient levels by more than a factor of 3.

Further release of metals to seawater could occur from remineralization at the sediment-water interface where organic compounds produced during the degradation of the sedimented organic-rich sludge could complex metals associated with the naturally occurring sedimentary material. These could be either resedimented, diffuse out, or released during sediment resuspension events. Pseudomonas bacteria extracted from Chesapeake Bay sediments have been found to be capable of alkylating tin and may be able to alkylate other metalloids. Such organotins are much more toxic to organisms than is inorganic tin. The extent to which metalloids become alkylated in sludge-rich sediments is not known.

Many trace metals have high concentrations in sewage sludge. It is not feasible to monitor all of them through the many possible chemical reactions that can occur after sludge discharge. Judgments must be made as to which metals and metalloids are more likely to affect public health or ecological communities. One possible way to eliminate some metals is to be found by looking at the areas around pre-existing sludge inputs. The Southern California Coastal Water Research Project (1982) has demonstrated that even though concentrations of cadmium, copper, and zinc are elevated in sediments and organisms adjacent to outfalls, the presence of a detoxifying complexing agent (metallothionen) in sea urchins and croaker fish enables these species to prevent concentrations of uncomplexed, toxic cadmium, copper, and zinc in their surroundings from entering sensitive cellular sites. In fact, less than 2 percent of the capacity of the metallothionen pool in animals near existing outfalls has been utilized. This has been interpreted as indicating that cadmium, copper, and zinc could be considered nonproblems.

TABLE 2.2 Comparison Of Predicted Soluble Trace-Metal Concentrations from Proposed Discharge with Those from the 1978 Water Quality Control Plan for Ocean Waters Of California (State Water Resources Control Board, 1978)

	Maximum Allowable Ambient Concentration, M	Predicted Maximum Soluble Concentrations, M, for Discharge at		
Element	(6-month median)	400 m	600 m	800 m
Cadmium	2.7×10^{-8}	9.3×10^{-9}	9.3×10^{-9}	9.3×10^{-9}
Chromium	3.9×10^{-8}	8.5×10^{-9}	8.6×10^{-9}	$7.1 \times 10\text{-}9^{-9}$
Copper	7.8×10^{-8}	7.6×10^{-9}	7.7×10^{-9}	6.6×10^{-9}
Lead	3.9×10^{-8}	8.0×10^{-10}	8.1×10^{-10}	6.8×10^{-10}
Nickel	3.4×10^{-7}	3.5×10^{-8}	3.5×10^{-8}	3.5×10^{-8}
Silver	4.1×10^{-9}	5.2×10^{-10}	5.2×10^{-10}	4.9×10^{-10}
Zinc	3.1×10^{-7}	1.5×10^{-8}	1.5×10^{-8}	1.3×10^{-8}

SOURCE: Jackson et al. (1979).

2.4 ULTIMATE FATE

Present studies of the fate and effects of contaminants rarely extend to spatial scales larger than several kilometers. Such scales are not always large enough to determine either fate or effects. For example, less than 10 percent of the trace metals that have been discharged from one southern California sewage outfall has been found in nearby sediments (Hering and Abati, 1978; Morel et al., 1975).

Discharged substances presumably stay with the water after discharge, either on slowly settling particles or in solution. They will travel with the different planktonic organisms in the water. There is little information on their interaction with these plankton. We can infer from the ability of DDT to accumulate in pelicans that there may be interactions. DDT was discharged for several years through a Los Angeles County Sanitation District outfall off Palos Verdes. Pelicans throughout the Southern California Bight accumulated it, and as a consequence were unable to maintain their population levels. Eggshell thinning and subsequent reproductive failures implicated the DDT from the outfall. This accumulation by pelicans implies that the sewage effluent was interacting with planktonic organisms because (1) pelicans must have derived their DDT from the surface fishes (such as anchovy and sardines) that they eat, and (2) surface fish accumulated DDT from the plankton in their diets or directly from the water. Thus, discharged waste can be in contact with plankton long enough to have an impact on these higher-trophic-level organisms.

The waste/seawater mixture created by a discharge does not simply drift into oblivion. In coastal areas such as the Southern California Bight, circulation and exchange with the open ocean can be inhibited by subsurface topography. Large-scale circulation rates as well as local advection determine contaminant concentrations over large areas. We expect slower water exchange in deeper waters because topographical constraints also increase with depth. Trace-metal concentration measurements in more than 500 m of water off the southern California coast by Barcelona et al. (1982) suggest that regional near-surface discharge has increased copper concentrations two to five times. This was in areas more than 15 km from the nearest major sewage outfall. Thus, far-field processes are important aspects of waste disposal. The spatial scale over which this occurs implies that releases from many distant sources combine to cause a substantial effect.

There is information on the rates at which trace metals are ultimately removed from the ocean by burial in the sediments. Transport to the sediments of naturally occurring trace metals is driven by adsorption to suspended particles and by incorporation in organisms. Characteristic times for the removal processes range from a few hundred years for lead, a highly reactive metal, to a thousand years for copper, a reactive metal, to 5,000 years for cadmium, a metal that has marine distributions similar to that of the nutrient phosphorus.

Comparable information does not exist for synthetic organics because they have not been in the ocean long enough. Chemical transformations as well as sediment burial are important removal processes. Their accumulation in biological matter suggests that transport to the sediments can be an important removal mechanism.

2.5 OVERVIEW

The Panel on Marine Sciences has identified its task regarding the ocean disposal option as one of assessing “our capabilities to make predictions of waste behavior in the ocean that will allow the estimation of impacts to public health, aquatic ecosystem health, and aesthetics.” The approach taken to assess these capabilities was that of outlining the framework used by marine scientists and engineers to predict spatial and temporal concentrations and fluxes of waste substances and ocean constituents impacted by wastes. The predictive framework can generally be used for any combination of waste, site, and disposal system design characteristics. The framework links predictions of concentrations of substances (such as pathogens, DO/BOD, nutrients, metals, synthetic organics, and suspended sediments) to disposal system description through consideration of physical, chemical, and biological processes. Critical substances and processes are identified for each disposal situation (specific waste, site, and discharge system) as one proceeds through the framework. An example of this activity is provided in an examination of a proposed sewage sludge outfall at a depth of 300 to 400 m off Orange County, California.

2.5.1 Assessment of Capabilities

Predictive capabilities exist now to provide first approximations in many cases of the concentrations and fluxes of substances critical to the examination of impacts of ocean disposal systems. Specific combinations of waste, site, and disposal system characteristics result in predictions with greater certainty than do other combinations. For example, predictions of transport and dispersion of wastes in the immediate vicinity of most discharge devices (near-field region) are generally well in hand, while predictions involving larger time and space scales are often less certain. Dissolved oxygen concentrations can be predicted with confidence in some cases, but knowledge of the processes governing the fate of synthetic organics is limited.

2.5.2 Information Needs Resulting from Prediction

In addition to the information needed for developing predictions of concentrations and fluxes, including model calibration, information needs are created by the prediction process itself. The primary purpose of a prediction is to link or integrate processes affecting concentrations in ways not obvious from independent consideration of each process. Examination of the responses of the linked system to variations in inputs (site, waste, and discharge system), as well as to the representations of the processes themselves, create new information requirements. Response and sensitivity studies provide insight into those data that appear most critical in the determination of concentrations and fluxes.

Early evaluation of predictions allows the collection of additional data to reduce uncertainty. Early prediction exercises are important to the design of programs for monitoring and assessing postconstruction performance.

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3

Report of the Panel on Land Disposal

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3.1 INTRODUCTION

Society has used land for the disposal of wastes since time immemorial. The history of earlier civilizations is reflected in ancient cities whose trash dumps became themselves the foundations for cities built as replacements. Not until the advent of public health concepts in the nineteenth century was any systematic attention given to the disposal of waste in ways aimed at minimizing harm to the public. Even after the most pressing aspects of waste treatment and disposal (e.g., sewage) were well recognized and modes of treatment established, almost another century passed before there was widespread recognition that land resources are finite and subject to multiple demands and that we could no longer indiscriminately dispose of unwanted residuals on the basis of institutional convenience. This recognition is reflected in the laws and regulations governing the disposal of wastes in or on the land.

The land option for disposal is difficult to understand because of the physical and chemical complexity of the medium, the wide range of regional land differences, and the long history of multiple land use in many regions that has altered many of its properties. Land is often in short supply; fills a wide variety of needs related to food, fiber, living space, recreational needs, and industrial development; and serves as a source of diminishing mineral and usable groundwater resources. We therefore need continuously to re-examine and refine our institutional mechanisms for waste disposal to avoid irrevocably committing land resources to a single-purpose use whose total costs will be greater than the benefits derived.

This chapter describes the criteria and information needed in considering land as a disposal medium. One set of criteria deals with the properties of the wastes to be disposed of and how these properties may affect the choice of a land option. A second set of criteria involves hydrologic, geologic, and other basic-properties of a land system that might affect or be affected by use of the land for disposal. The chapter also describes what information is needed to apply these criteria, where these data are obtainable, and what research is necessary to supply the data that do not exist.

The basic concerns with respect to land disposal are the long-term security of the disposal facility and the effects of both unexpected and routine discharges from the facility. Security is dependent to a large degree on the hydrology and geology of the disposal site, on whether the disposal process has been engineered to complement site conditions, on long-term maintenance and monitoring, and to a lesser degree on the toxic properties of the waste itself. The primary concerns in land disposal include, but are not necessarily limited to, two major categories: (1) long-term environmental effects, including contamination of surface or groundwater resources, potential threats to human health, and secondary effects on valuable natural or agricultural ecosystems, and (2) long-term commitment of land resources.

Consideration of land disposal options must include recognition of the significant time lag between waste inputs and outputs resulting from migration processes. Years or decades may elapse before a subsurface contamination plume moves far enough to create a problem. As a result, knowledge of the effects of recent land disposal practices, especially those focused on minimizing or eliminating contaminant migration, is limited. Many groundwater contamination problems today are the result of past disposal practices, and it is important to distinguish between those situations and the consequences of present-day practices. In other words, it is unfair and misleading to associate all groundwater contamination with land disposal inputs alone.

Ideally, information of all the types presented below would be used to model a disposal technique from the point of waste introduction to the ultimate effects on a receptor. The information needs discussed here are intended more as guidance than as a detailed methodology. This is because the relationships discussed are too complex to be reduced to a rigid modeling process. Often, even with unlimited resources of time and money, it will be impossible to simulate through testing or to estimate through calculation each and every phenomenon in a truly quantitative sense. Simplifying judgments on the major aspects of the situation will be necessary to reduce the problem to manageable proportions.

It may never be possible, for example, to predict from first principles the quantity of leachate from the mixture of wastes, the change in leachate after migration to a receptor, or the effect of the resultant mixture on a receptor. The problem can probably be made tractable only if simplified to account for major constituents of concern, the migration of which is influenced by major geologic and hydrologic phenomena. Sound scientific judgment, rather than cookbook methodology, must be used throughout the process.

3.2 WASTE PROPERTIES

Adequate understanding of the properties of waste is essential to evaluating a land disposal process. While properties like toxicity are intrinsic in defining potential environmental effects, other properties must be evaluated. The number and kinds of tests must be selected with the disposal method in mind. A number of the informational requirements listed below depend on facility and site-specific information, such as soil properties, to determine waste behavior. Generalized or typical information may be developed for order-of-magnitude comparisons (Table 3.1). However, further refinement of estimates requires the development of increasingly more detailed site-specific information. The amount of emphasis given the tests and the degree of accuracy sought are matters of judgment on the part of the evaluator.

Waste properties that can have a major effect on the acceptability of a land-based waste disposal site include pathogen content, levels of toxins, toxicant mobility, biodegradability/persistence, bioaccumulation, waste component interactions, phytotoxicity, incompatibility

TABLE 3.1 Waste Properties

Property	Availability of Data [a]	Relative Cost of Obtaining Data [b]	Comments
Pathogen content	Low	Medium	Especially critical for land application and agricultural use
Toxicity			
Acute	High	Low	
Chronic	Low	High	Major need is for organic mixtures
Toxicant mobility			
Water/soil	Medium	Medium	
Air	High	Medium	Organic data are far less available than are inorganic data;organic data are more expensive to generate
Biodegradability/ persistence	Low	Medium	May not be required if facility acceptable with no biodegradability
Bioaccumulation	High	Medium	Large data base on metals
Waste interaction	High	Low	
Phytotoxicity	High	Medium	Especially critical for agricultural use and land reclamation
Incompatability with containment system	Low	Medium	
Volume	High	Low	

[a] High, readily available; medium, fair data base; low, practically no data base.
[b] High, >$1 million; medium, $100,000 to $1 million; low, <$100,000.

with the containment system, and volume (Congress of the United States).

3.2.1 Pathogen Content

The presence of pathogenic agents in waste material will be a key factor in deciding on the acceptability of that waste for application to farmland or for other types of reuse. In those instances where the presence of pathogens does not preclude land application, it may restrict future use of the sites. Pathogens include bacteria, fungi, protozoa, viruses, and parasites. Some species of bacteria decline in number when exposed to soil. Viruses, however, can persist for a relatively long time in the natural environment. The most hardy of the pathogens appear to be parasites and those bacteria that form spores. These can survive in soil for years. Adding to the problems caused by pathogen survival are (1) the limited technology available for isolating and detecting viruses and other pathogens and (2) the introduction of increased concentrations of parasites into the environment in the United States and elsewhere.

Bacteria and parasites are detectable in most waste materials. Determination of virus die-off has been hampered by the difficulty in preparing samples and in detecting and enumerating viruses to assure sufficient data for statistical analyses. More work needs to be done on identifying natural indicators of viruses and formulating improved methods of virus recovery that can be used to determine survival and persistence. Toxicity must be defined in order to determine the potential problems if containment of the waste cannot be assured. Toxicity must be considered in conjunction with mobility; measurements should be based on the toxic mobile constituents rather than on the waste itself.

3.2.2 Acute Toxicity

The definition of what constitutes acute toxicity to mammals via oral ingestion or to aquatic organisms is of only moderate importance in evaluating a land disposal plan. Generally, significant toxic effects will occur at lower than lethal dosages (or environmental concentrations). Consequently, acute toxicity information should be used primarily as an indicator of unusually toxic materials that may require special scrutiny.

Acute mammalian and aquatic toxicity data are available for a significant number of chemical compounds, and many data are available for inorganic materials (metals). Virtually no information is available, however, for mixtures of wastes that may be encountered in practice. Relatively inexpensive and standardized test methods are available for determining the acute toxicities of waste mixtures, however.

3.2.3 Chronic Toxicity

In evaluating most land disposal plans, potential exposure over a long time to low concentrations or dosages in groundwater or surface waters must be considered. Chronic mammalian toxicity is a concern where there is potential for discharge to potable groundwaters or surface waters. Chronic aquatic toxicity may also be important where there is potential for discharge to surface waters. Estimates of the concen- tration that will produce harmful doses to target species are needed for the evaluation process.

Chronic toxicity information is more readily available for inorganic materials than for organic materials. There is little documentation of the chronic effects of mixtures. While test methods are available to determine the possibility of chronic effects, they become prohibitively expensive when applied to higher life forms (Brusick, 1978; EPA, 1979).

3.2.4 Toxicant Mobility

The ability of land-disposed waste or its constituents to move in the environment depends on the liquid nature of the waste as well as on its leachability (solubility of constituents), but movement may also depend on climate and soil properties. Soil systems may retard the mobility of or sequester certain constituents. Relative mobility in soils may be measured directly through partition coefficient determinations or indirectly through the use of such parameters as cation exchange capacity. An additional aspect of mobility is volatility, or the ability of waste constitutents to enter the atmosphere. Volatility can result in toxicity or odor problems. Volatility may also affect the transport of pathogens that are attached to particles.

There are few standardized test protocols or tabulations that delineate mobility directly. The Environmental Protection Agency (EPA) has developed an extraction procedure (Appendix II, EPA Toxicity Test Procedure, 40 C.F.R. §261.24) designed to simulate solubilizing effects and leachate production, but it is not site-specific (Lowenbach et al., 1977). While some partition coefficients are available for materials in certain soils, few are available for complex mixtures (EPA, 1979; Farrah and Pickering, 1977; Ramamoorthy and Rust, 1979). Methodologies have been developed to predict volatilization from liquid waste in impoundments, but there has been little work in predicting air emissions from covered landfills or land farms (Hwang, 1980; Thibedeaux, 1981). Consequently, additional research is needed to develop standardized test methods that simulate or allow calculation of mobility.

3.2.5 Biodegradability/Persistence

Toxic constituents may be mobilized from wastes that are applied as soil amendments or placed in shallow land burial, and the biodegrada- tion products of such mobile constituents may be more toxic than the parent compounds are. On the other hand, rapid disappearance of the parent compound may be beneficial. The long-term persistence of toxic components of wastes will be a key factor in determining disposal methodology. The probability of direct exposure of biota (including humans) is much greater with disposal as soil amendment than with disposal in landfills. Where there may be indirect discharge to surface waters from a land disposal facility, degradation of nonpersistent compounds may be significant. Where only groundwater is affected, assessments of persistence may not be fruitful because biodegradation may be slow.

Data and standardized methodologies are available to define rates of biodegradation and degradation of compounds in aqueous systems in treatment plants or surface waters. Far less information is available for these phenomena in soil systems or groundwaters. Even though experimental work is being done, considerably more research effort is needed to develop standardized test methods.

3.2.6 Bioaccumulation

The chronic toxic effects of organic and inorganic constituents in wastes are magnified by bioaccumulation. Uptake and incorporation of toxic components of wastes by foodstuffs intended for human consumption will limit the acceptability of wastes as agricultural soil amendments. However, wastes may be acceptable as soil amendments in silviculture. The potential for long-term releases to surface waters of substances that are subject to bioaccumulation may constrain shallow land disposal.

Considerable information is available on the ability of metals to bioaccumulate. While data on organic constituents are more limited, some estimates are available, such as those based on the octanol-water partition coefficients (Leo et al., 1971). Continued research effort is needed to relate experimental assessments to actual environmental effects.

3.2.7 Waste Component Interactions

Where significantly different types of waste are disposed of together, consideration should be given to reactions or interactions that may significantly alter waste properties. Mobilization or production of more hazardous constituents may occur, for example, when acids are added to metal hydroxide precipitates. Or, solvents may dissolve otherwise immobile toxicants.

The possibilities of violent incompatibilities are well known or can be anticipated by chemists. Solubilization and the like, while not cataloged, can be estimated for simple systems and simulated for more complex systems.

3.2.8 Phytotoxicity

The nutrient contents (nitrogen, phosphorus, potassium) of solid wastes can limit land application of wastes. Most of these limitations involve degradation of groundwaters as a result of excessive addition of nutrients.

Phytotoxicity may involve salts, e.g., sodium, or such trace elements as boron, cadmium, copper, nickel, and zinc. Phytotoxicity determinations have been made for years in the agricultural community, and sufficient information is readily available through agricultural extension services.

3.2.9 Incompatibility with Containment Systems

Incompatibility with containment systems is of primary importance in landfilling. Strong acids, bases, salts, and some liquid organics may interfere with the containment capabilities of natural or man-made barriers to migration. Considerable information has been developed on the ability of synthetic membranes to contain wastes (EPA, 1979). Recently, studies have been conducted on the compatibility of various concentrated organic materials with clay soils (Anderson et al., 1981; Sanks et al., 1975). More work is needed to better define these interactions for normally expected leachates.

3.2.10 Volume

In choosing land disposal options, both the amount of material requiring disposal and the generation of that material must be considered. Often, the continuous disposal of large volumes of waste on land is limited by climate and physical site, so that some form of interim storage between production and disposal is mandatory.

3.3 SITE PROPERTIES

Several objectives are associated with determination of site properties for evaluating land disposal of wastes. A major goal is to ensure that groundwater and surface water in and around the disposal site do not become contaminated beyond acceptable limits. Other objectives concern appropriate land-use patterns (past, present, and future) and preservation of cultural resources, such as archaeological or recreational areas, unique ecosystems, or special habitat features. Thus, site characterization may be undertaken from the viewpoint of hydrologic, land use, or ecological resources.

3.3.1 Hydrologic Considerations

Because of the linkage between land disposal operations and the potential contamination of groundwater and surface waters, a useful framework for identifying hydrologic information needs can be based on waste migration processes (Figure 3.1). It is assumed that

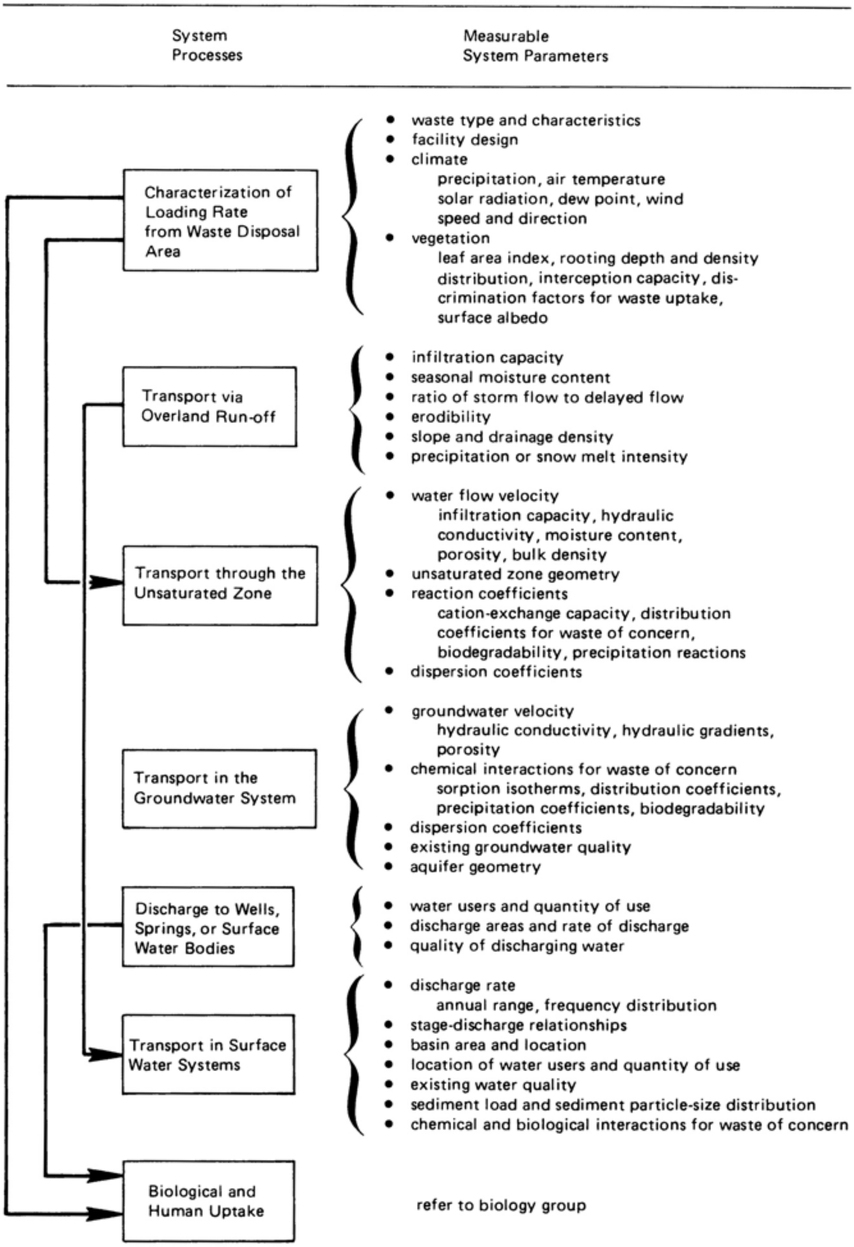

FIGURE 3.1 Waste migration process.

wastes will be present in a form that allows migration and transport. If sufficient measures are taken to prevent significant leaching or migration of wastes, many data will no longer be needed. Thus, there may be a trade-off between investment in pretreatment or engineered barriers and site characterization that is sufficiently detailed to allow prediction of waste concentrations over the period of concern for a given site. The measurable parameters listed in Figure 3.1 represent data needs, but not all of them will be required for every site. Furthermore, the level of detail or frequency of measurement will depend on an estimation of the degree of hazard associated with migration, and some judgment will inevitably be required.

3.3.1.1 Characterization of Loading Rate

The first step in the selection of a disposal method is to characterize the loading rate at potential disposal sites. Parameters relate to (1) land spreading, (2) injection wells, (3) shallow burial, and (4) a deep repository. For purposes of identifying key site properties, the second and fourth options may be considered a subset of the more general problem, which includes the potential pathway from the land surface through the unsaturated and saturated zones to eventual discharge in surface waters.

Migration Rate The waste form, loading rate, and selected disposal option will influence the migration rate. However, the information needed to evaluate the transport pathway will remain relatively unchanged regardless of the disposal option. The characterization of migration begins by specifying the loading rate of water and potential contaminants. This may be as simple as specifying the average flow rate and concentration of contaminants in a waste stream that is spread on the land surface or injected into a deep well or as complex as estimating the gradual decomposition and differential leaching of contaminants under the influence of temperature, aeration, and infiltrating or ponded water.

Migration Processes in Unsaturated Zone Hydrologic transport in the unsaturated zone may be divided between surface and shallow subsurface runoff pathways, and deep percolation of soil water and contaminants to the saturated (groundwater) zone. Climatic, vegetative, and topographic (slope and drainage density) properties of the site, together with soil hydraulic properties, control the degree of saturation of the near-surface soils and the partitioning of both precipitation and effluent waters between runoff and percolation.

Climate and Vegetation The most important long-term control on water movement is evapotranspiration, which includes evaporation from the soil surface and uptake and release of water by vegetation. Evapotranspiration controls the moisture in soils between precipitation and water application events. The rate of movement of water and associated contaminants through the soil depends in large measure on the soil's moisture content. The infiltration rate will generally decrease as moisture content increases, and the amount of additional water that can be stored in the soil and released slowly will also decrease with increased antecedent moisture. Thus, in considering the application of large amounts of water to a disposal site, either by irrigation or injection-type methods or storm events, a knowledge of water uptake by vegetation or direct evaporation is essential to quantitative estimation of subsequent transport pathways. In some situations, evaporation is the key factor in determining acceptable loading rates. Many methods are available for computing actual evapotranspiration and soil moisture status (e.g., Federer, 1982; Spittlehouse and Black, 1981; and Swift et al., 1975), which are the essential components of a water balance at the disposal site. The minimum data requirements on climate for modern computational models are daily solar radiation, maximum and minimum daily air temperature, and precipitation. More-complex models also require daily dew point temperature and wind movement. With the foregoing variables, a combined mass transport and energy budget approach will also allow simulation of snowpack accumulation, sublimation, and melt in addition to complete quantification of the site's water budget (Huff et al., 1977). Because rainfall intensity affects the partitioning of surface and subsurface flows, precipitation totals may be needed at hourly or shorter intervals for some water budget and transport simulations where the hazard is great. Often, data collected at a nearby site can be used.

Vegetation plays a dual role. It determines interception and transpiration losses of precipitation inputs, and it may bring buried contamination to the surface by water extraction from the soil. Vegetation species are altered from their natural state at many disposal sites; the consequences of such change should be explicitly estimated. The leaf area index (LAI: leaf area per unit of land surface) is a key variable in quantifying both interception loss and transpiration. Because of seasonal variation in the LAI, the full annual cycle must be known for continuous simulation of water budgets. Some models also use depth of rooting as well as root density distribution with depth. This is necessary information if surface-water withdrawal (as a function of depth) is to be used to estimate uptake of contaminants. Such estimates must also utilize discrimination factors to account for exclusion of some dissolved contaminants at the root-soil interface, which affects longer-term stability of a site. Natural succession may result in vegetation that taps into contaminated soils and translocates undesirable materials to the surface. A characterization is needed of climax vegetation species, including such properties as LAI and rooting depth and distribution, as well as estimates of discrimination factors and potential for transfer of contaminants into the human food chain.

3.3.1.2 Overland Runoff

Drainage and rapid runoff are primarily determined by the physical properties of the soils and underlying geologic materials, although topography also exerts an influence on both. For unsaturated materials, the drainage rate is a function of pore-size distribution and moisture content. If an impeding layer such as a natural clay or hardpan is present, the entire hydrologic cycle at the disposal site may be confined to the soil above the impeding layer. Given the soil porosity, it is possible to estimate the holding capacity of the overlying soils before saturation and surface runoff would occur. In addition, the relationship between the soil's moisture content, hydraulic conductivity, and water potential must be known to permit continuous estimates of infiltration rates and soil-water distribution, which in turn can be used to determine application rates of wastes. A general range of rates by season would be determined in the early stages of evaluating a land disposal option.

Runoff per unit area of disposal site can be estimated from stream discharge measurements at multiple points along streams draining the disposal area. This allows determination of the water budget, including estimated recharge rates to groundwater reservoirs and detection of anomalies in groundwater (low-flow) discharge. Continuous measurement of stream discharge at one or more points within or adjacent to the site will provide the necessary data and a basis for estimating flood frequency. At the very least, determination of discharge rates within the local and regional drainage system will allow estimates of expected concentrations of contaminants for comparison with applicable water-quality standards. Superimposed on the acceptable application rates would be natural precipitation events.

The movement of water as a carrier of contamination is governed by advection processes and by diffusion and dispersion relative to the carrier system. Perhaps of most concern is rapid surface or shallow lateral sub-surface movement that would result in rapid transport of contaminants to surface waters (streams, lakes, estuaries). These problems, once recognized, are generally amenable to engineering solutions, such as runoff collection and treatment systems. These have been employed in conjunction with shallow-land injection of sludges in California (Robson, 1974).

In some cases, however, successional processes may have to be prevented in order to maintain disposal-site integrity. For example, only shallow-rooted vegetation can be allowed where landfill caps are constructed to prevent infiltration. In such cases, site security will be of greater concern than will recovery.

An associated but also controllable process is erosion of contaminated surface materials. The erosion process may involve either windblown entrainment of contaminated sediments or waterborne transport of eroded clay and silt particles. Here, both topographic considerations (slope) and drainage density (slope length to developed channels) are important considerations. These characteristics are easily determined from topographic maps. In both of the foregoing situations, the thickness of the unsaturated zone above the water table or impeding layer affects the erosional process. Soil depth to an impeding layer of less than about 1 m will generally indicate that a site is unsuitable for land disposal unless engineered control of runoff is a part of the design or the climate is sufficiently arid to dissipate applied water before runoff occurs.

3.3.1.3 Water-Flow Velocity

Percolation of water or waste through the unsaturated zone is governed by the same physical parameters that control infiltration. To predict rate and direction of water movement, three subsurface hydraulic properties should be known. The first involves the standard soil physics characterization of “intergranular properties” of soil and bedrock. This includes porosity, bulk density, and variation of soil water potential and hydraulic conductivity with moisture content. The second includes characterization of fractures, macropores, or relatively large (compared with capillary pores) openings within the soil structure that may transmit water rapidly along pathways (vertical and horizontal) that do not necessarily coincide with expectations, based on assumptions of a homogeneous, isotropic medium (e.g., Hammermeister et al., 1982). Fracture density, aperture size distribution, orientation frequency, and interconnections should be determined if applicable (e.g., Neuzil and Tracy, 1981; Sledz and Huff, 1981). Although the amount of water or waste that travels such pathways may not be great, velocities can be several orders of magnitude greater than that for intergranular flow, thus raising the possibility of localized high concentrations of contaminants at significant distances from disposal sites.

Finally, the structural geology and stratigraphy of the intended site should be examined. This includes such features as synclines, anticlines, faults, bedding planes, and buried stream valleys, which may either collect or divert water or waste and force it to move along pathways that would not be predicted by simple analysis of piezometric gradients. Geologic maps of the depth to bedrock, as well as stratigraphic sections in the area of interest, will be useful in this evaluation.

Examination of the specific variables involved generally implies site-specific field studies using well-established field methodologies for measurements. At the early stages of option evaluation it may be impractical to collect such site-specific data. However, more general evaluations are possible. Soil surveys are generally available for all sites in the United States.

Often, the hydraulic properties of each soil type have been determined and provide a reasonable starting point for calculations.

Rainfall and runoff measurements can be used to obtain a rough estimate of evapotranspiration losses. Analysis of representative streamflow hydrographs can be used to further estimate relative natural partitioning of flow between groundwater and more rapid storm runoff.

Once deep percolation has been estimated, a basis exists for estimating recharge to aquifers. If this is coupled with simple descriptors of soil-water chemistry, such as distribution coefficients and dispersion coefficients for various contaminants, the contaminant loading associated with deeper percolation can also be estimated.

3.3.1.4 Transport in the Groundwater System

The movement of a waste or leachate in the saturated zone is a function of groundwater velocity, biological and chemical interactions between the media and the waste, the dispersion coefficient that characterizes the subsurface media, and diffusion. To quantify the rate of waste migration and to predict spatial and temporal concentrations, these parameters must be known.

<u>Groundwater Velocity</u> Groundwater velocity is a function of the hydraulic conductivity distribution in the saturated zone (aquifer), the porosity of the aquifer, and the hydraulic gradients. The hydraulic conductivity of aquifer materials is conventionally estimated by aquifer (pump) tests. This technique provides a good estimate of hydraulic conductivity within a small distance of the pumped well in relatively uniform media. However, it provides poor estimates in complex media and no estimate of hydraulic conductivity outside the radius of influence of the pumped well. A hydraulic conductivity distribution is conventionally estimated spatially for large-scale problems on the basis of a limited number of aquifer tests and on the basis of geologic logs by assuming that similar lithologies have similar hydraulic conductivities. These methods can provide good estimates of the hydraulic conductivity distribution in aquifers that are composed of relatively uniform materials, but they usually provide poor estimates for aquifers composed of nonuniform materials. Hydraulic conductivity in the

saturated zone can be altered by waste interactions with the media, but these processes are in general poorly understood, and empirical testing is the only method for determining their importance.

Effective porosity in unconsolidated aquifers can be estimated accurately by simple laboratory tests, but the effective porosity of most consolidated aquifers cannot be estimated accurately by any simple methods. In most consolidated aquifers the effective porosity (the ratio of the volume of interconnected pores to the volume of material) is predominantly a result of fracturing and dissolution, and the pores are not evenly distributed throughout the medium. Therefore, effective porosity cannot be estimated accurately from small core samples. Techniques have been developed for estimating the effective porosity of fractured media using fracture density, aperture size distribution, orientation frequency, and interconnections (Neuzel and Tracy, 1981; Sledz and Huff, 1981).

The hydraulic gradients in the saturated zone can generally be accurately estimated from piezometers installed in the saturated zone to measure both vertical and horizontal hydraulic gradients. Changes in vegetation, and filling and contouring during disposal operations, can alter recharge rates and cause substantial changes in hydraulic gradients. A rough estimate of these changes can be made by using analytical or numerical models.

<u>Chemical and Biological Interactions</u> Sorption and precipitation are generally the dominant chemical reactions that alter a waste in the saturated zone. The equilibrium distribution of a waste between soil and water is conventionally characterized by a distribution coefficient. Together with porosity and bulk density of the medium, the distribution coefficient is the simplest common empirical measure of mobility. The distribution coefficient for a given waste and medium is empirically determined by laboratory tests. There is some suspicion, though, that distribution coefficients measured in the laboratory do not apply in the field. Other chemical reactions and transformation and biological reactions that may occur between a waste and the medium are generally characterized by degradation coefficients.

Diffusion is an important process in waste migration only when groundwater velocities are low. Diffusion coefficients for many wastes have been determined empirically. Since diffusion coefficients for most wastes fall within a small range, the consequences of diffusion can generally be assessed relatively accurately.

Dispersion Coefficients Dispersion in porous media refers to the spreading of a stream (continuous source) or slug (instantaneous source) of contaminant as it flows through the subsurface. If a spot of dye is injected into porous material through which groundwater is flowing, the spot will enlarge in size as it moves downgradient. More specifically, in a three-dimensional Cartesian coordinate system where the average groundwater velocity is parallel to the x axis, a sphere of dye moving horizontally along the x axis will undergo longitudinal spreading or dispersion parallel to the x axis and transverse dispersion parallel to the y and x axis.

Scheidegger (1961) demonstrated that dispersion in a porous medium is a function of the velocity distribution and a fourth-order tensor termed the dispersivity. The dispersivity of an isotropic porous medium can be defined by two constants--the longitudinal and the transverse dispersivity. Dispersivity in a groundwater flow system is predominantly a function of heterogeneities in the hydraulic conductivity distribution. Measured dispersivity coefficients are a function of the scale of measurements, and the dispersivities for large-scale problems can usually be determined only by simulation. Laboratory tests are generally useless unless they are run on a large scale (Sauty, 1980). Typical values for dispersivity coefficients are reported in Anderson (1979). Simulation studies by Pinder (1973), Robson (1974), Konikow (1977), Andrews and Anderson (1979), and others have provided estimates of dispersivity coefficients for various aquifer systems. For generic studies it will be necessary to estimate a probable range for the dispersivity coefficients based on values reported in the literature.

Groundwater Discharge to Wells, Springs, Streams, and Oceans Contamination of groundwater does not present a threat to human health or biological organisms until the groundwater discharges to wells or to surface waters. The rate of groundwater withdrawal at or discharge to wells can usually be determined by field surveys. In almost all land disposal siting studies, it is necessary to determine how groundwater is used in the vicinity of the landfill and how much is used. Future groundwater use is difficult to project, and faulty predictions introduce uncertainties into long-term evaluations.

The rate of groundwater discharge to surface waters is generally difficult to determine accurately, but this information is essential for estimating waste fluxes to the surface-water environment. Groundwater discharge to springs can be measured directly, but most groundwater outflow occurs as nonpoint discharges. Water-balance-type models, using either analytical or numerical solution techniques, are conventionally used to calculate groundwater discharges to surface-water bodies.

To determine the waste concentrations that will result from groundwater discharges in surface-water bodies, it is necessary to know the flow and mixing characteristics of the surface-water bodies.

3.3.1.5 Transport in Surface-Water Systems

Two factors are associated with transport in surface-water (freshwater) systems: (1) migration rates and chemical and biological interactions within the stream or estuarial channels and (2) water resource utilization, such as proximity of discharge points to municipal or industrial water intakes or sensitive aquatic habitats. The former determine the temporal and spatial distribution of possible contaminants or nutrients throughout the aquatic system, while the latter deal with potential pathways into the human food chain.

<u>Migration Rates and Chemical and Biological Interactions</u> Experience has shown that discharge of wastes, nutrients, or contaminants into surface waters will probably result in a buildup of contaminants in sediments and biota, but spatial distribution will vary with the situation. Contaminant scavenging can be characterized in terms of the relative flow rates of the input and the receiving waters, the chemical composition and form (such as oxidation state or biological availability) of the waste or leachate and receiving waters, and the physical and chemical properties of streambed and suspended sediments. Because of the usually strong interaction between the water column and bed sediments, sediment transport characterizations (particle size distribution, exchange properties, and physical characteristics of the stream channel) must be specified. Continuous simulation models are available for examining the physical transport

processes and can be used as a framework for detailed specification of data needs.

Water-Resource Utilization Withdrawal of water resources in the vicinity of disposal sites inevitably raises the question of the contribution of disposal operations to the water's contaminant loading. As noted earlier, the fate of wastes or leachates that may be discharged from land disposal sites is a function of the physical, chemical, and biological conditions in the receiving waters as well as the relative magnitude of the input. An order-of-magnitude estimate of concentration may be made by simple conservation of mass calculations involving the flux of waste or leachate from the disposal site and the total water flow at the point of intake. Estimates can be refined by considering degradation, uptake reactions, and ambient flow properties, such as velocity and sediment transport capacity. The most practical approach to water-use considerations is to focus on specific indicator contaminants and their potential impact on downstream users.

3.3.1.6 Summary of Hydrologic Considerations

The hydrologic transport information needs are generally as listed above; it may be expensive and time consuming to collect the data. It is assumed that the nature of the waste constituents and their chemical form (especially the associated hazard) will dictate the degree of detail or observation frequency required. There are clear trade-offs between site performance characterization, predisposal waste treatment, use of engineered barriers, and the relative hazards associated with waste. The most common approach starts with a “worst case” involving minimum waste pretreatment and engineered barriers. If computed contamination levels for this case are unacceptable, more refined analyses are undertaken until a satisfactory engineering option has been found or until all options have been rejected. The main difficulty with such an approach lies in determining the worst case. From a hydrologic viewpoint, assumptions of homogenous isotropic aquifers and average annual (or even seasonal) flow rates for a given source are not valid. Much more emphasis should be placed on extreme events and on the anisotropic properties of the site, such as preferred flow pathways associated with aquifer structure and fractures, macropores, or faults.

3.3.1.7 Land-Use Considerations

Irrespective of other factors utilized in site selection for land disposal, current and prior use of the land and contiguous areas may need to be factored into the decision process. The presence of unique or fragile ecosystems that could be impacted on by either the construction or operation of a land disposal site needs careful consideration. Special situations involving habitats that contain rare or endangered species must also be evaluated. Cultural factors that require analysis include the presence or proximity of cultural resources, such as archaeological sites, recreational areas, population distribution, and past and current land-use practices.

3.3.2 Terrestrial Ecological Considerations

Because terrestrial ecosystems include communities with very different levels of resilience (deserts, rain forests, chaparral, grasslands, alpine habitats), it is difficult to generalize about the ecological effects of land disposal options. Even if soil conditions are undamaged, successional processes and rates are very different. In most cases the greatest damage results from activities associated with disposal. Once the disposal program is completed, restoration of surface conditions and natural succession may lead to recovery. In other cases (e.g., desert, riparian, marsh, or cypress habitats), however, in which propagules are not readily available or that need nurse trees or special habitats for colonization, artificial reclamation may be necessary. This is especially true for large areas in which dispersal limitations mitigate against adequate succession.

For land disposal involving surface amendments (sludges, liquid wastes), a primary concern is the buildup of potentially toxic substances in the soil zones where they may be taken up by plants or animals and thereby enter food chains. Particularly worrisome are organic compounds (e.g., PCBs, PBBs) that are readily transferable. Here the potential damage to the ecosystem per se has not been examined, but the ecosystem processes serving as vectors of transport to human consumption must be assessed. The significant role of microorganisms in bioaccumulation and food chain transfer needs to be understood.

Information on surface and subsurface drainage patterns is needed to evaluate the potential impacts of disposal on adjacent but different ecosystems. Examples of such potential problem situations are adjacent marshes or other water bodies that are habitats for waterfowl, fish populations, or other wildlife. Marshes are natural discharge points for regional groundwater flow systems. They are, therefore, more likely than other types of ecosystems to receive any contaminants or residuals that have become entrained in groundwater.

3.4 FACILITY DESIGN PROPERTIES

To determine the relative merits of a disposal system, the properties of a waste and the disposal setting must be considered together. This is because the relative hazard a waste may present and the relative cost of the disposal method will depend to a major degree on site-specific conditions. Site conditions, in turn, are dependent on the engineered design as well as the natural geographic, hydrologic, and geologic setting. Engineered design and operation may complement natural site conditions, improving the disposal method.

Characteristics that can have a major effect on the desirability and acceptability of a land-based waste disposal site include location; site selection considerations; engineered containment; segregation of wastes; maintainability of the site; site development, monitoring, and closure.

3.4.1 Location

The basic considerations in assessing the location of a site are accessibility (Can the waste be delivered to the site at the rate at which it is generated?) and proximity (Can the waste be transported to the site from its point of origin?). A site with ideal properties may be unacceptable if economic, social, or institutional constraints limit its accessibility. Transportation of wastes, especially those clearly recognized as hazardous, over long distances can create economic, social, or insti tutional concerns that override the benefits of a proposed disposal site.

3.4.2 Site Selection

The acceptability of a specific site can be enhanced by the ways in which waste is transported, stored, or treated prior to its incorporation within the soil. Often, barge or rail transport over long distances and pipelines over short distances can make a specific site accessible when truck transport is restricted for social/economic reasons. Certainly, barge or rail transport can expand the economically possible transport distance.

Most land application disposal sites and agricultural use sites are limited by climatological conditions to less than continuous use. Some type of interim storage is therefore required to assure the maximum capability of such sites. The proposed rate of application or emplacement of waste at the site will depend on the temporary storage facilities.

Often, use of a specific site is limited to a certain type of preconditioned waste. Landfills, for example, are often limited to wastes that are at least 25 to 50 percent solids. Injecting liquid wastes (<10 percent solids) into site soils sometimes is more economical and has less environmental impact than do spreading and disking dewatered waste.

3.4.3 Barriers to Waste Migration

Barriers to waste solids or liquid migration may be installed to improve natural sites. Such barriers may be used as liners to restrict leachate migration or as covers to restrict infiltration or percolation of water into the site.

The compatibility of the waste or its leachate with the barrier is of primary importance, since detrimental changes in the ability of the barrier to restrict migration will reduce the desirability of the site.

Tests of compatibility involve the following determinations:

- Permeability of the barrier to waste where preventing migration is important;

- Cation exchange capacity where immobilization of waste constituents is important;
- Potential for catastrophic failure of the barrier (seismic concerns or gross incompatibility of barrier);
- Thickness of the barrier, which will affect its ability to sequester waste constituents or define worst-case containment period.

Tests for determining the compatibility of anthropogenic materials with the barrier should be designed to determine loss of physical strength or deterioration.

3.4.4 Waste Segregation

Physical segregation of waste types may affect the mobility of various waste constituents and should be based on waste compatibility determinations described earlier.

3.4.5 Maintainability

Long-term containment ability is determined primarily by infiltration, percolation, or seepage barriers. The need for and the possibility of maintaining such barriers must be considered. Maintenance may thus be a considerable cost of disposal.

Another aspect of maintainability is monitoring. It may be necessary to monitor the integrity of the system through sampling of groundwater, for example. Such a need would constitute a significant long-term cost.

3.4.6 Site Development

Site development extends from the time a site is designated as a disposal area to the time when disposal activities have ceased. Consideration of this time continuum is especially critical for subsurface disposal, where the site may ultimately become a dedicated area with limited options for alternative future use, such as a site for disposal of hazardous wastes. It becomes less important for those sites where wastes are added as a soil amendment. Tasks that must be considered in site development include the following:

- Step-by-step stabilization of surface areas where waste has been emplaced through contouring and installation of surface seals to minimize infiltration and erosion.
- Use of techniques or construction to minimize infiltration and maximize surface-water diversion or runoff.
- Segregation of waste types (e.g., biodegradable versus nonbiodegradable) to minimize subsequent surface subsidence that might result in disruption of surface seals and slope continuity. Segregation of wastes can also result in a more effective surface-water and groundwater monitoring system, in that the required intensive monitoring of hazardous constituents can be localized to provide “early warning” of any breach of containment. Less intensive monitoring would be required in areas containing less hazardous wastes.
- Incorporating projected postclosure land use. Even where a site is considered “dedicated” and unavailable for future human activities (such as construction or recreation), all possible alternative land uses (e.g., for wildlife, silviculture) should be considered. Site maintenance keyed to land use should also be considered. The most economical alternative, for instance, might be to periodically cut surface growth in order to prevent the growth of deep-rooted plants. Sites for the disposal of hazardous materials will require the maintenance of a shallow-rooted ground cover that will not intercept the waste.

3.4.7 Monitoring

Effective monitoring systems are essential to establish the integrity of the disposal systems. Monitoring systems should be designed to take into account the properties of the waste (mobility and indicator constituents), the site's physical constraints (geologic and topographic considerations, especially as they relate to the geohydrology of the site), and environmental and social/institutional issues (ecological concerns, nuisance development, and regulatory limitations). Monitoring programs should always be one step ahead of site development to establish background conditions and to allow for additional programs as the site develops. An integrated monitoring system may be needed even after closure.

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4

Report of the Panel on Biological Effects

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4.1 SITE EVALUATION

4.1.1 Introduction

In order to evaluate the effects of waste inputs on various ecological systems, it is necessary to recognize and accommodate the key issue of multiple scales within ecosystems. This involves explicit consideration of spatial scales, e.g., the extent of the area under impact vis-à-vis the total areal extent of similar habitats. It also involves consideration of temporal scales, such as acute versus chronic exposure patterns and various times for recovery of affected systems. Finally, it involves organizational scales, e.g., impacts on single populations versus effects on higher-level processes such as primary productivity.

This chapter compares land and sea options for disposal of wastes from these ecological perspectives and provides a basis for comparison of ecosystems.

4.1.2 Marine Site Evaluation

The impact on the ocean resulting from discharged waste depends on the composition and volume of the waste and on the dispersal and transport characteristics of the site selected for disposal. Clearly, the distribution, fate, and effects of waste inputs are governed by the physical, chemical, and biological processes that alter the chemical forms of the waste and their bioavailability. These processes are discussed in detail in Chapter 2. Sewage sludge has been routinely discharged or dumped in the sea at several sites along both the east and west coasts of the United States. From these disposal activities, data bases are available that can be used to begin to evaluate the biological impact of waste disposal in the marine environment. Although our discussion is focused on the impact of sewage sludge disposal, the input of chemical contaminants from the disposal of dredged material and industrial wastes will result in similar effects in the marine environment.

4.1.2.1 Nearshore Disposal

Sewage sludge is discharged to U.S. coastal waters either by pipeline (commonly used on the west coast) or barge (primarily used on the east coast). Although the discharge method will generally determine the initial dilution of wastes, subsequent dispersal and transport will depend on advective processes. Along the coast of southern California, the city of Los Angeles discharges sewage sludge (1 percent solids) through a pipe 75 cm in diameter at a depth of 100 m along the rim of a submarine canyon 10 km from shore (Bascom, 1982). Initial dilution of the wastes by 10^2 occurs, and further dilution and transport of the waste plume are achieved by passive advection and lateral spreading (Brooks et al., 1982). Because of the highly stratified water column at this site, the waste plume usually remains at a depth below the pycnocline.

Contaminants of biological concern, such as pathogenic microorganisms, trace metals, and xenobiotic organic compounds, are primarily associated with particulate material, and transport of the sludge particulates is controlled by the same phenomena as is the transport of natural sediments. Only 10 percent of the sludge solids discharged settle on the bottom within 5 km of the discharge site; the extent of transport of the remainder of the material remains uncertain (Mearns, 1981).

On the east coast of the United States, sewage sludge (3 to 5 percent solids) from metropolitan New York City is released by barge to the shallow (25 m), seasonally stratified waters of the New York Bight Apex (Gunnerson et al., 1982). Initial dilution is by a factor of 103 to 10^4. The particulate phase of the waste accumulates along the pycnocline, and further horizontal transport is accommodated by advection of surface waters (Duedall et al., 1977). When stratification in the water column is diminished, horizontal transport of wastes is reduced, and deposition of wastes takes place (Hatcher et al., 1981).

Sludge particulates and associated contaminants, including pathogenic microorganisms and toxic compounds, accumulate in Christiaensen Basin, a depositional area to the southeast of the New York Bight Apex. Boehm et al. (in press) have recently documented the fate of organic contaminants, and Krom et al. (in press) have analyzed metal inputs to the New York Bight system. After the waste particles have accumulated on the bottom, the chemical form and bioavailability of metals and organic compounds may be modified by biological processes such as microbial degradation and bioturbation, as well as by chemical processes such as sorption and desorption reactions, oxidation, and dissolution. Deposit-feeding and suspension-feeding organisms may also ingest waste particles, enhancing the transfer of wastes and, potentially, of pathogenic microorganisms throughout the biotic environment. Metabolism of xenobiotic contaminants by both bacteria and multicellular animals may result in the production of more toxic metabolites, particularly when these compounds bind to cell membranes or macromolecules within the cell (Brown et al., 1982; Stegeman, 1981). The availability of these metabolites to benthic fauna or bottom-feeding fishes is a further concern (Malins and Collier, 1981; Malins et al., 1980).

Environmental concern with waste disposal in the ocean is focused on (1) the uptake and accumulation of pathogenic bacteria and viruses in commercially harvested species destined for human consumption, (2) the accumulation and transfer of metals and xenobiotic compounds in marine food chains, (3) the toxic effects of such contaminants on the survival and reproduction of marine organisms and the resulting impact on ecosystems, and (4) the release of degradable organic matter and nutrients to the ocean, resulting in localized eutrophication and organic enrichment.

In studies of the benthic impact of sewage and sewage sludge disposal from the outfalls on the southern California coast, the Southern California Coastal Water Research Project (SCCWRP) concluded that the area of most severe degradation, measured by changes in the benthic community and concentration of xenobiotic compounds, was limited to a small zone (0.01 to 8.4 km^2) in the vicinity of each outfall (Bascom, 1982; Mearns and Word, 1982). The degree of degradation was correlated with the rate of emission of suspended solids. A larger area around each outfall (0.01 to 94 km^2) was characterized by altered community structure in comparison with unimpacted stations. Data on sport and commercial fish landings from these same areas indicate either no effect of waste discharge or, in some cases, enhanced yields in areas adjacent to outfalls (Mearns, 1981).

Boesch (1982) reviewed the impacts of waste disposal on benthic communities within the New York Bight system for altered diversity and abundance and for energy flow and interactions of the benthos with higher trophic levels. He reported that the most severely altered area in the New York Bight Apex was restricted to a 10 to 15 km^2 area west of the sewage sludge disposal site on the margin of Christiaensen Basin. This zone is characterized by dense populations of only a few macrobenthic species, such as sibling species of the polychaete Capitella. Areas beyond this zone are characterized by dense populations of macrofauna typical of muddy-fine sand habitats. The high densities of macrofauna may be the result of organic enrichment, and a transition zone of approximately 240 km^2 of enriched benthos lies between the severely impacted and unaffected areas (Pearson and Rosenberg, 1978; Boesch, 1982).

Boesch (1982) suggested that alterations in the macrobenthos have resulted in reductions in populations of the predominant food items of demersal fishes and invertebrates; and, although increased productivity is apparent in the benthos, little productivity is transferred to higher trophic levels. There is also evidence to suggest that within the “enriched zone” the dominant macrofauna are capable of enhancing sediment-water exchange through bioturbation and sediment resuspension; and thus increased rates of microbial decomposition and nutrient regeneration may prevail. The latter two processes may result in localized conditions of low dissolved oxygen and accumulation of nutrients in the benthos. Boesch (1982) concluded that the altered benthic community within Christiaensen Basin is better able to cope with organic enrichment than is the indigenous community, but it is less suitable for support of higher trophic levels. Similar changes have been observed in the southern California Bight, and only generalist feeders among demersal predator populations appear to be unaffected by alterations in benthic communities (Allen, 1975; Word, 1979).

In addition to the high level of concern about toxic chemicals and pathogens in the marine environment, there is also concern about the release of degradable and nondegradable organic matter and nutrients to the ocean. If these substances are discharged in sufficiently high concentrations to oceanic areas of poor dispersion and mixing energy, depletion of oxygen as a result of the high rate of microbial degradation may occur. Eutrophication of coastal areas from nutrient enrichment may result in changes in species composition and dynamics of marine communities. Mearns et al. (1982) have recently reviewed the effects of nutrient and organic enrichment on marine ecosystems, focusing primarily on the data base available for the New York Bight Apex. Coastal waters of the New York Bight and adjacent estuaries receive high annual inputs of organic carbon, nitrogen, and phosphorus from multiple sources, including barge dumping of sewage sludge and dredged materials, inputs from the Hudson-Raritan estuary, and other coastal and atmospheric inputs (Mueller et al., 1976). In the New York Bight Apex, seasonal and annual variations in productivity and stratification of the water column may lead to periods of low dissolved oxygen or anoxia in the benthos, such as that experienced during the summer of 1976 following a bloom of Ceratium tripos. Nutrient enrichment has also been observed in the Southern California Bight (G. Jackson, Scripps Institution of Oceanography, La Jolla, California, personal communication)

On both the southern California coast and in the New York Bight, there are many sources of contaminants and nutrients, so biological effects cannot be attributed to the impact of dumping of sludge alone. Understanding the impact of other point sources is necessary to predict overall degradation of a receiving area. Clearly, the impact of waste discharges depends on the volume and composition of waste to be discharged and on dispersal characteristics at the site of discharge. Low-volume inputs from a small coastal city will not have the same impact as inputs from a large metropolitan area, in either total volume or contaminant loading. These factors must be taken into account in future permit decisions for ocean dumping.

4.1.2.2 Deep-Water Disposal

Deep-water disposal of wastes, such as sewage sludge, offers the advantages of greater dilution and dispersion, reducing the potential return of wastes to humans and reducing the potential impact on living resources in nearshore coastal areas. Two deepwater sites have been proposed for receiving sewage sludge: the 106-Mile Ocean Waste Disposal Site (Dumpsite 106) located 106 nautical miles southeast of New York Harbor on the continental slope in the northwest Atlantic at a water depth of 2,000 m; and the proposed Orange County deep-water disposal site located off the coast of southern California at a depth of 300 to 400 m.

Dumpsite 106 is typical of slope water regions of the northwest Atlantic, and experience with industrial waste dumping at this site provides a background of mixing and dispersal characteristics of waste inputs. Initial dilution and dispersion of wastes will be similar to those measured for barged wastes in the New York Bight, but the greater depth and proximity to the Gulf Stream ensures greater horizontal transport (O'Connor and Park, 1982). Despite a limited number of investigations that suggest that deposition rates of sewage sludge to deep benthic areas would be minimal, no accurate information exists on the potential deposition rates of sewage sludge to deep sea benthic areas in the vicinity of Dumpsite 106, the extent of the area of deposition, or the resulting impact on benthic systems.

Predictive transport models of the pipeline discharge from the proposed Orange County deep-water disposal plan indicate that initial dilution of wastes will be 5×10^2, or greater, depending on the prevailing current regime and the height to which the submerged plume may rise. Further mixing is accomplished by advection and lateral spreading. The most critical difference between other outfalls off the southern California coast and this proposed outfall is its proximity to the oxygen minimum layer, and the potential effects of high biodegradation rates on biota acclimated to a low ambient oxygen concentration (1 mg/L). Brooks et al. (1982), in conjunction with a baseline study conducted by the Southern California Coastal Water Research Project, have developed a comprehensive research program to address the feasibility and impact of this particular disposal option.

The paucity of documentation for the disposal of wastes at deep-water sites makes predictions of ultimate biological and/or ecosystem effects difficult to assess; further research is required for evaluation.

4.1.2 Terrestrial Site Evaluation

The use of terrestrial and affiliated freshwater ecosystems for the disposal of anthropogenic wastes involves issues that are quite distinct from those of the marine situation. Because of the more intimate contact that humans may have with the wastes, compared, for instance, with contact from open-ocean disposal, the central goal of disposal for terrestrial systems is containment of the various components of the wastes. In general, a properly sited terrestrial waste disposal system incorporates an area within which impacts on the natural ecosystem are not considered important and concern instead is focused on export to other ecological and human systems. As discussed in detail in Chapter 3, such concerns include (1) long-term environmental effects, including contamination of surface or groundwater resources, potential threats to human health, and secondary effects on valuable natural and agricultural ecosystems and (2) long-term commitment of land resources. Land spreading and reclamation may also be used as part of ecosystem management practices, such as the use of wastes for nutrient enrichment of park lands to enhance diversity and productivity.

There are a large number of terrestrial waste disposal options currently in use (Loehr et al., 1979), such as sludge applications to agricultural land (Council for Agricultural Science and Technology, 1976), sewage treatment by means of cypress domes, other wetlands and silvicultural areas (Ewel et al., 1982), and disposal in landfill and mine reclamation areas (Sopper and Kerr, 1981). Many of the constituents of waste enter surface-water and groundwater systems (Loehr et al., 1979) either through deliberate disposal (e.g., into some rivers and lakes) or secondarily (e.g., from leachate from agri cultural and forest systems). The purpose of this section is not to discuss specific disposal methods or recipient systems but to highlight those aspects of land disposal that need to be addressed.

The exports from disposal systems can be categorized as nutrients, organics, heavy metals, and pathogens. The pathways of concern for these include both those linked to other natural systems and those linked to humans. Disposal systems should be designed to prevent direct or indirect contamination of freshwater, groundwater, and estuarine systems, because such systems are characterized by extensive contact with humans, lack of containment, and high concern for system alterations (Ewel et al., 1982).

With respect to nutrients, direct enrichment of the disposal area may result in positive benefits, such as increased harvest of food or wood products. This is a key aspect of the application of wastes to managed terrestrial systems in that the resource value (i.e., nutrients) of human activities is recycled. Concern develops with the inadvertent nutrient enrichment of water systems from surface runoff and via percolation of leachate to groundwater. Nitrogen and phosphorus are of primary concern, particularly the movement into groundwater of nitrates and the movement of nitrogen, phosphorus, and oxygen-demanding organics into surface-water systems (Loehr et al., 1979). The latter is more problematical where climate, topography, and system management practices (e.g., agricultural cultivation) result in significant fluxes of runoff into streams and lakes. Movement of phosphorus into groundwater has been found to be far less significant (National Research Council, 1978).

For various terrestrial disposal systems, transport of toxic organics and heavy metals into both surface- and near-surface water systems remains an issue of concern, with respect to impacts on other natural systems and particularly with respect to pathways to humans. It is beyond the scope of this section to treat this topic in detail; rather, we will simply indicate that the waste stream must be characterized with respect to these toxicants and that their physicochemical characteristics and those of the environment are critical to determining the fate, transport, and effects of the toxicants.

Transport of pathogens to humans must be addressed for any terrestrial system. Potential pathways include direct consumption of food products grown in waste-amended systems, consumption of or other contact with animals that have had contact with the system, aeolian transport (e.g., windblown particulates), direct contamination of surface-water systems or groundwater (especially by viruses), and transfer by wildlife that are transient to the disposal area.

4.2 CRITERIA FOR EVALUATION OF ECOSYSTEM EFFECTS

4.2.1 General Aspects of Ecosystem Evaluation

This section introduces general ecosystem concepts that can be used to provide a basis for comparing impacts of waste disposal on different types of ecological systems. Considering the spatial aspects first, one can separate areas of waste input into two categories: (1) the direct, immediate locations where most of the waste materials are input and (2) adjacent and downstream areas into which some of the waste materials may be transported. In the case of terrestrial disposal, the first is the area of application, and to the extent that containment is provided, subsequent movement of pollutants to other locations or ecosystems is minimized. In the case of marine disposal, the first category is less well defined, but it is principally the benthic areas in the vicinity of the discharge sites. Significant dispersion into other parts of the oceans always occurs. Thus, in ocean disposal, there is a second much larger zone that is subject to low levels of waste input, although ultimate sinks may contain high levels of contaminants.

A fundamental question is whether, considering its value and uniqueness, the area under direct impact can be sacrificed. If it can be, impacts within the ecosystem are not critical, and attention focuses instead on ecological responses within systems to which constituents of the waste are exported. A closer look at ecological responses to direct waste inputs is required, however, for those cases in which the target system cannot or should not be eliminated, as is the case for waste additions to managed terrestrial ecosystems.

Ecological responses to environmental perturbations can be examined at both the population and process levels. The first includes changes in the distribution and abundance of particular species; the latter includes changes in biogeochemical cycles or energy-flow patterns. With waste disposal, the focus is largely on population- and community-level effects occurring within the area under primary impact rather than on processes, because of the limited spatial extent of the population perturbation relative to the spatial scale of most ecosystem processes.

When considering the effects of waste disposal at population and community levels, the first concern is potential elimination of a species, either through direct toxicological impacts or indirect effects, such as loss of habitat or reduction in some essential resource base. The issue of the spatial extent of the impacted area versus the spatial range of the species is critical. Further, for many species the area of concern includes the range for early life stages (e.g., nesting or spawning areas), which is smaller than the total geographical range for the mature stages. Similarly, the loss of a particular habitat that is both limited in its general occurrence and spatially of the same scale as the waste-impacted area presents a problem that must be addressed, as does the elimination of unique biotic communities.

Of less dramatic concern is the alteration of community structure. Community structure continually changes, even without significant anthropogenic perturbations; thus, alteration in the community structure per se may not represent a major problem. The situation becomes important, however, if such community alterations are major in spatial extent or unidirectional change and in the relationship and distribution of the constituent species. To evaluate these alterations, one must look at the interrelationships among the species. There may be some critical species whose presence is required for a significant part of the overall community to exist. Loss of the critical species from a location will result in concomitant indirect losses or population explosion of other species. Similarly, there are critical groups of species, i.e., a number of species may function redundantly within the system. The loss of all of them would result in the loss of their critical function in the overall ecosystem.

Another class of indirect effect that must be considered is the impact on some species that have particular aesthetic or economic value. For example, pollution-induced reduction in benthic habitat could result in depletion of fisheries, even though the fish were not directly affected by the pollution.

A final issue that needs to be addressed is the recoverability of the systems under impact. Rates of recovery of defaunated or defoliated areas depend on the rates at which the area becomes habitable again and on the rates of recolonization by the biota. These values depend to a large degree on the spatial scale of the impacted area. The recovery of a small area surrounded by undamaged systems is more rapid than is recovery of large impacted areas that have relatively fewer sources of colonizers.

In summary, the aspects of ecosystems that must be considered when evaluating waste impacts range from direct effects on individual species and effects resulting from interspecific interactions to effects on community structure and concomitant functional relationships. These aspects are overlaid by consideration of spatial and temporal scales. These are the types of information and understanding required, but it is quite another matter actually to attain them. For instance, while direct impacts on heavily affected areas may be readily discernible, effects of longer-term, more widely disseminated lower-level stresses are generally more difficult to detect. Such chronic stresses can affect a species in more subtle ways (e.g., behavioral changes versus immediate mortality) and often involve more indirect mechanisms. The response time is longer, and distinguishing stress-induced responses from normal environmental fluctuations and spatial heterogeneities is especially difficult. Temporal relationships, such as the one between biodegradation and accumulation rates, become important. Even more difficult to understand are synergisms, in which chronic pollutants have a greater impact in combination than separately. Each of these aspects contributes to the degree of uncertainty inherent in ecological evaluations. The significance of the uncertainty that remains after the ecosystems have been realistically characterized is a major issue in selecting among disposal options. This is particularly true since the uncertainty in predicting ecosystem-level effects of waste disposal includes not only a component related to the amount of information that has been collected about a system but also a component of intrinsic unpredictability. Further, the degree of uncertainty remaining even after a system is reasonably well studied varies from one ecosystem type to another.

4.2.2 Effects on Species

The ultimate impact to be avoided is the actual extinction of a species. Species extinctions have accelerated at an alarming pace in recent years, in most cases from the destruction of critical habitats. Such habitat destruction from sewage disposal is unlikely in open-ocean systems because the benthic species under impact have large ranges and, usually, good dispersal capabilities relative to the areas under impact. The degradation of coastal wetland habitats could be more serious, especially on the West Coast of the United States where wetlands are smaller and more isolated than East and Gulf Coast wetlands. The few remaining West Coast wetlands are refuges for several endangered insects, vertebrates, and plants (Onuf et al., 1978). Indiscriminate disposal of sewage and other materials in coastal wetlands should be avoided. Bastian (1981), however, suggests that both freshwater and marine wetlands could be effectively managed for waste disposal without deleterious effects on the ecosystem. The U.S. Environmental Protection Agency is currently developing guidelines for wetlands management practices that could be applied to the effective and environmentally compatible use and innovative treatment of wastewater in existing wetlands as well as to the establishment and management of artificial wetlands created for the primary purpose of wastewater treatment.

Landfill practices in terrestrial ecosystems can eliminate critical habitats of endangered species, and such elimination must be guarded against. At the same time, indirect effects on endangered species are also possible. These situations could include poisoning or the eutrophication of lakes, ponds, other surface waters, or groundwater in such a way that isolated critical habitats (including, e.g., caves) could be affected. In summary, species extinctions could result from sewage disposal if critical habitats are destroyed or seriously altered. This is unlikely for offshore systems; it is possible, but relatively easily guarded against in terrestrial systems; and it is a very real danger in West Coast wetlands.

It is important to emphasize that species extinctions are not adequate indicators of damage. Many important changes can occur without species extinctions. Furthermore, there is a tendency to be concerned about a small subset of species that are conspicuous or important to man to the exclusion of little-known or even yet-to-be described species. These species must also be protected, and the best protection is the preservation of their habitat. While local small-scale elimination of individuals may be an unavoidable consequence of some waste disposal procedures, the seriousness depends on the amount of habitat destroyed in relation to the total amount of that habitat in the area.

4.2.3 Community Stability

4.2.3.1 Resistance

One form of stability is the resistance of an assemblage to perturbation. Marine benthic communities can be expected to have different levels of resistance to waste inputs. For example, communities composed of active deposit feeders are less likely to be smothered than are associations of filter feeders, such as sponges. In most cases, the recently settled larvae and younger life stages are more susceptible to smothering or poisoning than are the adults, so there can be an illusion of persistence of an assemblage; but without successful recruitment, it will eventually disappear.

Waste disposal in wetland and terrestrial systems is complicated by the consequences of differential utilization of nutrients. Disposal may not affect the community structure in forests if all the species grow uniformly faster, but in some systems nutrient-limited populations can respond and displace others. Nutrient enrichment can have varying effects in different terrestrial ecosystems. The Park-Grass experiment and old-field studies revealed changes in biomass, reductions in species numbers, and loss of various life forms as a result of nutrient enrichment (Bakelaar and Odum, 1978; Milton, 1947; Silvertown, 1980). Nutrient-poor systems, however, will respond favorably to nutrient enrichment as evidenced by increased diversity and production (Willis, 1963; Specht, 1963). Eutrophication is common in many freshwater systems and may also occur in natural terrestrial systems following sludge deposition or runoff from waste disposal sites.

In summary, the resistance to change of natural communities in the sea and on land after waste disposal is variable and must be evaluated for each situation. In general, marine deposit feeding assemblages or assemblages with considerable natural bioturbation may be the most resistant; such communities are most common in shallow (<60 m) marine habitats that are already disturbed along many of the nation's coastlines.

4.2.3.2 Recoverability

Once a large area (measured in square kilometers rather than in square meters) of an assemblage has been altered, it is important to evaluate how long the consequences of the perturbation may persist and whether they are reversible. The recoverability of the original assemblages depends on the availability of propagules (seeds, spores, and gametes, for example), their settlement biology, their survivorship through reproductive age, and the difficulty involved in re-establishing interspecific relationships among the postperturbation species.

What little is known of deepwater soft-bottom marine systems suggests that, for the natural populations, dispersal is extremely limited and growth rates are very slow. Additionally, high species equitability juxtaposed with low densities also reduces the recoverability of the original assemblage, because different species may colonize and resist invasion of the previous populations. It is likely that recoverability of deep benthic communities is slower than that of any other marine community on Earth. The actual rates of succession are not known, but it is clear that impacts of deep-water dumping on benthic communities may last for hundreds of years.

Populations dominating shallow-water, soft-bottom benthic systems are characterized by much faster growth rates and more widely dispersing propagules. Disturbed habitats will be characterized by the presence of high densities of small numbers of opportunistic species. Species characteristic of undisturbed habitats generally have longer generation times and more unpredictable recruitment patterns. These include many important resource species (e.g., clams, crabs, shrimps, echinoderms, and some fish). Nonetheless, shallow (<60 m) habitats (Mearns, 1981; Rosenberg, 1972) appear to recover from large-scale perturbations much faster--tens of years compared with possibly hundreds of years--than do deep-sea populations (Grassle, 1977).

The populations in large wetlands have evolved along with occasional catastrophes, such as droughts, floods, or inundations of the sea following storms. The reduction of these habitats by man has threatened many species, and although these habitats enjoy certain resilience, they should be protected from further perturbations, especially the West Coast remnants.

Valid generalizations about the recoverability of terrestrial habitats are difficult to make. In many cases vegetation can be replanted, but this may be more difficult than it appears as other opportunistic species often resist invasion, and natural succession can be relatively slow. In practice, the consequence of most landfill operations is residential development, and this retards natural succession almost indefinitely.

In summary, relatively shallow (<60 m) offshore soft-bottom benthic communities may have the highest rate of succession and recovery from waste disposal. Many terrestrial systems can be replanted and fairly quickly recolonized by animals, but it is unlikely that landfill operations will follow this procedure.

4.2.4 Productivity Changes

Wastes such as sewage sludge can change biological productivity in dumping areas either by increasing plant nutrients and thereby stimulating photosynthesis or by direct input of organic matter. Sufficient quantities of waste can, of course, completely bury and smother the biota in any environment. In waters with restricted circulation (such as in shallow and semienclosed basins, soils with high water tables, estuaries, and small lakes), direct organic additions or increased primary production can result in anoxia.

In lesser amounts, organic enrichments can change the composition of communities. Generally, enrichment results in a reduction in diversity accompanied by an increase of biomass in soft-bottom marine systems (Pearson and Rosenburg, 1978), though in some cases (e.g., New England salt marshes) increased productivity increases diversity by enhancing the abundance of rare animals and increasing "evenness."

Additions of nutrients increase primary productivity in many systems. In experiments at the Marine Ecosystem Research Laboratory (MERL) in Rhode Island, high nutrient additions to shallow marine mesocosms increased phytoplankton productivity, decreased benthic species numbers, and had little effect on numbers of copepods in the zooplankton (Grassle and Grassle, in press; Nixon et al., in press). Nitrogen addition to salt marshes increases primary production, but more important is the increase in nitrogen content of the grasses, which subsequently increases production of herbivores and detritivores and enhances the attractiveness of the fertilized stands to more mobile herbivores, such as geese and voles. Alterations in the relative abundance of insect herbivores in comparison with those in unfertilized marshes also occurs (Vince et al., 1981).

In any nutrient-limited system, one could expect some species to be better able to make use of added nutrients than others, so that changes in plant community composition would occur, and the changes would depend on both the duration of the additions and the nutrient levels achieved.

Nutrients in sludge, along with the water in it, would drastically change the composition and productivity of desert and semidesert communities. Moderate additions to forests in humid regions would have less drastic effects on the plant communities themselves, although there could still be large changes in other components of the community, such as the abundance of insect herbivores.

4.2.5 Transport of Waste Constituents

It must be decided whether waste should remain concentrated locally in the environment into which it is placed or be dispersed throughout a greater volume (i.e., diluted), thus contaminating neighboring systems. Dilution can reduce the concentration of potential toxicants below the danger level but also make it impossible for complete biological decomposition to occur (Rubin et al., 1982). Dispersal of a substance makes it more difficult to recover, should that become desirable. Thus, the strategy of dilution markedly reduces the capability to mitigate undesirable consequences.

Fluid transport is responsible for most movement of materials between systems. Air transport can move materials throughout large areas as the Sea Air Exchange (SEAREX) studies demonstrate, but the amount and concentrations arriving at sites far from the source are small. Water is more effective and can move much larger amounts. Transport of contaminants to downstream systems must obviously be considered in any evaluation of aquatic sites.

In the case of marine disposal, once materials have reached the deep oceans, transport would ordinarily be expected between deep sites, often within the same general depth contours and therefore within one depth-controlled ecosystem. In shallow waters, movement can easily occur across system boundaries. Materials could move inshore into estuaries transported by bottom currents, or they could be moved offshore by turbidity currents. On land, surface waters can distribute pollutants over great linear distances and, eventually, into estuaries. Fluxes of waste components into groundwaters can also occur, and once the groundwater is contaminated, the potential exists for exceedingly long residence times of materials.

One must also consider the possibility of an organism moving a contaminant, especially a pathogen, from one system to another. If a parasite or virus introduced with a waste survives or even multiplies within a motile organism, it could significantly contaminate a system not being contaminated directly by the dumping. Populations of fish that migrate great distances along a coastline may result in contaminated organisms being harvested from uncontaminated areas.

4.2.6 Habitat Types

4.2.6.1 Uniqueness

Destruction of unique habitats and communities spatially separated from similar habitats by distances that create barriers to dispersal and recolonization should be avoided. In the marine environment, habitats of particular concern include coral reefs, deep-sea hydrothermal vent communities, seamounts, kelp forests and other sites of high recreational use, and communities on the walls of some submarine canyons or islands. Estuarine communities, although often separated from each other by considerable distances, are made up of some species with efficient dispersal mechanisms so that there is little or no evidence of genetic differentiation between individuals from widely separate estuaries (Gooch et al., 1972; Morgan et al., 1978). Other so-called cosmopolitan species, however, may be shown in the future to be genetically distinct in different estuarine systems within one climatic zone.

Deep-sea communities in general appear to comprise species with broad geographic ranges at certain depths.

However, the number of quantitative deep-sea samples examined, their distibution over the world oceans, and the relative paucity of information on the level of genetic differentiation among populations in different parts of a species range make generalization tenuous. There is also a lack of information on important life-history characteristics of biota, such as the dispersal capability of deep-sea species. In particular, little is known of the importance of biotic interactions in structuring deep-sea communities. Other life history features that characterize all but the most opportunistic deep-sea species (Grassle, 1977; Turner, 1973), such as long generation time, small population size, and low fecundity, suggest that these communities are likely to be affected more profoundly than are shallow-water communities when disturbed.

4.2.6.2 Recoverability

Disturbance in any ecosystem can be manifested by changes in the physical, chemical, and biological properties that govern the structure and function of communities within that ecosystem. Loss of habitat can be viewed as retrogression or reversed succession (Woodwell and Whittaker, 1968). Recovery, therefore, is dependent on the initiation of successional stages and the restoration of ecosystem properties (Holdgate, 1978).

For open-ocean plankton communities the scale and duration of habitat destruction is likely to be small compared with that of individual species. This, combined with the short generation time of individual species, suggests that these communities will receive less severe impacts (Capuzzo and Lancaster, in press). It is worth noting, however, that almost nothing is known about the level of genetic differentiation over the range of most planktonic species. There is some evidence that there are considerable genetic differences among populations of phytoplankton species collected in inshore and offshore coastal waters (Murphy et al., 1982), and these populations demonstrate considerable differences in sensitivity to environmental stress.

For marine benthic communities, community changes along either a spatial or temporal gradient of impact may reflect successional changes and a gradual restoration of stable community characteristics (Pearson and Rosenburg, 1978; Sanders et al., 1980). From their studies on organic enrichment in shallow benthic communities, Pearson and Rosenburg (1978) suggested that such successional stages include an initial zone of a few, small, rapidly breathing short-lived species with high genetic variability, followed by gradual changes in population with wider ecological and reproductive characteristics but lower genetic flexibility and contributing to increased community complexity.

In terrestrial environments the same principles apply. There are numerous examples of successional changes in terrestrial ecosystems as well (McIntosh, 1980). Successional properties and rates vary widely in different ecosystems, but once disposal is abated, natural succession should lead to recovery. Isolated rare habitats, however, may not be readily replaced by natural succession as propagules may not be readily available for colonization. These are exemplified by springs, especially in arid regions, isolated wetlands, rock outcrops such as limestone in regions of acid soils, and desert ponds.

Restoration of unique habitats in terrestrial and aquatic ecosystems may require artificial reclamation as propagules for recolonization are not readily available. Creation of artificial wetlands on the West Coast (Race and Christie, 1982), restoration of mangrove communities (Teas, 1977), and the restoration of plant communities in many terrestrial environments, ranging from high alpine to desert ecosystems (Cook, 1976), are but a few examples of successful restoration programs.

In practical terms, the uniqueness of a habitat may consist in its being the last fragment of the natural world surrounded by a greatly modified landscape. Urban ecologists have studied the importance of providing avenues for dispersal of city-dwelling fauna and flora by preserving the contiguous distribution of patches in the natural environment. Furthermore, social scientists have long appreciated the social importance of parks and natural sites in urban areas.

4.2.6.3 Nursery Grounds

It is apparent that areas that are intermittently used as nursery or spawning grounds should not be used as sites for waste disposal. Such areas include banks where hydrographic conditions promote concentration of developing embryos at certain times of the year (e.g., Georges Bank), benthic areas where spawning occurs, and estuaries that act as nursery areas for many resource species (e.g., penaeids, blue crabs, salmonids).

4.2.7 Monitoring Ecosystem Effects

Monitoring programs are an important component of ecosystem analysis in order to provide time-series data sets of long-term, unexpected changes in the ecosystem as a result of waste disposal. Ecosystem effects must be examined directly in the field by following the complete system, so that unsuspected interactive or synergistic effects can be detected. The study must be designed by researchers familiar with the systems at the site. Laboratory and field process-oriented studies would have to be conducted in parallel with monitoring of the system to detect causes for any changes observed in the system. The most difficult aspect of the research would be the selection of adequate control sites, which should be similar to the dump site in all aspects except for being subject to waste inputs. Sites similar in climatic and hydrographic conditions, and, with regard to sediment or soils, larval or propagule supply, predators, and productivity, for example, will usually mean that such sites are close together. But the water or air must not carry pollutants from the experimental site to the control site. An upstream positioning from the dumpsite seems logical but may be difficult to achieve with certainty. This is one reason that monitoring of the pollutant concentration must be done along with the biological measuring. Furthermore, one must be especially aware of the potential for storms or other rare events to spread wastes into the control sites.

4.2.8 Health Effects--Pathogens

In determining possible pathways that pathogenic organisms in wastewater sludge will follow from their point of disposal back to humans, one of the first considerations is the characterization of the distribution of such pathogens in sewage wastes. In general, sewage influents contain four broad groups of human pathogens: viruses, bacteria, protozoa, and helminths (Table 4.1). The occurrence of most of these microbial pathogens is dependent on the incidence of disease in the discharging population. The concentration of a given pathogen in wastewater effluents or sludge is not predictable, since it will increase or decrease with the number of sporadic cases of illness in the population. The difficulty of measuring individual species of microbial pathogens in sewage has been overcome by measuring, instead, a group of bacteria consistently found in the gastrointestinal tract of warm-blooded animals. This group, the coliform group, has traditionally been used to indicate the presence of fecal contamination and hence the potential presence of enteric pathogens. Much of the data in the literature on health effects related to thedisposal of domestic wastes into the environment has been described in terms of this bacterial indicator group. The distinction is of considerable importance, because the finding of coliforms in water indicates the probable occurrence rather than the absolute presence of pathogens.

TABLE 4.1 Pathogens Likely to be Associated with Sewage Sludge

Bacteria	Viruses	Helminths
Salmonella spp.	Poliovirus	Echinococcus granulosus
Shigella spp.	Coxsackie virus A and B	Hymenolepis nana
Vibrio spp.	Echovirus	Taenia saginata
Mycobacterium spp.	Adenovirus	Fasciola hepatica
Bacillus anthracis	Reovirus	Ascaris lumbricoides
Clostridium perfringens	Parvovirus	Enterobius vermicularis
Yersinia	Hepatitis	A Strongyloides sp.
Campylobacter spp.	Rotavirus	Trichuris trichiura
Pseudomonas spp.	Norwalk and related	Toxocara canis
Leptospira spp.	gastroenteritis viruses	Trichostrongylus
Listeria monocytogenes		
Escherichia coli	Protozoa	
Clostridium botulinum		
	Entamoeba histolytica	
	Acanthamoeba	
	Giardia	

SOURCE: Alderslade (1981).

During the formation of sludge in the sewage treatment process, up to 99 percent of the coliforms and pathogens can be removed from wastewater effluents. Although the sizes of the microorganism populations are significantly reduced in the sludge-forming process, many of the pathogens remain viable and present the potential of transport back to humans. Additional treatment, such as thermophilic digestion, is required for further disinfection of pathogens. The means of transport to humans is dependent on the form of disposal of the waste material. For instance, pathogens in sludge disposed of in water are highly mobile. They can be rapidly dispersed from their point of origin and transported to areas used for recreation or food resource harvesting. Pathogens in sludge disposed of on land, on the other hand, are not highly mobile and therefore are less likely to be routed back to humans. Countering this trend, however, is the proximity of the waste to human populations, which may be near in terrestrial disposal but distant in deep-ocean disposal.

4.2.8.1 Pathogen Problems Associated with Marine Disposal

Filter-feeding shellfish (e.g., clams, oysters, mussels) and contaminated recreational waters are two major sources from which pathogens from contaminated waste can reach man. Shellfish filter large volumes of water to obtain food and oxygen and, in the process, retain and accumulate particulates such as bacteria and viruses. While extensive epidemiological evidence relating sludge-contaminated shellfish to disease is lacking, there are ample data linking shellfish-associated disease outbreaks to waste-water microorganisms (F. L. Bryan, 1980; Levin, 1978; Metcalf and Stiles, 1968). The most important sewage assocated diseases appear to be infectious hepatitis and viral gastroenteritis.

Data are also available linking wastewater-contaminated bathing water to swimming-associated illness (Cabelli, 1980). Epidemiological studies have shown that there is a linear relationship between water quality and gastrointestinal illness (Atwood et al., 1979; Cabelli, 1978;

Cabelli et al., 1982), and disease outbreaks caused by wastewaterborne Norwalk agent virus have been documented (Baron et al., 1982).

Most of the evidence associating disease with wastewater-contaminated environments has been obtained from studies conducted in nearshore marine coastal waters or freshwater areas. The translation of these data to long-distance ocean outfalls or waste dump sites depends on the availability of information concerning two critical factors: (1) the ability of pathogens to persist and to replicate in marine environments and (2) an understanding of potential transport routes back to nearshore areas.

The evidence is clear that human pathogens, including protozoa, bacteria, filamentous fungi, yeasts, viruses, and parasites are present in sewage sludge and, very often, in dredge spoils as well (see Table 4.1, Alderslade, 1981; Sawyer et al., 1982). These organisms survive and persist, and as methods for isolation and outgrowth are improved, they are recovered with increasing frequency from water, sediment, and macrobiota samples from the Philadelphia and New York dumpsites (Table 4.2). Epidemiological evidence, however, has only linked a few of these pathogens to disease outbreaks in the United States.

A vexing issue is that of transport of these pathogens from the dumpsite regions and regions under impact from the dump to nearshore areas and, ultimately, to humans. Unfortunately, the hydrography of the offshore dumping areas has not been sufficiently described to make possible careful estimates of potential risks. This is especially so since measures of ocean currents (i.e., current speed and direction) and related physicochemical parameters of the waters do not, alone, define risks completely. For example, most microorganisms in aquatic systems tend to adhere to surfaces and to colonize particulates and organic aggregates. The particulates and aggregates provide protection against adverse environmental conditions, thereby enhancing survival and long-term persistence in the environment. Furthermore, the transport of pathogens attached to particulates and aggregates is different from that of free cells in the water column, and this may influence pathogen transmission to shellfish stocks. Sinking rates and transport velocities need to be known to determine whether transport of pathogens will occur from the dumpsites to areas of human activity and hence into contact with humans.

TABLE 4.2 Pathogens Documented to Occur in the Anacostia River, District of Columbia, and in the New York Bight

	Concentrations per 100 mL of water	
Isolates	Anacostia Riv	New York Bight
Total viable count	2 x 10^6-2 x 107	700-76,000
Total coliforms	3,000-28,000	4-800
Total fecal coliforms	100-4,900	0-90
Total anaerobic count	0-10,000	0-40,000
Aeromonas spp.	1,000-50,000	0-10
Bacteroides spp.	+	+
Clostridium spp.	+	+
Enterobacter spp.	+	+
Escherichia coli	+	+
Klebsiella spp.	+	+
Salmonella spp.	+	+
Vibrio cholerae	+	0
Vibrio parahaemolyticus	+	+
Group F Vibrio	0	+

NOTE: +, detectable levels present; 0, not detected.

Bacterial pathogens and indicator bacteria that are not attached to particulate or aggregate matter usually find the marine environment inhospitable. Predatory organisms, algal toxins, and natural antibiotics all contribute to the rapid die-off of these bacteria (Mitchell and Chamberlain, 1979). Viruses, on the other hand, are much more resistant to these environmental factors. Their persistence in seawater has been convincingly documented (LaBelle et al., 1980; Lo et al., 1976; Toranzo and Hetrick, 1982). Nonhuman hosts (i.e., shellfish and finfish) can serve as reservoirs of non-replicating viruses (Goyal et al., 1979; Gerba and Goyal, 1978; Gerba et al., 1980). It is not clear whether human pathogenic viruses replicate in nonhuman hosts.

Autochthonous pathogens (i.e., Vibrio parahaemolyticus, V. vulnificus, V. fluvialis, V. damsella, V. hollisae, and related organisms) grow, reproduce, and metabolize substrates in estuarine and coastal waters. They are not directly related to sewage effluent per se, and they occur naturally in estuaries and coastal waters but not in offshore, deep-water regions (Colwell et al., 1980). Two potential problems must be considered. First, under eutrophic or nutrient-enriched conditions could V. parahaemolyticus and 01 V. cholera flourish and achieve population sizes posing a serious threat to human health, notably via ingestion of contaminated shellfish? Second, will areas under impact, such as the Philadelphia dumpsite, the New York Bight Apex, and similar dumpsites, receive sufficient nutrients to support growth of these pathogenic marine vibrios and enteric pathogens not isolated heretofore from offshore waters? Preliminary data suggest that this may be the case.

Recent work (Xu et al., 1983) suggests that some pathogens including enterotoxigenic Escherichia coli and other pathogenic species, including 01 V. cholerae and Salmonella typhosa, enter a nonrecoverable stage (i.e., they cannot be cultured with the usual methods) under adverse conditions, yet they remain viable. Such conditions include low temperature (10°C), low nutrient concentration, and high salinity. This is clearly the case for V. cholerae (Singleton et al., 1982a, 1982b; Xu et al., 1983).

The survival and persistence of pathogens in deep-water dumpsites is unknown, although the presence of pathogenic Staphylococcus aureus discharged in pharmaceutical wastes to surface waters of the Puerto Rico Trench dumpsite has been recorded (Gunn and Colwell, 1982; Singleton et al., 1982b).

Because of costs and lower frequency of occurrence in sewage, pathogens themselves are, in general, not the subject of bacteriological surveys. Instead, indicator organisms are enumerated, such as E. coli and related members of the coliform group, including Klebsiella and Enterobacter. For nearshore waters and estuaries, the coliform group has provided a workable, if not totally reliable, index of sanitary quality. The extensive work of Cabelli and colleagues (Cabelli, 1980) has been focused on evaluating various indices for establishing human health criteria in recreational waters. A strong correlation exists between the incidence of gastrointestinal symptoms and enterococcus densities, but poor correlations exist between disease incidence and total coliform and Clostridium perfringens densities.

A major, well-recognized, documented shortcoming of the coliform (bacterial) indicator is that it does not provide an estimate of the presence of human pathogenic viruses. There have been documented cases of gastroenteritis induced by Norwalk and Norwalk-like viruses as a result of ingestion of contaminated shellfish harvested from waters judged to be safe for humans according to coliform counts. The currently accepted standard--70 total coliform per 100 mL of water, with not more than 10 percent of the samples exceeding 230 coliform per 100 mL--was established many years ago (circa 1939). This standard was not based on epidemiological evidence, and therefore it has little value with respect to predicting risks to the health of consumers of shellfish that have been exposed to sludge or sewage effluents. Fecal coliform indices are also used to assess shellfish contamination, but the relationship of these indicators to nonbacterial pathogens, is equivocal at best and nonexistent at worst. The relationship in offshore waters is totally unclear, and data are only now being gathered at the Philadelphia and New York dumpsites. If the relative merits of ocean versus land disposal of sludge is to be properly evaluated with respect to health risks, development of improved methods for identifying indicator and pathogen densities in shellfish, sediments, and water will be necessary.

4.2.8.2 Pathogen Problems Associated with Land Disposal

The application of wastewater sludges to land creates a potential hazard to human health. The transport of microbial pathogens through the soil can contaminate groundwater supplies under certain conditions. Surfacewater runoff can also carry pathogens to streams and lakes that are used as sources of potable water or as recreational resources. The degree of risk to humans is dependent on the type of land disposal site, the method of disposal, the proximity to human-related activities, the type of soil, and the depth to the groundwater table. Each of these factors or a combination of them is critical to the assessment of health risks associated with land disposal of sludge.

The normal habitat of most pathogens occurring in wastewater effuents and sludge is the human body or the body of some other warm-blooded animal. However, frogs, lizards, and other cold-blooded animals can also be carriers of potential human pathogens. Most bacteria and all viruses appear to be able to survive in terrestrial environments only for short periods. However, bacteria and protozoa that can form spores or cysts are able to survive for long periods under extreme conditions in the resistant form. These forms probably pose the greatest risk to human health. The risks associated with spore-forming bacteria and viruses are not known with any degree of certainty. There is a great need to gather data on their survival characteristics so that the information obtained can be applied to the assessment of potential hazards to health resulting from land application of sludge.

Scanty information available on the movement of bacterial and viral pathogens through the soil indicates that lateral movement does not usually cover great distances. Bouwer et al. (1974) found that fecal coliforms could not be found 92 m from their point of application to the soil. Similar results were observed in Santee, California, where coliforms were seldom found in coarse sand 61 m from the point at which they were applied to the soil (Wesner and Baler, 1970). The mobility of viruses through soil appears to be similar to that observed with bacteria (Wellings et al., 1975). These mobility studies would seem to indicate that the mobility of bacteria and viruses is such that the risk of transport to groundwater supplies would not be high at a properly chosen site.

Waterborne disease in humans has not been associated with pathogen contaminants from land sites used for the disposal of sludge, although the data are only now being gathered that may address this question directly

(H. Shuval, Hebrew University, Jerusalem, Israel, personal communication). The relative immobility and rapid die-off of pathogens in soil, the critical selection of appropriate sites, and careful site maintenance are probably the prime factors contributing to the lack of statistical data concerning disease associated with pathogens contained in sewage sludges applied to land.

An additional problem is that current methods for identifying fecal contamination cannot differentiate between animal and human wastes. Since it is unknown whether the hazards to health associated with exposure to animal and human fecal contamination are identical, it is impossible to determine with any degree of certainty whether a potential health risk is the result of sludge applied to the soil or of the deposition of animal wastes. Methodology should be developed to differentiate animal from human fecal contamination.

In summary, assessment of potential health effects related to dispersing of sludge on land or in the sea falls into four major areas: (1) survival of pathogens and indicator bacteria in marine and terrestrial environments, (2) the inadequacy of currently used bacterial indicators of fecal contamination for indexing potential health effects related to wastewater effluents and sludge, (3) the relationship of disease associated with shellfish to water quality, and (4) nutrient stimulation of growth of pathogens in open-ocean areas.

4.2.9 Toxicological Effects--Toxicants

The transfer of toxic chemicals through various components of marine and terrestrial food chains can induce specific changes at each trophic level or result in bioaccumulation and transfer to humans. Of specific concern is the uptake and transfer of metals, halogenated hydrocarbons, and other organic contaminants. Defining the potential risk of food chain contamination requires an understanding of the flux and bioavailability of the contaminant within each environment and elucidation of the potential routes to humans. Contaminants that demonstrate mutagenic, carcinogenic, or teratogenic potential must be carefully evaluated before a disposal option is selected.

Land application of wastes may result in the transfer of contaminants to soil surfaces, to plants grown in the sludge-amended soil, to groundwater, and to surface-water runoff that can transport contaminants to other terrestrial or aquatic sites. In soil amendment applications, contaminant uptake may result directly from ingestion of sludge-contaminated soil or forage or indirectly from consuming contaminated plants (Chaney, 1982). Contaminant bioavailability can vary among soil types because of the physicochemical nature of metal binding or sorption-desorption processes and differential rates of microbial degradation. Soil ingestion has been shown to transfer organic contaminants to grazing animals in fields covered with sewage sludge (Bergh and Peoples, 1977; Collett and Harrison, 1968; Hansen et al., 1981; Harrison et al., 1969; Harrison et al., 1970), and such contaminants may potentially accumulate in fatty tissues and milk. Some metals (such as lead or mercury) could also be transferred in this way.

Other metals (such as cadmium, manganese, and zinc) are easily translocated to plant tissue, but phytotoxicity can limit the amount of metal available to higher trophic levels. Chaney (1980) proposed a "soil-plant-barrier" theory to predict the uptake of metals in plants. The soil plant barrier protects the terrestrial food chain from metal toxicity through one of the following processes: (1) insolubility of metal in soil prevents uptake, (2) immobility of metal in fibrous roots prevents translocation to edible plant tissue, or (3) phytotoxicity of the metal occurs at concentrations in edible plant tissues below that injurious to animals. This concept holds true for most metals except cadmium, molybdenum, selenium, and cobalt, which can accumulate to harmful levels in plant tissue.

Storage of organic contaminants in plant tissue has also been documented, and Chaney (1982) proposed that compounds can enter plant tissue either through uptake from soil solution, with translocation from plants to shoots, or from adsorption by roots or shoots from volatilization of organic contaminants.

In the oceans, trace metals are primarily adsorbed onto particulate material, and their transport follows the sedimentary transport within an environment. Remobilization of metals from the sediments may occur through changes in the physicochemical nature of the sediments, bioturbation, and microbial processes. Bioaccumulation of trace metals by marine organisms is dependent on metal speciation and bioavailability. Although diffusion of dissolved metal through body surfaces is the easiest route of uptake by marine organisms, uptake through ingestion of food organisms and particulate matter are the most important ecological routes of transfer (G. W. Bryan, 1980). Bioaccumulation of metals in the marine environment may be further complicated by the physiological state of the animal, environmental conditions, and the ability of an organism to regulate metal uptake (George, 1982; Phillips, 1977). The role of bacteria in bioaccumulation has been demonstrated (Nelson et al., 1975), and effects on the food chain have been reviewed. Both metal and organic toxicants are taken up by bacteria, and the process may provide a rather significant mechanism for food-chain biomagnification and transfer.

Marine animals differ in their capacity to store, remove, or detoxify metal contaminants. Storage may involve deposition into tissues, skeletal material, and concretions or within intracellular matrices (George, 1982). Removal may take place via excretion or the production of particulate products, such as feces, eggs, and molts. G. W. Bryan (1979, 1980) classified organisms as to their relative metal regulatory ability: (1) where the contaminant is excreted at a rate proportional to body burden, (2) where the contaminant is stored rather than excreted, and (3) where an organism excretes most of the excessive input. Crustaceans and fish are the best regulators, and essential metals such as zinc and copper are better regulated than nonessential metals such as mercury and cadmium.

Certain metals may be detoxified by binding to metallothionein proteins in fish (Jenkins et al., 1982) and similar heavy-metal binding proteins (HMBP) in crustaceans and molluscs (Jennings et al., 1979; Noel-Lambot, 1976; Noel-Lambot et al., 1978; Roesijadi, 1981; Roesijadi et al., 1982). Both proteins are thought to exert their protective effect by sequestering free metal ions and partitioning them away from potential sites of toxic action (Jenkins et al., 1982). If the binding capacity of metallothioneins or heavy-metal binding proteins is exceeded, toxic effects of metal contaminants may be induced (Lee et al., 1980). George (1982) suggested that detoxification can also occur through compartmentation within extracelluar or particulate intracellular structures or blood cells.

The transport of xenobiotic compounds, such as polyaromatic hydrocarbons (PAHs) and polychlorinated biphenyl compounds (PCBs) is also linked to particulate transport, and thus such compounds tend to accumulate in sediments (Boehm et al., in press; Bopp et al., 1982). Bioaccumulation of lipophilic organic contaminants (PAHs, PCBs, or other chlorinated hydrocarbons) by marine organisms is dependent on both chemical factors, such as solubility, adsorption-desorption kinetics, and the octanol-water partition coefficients of specific components, and biological factors, such as the transfer of such compounds through food chains and the amount of body lipids in exposed animals (Neff, 1979). The availability of xenobiotics to marine organisms and their effects on those organisms are dependent on chemical and microbial processes within the sediments, including sorption/ desorption reactions and the production of potentially more toxic metabolites from degradation (Gibson, 1981). Uptake of PAHs and PCBs by bacteria and phytoplankton may further enhance the transfer to higher trophic levels (Iseki et al., 1981).

In both vertebrate and invertebrate systems, including mammals, fishes, crustaceans, and polychaetes, biotransformation of PAHs and PCBs may occur through cytochrome P450-mediated mixed function oxygenase (MFO) reactions. Such transformations result both in the production and excretion of more soluble, potentially more toxic metabolites (Stegeman, 1981). Some metabolites of xenobiotics have been shown to be carcinogenic, mutagenic, and teratogenic, and their potential long-term effects are of significant concern. Bivalve molluscs have only a limited capacity for biotransformation, and thus xenobiotic compounds will accumulate in their tissues and be directly available for human consumption.

Recent studies of the incidence of tumors and other histopathological conditions in demersal fish from the Duwamish River and Hudson River estuaries as well as the Southern California and New York Bights have suggested a possible link of chronic xenobiotic inputs and the increased incidence of such conditions (McCain et al., 1978; Perkins et al., 1982; Sindermann, 1980; Sinderman et al., 1980; Smith et al., 1979; Stegeman, 1981). Toxic effects of xenobiotics include impairment of reproduction, growth, and development (Califano, 1981), and potential effects on populations cannot be ignored.

4.2.10 Summary

It is understandable that those who must regulate should seek from natural scientists a single index that will indicate whether a population, community, or ecosystem has suffered irremediable damage. However, no such index exists, nor can one be devised. There is, on the other hand, no substitute for a functional understanding of how particular communities respond to change; and this understanding, though not perfect, is already available for some communities and should be used. For others, developing such understanding will require further research.

Table 4.3 summarizes the biological concerns for different types of environments. The numbers should not be added in either the columns or rows; they represent individual estimates that must be considered together, as a framework useful in biological aspects of decision making.

4.3 INFORMATION NEEDS

The objectives of waste disposal research are to (1) improve management schemes, (2) refine the types of remedial actions via better understanding of the natural recovery processes, (3) define pathways and fates of pathogens and toxic material, and, most importantly, (4) better understand the components and workings of the ecosystems to further improve and define the research/management goals and practices.

Specific information gaps that currently exist are as follows:

- An understanding of functional responses of ecosystems under impact is essential if we are to identify the possible routes by which toxicants and pathogens can reach humans and if we are to be better able to predict the outcome of chronic inputs to affected communities. This understanding, however, must be predicated on improved knowledge of community structure, such as what species are present in the ecosystem, their life histories, and the demographics of the important species, as well as the potential recovery rates of impacted ecosystems. Such information can be obtained both through the use of in situ long-term studies and experimental microcosms as parts of natural ecosystems.
- The contention in this chapter that most deep-sea species have wide geographic ranges is based on little evidence. This is especially obvious from an examination of species lists from some recent investigations of

deep-sea sites. They contain numbers of unidentified and unnamed species and undifferentiated groups of species making it impossible to compare results from different laboratories and different sites.

- As many contaminants are associated with particulate matter in the marine environment, improved understanding of hydrographic conditions, particle behavior, resuspension, and biological modification of particle fluxes is essential. This would include an understanding of the partitioning of contaminants between water and sediment particles, especially as this is affected by the activities of organisms in ventilating, irrigating, and passing material through their guts.
- Increased efforts are needed to estimate and quantify the relative contributions of pollutants from different sources in areas receiving multiple inputs, especially in shallow estuaries and coastal areas near large metropolitan centers.
- The survival and persistence of human pathogens (viruses, bacteria, fungi, protozoa, and helminths) in seawater are uncertainties that need to be addressed, in addition to identifying the parameters such as nutrient loading, salinity, temperature, and pressure that influence survival and persistence. The relationship between pathogens, notably viruses, and particulates must be established to determine survival and transport of pathogens to areas of human activity. This would include development of decay models for particle-associated bacteria and viruses in aquatic environments.
- Criteria on health effects for shellfish-associated disease must be defined. These should include the development of more specific bacterial indicator methods to monitor potential health hazards associated with marine sludge disposal sites.
- A similar data base on the survival and mobility of pathogens in soils must be established. This should include the development of methodology to differentiate human and animal contamination in an impacted ecosystem. Attention should also be focused on the development of predictive models to evaluate pathogen contamination of groundwater supplies and land crops.
- A better understanding of surface and subsurface drainage patterns from land disposal sites is needed to predict the export of toxicants and pathogens from disposal activities and their potential accumulation in groundwater and surface-water resources.

Table 4.3 Comparison of Ecosystem Response to Waste Inputs

Type of Enviornment	Species Extinction	Habitat Loss	Elevated Nutrients	Recoverability	Containment	Remedial Action	Uncertainity	Visibility	Pathogen Routes to Society	Toxicant Routes to Society
Land										
Disturbed lands [a]	1	0	1	0	1	0	1	5	5	5
Remnants [b]	0	5	1	1	1	1	1	5	5	5
Temperate forest	1	1	1	2	1	1	1	3	2	2
Temperature grassland	1	1	1	1	1	1	1	3	3	3
Pasture	0	0	0	1	2	1	1	5	5	5
Agricultural land	0	1	0	2	2	1	1	5	5	5
Arid land	3	2	1	3	1	3	2	2	1	1
Arctic land	0	1	1	5	1	5	4	1	1	1
Freshwater										
Lake	1	5	5	3	5	4	2	5	4	4
Stream	5	5	3	2	5	4	3	3	5	5
Wetland	5	5	5	3	5	5	4	2	3	3
Groundwater	3	1	5	5	5	5	5	0	5	5

Wetlands (U.S. East Coast)	1	4	3	3	5	5	3	5	5	5
Wetlands (U.S. West Coast)	5	5	3	3	5	5	2	5	5	5
Estuaries	5	5	3	3	5	5	2	5	5	5
Coastal areas	1	3	1	1	5	5	3	1	3	4
Open ocean	1	1	0	5	5	5	5	0	1	1

NOTES: Species extinction: 5 = greatest concern.
Habitat Loss--loss of a significant portion of a habitat type: 5 = greatest concern.
Elevated Nutrients: 5 = highest probability of change to ecosystem. Recoverability--ability of system to repair itself after input ceases: 5 = slowest recover, decades to centuries; 1 = rapid recovery, years.
Containment--ability of unmodified system to restrict spread of inputs: 5 = greatest difficulty.
Remedial action--ease with which we can repair damage to ecosystem: 5 = greatest difficulty.
Visibility: 5 = most visible.
Pathogen Routes to Society: 5 = highest probability of reaching society.
Toxicant Routes to Society: 5 = highest probability of reaching society.

[a] Disturbed Lands--land highly modified by human activities.

[b] Remnants--isolated natural spots within developed or otherwise highly modified areas.

- The effects of incomplete degradation or metabolic alteration of toxic chemicals on human health and ecosystems continue to be a subject of primary concern owing to the mutagenic, carcinogenic, and teratogenic potential of toxic parent compounds and their metabolites.

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5

Case Study A: Report of the Panel on Sewage Sludge

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5.1 INTRODUCTION

5.1.1 Why Must Wastewater Sludges Be Studied?

For every 1000 cubic meters of wastewater received in a wastewater treatment plant there will be of the order of 60 to 320 kilograms dry weight of solids that must be disposed of. The amount depends on factors such as wastewater strength, use of anaerobic stabilization, and the presence or absence of combined sewers. Since this residual is usally produced as a slurry of from 2.5 to 10 percent solids it is termed sludge. The sludge can be further processed by dewatering and drying prior to final disposal. However, because of its possible content of toxic contaminants, sludge whether wet or dry has had constraints placed on its use or disposal. These constraints apply to ultimate land, air, or water placement and have made the problem of sludge solids disposal a major one for almost all municipalities and industries. The magnitude of the problem can be illustrated by the treatment facilities of the City of Los Angeles, where a 113,550 m^3 per day secondary treatment plant must dispose of 18,000 dry kilograms per day and a 1,589,700 m^3 per day facility must dispose of 180,000 dry kilograms per day. For cities such as Chicago, Illinois, or Atlanta, Georgia, only land or air disposal seems feasible. For coastal communities, sea disposal would appear to be a logical alternative. However, as pointed out in the report The Role of the Ocean in a Waste Management Strategy by the National Advisory Committee on the Oceans and Atmosphere (1981), current legislation or its interpretation, or both does not allow a full evaluation of alternatives.

Decisions regarding the disposal of industrial and domestic sludges are now being made at the political, regulatory, project design, and operational levels. In many if not most cases, these decisions involve the commitment of large amounts of funds for capital expenditures, operations, and maintenance. Once such commitments are made, it becomes difficult, if not impossible, to make changes that take new environmental, financial, or scientific-technical developments into account. Given this fact, a basic question arises: Is the scientific-technical information now available about sludge disposal sufficient for making decisions within some reasonable risk factor?

Similarly, the data a decision maker must deal with in regard to sludge disposal must include public perceptions of what is safe and suitable. Wastewater sludge from the city of Los Angeles, for example, could have been pumped to solar drying beds in the desert and then utilized for agricultural purposes or disposed of in a hazardous-waste landfill near the drying beds. Although this idea had considerable merit (see Chapter 4, Table 4.3), it was never a realistic possibility because of intense opposition from desert residents and environmental organizations. If this plan had included some type of benefit for desert residents, such as a park or a camping area or swimming pools, their opposition might have been defused, thereby allowing the environmental soundness of the plan to be objectively reviewed. Court-mandated time schedules prevented this, however, and an incineration scheme in an air-quality nonattainment *area* was adopted. At present the commitment of construction funds to the incineration alternative prevents further consideration of any other option. A decision maker considering waste disposal alternatives must take into account environmental conditions, current scientific-technical knowledge, public perception, mandated time schedules, and capital investment already made. The scientist cannot be truly effective until he realizes that scientific facts are not the only things that must be considered by the decision maker in making choices (Region IX, U.S. Environmental Protection Agency, 1980).

5.1.2 Availability of Required Information

Given the need for immediate decisions about sludge disposal, the question is whether sufficient information exists to choose with reasonable assurance among air, land, and sea alternatives. The panel members believe that enough information now exists to make responsible decisions and that a procedure called "multimedia analysis" constitutes a powerful tool for evaluating environmental priorities for each specific case (Region IX, U.S. Environmental Protection Agency, 1980; New York City Department of Environmental Protection, 1983; Southern California Coastal Water Research Project, 1983). Since it is difficult to obtain all the environmental facts about the disposal of sludge in any particular medium, it will be necessary to make decisions in the absence of some facts that may have bearing on the problem. It is therefore important that some portion of the available research funding from the Environmental Protection Agency (EPA) and National Oceanic and Atmospheric Administration be used to identify the most important factual materials needed for a decision-making matrix. The panel directed its attention to the types of factual information that are needed to make valid multimedia assessments. While recognizing the legislative, administrative, and legal decisions that appear to mandate a single nationwide standard for sewage sludge disposal (Pregerson, 1977; U.S. Environmental Protection Agency, 1975, 1977), the panel recommends a total environmental analysis for each specific problem. This analysis should be aimed at identifying the disposal alternative with the least net negative environmental impact.

Use of a decision-making matrix is recommended, and one procedure is illustrated later in this chapter. A properly structured matrix will allow the goal of the least net negative environmental impact to be more closely approximated given the fact that knowledge in some areas of the concern is sparse or tenuous.

5.1.3 Approach to the Problem

Sewage sludge is the residue that results from treatment of domestic and industrial wastewater. During the treatment process, a percentage of the impurities in the wastewater are concentrated by gravity settling (with or without flocculant use), dissolved air flotation, or various mechanical methods and converted into a liquid slurry called sludge. Being an aggregate of settleable fats, carbohydrates, and proteins with wastewater contaminants a sludge may contain all the constituent elements found in domestic and industrial wastewaters. Because of the differences in the characteristics of communities, the composition of sludge can vary considerably from city to city. For example City of Los Angeles wastewaters receive 12 percent industrial-commercial waste, while Los Angeles County Sanitation Districts wastewaters have a 30-35 percent industrial flow. Other factors include the degree of wastewater pretreatment required of industry, the diversity of the processes utilized in treating wastewater, and the presence of combined or separate sewers. It is therefore essential that the physical, chemical, and biological characteristics of each specific sludge be taken into consideration in deciding on a disposal method. Currently, a variety of disposal/reuse options with differing potential impacts on land, air, and water resources are practiced. However, no option will ever have zero risk to society or the environment. Therefore, a comparative cross-media impact assessment incorporating uniform risk-analysis procedures applicable to all disposal media must be performed, and the results of this assessment must be used as a basis for selection. It is particularly important that the assessment be realistic, balancing risks, benefits, and costs.

Some of the materials in waste sewage sludge are biodegradeable, others are not. The end products of aerobic biodegradation are chiefly carbon dioxide and water. A major end product of anaerobic biodegradation is methane. These processes are time dependent and may produce a number of intermediate products that could potentially be more harmful than the parent material. Additionally, some organic compounds are persistent and require long periods of time (perhaps decades) to decompose. Inorganic elements, for the most part, are not biodegradeable. The choice of an option for sludge disposal/utilization will depend on physical, chemical, and biological character istics of the sludge, the availability of suitable disposal sites, and local considerations. Local considerations include environmental factors (hydrogeology, climate, soils, water quality and quantity, and topography, for example), sociopolitical factors (governmental structure, socioeconomic factors, and public attitudes, for example), and technological factors (scale, transport, and available energy sources, for example). These factors, among others, determine not only the suitability of land, air, and ocean receptors but also the specific type of land, air, or ocean option best suited for disposal/reuse. Specific land-based options include landfill, dedicated land disposal, agricultural reuse, land reclamation, and land maintenance (landscaping, for example). Ocean disposal options include transporting the sludge through pipes into deep or shallow water or dumping it from barges. Obviously, sewage sludges cannot be directly disposed of in air, but the products of thermal processing (incineration, pyrolysis) are disposed in the atmosphere, as well as on land (ash and fallout). Other disposal techniques include chemical or thermal fixation plus disposal in the ocean or on land.

The remainder of this chapter is divided into four sections. Section 5.2 describes the source in terms of a multimedia disposal evaluation of the quantity and quality of a sludge. Section 5.3 outlines available sludge disposal/reuse options. Section 5.4 presents a discussion of site-specific and local considerations that must be recognized in a multimedia assessment. Section 5.5 discusses elements of the evaluation process for a specific sludge and a specific site. Finally, a summary of salient points is given.

5.2 THE MATERIAL

The starting point in a multimedia assessment is a knowledge of sewage sludge quantity and quality, both currently and in the future. This section describes sludge characteristics and the various processes that influence those characteristics.

5.2.1 Source

Wastewater consists of a mixture of wastes from homes, commercial establishments, and local industries. Although mostly water (99.9+ percent), wastewater may also contain a wide variety of materials that could be harmful in concentration and that often must be substantially removed prior to discharge. Such removal is accomplished in wastewater treatment plants, where the pollutants in the wastewater are either degraded by biological processes or turned into residual solid material (sludge) that contains many of the contaminants in concentrated form.

The composition and volume of wastewater, and hence sludge characteristics, depend on the area served by the plant. Wastewater composition and volume tend to be specific for each plant, although it is possible to generalize when a plant primarily serves residential communities. In industrial communities, pretreatment of wastewaters by industries will obviously affect both the composition and volume of the sludge and effluents produced by the treatment plant.

5.2.2 Wastewater Treatment Processes

The objective of wastewater treatment is to protect public health and the beneficial uses (including aesthetics) of the land, water, or air resources that receive either the effluent or the sludge (State of California Resources Control Board, 1971, 1974, 1978). The effluent and the residual solids removed during treatment will have some environmental impact on whichever media to which they are disposed. Both effluent and sludge may contain materials of possible benefit (such as phosphorous, nitrogen, carbon, silica, and vitamin B12) and materials of possible danger (such as cadmium, selenium, DDT's, and PCB's). The design of a treatment plant must therefore take into account these factors as well as the characteristics of the specific area in terms of air, water, and land resources. It would appear unwise, for example, to incinerate sludge in an air-quality nonattainment area.

The treatment plant designer can choose from a number of processes. Basic to almost every process is gravity or flotation separation of particles. That is, over a period of 1-3 h, some materials will settle on the bottom of treatment tanks while others will float to the surface (grease, oils, plastic materials, cellulosics). The settled solids and floating scum are often mixed together in a slurry called primary sludge. The remaining fluid is termed primary effluent. The Federal Water Pollution Control (Clean Water) Act Amendments of 1972 and 1977 (Public Law 92-500 and 95-217) requires full secondary treatment of the wastewaters prior to discharge. In some cases, such as where a section 301(h) waiver of full secondary treatment is approved by EPA, primary effluent can be directly discharged into receiving waters.

The most usual secondary treatment involves the seeding of the primary effluent with an active culture of aerobic biological organisms capable of removing the dissolved and colloidal organic matter (fats, carbohydrates, proteins) by utilizing it as food or by coagulation and/or physically sweeping it into the aerobic organism culture (biomass). This process usually occurs in an aerated basin in which the primary effluent and the biomass are in contact for 3 to 6 h. Usually, more biomass organisms are grown than are needed for seeding, so some of the biomass must be removed from the basin. This material is called waste-activated (secondary) sludge and can be as much as two to three times the volume of the primary sludge. The secondary sludge is removed from the water by sedimentation, while the water, or secondary effluent, is either discharged to receiving waters or to land or is subjected to tertiary treatment to remove materials such as phosphorous, which can cause undesirable bioenhancement. Small amounts of trace contaminants, such as heavy metals and exotic organics, remain in this effluent stream.

The primary sludge, or the combined primary-secondary sludge, can be directly disposed of (e.g., by concentration and incineration), or it can be subjected to further biological breakdown. This biological step is usually termed sludge stabilization or digestion and can utilize either anaerobic or aerobic organisms. To grow, aerobic organisms need molecular oxygen, while anaerobic organisms only grow in the absence of molecular oxygen. Since aerobic digestion uses energy and anaerobic digestion produces energy, the latter process is the one most utilized. Approximately 70 percent of the wastewater sludge in the United States is anaerobically stabilized. Most usually, anaerobiasis is carried on in a mixed closed vessel, heated to 37°C. About 50 percent of the organic matter in the sludge is biologically converted to methane, carbon dioxide, water, bicarbonates, sulfates, and other metabolites. Approximately 15 to 18 cubic feet of gas, which is typically about 60-65 percent methane, are produced per pound of volatile organic material digested. The residual solids are called digested sludge. The digestion temperature is somtimes increased to the 49-65°C range. The higher temperature makes the residual digested sludge easier to dewater and deactivates a greater fraction of bacteria, viruses, and parasites (Garber, 1982b) . A considerable amount of work has been done on the question of whether residual digested sludge is microbiologically safe when used as fertilizer, and this work indicates that the microbiological hazard can be minimized. The extent of the hazard from heavy metals, such as cadmium, and from organics, such as PCBs, has remained the subject of controvery, although a number of guidelines have indicated practices that are considered safe.

Anaerobic digestion has been used to stabilize primary and primary-secondary sludge solids at most of the major wastewater treatment works over the last 50 years. It should be pointed out that only about a third of the solids are destroyed in this manner, so the residual digested sludge must still be satisfactorily disposed of. In the City of Los Angeles it is disposed of by sea, in Atlanta by incineration, and in Chicago on land. Digestion can be accomplished in as little as 3-5 days by maximizing the bacteriological breakdown process. More recently 37°C digestion followed by 49-65°C digestion with periods to allow maximum formation of organic acids to increase organic breakdown followed by wet combustion has been studied (Torpey and Andrews, 1983). The purpose of this work is to destroy organic solids so completely that only an ash will remain. Maximizing the biological breakdown process would minimize the energy needed for final disposal.

The agricultural and horticultural uses of residual digested sludge are affected by both the sludge's microbiological and trace contaminant quality. As noted, it is possible to achieve satisfactory microbiological quality, but background values for trace contaminants probably cannot be further reduced. While government agencies can identify and establish controls over industrial or commercial sources (point sources) of water pollution, they have little control over domestic (nonpoint) sources of trace contaminants, such as foodstuffs, cookware, plumbing pipes, cosmetics, and household cleaning materials. In order to know whether background levels of trace contaminants from nonpoint sources are such that the use of digested sludge in horticulture and agriculture is viable, a considerable amount of toxicity testing and evaluation of risk must be performed. The risk evaluation model, now used in EPA water-quality criteria (U.S. Environmental Protection Agency, 1979b) may be so conservative that its results in specific cases preclude what may actually be viable uses of sludge.

5.2.3 Sludge Conversion

In the past few years, most major cities have experienced considerable difficulty in obtaining new sites for sewage sludge disposal. Consequently, a variety of methods for stabilizing sludge or converting it to other products have been explored.

One of the older techniques for sludge conversion is that of drying. In this process, heat is applied to reduce the moisture content of sludge to approximately 10 percent. The temperatures attained during this process are high enough to sterilize the sludge. The resulting product is a low-grade soil amendment-fertilizer. Sludge drying has been used successfully by several major cities, including Milwaukee, Houston, Los Angeles, and Chicago. Sludge drying is expensive, however, since it requires substantial amounts of energy. Furthermore, marketing of the product can sometimes be difficult, and there is now greater concern over trace contaminants, such as cadmium. It seems clear that cities cannot totally rely on this method of disposal.

Another conversion process is composting. This is an aerobic-facultative organism process in which partially dry sludge is mixed with composted sludge or other materials. The resulting combination is dry enough so that oxygen can reach the microbiological organisms. Forced ventilation is used in static pile procedures, while windrow agitation often relies on natural ventilation. Stirring can be accomplished by the regular turning of windrows of the sludge or in mechanical units similar to furnaces used for cement production or multiple-hearth incineration. Composting provides further exothermic biological breakdown of the organic material, with the principal breakdown products being carbon dioxide and water. Temperatures high enough to inactivate bacterial, viral, and parasitic contaminants can be achieved. The final compost is a stabilized material well suited for application on land. Several major population centers, including Philadelphia, Washington, D.C., and Los Angeles County have used composting to dispose of part of their sludge. Composting can be an expensive operation, however, and marketing of the product can sometimes be difficult. Obtaining public acceptance of the process can also be difficult.

Other sludge treatment techniques include chemical stabilization and irradiation. Chemical stabilization is usually used as a temporary expedient prior to landfilling or other land disposal. Only limited experience is available on the use of irradiation.

A variety of thermal conversion processes is also available. These processes, which result in the conversion of sludge to gases and a residual ash, will be considered in a separate section, since air emissions and land disposal of an ash with concentrated trace contaminants introduce other disposal problems.

5.2.4 Pathogens

Domestic wastewater contains pathogenic microorganisms, such as bacteria, viruses, parasites, and fungii. These organisms are concentrated in sludge and must be inactivated to levels satisfactory to regulatory agencies prior to disposal.

Reductions in pathogen levels can be accomplished in secondary wastewater treatment and sludge processing operations (anaerobic and aerobic digestion). Further reductions take place in sludge storage systems and after discharge to the ultimate disposal medium. It has been suggested that ocean disposal causes increases in pathogen levels, but further work is required to provide or disprove this thesis (O'Malley et al., 1982; Sawyer et al., 1982).

The best evidence of the limited degree of microbiological hazard is the fact that digested sludge has been applied to land for many years with no known outbreaks of disease. Some agencies (State of California, 1956), as an extra precaution, recommend that root crops that can be eaten raw not be grown on land to which digested sludge is applied. This precaution may not be necessary for thermophilic digested sludge processed at 49-65°C.

Anaerobic or aerobic digestion, as well as composting and sludge pasteurization, probably lead to minimal hazard. Table 5.1 presents the results of bacteriological analyses conducted by the Municipal Environmental Research Laboratory (MERL), Environmental Protection Agency, Cincinnati, Ohio, on samples of raw mesophilic (36°C) and thermophilic (50°C) sludges from the Hyperion plant in Los Angeles (Garber, 1982b; Berg and Berman, 1980). The results indicate that both mesophilic and thermophilic digestion bring about substantial reductions in the numbers of the indicator organisms as well as Salmonellae. In fact, Salmonellae populations were reduced below detectable limits by thermophilic digestion. Twelve animal enteric virus analyses were also conducted, and the results are presented in Table 5.2. Mesophilic and thermophilic anaerobic digestion substantially reduced virus concentrations.

At 37°C, parasitic eggs are more resistant to destruction under mesophilic digestion than are viruses and bacteria. For this reason thermophilic digestion (50° C) is the process used to treat sewage in Moscow, USSR. Poppua and Bolotina (1963) state: "The most essential advantage of this process is the sanitary quality of the

TABLE 5.1 Reduction in Bacterial Densities in Mesophilic and Thermophilic Anaerobic Digestion (20-Day Detention) a

	Bacterial Densities (number/100 mL) [b]		
Bacteria	Raw Sludge Feed	Mesophilic Digestion (36°C)	Thermophilic Digestion (50°C)
Fecal streptococcus	2.7×10^7	2.0×10^6	3.7×10^4
Fecal coliform	3.6×10^8	5.5×10^6	2.9×10^4
Total coliform	5.2×10^9	7.0×10^7	6.4×10^4
Salmonella	7530	62	BDL [c]

a From Garber, 1982.
b Average of measurements taken over 2-year period.
c BDL, Below detection limits (<3/100 mL).

thermophilic (50°C) sludge. According to the health officials, viable eggs of helminths are absent from such a sludge." The work on sludge pasteurization tends to confirm the Soviet conclusions (Andrews, 1971).

TABLE 5.2 Temperature and Time for Pathogen Destruction in Sludges

	Exposure Time (Minutes for Destruction at Various Temperatures (°C)				
Microorganisms	50	55	60	65	70
Cyst of Entamoeba histolytica	5				
Eggs of Ascaris lumbricoides	60	7			
Brucella abortus		60		3	
Corynebacterium diphtheriae		45			4
Salmonella typhi			30		4
Escherichia coli			60		5
Micrococcus pycogenes var. aureus					20
Mycobacterium tuberculosis var.					20
Viruses					25

5.2.5 Trace Metals

To assess properly problems that may manifest themselves during the utilization or disposal of sludge, as well as possible long-term environmental consequences, detailed information on trace metals is necessary. The trace-metal concentrations in sludge are highly variable and, as stated earlier, are dependent on the source of the wastewater. Trace-metal concentrations may also be affected by the type of treatment process used and the degree of treatment. Table 5.3 is a tabulation of the trace-metal concentrations in sludges from treatment plants throughout the United States. These values are considered to be reasonably accurate for planning and analytical purposes. Sludges are seldom completely dry, but all data are expressed on a dry weight basis to facilitate comparisons.

TABLE 5.3 Typical Concentrations of Various Trace Elements in Municipal Sewage Sludges from the United States

	Milligrams per Kilogram		
Element	Range	Median	Mean
Arsenic	5-50	10	25
Boron	5-150	40	70
Cadmium	5-500	15	80
Cobalt	1-20	10	10
Chromium	20-20,000	400	2000
Copper	100-45,000	750	1000
Lead	20-20,000	450	1800
Mercury	0.5-50	7	10
Molybdenum	1-40	5	15
Nickel	10-4000	60	350
Selenium	2-20	4	10
Zinc	1000-6000	1800	2800

5.2.5.1 Potentially Harmful Trace Contaminants

Trace contaminants in the environment can affect human health as well as ecological webs involving the air, land, and water. In evaluating the effect of trace contaminants, the effect on societal health was considered the most important parameter to review. Effects on ecological webs insofar as they affected humans would then be of next importance, with the effects on general ecological balance including aesthetics following.

Elements Potentially Harmful to Man Trace contaminants that may be harmful to humans may enter the food chain through drinking water, food, or atmospheric fallout. Although sludges may contain many toxic chemical elements at elevated concentrations (e.g., arsenic, lead, and mercury), the extent to which these contaminants appear in food crops is dependent on the chemical characteristics of the soil, with pH levels being important. Similarly, the contribution of these contaminants to surface and underground waters that may be used as potable water supplies is dependent on the chemical nature of the contaminants and of the soil. Some organic materials, trihalomethanes, for example, may reach such waters quickly.

Cadmium often occurs in sewage sludges at concentrations higher than those found in soils, and it can accumulate in crops levels that may be potentially harmful to consumers if the pH is low and the cadmium is mobilized. The possibility of cadmium gaining entry to the human food chain is considered to be the most critical long-term environmental problem related to agricultural and horticultural uses of sludge. Elevated cadmium concentrations in food crops may not pose an immediate danger to the consumer, but accelerated dietary intake of cadmium could significantly increase the total body burden of cadmium over one's life span with consequent risk to renal functions (Hinesly, 1971).

The U.S. Environmental Protection Agency has established guidelines to limit both the annual and the cumulative amounts of cadmium from sludge that may be added to soils that are used for the production of food crops. For unrestricted land applications of sewage sludge, current EPA criteria for annual and cumulative total inputs of cadmium are 0.5 kg annually and 5.0 kg cumulatively. Where the pH of the soil-sludge mixture is maintained at levels greater than 6.5 and the cation exchange capacity of the soil is greater than 5 meq per 100 g of soil, the cumulative limits are somewhat higher. No limit on cadmium input is imposed on land used exclusively for producing livestock feed, but the operator of the disposal facility must ensure that the resulting animal food product is safe for consumption and that the sludge treated land will not be used at some future time for the production of crops that are connected to the societal food chain. Cadmium does concentrate in leafy crops such as tobacco, lettuce, and spinach, and input from these sources can be of concern to humans.

Lead is generally not considered to be a cumulative element in terms of crop absorption, but it may be harmful to young children who ingest sludge-contaminated soil. Use of sludge horticulturally in homes or in city parks increases the possible problem with children. The whole field of the assessment of toxicity for all trace contaminants needs thorough scientific evaluation. Definitions of toxicity, what to test, and how to test are also needed.

Elements Potentially Harmful to Animals A number of chemical elements (i.e., As, B, Cd, Cr, Co, Cu, Fe, Pb, Mn, Mo, Ni, Se, V, and Zn), when present in feed above certain critical levels, may be harmful to the domestic animals and wildlife. But the information available so far indicates that most of these elements are not likely to accumulate in crops grown on sludge treated lands to such levels (Sharma, 1983). Selenium and molybdenum--and possibly cadmium, cobalt, copper, and zinc--may be exceptions, however, since feed and food crops grown on soils receiving sludge at rates within EPA standards may contain levels of Mo, Se, and possibly Cd sufficiently high to arouse concern. It has been shown, for example, that when selenium is added to soils to a level of approximately 2 kg, it results in toxic levels of Se in forage crops (Sharma, 1983). Information on the concentrations of Se present in sewage sludges, while not extensive, tends to suggest that repeated applications of sludge to the soil could cause forage to absorb sufficient quantities of Se to be toxic to foraging animals. Lead may be harmful to foraging animals ingesting soil, but the hazard can probably be minimized by applying the sludge when the land in question is not being used for grazing.

Molybdenum, like Se, is an essential micronutrient, but it is toxic at concentrations slightly above the optimum. Toxic concentrations of Mo are related to the levels of Cu and SO_4 in the diet. When Cu is low, amounts of Mo as low as 5 $\mu g\ g^{-1}$ in forage may cause molybdenosis, or molybdenum-induced copper deficiency. Recent studies conducted in the United Kingdom indicate that forage grown on sludge-treated soils may accumulate sufficient quantities of Mo to be toxic to animals (U.S. Environmental Protection Agency, 1979a).

The problem of cadmium toxicity in human foods grown on sludge-treated soils has already been considered. As for animals, it has frequently been observed that elevated concentrations of cadmium in feed obtained from soils treated with sewage sludge have not adversely affected animal health. However, soils vary in acidityalkalinity. Studies did not necessarily take this into account. In acid soil, cadmium is picked up in leafy plants and ingested. Additional observations are needed to determine whether cadmium has any effect on animals under various soil conditions.

Excessive dietary Co can cause toxicity in ruminant animals, but nonruminant animals are hardier. Diets containing more than 10 μg of Co g^{-1} have injured cattle and sheep. Crop plants used as animal feed have been shown to accumulate Co in excess of 10 μg g^{-1} when cobalt was present in the soil and other soil conditions were favorable (A. Page, University of California, Riverside, personal communication). It is considered improbable that ruminant animals would be at risk when foraging on sludge-treated soils, however, since only a few plant species accumulate Co and the concentration of Co in sludge is low.

Copper fluoride toxicity has been reported where excessive amounts of this compound occurred in the diets of sheep, cattle, and swine. As with Co, however, the probability that this element would accumulate to harmful levels in forage crops grown on sludge-treated soils is remote.

5.2.5.2 Phytotoxic Elements in Sludge

Studies have shown that repeated applications of sewage sludge to land may result in the accumulation of B, Cd, Cu, Ni, and Zn to levels that are phytotoxic. While B is quite mobile in soil and can be leached to nontoxic levels, Cd, Cu, Ni, and Zn are quite immobile and tend to accumulate at or near the depth of incorporation.

Plant species vary substantially in their ability to tolerate B, Cd, Cu, Ni, and Zn in soils. Even varieties of the same plant species frequently show marked differences in tolerance. The phytotoxic effects of Cd, Cu, Ni, and Zn are far more prevalent and acute on plants grown in acid soil than on plants grown in neutral or calcareous soils. Except for cadmium, EPA does not currently regulate the amounts of these elements that may enter cropland soils through sludge applications. However, a number of the states have developed land disposal guidelines that specify limits for Cu, Ni, and Zn. Limits vary somewhat among the various states. The limits most commonly cited call for the affected soils to be maintained at a pH of 6.5 or greater and require cumulative loadings of Cu, Ni, and Zn not to exceed 250, 500, and 1,000 kg ha^{-1}, respectively. The upper limits of metal loading are based in general on the cation exchange capacity (CEC) of the soil; as the CEC increases, the quantity of metal may increase accordingly. These limits are designed to protect the productivity of soils and apply only to soils used for the production of crops.

Phytotoxicity is also considered in Chapter 3, Section 3.2, and Chapter 4, Section 4.2.9. Specific concerns are possible long-term biological effects and possible long-term removal from the soil to the groundwaters.

This chapter considers the effects of inorganic contaminants on societal health and sensitive ecosystems since the panel had available to it information regarding inorganic materials arising from actual sludge applications. The effect of undesirable concentrations of xenobiotic contaminants is considered in Chapter 4 on biological concerns since much of the ongoing work has been carried on by biological scientists.

5.2.5.3 Selection of Disposal/Utilization Sites

The same hydrological and geological considerations that apply to land disposal of other contaminant containing wastes must be taken into account in selecting a site for sludge disposal. Except for B, Mo, and Se, trace elements are chemically more active (i.e., more soluble and mobile) in acidic than in neutral and alkaline soil. Therefore, the chemical characteristics of the soil receiving the sludge are also important.

Also of importance is the extent to which these trace contaminants will remobilize and therefore how long the sludge-treated soil will present a hazard. The relative permanancy of disposal into any medium should be an important part of any environmental evaluation. Sludge deposition on soil may result in a relatively permanent potential hazard. Although it has often been assumed that these contaminants could be better controlled through land disposal than through air or water disposal because of the relatively inert nature of trace elements in the soil, it is not believed that this postulate has been adequately proven. Certainly, a review of the pollutants to which EPA gives highest priority (U.S. Environmental Protection Agency, 1979b) would indicate that there are many chemicals whose mobility in soil and plants is poorly understood. It would therefore always appear to be necessary to make a careful review of all media for each specific disposal problem, since changes in soil and groundwater caused by sludge disposal may be long lived.

5.3 AVAILABLE DISPOSAL AND/OR REUSE OPTIONS

Available technologies for sewage sludge management are discussed in the following subsections.

5.3.1 Thermal Processes

Thermal processing accomplishes volume reduction of sludges and partial disposal to the atmosphere. All processes produce a residual that generally requires land disposal.

5.3.1.1 Incineration

Wastewater and other organic sludges can be disposed of or reduced in volume by incineration processes. The temperature of the incinerator must be on the order of 705°C to prevent smoke and odors and as much as 1650-2200°C to ensure that the bulk of the organic matter is converted to heat and carbon dioxide. Approximately 30-50 percent of incinerated wastewater sludge is left as nonvolatile ash, depending on such factors as the sources of the solids (i.e., separate or combined sewers and percentage and types of industrial-commercial flow). Such ash may contain concentrated amounts of toxic metals that are also likely to be more soluble/mobile because they are no longer tied to organic material and that may consequently result in its being classified as a hazardous waste that must be disposed of at specifically permitted sites. If on land, these disposal sites must be designed to minimize the possibility of groundwater contamination.

Because incineration processes produce gaseous and particulate effluents, a careful review of their use in air-quality nonattainment areas is quite necessary. Cleanup trains, which partially remove particulates NO_x, SO_x, and residual hydrocarbons are available and must be evaluated. Heat and carbon dioxide produced by incineration may also be serious pollutants in enclosed basins, such as those at Salt Lake City and Los Angeles (G. Csanady, Woods Hole Oceanographic Institution, personal communication). Such materials as arsenic, mercury, cadmium, and lead may pass through the cleanup trains and enter the air, with eventual impacts on public health and land and water resources.

A special caveat is that for some percentage of the time there will be operational problems, and the likelihood of such problems increases with process complexity. Any overall evaluation of a sludge disposal operation must therefore include an estimation of the reliability of the disposal process. With respect to incineration, representative operating data that make it possible to estimate rates of volatilization of metals and other constituents at various operating temperatures is important information.

The operational and maintenance costs of incineration are usually high because capital, energy, maintenance, and personnel costs are high. Including interest on capital, operating costs range up to $300-400 per dry ton of sludge combusted.

5.3.1.2 Pyrolysis

Pyrolysis is a starved air (oxygen) combustion process that is used to burn sludges at between 500 and 900°C, producing either fuel oil or combustible gas and an ash. The fuel oil or gas is then used as an energy source for boilers, gas turbines, internal combustion engines, or similar units that produce electrical current or directly drive other equipment. The earlier comments about ash disposal and air emission problems caused by incineration also apply to pyrolysis through the gas-oil combustion step. Because pyrolysis is an incineration process, gaseous and particulate effluents from its separation and from use of its products must be reviewed in terms of air-quality requirements and land disposal of its ash. Organic contaminants are broken down, but trace metals will concentrate in the ash. Costs are as high. A difficult-to-treat liquid side stream, as from ash quenching or air-pollution control processes, may occur and have to be treated in wastewater treatment facilities.

Pyrolysis generally requires sludges that are 25-35 percent solids (cake). This is necessary to permit the combustion process to be self-sustaining. Cake is fed into the combustion chamber at a maximum rate of 8-12 pounds wet weight per square foot of hearth area per hour. Multiple-hearth and rotary-hearth furnaces may have differing requirements, but the oil or gas produced is essentially similar in composition.

When a pyrolytic combustor is used to make an oil, the fuel value of the composite liquid residue is approximately 69,300 BTUs per gallon at 500°C, and 87,800 BTUs per gallon at 900°C. By comparison, heavy petroleum has a fuel value of 120,000 BTUs per gallon.

5.3.1.3 Wet Combustion

This process consists of feeding a ground-homogenized liquid-sludge slurry of approximately 3 percent dry solids into an enclosed reactor operated at about 315°C and 1800 psi at a rate of about 2 tons per hour. The reactor may or may not be operated at controlled oxygen levels. At this temperature and pressure the system is capable of converting organic matter to CO2, which is vented to the atmosphere, and water. The system is not self-sustaining; fuel requirements range from 900 to 1,000 BTUs per gallon of sludge.

The gases generated by the reactor must be treated by an afterburner operated at between 345 and 400°C to control odorous emissions. The combusted material is sent to a liquid/solid separator, from which a liquid, high in organic content, is returned to the wastewater treatment plant for additional treatment prior to discharge. The solids/ash are disposed of in a landfill.

Construction costs are reported to be about $130,000 per dry ton per day of capacity, while operation and maintenance costs range from $35 to $90/dry ton.

5.3.2 Land-Based Alternatives

Land disposal of sludges has been considered in detail in Chapter 3. However, certain features common to all the available land-based options for wastewater sludges should be emphasized. These include the need to protect surface waters from contaminants in runoff and to protect potable groundwater from contaminants in leachates. In situations where commercial crops that enter the food chain are grown on sludge-treated soils, the productivity of the soil must be maintained or improved and potentially harmful elements must not accumulate in the crops. In situations where sewage sludge is used to produce crops used as feed for farm animals, which in turn are used as food for human beings, it is necessary to ensure that the food does not contain contaminants at concentrations that may be harmful to man. Where sewage sludge is used to reclaim marginal or drastically disturbed lands, surface water and groundwater must be protected as well as the health of the livestock.

Current federal regulations address all the above issues (U.S. Environmental Protection Agency, 1978) and are more than adequate to provide the margin of safety required.

Landfills have been increasingly used to isolate wastewater sludges containing trace contaminants at levels high enough to be of regulatory concern. The assumption has been that remobilization of such contaminants is minimized by using landfills and that release of contaminants to the environment is unlikely. The panel believes that the data supporting such a conclusion are scant and that remobilization of contaminants in surface and groundwaters as well as to the atmosphere is possible. The finding of substantial vinyl chloride in landfill gases (Los Angeles Times, 1983), as well as the mobilization of metals because of the generation of carbon dioxide and conversion to carbonic acid with lowered overall pH, are well known. The effects and risks of using landfills for sludge disposal, however, are still largely unknown. Suitable land disposal sites are also becoming increasingly scarce, particularly in urban areas, and an assumption that such sites will always be available may no longer be valid.

5.3.3 Ocean-Based Alternatives

A considerable amount of effort has been devoted to measuring the constituent elements in domestic and industrial wastes, analyzing concentration and dispersion in the marine environment, and assessing the effects of disposal on marine organisms that have direct or indirect contact with wastes. An inherent problem in attempts to analyze and synthesize the available information is the substantial effort required to locate material published in a wide variety of disciplinary journals and symposium volumes, as well as numerous unpublished reports and data files. The problem is further complicated by the difficulty of assessing the quality of this information, since critical details about analytical techniques and accuracy are often not readily available. Some previous attempts to assess the problem of waste disposal (for example, National Research Council, 1976) have concluded that the problem is basically intractable from the standpoint of existing knowledge and that a long-term and large-scale examination is needed to obtain a sufficient base of information about the marine environment. The implication is that the marine system is so complex and so poorly understood at present that there is no reasonable hope of evaluating the problem of safely disposing of waste including sludge. The present state of knowledge does not appear to support this conclusion. Enough information now exists about the New York and southern California Bights to allow a reasonable evaluation of the environmental impacts of specific discharges. This in turn allows a comparison with land or air disposal options to determine the disposal procedure most likely to cause the least net negative environmental impact. Further studies in areas such as pathogen persistence may well be desirable, but much of the work now going on can best be categorized as a perpetuation of data gathering rather than attempts to synthesize and interpret information so as to achieve more rational decisions. The panel believes it is important to redirect some of the funds now being used for monitoring toward filling gaps in present knowledge. Chapter 2 discusses these gaps.

5.3.4.1 Chemical Fixation

Wastewater sludges and sludges arising from certain chemical manufacturing operations now disposed at certain ocean dumping sites (chemical sludges) can be fixed or solidified by the addition of chemical materials. Limestone, cement, fly ash, polymers, clay products, sand, gravel, and plastics are mixed with sludges to produce a product with high solids content that will reduce leaching of pollutants when finally disposed of (e.g., in a landfill or ocean dumping site).

The largest known use of this procedure is in Japan, where the solidified material is deposited within a massive sea wall in Tokyo Bay (Fifth Japanese-United States Conference on Solid Waste Management, 1982). Floating pump barges and finally wells within a completed fill area continuously remove water, which is treated to remove organic material from biological breakdown in the fill and trace hazardous contaminants. Experience has shown that solidification-encasement does not entirely eliminate leaching of hazardous materials.

There are currently a number of solidification processes available, but in the United States it is estimated that less than 10 percent of wastewater sludge is handled in this way. The procedure seems equally applicable to small and large facilities. Current costs are in the range of $10-30 per wet ton of sludge treated.

5.3.4.2 Thermal Fixation

An example of thermal fixation is Philadelphia's EcoRock project, developed in conjunction with The Franklin Institute. Sewage sludge and municipal incinerator residue are mixed, dried, and burned in a rotary kiln, then fused into a rocklike material in a fusion furnace. The resultant product is intended for use as a pavement aggregate. The system is oil-fired, and emissions to the air are controlled by cyclones and a venturi scrubber. For odor control there is a column packed with potassium permanganate that can be applied to exhaust gases. Solids from the emissions control system are cycled back to the waste input stream.

Inputs to the EcoRock system are anaerobically digested sludge cake that is approximately 20 percent solids and unscreened incinerator residue. Large pieces of scrap metal are removed from the residue and sold. The rocklike output is crushed prior to utilization in pavement. The system produces three revenue streams that offset part of the operating costs: fees for incinerator residue delivered to the process, revenue from the sale of recovered scrap metal, and revenues from the sale of the final product.

EcoRock began as a pilot plant processing 20 dry tons per day in April 1982, and operational data are not yet available. Mechanical performance, maintenance experi ence, energy consumption, emissions, product characteristics and marketability, and costs will be assessed annually. Current estimates are for a net unit cost of $155 per dry ton of sludge.

Another example of this technology is a project developed by the Washington Suburban Sanitary Commission (WSSC) in cooperation with the University of Maryland for converting sewage sludge to brick. Sewage sludge of various types is dewatered to about 20 percent solids (cake) and mixed volumetrically with clay and shale as follows: sludge cake = 30%, clay = 46.66%, and shale = 23.34%.

The mixture is molded in regular brick molds, after which it is extracted and air dried for about 24 h. The material is then further dried by the hot gases from a brick kiln (200 to 300°F) for another 24 h. The material then travels through the brick kiln at an initial temperature of about 200°F and progresses through a maximum temperature of 2000°F down to a temperature of about 200°F at exit from the kiln. The total firing time ranges from 30 to 40 h.

The finished brick is said to be of construction quality and is currently priced at 12¢ to 14¢ per brick. The technology is said to be ready for full-scale production, with between a half million and a million bricks already produced.

5.4 LOCAL CONSIDERATIONS

Any environmental legislation or regulations to control the disposal of sewage sludge must allow decision makers sufficient flexibility to take local considerations into account. Current regulations for air, land, and ocean disposal have been developed separately and do not necessrily provide equal levels of protection for a specific location. The physical and sociopolitical characteristics of the locality where the sludge is generated, as well as those of the locality where it is to be disposed of or utilized, will so strongly affect acceptability or implementation that they may have to be given precedence over questions of cost-effectiveness and optimal technology. They include:

5.4.1 Environmental Considerations

5.4.1.1 Geography

The location of a community can limit the disposal options available to it. A city without port facilities or access to the ocean shoreline is virtually precluded from choosing an ocean-based option. A city in an airshed that is a nonattainment area may find that the construction of a sludge incinerator would cripple its air-quality management efforts or that air-quality regulations prevent use of air disposal. A land disposal plan may be eliminated by a shortage of permitted landfill space, steep terrain, an absence of disturbed land for reclaiming, or lack of appropriate types of agriculture. The information needs are rather obvious and can be met early in the decision process by developing a geographical description of the region.

5.4.1.2 Climate

In much the same way, climate can limit disposal or utilization options and the community's ability to modify its sludge through changes in wastewater treatment or sludge conversion. Stripmine reclamation, for example, cannot be practiced in Pennsylvania between October and April because of frozen ground and snow. If sufficient storage space for sludge is not available, other options must be included in the sludge management scheme. When planning a system for a specific location, information needs regarding climate can generally be met by available descriptive data on the region. In terms of sludge disposal data, the experience of other agencies in the general areas regarding conversion, disposal, and utilization is usually of great help.

5.4.1.3 Unique Environmental Features

Unique environmental features are such things as aquifer recharge areas, wetlands, deserts, endangered species habitats, and recreational areas. The extent to which such features limit the selection of sludge-management options can be partially determined by an analysis of their proximity to the proposed disposal and of their susceptibility to physical, chemical, biological, or aesthetic alteration. The extent to which the public values these features and perceives them to be threatened by sludge disposal plans is sometimes a large component of the decision. The former type of information can be obtained through an inventory of regional features, but the latter may not become evident except through public participation in the decision process.

5.4.1.4 Other Factors

Other environmental considerations that may influence the selection of a sludge option include available water supply and energy resources. Again, both an objective analysis and an understanding of how these features are perceived by the public are necessary to determine how they may affect the selection of a sludge disposal option. A groundwater-dependent community, for example, will need to be especially careful in selecting and designing a land-based disposal plan. It will be important that such a plan minimize the risk to groundwater. However, perceptions of the risk by local citizens can overshadow the results of any risk analysis and preclude certain options. Information needs are much the same as in Section 5.4.1.3.

5.4.2 Sociopolitical Factors

The sociopolitical factors that are important facets of any environmental decision are reviewed in Chapter 1. It will therefore suffice here simply to reiterate items that have been important to many decisions regarding the disposal of wastewater sludges. Included are a review of the relation of the governmental structure to the decision process. To what degree does the local government control decisions? Population density, per capita income, educational levels, patterns of land use and ownership, recreational habits, open-space availability, agricultural practices, types of businesses, and regional economic conditions may all be important aspects of any decision. For example, it is often easier to place facilities such as landfills in poorer regions than in those where there is a concentration of wealth. Regional rivalries and the concept of a big rich area transferring its wastes and problems to a poor and sparsely populated area may often be pivotal in a project. This was appar ent in the attempt by Los Angeles to use the desert for solar drying and disposal of wastewater sludges. Similarly, political rivalries may overshadow environmental or technical merits. An example of this would be rural legislators joining together to prevent a large metropolitan area from getting state funds or from effecting disposal plans outside its own boundaries.

5.5 EVALUATION PROCESS

5.5.1 Introduction

There are five components of an overall multimedia evaluation. Each of these must be addressed for each feasible disposal alternative. Since a review of all possibilities would be an almost impossible task, however, it is assumed that site-specific or local considerations and technological evaluation will narrow the number of alternatives. These can then be considered in a more definitive evaluation. It should then be possible to select a cost-effective alternative that permits a balancing of available and reliable technologies, their environmental effects, and relative costs. Public participation and agency responsiveness to public concerns should help to foster public acceptance of the optimum disposal practice.

The first element is the process of informing the public and political decision makers about the scientific-technical and environmental facts pertaining to a specific proposal. Present laws and common sense both make public participation a part of the solution. In Japan (Fifth Japanese-United states Conference on Solid Waste Management, 1982) and Scandinavia the citizens most likely to be affected by a sludge disposal plan are told of its possible effects and given some substantial benefit for allowing a project to be placed in their area. They may be supplied with free gas from a landfill, for example, or free hot water from an incinerator or pyrolysis unit. This would appear to be a procedure that should be considered in the United States.

The mass communications media must also be fully advised about any sludge diposal plan. There appears to be a great deal of emotion surrounding the ultimate disposal of wastewater sludge or its products, such as ash. By stressing factual information and identifying emotional issues, the mass media can help to achieve a responsible solution. Similarly, the agency or agencies involved in finding a solution to a sludge disposal problem must not concentrate on technical-scientific aspects alone. They must also consider the public interest or concern, mass media desires to keep readers or viewers informed, pressures on regulators to ensure that beneficial environmental values are preserved, and needs of legislators to assure constituents of their continuing attention to environmental problems. A complete plan for a project must consider these aspects as well as pure engineering or scientific concerns.

Resolution of intergovernmental relationships is considered the second element. The problem of complex and possibly conflicting laws and of differing court interpretations of portions of these laws has been covered in the literature (Garber 1982a). Although further resolution of this area of possible conflict is necessary, it is essential that industrial, municipal, state, and federal governmental agencies work as cooperatively as possible toward a solution to any specific environmental problem. It is probable that areas of disagreement will be present; but adoption of an adversarial approach to a problem makes an environmentally responsible solution much more difficult. A decision matrix must include a factor covering intergovernmental relations. A cooperative effort, within the interpretations of the laws, skews the matrix less and makes a reasonable scientific-technical and cost-benefit solution much more probable.

An example of a factor that has caused difficulty in this necessary interrelationship is the fact that with municipal sludges there are nutrient materials such as nitrogen, phosphorous, silica, carbon, vitamin B12, and certain necessary trace elements. The positive value of these nutrients for sludges used as soil amendments as well as diffused in the ocean must be balanced against the negative effects of trace contaminants such as cadmium and lead or xenobiotic compounds. This has been difficult to do because of a general plan from federal and state regulatory agencies as to the uncertainty of knowledge regarding the possible negative effects of trace contaminants such as those in the EPA list of priority pollutants. Standards for allowable effluent or receiving-water concentrations of the priority pollutant reflect the uncertainty felt by the regulatory agencies.

For discharges to ocean waters EPA Water Quality Criteria (U.S. Environmental Protection Agency, 1979b) have been used by some states as the basis for effluent and receiving-water concentration standards, despite EPA's indicated reluctance to adopt them as standards on their own, because of questionable scientific basis. The Criteria includes guidelines for all the priority pollutants. Each Criterion is based on available toxicity information for that pollutant as processed in a computer model based on the assumption that for all pollutants there is no tolerance and that in humans there should be no more than one teratogenic, mutagenic, or carcinogenic event in 100,000 to 10,000,000 exposures. The panel believes that the concentration guidelines so developed are overly conservative since many of them were lower than concentrations found in most potable drinking waters or were one or more orders of magnitude lower than the detection limits for known methods of analysis. Additionally strict application would probably preclude any use of the ocean medium. The State of California Ocean Plan (State of California Water Resources Control Board, 1978) was based on a study of acute and chronic toxicities for each pollutant, made provisions for initial dilution, and included corrections such as that for the amount of the trace contaminant already natural to the receiving waters. The panel considered it to be quite conservative but a more scientifically satisfactory approach to the problem of setting receiving-water concentration standards.

Environmental criteria, such as the EPA Water Quality Criteria, can have substantial impact on the design, construction, operation, and cost (capital and operational) of treatment and disposal facilities. Criteria are often misapplied (for example, by using freshwater-toxicity-based standards for saltwater). Evaluation of specific criteria as well as evaluation of specific projects must take place if the lowest environmental impact is to be achieved.

The third element in developing a multimedia decision matrix is the group of scientific-technological factors that are now known and that can contribute to a responsible environmental solution. That is, is there a technology or a group of technologies that will provide compliance with regulatory standards and a process that is reliable and has a reasonable operation and maintenance cost? It may also be of value to the decision makers to know whether the evaluation shows that there is a procedure that appears to have a lower (or the lowest) net negative environmental impact but that the laws and regulations appear to prevent. This may allow them to work toward necessary changes in the laws, regulations, and standards for the receiving medium to make a previously prohibited procedure possible.

To be considered truly available, a sludge disposal process must have been either in operation for a number of years or must have recently gone through a procedure of laboratory evaluations, pilot plant operation, and expansion to a full-scale operational unit from which operational data suitable for design decisions are available.

Sludge management or disposal systems are substantial public works projects with high capital costs that are designed to operate from 20 to 50 years. Consequently, local policy makers would be well advised to evaluate carefully the state of the true availability of a technology before committing themselves to its implementation. Failing this, they may have to seek substantial funds to rebuild the system.

Both proprietary and nonproprietary systems should be considered. Imported and sole-source systems should be studied with special care, since maintenance costs can become a serious problem.

System reliability is important. This means that a system should have a good performance and maintenance record over an extended period of time. This approach, however, can be criticized as discriminating against emerging technology. Local policy makers should therefore decide whether to purchase a well-established system or a new system with little experience but strong advocates.

The case of operation and maintenance is another important factor that is sometimes overlooked or given inadequate attention during the evaluation process. This may be addressed in the following ways:

- Assure that the control funding agency and the designers have full recognition of startup and continuing costs. Such costs should be an important part of the evaluation of competing systems for handling sludge processing and disposal.
- Funding agency and designers should recognize the impacts on the community and the workers of upgrading the plant staff to handle the new system.
- The funding agency and local decision makers should make sure that an adequate engineering staff is available to design the system and aid in startup. A

premium should be included in financing plans to cover projected engineering costs.

An environmental assessment should include examination of ecological effects, human health effects, and cultural effects. Such an evaluation would include a balancing of the environmental effects of sludge disposal options, utilizing a matrix approach that compares the potential effects of each option in each medium. Some of the comparisons will be quantitative, while others will be semiquantitative or qualitative. The panel believes that enough data are currently available or are readily obtainable to make well-reasoned, technically supported decisions about sludge disposal. Where the data are not so well developed as may be desired, ongoing multimedia assessments can be used to identify further research needs.

The starting point in an environmental evaluation is a comprehensive characterization of the wastewater sludge itself. Such a characterization should include estimates of sludge quantity, solids content, nutrient content, biological oxygen demand (BOD), heavy metals, persistent organics, and pathogens. It is also important to understand how quantity and quality are expected to change over the next 5, 10, or 20 years in response to service-area changes, pretreatment programs, or general regulatory actions, such as bans on DDT and PCB manufacture and use.

After the present and future characteristics of the sludge are determined, technological and local agency operational preferences are used to select the most feasible disposal option for further evaluation. For each of the disposal technologies identified in this process, the affected media (air, land, water) are then specified. Most of the potential effects of disposal in the ocean will be contained there, although it is theoretically possible that some of the effects will reach human being through the consumption of seafood or exposure in recreational beaches. Disposal in the air will cause increased air emissions, as well as deposits of incinerator ash or pryolysis residues on land. A portion of the air emissions ultimately come back to the land or oceans in precipitation or by particulate settling. Disposal on land will mean direct depositing of the sludge in landfills or in other soils. Constituents of the sludge will eventually move into soils and may potentially move into ground or surface waters.

After the affected media are identified, it is necessary to develop estimates of the dispersion of the constituents of concern in each medium. It is important to estimate both short-term dispersion (to address potential acute toxicity effects) and longer-term dispersion (to address potential chronic effects). Models of the physical transport relationships are fairly well developed for the ocean and air and to a lesser extent for soil-groundwater movement. Where possible, an additional element signifying the chemical fate of the constituents should be added to the model. Analysis should focus on those materials that may affect the ecosystem or human health.

In developing an understanding of trace-contaminant movements in a medium to be used by a sludge disposal project, it is important to contrast proposed contaminant loading from the sludge with the comtaminant loading already in the medium at the proposed disposal point. That is, it is important to put into perspective whether the contribution by the wastes will represent a large or small percentage of the amount of contaminant that will be present after a project is in service.

While ecosystem effects are addressed in detail in Chapter 4, a brief reiteration is appropriate here. It is important to collect background ecosystem information for each medium. In designing programs to collect this information, it is important to have adequate control areas--as similar as possible to the discharge area but distant from proposed discharge points. This should help to identify effects that may be the results of more widespread environmental factors.

The background data that should be collected should include hydrological or meteorological data; sediment or soil quality data; fisheries or terrestrial population data and associated diseases; benthic or soil community data; and information on plant, animal, and human uptake of constituents of concern. Enough information on uptake has been collected to make reasonable multimedia assessments of potential effects on ecosystems or areas of particular interest such as public beaches.

In addition, bioassay data should be assembled or developed on the acute toxicity of constituents to aquatic or terrestrial organisms. Further analysis should include laboratory studies that address bioaccumulation potential and its possible effects on growth or other sublethal responses. Such test methodology is well developed. Short-term tests can be used in conjunction with dispersion models to assess potential acute effects, while long-term tests, such as bioaccumulation evaluations, can identify potential chronic effects that require further evaluation.

Using the various kinds of data mentioned above, comparative assessments can be made to determine whether disposal will lead to unreasonable short-term effects, unreasonable chronic effects, or the destruction of unique habitats.

Another ecosystem issue that should be addressed is whether changes in the media that are caused by sludge disposal will be temporary or permanent.

Ecosystem data can be compared to determine the relative effects of different disposal options. Some of these will be quantitative comparisons, but many will be semiquantitative or qualitative. As more data become available, such multimedia comparisons will become more quantitative in nature.

5.5.2 Human Health Risk

A multimedia environmental evaluation should also include comparisons of the human health risks of a proposed disposal option. These comparisons should focus on the potential health risks associated with metals, persistent organics, and pathogens. Many of the required elements needed for these comparisons are available, having been developed in the process of establishing regulatory criteria for the individual media. What is needed is an integration of the individual media assessments into a multimedia, multichemical analysis. Such an analysis is currently possible.

It is necessary to identify human populations at risk, estimate average population exposures to the pollutant, and determine the probability of a health effect for each pollutant in each medium. At this point a multimedia comparison should have been developed for each individual constituent of concern to human health. The next step is to normalize the data on a wide variety of constituents and their associated health effects into a common index that permits overall relative health risk comparisons. Methods have been developed to rank the severity of various health problems (e.g., nasal cancer, lung cancer, kidney disease, dermatitis) and to use such rankings to normalize individual health effects into a common denominator.

It is also important that the assessment address the “added,” or “delta,” risk to human beings rather than the absolute risk. Such risk indices are useful in making disposal decisions by showing how risk can be minimized.

Although multimedia risk assessment is a relatively new concept, much of the necessary methodology already exists. The sophistication of these multimedia assessment techniques will improve in the future, but valid assessments are possible today.

The last element of a multimedia environmental assessment includes cultural considerations (aesthetic, recreational, and historical resources, for example). These judgments are qualitative in nature and include direct comparisons of whether any unique cultural resources exist in each medium and how such resources will be affected by constituent loading.

5.5.3 Conclusions

A combination of quantitative, semiquantitative, and qualitative assessments will make it possible to address the issues raised by sludge disposal in each medium. These evaluations will be based on sludge characteristics, local physical characteristics, and the characteristics of local or regional media. The simplistic approach to the problem of sludge disposal suggested by a single national standard is inappropriate and could result in decisions that are environmentally and financially irresponsible.

Scientists can always argue that more definitive data would be helpful or that better models should be used. To some degree this is always true. However, enough data are available within the many technical disciplines to allow adequate multimedia analyses. What is required now is open, effective communication among the scientists working on these problems so that the best information can be integrated into the analyses. Table 5.4 is a simplified version of a multimedia matrix as used by environmental consultants to assess the barge disposal of sludge in the New York Bight by New York City (Hinesly, 1971).

TABLE 5.4 Assessment Data Requirements and Availability [a]

	Medium of Disposal		
Data Element	Ocean	Air	Land/Groundwater
Sludge characteristics	Readily available and usable in each medium		
Dispersion models	Good [b]	Good	Average
Constituent mass contribution to media	Good	Good	Average
Ecosystem data			
Biological characteristic	Good	Average	Good
Physical characteristic	Good	Good	Average
Chemical characteristic	Good	Average	Average
Acute toxicity data	Good	Average	Good
Chronic toxicity data	Average	Average	Average
Disposal technology			
Operational engineering data	Good	Average	Average
Human health data			
Population at risk	Good	Good	Average
Pathway to man	Good	Good	Good
Estimate average exposure	Good	Good	Average
Data on probability of health effects given a certain average exposure	Good availability of the data and is equally useful		

[a] It should be recognized that these rankings are relative. The data base for ocean use is the best; but in actuality there is much more work to be done on the ocean to understand really how best to utilize it. This then means that the data bases for air and land are much poorer. In terms of making a decision needed now with the data bases we have now the rankings of Table 5.2 are reasonable. In terms of knowledge that would be desirable to have in order to insure the most responsible answer to a question that we might term the ocean data base as poor to fair, the air data base as poor, and the land data base as poor to very poor.

[b] Data availability ranking: good, average, lacking.

5.5.4 Financial Assessment

There is no doubt that the funds spent on a major public-works project, such as a sludge processing and disposal plant, will produce jobs and other economic benefits for a community. A cross-media analysis to identify the most beneficial disposal option should precede any such expenditure. Other financial considerations that must be included in the analysis include operation and maintenance, financial effects on industry and commerce, and effects on property values.

5.6 SUMMARY

The question of how to dispose of wastewater sludge is now surrounded by a considerable body of law, regulations, and judicial decisions. Making a decision about sludge disposal on the basis of relatively simple scientific, technical, and economic analyses is impossible. Since there is a keen public interest in each decision, public perceptions become important. The City of Los Angeles, for example, disposes of approximately 22,000 tons of domestic, commercial, and industrial solid wastes in landfills per day. The city must also dispose of approximately 200 tons of anaerobically digested waste-water sludge per day, or less than 1 percent of all the city's wastes. But disposal of the sludge in the landfills is politically unreasonable, since citizens have demanded closure of the landfills if they are used for the sludges. Rational or not, public opinion must be considered as important as the scientific, technical, and economic facts that would have been relied on for a decision if there had been no public interest.

Notwithstanding the situation in Los Angeles, however, the members of the panel found that the public and its professional advocates have become increasingly sophisticated about waste materials and their disposal. There now seems to be no preferred receptor medium, and searching questions are asked about each specific waste disposal project and about each part of the environment proposed for disposal. There is no longer a strong skewing toward land disposal only, and support for the use of a single national standard, regardless of environmental impact, has waned. Instances of serious groundwater pollution, and a finding that reclaimed storm water in a metropolitan area was more mutagenic than reclaimed sewage (Nellor et al., 1982), have illustrated the need for full receptor media analysis. The Environmental Impact Study required under the National Environmental Policy Act was intended to accomplish this, but such studies have been constrained by legal and regulatory decisions. A return to objective and scientific evaluations is now required.

The panel believes that a definition of sludge is required. Wastewater sludge and its disposal or reuse can be significantly affected by such factors as whether sewers are combined or separate, the amount of industrial sewage, industrial waste control and pretreatment practices, the types of industries using the sewage system, the characteristics of the potable water discharged into the sewer system, the length and temperature of the sewer system, and the food consumption habits of the local population. A definition of sludge must also take legal-regulatory semantics into account. In California, for example, a composted sludge at 20 percent moisture can be taken to a nontoxic solid-waste disposal site, while the same sludge taken from a centrifuge at 80 percent moisture must go to a toxic-waste disposal site. Since toxic-waste sites are rare and expensive, a multimedia analysis would have to factor in this important difference in interpretation of the toxicity of the sludge. Similarly, sludge that has been composted may be usable as a soil amendment, but the ash from sludge that has been incinerated may be considered a toxic waste. Because of problems like these, a major food processor will not accept fruits or vegetables grown on soil where sludge has been deposited (Rawls, 1982). The problem from a scientific point of view is not real, but the processor is governed by possible public rejection of products so fertilized.

How then can sludge reuse or disposal be evaluated so that the public will accept a decision that by normal scientific-technical procedures is responsible? The panel believes that a multimedia evaluation that includes public meetings, full public information, maximum media participation, and full review of the concerns of environmental organizations is necessary. Such an evaluation would also include a study of the important political factors surrounding a project. In short, a multimedia analysis must be concerned with several types of facts. These would include a scientific-technical evaluation of the various options, public perception of the problem, the options believed acceptable to the public, economic assessment, regulatory-political-judicial concerns, and the concerns of national environmental groups. Regardless of what is correct from a strictly scientific point of view and of what measures might be the best ones to protect known environmental values, public concerns may take precedence. The multimedia matrix allows all of this information to be weighed prior to a decision.

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6

Case Study B: Report of the Panel on Industrial Wastes

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6.1 INTRODUCTION

The 106 Mile Ocean Waste Disposal Site (Dumpsite 106) and the New York Bight Acid Waste Disposal Site (Acid Waste Dumpsite) are two of the four industrial disposal sites currently in use in the marine environment under the jurisdiction of the United States. The other two are the Fish Cannery Waste Site in American Samoa and the Waste Incineration Site in the Gulf of Mexico. Located 106 nautical miles southeast of New York Harbor, Dumpsite 106 in recent years has received primarily acid-iron wastewater left over from the manufacture of titanium dioxide and alkaline sodium sulfate wastewater. Other organic and inorganic residues have also been released at the site. The Acid Waste Dumpsite has received primarily acid-iron wastewater and a smaller amount of hydrochloric acid wastewater.

The Marine Protection, Research and Sanctuaries Act of 1972 requires an evaluation of land-based alternatives to ocean dumping before a dumping permit can be issued. Studies carried out over a period of several years of studying on the effects of dumping at the two sites provide a basis for comparing ocean disposal with other waste management alternatives. This section addresses the scientific, engineering, and social issues involved in a multimedia assessment and identifies the information necessary for such an assessment.

The first step in a multimedia assessment is to develop a preliminary list of alternatives for a specific waste and then, through standard screening techniques, to reduce the list to more manageable size before making more detailed engineering and cost analyses. Environmental criteria are applied during this first step to eliminate alternatives that would clearly contravene existing regulatory standards or that would be clearly unacceptable environmentally. Also important in narrowing the list is technical feasibility. Incineration of hydrochloric acid, for example, would make no sense technically because the end product would be hydrochloric acid. At this stage revisions of the waste-generating processes that would reduce waste volume or change its character should be considered. Pretreatment of waste streams should also be evaluated.

The results of this first step will be the identification of perhaps two to five waste-management options, each of which can be characterized in terms of waste volume and character, capital and operating costs, needed equipment, flow sheets and material balances, construction materials, and other data that are required for an environmental assessment.

The second step is to compare these alternatives to determine their potential effects on the environment and on the general welfare. This approach is useful for any waste and for any location. Because the information available for assessment will range from qualitative to semiquantitative to quantitative, a matrix approach is best suited to reducing this multitude of inputs to a common base that can be used by decision makers to assess alternatives. As experts in each medium consider the environmental effects of each alternative, they will generate more detailed needs for information and testing, as is illustrated in the other chapters of this report.

The matrix discussed here (1) identifies environmental and institutional variables of concern, i.e., air, surface water, groundwater, land, ocean, and community effects; (2) rates the variables in terms of relative importance among alternatives; (3) provides ratings of the short-and long-term effects of the alternative being considered on these variables; and (4) results in a composite numerical rating of the long- and short-term effects of each alternative.

The proposed matrix is by no means the only one possible, but it is based on years of experience at Dumpsite 106 and the Acid Waste Dumpsite and is believed to be balanced. It lends itself to elaboration at any level of detail. It also lends itself to refinement by integrating evaluations of short- and long-term environmental impacts into a single predictive statistic and by integrating institutional and environmental factors. This matrix is similar to one that was developed to compare the environmental impacts of ocean disposal with those related to alternative disposal practices and submitted in conjunction with an Ocean Dumping Permit Application to the Environmental Protection Agency (Energy Resources Co., Inc., 1981).

The case study of the Acid Waste Dumpsite presented later in this chapter evaluates ocean disposal of acid-iron wastes and an alternative method under which the wastes would be neutralized and then placed in a landfill. This is a simplified case that demonstrates how the assessment system can be used; it is derived from numerous actual studies and experience.

The third step in a multimedia assessment is decision making. This is the point at which the results of the impact assessment, the capital and operating costs, and the policy and strategy considerations of the owner of the waste confront the regulatory and political processes. The matrix approach helps to ensure that all possible media and processes are considered in determining how to manage a particular waste.

The proposed approach is sometimes subjective and sometimes objective. It approximates natural decisionmaking processes but is more structured to facilitate decision making. The analysis may be simple or detailed, depending on the complexity of the problem, the wishes of decision makers, and the availability of data.

6.2 CONSIDERATION OF ALTERNATIVES

Decisions on how to deal with wastes first require identification of the available options. In addition to various disposal techniques, those options may include changes in processes, recycling of wastes, changes in raw materials, the separation or relocation of manufacturing steps, and others (see Table 6.1). The use of alternatives is likely to result in wastes of differing quantity with different physical, chemical, and biological characteristics. These wastes may be gaseous, liquid, or solid in varying proportions.

TABLE 6.1 Examples of Waste-Management Alternatives

- Revise process to reduce waste volumes or change waste character
- Change raw materials
- Sell, use, or recycle wastes
- Pretreat wastes or intermediate streams
- Apply wastes to land for treatment
- Send wastes to landfill
- Dispose of in ocean
- Incincerate
- Inject into a deep well
- Store
- Develop other land-based alternatives
- Shut down generating process

6.2.1 Screening of Alternatives

Once all the alternatives are identified, some type of screening process is necessary. The alternatives are normally screened to determine:

- Technical feasibility
- Economic reasonableness
- Environmental acceptability

An evaluation of technical feasibility involves determining whether a particular alternative can be accomplished. Deep-well disposal of a waste, for example, might be quickly eliminated as an option if no geologic formations that could accept wastes in the quantities anticipated were known to exist.

An examination of economic reasonableness might show that certain options would be prohibitively expensive because of the effects of utilizing such options on the price of a product. An example would be incineration of a waste. Although incineration might be technically feasible, a preliminary evaluation might show that the additional cost of the fuel needed for incineration would make a particular product uncompetitive in the marketplace.

Some options might be shown, with relatively little study, to be environmentally unacceptable. An example would be incineration that caused an unacceptable level of emissions into an air-quality nonattainment area. Furthermore, the cost of assuring that emission levels are in compliance with air-quality rules could make the option fail the test of economic reasonableness.

The initial screening process provides a list of possible alternatives. Whether the list is long or short, each item must be technically feasible, economically reasonable, and environmentally acceptable. This process should not preclude reconsideration of rejected options. Often, further discussion or the development of new information will show that these options were rejected prematurely.

6.2.2 Detailed Evaluation

The alternatives identified during the initial screening process need to be examined in greater detail to confirm that they actually can be accomplished. This closer examination may involve such things as laboratory studies, construction and operation of pilot plants, identification of alternate raw materials, test drilling of geologic formations, or field studies on the effects of ocean dumping.

The detailed evaluation should specify energy requirements, services requirements, sources of raw material, auxiliary manufacturing of intermediate products, and other needs. It is at this point that air emissions, wastewater effluents, and solid residuals are characterized and quantified. Resultant changes in the air, surface water, groundwater, ocean, and land should also be identified.

This detailed evaluation will narrow the list of alternatives. Capital and operating costs can then be addressed.

6.2.3 Cost Estimation for Alternatives

The desired degree of accuracy of a cost estimate will influence the amount of effort involved in making the estimate. For example, to obtain an estimate accurate to within 10 percent of the actual cost may involve identification of a specific location; field topographic surveys; partial design of processing equipment; price quotations for major pieces of equipment; forecasts of inflation and interest rates; and quantification of management, operating, and service personnel. The sources and costs of required raw materials, utility services, and transportation should be specified. The salability of by-products will need to be identified, or, if such by-products must be disposed of, the costs of such disposal must be specified.

Various secondary problems should be addressed as well, such as a need for increased production of raw materials. An instance of this would be the need to produce more lime to neutralize acidic wastewater. This might entail additional mining operations and consumption of larger amounts of the fuel used in calcining the limestone.

6.2.4 Final Listing of Alternatives

The final alternatives, typically two to five, will include those that are considered technically and economically feasible and environmentally acceptable. Of critical importance, too, is whether each alternative can be implemented within a required time. The time constraints will be determined by marketing strategies, by statute, or by management or political edict.

If the final alternatives are found to be equally desirable or satisfactory, an environmental impact assessment can expedite the process of deciding which one to adopt. Table 6.2 lists the types of information required for environmental and institutional assessments.

6.3 IMPACT ASSESSMENT

In this section, the panel proposes a matrix for comparing final disposal alternatives with respect to their impacts on air, water, land, ocean, institutional, and community variables. The system is illustrated by an example from the Acid Waste Dumpsite.

Waste-management alternatives that still appear feasible after the initial evaluation can be compared by using a matrix approach. By using a matrix it is possible to make a comparison on the basis of composite point totals that indicate short- and long-term impacts. The higher the point total, the greater the impact of the alternative. Table 6.3 shows the matrix that was developed. There are five environmental media to be considered (air, surface water, groundwater, land, and ocean), and there is also a set of institutional considerations (the effect of each disposal option on community attitudes, services, economy, and safety). Each environmental medium has four areas of concern--human health, human welfare, biota, property--with each area of concern rated on a scale of 1 to 3, with 3 signifying greatest relative importance. Since human health is the most important environmental consideration, it is assigned a 3. Human welfare, a category that includes such factors as aesthetics and recreational value, and biota are generally assigned an intermediate rating of 2. Property (land, buildings, roads, for example) is assigned a relative importance of 1.

TABLE 6.2 Information Needed for Environmental and Institutional Assessments of a Waste-Management Alternative

- Description of the waste-management alternative
- Process flow sheets
- Material balances
- Detailed design
- Equipment lists
- Materials of construction
- Capital costs
- Operating costs
- Manpower requirements
- Transportation needs
- Waste streams--character, quantity
- Time needed to install
- Energy requirements
- Definition of major secondary factors (energy, raw material, waste, or new product)
- Market information if sale of a by product or new product is involved
- Key economic ratios (e.g., return on assets and discounted cash flow)
- Technical feasibility--what are the odds that the alternative will work?
- Location of generating process
- Waste disposal locations
- Character of waste disposal location (e.g., geology, hydrology, climatology, topography)

The relative impacts of each waste disposal option are then rated in terms of their short-term and long-term effects. Short-term effects are those that are evident shortly after an alternative is implemented; they are normally reversible. Long-term effects are those that persist far into the future and may be irreversible. Separate ratings are assigned for short- and long-term effects. These ratings are assigned on a −5 to +5 scale where 0 signifies no effect, +5 signifies a serious adverse impact, and −5 signifies a very positive benefit. These impacts can best be determined by experts who have studied cause-and-effect relationships and can properly quantify impacts. If the impact of a waste disposal option is so severe as to be environmentally unacceptable, the option probably will have been rejected during the screening phase. But it is also possible that new information indicating environmental unacceptability will only come to light during the assessment.

Environmental evaluation factors (EEF) for each area are then derived by multiplying the relative importance score by the relative impact score. The EEFs are summed to derive separate scores for environmental/short-term, environmental/long-term, institutional/ short-term, and institutional/long-term impacts.

The matrix approach offers flexibility in choosing among waste disposal alternatives, but balance in the matrix must be maintained. The relative emphases in the subcategories should be similar, and no medium should be weighted much more heavily than another. Indeed, an alternative that impacts substantially on more than one medium could be difficult to manage in an environmentally acceptable manner. Institutional considerations should also be weighted more or less equally.

The matrix is a semiquantitative tool based on information of varying degrees of accuracy. Thus, small differences in summary scores probably will not be important.

6.3.1 Environmental Impacts

The impact of a waste disposal option on the air would consist of changes in air quality affecting human health, human welfare, biota, and property. Human-health impacts would be the impacts resulting from excessive concentrations of sulfur dioxide or carbon monoxide. Human welfare impacts would be such things as decreased

TABLE 6.3 Impact Assessment Matrix

		Relative Importance	Impacts (−5 to +5)			
			Short Term		Long Term	
	Medium and Areas of Concern	Scale 1-3	Mag. [a]	EEF [b]	Mag.	EEF
A.	Environmental Considerations					
1.	AIR					
	a. Human:					
	health	3				
	welfare (e.g., aesthetics)	2				
	b. Biota	2				
	c. Property	1				
2.	SURFACE WATER					
	a. Human:					
	health	3				
	welfare	2				
	b. Biota	2				
	c. Property	1				
3.	GROUNDWATER					
	a. Human:					
	health	3				
	welfare	1				
	b. Biota	1				
	c. Property	1				
4.	LAND					
	a. Human:					
	health	3				
	welfare	2				
	b. Biota	1				
	c. Property	1				
5.	OCEAN					
	a. Human:					
	health	3				
	welfare	2				
	b. Biota (benthos, plankton and fish)	2				
	c. Property	1				
	TOTALS					

B. Institutional Considerations	
Effects on Community	
a. Attitudes	2
b. Services	1
c. Economy	2
d. Safety	3
TOTALS	

[a] Mag.: Magnitude of impact on each environmental and institutional resource considered is estimated on a scale of −5 to +5 with negative numbers indicating beneficial impacts (−5 = greatest beneficial impact) and positive numbers denoting harmful impacts (+5 = greatest harmful impact).

[b] EEF: Environmental evaluation factor = (relative importance of environmental variable) x (magnitude of impact).

visibility and odors. Impacts on biota would be such things as damage to fish populations from acid rain or damage to vegetation from air pollutants. Impact on property might consist of damage to buildings caused by acid rain.

Similarly, a waste disposal option could have impacts on surface waters. A human welfare impact on surface water might be loss of recreational fishing in a lake because of pollution. Damage to a freshwater aquatic community because of a change in water quality would be an impact on the biota. An impact on property would be damage to structures in the water, such as boats or docks. A human-health impact would be contamination of a surface-water source of drinking water.

In the groundwater medium, the major human-health concern would be the protection of drinking-water supplies. An impact on human welfare would be the rendering of groundwater less usable for agricultural irrigation. Although neither biota nor structures are normally present in groundwater, biota and property are included in the matrix to take account of the impact of groundwater pollution on aquatic species in underground waters and on structures elsewhere.

With respect to land, a direct impact on human health would be caused by contact with contaminated soil; an indirect impact would be human consumption of contaminated crops. Human welfare, such as aesthetic or recreational opportunities, could be impacted by the use of large areas of land for waste disposal. The impact would be short term or long term, depending on whether the land could be returned to other use after disposal was completed. A waste disposal option could have an impact on land biota, such as vegetation or animals, if the waste was placed in a vegetated area. An impact on property would be the razing of buildings to make space for a power plant's waste disposal.

Ocean disposal considerations are discussed in greater detail in Section 6.5. Ocean disposal might affect human health if fish contaminated at disposal sites were consumed as food, although the possibility of this appears to be slight. An impact on human welfare would be perceived loss of recreational and aesthetic values, such as beach damage caused by ocean dumping. Damage to the biota could be short term (reversible) and long term (irreversible). The impact on property would again be confined to physical structures, such as boats and docks affected by the physical and chemical changes in ocean water.

6.3.2 Institutional Impacts

The panel has identified four institutional considerations: community attitudes, services, economy, and safety. Their relative importance ranges from 3 for safety to 1 for services. The magnitude of the impact of waste disposal options on these institutional considerations is rated on a scale of −5 to +5.

Attitudes relate to the acceptability of the alternative to the local community and to the general public. Services are the new demands on roads, sewers, utilities, and other public facilities resulting from selection of a disposal alternative. Economic considerations include employment, taxes, commercial activity, property values, and other economic factors in the local area that would be affected by a disposal option.

Safety refers to all aspects of public safety, e.g., transportation or storage of hazardous materials in or near inhabited areas.

6.4 DECISION MAKING

The matrix approach makes it possible to weigh the environmental and institutional effects of waste disposal alternatives. Economic considerations clearly play an important role. For a manufacturing process, the economic question is whether the alternatives are affordable. That is, what impact will the alternatives have on, for instance, product profitability, competitiveness in domestic and foreign markets, alternate use of funds, and manufacturing processes? Similar questions arise in the case of municipal waste disposal options. Can the local community afford the options (taxes and user fees)? How will the options affect the financing of other needs of the community (schools, roads, parks)?

A significant benefit of the matrix approach is that a relatively objective view can be taken of the environmental and institutional impacts of waste disposal alternatives. Economic, as well as policy or legal factors, can then be applied to reach a decision. The decision, for example, may be to adopt a plan of action that is less than totally satisfactory with respect to environmental protection but so much less costly that some adverse environmental effects can be tolerated. Conversely, a less costly alternative may have too high an environmental impact on one or more media.

The matrix approach to decision making is useful both to those who will be required to pay for an option and to those whose approval is needed. This includes not only those from whom permits must be obtained but also public groups that have a direct or indirect interest in a project.

Fundamentally, decision making is a series of compromises among factors that pull the decision maker in a variety of directions. Use of the matrix approach can assist the decision maker in selecting a disposal option that best responds to the needs of society.

6.5 CASE STUDY

6.5.1 Background

Alternate methods of managing wastes from a sulfate process titanium dioxide manufacturing plant in New Jersey are examined in this section.

The sulfate process used in the plant yields waste streams containing sulfuric acid (approximately 8 percent), ferrous sulfate (approximately 10 percent) , small quantities of other metal sulfates, and an inert solid mineral gangue (Goldberg, 1979).

Until 1948, acid-iron wastes from the plant were discharged into the Raritan River. Since then, however, these wastes have been dumped at the Acid Waste Dumpsite. Before dumping started, environmental studies were conducted at the dumpsite by the Woods Hole Oceanographic Institution and others. Although much of the information discussed here was obtained in recent years, the chronological sequence of the studies and their details differ. Thus, some of the data used here have been synthesized from existing studies.

Alternative methods for managing the wastes from the plant were examined by use of the techniques described in Section 6.2, and these alternatives were screened out: waste acid concentration, deep well disposal, ammonium neutralization, ore beneficiation, and a change from the sulfate process to the chloride process. Two alternatives remained: (1) neutralization of the acid-iron waste with limestone, disposal of the resulting solid waste on land, and release of treated effluent to the river or (2) disposal in the ocean (the existing practice).

6.5.1.1 Neutralization and Land Disposal (Table 6.4)

The neutralization and land disposal alternative would consist of neutralization of the acid in the waste with ground limestone, followed by separation of the produced gypsum and metal hydroxide, which would then be disposed of on land. Approximately 3,100 tons of 80 percent solids would be generated per day, 365 days per year.

At an increase in cost, the neutralization process can be modified to produce high-grade gypsum that would be suitable for wallboard manufacture. A study conducted by Battelle Memorial Institute determined that the gypsum could be used in cement manufacturing, provided the material was pelletized. Despite successful pilot runs (in a Canadian plant), however, no local producer of wallboard was interested in using the synthetic gypsum. The cement industry would, at best, consume less than 10 percent of the gypsum.

Because prospects for the use of the gypsum were so small, a study of the land disposal requirements was then made. Three potential land disposal sites totaling 300 acres were identified. These sites could be used for 12 years, assuming the wastes were deposited to a height of 24 feet. Thereafter, additional sites at greater distances from the point of generation would be needed. The wastes would amount to approximately 145 truckloads per day; that is, a truck would be entering or leaving the plant every 5 minutes, 24 hours per day, 365 days per year. That would mean a significant increase in truck traffic and thus a significant increase in energy consumption, air pollution, and noise.

An engineering study of the neutralization process produced a detailed estimate of capital, operating, and maintenance costs and of engineering parameters (material balances, manpower, material usage, for example).

The impact ratings of the neutralization process were judged to range from 0 to +5 (on a scale of −5 to +5).

This option was found to have a small but quantifiable effect on the ambient air as a result of the increase in truck engine exhaust gases and particulates that would occur. Therefore, a value of 1 was assigned to this alternative's impact on human health. Since no irrever

TABLE 6.4 Impact Assessment Matrix for Neutralization and Land Disposal Alternative

		Relative Importance	Impacts (−5 to +5)			
			Short Term		Long Term	
	Medium and Areas of Concern	Scale 1-3	Mag. [a]	EEF [b]	Mag.	EEF
A.	Environmental Considerations					
1.	AIR					
	a. Human:					
	health	3	1	3	0	0
	welfare (e.g., aesthetics, odor, noise)	2	1	2	0	0
	b. Biota	2	0	0	0	0
	c. Property	1	0	0	0	0
2.	SURFACE WATER					
	a. Human:					
	health	3	0	0	0	0
	welfare	2	0	0	0	0
	b. Biota	2	1	2	0	0
	c. Property	1	0	0	0	0
3.	GROUNDWATER					
	a. Human:					
	health	3	0	0	1	3
	welfare	1	0	0	1	1
	b. Biota	1	0	0	1	1
	c. Property	1	0	0	0	0
4.	LAND					
	a. Human:					
	health	3	0	0	0	0
	welfare	2	4	8	4	8
	b. Biota	1	1	1	0	0
	c. Property	1	1	1	1	1
5.	OCEAN					
	a. Human:					
	health	3	0	0	0	0
	welfare	2	0	0	0	0
	b. Biota (benthos plankton and fish)	2	0	0	0	0
	c. Property	1	0	0	0	0
	TOTALS			17		14

Institutional Considerations					
Effects on Community					
a.Attitudes	2	4	8	2	4
b. Services	1	2	2	2	2
c. Economy	2	1	2	2	4
d. Safety	3	2	6	2	6
TOTALS			18		16

[a] Mag.: Magnitude of impact on each environmental and institutional resource considered is estimated on a scale of −5 to +5 with negative numbers indicating beneficial impacts (−5 = greatest beneficial impact) and positive numbers denoting harmful impacts (+5 = greatest harmful impact).
[b] EEF: Environmental evaluation factor = (relative importance of environmental variable) x (magnitude of impact).

sible effect was anticipated, the long-term impact was estimated at 0. Human welfare was judged to be affected by truck noise and was rated as 1.

The option was expected to have a modest short-term effect on biota as a result of changes in surface-water quality, but no long-term effects were anticipated. No short-term effects on groundwater were anticipated. As with all landfill operations, however, the possibility of long-term effects could not be ignored. Hence, potential long-term effects on human health and welfare and on the biota were each rated at 1.

Obviously, the medium that would sustain greatest impact would be the land itself. Although there would be no effect on human health, the impact on human welfare (e.g., aesthetics and recreation) would be considerable on both a short-term and long-term basis. The effects on biota were short-term and confined to the three sites and their immediate surroundings. By definition, effects on real property were confined to the value of the sites themselves.

Adoption of this alternative would have no effect on the ocean.

We now turn to institutional considerations. The attitude of the community, in this case, the local community, would be very negative in the short term and less so in the long term. A high rating of 4 was assigned to short-term impacts, with long-term impacts judged at 2.

Community services would be significantly impacted by the 700,000 truck-miles traveled per year; road maintenance and traffic control would be two major aspects of concern.

The economy of the community would be affected in a positive way by the creation of jobs in the neutralization plant and the disposal operation. However, property values in the vicinity of the disposal sites would be adversely affected, and the cost of maintenance along the traffic routes would increase. Not only streets and bridges but also homes and commercial establishments would be affected. The net result is a rating of 1.

The most significant institutional impact of this alternative would be in the area of public safety. Although no attempt was made to quantify this impact, it would be possible to forecast the number of accidents on the basis of the truck-miles traveled per year. A rating of 2 was assigned.

6.5.1.2 Ocean Disposal (Table 6.5)

Ocean disposal of the wastes from the New Jersey plant did not change the quality of the air, surface water, groundwater, or land. Since there were no impacts in these media, there were no ensuing effects on health, welfare, biota, or property.

The dumping of the wastes in the ocean has had environmental effects on that medium, however. As a result of analysis of the approximately 50 studies of the effects of acid-iron waste on the ocean ecosystem, a rating of 1 was assigned to both short-term and long-term effects on the marine biota. This is a rating of the effects on small numbers of plankton and fish that are exposed to the waste within minutes of its discharge. There is no effect on human health. The effect on human welfare is the impact on aesthetics and recreational use of the dumpsite, and the short-term effect was assigned a value of 1. Since this is reversible, there is no long-term effect.

With respect to institutional considerations, the principal impact of ocean dumping is on community attitudes. Notwithstanding the results of scientific studies showing that the impact is small, the inhabitants of the New Jersey shore communities are unquestionably opposed to ocean dumping. The attitude in the communities from which the plant draws its workers is more benign.

Public safety was given a rating of 1. Although there has not been an accident involving ships in the 34 years of ocean disposal, there have been collisions with bridge structures and the potential of shipping accidents does exist.

6.5.1.3 Conclusion

Assessment of environmental considerations produces EEF sums of 17 (short term) and 14 (long term) for the neutralization and landfill option (Table 6.4). EEF sums for ocean disposal (Table 6.5) are 4 (short-term) and 0 (long-term).

Assessment of institutional considerations also produces ratings for ocean dumping that are more favorable than those for neutralization. However, the differences are not so great as in the environmental assessment. The EEF sums for neutralization are 18 (short-term) and 16 (long-term). For ocean disposal, the short- and long-term EEF sums are both 11.

TABLE 6.5 Impact Assessment Matrix for the Ocean Disposal Alternative

		Relative Importance	Impacts (−5 to +5)			
			Short Term		Long Term	
	Medium and Areas of Concern	Scale 1-3	Mag. [a]	EEF [b]	Mag.	EEF
A.	Environmental Considerations					
1.	AIR					
	a. Human:					
	health	3	0	0	0	0
	welfare (e.g., aesthetics)	2	0	0	0	0
	b. Biota	2	0	0	0	0
	c. Property	1	0	0	0	0
2.	SURFACE WATER					
	a. Human:					
	health	3	0	0	0	0
	welfare	2	0	0	0	0
	b. Biota	2	0	0	0	0
	c. Property	1	0	0	0	0
3.	GROUNDWATER					
	a. Human:					
	health	3	0	0	0	0
	welfare	1	0	0	0	0
	b. Biota	1	0	0	0	0
	c. Property	1	0	0	0	0
4.	LAND					
	a. Human:					
	health	3	0	0	0	0
	welfare	2	0	0	0	0
	b. Biota	1	0	0	0	0
	c. Property	1	0	0	0	0
5.	OCEAN					
	a. Human:					
	health	3	0	0	0	0
	welfare	2	1	2	0	0
	b. Biota (benthos and fish)	2	1	2	0	0
	c. Property	1	0	0	0	0
	TOTALS		4		0	

B.	Institutional Considerations					
	Effects on Community					
	a. Attitudes	2	4	8	4	8
	b. Services	1	0	0	0	0
	c. Economy	2	0	0	0	0
	d. Safety	3	1	3	1	3
	TOTALS			11		11

[a] Mag.: Magnitude of impact on each environmental and institutional resource considered is estimated on a scale of −5 to +5 with negative numbers indicating beneficial impacts (−5 = greatest beneficial impact) and positive numbers denoting harmful impacts (+5 = greatest harmful impact).
[b] EEF: Environmental evaluation factor = (relative importance of environmental variable) x (magnitude of impact).

Thus, the matrix shows that ocean disposal has a less unfavorable impact on the environment and society than does the alternative of neutralization and land disposal.

Economic analysis of the two alternatives provided striking results. The use of neutralization and land disposal would require approximately $48 million in capital expenditures and increase the cost of titanium dioxide by 10 to 15 percent. However, the capital cost for the ocean disposal alternative is approximately 50 to 60 percent of the land-based option. It is estimated that ocean disposal of wastes accounts for 3 to 4 percent of the product cost.

6.6 CONCLUSIONS AND INFORMATION NEEDS

Following are conclusions of the Panel on Industrial Wastes:

1. All feasible alternatives and the resulting impacts in all media should be considered before a decision on a method of managing wastes is made.
2. An orderly and flexible procedure is needed for comparing options on a common basis and handling masses of information of varying quality.
3. A matrix is such a procedure to compare the short-and long-term effects of different disposal methods on air, surface water, groundwater, land, and ocean and on institutional considerations.
4. Further refinement of the matrix is needed to (1) determine whether the variables being used are the proper ones; (2) establish the relative importance of the variables, possibly by using a more sensitive scale; (3) decide how the resulting scores should be integrated (short-term effects with long-term effects, institutional effects with environmental effects); and (4) decide to what extent secondary impacts should be rated.
5. Further work by experts is needed to develop the factors that should be considered in rating impacts. Such improved factors would reduce uncertainty in assigning environmental impact values on both a short- and long-term basis.
6. Because the matrix approach is flexible, its use may lead to excessive study. It is therefore important to use experienced people who are able to determine the point beyond which further study is unproductive.

7. Additional research on the impacts of waste disposal at Dumpsite 106 and the Acid Waste Dumpsite probably would not significantly affect the relative ratings of the ocean disposal option in comparison with other options.
8. The matrix approach can be used within the present legal/regulatory framework. It is an effective guide in decision making.

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