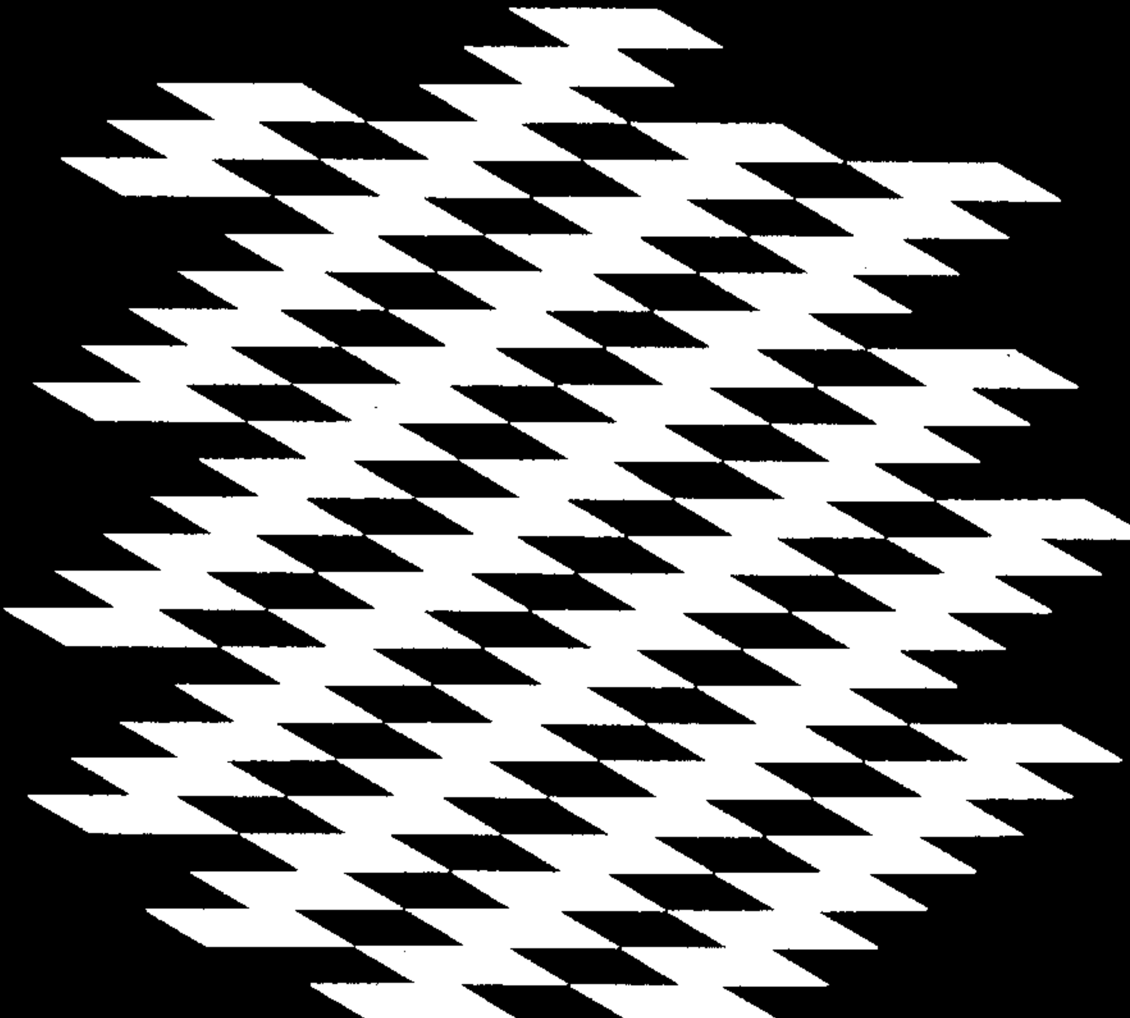
Public Waste Management and the Ocean Choice

Edited by
Keith D. Stolzenbach
Judith T. Kildow
Elizabeth T. Harding



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MIT Sea Grant College Program
Cambridge, Massachusetts 02139

MITSG 85-36
April 1986

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Public Waste Management and the Ocean Choice is the proceedings for the 1985 MIT Sea Grant College Program Lecture and Seminar, "Ocean Disposal of Public Wastes: Technology and Policy for the Future."

Copies of this book are available through
Sea Grant Information Center
Sea Grant College Program
Massachusetts Institute of Technology
77 Massachusetts Avenue E38-306
Cambridge, Massachusetts 02139

Cataloging in Publication Data
Public Waste Management and the Ocean Choice
1 Waste disposal in the ocean – Congresses
2 Sewage disposal in the ocean – Congresses
3 Marine pollution – measurement – Congresses

Foreword

In the late 1960s the United States and other industrialized nations became alarmed at the visible damage to air, land, and water quality. By the 1970s complex and often highly restrictive regulations had been implemented to govern many activities including the use of the oceans for public waste disposal. Although extensive documentation indicated that the oceans could assimilate large volumes of sewage, sludge, and dredge spoils, there was also evidence that in a few cases environmental thresholds had been reached. The public demanded greater assurance that the oceans and coastal waters would not be seriously degraded by future waste disposal.

This volume examines these issues and places ocean disposal within the context of other waste management options. Theory is blended with practice drawing upon scientists and public and private sector interests. Experience and planning in Philadelphia, Chicago, and New York illustrate the technical, economic and institutional aspects that communities face in disposing of their wastes. Throughout, there is a common theme – ocean disposal today is as safe as the other options, *but* the removal of persistent toxics at the source is *essential* to long-term safe waste disposal – whether on land, in the air, or at sea.

In considering the ocean option, the authors pose many questions: Are there long-term scientific issues not being considered? How can we improve environmental monitoring? Why are we so slow in adapting innovative technologies for waste management? Who should operate waste management systems – the public sector, private enterprise, or a combination of both? How can public intervention be effective in the waste management process? Can we continue to manage public wastes by balancing economic pressures with environmental integrity? And finally, is global survival at stake if society does not shift to a chemically tight system?

Keith D. Stolzenbach
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Elizabeth T. Harding

Acknowledgments

The papers in this volume were presented at a symposium, "Ocean Disposal of Public Wastes: Technology and Policy for the Future," in April, 1985 at the MIT Sea Grant College Program Lecture and Seminar Series. Within MIT, support for the Lecture/Seminar was generously provided by a permanent endowment from the Henry L. and Grace Doherty Foundation, the Sea Grant College Program, the Department of Civil Engineering, the Department of Ocean Engineering, and the Industrial Liaison Program. Generous industry support for this major symposium came from Camp Dresser and McKee, Inc. in Boston, and ENDECO, Inc. in Marion, Massachusetts.

The organization of the conference and the production of these papers would not have been possible without the considerable help and efficiency of the MIT Sea Grant Program support staff, particularly Ms. Re Quinn and Therese Henderson. C. Chryssostomidis, Director of Sea Grant, was a thoughtful critic and contributor as the program and planning for the Lecture/Seminar evolved.

An important role was also played by a Technical Advisory Committee of people chosen for their dedication to the prudent use of ocean resources. They include: Judith Capuzzo, Woods Hole Oceanographic Institution; Michael Connor, US Environmental Protection Agency, Boston; Harriet Diamond, Massachusetts Office of Coastal Zone Management, Boston; Robert P. Eganhouse, University of Massachusetts, Boston; Merton Ingham, National Marine Fisheries Service, Woods Hole; and Richard Kolf, Office of Sea Grant, NOAA, Washington, DC.

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Thomas Jorling, an environmental scientist and lawyer, has had a career that included university research and teaching as well as government service in Washington. In 1962 he received a B.S. in biology from the University of Notre Dame, in 1966 an L.L.B. from Boston College and in 1969 an M.S. in Ecology from Washington State University. From 1972-1977, he helped to establish the Center for Environmental Studies at Williams College. Now a Professor at the Center, he served as its Director from 1979-1982. In 1977 he was appointed by President Carter to be the Assistant Administrator for Water and Hazardous Materials in the US Environmental Protection Agency. In that position, and earlier as Minority Counsel for the US Senate Committee on Public Works, 1969-1972, he was responsible for developing and implementing the Clean Water Act, the Ocean Dumping Act, the Safe Drinking Water Act and the Resource Conservation and Recovery Act. As Assistant Administrator he helped guide the distribution of $200 million in grants to states for program administration and $4.5 billion in water and sewer construction grant assistance to states and municipalities. He is a member of the Bar in the District of Columbia and the Commonwealth of Massachusetts and is on the Board of Directors of the Natural Resources Defense Fund. On April 17, 1985, he delivered the 13th Annual Sea Grant Lecture as reprinted in this volume.

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Two Perspectives on Waste Management

The Future of Public Waste Disposal

Thomas C. Jorling
Williams College

I am deeply honored by the invitation to deliver this MIT Sea Grant lecture. It is especially pleasing because this lecture is the initial activity in the effort of this conference to look to the future, specifically the future of ocean management, a subject in which I am, no pun intended, deeply interested.

Some may be surprised that a member of a faculty at a small college in the Berkshire mountains of western New England would be giving a Sea Grant lecture. Let me remind you that only a few miles away from Williamstown stands the house where Herman Melville wrote the classic Moby Dick. Perhaps those of us somewhat removed from proximity to the ocean don't take it so much for granted.

I am by nature an optimist. And part of my interest in the oceans arises from an intuitive sense that the oceans, including Antarctica, can represent the subject, and the object, of activities that can begin first, the reconciliation of dangerous differences within the community of mankind, and second, the harmonization of mankind with the life-supporting biosphere. I warned you I am an optimist.

The oceans have always had a powerful influence on the value systems of man. Whether one looks to mythology, to the Bible, or to literature; the ocean and the life it harbors stand out prominently as symbol, as metaphor, and as inspiration.

There is a universal interest in the oceans; an interest that may have deep biological origins. Certainly, it has roots in the power, if not the mystery of the sea and its creatures. Whatever, the sources of this interest, and I defer in such exposition to those with greater analytic and literary talent than I, it provides an opportunity; nay, a range of opportunities that we, especially those of us who share a concern about the oceans, should foster and nurture. Whether it be to influence in a positive direction superpower relations, as for instance, in the form of cooperative research and management agreements on Antarctica or treaties on fisheries; or to mediate and accommodate Third World interests, as in the Law of the Sea; or to regulate the management of materials and waste in the domestic setting; we should not lose sight of the grip the oceans have on the human imagination.

And I just don't "feel" optimism; it is not blind faith. I see solid evidence all around. I see it in the Japanese agreement to terminate whaling announced a few weeks ago. I see it in the continuing efforts and some successes, regarding Antarctica and its surrounding Southern Ocean. And, notwithstanding the current U.S. position, I see in the Law of the Sea Treaty a level of cooperation and understanding in the global community that breeds hope and confidence. Walter Orr Roberts' recent proposal for a Research Park on both sides of the Bering Sea is a bold proposal,[1] not just to manage properly an important geographical area, it can be a powerful step on the path to a new set of relationships between the two most powerful and contending factions of mankind. And new relations are essential if man is to have a sustainable, peaceful future on this mostly liquid planet.

But this vision of optimism regarding the role of the oceans cannot obscure the fact that there are immense difficulties in mankind's relation to the biosphere now, and surely there will be difficulties into the future. One of the most significant aspects of this relationship is the subject of this conference; the generation and management of public waste, or put more accurately, materials which by intention, design, or accident escape man's control and are released into the environment.

The focus of this conference is the future management of public wastes. The conference organizers should be complimented for not only have they recognized that the controversies of today need a new context; they have at the same time afforded us a theme that enables us to see, and understand how the present relates to, if not binds the future. We can imagine how visions of the future might enable us to begin to make choices between where we are, and different visions of what we might become. When viewed in this light, the present doesn't necessarily bind to more of the same; considering the future can liberate us to see different possibilities and different methods; different ends and different means. But, it must be responsible. Our starting point is the present, it is where we are and where we have been. The difference between science fiction and speculating about future public policy is the recognition that at the same time we look to the future, we must start from the present. The two must be connected by honest statements of assumptions, principles, and values, as well as honest attempts to suggest how -- not just why -- we should get from here to there. And this is what I propose to do in context of the management of public waste.

At the outset we must look beyond the current dynamics of domestic society, we must look to the global dynamics. It is at once trite and profound to acknowledge that all mankind is now linked together, as often as not by the oceans. But there are other links; the movement of materials and products, the communication of information, values, and expectations; and the convergence of the patterns, linkages, and forces of economic development.

Over the past several decades, monitoring and assessing population growth has become, at least in a relative sense, quite accurate. The predictions and accompanying assumptions of a decade ago are remarkably close to conditions actually realized. And, if we are to talk about any aspect of the human future, we must start with some sense of how many humans there are likely to be. I'll use the population information of the World Bank's 1984 Annual Report,[2] not as a prediction, but as a likely context; a device to assist our imaginations in seeing the future.

In 1982, the world's population was about 4.57 billion, up from about 2 billion in the year of my birth, 1940. The World Bank's annual report now includes a projection to a stationary population, a population that under the model's assumptions -- analyzed and projected country by country -- is reached sometime in the second half of the next or 21st century. The projection gets there in stages. In 1990, the global population would be 5,050 billion; at the turn of the century, 6,082; and stable at 11.050 billion. Large numbers indeed, but numbers that represent people, people whose demands on the biosphere will control strongly the amounts and character of material that must be managed. Of the estimated population in 1990 -- a date so close and the population dynamics so structurally "entrained" as to allow considerable confidence in characterizing the number, short of war or other apocalypse, as a prediction; of those 5.2 billion, some 4.613 billion, that is a number greater than today's population, is in nations with GNP per capita currently below $6,000. This compares to $13,000 GNP per capita in the U.S. in 1984. These figures can be interpreted to represent a great deal of, as yet, unsatisfied demand for materials -- and maybe the production of waste. In the year 2000, something on the order of 5.556 billion will be in nations currently below $6,000 per capita.

Urbanization, that is people moving to more densely populated units of geography, often adjacent to coastlines, also seems to be accelerating. In the industrial world something on the order of 70% of the population is urban. The population of the rest of the world appears, at least now, to be proceeding toward a similar pattern of urban settlement and at a rapid rate.

These figures represent, according to one's assumptions, the possibility of tremendous fluxes in the use of materials, of water, of energy, of plants and animals, and of geographical space.

Again, to help us imagine a context and not as an extrapolation, consider the curve of synthetic organic chemical production in the U.S. Using my favorite reference year, 1940, once again, 1.3 billion lbs. were produced. This grew to 49 billion lbs. in 1950; 97 billion in 1960; 233 billion in 1970; nearly 400 billion in 1980.

I introduce the numbers obviously to support a point -- the point being that as we consider what appears to be the open-ended dynamics of the human population in the biosphere, our assumptions, our premises, and our conduct in the use of materials, energy, water and space must change; must change, that is, if our goal is sustainable, just, and free society.

But, there is some good news -- some hard evidence that it is changing, in at least some important elements. Energy demand in the U.S. is my source of optimism here. As recently as a decade and a half ago, official governmental estimates of U.S. energy demand for the year 2000 was 190 quads. Independent analysts, with different assumptions, but no lessening of "quality of life" were projecting 125 quads. By the mid '70s, the government was projecting 140 quads of demand for the year 2000 and the independents were projecting 75 quads -- parenthetically, it might be noted about our current level of demand. In 1980, the government estimates were down to 102 quads and the independents, in the person of Amory Lovins, suggest 15 quads.[3] What is going on here? Tremendous changes in technology for one thing; changes that have a powerful message. Using resources efficiently for another. This pattern of change can become a very bright beacon.

As I look out on human community today, and as I see it becoming in numbers and expectations, I conclude that if we are to match human wisdom and actions to our biospheric foundation then we must become a chemically tight society; that is, the linear pattern of extract, process, produce, and dispose of materials is simply not able to support human populations indefinitely. Nor is it necessary. We can, based on what we now know, begin to develop societies structured and functioned as if there is a tomorrow.

In the context of waste, this means societies in which we conserve and recycle; where we confine and contain disposal. Certainly, such societies would place different demands on the oceans than current societies do, or will likely do, if we assume current trends and policies are merely extended. Unavoidably, in societies that are mere extensions of the present, oceans will be looked upon lovingly as a depository.

But, we must start from where we are, not where we might like to be and we must recognize the important and determinative role of the processes and pathways through which public policy acts. Public policy and public law are interactive processes. Regulation or policies affecting a practice -- say ocean disposal of public waste -- cannot be evaluated solely in terms of the specific practice or adverse effect to which they apply; they are also part, a casual part, of a much larger, general pattern of the generation, production, and use of materials and products, including waste.

Academics and analysts might wish -- even act as though it were otherwise -- but we won't move to a sustainable or any other model society in one grand leap. We can articulate principles and assumptions, strategies and tactics until we are blue in the face, but the actual course of getting from today to tomorrow and the next day -- much less to the future, is a rugged, tough, messy, competitive process. But, in that process if one focuses too narrowly -- say, for instance, on the question is it o.k. to dump this stuff here? -- and doesn't have some overall frame of reference regarding the general policies or principles that should govern the management of material in human society over the long term, then the incremental processes will result in a society, as we would expect would result from a collection of unrelated, random, incremental decisions, that looks like H.G. Wells' classic description of modern society, "everything is driving anyhow to anywhere at a steadily increasing velocity." Surely, it won't be a sustainable society. It is unlikely even to be very pleasant.

Modern man has immense power. The Institution where we gather is a citadel to that fact. The range of choices of technology that is available and will be available in the future is limited only by human imagination. I tell my students never to doubt technology. I say this to convey the lesson that there will be no easy answers of the sort implied by, for instance, the objection that this or that technology "won't work" or "is not safe." We will have no easy path in our choices of technology and there will be choices, if for no other reason than simply because we can't afford to do it all. But, we will need to develop criteria to assist us in making individual, national, and international choices over technology. And our policy now regarding the oceans and the disposal of public waste are very much involved in that process. But, the rhetoric, even the rhetoric of the decision-making processes of today ignores it. And by ignoring it assures that consideration of factors relating to the future are meaningless.

One of the most popular flourishes of rhetoric at the present time, especially from the Administration, including the Administrator, William Ruckelshaus, who recently resigned, deals with risk assessment and management. Risk assessment, in the advocated formulation suggests we can evaluate how much material can be released to the environment at what level of effect and regulate to that level of release. Used this way, it is the determination of an end, not as a means to an end -- it is a process that defines its own end. It suggests a very static view of the world, accompanied by an assumed ability to make judgments about the biosphere and its components that simply does not exist. It is premised upon and assumes the release of chemicals to the environment. And as I have attempted to suggest, if we make such an assumption as the method of managing chemicals, it becomes a self-fulfilling prophecy.

As a general principle describing the dynamics of industrial society that is a flawed assumption, especially when considered from a perspective of hundreds of years. A policy of reacting to releases after they occur, in a society where chemicals are introduced at a rate of several thousand per year and where quantities of production are growing rapidly, is simply irresponsible.

Risk assessment and its semantic antecedent, an ambient control strategy, share many flaws when advocated as a public policy that need to be understood. First of all, it puts the entire burden of proof on the government, on the public and on the environment to support control.

It is not just a burden of proof in a legal, evidentiary sense. It is a burden that is made exceedingly heavy by what it in fact assumes. It assumes an ability to acquire knowledge about the release of chemicals, their transport and fate throughout the biosphere, their effects throughout the myriad of organisms in the biosphere, over time, and an ability to evaluate possible synergistic effects. If we are concerned about the health of persons, then we must in addition incorporate all of the uncertainty on how and why chemicals react, and then effect the structure and function of a human organism, an organism already immersed in an environment with countless chemicals in countless circumstances.

Each of these elements of the burden of proof deserve attention. The ability to acquire knowledge about the release of chemicals into the environment assumes a monitoring and analytical methodology that is not, nor will soon be routinely available. Whether in air, water, or solid waste streams we generally only sample for the most gross parameters. Where a given chemical is targeted, specific methods and protocols must be tailor-made -- always with the burden on the government, constantly scrutinized and contested by the affected interests.

Once released, understanding the pathways by which chemicals move through the biosphere, how they interact and with what effect on organisms is assuming that we can know in detail what many ecological scientists believe is a system -- the biosphere -- that is more complicated than we can think. Yet, if we assume the release of chemicals -- we willingly take on that burden.

Time, and the effect its passage causes as chemicals are released and move through the biosphere is over the edge of uncertainty. We may know after an extensive research project what a given chemical exposed to an organism or community of organisms does in one day or one week or one year, but we know precious little what it may do after ten years. Yet, such an understanding is implicit in carrying the burden of proof in a risk assessment that may lead to the authorizing of the release of certain concentrations of that chemical for many, many years.

Synergistic effects, that is the action of one chemical as it may be influenced by the action of others -- probably hundreds of others -- is currently outside of our understanding.

There is a different principle of management, and I'll be provocative and more simplistic than need be, to make the point and that is a principle of "do not release."

Given such a principle, and incremental decision-making beginning now which is informed by, and moves toward that principle, will steadily block the path of least resistance and a change in our management of materials will be forthcoming; not abeuptly, disrupting the fabric of society, but surely and steadily and with great effect on the sustainability of mankind. Specifically, the creative individuals who occupy this and similar institutions will apply their talents and imaginations to developing systems and technologies in which materials will not be accumulated as waste. Especially in an institution such as this; an institution premised on the notion we can do anything, anything that is, which is consistent with the natural system, it is appropriate to ask what should we do. We need to keep tightening the circle of controls around the release of materials to the environment. And as we do, the path of least resistance will become a chemically tight society, and with it a sustainable future. Visionary? Perhaps, but I believe achievable.

Will the future be an extension of today? In 50 years will we be deliberating whether this or that "waste," in the classic Hobbesian formulation, should be put on land, sea, and the air? Only if we keep asking it will the answer be yes. The dynamics of public policy assure that if we keep asking where to put it, we will have material to be put somewhere. The "law" that assures this result is that in any incremental decision there is a compelling force operating on the generator to take the path of least resistance -- often defined in contemporary economic jargon as least cost. And the path of least resistance, now at least, is to shift the burden of waste to some public agency, or to the environment and public health. Merely asking the question where to put it assures that this compelling force has an outlet.

As I have implied, a global community of some 8 or more billions of people with high expectations for standards of living cannot be sustained by principles of management of waste that assume that our option is simply where to choose to release it.

As I search for supporting evidence that we can move to a chemically tight society, I find it in the recent statement by Warren Anderson, Chairman of Union Carbide Corporation. While I share his and other peoples' sense of loss at the incident in Bhopal which precipitated his statement, he said, "Maybe out of this will come a whole new approach to this issue of health, safety, and environment . . . . so we have a commitment and an obligation to show the way if we can. The world's going to be a better place."4

Yes, I say when the CEOs give the direction and the resources, in the chemical industry especially, we can move to a chemically tight system.

One of the most perplexing things I have experienced lately is the responses to my thesis from individuals in the scientific and technical community. Surprisingly from this source I hear "it can't be done." It's a real reversal of roles. It used to be environmentalists were accused of being anti-technology. And most of that criticism was directed at the easy target of inartful articulation of their position by environmentalists. They weren't anti-technology, they are pro-choice technology — that is, choose technology that is consistent with life support sustainability; consistent with human freedom, consistent with beauty and joy. They were, they are saying, "don't choose heavy-handed technology; choose liberating technology." But because so much heavy-handed technology was being foisted on society in the form of big dams, big central utilities, and dangerous products, opposition had to be raised, and it was. But because the forum was generally over whether to go forward with some technological monster, it appeared the environmentalist opposition was anti-technology. But it was not opposition to technology -- although in the framework of a specific project it may have appeared that way (and in all candor, it must be admitted some was), the opposition was grounded in opposition to what was perceived as the wrong technological choice.

The environmental community is now becoming one of the few sources of optimism about the future. We -- even 8 or more billion of us -- can have a joyful, fulfilled life, if we tailor our technological choices to human values and biospheric requirements. And we are smart enough to do that -- if our public policy moves us in that direction.

The comments of the author of a much acclaimed recent study of the evolution of space technology, entitled, "The Heavens and the Earth: A Political History of the Space Age" should give us pause at the same time they provide insight into the role of technology in the human predicament. Walter McDougal said, "But the social mechanisms required to tap the full technological potential of a nation, particularly in the context of cold war competition, mean we have to pay a price for our advances in science and technology and the price is usually a sacrifice in human values. I believe it is inevitable, as long as international competition is the primary engine moving history, and technology is brought to bear in the competition, that we will move more and more toward management of people by a huge bureaucracy, by technocracy."[5]

In our context I would restate the point that a system that assumes a linear pathway in the management of material assures not just a sacrifice in human values -- but a sacrifice of the biosphere itself.

Preventing the oceans from becoming the path of least resistance -- is crucial in this long-range plan. Now here, obviously, I have to add specific recognition that we can't get there in one fell swoop. While at EPA I even made decisions to allow dumping and authorized the U.S. Air Force to incinerate Agent Orange at sea. The way to the future may require some dumping -- but it shouldn't <u>presume</u> dumping -- on land, air, or sea. The public policy should be firm and credible and carry a real message that if there is a compelling reason that the only recourse is to dump this or that; it is only a temporary situation. And that public policy must apply real pressure to change the conditions leading to the practice. I can't emphasize enough that we must free our imaginations to do things differently.

And so perchance to dream, to dream of a chemically tight future. What might it, or parts of it look like. I see a house -- perhaps called the MIT House after the team in 1990 that designed it -- into and out of which very little flows and that which does is not released to the environment. Energy is supplied by different manifestations of the sun. And I should add as a note in support that such a dream is not unrealistic. A recently retired MIT physicist has built a house in Williamstown -- not, unfortunately, the sunniest part of the world -- with no heat source other than passive solar and the heat dumped from light bulbs, refrigerators and human bodies. It is exciting -- it is now. But our MIT House will go further, water will be used for consumption not as a waste carrier. Human and organic waste will be converted to resources. I see a convenience room, presumably still called a kitchen, where it will be easy to separate and make readily accessible for collection and recycling all manner of things we no longer want, batteries, paints, chemicals, metals, plastics.

I see the sewer systems of today becoming the conduits of optical fibers of tomorrow. Waste treatment plants could be communication centers. Now that is a dream! Perhaps to help Willard Bascom's work on the south coast of California, the ocean outfalls will become devices for routine collecting samples of deep ocean water to support basic research.

If we take our system which now, in the aggregate, generates so much waste, and look at the component parts that lead to "waste" it is possible to say we can do it. Why not start recycling batteries? I find it profoundly disturbing to have to throw away every 2 to 3 weeks the batteries that energize my hearing aid. But if there is a path of least resistance, it will be taken.

If past reaction is any guide most of you out there are saying (if you are charitable), "pie in the sky" -- what about pesticides, what about chemicals released from millions of wearing automobile tires? What about this? This? What about that? Well, what about it? I dream of an energy future with very little combustion of fossil fuels and no use of nuclear fuels. I see a future where chemical production, especially long-lived chemicals, is dramatically reduced. I see a future where little if any new ore is mined.

I see a future, in short, where "waste" -- something that must be released somewhere is a tiny portion of what we now call waste. And in fact it won't be released, it will be stored.

I don't wring my hands at the intractability of problems. I see an exciting future, a future where we can agree on human values, and develop technology to serve them. Among those human values, is the recognition of the imperative of the protection and maintenance of the biological, chemical and physical integrity, not just of the oceans, but the life-giving and life-supporting biosphere itself.

If we can stabilize the human population within the next century and we apply the wisdom and humility that should result from the lessons of some four billion years of evolution, it should be possible to sustain the human population in the biosphere indefinitely. Not just sustain it -- to enrich it, and the individuals who comprise it.

Aldo Leopold wrote,

> "Examine each question in terms of what is ethically and esthetically right, as well as what is economically expedient. A thing is right when it tends to preserve the integrity, stability, and beauty of the biotic community. It is wrong when it tends otherwise."[6]

Our task is to push, steadily but steadfastly, the tendencies in the right direction.

END NOTES

[1]Roberts, Walter O., "Out of an Impasse Through a Strait," Journal '85, World Resources Institute, p. 68.

[2]World Development Report, 1984, World Bank, Oxford University Press.

[3]Lovins, A., and H. Lovins, Energy Strategy for Low Climate Risks, Report for the German Federal Environmental Agency, June, 1981.

[4]Interview, Chemical and Engineering News, January 21, 1985, p. 15.

[5]Interview, New York Times Sunday Book Review, April 7, 1985, p. 16.

[6]Leopold, A., A Sand County Almanac, Oxford University Press (1966), p. 240.

Acceptable Environmental Change from Waste Disposal

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ABSTRACT

Environmental alterations result from the disposition of waste materials. The identification of acceptable changes is a major criterion of effective waste husbandry. The protection of public health and of ecosystem integrity constitute the primary guidelines for appropriate waste management. Still, subjective judgments are often necessary and can supplement the assessments based on scientific and technological data. Three examples of possible changes in the marine environment as the consequence of the entry of societal discards are considered: (1) the disposal of sewage sludge; (2) the release of tin butyls through their use in anti-fouling paints on boats and ships; and (3) artificial radionuclide releases. Once the maximum acceptable levels or fluxes of polluting substances are determined, the translation of these numbers into regulatory and monitoring activity is essential. Public perception of environmental problems can conflict with rational disposal tactics.

INTRODUCTION

The introduction of waste materials to any environment will result in changes of its physical, chemical and biological properties. Sometimes the change will be beneficial to societal interests; sometimes it will be detrimental. For over thirty years marine scientists have recognized that the nature of seawaters can be altered by human activities (Goldberg, 1976). Initially, there was the concern about the widespread loss of resources through the improper discard of radioactive debris from nuclear power generating facilities. As a consequence the goals for the management of radioactive wastes were formulated to minimize any deleterious effects of the artificial radionuclides upon human health.

Subsequent impacts of pesticides upon non-target organisms, first recognized in the early 1960's, directed attention to the protection of ecosystem integrity. The potential destruction of entire groups of organisms by biocides brought about use restrictions in many northern hemispheric countries. At the present time there are fears that the excessive use of halogenated hydrocarbon pesticides in the tropics and southern hemisphere may cause a repetition of the ecodisasters of the 1960's in the northern hemisphere (Goldberg, 1983).

Once the polluting material or collective of substances is identified, cause/effects relationships are sought with regard to the impact upon the receptor. Environmental levels are then established which define acceptable alterations to the makeup of the environment. The British in their husbandry of radioactive wastes discharged to their coastal waters developed the "critical pathways" approach which sought to regulate the release of those radionuclides which might jeopardize the health of the most exposed individuals (Hunt, 1982). In the United States a similar model was developed which was applied not only to public health but also to the maintenance of ecosystem vitality, "the assimilative capacity" approach (NOAA, 1979). It is emphasized that in both concepts concentration levels, which could be monitored, were sought such that unacceptable alterations of the environment could be detected.

Of crucial importance to those responsible for managing our environment is an ability to regulate inputs of materials that can threaten resources. Scientific wisdom must be translated into simple numerical concepts which can be written into our laws. But the formulation of such concepts often may require subjective inputs. Herein, I will consider several waste disposal problems whose resolution may depend both upon judgment and upon hard scientific data. Finally, it must be recognized that public perceptions on the resolution of waste management problems can be in conflict with scientific evaluations. As a consequence, the management strategy finally adopted may not be the more rational one but one based upon public acceptance.

SEWAGE SLUDGE

An increasing world population with an increasing material usage is producing an increasing amount of waste which ends up in domestic sewage systems. Further, with increasing treatment of the sewage, an increasing volume of sewage sludge is being produced. Where will it go? The sludge does have a restricted land usage, especially as a soil additive, inasmuch as it may contain high concentrations of metals, some of which may be deleterious to plant growth. With large populations of the world living near sea coasts, marine disposal appears especially attractive. However, the materials do contain large amounts of organic matter whose dissolved oxygen demand can potentially interfere with life processes of animals. What amounts of alteration are allowable from marine discard of sewage sludge and how can disposals be managed through appropriate monitoring procedures.

A rather unique approach to the problem has been proposed by Jackson (1982) who adopted the concept that an unacceptable change involved the reduction of the oxygen levels of the water to less than 4 micromolar. Below this level, some organisms cannot survive. For any given site location and depending upon the flux of sewage sludge, models can be formulated to predict whether this level will be maintained. Using accepted parameters on mixing processes in the San Pedro-Santa Barbara basins of Southern California, a model was constructed. Pipe disposal at 800 meters of varying amounts of digested sewage sludge appeared to have crucial impacts on the oxygen concentrations but shallower disposal, say at 400 meters would not. The simple and easily measurable dissolved oxygen gas in principle can flag an unacceptable alteration to the environment. Other parameters can also be considered, such as the biostimulants nitrogen and phosphorus that could possibly lead to eutrophication, perhaps an unacceptable alteration.

Possibly a more subjective approach merits consideration. For example, in the Southern California Bight region about five percent of the fish, the Dover Sole, have surface abnormalities such as lesion, discolorations or fin-rot (SCCWRP, 1984). These alterations are associated with the inputs of domestic wastes from the eleven million citizens inhabiting the adjacent lands. Is this five percent figure an acceptable trade-off for waste accommodation? Should the number be one percent or ten percent? Most important is the basis for the designation of a number. Perhaps, the assimilative capacity of the area for domestic wastes might be based upon the area whose community structure is measurably altered by the discharge. Would a five percent change be acceptable? Possibly, marine ecologists can develop substantial and usable schemes in which to designate the acceptability of change due to waste disposal.

THE PROBLEM OF ORGANOTINS

The entry of organotins to coastal environments provides a significant problem in the management of toxic chemicals. Although not technically a waste, they do provide an interesting "for instance" for the control of a societal discard that disturbs marine ecosystems. Many countries of the world are now reviewing the problems associated with the release of the organotins from boats and ships to harbor waters. Can we ascertain with present knowledge the flux of organotins to a given part of the coastal ocean that will result in acceptable change?

The biocidal properties of organotin compounds were initially recognized in the early 1950's by Dutch scientists. These substances have subsequently been used as fungicides, bactericides, and preservatives for woods, textiles, paper and electrical equipment (a review is given by Bennett, 1982). The major compounds are tributyl tin oxide, tributyl tin fluoride, triphenyltin chloride, triphenyltin hydroxide and tricyclohexyltin hydroxide. Two of these compounds, tributyl tin oxide and tributyl tin chloride are incorporated into marine paints as anti-fouling agents. There are a number of attributes of these compounds that make them especially attractive and preferable to conventionally used anti-fouling compounds such as cuprous oxides. First of all, they provide more effective protection against fouling organisms for long periods. Secondly, they do not promote corrosion. Finally, they degrade to relatively harmless compounds with time.

The release of the organotins from the paint, usually given in weight per unit area per unit time, is the significant parameter in their use. The minimum value, consistent with anti-fouling activity, is sought. A greater release is non-economic. A smaller release is ineffective. There are three types of formulations: a direct mixing of the organotins with the paint constituents; incorporation into a chloroprene rubber; or incorporation into acrylate or methacrylate polymers.

There are other entries of organotin biocides to the marine environment. For example, toxic impacts of organotin effluents from the manufacture of "odor-free" socks where the biocide is applied as an anti-bacterial agent have been uncovered in North Carolina (Cardwell et al., 1984). The wastes from their manufacture have been implicated in fish kills through discharge into natural waterways which enter the oceans.

Both the military and civilian operators of boats and ships find the use of organotin antifouling coatings especially desirable. They have a five to seven year service life which reduces the need for more closely spaced underwater cleanings of ship bottoms. Their use results in a fifteen percent lower fuel consumption, as a consequence of a reduction in frictional effects due to biofouling. These applications translate into savings of hundreds of millions of dollars per year for large fleets. Presently used copper oxide based anti-fouling paints last for periods of only two years.

The butyl tins are introduced to the marine environment both from leakage directly from the applied paints and from painting and blasting operations. A typical release rate from ship hulls is of the order of 0.1 micrograms per square centimeter per day. The most common fouling species are barnacles and seagrasses although molluscs, tubeworms, hydroids and sponges are other known offenders. Established communities reduce the smoothness of the hull and consequently increase friction and drag through the water. Typical paints contain up to ten percent tributyl tin compounds.

There has been far more extensive work upon the impact of butyl tins on organisms rather than on humans. Health effects and acceptable daily intake rates are yet to be identified through public health studies. There is a limited amount of field data on the persistence of the compounds in the marine environment. The most sensitive organism so far studied is the juvenile mysid shrimp which has a toxicity value of 0.5 micrograms/liter for a 96 hour LC50. Acute (short term responses) toxicities have been measured for many organisms. Long term (chronic) toxicities have not as yet been investigated. They can be approached by applying a safety factor of ten to the acute levels, i.e. on the basis of present wisdom, a level of 0.05 micrograms per liter or less would protect marine organisms from adverse effects.

The persistence of butyl tins in the marine environment is poorly known. Photolysis may be the most important mode of abiotic degradation with environmental half-lives estimated to fall between 18 days and somewhat greater than 90 days. Another pathway from the aqueous system is adsorption on particulates and subsequent sedimentation. Degradation by micro-organisms can limit the persistence of the tributyl tins in the marine environment. Laboratory studies indicate that butyl tins in fungal and bacterial cultures can have biological half-lives of weeks or fractions of a week. In abiotic sediments where sulfides are present, the tributyl tin sulfide forms irreversibly and is removed from any toxic activity. In sediments biodegradation is the principle removal mechanism and the half-lives in aerobic and anaerobic sediments are reported to be of the order of 116 and 815 days, respectively.

Although present information is fragmentary, we have, I submit, adequate information to make a judgment as to whether to allow ships treated with tin butyls to utilize a specific harbor. If the yearly average area of ship bottoms painted with tributyl tins in a given harbor is known, the leakage of tin butyls into the waters can be estimated. Knowing the flushing times of harbor waters, coupled with the particulate loading of the waters and with persistances based upon biological and abiotic degradations, estimates of tributyl tin levels can be ascertained. The results can be compared with the upper limit of 0.05 microgram/liter proposed on the basis of the acute dose to shrimp larvae. Even though this approach is somewhat primitive, it can be checked by monitoring programs for the tributyl tin levels in the water.

An optional procedure to manage the use of tin butyls might be based upon a tradeoff. Clearly there are operational and economic benefits from the use of tin tributyls. Also the biocides will impact upon the more sensitive organisms, especially in areas where the tin tributyls have a short path to the organisms from the ship bottom, such as in dock areas. There are reports that arthropods and bivalves have disappeared from the dock areas in some marinas. The question then becomes how much of a harbor area can be sacrificed in order to minimize ship and boat maintenance and fuel costs. This is the same type of question that can be posed for the disposal of domestic wastes. A possible solution is then to accept changes in the flora and fauna of the harbor area. Is the displacement of some indigenous organisms by those that are more tolerant of butyl tins a reasonable exchange for the economies resulting to the ship and boat owners?

4. THE RELEASE OF ARTIFICIAL RADIONUCLIDES

There are many instances in which public perception, as opposed to scholarly assessment, has defined acceptable environmental alterations. Several recent instances involving low level artificial radioactive waste disposal address this point. The planned marked reduction in the release of low level radioactive wastes into the coastal waters of the United Kingdom and the decision to dispose of nuclear submarine hulks on land as opposed to the deep sea are responses in the main to public opinion rather than to conventional scientific and engineering wisdom and to economic considerations. It must be emphasized that the informed public can be put into a state of confusion by exposure to conflicting views of knowledgeable scientists.

Obtaining public acceptance of a rational discharge strategy based upon scholarly assessments can be difficult with the prevailing mood that the oceans should be kept inviolate with respect to societal wastes. This became especially evident with the problem of the disposal of decommissioned, defueled naval submarines. The United States Department of the Navy prepared an environmental impact statement in which land versus sea disposal options were considered (NAVY, 1984). The former involved storage of the radioactive parts of the submarines at the Savannah River Plant in South Carolina or at Hanford, Washington, both presently existing nuclear waste disposal sites. An alternative would be the sinking of the entire submarine, without the reactor core, to the seafloor in waters deeper than 4.0 km. from a U.S. coast.

The assessment of the options involved potential impacts upon the environment, the uses or resources such as land or materials, the impact upon ecosystems, the effects on public health, especially during protective custody of the vessels, and the relative costs. Sea Disposal turned out to be the least costly option. There were no evident impacts upon public health through exposure to radiation or to the environment in the land or sea disposal options. All alternative disposal techniques could not be examined inasmuch as such information gathering would have added additional costs to what appears to be an already expensive undertaking. Disposal at sea would have cost two million dollars, while burying the radioactive reactor compartments and the disposal of the non-radioactive parts of the ship would cost a little over seven million dollars. With one hundred ships slated for disposal, the sea option would result in savings of around a half a billion dollars.

Environmental groups, citizens and the U.S. Environmental Protection Agency challenged the validity of the assessment. For example, a fundamental concern of one environmental group, The Ocean Society, involved the absence of detailed knowledge about some aspects of deep sea ecology. Yet there were neither identified problems in the deep sea involving the transfer back to society of the radionuclides nor serious impacts upon ecosystems. Some scientists were concerned about the precedent of disposing of these low level radioactivities as providing a precedent for more extensive ocean dumping activities.

As a consequence of the public furor, the Navy abandoned its plan to sink the retired nuclear submarine hulks to the seafloor and instead to bury them in disposal yards. Eight ships are presently awaiting this fate. This example illustrates the urgency for marine scientists to formulate more persuasive arguments to convince the public of economically and scientifically sound courses of action in cases where the sea is proposed as waste receptacle. In this case the U.S. citizenry will pay more for a land disposal tactic that is not better than the sea alternative.

For nearly a quarter of a century, the United Kingdom nuclear establishments have discharged wastes to the coastal waters. The Fisheries Radiobiological Laboratory of the Ministry of Agriculture, Fisheries and Food have monitored these discharges to establish whether the resulting public radiation exposure is within nationally acceptable limits. Annually, reports are issued to identify the most exposed individuals and the radiation doses they received, either from the consumption of seafoods or from exposure on beaches or in boats (Hunt, 1982). The monitoring programs indicated that at no times were any individuals receiving amounts of radiation greater than those recommended by the International Commission on Radiological Protection.

However, in 1983 a television program suggested that there was an increased incidence of childhood leukemia in the neighborhood of the Sellafield Nuclear Facilities perhaps as a consequence of the artificially produced radioactivity entering the environment. A Commission was formed to inquire into the problem and headed by Sir Douglas Black (Black, 1984). The Commission concluded that the hypothesis can neither be categorically dismissed nor can it be readily proven. Mortalities from childhood cancer, particularly from those other than leukemia, appeared to be near the national average, but the possibility of local pockets of high incidence could not be excluded. The Commission pinpointed some great difficulties in relating measured environmental levels to actual exposure. Population exposures are determined on environmental measurements of radionuclides in various parts of the environment. Perhaps, there are unidentified sites of concentration which act as a path back to human society. Also, unplanned discharges, not detected by the monitoring programs, could have delivered a significant dose via an unsuspected route. Still, using models with most conservative assumptions, the Commission found no evidence of a general risk to children or adults living near the nuclear facilities compared to their near neighbors.

The report had critics both among the lay public and within the scientific community. Shortly after its issuance in 1984, there were a series of articles in Nature and in the popular press contesting the conclusions of the Black Commission (see for example Pomiankowski, 1984). Sophisticated statistical analyses of the data, only transferable with difficulty, if at all, to the lay public formed the bases of the disagreement. Both the report and its critics call for further research on the possible health consequences of the discharges. However, the Black Commission Report, suggesting that there is no demonstrable relationship between the discharges and the incidence of childhood cancer, will not overcome the momentum to greatly decrease the disposition of the radwastes to the sea. Present intents of the British Government are to markedly reduce the marine discharges over the next decade. Can we more effectively and in an economically reasonable way manage these wastes on land?

5. OVERVIEW

The first step in pinpointing an unacceptable alteration to the marine environment through waste discharge is the identification of what resource is being protected -- human health, ecosystem integrity, recreational use, transportation or aesthetics. Once the acceptable levels of a given waste in the ocean system is agreed upon on a scientific basis (or the rate of introduction of the waste), the translation of numerical levels or rates into public law is possible. However, sometimes objectively determined criteria can not be formulated and recourse to subjective judgments takes place. For example, in the case of the entry of tin butyls to the environment through use in ship and boat paints, the level to protect the most sensitive organism so far investigated, the mysid shrimp, is known? However, it may be worthwhile to consider an alteration in the community structures of organisms in the harbor as a rational trade-off for the thousands upon thousands of dollars savings in fossil fuel and maintenance costs to operate the ships. The determination of the area of the harbor that can be sacrificed is a subjective judgment. Similar trade-offs can be formulated for the disposition of sewage sludge to a coastal area where the waste problems of large populations can be rationally handled.

The identification of the minimum amount of information necessary to evaluate whether or not to put a given waste into the ocean is also a very important consideration. (Goldberg, 1984). In general the essential scientific and technological information to consider the marine disposal of wastes can be divided into three parts: the source term; the impact upon marine resources; and the mixing or dilution with the receiving waters. These three sets of data are site specific and interrelated. With such information, monitoring programs can be devised to ensure that unacceptable impacts have not occurred or are not possible of attainment in the near future.

6. REFERENCES

Bennnett, R.F. (1983). Industrial development of organotin chemicals. Ind. Chem. Bull. 2, 171-176.

Black, D. (1984). Investigation of the possible increased incidence of cancer in West Cumbria. Her Majesty's Stationary Office, London, 104pp.

Cardwell, R.D., Pavlou, S.P. and Kadeg, R.D. (1984). An assessment of the environmental chemistry and aquatic toxicology of trialkyltin compounds. Prepared for the Organotin Environmental Programme Association. September, 1984. 68pp.

Goldberg, E.D. (1976). The Health of the Oceans. UNESCO Press, Paris. 170pp.

Goldberg, E.D. (1983). Can the oceans be protected? Can. J. Fish. Aquat. Sci., 40, Suppl 2, 349-353.

Goldberg, E.D. (1984). Information needs for ocean waste disposal. La mer 22, 327-33.

Hunt, C.J. (1982). Radioactivity in surface and coastal waters of the British Isles, 1980. Ministry of Agriculture, Fisheries and Food. Aquat. Environ. Monitoring report No. 8. 35pp.

Jackson, G.A. (1982). Sludge disposal in the Southern California Basins. Environ. Sci. Technol. 16, 746-757.

NAVY (1984). Final environmental impact statement on the disposal of decommissioned, defueled naval submarine reactor plants. United States Department of the Navy. 3 volumes.

Pomiankowski, A. (1984). Cancer incidence in Sellafield. Nature 311, 100.

NOAA (1979). The assimilative capacity of U.S. coastal waters for pollutants. Proceedings of a Workshop at Crystal Mountain Washington, July 29-August 4, 1979.

SCCWRP (1984). Southern California Coastal Water Research Project Biennial Report 1983-1984. Long Beach, California. 332pp.

Waste Treatment and Transport Systems

Treatment Technologies and Effluent Quality of the Future, 2000 +

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1. INTRODUCTION

The wastewater treatment and sludge management technology which will be used in the future will undoubtedly be dictated by many forces which are difficult to predict from today's vantage point. But, it is putative that the present state of these technologies and how they were developed offers insight and permits predictions about the state of wastewater treatment and sludge management technology more reliable than one might initially think.

This paper will attempt to discuss what wastewater and sludge management technologies will be used in the year 2000+. These predictions will be based upon a review of the history of the development of these technologies which the authors believe is the most useful information which can be used for making these predictions.

2. HISTORY OF WASTEWATER TREATMENT TECHNOLOGY

Prior to 1900

The development of wastewater treatment technology is closely linked to man's move from a rural to an urban environment. After 1800, the economy of the United States like most western nations no longer was primarily based upon agriculture. This was caused by the beginning of the Industrial age which required manpower to be present in highly concentrated areas. From 1800 through 1850, waste disposal consisted mainly of the systems used in the rural environment. The pit privy was often the waste disposal method of choice, or worse, disposal consisted of dumping in nearby ditches which occasionally were flushed by rainfall to nearby streams.

Table 1 contains a listing of the municipal wastewater treatment systems and water quality impacts which were prevalent in the United States from 1850 to 1900. As can be seen, this half of the nineteenth century saw the beginning of modern wastewater treatment technology. The large cities began to build sewer systems to serve the residents and industries of such cities as Boston, New York, and Chicago. Sewers were built in this period to solve drainage problems in low or flat areas and to give a more aesthetically acceptable alternative to the privy, open drainage ditch, and other indiscriminate disposal practices.

Early sewer systems were designed to carry sewage to rivers and/or lakes in a convenient and more efficient manner but did nothing to solve the problem of pollution of these surface waters. In fact, early sewer construction probably aggravated pollution problems in some instances since more wastes were conveyed than previously. In addition, industries such as packing plants, tanneries, and steel mills began to spring up and contributed to the water pollution problems as well.

Basically, United States cities, before 1900, relied upon the water available for dilution in the receiving water-bodies to prevent or ameliorate pollution. European cities such as Paris

Table 1
Municipal Wastewater Treatment Before 1900

Domestic Sewage	Industrial Sewage	Municipal Processes	Water Quality Impacts
Kitchen and Human Wastes:	Packing Plants:	Dilution	Dilution Sometimes was Insufficient Causing Contamination of Drinking Water
Pathogens	BOD	Sewage Farms (Paris, Berlin)	
BOD	SS		
SS	Tanneries:	Chemical Precipitation	Minimal Treatment Resulted in Severe Water Quality Degradation in Some Cases
TOC	Cyanide		
	Sulfur	Intermittent Sand Filtration	
	Steel Mills:		
	Cyanide	Little Industrial Waste Treatment	
	Phenol		
	NH_3-N	Onset of Chlorination of Drinking Water Supplies	

and Berlin, which had to deal with pollution problems long before the United States did, were using sewage farming, sometimes referred to as broad irrigation. Such technology did not find favor here primarily due to asthetics and the nature of the climate.

Although the number of wastewater treatment systems employed in the United States before 1900 were not many, the use of chemical precipitation and intermittent sand filters was considered by some United States cities. Most industries, except in isolated cases, did not consider wastewater treatment as being necessary.

Water quality impacts, of course, were not as well recorded as they are today. But we do know that oftentimes water supplies were severely deteriorated by wastewater disposal as evidenced by the outbreak of waterborne diseases reported. Since drinking water disinfection did not become widespread until after 1900 such problems were perhaps inevitable.

The city of Chicago had one of the most severe problems of contamination of its drinking water supply by wastewater. Until 1900, the Chicago River drained directly into Lake Michigan, the source of drinking water for the city of Chicago. The city's sewage drained into this river and unlike today, typhoid, cholera, and dysentery were widespread in the community. To compensate for the problem, city officials kept extending drinking water intakes further from shore but this solution would only last for a few years. Finally, in 1885, a severe rainstorm flushed a large amount of waste into the Chicago River and into Lake Michigan. It was reported that twelve percent of the city's population died from the resulting outbreaks of cholera, typhoid and other waterborne diseases. Driven by this catastrophic event, the city decided to take drastic action. Given the fact that the wastewater treatment technology of the time was crude at best, the decision was reached to simply reverse the flow of the Chicago and Calumet Rivers and to take them out of the Lake Michigan drainage basin and put them instead into the Mississippi River basin. The Metropolitan Sanitary District of Greater Chicago (District) was created in 1889 and embarked upon the task of reversing the flow of the Chicago River.

The reversal of flow was accomplished by building a canal of sufficient size (10,000 cfs) to drain the Chicago River during storm flows. Figure 1 shows the man-made waterway system created to divert sewage into the Des Plaines River (eventually emptying into the Illinois River which ultimately enters the Mississippi). The main channel was completed in 1900, the North Shore Channel in 1911, and the Cal-Sag Channel in 1922. To prevent foul odors, and to dilute the pollutional level in the waterway for downstream uses, Lake Michigan water was discharged through locks into the system. Basically, Chicago had decided to relocate its source of drinking water contamination and dilute it sufficiently to control water pollution problems for downstream users.

Figures 2 and 3 show the success that Chicago achieved in reducing the level of typhoid fever deaths. As can be seen in Fig-

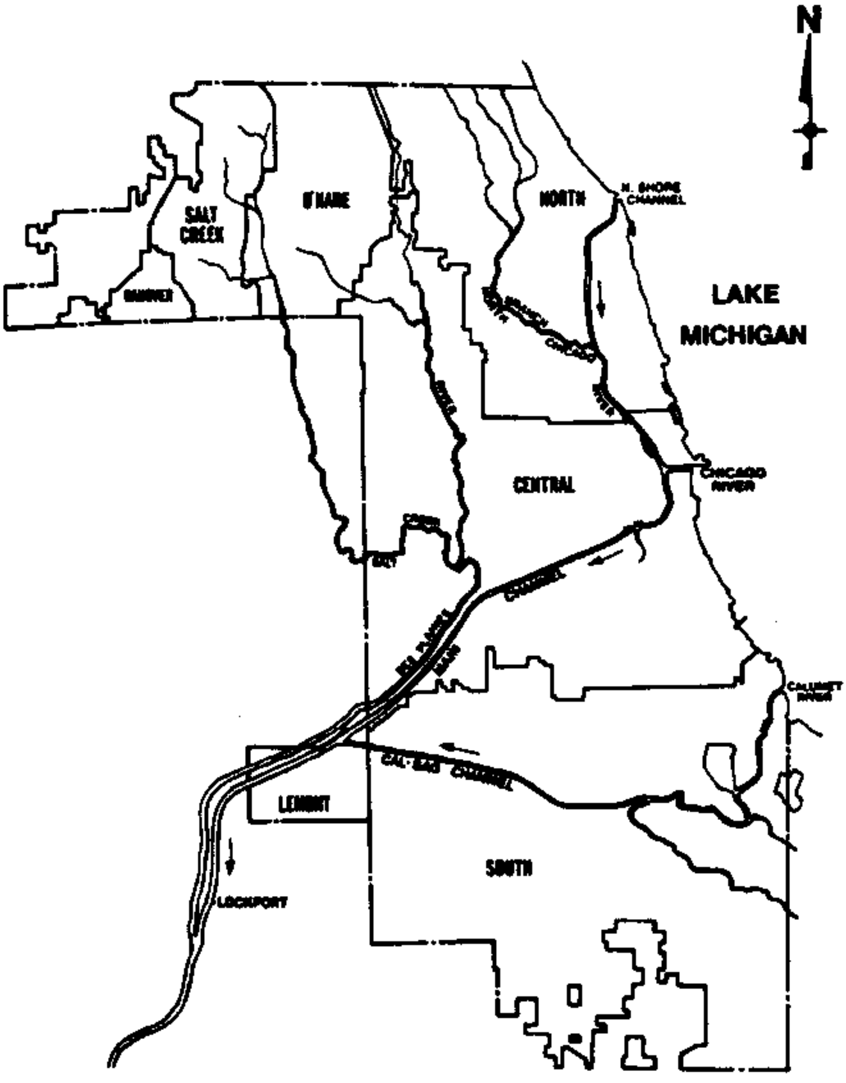

Figure 1
The Man-Made Canal System of the Metropolitan
Sanitary District of Greater Chicago

ure 2, the typhoid death rate before 1900 was variable but nevertheless very high in relation to the period after 1900. As indicated in Figure 3, by 1917 the typhoid death rate in Chicago was far below that of other major United States cities.

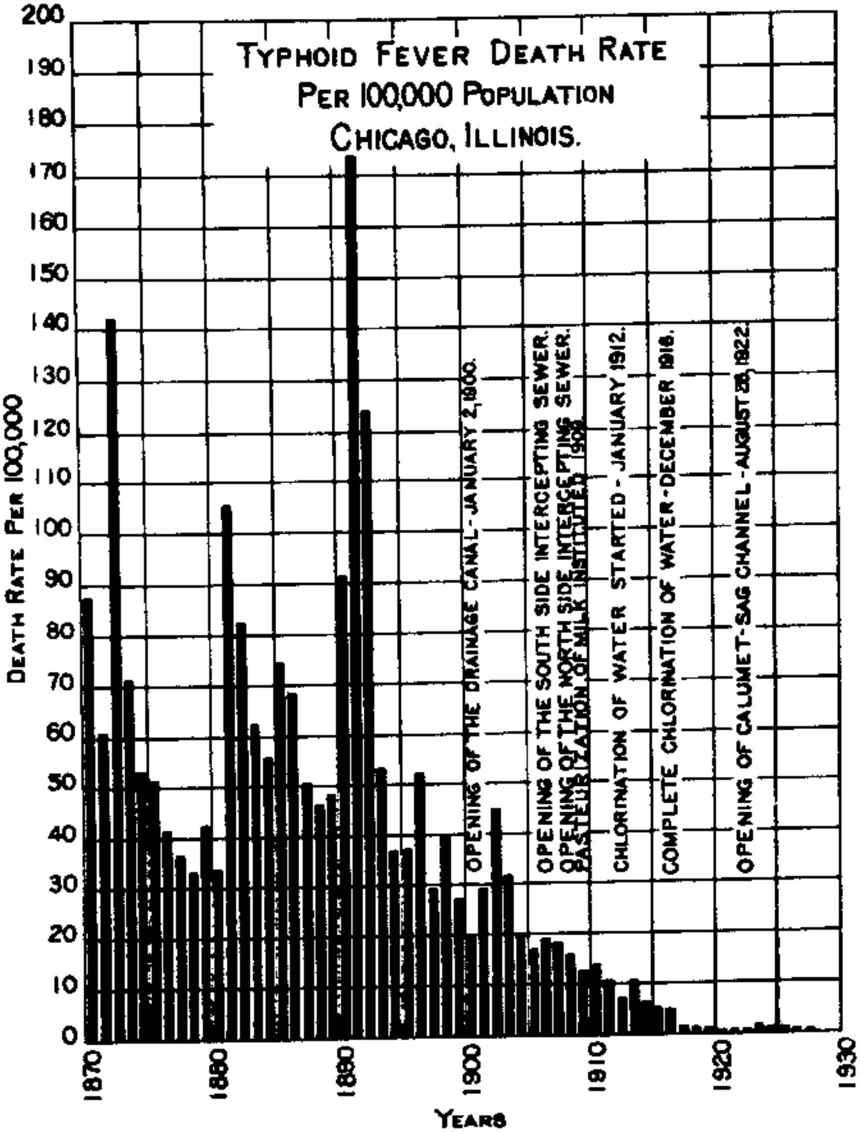

Figure 2
Typhoid Fever Death Rate Per 100,000 Population
Chicago, Illinois

The reduction in pollution culminated not only in a drastic reduction of Chicago's typhoid death rate but also in maintaining the excellent chemical quality of Lake Michigan water. As shown

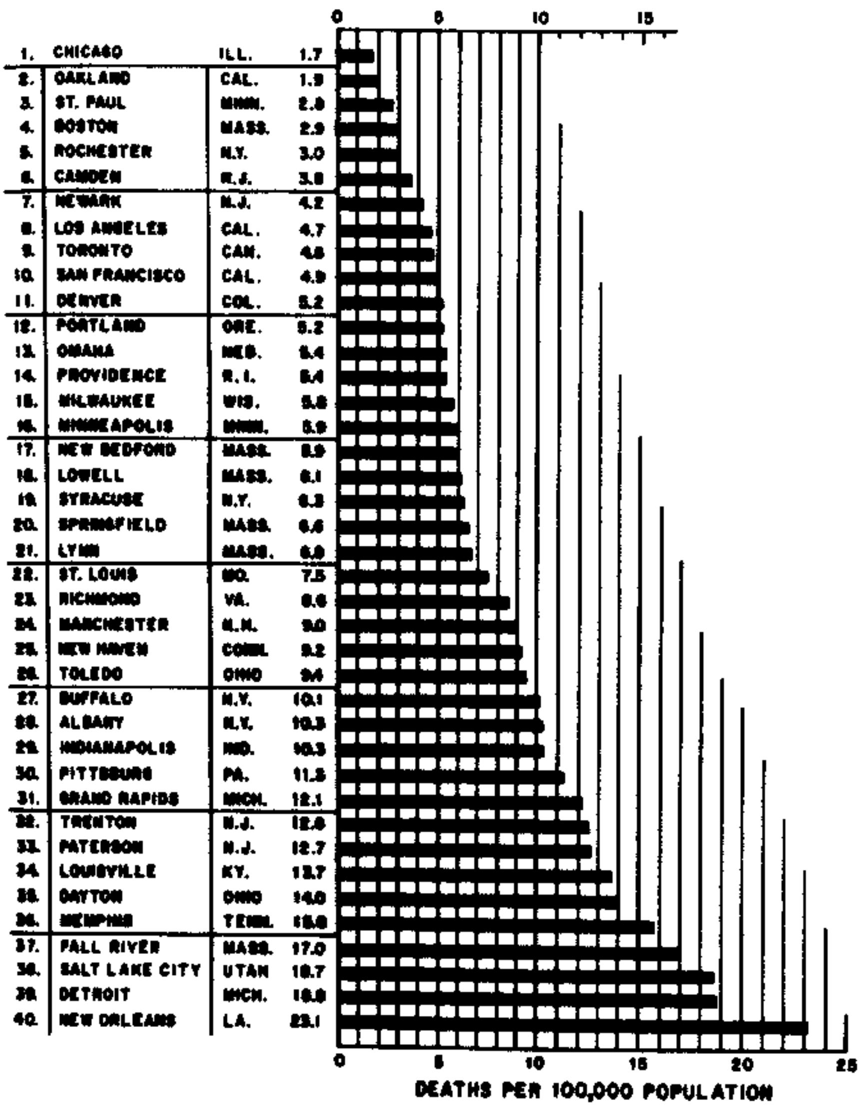

Figure 3
United States Typhoid Fever Mortality Rates
1917

in Table 2, there is little difference in the levels of total solids, ammonia, total nitrogen (albuminoid-N), chloride, nitrate, nitrite, and oxygen depletion in Lake Michigan water between 1908

Table 2
Selected Comparative Analyses of Chicago Lake Michigan Water and Chicago Sewage

	Lake Water		Raw Sewage	
	1908	1984	1909-10	1984
Total Solids	156	155	471	570
Ammonia-N	0.009	<0.01	8.8	11.4
Albuminoid-N	0.104	0.16	7.02	12.4
Chloride	5.0	8.8	40.0	98
Nitrate-N	0.29	0.26	0.35	0.21
Nitrite-N	0.00	0.00	0.11	0.16
Oxygen Consumed	2.6	<5	38+	221

All values are in mg/L.

and 1984. Remarkably, this same statement can be made for the raw sewage in Chicago. As also shown in Table 2, the sewage in Chicago in 1908 is really not significantly different from that of 1984 with respect to the parameters indicated.

The reduction in Chicago's typhoid death rate clearly demonstrates that water pollution control will improve public health not only locally but nationally. As shown in Figure 4, the life expectancy of Americans has increased dramatically since 1900. Many factors besides water pollution control measures have undoubtedly contributed to the rise of life expectancy, but those attributable to water pollution control must share some of the credit.

1900 to 1930

The first 30 years of the twentieth century must be considered to be the age of the most rapid development of wastewater treatment technology. In reality, this period marks the beginning of modern sanitary or environmental engineering as we know

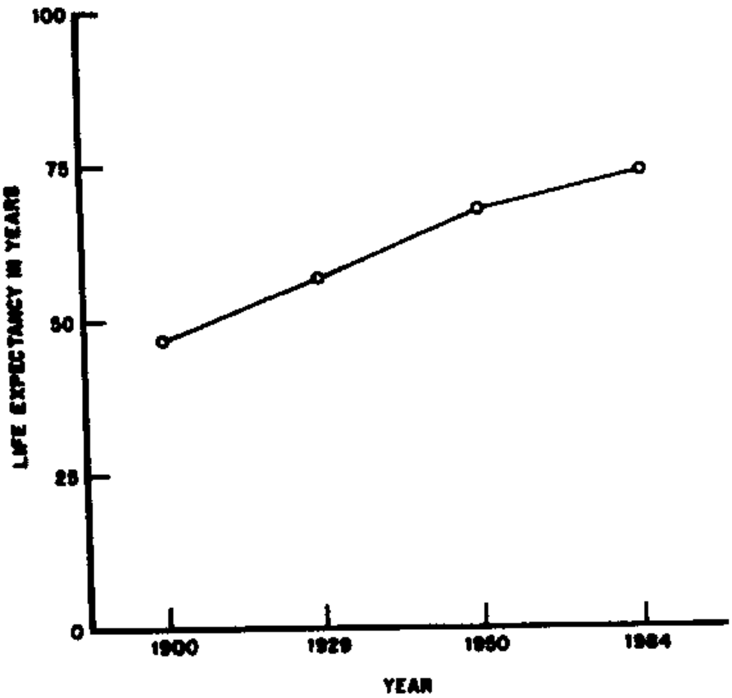

Figure 4
Life Expectancy 1900-1904

it today and represents a great improvement in the technologies available to engineers for treating wastewater. During this period, use of the water carriage system for wastes was becoming widespread in residences as indoor plumbing became the rule rather than the exception.

Table 3 presents a qualitative listing of municipal wastewater components, treatment technology, and water quality impacts for the period of 1900 to 1930. Treatment unit processes using Imhoff tanks, trickling filters, and activated sludge were invented and put into widespread use during this period. Obviously, the deterioration in water quality during this period was lessened by increased use of sewage treatment. The oxygen levels and aesthetic conditions in streams were improved. As Table 4 shows, the components in the sewage of this period were very similar to what we have today. For example, ammonia nitrogen levels range from 10.6 to 38.9 ppm and suspended solids from 214 to 384 ppm. Per capita water consumption ranges from 69 to 178 gals/day and is typical of what we expect today.

Table 3
Municipal Wastewater Components and Treatment Technology
1900-1930

Domestic Sewage	Industrial Sewage	Municipal Processes	Water Quality Impacts
Greater Use of Indoor Plumbing:	Food Processing Plants:	First Trickling Filter in U.S. - 1901	Putresible Conditions in Receiving Waters Alleviated
Pathogens BOD SS FOG	Canning Dairies	Imhoff Tank Invented - 1904	Aesthetics Improved
	Commercial Laundries:	First Imhoff Tank in U.S. - 1911	Waterborne Diseases Decreased
Increased Use of Soaps and Domestic Cleaning Agents:	FOG SS MBAS	Invention of Activated Sludge - 1914	Dilution Water Quantity Decreased
MBAS	Tanneries:	First Activated Sludge Plant in U.S. - 1916	
	Chromium Cyanide	District's North Side STW (175 MGD) Begins Operation - 1927	
	Automobile Industry: Copper Zinc Lead	A Little Industrial Waste Treatment, Mostly Combined Industrial and Municipal Waste Treatment	
	Cooking Operation: Cyanides Phenols NH_3-N	Chlorination of Water Supplies Becomes an Accepted Practice	
	Others	Practice of Sewage Dilution Progressively Decreases	

Table 5 shows the analyses of the sewage and water supplies of five cities in the United States during the period of 1905 to 1911. Again, it is indeed remarkable that the sewage of these cities is quite similar to that found today. This is also true for the water supplies of these cities which are similar to the quality found today. Industrial waste showed dramatic increases from 1900 to 1930 with the rapid development of the food processing industry, tanneries, and automobile manufacturing, to name a few.

1930 to 1950

The period of 1930 through 1950 represents a phase of consolidation and optimization for the sanitary and environmental engineering fraternity. As Table 6 shows, major treatment processes were developed. Activated sludge was generally the process of choice but fixed film systems were still used for smaller plants where less manpower and expertise were available.

Table 4
Comparative Average Sewage Analyses, 1920*

Constituents	Large American Cities (Combined Sewers)	American Mfg. Cities	Small American Mfg. Cities	American Residential and Rural Communities
Sewage Flow, gal. per cap. per day	178.0	95.0	69.0	80.0
Nitrogen as:				
Free Ammonia	10.6	26.5	38.9	27.2
Albuminoid Ammonia	7.0	11.9	11.3	7.8
Organic Nitrogen	8.0	24.1	23.8	18.0
Nitrites	0.11	0.26	--	--
Nitrates	0.44	1.19	--	--
Oxygen Consumed	59.0	133.0	107.0	71.0
Chlorine	48.0	109.0	83.0	47.0
Alkalinity	153.6	129.0	--	--
Solids:				
Total	1355.0	1058.0	730.0	603.0
Volatile	432.8	635.0	448.0	293.0
Fixed	902.8	423.0	262.0	210.2
Suspended Solids:				
Total	214.2	384.0	242.0	342.0
Volatile	144.3	288.0	203.0	260.0
Fixed	69.3	96.0	39.0	82.0
Dissolved Solids:				
Total	1052.5	608.0	488.0	261.0
Volatile	241.5	270.0	245.0	133.0
Fixed	811.0	338.0	243.0	128.0
Fats	25.3	37.0	--	--

*Values are in PPM.
REFERENCE: Metcalf L., and H.P. Eddy, Sewerage and Sewage Disposal, McGraw-Hill Book Company, Inc., New York, N.Y., 1922.

Industrial wastes were becoming more and more complex during this period with the tremendous growth of the organic chemicals industry and the sophistication of the defense industry.

Table 5
Comparative Analyses of Water and Sewage
(Parts per Million)*

Location	Residue on Evaporation	Free Ammonia	Chloride	Nitrates	Oxygen† Consumed
Brocton, Mass.					
Water Supply (1905-09)	32.7	0.013	6.2	0.009	2.2
Sewage (1905-09)	2210.0	47.8	144.3	—	433.7
Worcester, Mass.					
Water Supply (1905-09)	31.4	0.019	2.1	0.052	3.0
Sewage (1905-09)	871.0	17.7	113.4	—	122.0
Providence, R. I.					
Water Supply					
Filtered (1911)	56.0	0.009	6.3	0.12	3.5
Sewage (1909)	1715.0[1]	15.40	496.5	—	89.3
Chicago, Ill.					
Water Supply (1908)	156.0	0.034	5.0	0.29	2.6
Sewage (1909-10)	471.0	8.8	40.0	0.35	38±
Mansfield, Ohio (1910)					
Water	391.0	0.081	8.5	1.5	0.05
Sewage	876.0	13.3	108.7	0.2	51.6

*Reference: Metcalf, L., and H.P. Eddy, Sewerage and Sewage Disposal, McGraw-Hill Book Company, Inc., New York, N.Y., 1922.
[1]Estimated.
†Possibly by $KMnO_4$ in 4 hr.

Table 7 lists the number and types of industries inventoried by the District (The Sanitary District of Chicago in 1933) in 1933. Also included are the suspended solids and BOD loading rates to District plants and a population equivalent for these loadings. As can be seen, in 1933, the city of Chicago was a thriving industrial area with a multitude of industrial plants with an equivalent loading of over 2.2 million people. It is no wonder that sewage treatment was becoming increasingly important if not popular at this time.

Table 8 shows the basic method by which industrial wastes were handled in 1933 in 417 United States cities. In a similar fashion today, most industrial wastes were discharged to city sewers for treatment at domestic wastewater treatment facilities.

In 1933, the concept of the user charge was in its infancy since only one industrial waste plant was charged directly for treating its wastewater.

Domestic sewage also changed somewhat during this period with

Table 6
Municipal Wastewater Treatment
1930-1950

Domestic Sewage	Industrial Sewage	Municipal Processes	Water Quality Impacts
Garbage Grinding: BOD SS	Metal Plating Industry: Metals Cyanide	Activated Sludge Predominates Over Fixed Film Systems.	Aesthetics Improved.
Household Cleaning Supplies: Phosphorus Hard Detergents	Organic Chemical Industry: Organo-Phosphorus Pesticides	Improvements in Clarifier Design.	Anaerobic Conditions Decrease.
	Defense Industry: Nitrates Metals Nerve Gases/ Components Toxic Phosphorus Esters TNT Wastes	Industrial Pretreatment Begins.	Water Quality Improvements.
		Large Increase in Sewered Population.	Increased Interest in Water Sports.
		Large Increase in Plants Built.	

Table 7
The Sanitary District of Chicago, 1933
Summary of Industrial Wastes (Preliminary Estimates)

Number of Plants	Industry	Flow (MGD)	Suspended Solids (lb. per 24 hr.)	5-Day BOD (lb. per 24 hr.)	Population Equivalent
41	Packinghouses	29.37	190,000	152,000	912,000
10	Tanneries	2.93	33,500	22,650	135,900
20	Breweries	3.50	18,800	35,000	210,000
2	Yeast and Vinegar	2.43	6,040	26,750	160,500
2	Paper	6.35	22,760	6,530	39,180
2	Tomato Canning*	2.92	11,200	21,650	129,900
4	Malt Plants	1.93	3,470	14,960	89,760
4	By-Products Coke	1.73	1,280	18,600	111,600
3	Vegetable Oil Refining	2.00	3,470	8,060	48,360
3	Tar Distillation	1.06	570	3,910	23,460
12	Candy (Est.)	4.00	3,600	11,000	66,000
1	Corn Products	15.00	4,000	6,000	36,000
--	Laundries (Est.)	2.00	3,600	12,000	72,000
--	Dairies (Est.)	3.00	9,600	13,000	78,000
--	Miscellaneous (Est.)	10.00	10,000	20,000	120,000
	Total	88.92	281,890	372,010	2,232,060

*During canning season, August-October.

the beginning of the use of garbage grinders and the increased use of detergents.

Water quality continued to improve during this period with increased construction of sewage treatment plants. Secondary treatment of sewage was becoming more widespread during this period.

Table 8
Industrial Waste Treatment and Disposal in 417 U.S. Cities Over 1000 Population, 1933

Method of Treatment	Number	Percent
Waste Mixed with Sewage	229	54.9
Waste Treated Separately	4	0.96
Untreated Waste Discharged into Sewer	141	33.8
Waste Treated at Origin	13	3.12
Waste Treated by City	1	0.24
Waste Treated and Effluent Discharged to Sewer	21	5.03
Industry Charged for Waste Treatment	1	0.24

REFERENCE: Powell, W., Industrial Waste Treatment in Cities, Chemical and Metallurgical Engineering, 40, 359 (1933).

<u>1950-1960</u>

The period of 1950 to 1960 can be seen as transitional during which time many of the concepts that are used today were beginning to find favor. Table 9 shows some of the major trends in municipal wastewater treatment technology from 1950 to 1960. During this period, industrial pretreatment in some cities was used to control inputs to municipal systems. Water conservation in the suburbs became common place as the demand for water increased. Municipalities found that regulations at the state and federal level were playing an ever increasing role in the decision making process.

Municipal wastewater treatment was common in most cities and treatment facilities were operated more efficiently due to an increased understanding of the underlying theoretical and operating principles. Although industrial wastewater continued to become more diverse in their makeup, better municipal treatment coupled with increasing industrial pretreatment undoubtedly contributed to better water quality.

Table 9
Municipal Wastewater Treatment
1950-1960

Domestic Sewage	Industrial Sewage	Municipal Processes	Water Quality Impacts
Garbage Grinders	Chlorinated Hydro-carbons	Regulatory Control at Federal and State Level Begins.	Decrease in Direct Dischargers.
Hard Detergents: Phosphorus, Foam	Pesticides	Activated Sludge Still Predominates.	Improved DO and Transparency of Receiving Waters
	Herbicides		
Water Conservation in Suburbs: Higher BOD, Higher SS	Industrial Pre-treatment at Major Cities	Better Understanding of Biological Treatment Systems.	Decreased Inputs of Toxic Compounds.
Household Use of Weed Killers and Pesticides		Laboratory Methods Improve.	
		Operator Certification Increases.	

1960-1985

The last 25 years represents a period similar to that during 1900 to 1930 in that during both of these periods a large number of new treatment technologies were introduced and there was increased awareness and demand for better water pollution control by the public. Table 10 shows some of the treatment technology developed and the general make-up of industrial and domestic sewage from 1960 to 1985.

In the last 25 years, there has been a multitude of new processes developed including physical-chemical treatment, the UNOX process, ammonia stripping, nitrification and denitrification, etc. These processes have been intensely researched and promoted. Yet, few have been widely applied. Physical-chemical treatment is only used for specific industrial wastes and has not been accepted by municipalities despite strong promotion for this application. Despite the widespread publicity and research done on pure oxygen activated sludge systems, the standard activated sludge process still is generally chosen for application at municipal plants. Ammonia stripping was highly touted as a mechanism for ammonia removal from wastewater, yet biological nitrification in activated sludge systems is usually chosen for application at municipal plants even in colder climates.

Before 1960, the forces in environmental pollution control were more or less inherent in the municipalities. Municipalities adopted wastewater treatment schemes because of their own concern about effects upon local public health and receiving water qual-

Table 10
Municipal Wastewater Treatment
1960-1985

Domestic Sewage	Industrial Sewage	Municipal Processes	Water Quality
Soft Detergents: Less Foam	Nuclear Wastes	Clean Water Act-1972	Toxic Reductions Reduce Direct Discharges
Low Phosphate Detergents: Lower P	Pharmaceutical Wastes	NPDES Permits-1975	
User Fees Lower Water Consumption: Higher BOD Higher SS	Hazardous Wastes	Biological Nitrification & Denitrification	
Increases in Garbage Disposers: TOC BOD	Chemical Warfare Components	Ammonia Stripping	
	Pesticide Industry	Physical-Chemical Treatment	
	Organic Chemical Industry	Phosphorus Removal: Biological Chemical	
	Plastics Industry Wastes	UNOX Process	
	Adhesives Industry Wastes	Advanced Waste Treatment: Sand Filtration Coagulation-Sedimentation Carbon Adsorption	
		Pretreatment Standards: Categorical Standards Removal Credits	

ity. After 1960, federal and state regulatory forces were the chief reasons for the construction and expansion of treatment plants to deal with nutrients and toxics.

Naturally, the regulatory forces were the result of social and political changes in the United States, brought about by increased citizen awareness of, and desire for, better environmental protection. But the fact remains that many forces which influenced municipal sewage treatment policy were generally external to the municipalities themselves. State and federal agencies must, however, be prudent when they endeavor to require new and more stringent control of effluent contaminants lest the mistakes of the past are repeated. The issue of effluent chlorination offers real insight into how the setting of a treatment standard can lead to serious problems if foresight is not exercised.

In 1974, the USEPA required all municipal secondary effluents to meet an effluent fecal coliform count of 400 per 100 mL. To meet this standard, municipal plants used chlorination. By 1977, the USEPA removed the fecal coliform requirement from its secondary effluent standards. It became widely known that effluent chlorination produced harmful effects upon fish life and could elevate the levels of chlorinated hydrocarbons in drinking water supplies.

Table 11 shows what happened to fish populations downstream of the District's 350 mgd North Side Sewage Treatment Works (STW) before and after effluent chlorination was discontinued. As can be seen during effluent chlorination, from 1974 to 1980, fish pop-

Table 11
Comparison of the Results of Fish Collections Downstream of the North Side Sewage Treatment Works, 1974 through 1980 and During 1984
(Chlorination Discontinued April 1, 1984)

Fish Species Collected	Number of Fish Collected			
	0.7 to 1.7 miles[a]		1.8 to 2.5 miles[e]	
	1974-1980[b]	1984[c]	1974-1980[d]	1984[e]
Goldfish	1	1	0	7
Carp	5	2	3	8
Carp x Goldfish Hybrid	1	0	2	0
Golden Shiner	0	0	0	1
Bigmouth Shiner	0	0	0	1
Spottail Shiner	1	0	0	1
Bluntnose Minnow	0	12	0	99
Fathead Minnow	1	70	0	171
Longnose Dace	0	2	0	6
Black Bullhead	0	1	0	0
Brook Stickleback	0	18	0	68
Green Sunfish	10	7	4	0
Bluegill	1	0	0	2
Yellow Perch	0	2	0	2
TOTALS	20	115	9	366

[a]Distance below North Side Sewage Treatment Works outfall.
[b]Results of seven collections, 1974 through 1980, Touhy Avenue to Devon Avenue.
[c]Results of one collection, November 5, 1984.
[d]Results of six collections, 1974 through 1980, Granville Avenue to Bryn Mawr Avenue.
[e]Results of one collection, October 30, 1984.

ulations in the North Shore Channel were virtually nonexistent downstream of the North Side STW. After chlorination ceased (April 1, 1984), the number of fish collected equalled 366 as compared to only nine fish total in six previous years of collections during effluent chlorination.

It is hoped that state and federal agencies will be more prudent in the future in setting environmental standards and that the full environmental consequences of their actions will be evaluated before standards are finalized.

Because of huge amounts of construction money originally available through the Federal Water Pollution Control Administration and eventually the United States Environmental Protection Agency (USEPA), it was inevitable that the period of 1960 to the present would result in a proliferation of new technology to deal with the concern about nutrients and toxics. Physical-chemical treatment, ammonia stripping towers, nitrification-denitrification, biological phosphorus removal, rotating biological contactors, and different types of sludge treatment and disposal techniques are but a few of the methods which emerged in the 1960s and 1970s. Many municipalities were faced with the problem of meeting very short deadlines to build facilities to deal with impending effluent standards for nutrients and toxics. Oftentimes, municipalities were forced into construction with little data to support the designs used.

The District, too, was faced with the imposition of strict effluent standards during the 1960s and 1970s. However, these standards were imposed mainly upon those facilities of the District which discharged to natural streams, not the man-made waterway system which received so much attention from 1910 to 1960.

Table 12 is a listing of several nitrogen removal studies conducted by the District from 1972 to 1980. Based upon the results of these studies, the District constructed two new wastewater treatment facilities utilizing the two-stage activated sludge process, the John E. Egan and O'Hare Water Reclamation Plants. Another facility, the Hanover Park Water Reclamation Plant, was expanded to a single stage activated sludge plant.

As early as 1965, the District became aware of the possibility of effluent BOD and SS standards being more stringent than secondary treatment for facilities discharging to natural streams. From 1965 through 1978, a variety of granular media filters and microscreens were evaluated on both pilot and full-scale basis. Table 13 presents a chronology of this testing program. As a result of these studies, the District installed granular media filters at the above mentioned Hanover Park, John E. Egan, and O'Hare Water Reclamation Plants. The last 25 years has added numerous processes to those available for selection by engineers but the time proven systems developed from 1900 to 1930 as the major components of several treatment schemes still predominate.

The last 25 years has seen a proliferation of state and federal regulations on municipalities and industries. These regulations and the availability of federal construction dollars have caused a rapid increase in the size and number of municipal treat-

Table 12
Nitrogen Removal Studies by the Metropolitan Sanitary District of Greater Chicago 1972-1980

Process	Size	Objective	Results
Biological Rotating Disc (BRD)	Pilot plant ($2.0\ gpd/ft^2$)	Study BRD as means of removing high NH_4-N concentrations from sludge lagoon supernatant	NH_4-N removal 99%, pH control required, need added Na_2CO_3
Two-state Nitrification	Pilot plant (150 gpd)	Study nitrification in high industrial waste sewage by two-stage process	NH_4-N removal 88%, NH_4-N oxid. 145-420 mg/L.d, Avg. SRT = 19.3 days
Single-stage Nitrification	Full-scale (21 mgd)	Test of design criteria for single-stage nitrification at West-Southwest STW	HRT = 8.9 hrs SRT = 9.2 days NH_4-N removal 95%, high NOD in effluent
Single-stage Nitrification	Full-scale (80 mgd)	Test design criteria for single-stage nitrification at North Side STW	SRT $\geq$ 7 days for $\leq$ 90% NH_4-N removal at 10°C; DO <2 mg/L inhib. nitrif.
Single-stage Nitrification	Pilot plant (100 gpd) & full-scale (30 mgd)	Study nitrification in high industrial waste sewage by single-stage process	HRT = 10 hrs SRT = 10 days for year round nitrification, NH_4-N removal 90%
Two-stage Nitrification	Pilot plant (0.6 mgd)	Develop design parameters for two-stage nitrification at John E. Egan WRP	Optimum 2nd stage nitrification. HRT = 4.4 hrs SRT = >10 days
Single-stage Nitrification	Full-scale (300 mgd)	Test design criteria for plant expansion at West-Southwest STW	SRT controlling factor. 10-day SRT for year round operation HRT = 7.9 hrs

HRT = Hydraulic Retention Time.
SRT = Solids Retention Time.
NOD = Nitrogenous Oxygen Demand.

Table 13
Testing of Tertiary Treatment Systems for Effluent Polishing at the Metropolitan Sanitary District of Greater Chicago

Year	Facility	Type of Effluent	Systems Tested	Scale
1965-68	Hanover Park	Secondary	Hardinge continuous back-washing sand filter, Glenfield and Kennedy micro-strainer	Full
1969-70	Hanover Park	Secondary	Delaval upflow filter, Neptune Microfloc mixed media filter, Graver dual media pressure filter	Pilot
1971-73	North Side	Secondary	Crane-Cochrane micro-strainer	Full
1973-74	West-Southwest	Two-stage Nitrified Secondary	Neptune Microfloc mixed media filter	Pilot
1974-75	North Side	Nitrified Secondary	Roberts dual media gra-vity filter	Pilot
1975	Calumet	Nitrified Secondary	Roberts dual media gra-vity filter	Pilot
1977-78	John E. Egan	Two-stage Nitrified Secondary	Dual media gravity sand filters	Full

ment plants. Obviously, improvements in water quality have resulted. Improvements in water quality have also resulted from the development of biodegradeable detergents and the use of low phosphate household cleaners.

Industrial wastes have continued to become more complex in the past 25 years. However, more stringent effluent standards and the National Pollution Discharge Elimination System (NPDES) have reduced the number of direct industrial dischargers thereby lessening water quality impacts.

3. WATER REUSE

Water Reuse Technology

With the progress made in the municipal wastewater treatment field from the early era of dilution to the current use of complex and advanced wastewater treatment processes, the quality of the effluent derived from such processes has also improved. Primary sedimentation of sewage following screening and grit removal, perhaps removes about 30 to 50 percent of suspended solids and the biochemical oxygen demand associated with these solids. Following primary treatment, sewage can be subjected to secondary treatment by suspended growth systems such as the activated sludge process and by attached growth systems such as trickling filters and rotating biological contactors.

The quality of the effluent that can be expected from these processes and from additional treatment (advanced or tertiary waste treatment) is given in Table 14 (Metcalf and Eddy, Inc., 1979). It can be seen from this table that there is a progressive increase in the quality of the effluent as additional treatment units are added. The land treatment of primary or secondary effluent gives a fairly good quality effluent, which can be put to many uses. The effluent derived from advanced wastewater treatment technologies can be further processed by groundwater recharge and subsequently used.

Depending on the quality of the effluent achieved from a wastewater treatment system, it can be used for various purposes such as irrigation, recreation, and industrial uses. Secondary effluents can be processed further to render them fit for potable use (direct reuse).

A very important issue that will increasingly confront environmental engineers in the future, particularly in areas of water shortages, is the use of processed municipal effluents for various purposes as the demand for water increases. If the quality of the effluent produced is within the limits specified for the intended use, it will be a valuable water resource. In many parts of the world, including the United States, nonpotable use of processed effluents is already being practiced.

Today, we have the technology to accomplish direct use of processed wastewater to supply the needs of very small numbers of individuals. For example, the National Aeronautics and Space Administration (NASA) has developed systems of direct use of processed wastewater for astronauts who are put into space for extended periods. However, the discussion contained herein will deal with systems developed for large volume reuse of water rather than the NASA type systems which are designed for atypical situations.

Broadly the pollutants in wastewater can be characterized as soluble and suspended matter. Different unit processes have been developed over the years to remove pollutants to varying degrees from wastewater to achieve an acceptable effluent. The degree of

Table 14
Treatment Levels Achievable with Various Operations and Processes Used for Advanced Wastewater Treatment

		Typical Effluent Quality						
Secondary Treatment	Additional Treatment	Suspended Solids mg/L	BOD mg/L	COD mg/L	Total N mg/L	PO_4 as P mg/L	Turbidity mg/L	Color Units
Activated-sludge (suspended growth process)	None (Secondary Effluent)	20–30	15–25	40–80	20–60	6–15	5–15	15–80
	Granular-medium filtration	<5–10	<5–10	30–70	15–35	4–12	0.3–5	15–60
	Coagulation plus settling and granular-medium filtration	<1	<5	30–60	15–30	0.1–1.0[a]	0.1–1.0	10–30
Land Treatment	Irrigation[b],	<1	<2	—	3	0.3		
	rapid infiltration[c],	2	2	—	10	3		
	Overland flow[d]	10	10	—	3	12		
Trickling-filter process	None (Secondary effluent)	20–40	15–35	40–100	20–60	6–15	5–15	15–80
Rotating biological Contractor (attached growth processes)	granular-medium filtration	10–20	10–20	30–70	15–35	6–15	<10	15–60

a–Reduction of PO_4 to this level will typically require 200 ppm of alum or 400 ppm of lime; if greater PO_4 concentrations can be tolerated, coagulant dosage is decreased.
b–Percolation of primary or secondary effluent through 1.5 m of soil.
c–Percolation of primary or secondary effluent through 4.5 m of soil.
d–Runoff of comminuted municipal wastewater over about 45 m of slope.

processing obviously depends on the quality of the effluent desired for the intended use. The unit processes that have been developed to remove the dissolved and suspended matter contained in wastewater are outlined in Figure 5. A combination of these unit processes can be judiciously selected to obtain an effluent of a desired quality (Stander, 1977).

Typical effluent qualities that can be achieved in the processing of municipal wastewater by using various unit operations

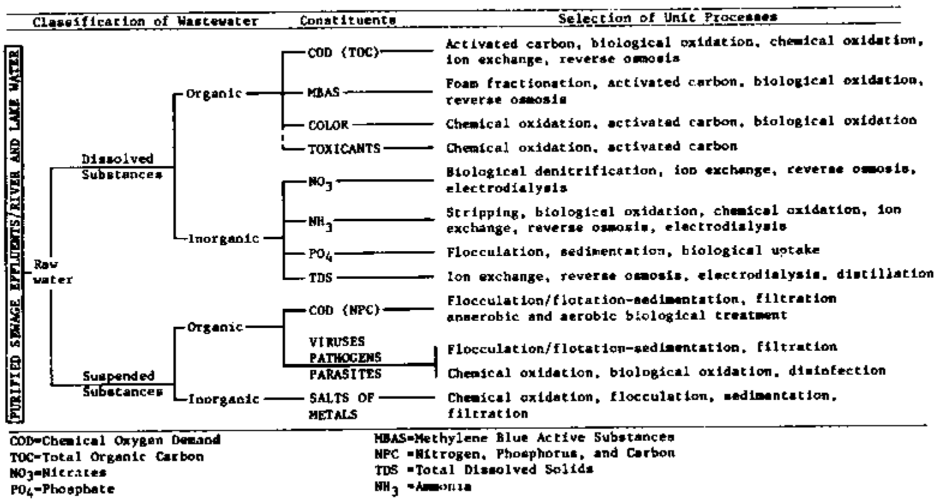

Figure 5
Reuse of Water for Municipal Purposes
Classification of Wastewater Contaminants and
Selection of Process Units

were shown previously in Table 14. A similar effluent quality may be achieved by different combinations of the indicated unit processes as well. However, their selection depends on not only the quality of effluent desired, but also the need for such effluents, and the economics of the treatment train employed.

Unit processes such as ion exchange, dialysis, reverse osmosis, etc., can be used to further reduce the dissolved inorganic matter contained in processed effluents.

To illustrate the extent of purification that can be achieved by processing wastewater using various unit processes for direct reuse and indirect reuse, some typical case histories will be presented. These are real life situations where wastewater is processed for a variety of uses including immediate human consumption, recreational purposes, and artificial recharge of groundwater. These are: (1) the water reclamation plant of Windhoek in South Africa; (2) Water Factory 21, Fountain Valley, California; (3) the Santee County Water Reclamation Facility; and (4) the Flushing Meadows, Arizona Groundwater Recharge Project.

Windhoek, South Africa

The original Windhoek Water Reclamation Plant in South Africa (Figure 6) processed trickling filter effluents through algae maturation ponds. The algae were separated by flotation and foam

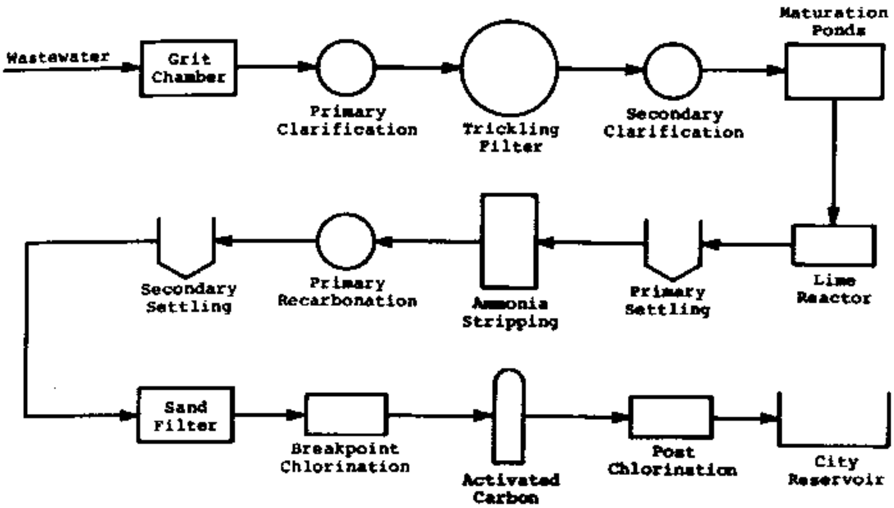

Figure 6
Sewage Purification Works, Windhoek, South Africa

fractionation, and the effluent treated by activated carbon, and break point chlorination. This process was subsequently modified to eliminate the algae flotation and foam fractionation units and now includes lime treatment, ammonia desorption, and recarbonation. Ammonia desorption is employed as ammonia uptake by algae was found to be unsatisfactory. Denitrification and phosphate removal are also planned to be integral unit processes of this plant to improve the quality of the final effluent. The quality of the effluent, after carbon adsorption (before post chlorination) (Von Vuuren et al., 1980), is given in Table 15. These data indicate that the chemical and bacteriological quality of the processed effluent was highly satisfactory. The heavy metal concentrations never exceeded acceptable criteria for potable water. The public acceptance of the processed effluent for potable use is favorable because the public is aware of the need for water reuse as a consequence of water shortages.

Fountain Valley, California

Water Factory 21, located in Fountain Valley, California is named as such to denote a prototype of the plants that may be used in the twenty-first century to meet the demand for water in arid areas. Besides offsetting some or all of the water demand, the recharge of the effluent of such plants into the groundwater table is expected to provide a barrier for saltwater intrusion.

Table 15
Comparison of the Quality of Renovated Wastewater with Drinking Water Quality

Parameter†	Chicago Drinking Water	Water Factory 21 California	Windhoek, SA	District Secondary Effluent	USEPA Drinking Water MCL
Total Dissolved Solids	163	N.D.	646	539	500*
Nitrate Nitrogen	0.25	N.D.	9.7	4.1	10
Cadmium	0.001	0.0007	N.D.	0.01	0.01
Chromium	0.001	0.002	N.D.	0.02	0.06
Copper	0.001	0.009	N.D.	0.02	1.0*
Total Iron	0.010	0.02	N.D.	0.1	0.3*
Lead	0.001	0.0009	N.D.	0.02	0.05
Manganese	0.001	0.002	N.D.	0.02	0.05*
Mercury	0.0001	0.0005	N.D.	0.0003	0.002
Zinc	0.001	N.D.	N.D.	0.1	5.0
Silver	0.001	0.0005	N.D.	0.00	0.05
Selenium	0.001	0.005	N.D.	0.00	0.01
pH	8.3	N.D.	7.33	7.5	6.5-8.5*
Turbidity	0.35	0.11	N.D.	N.D.	1-5
Color	0	N.D.	N.D.	N.D.	15
Total Coliforms	0.0	0.01	0.0	9960	1.0

†All units are given in mg/L except for pH, color and total coliforms (counts/100 mL.
*Secondary standards.
N.D. = Not Determined.
MCL = Maximum Contaminant Level.

A schematic of Water Factory 21 is given in Figure 7 (Donovan and Bates, 1980). The process of this facility consists of treating activated sludge effluent by lime clarification, ammonia desorption by air stripping, recarbonation, prechlorination, and mixed media filtration (coarse coal and silica sand supported on garnet gravel). Two-thirds of the filtered effluent is processed through a granular activated carbon filter and the remainder through a reverse osmosis unit. The demineralized effluent and the carbon-treated effluent are then blended in a reservoir and subsequently recharged into the groundwater table. The quality of the effluent from the Fountain Valley, California facility is presented in Table 15. The quality of the finished water of the central water filtration plant of the city of Chicago, along with the USEPA drinking water standards are also given in Table 15 for comparison. It can be seen from the data presented in this table that technol-

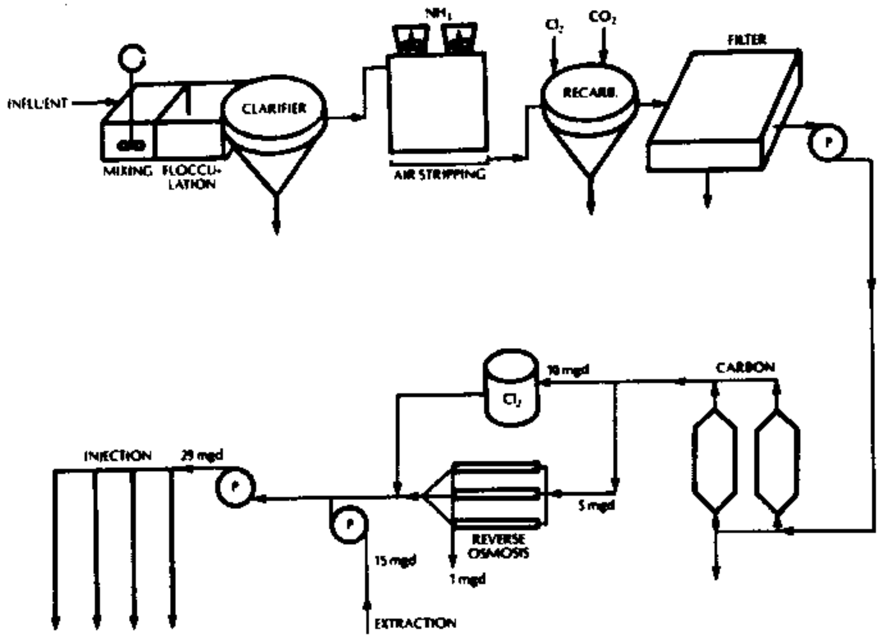

Figure 7
Process Schematic for 15 mgd Water Factory 21 Operated by the Orange County (California) Water District

ogy is currently available to produce an effluent comparable to the quality of finished drinking water.

Santee Water Reclamation Facility, California

A schematic diagram of the Santee County Water Reclamation Facility, California, is given in Figure 8. The effluent from the activated sludge plant is discharged into a 30 mgd oxidation pond from which a portion of the effluent is pumped one-half mile to three acres of infiltration beds. The infiltrated water is then fed to the four lakes located downstream. The infiltrated water is chlorinated in a contact chamber prior to its entry into the uppermost part of the lakes (Lake 5). A swimming area is set aside for recreational purposes. Typical effluent quality of the Santee County Water Reclamation Facility (Table 16) indicates that there was no untoward reaction of the public to use the water in the lakes for recreational purposes (swimming and fishing).

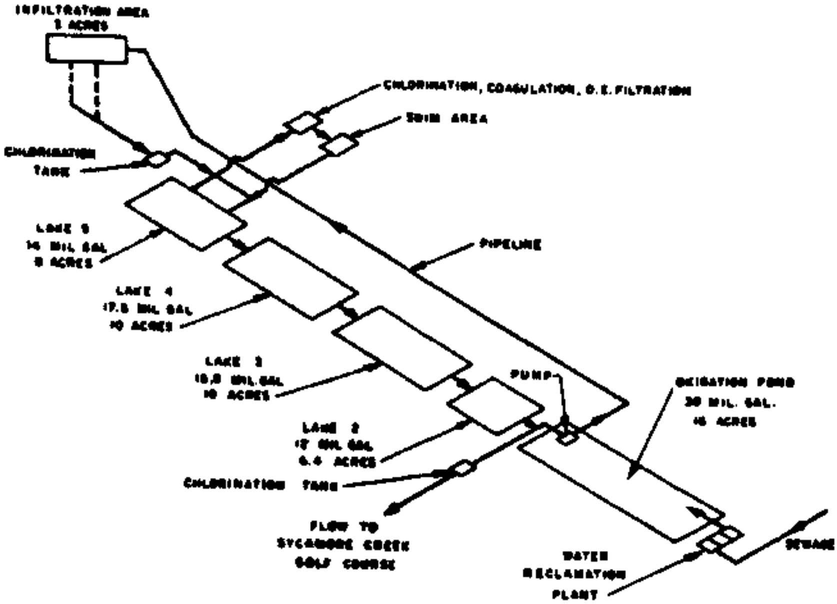

Figure 8
Isometric Sketch of Lake System
Santee, California

Flushing Meadows, Arizona Groundwater Recharge Project

The Flushing Meadows Project consists of six parallel horizontal basins (20' x 700') which are recharged with secondary effluent. The water depth in these basins is maintained at 6" or 12". The soil in the basins consists of 3' of fine loam sand followed by coarse sand and gravel layers to a depth of about 240', where a clay layer occurs. By alternating the flooding period of 2 to 3 weeks with 10 to 20 days of drying, even after 10 years of operation, there was no reduction in the hydraulic loading of the aquifer. The renovated water is free from viruses. A 100' lateral movement appears to be adequate to render the renovated water free from coliform organisms. The renovated water is of a quality that is suitable for irrigation, primary contact recreation and other comparable purposes. The quality of the secondary effluent and the renovated effluent is given in Table 17.

Toxic Organics and Water Reuse

It is quite possible to achieve highly satisfactory removals

Table 16
Santee Water Reclamation Plant Effluent Quality

	Santee Water Reclamation Plant		
Parameter	Oxidation Pond Effluent	Infiltrated Effluent	Lake System
Turbidity JTU	30	5	0-20
pH	7.7	7.7	8.8
TDS	1,168	1,150	1,150-1,600
SS	8.6	5-10	--
Alkalinity ($CaCO_3$)	250	240	50-170
Total Hardness ($CaCO_3$)	380	400	210
ABS	--	--	--
Threshold Odor Number	--	--	--
NO_3-N	1	1	1
BOD	5	3.5	--
COD	--	41	--
Total PO_4	8	3.6	0.1-4.2
Cl	230	250	270-480
SO_4	450	340	380-575
Coliforms (MPN/100 mL)	--	<2	<2.2
Visible Solids	--	--	--
Floatable Oil and Grease	--	--	--
Emulsified Oil	--	--	--

All units are in mg/L except pH and others which are indicated.

of the conventional pollution parameters such as biochemical oxygen demand, suspended solids, and coliforms by secondary and tertiary waste treatment technology. Although efforts have been made to remove taste and odor causing organics, such as phenol, from drinking water, very little attention has been given in the past to understand the nature and removal of the toxic organics present in wastewater effluents. Recently, the concern for toxic effects of such compounds, which are present in drinking and wastewater, has increased dramatically and has become an important issue. Although conventional water treatment technologies are able to remove most of the taste and odor causing compounds effectively from drinking water sources, they are not effective in removing pesticides. Table 18 shows the removal by various pesticides that can be expected by using various water treatment processes (McJunkin, 1982). Treatment of waters by activated carbon provides excellent removals of the toxic organics as indicated by the data provided in Table 18.

Table 17
Quality of Secondary Effluent and Renovated Effluent at the Flushing Meadows, Arizona Infiltration Site

Parameter*	Secondary Effluent	Renovated Effluent
pH (units)	7.6-8.1	7
TDS	1,075	1,095
SS	10-70	0
BOD	10-20	0-1
COD	30-60	10-20
TOC	10-30	0-10
NH_4-N	20.7	2-8
NO_3-N	3	6.25
NO_2-N	<0.1	Trace
ORG-N	3.7	0.58
PO_4-P	79	0.51
F	2.08	0.9-1.66
B	0.59	0.59
Zn, µg/L	193	35-108
Cu "	123	16-17
Cd "	7.7	7.2-7.3
Pb "	82	66
Hg "	2.1	1.2-1.4
Fecal coliform, counts/100 mL	10^6	0 (at 300' depth)
Viruses, PFU/100L	2,118	0

*All units given are mg/L except where indicated.

A similar technology can be applied for the processing of secondary effluents.

The degree of organic removal that can be accomplished by current technology is typified by the data in Table 19 available from Water Factory 21, Fountain Valley, California (Argo, 1985). Data available for the organics present in the secondary effluent and recharged groundwater at the 23rd Avenue site in Phoenix, Arizona (Bouwer and Price, 1984), are also given in Table 19. Argo (1985) has reported that Water Factory 21 has the ability to meet all the USEPA drinking water standards and has the ability to reduce the primary pollutants below the mandated limits particularly when reverse osmosis was used. Thus, the technology at this facility has the ability to produce a product water that is suitable

Table 18
Percent Organics Removed by Water Treatment Processes*

Process	Endrin Reduction Percent	Lindane Reduction Percent	Toxaphene Reduction Percent	2,4-D reduction, percent: Sodium Salt	Isopropyl ester	Butyl ester	Isooctyl ester
Coagulation, filtration	35	<10	<10	<10	<10	<10	<10
Coagulation, filtration and adsorption with:							
Powdered activated carbon, mg/L:							
5-9	85	30	93				
10-19	92						
	80	55			90	90	90
20-29	94	80-90					
30-39				90			
40-49					97	97	
50-59	98						97
70-79		99				98	
Granular activated carbon, 7-5-minute full bed contact time	<99	<99					
Oxidation:							
Chlorine, mg/L:							
5	<10	<10					
8		<10					
30		<10					
100			<10	<10	<10	<10	<10
Ozone, mg/L:							
11		<10					
38		55					
Potassium permanganate, mg/L:							
10		<10		<10	<10	<10	<10
40		<10					

*Reference: McJunkin, F.E. Water and Human Health, U.S. Agency for International Development, Washington, D.C., 1982.

for a number of uses including direct and indirect reuse by artificial groundwater recharge. However, the results reported for the renovated wastewater from the 23rd Avenue infiltration project shows that the concentration of the toxic organics in the secondary effluent were considerably higher than those present at the Water Factory 21 facility, and that some of the organics have, in fact, increased in concentration with infiltration. Nevertheless, the produced water met the water quality criteria for unrestricted irrigation and primary contact recreation (Bouwer and Price, 1984).

The effect of various treatment processes on the removal of organics was studied in South Africa on a pilot-plant scale. This pilot plant was modeled after full-scale water reclamation plants intended to produce effluent for unrestricted use. The percent removals of selected toxic organic compounds are given in Table 20 (Van Rensburg, et al., 1981).

Small Volume Direct Reuse Systems

As mentioned previously, NASA has developed small volume systems for direct reuse of wastewater during extended trips into space. Such systems have been found to be quite acceptable for this purpose but the technology used is quite expensive and is not pragmatic for large scale systems.

Table 19
Concentration of Organic Compounds in
Wastewater, and Effluents

	WATER FACTORY 21 Fountain Valley, CA (1985)		23rd AVENUE RAPID INFILTRATION PROJECT Phoenix, AR (1984)	
Parameter	Chlorinated Effluent (ng/L)	Reverse Osmosis Effluent (ng/L)	Chlorinated Secondary Effluent (µg/L	Renovated Water (µg/L)
Carbon Tetrachloride	210	60	0.12	0.07
Trichlorobenzene	70	70	0.38	0.11
Monochlorobenzene	70	70	NA	NA
1,1,1,Trichloroethane	180	50	1.41	0.22
Dichlorobenzene	100	60	2.40+	2.16+
Ethylbenzene	20	40	0.15	0.05
Napthalene	30	60	0.63	0.06
Pentachlorophenol	50	50	0.04	0.04
Phenol	ND	50	NA	NA
Phthalate Esters	1.7 (µg/L)	1.7 (µg/L)	10*	1*
PCB	50	ND	NA	NA
Tetrachloroethylene	710	80	1.69	1.17
Toluene	390	130	NA	NA
Trichloroethylene	60	10	0.39	1.43

*diethyl phthalate.
ND = Not Detectable.
NA = Not Available.
+Ortho.

4. COSTS FOR WASTEWATER AND SLUDGE MANAGEMENT

Introduction

Forecasts of the future of wastewater and sludge management technology must include some mention of the funding that will be required in the future. Without available funding, the future of municipal wastewater treatment and sludge management technology would be bleak indeed. This portion of the paper will deal with the past and future funding associated with municipal wastewater and sludge management technology.

Table 20
Efficacy of a Pilot Plant in Removing Toxic Organic Compounds

		Unit Process									
		High Lime Treatment		Secondary Clarification		Sand Filtration		Chlorination		Activated Carbon Treatment	
Compound	A (µg/L)	B (%)	C (%)	B (%)	C (%)	B (%)	C (%)	B (%)	C (%)	B (%)	C (%)
Lindane	20	0	0	0	0	0	0	80	80	98.3	99.5
Dieldrin	40	12.5	12.5	0	12	22	30	4	33	99	>99.3
Chlorodane	300	18	18	2	20	13	30	32	52	99	>99.5
Demetonmethyl	3000	67	67	0	67	21	74	99.6	>100		
Parathion	4000	19	19	0	19	18	33	99.9	100		
Fenitrolhion	4000	22	22	0	22	21	38	99.9	100		
Fenthion	4000	22	22	0	22	19	37	58	74	100	100
Phenol	600	5	5	30	33	0	33	82	88	98	99.7
Hexachlorobutadiene	100	72	72	97	>99						
Trichlorophenol	400	28	28	0	28	0	29	25	67	97	99
Hexachlorobenzene	80	94	94	5	100						
Acenaphthene	600	27	27	95	97	13	98	75	100		
Fluoranthene	500	79	79	80	96	33	98	22	98		
Pyrene	500	85	85	86	98	20	99	50	100		
Dibutylphthalate	400	87	87	0	53	0	35	28	53	99	100
O-Nitrotoluene	400	24	24	73	80	10	82	18	85	93	100
Tetradecane	200	96	96	89	100						

[a] A = concentration of substances in feed water; B = percentage removal by unit process; C = percentage overall removal.
[b] Increase in dibutylphthalate probably due to plastic piping used in pilot plant.

1972 to 1984

To gain a true perspective of future funding, first the past funding and its impact on water quality must be assessed. In this section of the paper, the funding of wastewater treatment and sludge management technology in the past will be discussed.

Since the passage of the Clean Water Act in 1972 until 1984, the United States has spent more then 56 billion dollars in federal, state, and local funding for wastewater and sludge management technology. Construction grant funding has resulted in the construction or improvement of 3,500 treatment facilities, and these facilities serve a total of 21 million people.

The Association of State and Interstate Water Pollution Control Administrators (1984) has estimated that from 1972 to 1982, out of 758,000 stream miles evaluated, 296,000 miles have maintained the same water quality, 47,000 miles improved, and 11,000 miles have been degraded. Pollutants discharged from municipal plants were reported to have decreased by 46 percent in that period.

Currently, 170 million people send a daily load of 23,000 tons of pollutants to 15,000 municipal sewage treatment facilities. By the year 2000, the sewered population will grow to 248 million people, who will send 41,000 tons per day of pollutants to more than 21,000 municipal sewage treatment facilities.

Water quality improvements in the period between 1972 to 1982 reflect the efforts of states and municipalities to achieve secondary treatment following the enactment of the Clean Water Act. Secondary treatment has resulted in significant improvements in water quality, especially at major rivers near large population centers. However, the USEPA estimates that there are 202 remaining raw sewage discharges and more than 2,800 facilities now operating at insufficient levels of treatment. Obviously, more work needs to be done in the future to minimize the adverse impacts of these uncontrolled discharges.

Future Funding Needs

By the year 2000, the USEPA estimates that the sewered population will increase by 55 percent to 248 million people and the effluent level of pollutants by nine percent. In order to deal with the increase in sewered population, and upgrade and build treatment facilities to resolve existing water pollution problems, more dollars will be required.

In 1978, 1980, and 1982, the USEPA (1985) prepared needs surveys designed to document the expenditures needed to deal with the nation's municipal water pollution control problems through the year 2000. Table 21 presents the results of these needs surveys. As can be seen, the funding amounts are listed according to various categories. There are differences in the estimates for each of the categories from year to year. Most notably, the USEPA, in recent surveys, has downgraded the projected funding needs for advanced treatment since they believe it is not as necessary as originally thought. In general, however, the total funding needs have remained virtually the same for each needs survey ranging from 106.2 to 119.9 billion dollars in 1984 dollars. Based upon these USEPA estimates, the United States will require about 110 billion dollars in order to correct the municipal water pollution problems of the United States through the year 2000.

Based upon the 1982 USEPA needs survey, results shown in Table 21, of the total of 132 billion dollars (1982), 34 billion dollars will be needed for secondary treatment, and only 6.3 billion dollars will be needed for advanced treatment. All of the rest of the needed municipal funding through the year 2000 will be required for sewage collection facilities (sewers and other appurtenances) and control of combined sewer overflows. In other words, relatively little advanced treatment funding, is believed by the USEPA, necessary for meeting future water pollution control needs. Obviously, if the need for advanced wastewater treatment is relatively small through the year 2000, it can be anticipated that the pressure for better and more efficient advanced wastewater treatment would also be small. We can, therefore, expect to see an increased interest in secondary treatment since nearly six times the funding for advanced treatment will be spent on secondary treatment. Further, the cost for advanced treatment per unit volume is usually more expensive than for secondary treatment.

5. TRENDS IN SLUDGE MANAGEMENT TECHNOLOGY

Table 21
Comparison of Year 2000 Needs Surveys Compiled by
United States Environmental Protection Agency

Needs Category	1978 Survey	1980 Survey	1982 Survey
	------------$ Billions------------		
Secondary Treatment	23.1	35.9	34.0
Advanced Treatment	31.4	7.0	6.3
Infiltration/Inflow	3.8	3.2	2.9
Replacement/Rehabilitation	7.7	7.7	5.3
New Collector Sewers	29.9	23.8	23.2
New Interceptor Sewers	29.1	27.5	20.1
Combined Sewer Overflows	40.5	48.0	40.2
Total	165.5	153.1	132.0
Total (1984 Dollars)	106.2	119.9	118.4

REFERENCE: United States Environmental Protection, 1984 Needs Survey - Assessment of Needed Publicly-Owned Wastewater Treatment Facilities in the United States - Final Draft, USEPA, February 10, 1985.

Introduction

Sludge management is a difficult and expensive problem for wastewater treatment agencies. As the level of wastewater treatment increases, sludge quantities invariably increase also. Sludge management technology is, therefore, just as important as wastewater treatment technology and has received considerable attention in the past, and will undoubtedly receive additional attention in the future.

History of Sludge Management

As sewage treatment advanced, the Imhoff tank, the trickling filter, and the activated sludge process became accepted. These processes produced sludge solids in copious quantities. In the early years of treatment facilities, sludge lagooning was often the method of choice. In some cases, municipal sewage treatment plants air-dried their sludge and gave it to farmers.

Table 22 lists some of the important dates in the development of municipal sludge management processes. It is only since 1900 that modern sludge management processes were put into practice.

Many of the sludge management technologies practiced currently were developed in the first three decades of this century and remain in wide-spread use today. Anaerobic digestion and centrifugation remain as viable options for municipal sludge management today despite the fact that these processes were developed over 60 years ago.

Many municipalities have moved away from older energy intensive operations to operations with low energy utilization such as land application. Incineration in the United States has only found favor for sludges which have very high organic content and

Table 22

Important Dates in Municipal Sludge Management

1904	Imhoff Tank Invented in Germany
1908	First Sludge Centrifuge Used in Germany
1911	First Imhoff Tank Built in the United States at Madison, New Jersey
1912	First Separate Sludge Digestion in Baltimore, MD.
1921	First Vacuum Filters at Milwaukee, Wisconsin
1921	First Continuous Flow Sludge Centrifuge Used in the United States at Milwaukee, Wisconsin
1928	Use of Ferric Chloride for Vacuum Filtration is Discovered by the District
1928	First Heated Digester Installed at Antigo, Wisconsin
1932	First Large Scale Sludge Dewatering and Incineration Plant at the District's West-Southwest STW
1957	First Dissolved Air Flotation Installation in the United States at Nassau County, New York
1958	Zimmerman Wet Air Oxidation Process for Sludge Disposal Invented

in situations where land is scarce. The key to incineration, and for that matter, any sludge management operation, is removal of water from the sludge. Without a high solids content, sludge transportation costs soar, and auxiliary fuel costs for incineration and sludge application to land costs become prohibitive.

The sanitary engineering profession has learned over the years that one of the keys to efficient and economical sludge management is volume reduction through the use of sludge dewatering processes. The first mechanical dewatering process to gain wide-spread use was the vacuum filter. Vacuum filters were installed in the United States, beginning in the 1920s, and their popularity continued through the 1960s. The District installed vacuum filters at its Calumet facility in 1935 and its West-Southwest facility in 1939.

In the 1960s, centrifuges began to be successfully used for sewage dewatering with the advent of the continuous flow horizontal bowl design. In the 1970s, belt filter presses were successfully employed at many treatment facilities.

In 1973, the District began to explore the possibility of mechanically dewatering its sludge in order to allow it to be handled as a solid for land application uses. Desk top studies indicated that vacuum filters, centrifuges, and belt filter presses had the greatest potential. Based upon this review, extensive pilot-scale studies of these three types of dewatering devices were conducted at District sewage treatment facilities from 1974 through 1976. The performance of each unit was evaluated with respect to cake solids, solids capture, and chemical conditioning agents. The results of these field tests indicated that horizontal bowl centrifuges would best fit the District's needs and three treatment facilities now have these centrifuges.

Future of Sludge Management

It is difficult to see clear trends in sludge management over the past 20 years. Certainly, composting has gained in favor as a sludge processing method, but ultimately this process merely makes the sludge more acceptable for general use, a long standing practice by many municipalities.

The USEPA (1977) conducted a survey of the municipal sludge handling practices of members of the Association of Metropolitan Sewerage Agencies. This survey included 75 individual agencies handling over 2,700 dry tons of sludge per day or about 25 percent of the national total. Data from this survey are shown in Table 23.

Sale and give-away programs accounted for the largest amount of sludge in this survey. Ocean disposal was the option used for 30 percent of the sludge being disposed. Landfilling accounted for 17 percent whereas 15 percent was applied to land and less than four percent was lagooned.

Table 24 lists predictions of future trends in municipal sludge management through the year 2000+. These predictions were

Table 23
Sludge Management Options Used by Municipal Agencies

	Treatment Facilities	Dry Tons	Percent of Total
.e and Giveaway	8	953	34.5
:an Disposal	19	833	30.2
ndfill	27	461	16.7
nd Application	14	402	14.6
.goons	7	109	4.0
Total	75	2,758	100

:FERENCE: United States Environmental Protection Agency, Sludge Handling and Disposal Practices at Selected Municipal Wastewater Treatment Plants, USEPA 430/0/77/077, 1977.

Table 24
Future Trends in Municipal Sludge Management
Through the Year 2000+

1. Lagooning as a final sludge disposal method will be used infrequently by municipal agencies.
2. Ocean disposal and sale and give-away programs will be strongly influenced by Federal Regulation.
3. Dewatering processes will continue to be improved.
4. Digestion will continue to be used.
5. Incineration will be influenced by:

   A. Fuel Costs
   B. Maintenance Costs
   C. Sludge Dewatering Technology

based upon the current trend in municipal sludge management and the regulatory forces at work today.

The USEPA has taken a less than enthusiastic stance in the past ten years in support of the ocean disposal of sludge and has issued permits for such disposal for only a few agencies. This method of sludge disposal is, however, viable and can be practiced with adequate environmental protection and safeguards.

It is also clear that lagooning as a final disposal method for municipal sewage sludge will be virtually nonexistent in the near future. The USEPA survey in Table 23 shows that this practice is used now for four percent of the sludges produced. Certainly, with the increasing competition for land, this practice can only remain at its present low usage or diminish.

The future of the other various options available to sludge management agencies will be influenced mostly by future regulatory action and public attitudes. For example, there are no current USEPA regulations governing the sale and giveaway of sludge products. When and if these regulations are promulgated, they will have a profound effect upon existing sale and giveaway programs.

Land application of sludge has a good future in the United States provided municipalities can continue to reduce the costs associated with this operation. It is important that municipalities find ways to reduce the water content of sludges to reduce handling, transportation, and application costs. Also, sites for sludge application must be found in relatively close proximity to wastewater treatment sites.

The dewatering of sewage sludges will remain a difficult problem for most municipalities. Centrifuges, vacuum filters, belt filter presses, dissolved air flotation, and filter presses are all used today, but capital and operating costs are high and results are often unsatisfactory. A cheap, reliable and effective method for producing a well dewatered sludge would be a boon to the sanitary engineering profession. Hopefully, the future will produce such a dewatering system.

5. FUTURE TRENDS IN EFFLUENT QUALITY AND WASTEWATER TREATMENT TECHNOLOGY

Introduction

This paper has put considerable effort into a discussion of the history of wastewater treatment technology. This history inevitably leads to conclusions regarding the changes in effluent quality that have occurred. Table 25 shows a summary of the effluent quality experienced at municipal wastewater treatment facilities over the past 100 years. Obviously, there has been an improvement in effluent quality as we approach the turn of the next century. However, the last 15 years demonstrate that the move to levels of wastewater treatment higher than secondary may not be necessary, practical, or affordable. It would therefore appear that many, if not most, municipal treatment facilities will not

Table 25
Municipal Wastewater Treatment Plant Effluent Quality

Period	EFFLUENT QUALITY
1 (Before 1900 AD)	-Mainly raw sewage quality ·Not many treatment facilities
2 (1900-1930)	-Primary effluent quality in many cases ·Some cleaner effluents due to use of intermittent sand filtration, trickling filters, broad irrigation and a few activated sludge plants
3 (1930-1970)	-Secondary effluent quality ·Intensive efforts to build secondary treatment facilities ·Industrial waste treatment activity intensification ·Development of AWT technology and water reuse technology ·A few tertiary treatment facilities
4 (1970-1985)	-Secondary effluent quality ·Increased efforts to meet secondary effluent quality ·Pretreatment regulations ·RCRA regulations ·Efforts to reduce toxic organics and metals in effluents ·Closed loop technology existence
5 (1985-2000 AD and beyond)	-Secondary effluent quality mostly ·Higher removal of toxic organics and heavy metals ·Increased compliance with effluent standards ·Increased water reuse technology in industries and municipalities of arid and semi-arid regions ·Closed loop technology applications rise ·More exacting site-specific criteria for funding and construction of AWT facilities

be using tertiary treatment by the year 2000+.

In general, it appears that future effluent quality will be dictated by several factors and issues. Among these are:

1. The enforcement of pretreatment regulations for discharging industrial waste into receiving waters or publicly owned treatment works (POTWs).

2. The proper management and disposal of hazardous wastes.

3. Increased demand for additional quantities of water and the increased interest in treated wastewater for subsequent use.

4. Availability of funding to build new treatment facilities, or for the upgrading and expansion of existing facilities.

5. Improved operation and maintenance of existing wastewater treatment facilities.

6. Controlling the pollution from combined sewer overflows and nonpoint sources.

Using these factors and assuming no radical changes in the environmental control forces already at work, future trends in wastewater treatment technology have been formulated and these are contained in Table 26. In the sections that follow each of the major topics covered in Table 26 are discussed.

Industrial Pretreatment

The enforcement of pretreatment regulations will undoubtedly reduce the loads of various pollutants to wastewater treatment facilities. In turn, this should result in fewer facility upsets and better effluent quality. Overall, the quality of effluents, particularly with respect to the priority pollutants, will be enhanced as these pollutants will be monitored and controlled more closely in the future.

Since the distribution of the various priority pollutants (both inorganic and organic) between the solid and aqueous phases is a function of their partition coefficients, a decrease in their concentration in the influent will result in a proportionate decrease in the liquid and solid phases of the effluent and sludge.

Strict compliance with the Resource Conservation and Recovery Act will ostensibly stop the disposal of hazardous wastes into POTWs. With compliance of these regulations by industries which produce such wastes, one can expect an overall improvement in the quality of POTW effluents.

Water Reuse

According to one estimate, the extent of water reuse in the United States amounts to less than two percent of the total water consumed. Barring unforeseen circumstances, it is unlikely that, overall, reclaimed water will be used to a significantly higher degree by the year 2000. It is quite possible, however, that wastewater processing for subsequent reuse will increase in certain areas of the country, particularly the arid zones. It is in

Table 26
Future Trends in Wastewater Treatment Technology
Through 2000+

Water Reuse

1. Reuse, overall, will not increase significantly in the Great Lakes Region
2. It will increase in Western and Southwestern regions
3. It may be economical in isolated cases to practice direct reuse

Industrial Pretreatment

1. Reduction in quantity and contaminant levels of wastes discharged to receiving waterways and POTWs
   a. Pretreatment
   b. RCRA
   c. TOSCA
   d. Increased by-product recovery
   e. Increase in direct reuse
   f. Increased use of Closed Loop technology

Biological Waste Treatment

1. Genetic engineering applications will increase
2. Activated sludge will predominate
3. Process optimization will continue
4. Automation will increase
5. Energy conservation and recovery will increase
6. Use of anaerobic processes will increase

Control of Combined Sewer Overflows (CSOs)

1. Increased interest in CSO control
2. CSO control vs. advanced waste treatment
3. Improved CSO control technology

Future Technology

1. Genetic Engineering
   a. Flocculation of sludges
   b. Ammonia removal by biological oxidation
   c. In-situ detoxification of soils
2. Automation
   a. Control
   b. Maintenance
   c. Optimization
3. Treatment Processes
   a. RBCs
   b. Physical-chemical treatment
   c. Molecular oxygen
   d. Lagooning
4. Closed-Loop Treatment for Small-Scale Systems

these areas that more and more attention will be given to advanced wastewater treatment. In most areas, however, where water is available in adequate quantities, it is unlikely that water reuse will dramatically increase. It is also unlikely that wastewater will be renovated to routinely produce effluents of a quality analogous to that of drinking water to any significant extent in the future. The surface and groundwater supplies currently available will probably meet future demand and the public will continue to be adverse to the use of renovated water for direct potable use due to aesthetic considerations. Industries will continue to seek ways to conserve water by adopting new production practices, and employ water reuse and recycling without decreasing productivity (Prakasam, 1975).

Water reuse is an age old phenomenon. Since it has not yet become a major practice, it is unlikely that it will be a major national pursuit by the year 2000. Hence, although advanced wastewater treatment technology is currently available for the purpose of producing high quality effluents, it is unlikely that such a technology will be implemented for producing a high quality effluent for subsequent reuse, except in specific cases where it is justified.

Biological Wastewater Treatment

If we are guided by history, we know that we have not yet really accepted wastewater treatment technologies beyond biological waste treatment. The 70-year old activated sludge process is still unrivaled, and is the technology of choice for many large sewage treatment facilities. Thus, it is unlikely that in the next 15 years there will be a dramatic shift in wastewater treatment technology, particularly when no significant breakthroughs have occurred in the past 20 years. No plans are on the drawing board today to build sewage treatment facilities in significant numbers using technologies other than the proven biological waste treatment systems which have been available for the past 75 years.

Funding and Effluent Quality

As the funding level for the construction grants program becomes limited, the construction of new and expanded facilities will proceed at a slower pace than in the last decade. However, an improvement in the operation and maintenance practices of the facilities already built through increased training of operating personnel, should bring about an improvement in effluent quality and a considerably higher degree of compliance with the effluent limits set by the state and federal environmental protection agencies. Also, one can expect increased investment in instrumentation in the future at various treatment facilities to facilitate better control and monitoring of unit processes, thus enabling better effluent quality.

Control of Combined Sewer Overflows

The control of combined sewer overflows (CSO) and their treatment should bring about an improvement in the quality of effluents discharged from municipal wastewater treatment facilities under wet weather conditions. The handling of CSOs during wet weather is typified by the District's Tunnel and Reservoir Plan (TARP). As we progress, increased attention will be given to the control of CSOs. Hence, the quality of effluents discharged from various treatment facilities can be expected to be more uniform irrespective of the seasons of the year. Also, the amount of CSOs which discharge directly to waterways will diminish.

Future Wastewater Treatment Technology

Existing primary and secondary sewage treatment processes will continue to dominate the scene for the next few decades, as billions of dollars have already been invested. Due to a lack of significant breakthroughs as yet in other technologies, it is reasonable to expect that conventional sewage treatment processes such as activated sludge and trickling filters will continue to be built. Technologies such as rotating biological discs and other forms of attached growth systems may be built in small communities, but not in large communities because of the lack of a proven record of their performance on a large scale.

Physical-chemical processes will continue to find application in polishing secondary effluents. However, the optimistic prediction made by some investigators in the 1960s that they will replace biological waste treatment, has not been realized and it is unlikely that by the year 2000, existing biological waste treatment facilities will be replaced by physical-chemical treatment. Physical-chemical treatment does not produce effluents significantly better than the activated sludge process and is more expensive and complicated to operate. Despite claims by their protogonists and much research, physical-chemical treatment has not been adopted in significant numbers by municipal agencies. There is no reason to believe that the current limited use of physical-chemical treatment processes by municipal agencies will improve by the year 2000.

The era of computerization will definitely have an impact on the way treatment facilities will be operated and maintained in the future. Skilled operators trained in computer operation of wastewater treatment facilities, will have a positive impact on the quality of the effluents that will be discharged.

The new fields of biotechnology and genetic engineering will come of age in the twenty-first century and the concepts developed in the fields of pharmaceutical, fermentation, and biomedical sciences will undoubtedly be applied to the field of wastewater treatment technology.

6. CONCLUSIONS

This paper has attempted to predict what the effluent quality and wastewater treatment and sludge management technology will be in the year 2000+. Basically, the history of wastewater and sludge management technology over the past 150 years was drawn upon extensively as a guide to helping us in making these predictions. Based upon this history and other factors, the following predictions and conclusions were drawn.

General

Future technology development is likely to be driven primarily by regulatory forces.

While issues of public health concern will certainly influence this development, it is unlikely that catastrophic public health episodes will occur because of a lack of wastewater treatment technology. Hence, public health catastrophies are not likely to be the primary driving forces behind significant advances in wastewater treatment technology in the future.

Water Reuse

The technology exists today for direct reuse of treated wastewater, but public attitudes regarding aesthetics will continue to severely limit this practice in the future.

Although reuse of treated wastewater represents only two percent of water use in the United States today, it is unlikely that this percentage will rapidly increase in the future except in the more arid western regions of the country.

Biological Treatment

The wide spectrum of existing biological wastewater treatment systems will continue to play a significant role in future municipal wastewater treatment programs.

The 70-year old activated-sludge process is still unrivaled, and is the technology of choice now and into the future for most municipal sewage treatment facilities.

Optimization using genetic engineering and biotechnology to reduce cost and improve biological process efficiency should be expected.

Physical-Chemical Treatment

Interest in physical-chemical treatment technology for municipal systems will diminish.

Sludge Management

Ocean disposal of sludge will remain a viable option, and will continue under more effective management schemes, including improved stabilization and more intensive and extensive operational and long-term monitoring.

Lagooning as an ultimate sludge disposal system represents a very small percentage of existing sludge management operations and will probably continue on a downward trend through the year 2000 and beyond.

The dewatering of sludge still remains a difficult problem for all sludge management systems. A cheap, reliable and effective method for providing a well dewatered sludge could be developed by the year 2000. Genetic engineering is likely to be used for improving process efficiency.

The role of privatization in sludge management and ultimate disposal will increase significantly.

Operation and Maintenance

There will be an emphasis on improved maintenance practices and instrumentation, more effective monitoring and process control, and more efficient operation of wastewater treatment facilities in the future.

More energy-efficient prime movers and power generating systems will be developed and utilized.

CSOs

Control of CSOs will increase in the future as the demand for better water quality focuses on this environmental problem.

Improved CSO control technology will emerge.

Effluent and Waterway Quality

The effluent quality of the future will continue to improve, but will not routinely equal drinking water quality except in special cases; e.g., NASA space activities, or where local demand for reuse force the issue.

Improvement will occur in the inorganic, organic, and volatile components of effluents.

Environmental recovery/repair will continue both with respect to water column chemistry and aquatic biota, but at a slower rate for the latter.

Even in the absence of new technology between 1985 and 2000+, receiving water quality will continue to improve by the efficient operation and maintenance of existing facilities.

Advanced treatment; i.e., beyond improved secondary treatment, will not become common practice because of costs and the ability of receiving waters to improve in quality with secondary treated effluents.

Costs

To meet the wastewater treatment and sludge management needs of the year 2000, the USEPA estimates that about 110 billion dollars must be expended in the next 15 years. Since nearly 56 billion dollars have been expended by the USEPA since 1972, it appears unlikely that the estimated environmental needs of the United States will be completely fulfilled by the year 2000.

7. BIBLIOGRAPHY

Argo, D.G., "Water Reuse: Where are we headed?" Environmental Science and Technology, 1985.

Association of State and Interstate Water Pollution Control Administrators, "States Evaluation of Progress," In America's Clean Water, 1984.

Bouwer, H., R.C. Price, J.C. Lance, and R.G. Gilbert, "Rapid infiltration Research at Flushing Meadows Project, Arizona," Journal Water Pollution Control Federation, 52, 1980.

Bouwer, H., and R.C. Price, "Renovation of Wastewater at the 23rd Avenue Rapid Infiltration Project," Journal Water Pollution Control Federation, 56, 1984.

Cain, Louis P., Sanitation Strategy for a Lakefront Metropolis - The Case for Chicago, Northern Illinois University Press, Dekalb, Illinois, 1978.

Donovan, J.F., and J.E. Bates, Guidelines for Water Reuse, USEPA Report 600/8-80-036, 1980.

United States Environmental Protection Agency, 1984 Needs Survey - Assessment of Needed Publicly Owned Wastewater Treatment Facilities in the United States - Final Draft, USEPA, February 10, 1985.

United States Environmental Protection Agency, Sludge Handling and Disposal Practices at Selected Municipal Wastewater Treatment Plants, USEPA 430-0-77-077, 1977.

McJunkin, F.E., Water and Human Health, U.S. Agency for International Development, Washington, D.C., 1982.

Metcalf and Eddy, Inc., Wastewater Engineering Treatment, Disposal, Reuse, McGraw-Hill, Inc., New York, 1979.

Prakasam, T.B.S., "Processing of Wastewater for Further use - Some Concepts and Considerations," Proceedings 2nd National Conference on Complete Water Reuse, American Institute Chemical Engineers and USEPA Technology Transfer, 1975.

Rogers, S.E., and W.E. Laner, "Start-up and Operation of the Denver Potable Water Reuse Demonstration Plant," Paper presented at the Annual Conference of Water Pollution Control Federation, New Orleans, Louisiana, 1984.

Schmidt, C.J., and E.V. Clements, Demonstrated Technology and Research Needs for Reuse of Municipal Wastewater, USEPA Report 670/2-75-603, 1975.

Stander, G.J., "Reuse of Water for Municipal Purposes," In Water Renovation and Reuse, Academic Press, New York, 1977.

Van Rensburg, J.F.J., A.J. Hassett, and S.J. Theron, "Control and Measurement of Organic Micro-pollutants in South Africa's Water Reclamation Plants," In Chemistry in Water Reuse, Vol. I, Ann Arbor Science, Ann Arbor, 1981.

Van Vuuren, L.R.J., A.J. Clayton, and D.C. Van der Post, "Current Status of Water Reclamation at Windhoek," Journal Water Pollution Control Federation, 52, 1980.

The Effluent of the Future: Trends in Quality and Quantity

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INTRODUCTION

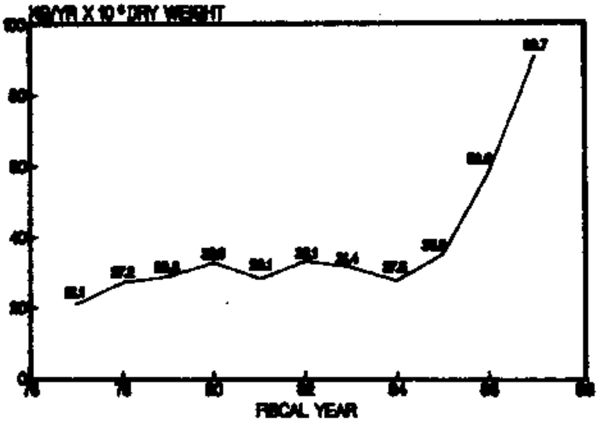
This paper should probably be the last in the program, rather than the first, for the speakers who follow me will discuss technical and institutional factors which will have a great deal to do with determining the characteristics of the effluent of the future. As a starting point for the series, however, I am going to take the somewhat conservative view that effluents in the year 2000 -- both treated wastewater and sludge -- can be fairly well described by extending presently obvious trends. Among the influences which will impede any dramatic departures in either direction from these trends are today's regulatory climate, the magnitude of the investment already made in improved wastewater treatment, diminishing returns and spiralling operating costs from further treatment improvements, and public resistance to anything which might be perceived as backsliding from achievements in environmental quality. The City of Philadelphia's experience over the preceeding 13 years in complying with the Clean Water Act can thus serve as the basis for my vision of what we can expect during the next 15.

WASTEWATER TREATMENT IN PHILADELPHIA

Philadelphia Water Department provides wastewater collection and treatment services to approximately 2.3 million people. In 1975, 1 136 x 10^3 m^3/d (300 mgd) received primary treatment at two plants, and 719 x 10^3 m^3/d (190 mgd) received treatment intermediate between primary and secondary at a third. By 1980, one of the primary plants had been upgraded to provide high-quality (90 percent removal) secondary treatment to 795 x 10^3 m^3/d (210 mgd). At the end of 1985, most of a $900 million expansion and upgrading program will have been completed, affording full secondary treatment capacity for 2 044 x 10^3 m^3/d (540 mgd). Discharge permits require effluent characteristics typical of secondary treatment (30 mg/l BOD_5 and suspended solids, and 200/100 ml fecal coliform), but each plant is also assigned a carbonaceous BOD wasteload allocation which represents a practical limitation on BOD_5 of approximately 20 mg/l.

With the improvement in wastewater treatment comes of course a substantial increase in sludge production. Daily production on a dry weight basis has risen from 60 x 10^3 kg/d to a current level of 160 x 10^3 kg/d. By 1987, production will exceed 270 x 10^3 kg/d. Figure 1 shows total annual sludge production for the 10 years during which the plant upgrading will have been in progress.

FIGURE 1
TOTAL SLUDGE PRODUCTION



In gross terms, I would not expect to see major changes in wastewater flow, effluent characteristics or sludge production in Philadelphia between 1987 and the year 2000. Population has been declining in most parts of the Water Department's 595 km^2 service area. Domestic and industrial water consumption has declined in response to recurring drought conditions and sharp increases in treatment costs, and an aggressive leak detection program in the water system has reduced infiltration to the collector system. Wastewater volume has consequently decreased. Effluent quality has improved, but the cost of the improvement (see Figure 2) raises real questions about the ability of customers to pay and the ability of the utility to allocate resources to other pressing needs such as water distribution system maintenance. Receiving water quality could be further improved by advanced treatment, but at a cost which would probably outweigh any benefits. If further major capital expenditures are to be made on managing Philadelphia's water resources, benefit-cost ratios would be much more favorable in the area of water supply. These all represent practical barriers to any program for higher levels of wastewater treatment. At the same time, I find it difficult to imagine any program which would diminish effluent quality, unless it be seasonal rather than year-round disinfection, which would not adversely affect uses of the Delaware estuary.

FIGURE 2
COST OF WASTEWATER TREATMENT AND DISPOSAL

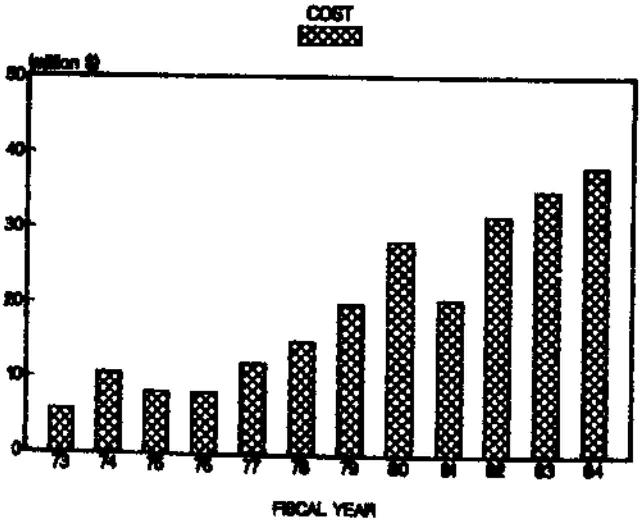
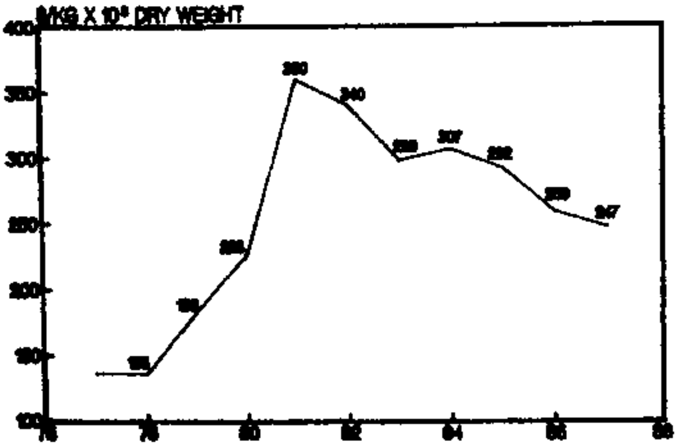


FIGURE 3
UNIT COST OF SLUDGE MANAGEMENT



Sludge production should also remain stable, although the methods of disposal or utilization will probably continue to evolve. The City moved from ocean dispersal in 1976 to a completely land-based program by 1981. That program's principal components today are strip-mine reclamation, use on agricultural land for animal feed productions, recreational land application, and marketing to landscapers and commercial plant growers. By 1988, Philadelphia expects to be marketing more than one-half of its composted sludge products. Marketing is the most promising means to significantly reduce the high cost of sludge management, shown in Figure 3 excluding the cost of dewatering.

WASTEWATER TREATMENT FROM A NATIONAL PERSPECTIVE

In 1982, approximately 78 percent of the discharge from publicly-owned treatment works received secondary treatment. Only 2.6 percent was undergoing some form of advanced waste treatment (U.S. EPA, 1983). The most recent national survey of wastewater treatment needs contains an estimate that $23.0 billion is needed to bring the remaining publicly-owned treatment plants into compliance with requirements of the Clean Water Act. In contrast, advanced waste treatment needs total only $4.1 billion (U.S. EPA, in press). One can conclude from this, first, that advanced waste treatment systems, while significant in managing the quality of certain streams, will continue to account for a relatively insignificant fraction of total effluent. Second, the country's utilities will be hard pressed to accumulate the financial resources necessary to provide basic secondary treatment, regardless of the 1988 statutory deadline, by the year 2000. In fact, EPA predicts that, in that year, 84 percent of effluent will be from secondary plants and only 3 percent from tertiary systems (U.S. EPA, 1983).

Will waivers of secondary treatment requirements relieve some of the financial burdens and result in retention of a significant amount of primary treatment? To date, only 10 of the 208 301(h) waiver applications received have been approved, while 45 have been rejected and 31 withdrawn. There are 122 still to be processed. It is clear that the present regulatory and legislative climate is not favorable for waivers, particularly where the receiving water is estuarine rather than marine. This may change; however, the burden of proof that there will be no adverse impact rests with the utility, and the task of proving it is onerous. EPA projects that only 3.4 percent of effluent will be receiving less than secondary treatment in 2000 (U.S. EPA, 1983).

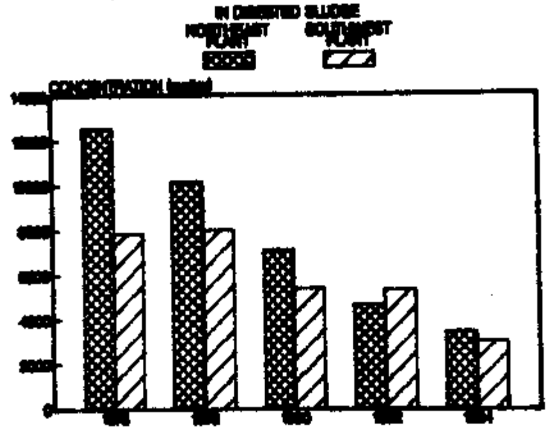
Nationwide predictions of effluent quantity are primarily dependent on predictions of population growth. The so-called sunbelt cities will be generating more effluent, whereas cities which are not growing will not. There are, though, some factors which will eventually modify this relationship. They are evident in older cities now and will be, I believe, even in high-growth regions by 2000. First among them is the cost of water. We have seen in Philadelphia reductions in industrial water use, and even in residential consumption, as wastewater treatment charges have begun to reflect the full cost of compliance with the Clean Water Act. Second is the availability of water. The increasing cost of supplying water, especially in naturally water-short areas, will result in changed water use patterns. Treatment to remove trace organic compounds, which is quite expensive, will reinforce this trend, and we believe such treatment will be required at some point as a result of regulations under the Safe Drinking Water Act. Finally, as the backlog of infrastructure rehabilitation work is attacked, in older utilities, part of the result will be reduced leakage into the sewer systems. The combined effects of these factors will be reduced per-capita water consumption and greater reuse and recycle of effluents from wastewater treatment plants themselves and from their tributary industrial customers. EPA estimates that total effluent was approximately 99 0 x 10^6 m^3/d (26 402 mgd) in 1982 and will reach 155 5 x 10^6 m^3/d (41 193 mgd) by 2000 (U.S. EPA, 1983).

The trend in sludge production is clearer; it will increase as treatment is upgraded and as population increases. EPA estimates that 6 2 x 10^9 kg of sludge are produced annually today and that the amount will double by 2000 (U.S. EPA, 1984). The quality of the material in gross terms will not be much changed from today. However, there will be changes in terms of potentially toxic constituents, as Philadelphia's experience shows.

FIGURE 4

AGGREGATE HEAVY METAL CONCENTRATION

IN DIGESTED SLUDGE



CHEMICAL CHARACTERISTICS OF SLUDGE AND EFFLUENT

In Philadelphia, the transition from ocean disposal to land application of sludge provided the impetus for implementing an industrial pretreatment program in 1977, on a more rapid schedule than EPA has followed in issuing categorical pretreatment standards. Figure 4 shows, in the aggregate, the decline in concentrations of heavy metals in sludge in recent years as a result of the pretreatment legislation enacted by the City. Figures 5 and 6 show the changes for six metals in more detail and project the further reductions in concentrations of some which can be expected.

FIGURE 5

PHILADELPHIA WATER DEPARTMENT NORTHEAST PLANT

METAL CONCENTRATION IN SLUDGE (Dry Wt.

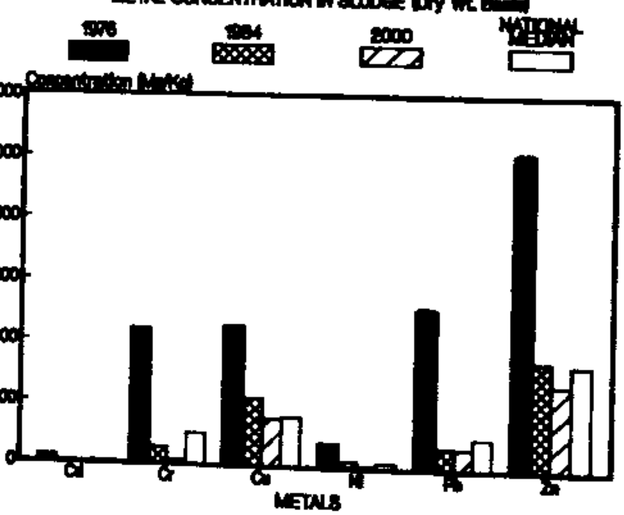
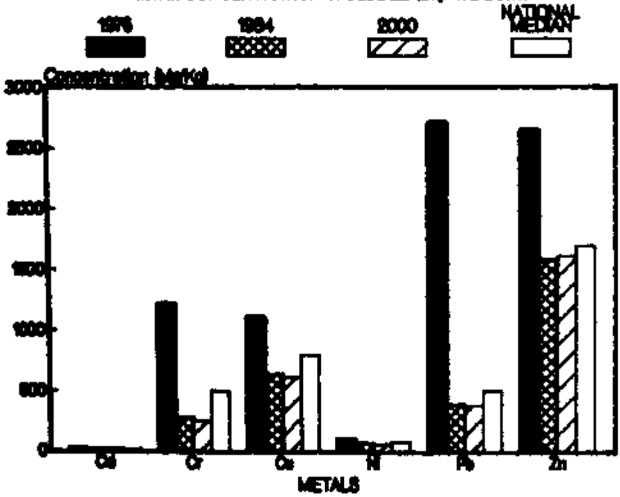


FIGURE 6

PHILADELPHIA WATER DEPARTMENT SOUTHWEST PLANT

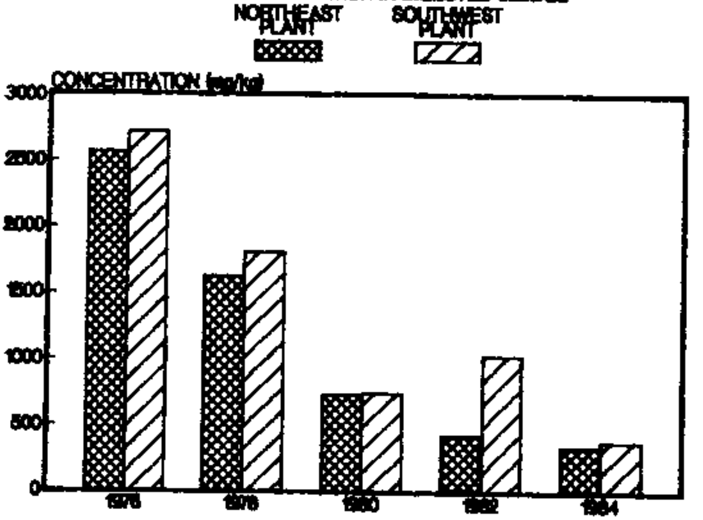
METAL CONCENTRATION IN SLUDGE (Dry Wt. Basis)



Compliance with Federal categorical pretreatment ındards now being issued will produce some further luction, but in general, I would expect the metal ıcentrations to remain at about current levels for ı next 15 years. Sludges exhibiting these charac- ristics are quite suitable for a variety of uses on ıd, which is the objective we have had to meet. ere are some interesting possibilities, however, ich are beyond the control of wastewater utilities, : which may result in further improvements. First, :ause zinc is the metal which limits the application :e for Philadelphia sludges, we are exploring the use water distribution system corrosion inhibitors other ın zinc compounds. (Clearly, there is here an advan- ;e to being a combined water and wastewater utility; ıarate utilities would have more difficulty imple- ıting such a change.) Second, trends toward :reased use of plastic piping may lead to reduction

FIGURE 7

LEAD CONCENTRATION IN DIGESTED SLUDGE



in the copper content of sludge, which is affected by leaching from plumbing. Third, as Figure 7 shows, there has been a gradual decrease in Philadelphia sludge lead concentration since 1976. This is apparently related to more common use of unleaded motor vehicle fuels in an area served by combined sewers. EPA's recent decision to eliminate leaded fuels should cause this trend to continue.

Philadelphia sludge has been subjected to EPA toxicity analysis since 1981 and has not exceeded or even approached maximum permissible concentrations for metals and pesticides. In addition, it is periodically screened for priority pollutants. Fourteen organic compounds have been identified in our composted sludge, but only three -- toluene, dichloroethane, and phenol -- are consistently present in significant amounts. Their major sources are known, and efforts of those sources in response to City actions over the last four years have resulted in 90 percent reductions of toluene and dichloroethane. Phenols, because they are discharged from a larger number of smaller sources, have been more difficult to manage. Pretreatment programs nationwide should be able to achieve results similar to these.

As far as effluent quality is concerned, Philadelphia's discharge permits currently include limitations for eight metals which are virtually never exceeded. We are currently in the midst of our permit renewal process, however, and believe that new permits nationwide will be written based on the existence of categorical pretreatment standards for 25 industries. We also would not be surprised to see all 126 priority pollutants incorporated in the permits in some fashion, perhaps with requirments to monitor for those known to be present. At a minimum, this will produce a great deal of data regarding effluent quality which can be used in future permit renewals. The burden of compliance with limitations for such materials will undoubtedly be passed on to industry by the wastewater utilities. It appears that wastewater utilities are not going to be able to rely heavily on removal credits to ease the burden on industrial customers, especially for volatile compounds, in view of recent evidence suggesting that municipal treatment plants can be significant sources of volatiles in the atmosphere. Removal of organics will be chiefly an industrial problem, as will disposal of any resulting solid or liquid residuals.

Our monitoring of organics in plant effluents to date indicates the presence of:

Chloroform	Benzine
1,1,2, Trichloroethylene	Cresols
Tetrachloroethylene	Toluene
1, 2, Dichloroethane	Phenol
Dichlorobenzene	Acrylonitrile

This will be highly variable from one service area to another, depending on the mix of industrial and commercial customers.

CONCLUSION

To summarize, in 2000 there will be approximately 155 9 x 10^6 m^3/d of POTW effluent to be dealt with. Treatment of that wastewater will produce 12 x 10^9 kg of sewage sludge annually. The effluent will be, for the most part, from secondary treatment plants. The sludge will be generally suitable for land application unless pretreatment regulations are radically altered.

REFERENCES

U.S. Environmental Protection Agency, The 1982 Needs Survey: Summaries of Technical Data (EPA/43019-83-002), Washington, 1983.

U.S. Environmental Protection Agency, Environmental Regulations and Technology: Use and Disposal of Municipal Wastewater Sludge (EPA 625/10-84-003), Washington, 1984.

U.S. Environmental Protection Agency, The 1984 Needs Survey, in press.

Innovative Management of Residuals – Research Needs

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ABSTRACT

The past twenty years since passage of the "Clean Water Act" have witnessed a vast increase in the amounts of residual sludges that must be managed. Passage of the "Clean Air" and "Ocean Dumping" Acts effectively foreclosed the indiscriminate use of two of the three earthly media for final placement of sludges leaving land as the only viable alternative. While the practice of land application of wastes dates back to antiquity, the sheer volume and nature of the wastewaters and sludges resulting from full implementation of water and air pollution control legislation is without historic precedent. Research by the U.S. Environmental Protection Agency (EPA) in cooperation with the Department of Agriculture has resulted in new methods for composting of sludges prior to placement on land. Research on methods to disinfect sludges for protection of public health, which was thought would be placed in added jeopardy by the massive increases in the amounts of sludges that were destined for land placement, was supported by EPA in cooperation with the Department of Energy involved the use of gamma radiation derived from sources made artificially radioactive such as Cobalt-60, and Cesium 137. As part of its Program of Research Applied to National Needs, the National Science Foundation (NSF) provided support for research on the potential applicability of energized electrons as the source of ionizing radiation for disinfection of wastewaters and sludges. This paper presents a concept that emerged from research supported by NSF as a step toward a more fundamental approach to the management of sludges than is now practiced or possible by simple refinement of known processes. Research needs relating to this concept are suggested.

Opinions expressed in this paper do not represent an official position of the National Science Foundation.

INTRODUCTION

Evidence from the earliest records through the 16th Century indicates that the quest for pure water began in prehistoric times (1). Processes for water purification mentioned in literature and inferred from other sources preceding the 18th Century include contact with copper, exposure to sunlight, filtration through charcoal, filtration through sand, boiling, and sedimentation. Sedimentation and filtration were the large-scale processes of choice for treatment of obviously polluted water during the 17th and 18th Centuries.

Advances in microbiology during the late 1800's proved the existence of linkages between public health and sanitation. The germ theory of disease emerged as a byproduct of Pasteur's classical research on fermentation. Lister applied this new knowledge by introducing antiseptic procedures in surgery and Koch systematized experimental procedures for scientifically proving the relationships between diseases and causal bacteria (2).

The relationships between ingestion of contaminated water and incidence of gastrointestinal diseases were firmly established by the turn of the century. Occasional outbreaks now of typhoid fever, cholera and infectious hepatitis serve to remind us of those linkages. Disinfection of public water supplies by use of chlorine, although of research interest earlier, came into use rapidly following its initial adoption in 1908 at the Boonton Reservoir of the Jersey City Water Works (1).

The practice of primary treatment of wastewaters by screening and sedimentation to remove suspended solids was followed by adoption of chlorination for disinfection of the effluents to protect the downstream users of water into which effluents were discharged. When it became evident that removal of suspended solids and disinfection were no longer adequate to protect receiving waters from unacceptable degredation, processes began to be introduced to convert the colloidal and soluble organic substances remaining in the primary effluents into settleable solids.

Originally, following primary treatment which was sometimes enhanced by use of chemical coagulants, wastewaters were spread over beds of crushed stone called trickling filters. Biologically active films that were formed on the surfaces of the stones and nurtured by nutrients in the wastewaters simulated the purification process that occurred naturally in lakes and streams. However, convergence of problems caused by population growth and its concentration, and by the continued advancement of knowledge about relationships between contaminated water and illnesses forced a search for an alternative process to trickling filtration. The almost desperate need for an alternative process is suggested by the fact that only eleven years after demonstration that the activated sludge process worked, more than twenty full-scale wastewater treatment plants based on this process were operating in cities such as Chicago, Houston, Indianapolis and Milwaukee in the

United States; Birmingham, England; Glasgow, Scotland; and Shanghai, China (2). In this process, a mixture of wastewater and biologically active solids is aerated to supply oxygen to living organisms in the solids as they utilize the dissolved organic matter for their growth. The solids are then separated from the mixture in a final sedimentation tank and returned to the aeration tank for reuse. Rapid adoption of the process for secondary treatment of wastewater occurred despite a problem that was evident as early as 1926 when Fuller noted that: "...occasionally aerating tanks have failed to function and the volume of sludge has become unusually great. This is called Bulking" (3).

The bulking of sludge continues to be a problem in operation of activated sludge processes despite almost 60 years of effort to understand its causes and to remedy its negative effects on treatment plant operation as well as the pollution of receiving waters when the sludges overflow the final sedimentation tank. The quantities of excess sludges that were a byproduct of this process were small to begin with and easily disposed of by conversion to fertilizers (e.g. Milorganite), incineration or storage and placement on land. However, the quantities of excess sludges grew in direct proportion to the increases in population that are tributary to collection systems using this process, as well as by the acceleration of the adoption of secondary treatment for wastewaters to protect public health and the quality of receiving waters.

WASTEWATER TREATMENT SLUDGES

Sludges are concentrates of the suspended solids that were originally present in the wastewaters processed or generated during treatment. They contain the major fraction of all original wastewater constituents that posed a threat to public health or the creation of nuisance if discharged untreated. As an inevitable consequence of their origin, sludges derived from treatment of human wastewater must be presumed to be infected with pathogenic organisms and contaminated with toxic substances. They contain unstable organic substances which can cause or contribute to nuisance conditions at or near the locations where they undergo stabilization. Sludges must therefore be managed after their production in such a way as to reduce the threat they pose to public health and the receiving environment.

Recent passage of the Clean Water, Clean Air and Ocean Dumping Acts have effectively closed the indiscriminate use of water and air for disposal of sludges or for dilution of the combustion products of incineration without additional capital investment and operating costs for air pollution abatement equipment and the energy resources to operate it. Sludge processing and management now accounts for more than half of the total cost of wastewater management, a noticeable fraction.

After their production, sludges are processed to change their composition and form. In a typical sequence of operations, sludges are collected; thickened; digested; elutriated; chemically or thermally conditioned; mechanically dewatered by vacuum filtration, filter press or centrifugation; transported and then applied to land, dumped into the ocean or incinerated. During these processes, the sludges are subjected to physical, chemical and biological conditions that affect the presence, kind and relative proportions of chemical substances and living organisms.

All of the unit operations to which sludges are subjected following their production contribute to one or more of the following effects:

- Disinfection - reduction of risk to health posed by pathogenic organisms.

- Detoxification - destruction, removal or neutralization of toxic substances.

- Stabilization - reduction of the nuisance associated with sludges undergoing putrefaction.

Much of the conventional processing of sludges is aimed at reducing their moisture content for more efficient operation of subsequent processes, to reduce the cost of transporting them, or to reduce the volume required for their storage. Most of these processes result in a byproduct liquid which must be returned to the wastewater treatment plant for processing with the incoming wastewater. Neither the costs of additional treatment plant capacity needed to accommodate these return flows nor their disruptive effects on the efficiency of treatment plant operation are usually considered in selecting the sludge management option.

It follows from consideration of the nature of sludges that the most efficient system would be one that minimizes the costs of disinfection, stabilization, detoxification and transportation. Since the costs relating to sludge management are directly proportional to the amounts produced, an initial step toward reducing those costs would be to select wastewater treatment processes that minimize production of sludges consistent with other treatment objectives. Regulation of the entry of heavy metals and toxic organic substances of analogous significance into the collection system may prove to be more efficient than decontamination of resultant sludges. However, the presence of unstabilized organic matter of nuisance potential, and pathogenic bacteria, protozoa, viruses and intestinal parasites is inherently characteristic of wastewater from human sources.

DISINFECTION AND DETOXIFICATION

Until recently, except for waste and wastewaters from known sources of large numbers of pathogens such as medical laboratories and hospitals, the degrees of disinfection accomplished implicitly by conventional wastewater treatment processes was acceptable. Explicit disinfection by use of conventional physical, chemical and biological agents is of limited interest because the quanity of substance needed, time of contact, capital and energy requirements are such as to make their use technically feasible but too costly, and/or result in production of byproducts that are undesirable for discharge to the environment.

Research conducted in the period from about 1950 to 1970 indicated that ionizing radiation derived from sources made artifically radioactive such as Cobalt-60 and from cathode-tube emissions were potentially useful for disinfection of wastewater and sludges (4)(5)(6). Exploratory work on disinfection of wastewater conducted at MIT with support from its Sea Grant Program, using an electron accelerator located in the MIT High Voltage Laboratory, was sufficiently promising (7) to justify preparation of a joint proposal by MIT and the High Voltage Engineering Corporation for conduct of large-scale experimentation at the Metropolitan District Commission's Deer Island Wastewater Treatment Plant in Boston. This proposal was originally submitted to EPA, but as a consequence of its decision to support a cooperative research effort with the Atomic Energy Commission involving use of ionizing radiation from Cobalt-60 as a step toward eventual use of Cesium-137, a nuclear fuels reprocessing byproduct, the proposal to investigate use of electron-beam radiation for disinfection of wastewater was referred to NSF.

Receipt of this proposal at NSF coincided with reports of results from a project being conducted at the University of Texas in Austin (8) which indicated that viruses originally present in wastewaters became concentrated in sludges during conventional treatment. These studies also indicated that viruses in the sludges remained viable despite additional processing prior to placement on soil, where they were initially adsorbed, and from which they could be desorbed under conditions simulating rainfall. The potential threat to public health resulting from application to soil of infected sludges was clear and the MIT research team was encouraged to concentrate their new efforts on disinfection of sludges by electron beam radiation.

A one-year exploratory study was initiated by Trump and his coworkers (9) in 1974 and shortly thereafter they were joined by Metcalf (10) at the University of New Hampshire who conducted associated virological inactivation studies in cooperation with MIT personnel. Results confirmed the capability of the electron beam to inactivate viruses and provided the design basis for a large-scale, 100,000 gallon per day experimental electron beam accelerator facility that was constructed and operated in a research program at the Deer Island Wastewater Treatment Plant until 1980 (11)(12)(13)(14). Knowledge gained during this seven-year period was used to redesign the Deer Island unit (15) which served as the construction prototype for a large scale electron beam system that was recently placed into operation at the Miami-Dade Water and Sewer Authority's Virginia Key Wastewater Treatment Plant. Both the unit at Deer Island and the one on Virginia Key in Miami have a design capacity for subjecting 170,000 gallons per day of sludges to a dose of 400,000 rads, which at Miami is one-fourth of the capacity needed to disinfect all of the sludge produced and transported to the Virginia Key plant. Operation of the initial stage there was planned to provide the engineering design basis for possible expansion of the system to full scale.

Research conducted at MIT also led to findings that ionizing radiation from an electron accelerator destroyed polychlorinated biphenyls in water solution (16). The significance of these preliminary observations as they may relate to the detoxification of sludges remains to be determined. However, despite some evidence that regulation of discharges from industrial sources may not completely address the contamination of sludges with heavy metals, the combination of regulation and pretreatment appears to offer the most cost-effective approach to dealing with the removal of contaminants such as heavy metals and organic chemicals of similar significance.

STABILIZATION AND TRANSPORT

Sludges are transported from their sources, through processes for their modification, and finally to locations where they can be stabilized under conditions that minimize threats to public health and nuisance conditions. In his 1912 book, Fuller (17) noted that "...disposal of sewage by dilution is a proper method when by dispersion in water the impurities are consumed by bacteria and larger forms of plant and animal life or otherwise disposed of so that no nuisance results." Only 16 of the 767 pages in his book deal with sludge processing while 192 pages are devoted to dilution in inland streams, large lakes, tidal estuaries and oceans. Processes covered are essentially those still in use, viz. lagooning, drying beds, pressing, digestion, filling or dumping, trenching, mechanical drying, incineration, and destructive distillation.

Just 4 years later, sludge management issues being raised by emergence of the activated sludge process from its "highly experimental stage" were presented in Metcalf and Eddy's book (18) including potential agricultural utilization of both sewage and sludge. Fuller's 1926 revision, coauthored by McClintock (3) has two chapters on the activated sludge process, and two on disposal of activated sludge, one of which dealt with its use as a fertilizer. During the next 35 years, there were numerous refinements in physical and chemical processing of sludges for reducing their water content, some new concepts in heat processing and incineration, were developed and some interest continued in use of waste activated sludges as fertilizers.

Land application of wastewater and the use of land as a final receptor for sludges experienced a resurgence of interest in the 1960's, attributable partially to the expected vast increase in the amounts of sludge that would be the consequence of adopting the activated sludge process as the preferred one for full implementation of water pollution control legislation. The U.S. Department of Agriculture working closely with the U.S. Environmental Protection Agency developed a process for composting sludges which involved mixing dewatered sludges with a bulking agent such as wood chips and drawing air through the mixture to maintain aerobic conditions during the period of time needed for stabilization of the sludge (19). Costs for composting are in addition to those necessary to prepare the sludges for this process, and the process can result in nuisance conditions at and near the composting location. The potential use of ionizing radiation for disinfection of composted sludges was investigated in studies supported by the U.S. Department of Energy as a responsive measure to inactivate pathogens that would survive during composting (20).

An alternative concept for placement of stabilized sludges on land or utilizing land application as the stabilizing process emerged from research conducted by Smith at Colorado State University (21). In this concept, advantage is taken of the water content of sludges to permit their transport by pipeline to sites where an implement, connected to the pipeline with a flexible hose, would inject the sludges into topsoil where they would undergo stabilization. Disinfection could occur at either end of the pipeline and the injection system would be operated to eliminate any possibility of nuisance. Accumulations of sludge-modified topsoil would be removed as necessary to maintain the viability of the site and as determined acceptable from quality assurance evaluation of the material to be removed in the context of its destination. In addition to providing a discrete method for transport of sludges analogous to the wastewater collection system, this concept would eliminate costs of dewatering chemicals and equipment, contamination of sludges by those chemicals, and costs as well as operational problems from return-flows. If found to be necessary, the seepage from the land application site collected by an underdrainage system, and surface runoff would be treated to prevent pollution of surface and groundwaters (22)(23). For coastal communities, the analogous alternative would be the transport of disinfected sludges by pipeline or barge to locations in the ocean where the sludges could be safely dispersed to eliminate any defilement of beaches or contamination of fisheries.

Both land and ocean sludge stabilization concepts require additional research to determine their technical and economic feasibilities, degree to which they meet public health requirements, and to establish their engineering design parameters as well as confidence in their use. Land stabilization appears to offer a somewhat more conservative approach than ocean placement because it would be subject to management and control by isolation of the location where stabilization takes place, monitoring of the site, controlled public access and isolation of the area to eliminate any possibility of dispersal of contaminants.

SUMMARY

Future concepts of sludge management are more likely than they were in the past to be based on the efficiency with which they utilize capital, material, human and energy resources. Disinfection, decontamination, stabilization and transportation are fundamental considerations in comparing alternative systems for management of sludges. The use of ionizing radiation from electron accelerators coupled with direct injection of sludges into topsoil where they undergo stabilization appears to offer an attractive alternative to existing systems for management of sludge. The unit operations comprising this concept have been sufficiently studied to warrant conduct of a large-scale, experimental evaluation of its technical and economic feasibility which would take into account environmental issues that cannot be predicted from the results of research on concept components. The comparable ocean placement concept likewise suggests need to conduct large-scale, experimental assessment of its feasibility.

REFERENCES

1. The Quest for Pure Water by M. N. Baker, The American Water Works Association, Inc., New York (1948).

2. Textbook of Bacteriology by Edwin O. Jordan and William Burrows, 13th Edition, Revised, W. B. Saunders Company, Philadelphia (1938).

3. Solving Sewage Problems by George W. Fuller and James R. McClintock, p. 432, McGraw-Hill Book Company, Inc., New York (1926).

4. Dunn, C. G. "Treatment of Water and Sewage by Ionizing Radiations, Sewage and Industrial Wastes, Vol. 25, No. 11, pp. 1277-1281 (1953).

5. Gerrard, Martha. "Sewage and Wastewater Processing With Isotopic Radiation: Survey of the Literature," Isotopes and Radiation Technology, Vol. 8, No. 4 pp. 429-435 (1971).

6. The Effects of Radiation on Chicago Metropolitan Sanitary District Municipal and Industrial Wastewater, Research Report to the Metropolitan Sanitary District by Pacific Northwest Laboratories, Battelle Memorial Institute, Richland, Washington (1968).

7. Trump, John G. "Wastewater Treatment With High Energy Electrons," Sea Grant Project Summary, Project No. R/Pb-2, July 1-December 31, 1972.

8. Duboise, S. M., B. P. Sagik, B. E. D. Moore and J. F. Malina, "Virus Migration Through Soils," in Virus Survival in Water and Wastewater Systems, pp. 233-240, Water Resources Symposium No. 7, Center for Research in Water Resources, University of Texas at Austin (1974).

9. High Energy Electron Irradiation of Wastewater Liquid Residuals, NSF Grant No. GI 043112 to the Massachusetts Institute of Technology, Principal Investigator: John G. Trump (April 29, 1974).

10. Control of Virus Pathogens in Municipal Wastewater and Residuals by Irradiation With High Energy Electrons, NSF Grant No. AEN 75-14279, University of New Hampshire, Principal Investigator: Theodore G. Metcalf (March 18, 1975).

11. Trump, John G. "High Energy Electron Radiation of Wastewater Liquid Residuals," Final Report, NSF Grant No. ENV 74-13016 National Technical Information Service (NTIS) No. PB 279489 (December 1977).

12. Trump, John G., Edward W. Merrill, and Anthony J. Sinskey, "High Energy Electron Radiation of Wastewater Liquid Residuals," Final Report, NSF Grant No. ENV 77-10199 NTIS No. PB 297593 (February 1979).

13. Metcalf, Theodore G., "Control of Virus Pathogens in Municipal Wastewater and Residuals by Irradiation With High Energy Electrons," Final Report, NSF Grant No. ENV 75-14279 and ENV 77-14454, NTIS No. PB 272347 (September 1977) and PB 80-104086 (November 1979).

14. Trump, John G., "High Energy Electron Treatment of Wastewater Liquid Residuals," Final Report, NSF Grant No. PFR 78-24092, NTIS No. PB 101577 (December 31, 1980).

15. Merrill, Edward W., D. R. Marby, H. B. Schulz, W. D. Coleman, J. G. Trump, and K. A. Wright, "Destruction of Trace Toxic Compounds in Water and Sludge by Ionizing Radiation," American Institute of Chemical Engineers Symposium Series, Water 74:245-50 (1977).

16. Sewage Disposal, by George W. Fuller, McGraw-Hill Book Company, New York (1912).

17. American Sewerage Practice, Volume III - Disposal of Sewage, McGraw-Hill Book Company, New York (1916).

18. Epstein, E., G. B. Willson, W. D. Burge, D. C. Mullen, and N. K. Enkiri, "A Forced Aeration System for Composting Wastewater Sludge," Journal of the Water Pollution Control Federation, Vol. 48, No. 4, pp. 688-694 (1976).

19. Ahlstrom, Scott B., "Evaluation of Sludge Irradiation as a Process to Further Reduce Pathogens," Proceedings of the 1981 National Conference on Environmental Engineering, pp. 524-531, American Society of Civil Engineers, New York (1981).

20. Brandon, Jerry R. "Sandia's Sludge Irradiation Program," in Proceedings of the Third National Conference on Sludge Management, Disposal and Utilization, pp. 135-137, Information Transfer, Inc., Rockville, Md., (1977).

21. Smith, James L., Maurice H. Lutkin, James S. Latham, and Alan de Haai, "Land Management of Subsurface-Injected Wastewater Liquid Residuals," Interim Report, NSF Grant No. AEN 74-08082, NTIS No. PB 280162 (November 1977).

22. Bryan, Edward H. "Research Supported by the National Science Foundation Relating to Treatment of Wastewater and Management of Residual Sludges," in Proceedings of the 8th United States/Japan Conference on Sewage Treatment Technology, pp. 54-64 U.S. Environmental Protection Agency publication, (1981).

23. Bryan, Edward H. "Future Technologies of Sludge Management," in Proceedings of the 1980 Spring Seminar, National Capitol Section, American Society of Civil Engineers, pp. 52-62 (1980). Also in Proceedings of a Seminar on Electron Disinfection of Municipal Sludge, pp. 227-238, Massachusetts Institute of Technology, NTIS No. PB 82-101577; and in Dangerous Properties of Industrial Materials Report, Vol. 2, No. 3, pp. 2-7, Van Nostrand Reinhold Company, New York (May/June 1982).

Ocean Disposal Systems for Sewage Sludge and Effluent

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INTRODUCTION

This paper is a condensed version of the report titled Ocean Disposal Systems for Sewage Sludge and Effluent authored by the Committee on Ocean Waste Transportation of the Marine Board, Commission on Engineering and Technical Systems of the National Research Council (NRC). The report contains 126 pages and was published by the National Academy Press, Washington, D.C. in 1984. Although I served as chairman of the committee, the members, the NRC staff, and two consulting organizations prepared the report. Members of the committee were Norman H. Brooks, Judith M. Capuzzo, Gabriel T. Csanady, C. R. Firth, William F. Garber, Francois M. M. Morel, Nathan Sorenshein, and NRC staff, Donald W. Perkins. This paper focuses on the transportation aspects of the study set in the context of the conclusions and recommendations of the total study. Much of this paper comprises direct quotations from the report. To provide the reader with the most coherent, yet abbreviated, text I have not indicated quotations where they do in fact occur. The reader is encouraged to read the entire report, plus supporting appendices, to appreciate the scope as well as the limitations of the study.

The study had three objectives:

- To provide a rationale for considering ocean disposal of sewage sludge and effluent,

- To define the characteristics of acceptable ocean disposal sites, and,
- To compare surface and submarine pipeline options for sludge and effluent transportation.

STRATEGIC CONCLUSIONS

Two strategic conclusions were reached early in the study. These guided the conception and exploration of physical systems.

- Systems to promote dispersal of sewage sludge and effluent in the ocean are feasible.
- Systems to contain or isolate sewage sludge in the ocean or on the bottom of the ocean are not feasible.

CRITERIA FOR ACCEPTABLE MARINE DISPOSAL OF SLUDGE

For sludge discharge into the ocean by surface vessel, it is necessary to designate a disposal site, or an area within which sludge vessels may discharge their cargo. Alternatively, for a sludge outfall pipeline, the alignment, pipe length, and discharge point must be selected.

For a site to be environmentally acceptable, it should meet all the conditions enumerated below. The discussion that follows is based solely on environmental grounds and does not take account of existing regulatory policies, administrative procedures, or costs of alternative disposal methods. The allowable discharge area for vessels may be elongated for hundreds of kilometers along an isobath, but probably restricted to 10-50 km perpendicular to the isobath. A policy of maximum dispersal rather than containment applies for sewage sludge; the same policy and the same criteria do not necessarily apply for other types of wastes.

1. The site must not be within, nor too close to, sensitive areas, including: the shoreline; productive fishing grounds, such as are found at a shelf-break front of the East Coast; and a unique ecosystem such as a coral reef or a marine sanctuary. The distance of the site from any such sensitive area should be so large that the probability of immersion in a diluted sludge plume advected from the source in less than a day is very small. The general criterion is that, at the time of approach to the sensitive area, the dilution of the liquid sludge should be 10^4 or more.

2. The long-term, background concentration of fine sludge particles caused by the sludge discharge in the water column should be comparable to the concentration of natural organic particles. In addition, the depth over which the sludge is distributed is important; deep-water sites are potentially more favorable than shallow-water sites.

3. The incremental benthic accumulation for organic solids in the nearfield, generally particles with fall velocity w 10^{-2} cm/s, should not be much greater than the natural sedimentation rate at the dumpsite. The exception to this objective is a fixed point outfall discharge of sludge, where the deposition rate may be allowed to be much higher in a limited designated area near the outlet and larger benthic impact is expected. Adequate depth and current strength are significant factors in determining whether this criterion is met.

4. The flushing of the site must be vigorous enough to prevent excessive nutrient enrichment that would cause algal blooms and to prevent serious depletion of oxygen either above or below the pycnocline.

5. The increase in long-term mean concentration of all important trace contaminants in the water column and in sediment must remain below background levels or below levels considered safe on the basis of toxicity and bioaccumulation tests. However, for most normal sludges from sewerage systems having reasonable source control, compliance with the sedimentation requirement (item 2) will most likely mean that trace contaminant levels are also acceptable.

6. For disposal from vessels, near-surface discharge into the wake of the ship is preferred to maximize dilution. The rate of discharge must be limited to that which ensures adequate dilution. (The large distances from shore eliminate aesthetic problems.) Discharge below the pycnocline by means of stingers extended down from the ship is not desirable because of the lower initial dilution achieved.

7. For an outfall site to be acceptable there must be adequate circulation and dispersal of the water mass at the plume depth to meet the above criteria. The plume is likely to be submerged due to stratification. Particular attention should be given to the dissolved oxygen requirement.

8. The size of the disposal site for vessel discharge can be large--the larger it is, the greater the dispersal will be. All parts of site must, however, meet the criteria above, even if it is assumed that most sludge release is concentrated in one portion of the area, such as the area nearest to a source city. Disposal areas can most logically be established as an elongated region with the long axis parallel to the bottom contours and the short axis perpendicular because the characteristics of the ocean change slowly along the contours on an open coast.

9. Areas of vigorous resuspension of bottom sediments (as indicated by coarser bottom sediments) are preferred; areas of quiescense are to be avoided. This is a relative criterion and indicates a preference (not a fixed criterion) if there is a choice within a region, and the other criteria are satisfied.

10. Floatable solids should be removed from the sludge at the treatment plant to an extent that will avoid the occurrence of a slick or floating solids in the ocean. Particles with high

settling velocity (greater than 1 cm/s) should also be removed to avoid heavy buildup around outfalls or in disposal sites of small area. Screened digested sludge usually meets these restrictions, but undigested primary sludge would be unsatisfactory.

The above criteria are suggested as guidelines to indicate a methodological approach. In any given application the criteria may be made more or less stringent if desired. In case of multiple sources, the sum of the effects must meet the suggested guidelines and criteria.

SITE MONITORING

The importance of a carefully planned and diligently executed monitoring program cannot be overemphasized. In addition to the operational information that it will provide, it constitutes the cornerstone of public confidence in system safety. Not only should the monitoring system provide information concerning the day-to-day and long-term effects of disposal operations, but it will also provide baseline information that will permit the separation of the effects of large-scale natural phenomena from the effects of disposal operations.

Four monitoring objectives are primary:

- To determine whether or not the disposal system is operating as planned and meets requirements;
- To obtain information that can be used to feed back to the system so that treatment processes and transport operations can be changed (tightened or relaxed) as operating experience is obtained;
- To obtain information with which the public can be informed that the system is or is not operating as planned; and
- To obtain valid scientific information on processes that are important to understanding the ultimate fates and effects of contaminants.

ENGINEERING APPROACH TO MARINE DISPOSAL TO MEET ENVIRONMENTAL OBJECTIVES

Successful wastewater and sludge disposal in the ocean depends on designing an appropriate engineering system where the input is the waste water and the output is the final water quality in the region of the discharge.

For treated wastewater and sewage sludge, the disposal strategy should be wide dispersal, rather than containment. With proper engineering design including choice of disposal site, it is feasible to achieve wide dispersal of treated wastes, and degradation through natural biogeochemical and microbial processes, with minimum risks to ecological systems and human health.

For municipal sewerage systems, the principal variable components are listed below and shown in Figure 1.

- Source control (or pretreatment) of industrial wastes before discharge into municipal sewers;
- Sewage treatment plants, including facilities for processing of sewage solids (sludge); and
- Outfall pipes and diffusers for disposal of effluents into the ocean, and either barges or pipelines for injection of sewage sludge into receiving waters at the desired location.

GENERIC SITES

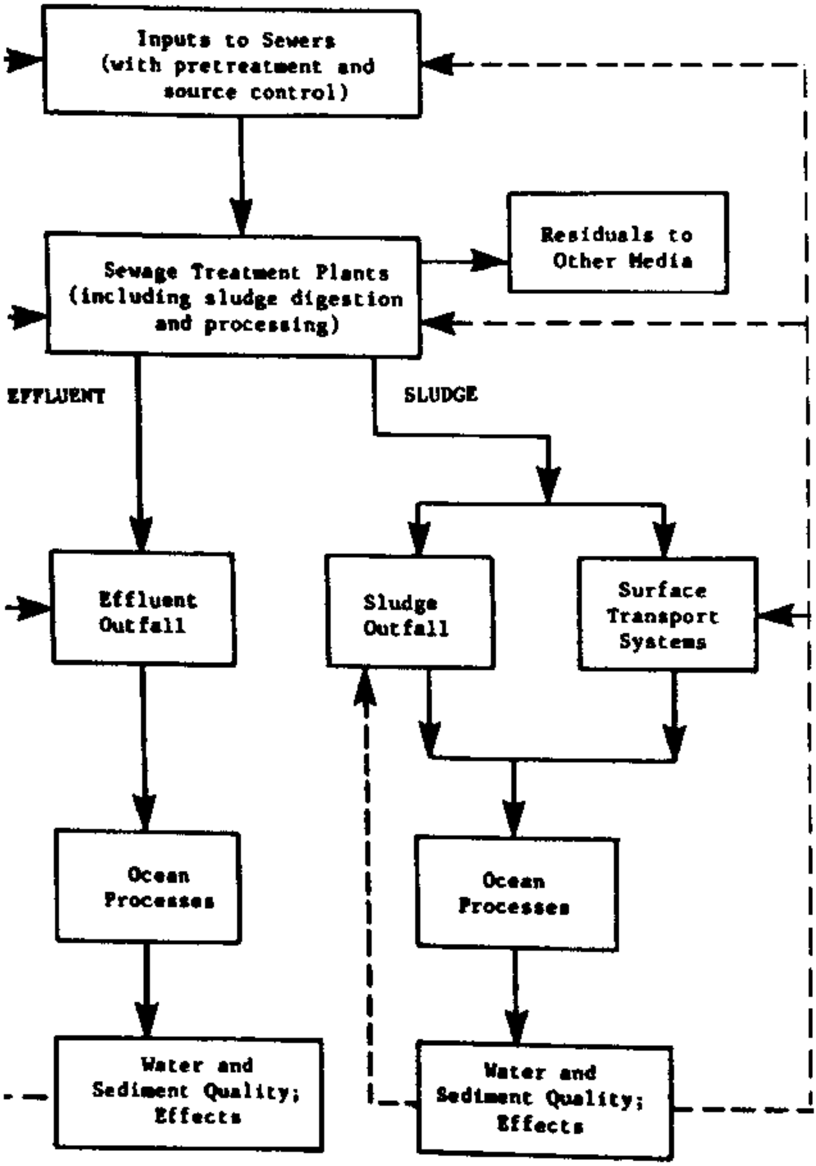
Two characteristic bathymetric conditions were considered. One was a broad, flat, underwater slope of -0.1 percent, which is typical of the upper mid-Atlantic shelf. The other was a steeper slope of -3.3 percent to deeper water of 600 m, which is typical of the Pacific Ocean off southern California.

ECONOMIC MODELS

Economic models were used to compare the two systems on a parametric basis. The pipeline model was prepared and run by Bechtel Petroleum, Inc., of Houston, Texas. It derived from pipeline models used to analyze submarine pipelines for the transport of petroleum and petroleum products.

The surface transport model was prepared and run by M Rosenblatt and Sons of New York, New York, a ship design organization. This model incorporates features of ship system cost models typically used in commercial practice. In addition, system capital and operating cost information was obtained from the New York area sources as well as from naval architectural sources and the federal government.

Care was taken to ensure that common economic assumptions were used in both parametric models so that output information would be directly comparable. Two measures of costs were



URE 1 System overview for marine disposal of wastewater
't side) and sludge (right side). Dashed arrows indicate
sible design adjustments to the system to achieve different
l results (e.g., water and sediment quality, and effects).

generated by these models for various throughput rates. One was a unit transportation cost expressed in first quarter 1983 dollars per dry metric ton of solids throughput. The unit transportation costs were for the first year of operation. The second cost measure computed by the models was the present value of 25-year costs.

The distances to the disposal site, or discharge point, used in the economic analyses were chosen to cover ranges that can be considered most likely to be useful or feasible. For pipelines, the selected ranges were 6 km to 18 km (at steep slope of 3.3 percent) and 60 to 100 km (at flat slope of 0.1 percent). For surface transportation, the range selected was 5 to 200 km, which represented the likely maximum range to accommodate U.S. coastal environmental concerns.

Common Assumptions for Surface and Pipeline Systems

To generate comparable cost data, it was necessary to establish common ground rules for the pipeline and surface transport models. Some of these ground rules were directly applicable to both systems; others, because of the unique physical features and operating practices of the two different systems, were rationalized to provide as nearly equal operating constraints as practical. Common, or normalizing, assumptions are described in the following paragraphs.

Location of plant. Output from a typical coastal municipal treatment plant was considered to be the input to both pipeline and surface transport systems. For the pipeline this location was immediately behind a sandy coastline with a direct and unobstructed access route to the terminal (discharge) point of the pipeline. For surface transport systems the plant location was assumed to be at a sheltered harbor site with immediate access to the ocean so that either ships or tug-barge units, could proceed directly to and from the designated dumpsite.

Sludge characteristics. The composition of sludge by weight, as it leaves the treatment plant, was assumed to be either 2 or 3 percent solids and 98 or 97 percent water respectively with a specific gravity of 1.01.

Annual throughput. The annual throughput is constant throughout the project life.

Useful life of systems. Costs were computed using a 25-year economic life. For surface transport systems this time period was assumed to be the total useful life and no salvage value was assumed. For the pipeline system a useful life of 50 years was assumed, so that at the end of the 25-year period used for the cost comparison a residual value was assigned.

Cost levels. All costs were computed using first quarter 1983 dollars. As a simplifying assumption, cost-level changes (inflation/deflation) were ignored. Consistent with this assumption, a real interest rate was used to approximate the cost of capital.

Financial structure. All capital costs were assumed to be 100 percent debt financed through the sale of municipal or public authority bonds.

Bond terms. Bond maturity was assumed to be 25 years at a real interest rate of 7 percent. This gives a capital recovery factor of 8.58 percent of investment.

Discount rate. The disocunt rate is equal to the real interest rate, which is assumed to be 7 percent for this study.

Taxes and profit. Income taxes, property taxes, and profit were excluded from financial calculations.

Indirect capital costs. Included in the financial calculations as indirect capital costs were sales taxes; engineering, procurement, and construction management costs; and contingencies. Excluded from the financial calculations were financing costs and working capital.

Ownership. Both pipeline and surface transport systems were assumed to be built new and owned by municipal or public authority operators.

Material sources. Pipe and ship steel were assumed to be of U.S. manufacture.

Accuracy of cost analysis. The accuracy of the capital and operating costs was estimated to be within a range of ±25 percent.

Other assumptions used in the analysis of surface and pipeline systems were unique to each system and will be presented in the following sections devoted to a more detailed discussion of each system.

SURFACE TRANSPORTATION SYSTEM

The analysis of surface transport systems included four alternatives to move sludge from a treatment plant to an ocean disposal site. These alternatives were (1) self-propelled sludge vessels, (2) barges-in-tow, (3) articulated tug-barges (ATB), and (4) inflatable rubber barges-in-tow.

A generalized flowchart of the ocean waste transportation system is given in Figure 2.

The four systems considered in the comparative cost analysis are described in the following summaries. The prime parameters that determined the optimum selection of cargo deadweight transport size (CDWT) and the corresponding number of transport units for a given sludge throughput for a given distance to a disposal site are: (1) vessel type, (2) vessel speed, (3) delays (weather and port), (4) loading and discharging rates, (5) annual dry docking and repair time, and (6) cost. The philosophy for sizing a particular transport system fleet was to minimize the number of transport units, favoring large CDWT capacities, for a given sludge throughput and disposal site distance.

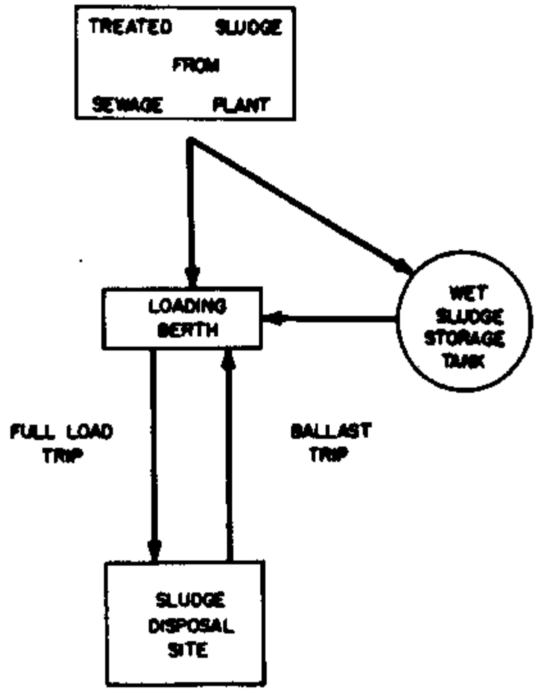


FIGURE 2 Ocean waste surface transportation flowchart.

Self-Propelled Sludge Vessels

Self-propelled sludge vessels are tank vessels divided longitudinally and transversely into individual tanks, each with bottom dump gates. Tank bottoms are positioned in the ships above the still water level when the ship is empty so that the tanks will discharge by gravity. A simplified center line profile of a typical self-propelled sludge ship is shown in Figure 3.

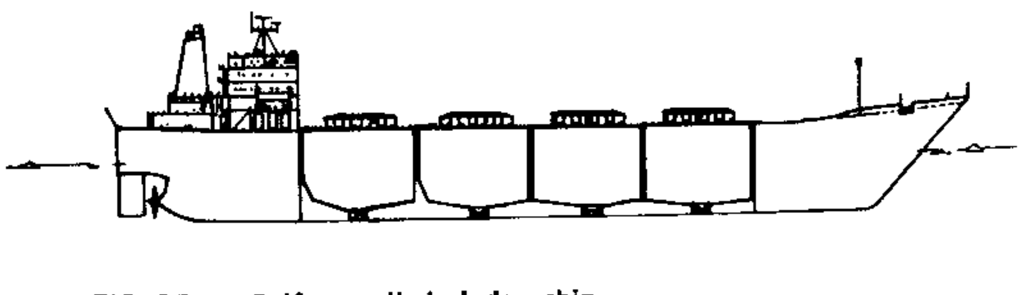


FIGURE 3 Self-propelled sludge ship.

Barges-in-Tow

The barges-in-tow system assumes one tug and one barge with the tug pulling the barge by hawser as shown in Figure 4. The barges are divided longitudinally and transversely into as many tanks as appropriate for the vessel strength and stability.

FIGURE 4 Barges-in-tow.

Articulated Tug-Barges (ATB)

The articulated tug-barge system offers some advantages of self-propelled surface vessels and some advantages of barges. In all but severe weather the tug pushes the barge by fitting into a stern "notch" that permits the tug-barge to enjoy many of the efficiencies of economy and speed associated with self-propelled vessels. An articulated tug-barge is shown in Figure 5.

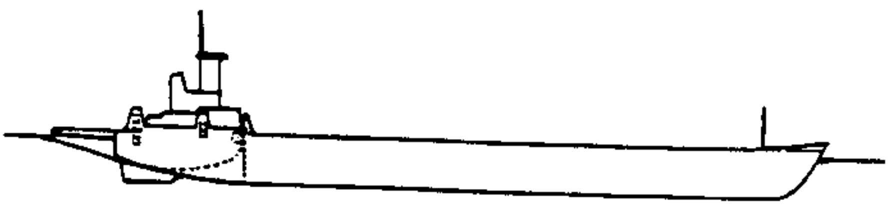


FIGURE 5 Articulated Tug-Barge.

In severe seas it will be necessary for the tug to leave the notch and move into a towing position, thus losing the advantage of articulation.

Rubber Barges-in-Tow

This system of transporting sludge consists of a standard tug towing an inflatable, compliant, streamlined cylinder which is towed nearly submerged. Its advantages include lower capital cost due to fewer structural requirements. Its disadvantages include shorter life and lack of extensive experience in this service. A rubber barge-in-tow is shown in Figure 6.

FIGURE 6 Rubber barge-in-tow.

Discharge of sludge from the rubber barge can be accomplished in two ways. The first way is to pump out sludge through the flexible hose attached to the nose of the barge. This is the same flexible hose that is used in loading the barge. The second method is to open a radio-controlled valve at the tail plate of the barge, in which case the sludge will be discharged through this opening while the barge is under tow.

ASSUMPTIONS UNIQUE TO SURFACE TRANSPORT SYSTEMS

Assumptions, in addition to those common assumptions discussed earlier, were uniquely necessary for the surface transport system. These were:

- For all barge transport systems, one tug for each barge.
- Constant volume of sludge flow from treatment plant.
- Single point pickup and discharge.
- Shoreside downtime relates only to time during which sludge cannot be loaded onto the transport units for "nonmarine" reasons.
- Two hours for port hookup and disconnect.
- Fifteen days for annual dry docking and repairs.
- Full load transit speeds for transport units assumed will be:
  -10 knots for self-propelled sludge vessels;
  -8 knots for ATB and rubber barges in tow; and
  -6 knots for steel barges in tow.
- 20 percent of the time, barges will be in tow and ATBs will have to reduce speed due to weather as follows:
  -3 knot speed loss for the ATB when changed from push to tow mode due to weather; and
  -1 knot speed loss for barge in tow due to weather.
- Ballast trip speed 1 knot greater than full load trip speed, except for towing rubber barge, where there is a 3 knot increase in speed.
- Loading rate is equal to discharge rate, but limited to a maximum of 6 hours.
- Distance to sludge disposal site: 5-200 km, but only data for 20 to 200 km could be conveniently plotted.

- Dry sludge throughput range: 5-500 metric tons per day (except inflatable barge).
- 3 percent dry sludge concentration.
- 50 dry metric tons per hour discharge rate.
- 10 percent shoreside downtime; 4-hour repair time.
- 90 percent vessel utilization efficiency (i.e., required number of voyages to transport sludge is 90 percent of the total number the fleet is capable of making).

Output of the simulation program provides data points for transport distances of 5, 20, 50, 100, and 200 km from the sludge pickup point to the dumpsite. Throughput values of 5, 50, 150, 300, and 500 dry metric tons per day were considered as noted earlier. One exception is that the inflatable barge-in-tow system analysis extends only to 300 dry metric tons per day capacity due to its size limitation.

The simulation program assumes continuous operations for the transport vessels, except for maintenance, repair, and weather down time. Such operations are normal maritime practice and are feasible from a human factors point of view through appropriate watch scheduling and crew rotation techniques. If a municipality elects to limit operations to less than continuous ones, the fleet size and transport cost would be increased correspondingly. The actual cost increase will depend, of course, on site and specific schedular considerations.

This study, which compared the alternative surface transport systems, is parametric and is not directed towards an analysis of a specific site. The net effect of not running the transport vessels round-the-clock could increase the transport cost by 15 to 20 percent. However, the actual cost change will depend on conditions at a specific site.

RESULTS OF COST COMPARISON

The study model is a static simulation. The characteristics of the marine system are provided as input data to the model and the system is analyzed. By varying the defined base transport system and sludge throughput/discharge rate, the system's capability to move the required sludge and the associated cost can be estimated. The accumulated sludge that temporarily cannot be moved will determine the required sludge storage capacity. Figure 7 shows the results of the least-cost alternative, barges-in-tow. This is the level of detail produced by the model.

Figure 8 shows a comparison of all surface transport systems considered in the study.

SLUDGE THICKENING

Sludge typically leaves the digestion tanks in a treatment plant as a suspension consisting of about 97 percent water and 3 percent suspended solids. The cost of shipping sludge per

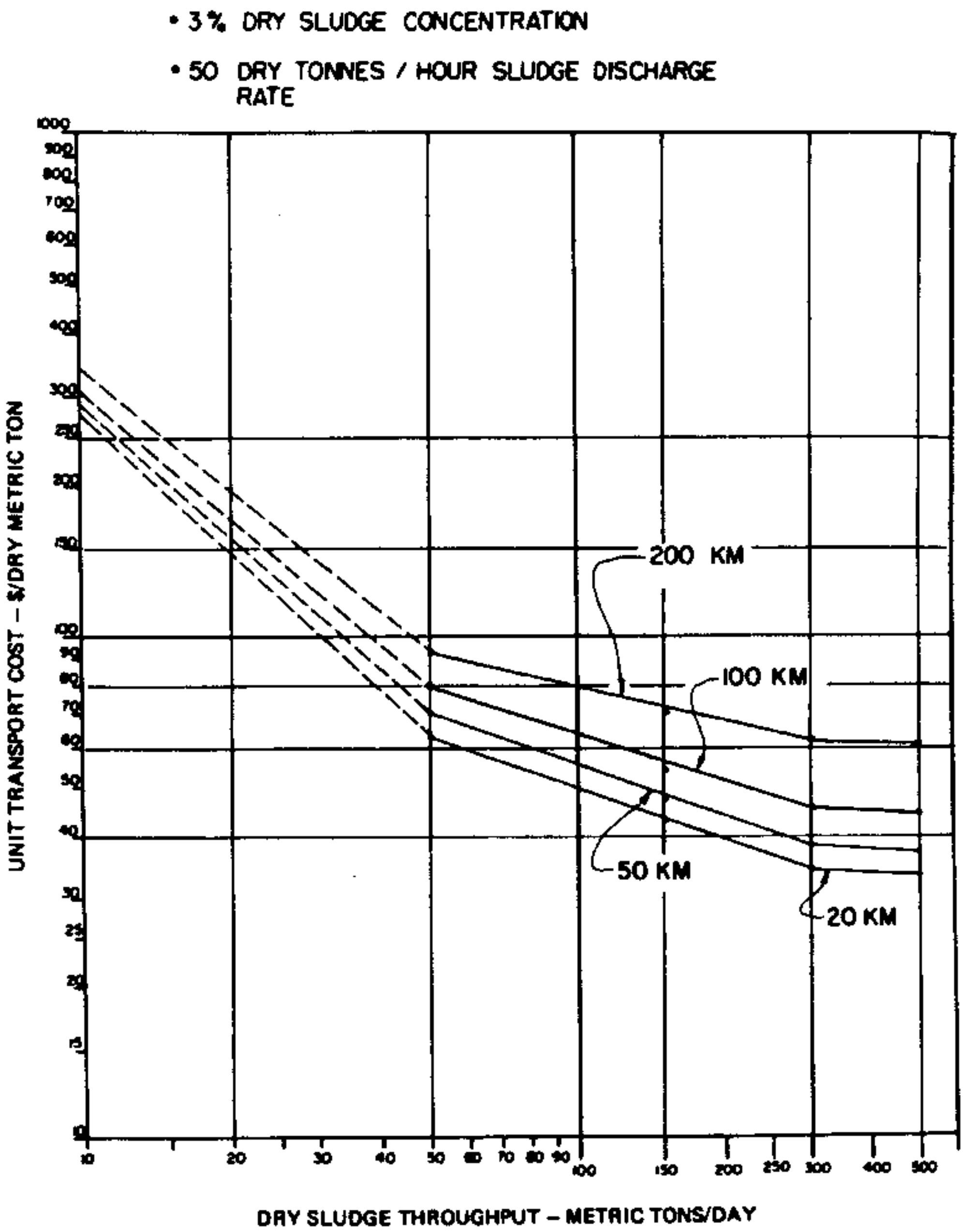


FIGURE 7 Barges-in-tow.

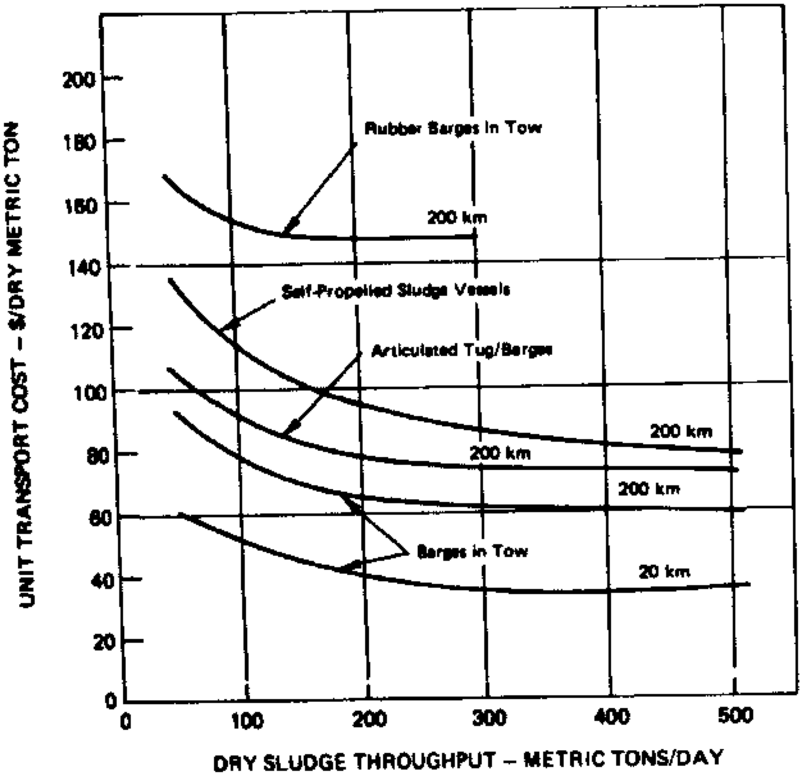


FIGURE 8 Comparison of surface marine transport system costs (3 percent dry sludge concentration).

metric ton of solids is the only measure by which the surface and pipeline systems can be compared in this study; these costs may be stated in either unit transport cost per metric ton or present value per unit capacity.

It is a distinct advantage to the surface vessels if they can carry sludge containing a higher percentage of solids and less water. Thickening sludge has its disadvantages, however. It adds costs since it involves a mechanical separation process such as centrifuging, thus offsetting some of the savings resulting from a smaller fleet and fewer trips. Sludges can be thickened by methods other than centrifuging, such as using belt filters. The choice of method is based on percentage of removal needed and cost, and such decisions sometimes must be made after operational experience is gained. Figure 9 shows cost comparisons for sludge thickening in the range from 3 percent (no further thickening) to 8 percent dry sludge for trips of 100 km and 200 km and throughputs of 150 and 300 dry metric tons per day. The

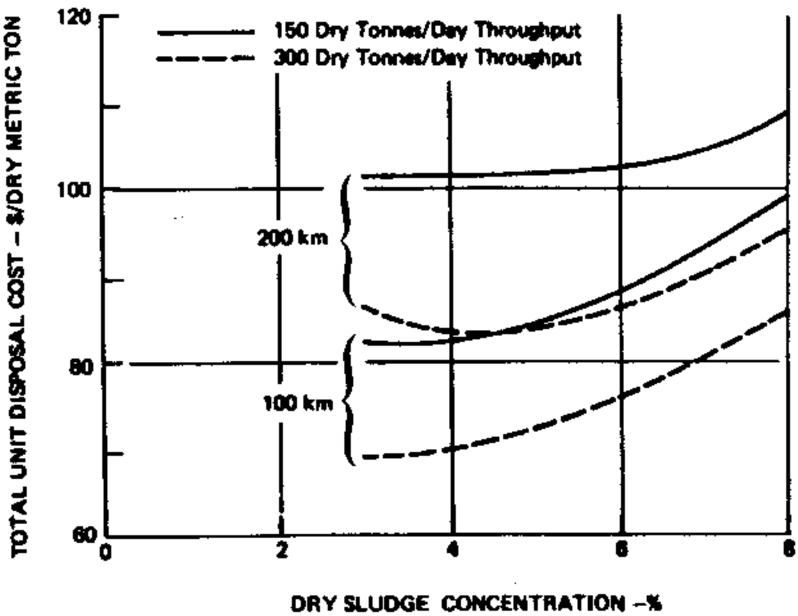


FIGURE 9 Effect of dry sludge concentration on total unit disposal cost, based on use of self-propelled sludge vessels. Sludge thickening cost included in total cost.

empirical equation for sludge thickening costs,

> Cost in \$/dry metric ton = 7.35 C_t - 22, where C_t is percent concentration after thickening,

is based on data from a sludge thickening plant operated by the Bureau of Sanitation of the City of Los Angeles. The curves in Figure 9 are illustrative only because many site specific factors such as construction and operating costs and the physical and chemical characteristics of the sludge will determine the actual cost and can differ markedly from site to site.

PIPELINE TRANSPORTATION SYSTEMS

Initial consideration was given to both concrete and steel pipelines for the transport of sludge to bottom disposal sites. Due to the relatively long distances considered, particularly on the Atlantic shelf (slope of -0.1 percent), the well-established technology for installing steel pipelines was adopted for the cost analysis.

Petroleum pipelines are characterized by their relatively long length and high-pressure requirements. Thus good practice has dictated the use of all-welded steel pipe, on land and on the ocean bottom. When these lines are installed at sea, the pipe walls must be designed to guard against buckling during installation. When the line is in place, the extra wall thickness is available to accommodate higher internal pressures.

Sea outfalls generally traverse three zones: onshore, surf, and offshore with discharge through a linear diffuser, diffuser structure, or open-ended pipe.

Soil properties are important due to the diversity of possible trenching and burial conditions and the alternatives for dealing with them. Optimal subsea conditions consist of gradually sloping sandy soil or mud. Under these conditions, the pipe is laid and buried by jetting within the surf zone.

Difficult conditions consist of rock or coral sea floors and are dealt with by anchoring the pipeline at intervals or by laying the pipeline in a blasted or dug trency.

Internal forces in a pipeline result from the hydraulic pressure created by pumping. Operating pressures in short length outfalls are normally low and, thus, have little influence on the design of the pipeline. Longer distance outfalls, however, require higher operating pressures to compensate for friction losses; these pressures influence wall thickness.

The external forces on an outfall can be very significant. These result from the oscillatory movement of waves or currents where horizontal drag forces and vertical lift forces may cause undesired pipeline movement.

Boats or ships may drop anchor on the line and cause damage. Bottom trawling fishing vessels are also hazards. In the surf zone, where wave and current forces are exceptionally high, it is common practice to bury the pipeline with adequate cover--enough for the pipeline to remain below extreme seasonal erosion. Beyond the surf zone, some type of anchorage is advisable. This can be accomplished by tying to rock or soil using weighted concrete anchors, exterior concrete coating, or screw devices.

In this study all required pumping was assumed to be at the input point and no booster pumping was required for 100-km distance, the maximum pipeline length considered.

CONSTRUCTION METHODS

The following methods may be used in submarine pipeline construction for sludge outfalls constructed of steel pipe:

- Bottom pull,
- Bottom tow,
- Float and sink, and
- Continuous pipe lay barge.

Bottom Pull

The bottom-pull method eliminates almost entirely the need for sea bottom connections. The pipe is assembled, preferably welded, into short strings at the onshore yard. These strings are then rolled onto a centrally located launchway, where pipe on rollers can be pulled into the sea. The strings are then welded into a continuous pipeline on the launchway, concrete weight coating added, or completed at joints, and subsequently pulled by tugs or anchored winching arrangements. A problem with this method is the substantial frictional resistance of the sea bottom (unless supplementary buoyancy is attached); thus, only fairly short pipelines can be installed using this method. This method is shown schematically in Figure 10.

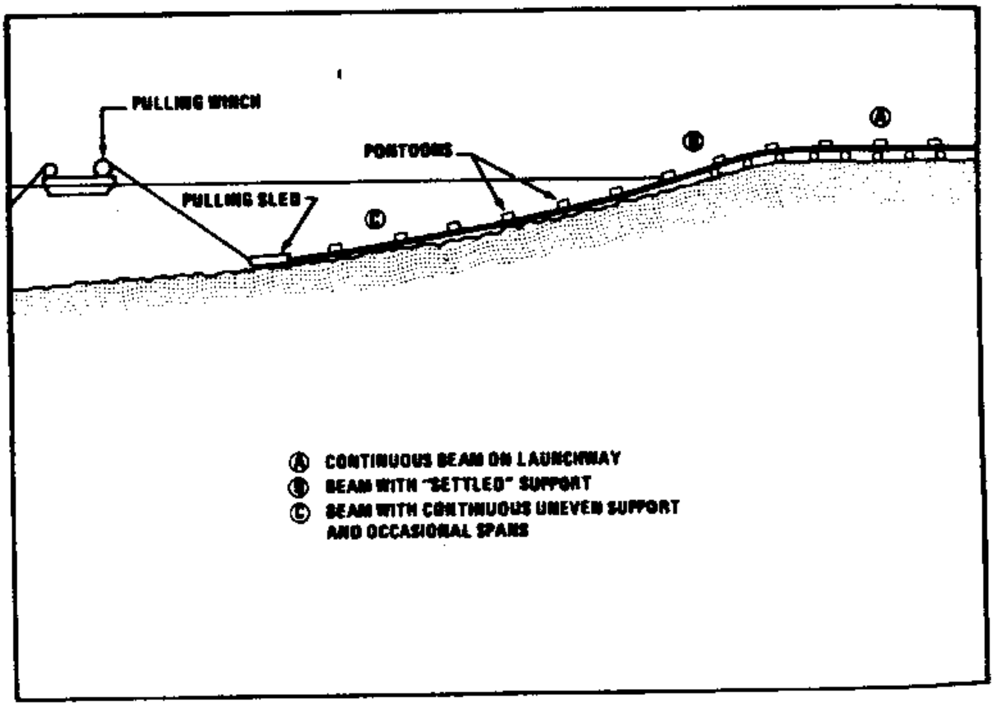


FIGURE 10 Bottom-pull pipe configuration.

Lay Barge

Lay-barge methods are used where substantial lengths of pipe have to be laid continuously.

Conventional and semisubmersible lay barges ("S" configuration) and high-angle pipelayers ("J" configuration) are most suitable for construction of deepwater lines. "S" and "J" describe the configuration of the pipe as it is being laid from the barge to the sea bottom. See Figures 11 and 12. The "J" method is suitable for laying pipe in water depths greater than 350 m; in shallower waters, this method is too expensive relative to conventional or semisubmersible barge methods. Conventional lay barges typically operate in water depths up to approximately 200 m. Both of these systems rely on proven devices to control each phase of the construction procedure.

Deepwater lay vessels are sometimes dynamically positioned (possibly with tug assistance), eliminating the necessity for anchor lines and winches. They have high mobilization and operating costs, however, and their use can generally be justified only where less sophisticated pull methods cannot be used.

The recent introduction of dynamically positioned reel pipelay ships provides an extension of conventional stinger pipe laying.

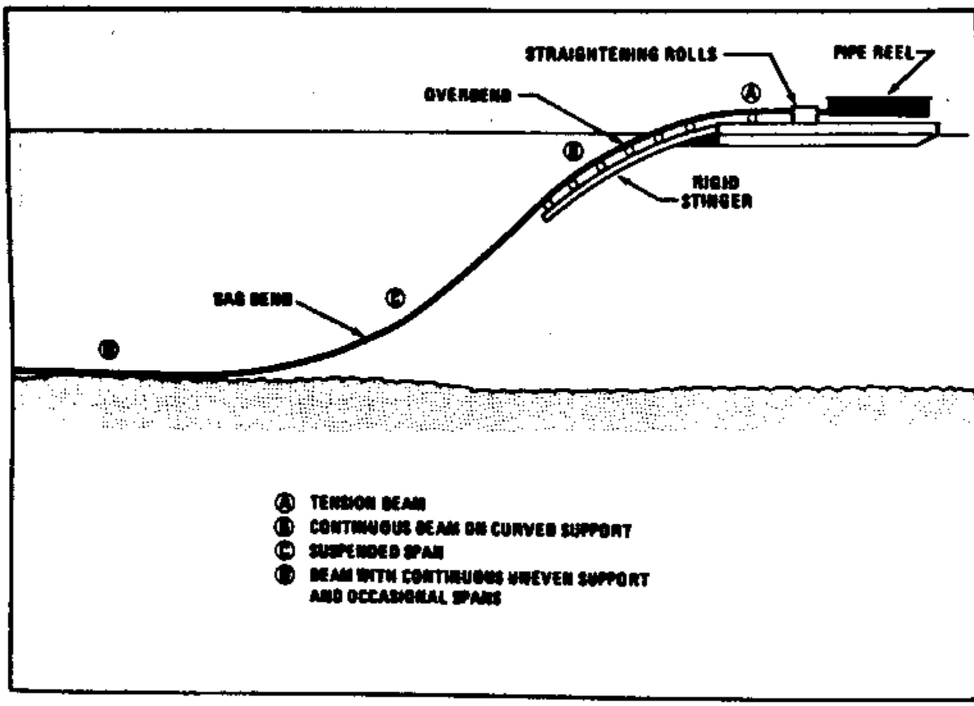


FIGURE 11 Reel barge, pipe configuration.

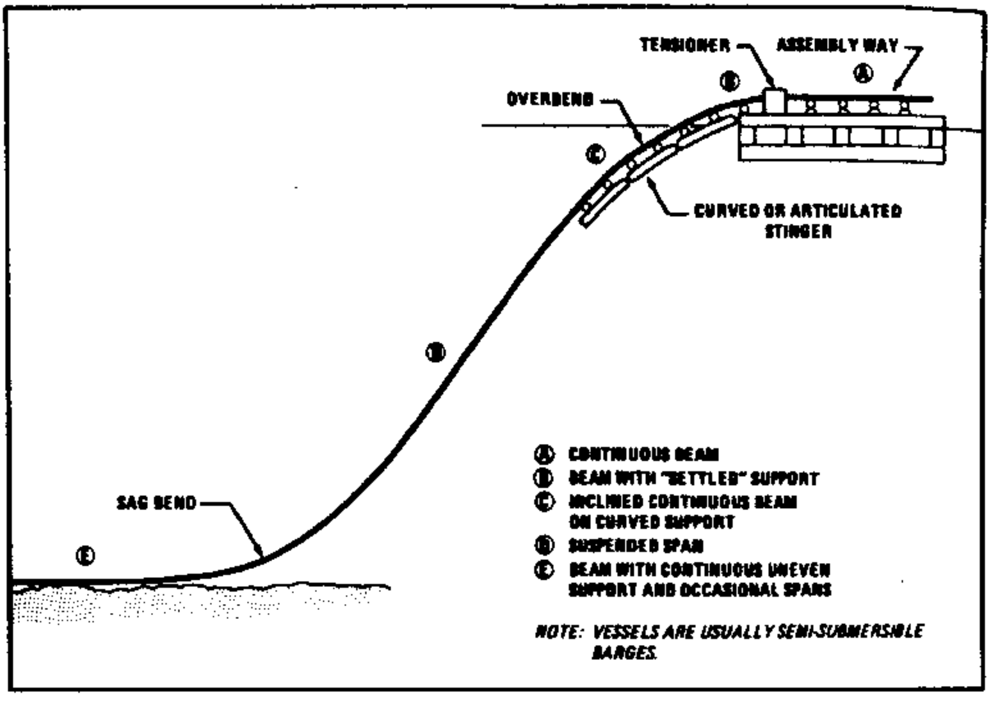


FIGURE 12 Deepwater lay barge, pipe configuration.

ASSUMPTIONS UNIQUE TO PIPELINE TRANSPORT SYSTEMS

The computer model to make comparative cost studies for pipelines was a modification of a Bechtel Petroleum, Inc. model used regularly in its analysis of submarine petroleum pipelines.

It was necessary to make certain assumptions, in addition to those common to both pipelines and surface transport systems noted earlier in this report, to analyze pipeline alternatives. Unlike the surface transport systems, the pipeline cost analysis program was required to account for differing methods necessary to lay pipe of varying sizes and lengths in varying water depths. Additional assumptions unique to the pipeline systems were:

1. Units of Measure
   - Throughput rates: dry metric tons/day and million U.S. gallons/day
   - Pipeline diameters and wall thicknesses: inches
     Other: metric

Pipeline System

- Pump station adjoining sewage treatment plant
- No booster pump stations included at sea
- No storage tanks required
- Electric-driven pumps installed in parallel (no spares)
- Underwater carbon steel pipeline
  Concrete-coated
  Cement-lines
- Pipeline buried from shoreline to 60-m depth

Pipe-laying Limitations:

Pipe outside diameter (in.)	Maximum water depth (m)
24	600
30	525
36	425
40	375
44	300
48	250

- No diffuser structure provided (i.e., open-ended pipe discharge)

Hydraulic Analysis

- Sludge composition (weight): 2 percent solids, 98 percent water
- Homogeneous flow in turbulent regime at velocities of 1.5-3.0 m/s
- Viscosity of sludge: 4.5 centistokes
- Load factor: 95 percent
- Internal lining thickness: 0.375 in.
- Pipe roughness: 0.015 in.
- Pump efficiency: 80 percent

Capital Cost Estimates

- Pipeline construction market supply/demand factors in balance
- Facilities included: pump station and underwater pipeline
- Pipe characteristics:
  Carbon steel
  Less than or equal to 20-in. outside diameter:
    Seamless
    Grade X52
    Standard wall thicknesses, subject to maximum diameter-to-wall thickness ratio 40 to 1
  Greater than 20-in. outside diameter:
    Double submerged arc welded
    Grade X60
    Target diameter-to-wall thickness ratio: 40 to 1
  Two-inch concrete coating
- Pipe buried by jetting-in method
- The most advanced large lay barges are required when water depths exceed 200 m; in water depths less than, or equal to, 200 m. smaller lay barges suffice

6. Operating and Maintenance Expense Estimates
   - Labor required: one person on duty at all times
   - Maintenance by contract
   - Electric power costs: mid-range for U.S. coastal communities
   - Exclusions: administrative and general expenses

7. Cost Analysis
   - Basis of comparison of transportation modes in cumulative discounted cost of service (CDCS) over the life of the project.

RESULTS OF COST ANALYSIS

A computer program was developed that would: perform a hydraulic analysis, size the major pipeline facilities required, calculate capital costs and annual operating and maintenance expenses, and derive transportation costs. The analysis consists of three sections:

- Design information;
- Capital cost information; and
- Operating expenses and transportation costs.

Design information comprises input and calculated data. Among the calculated data are the total installed horsepower required and the total pipe tonnage required. The capital cost information includes input unit costs for the given pipe size and water depth, and calculated capital costs. Operating expenses and transportation costs include input unit expenses, calculated annual operating and maintenance expenses, and calculated transportation costs.

The transportation costs produced by running the computer program are plotted using log-log scales in Figures 13 and 14. These plots show unit transportation costs for the two generic cases of sea floor slope. Figure 13 is for -0.1% slope while Figure 14 is for -3.3% slope typical of the west coast.

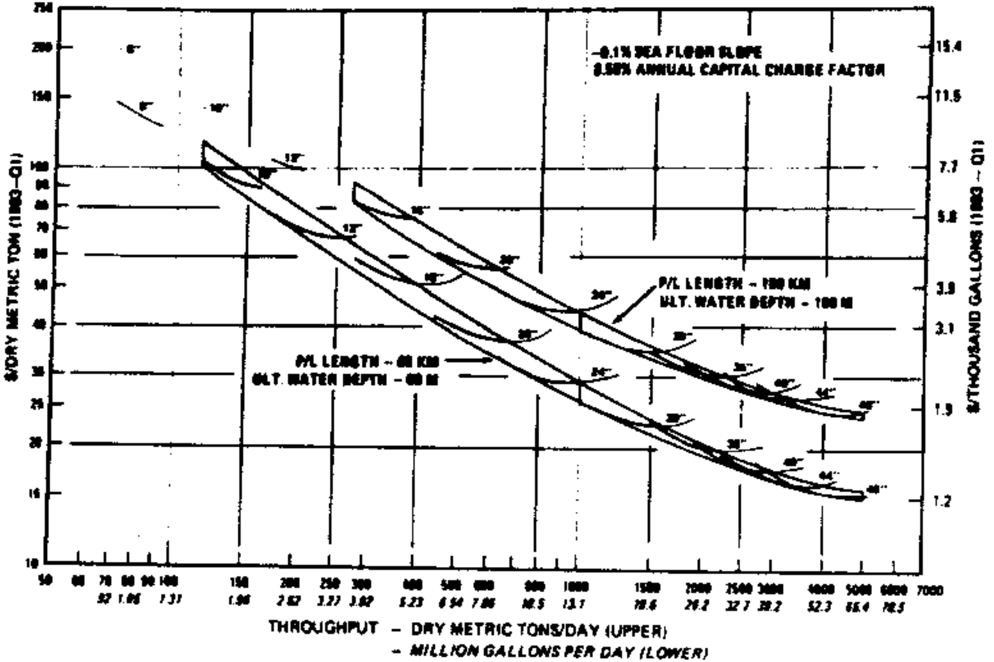


FIGURE 13 Unit transportation costs for disposal of sludge at sea by pipeline (-0.1 percent seafloor slope), pipeline lengths of 60 and 100 km.

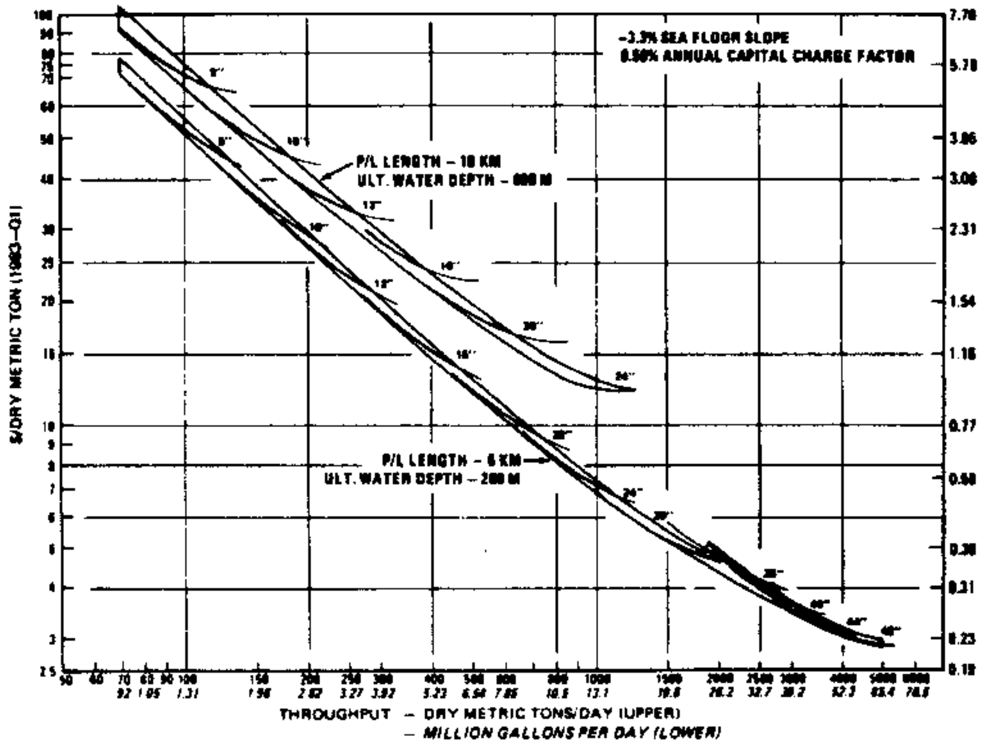


FIGURE 14 Unit transportation costs for disposal of sludge at sea by pipeline (−3.3 percent seafloor slope), pipeline lengths of 6 and 18 km.

Finally, Figure 15 presents unit transportation cost data versus dry sludge throughput in the same format and over the same throughput ranges as presented in Figure 8 for surface transport systems.

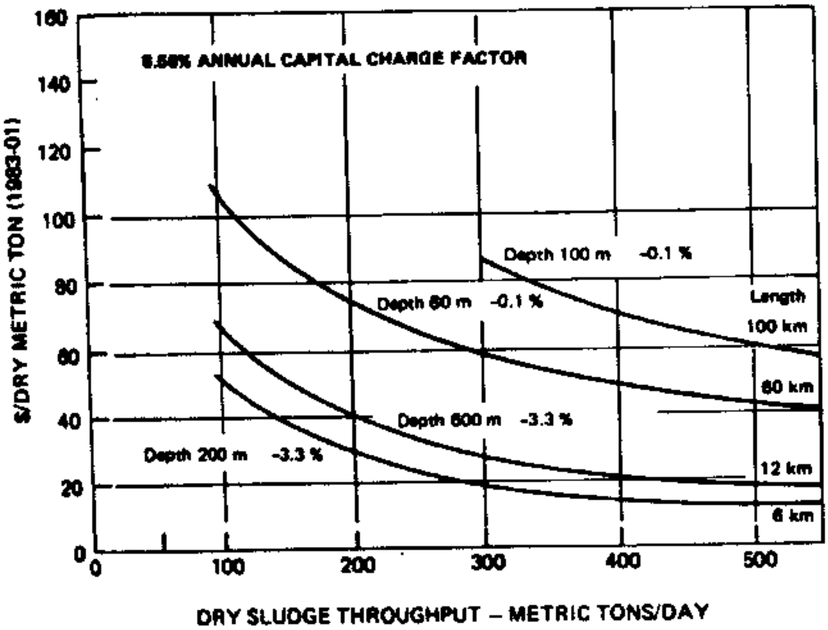


FIGURE 15 Comparison of pipeline systems unit transportation cost in dollars per dry metric ton versus dry sludge throughput.

Pipeline Pigging (Periodic Internal Cleaning)

To enhance pipeline efficiency it is common pipeline practice to pump "pigs" (i.e., a short cylindrical or spherical unit slightly smaller than the inside diameter of the pipeline) through the line as a scheduled maintenance activity. For this study pipelines are assumed to be open ended so that the pig can be discharged and will float to the surface for recovery after passing through the full length of the line.

The majority of the material deposits that build up on the inside of pipelines are comprised of grease and grit. The floatables (principally grease), after pigging, are picked up on the ocean surface; grit falls to the seabed. The infrequent discharge of materials resulting from pigging poses no significant environmental problem.

COST COMPARISON OF SLUDGE TRANSPORT SYSTEMS

For the purpose of comparison, the unit cost data for both surface transport and pipeline were plotted on log-log scales using the present value (PV) unit costs versus throughput.

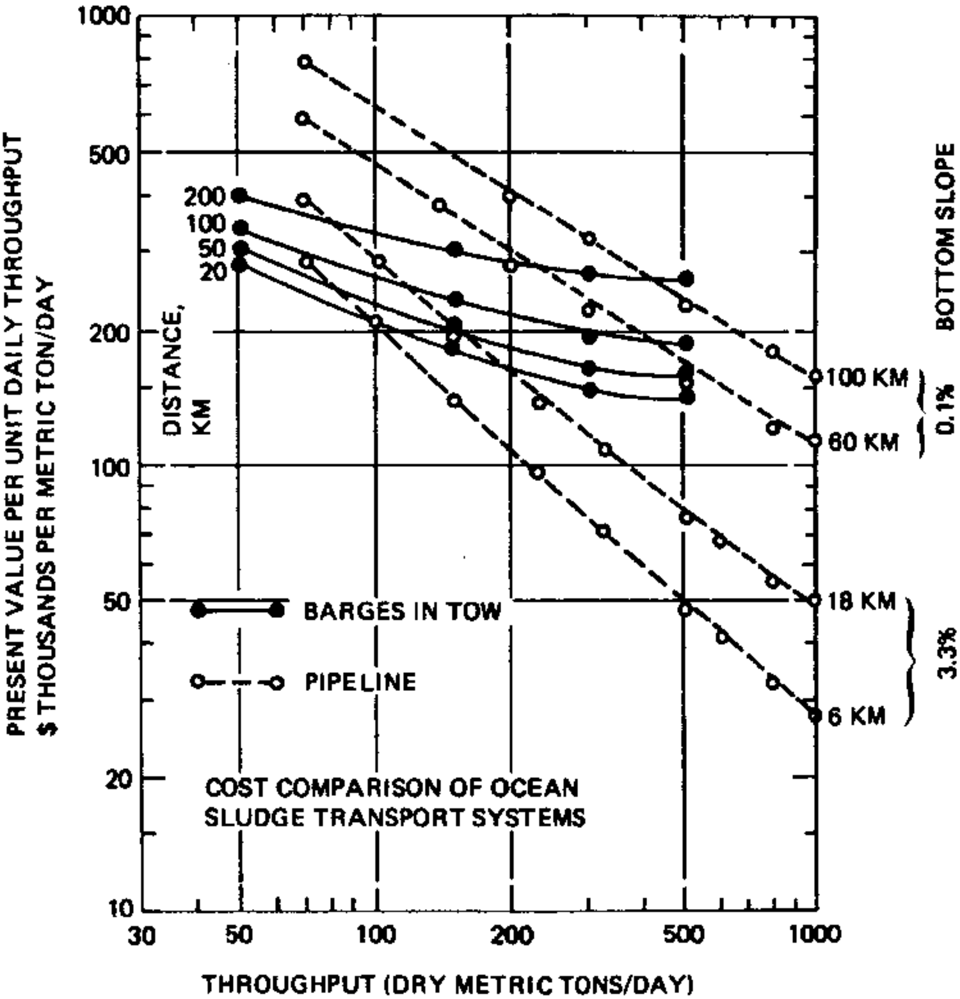


FIGURE 16 Cost comparison of two ocean sludge transport systems: pipeline and barges-in-tow, the least expensive of surface transport options. ("Present value per unit daily throughput" is equal to the present value of 25-years' operation divided by the average daily throughput in metric tons per day.)

° The curves in Figure 16 show clearly that barge systems are less cost for lower throughputs and longer distances, while pipelines are less cost for higher throughputs at shorter distances.

° Surface transport systems approach a constant unit cost at throughputs of about 300 to 500 metric tons per day, indicating no further economies of scale with increased throughput. However, pipeline unit costs continue to drop sharply with increasing throughput (over the range studied) indicating that the limits of economies of scale have not been reached.

° It is easier to expand a surface transport system over a wide throughput range (say half an order of magnitude or greater) than a pipeline system.

° Surface systems afford great flexibility in location of disposal. The pipeline is relatively inflexible in this regard.

° A pipeline system sized for sludge disposal for a U.S. coastal community is limited to approximately 100 km with an onshore pumping station. Longer distances are possible, but may require one or more at-sea pumping stations.

° Surface transport systems are subject to weather conditions, whereas submarine pipelines are immune to weather restrictions.

° A pipeline system has less surface visual impact than a surface transport system.

CONCLUSIONS

The principal conclusions of this study are:

1. The present knowledge of coastal oceanography and of environmental engineering is sufficient to support the design of ocean disposal systems for treated wastewater and sludge to meet environmental objectives.

2. Sewage treatment plants and ocean disposal facilities are complementary parts of the overall disposal system.

3. Effluent and sludge contain similar contaminants for which behavior in the ocean is predicted on the basis of the same physical, chemical, and biological principles.

4. Sludge particles behave physically as fine natural particles in the marine environment.

5. Although considerable progress has been made in the capability of modeling the transport, fates, and effects of ocean discharges into the ocean, this ability can be significantly improved as more research results are obtained.

6. The trace contaminants of environmental concern in sludge are primarily metals and xenobiotic organic compounds.

7. Trace contaminants appear to be chemically and physically associated primarily with solids in the waste stream.

8. The fate and effects of xenobiotics in the ocean are not well understood. Therefore, their disposal should be minimized.

9. Certain xenobiotics are so persistent that they cannot be removed in treatment plants or diffused to safe limits in the ocean.

10. For disposal of sludge, containment in specific ocean sites is not technically feasible. Therefore, the opposite strategy of wide dispersal, at adequate offshore distances, is the best way to minimize environmental degradation and potential pathways of pathogens and contaminants to humans.

11. A key design criterion for disposal systems is the deposition rate of sewage and sludge particles on the seafloor.

12. Suitable sludge disposal sites or areas exist over wide regions of the continental shelf and slope.

13. Costs of surface transport are lower for smaller throughput and longer distances, while pipeline costs are lower for higher throughput and shorter distances.

REFERENCES

Ocean Disposal Systems for Sewage Sludge and Effluent, Report of Committee, on Ocean Waste Transportation, Marine Board, Commission on Engineering and Technical Systems, National Research Council, National Academy Press, 2101 Constitution Avenue, N.W., Washington, D.C. 20418; 1984: 126p. (91 References).

Environmental Monitoring and Assessment

Public Wastes and Public Health

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ABSTRACT

The results of the prospective epidemiologic studies on acute gastroenteritis (AGI) associated with swimming in sewage contaminated marine waters and the more than 45 shellfish-associated outbreaks of the same illness reported from New York State in 1982 and 1983 suggest that people are becoming ill from contact with presumably acceptable waters via these two major waterborne routes of transmission. In both situations, a more active disease surveillance program for this relatively benign and nonreportable illness was responsible for the findings. On the other hand, the epidemiological record indicates that currently used wastewater disposal and resource utilization strategies have largely eliminated typhoid fever and infectious hepatitis, respectively, as diseases of major concern via these two routes. The relatively high illness rates and the relatively benign nature of the illness suggest decisions on resource utilization and wastewater disposal strategy based on risk acceptability.

At least along the Atlantic coast, where the option for long deep outfalls generally does not exist and where there often are combined sewer systems, there is a heavy dependence on wastewater treatment and disinfection, the former in part to physically eliminate larger sewage particles and the latter to kill pathogens and indicators in those particles that

remain. Because of the environmentally unstable nature of the coliform indicator systems used in arriving at these strategies, we probably have a false sense of security concerning their effectiveness relative to AGI. Thus the extent to which the available information on the effectiveness of disinfection and die-off during transport in water and in the sediments can be relied upon needs to be evaluated. The latter can be especially important in the shellfish growing waters in estuaries and embayments subject to combined sewer overflows. During the next decade we have to examine and, as required, modify these strategies. At least one state, New Jersey, is moving to disinfected secondary treated effluents discharged from "longer" ocean outfalls as a solution.

INTRODUCTION

It has been more than a decade since the "Clean Water Act" (P.L. 92-500) was enacted to accelerate the improvement of the environment with regard to public health and ecological considerations. Considerable research towards these ends has been performed; some of the findings from the research have been applied with salutary effects; but much remains to be done in both areas. This review will attempt to define the research needs* for the next decade with regard to the public health consequences of the ocean disposal of wastewater and sludge in the United States. The consequences include acute and chronic disease due to an increasing number of chemical contaminants which are or could be present in municipal wastewaters and sludge and which reach man through the consumption of marine biota. A consideration of these consequences is beyond the scope of this paper and the expertise of the author.

Additional pharmacological data on the levels of the contaminants which produce chronic disease, information on additive or synergistic effects of the chemicals, and data bases for extrapolating the unacceptable levels of the toxicants in the biota consumed as food back to corresponding levels in wastewater and sludge remain as critical research needs. The correlative technological and administrative needs are the development of cost-effective methods for the pretreatment of industrial effluents to remove these anthropogenic chemicals and the promulgation and enforcement of standards to insure their removal.

The public health consequence to be considered herein is the potential for infectious disease due to pathogenic agents which are present in fecal wastes and, hence, in municipal

*For emphasis, research and, in some cases, technological and administrative needs will be underlined.

wastewater and sewage sludge. There are two major routes by which these agents reach man. They are direct contact with the water, primarily during recreational activities such as swimming, surfing, scuba diving, etc., and the consumption of raw or marginally cooked molluscan shellfish. In the United States, enteric diseases such as typhoid fever, hepatitis A and gastroenteritis due to ingestion of the agents in the contaminated water or in the biota are or have been the major consequences of such discharges. In addition, there has been increasing concern over the past three decades with the potential for human disease due to infectious agents which are autochthonous to the marine environment. The agents are members of the genus *Vibrio*; and the illnesses reported include infected wounds via contact with the water and enteric disease through the ingestion of contaminated crabs and molluscan shellfish. There have been relatively few cases and outbreaks of these diseases; however, the illnesses are severe and, at times, fatal. Since the infectious agents are autochthonous to the marine environment, the potential for disease cannot be indexed by the fecal bacteria used as water quality indicators. This notwithstanding, the water levels of one of the agents, *V. parahaemolyticus*, are increased by municipal wastewater discharges, presumably because of nutrient enrichment of the water (Watkins and Cabelli, 1985). Additional information is needed on the extent to which wastewater and sludge discharges increase the potential for such diseases.

The approach to be taken in forecasting the research needs in the period from 1984 to 1994 will be to examine three assumptions prevalent in the decade from 1964 to 1974 in the light of information obtained from 1974 to 1984. These three assumptions have served as the basis for wastewater treatment and disposal strategies and the regulation of the affected resources, at least along the Atlantic and Gulf coasts of the United States where the option for long, deep ocean outfalls for sewage disposal is much more limited than that along the Pacific coast. The validity of these assumptions is especially critical in the vicinity of the large coastal population centers in the East and Southeast since most of the cities are located on relatively shallow estuaries and embayments into which large volumes of sewage are discharged via relatively short outfalls. Furthermore, these same receiving waters contain much of the shellfish resource and constitute a much needed source of outdoor recreation, especially for individuals living in the "inner cities."

ASSUMPTIONS OF CURRENT WASTEWATER DISPOSAL STRATEGIES

The first assumption is that the risk of infectious disease via the two transmission routes from waters which meet the existing water quality standards is negligible or, at least, undetectable. Its corollary is that the water quality standards and monitoring programs, including the choice of microbial indicators, used to evaluate the quality of the potentially affected water resources are effective. The second is that, because of disinfection, wastewaters could be "safely" discharged through relatively short outfalls into estuaries, embayments, and near coastal waters. The third, which in a sense paraphrases the other two, is that all coastal and estuarine waters could be rendered both fishable and swimmable with state of the art technology (secondary treatment followed by chlorination) and that this can be done at costs which would not be prohibitively expensive. It would seem that this assumption is implicit to the provisions of Section 101(a)(2) of the Clean Water Act as amended in 1972 (P.L. 92-500).

All three assumptions were probably correct with regard to the potential for the enteric diseases of most concern at the time the wastewater disposal and resource utilization strategies were being developed, that is, the bacterial diseases typhoid fever and bacilliary dysentery and possibly hepatitis A caused by a virus. However, it is now reasonably clear that the assumptions are not valid with regard to the most prevalent water-related illness, an acute gastroenteritis of viral etiology (Cabelli, et al., 1983). In addition, although most coastal and even estuarine waters can be made swimmable and fishable with regard to the potential for infectious disease, in most cases this will require the acceptance of some measurable risk of acute gastroenteritis; and, in many instances, currently used wastewater disposal strategies may have to be modified.

RISK OF WATER-RELATED AND WATER QUALITY STANDARDS

Swimming-Associated Disease

As late as 1974, many public health officials and environmentalists believed that there was little, if any, risk of swimming-associated, pollution-related illness except in waters so grossly contaminated with fecal wastes that they were aesthetically undesirable (Moore, 1975). Because of this, they questioned the need for recreational water quality guidelines and standards based on fecal indicator levels in the bathing water such as those recommended in 1968 by the National Technical Advisory Committee (NTAC) to the Federal Water Quality Administration (Henderson, 1968). In general, even those individuals who believed in the necessity for such guidelines and standards felt that those recommended by NTAC, later adopted by the USEPA (1976), and subsequently promulgated by most states as standards were sufficiently restrictive so that the risk of such illness was negligible. The results of the prospective epidemiological studies conducted by the United States Environmental Protection Agency over multiple years at several locations argued against this assumption (Cabelli, 1980). These studies clearly showed that measurable and appreciable rates of a swimming-associated, pollution-related illness, acute gastroenteritis, occurred at beaches which met the existing USEPA recreational water quality guidelines and local standards, and these results were consistent with recent information on waterborne disease outbreaks (Cabelli, 1983). Two aspects of the findings from the studies have particular relevance to the issue of wastewater disposal. The first is the relatively benign nature of the illness and the absence of evidence for more serious swimming-associated, pollution-related disease. The second is that swimming-associated acute gastroenteritis rates of about 10 cases per 1000 swimmers were associated with very low enterococcus (the water quality indicator of choice) or Escherichia coli levels in the bathing water, about 10 colony-forming units (CFU)/100 ml (Figure 1).

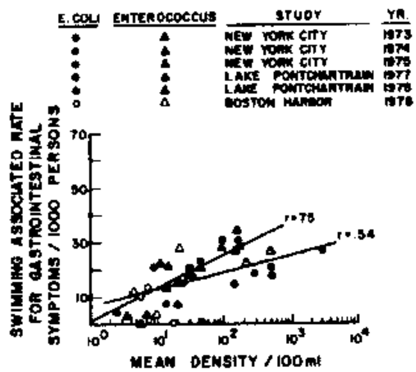


Figure 1. Regression lines for enterococcus and E. coli in the bathing waters against the swimming-associated rates of acute gastroenteritis (highly-credible gastroeintestinal symptoms) as obtained from epidemiological studies.

The attainment of enterococcus levels of about 10/100 ml in the bathing water would require more than a three log reduction from those levels found in secondary effluents from sewage treatment plants (Miescier and Cabelli, 1982). Without recourse to wastewater disinfection such reductions would be very difficult to achieve with relatively short outfalls discharging into estuaries or embayments, especially when the recreational resources are near the discharge points and large volumes of sewage are discharged into the receiving waters. Even with these reductions, however, illness rates of about 10/1000 swimmers would have to be accepted, a distinct possibility considering the relatively benign nature of the illness in question. Since economic and sociologic factors properly enter into decisions on the acceptability of the risk of swimming-associated disease, it is anticipated that, during the next decade, these decisions and the water quality and effluent guidelines and standards which derive from them will increasingly be made at the local level, albeit with review by the responsible federal officials. The economic and sociologic data bases and procedures for using them in arriving at such decisions need to be developed. A recent report examines the economic consequences of foodborne gastroenteritis (Archer and Krenberg, 1985).

Shellfish-Associated Disease

As early as the turn of the century, it was clearly recognized from reported disease outbreaks that the consumption of raw molluscan shellfish harvested from sewage-polluted waters carried a significant risk of typhoid fever and cholera (Verber, 1984). It was not until 1925, however, following an extensive outbreak of oysterborne typhoid fever, that the United States Public Health Service recommended a program for the sanitary control of the shellfish industry in the United States (Frost, 1925). Included in the recommendations was a limit on the *Bacillus coli* levels (they were probably referring to total coliforms) in the growing water. The manner in which the limit was expressed was both vague and variable until 1935 when it was stated in a manner from which an average *B. coli* MPN in the overlying water of 70/100 ml could be derived (see Furfari, 1968). In 1941, Kehr, et al., from their analysis of the 70/100 MPN coliform limit, concluded that this standard, though arbitrary, would provide the necessary protection against shellfishborne typhoid fever. They noted, however, that "--- the limiting standard should be more accurately evaluated by extensive studies at the earliest opportunity." The proposed standard was adopted in 1946 (USPHS, 1946) and has remained unchanged with two exceptions. They were the addition of 90 percentile limits and extrapolation to fecal coliform limits based on the comparison of total to fecal coliform (a more, but not completely, fecal-specific indicator) levels at the closure lines as reported by Hunt and Springer in 1974.

Three aspects of the development of the coliform standard should be noted as background for the information on the outbreaks of shellfishborne disease shown in Figures 2 and 3. The first is that the standard was developed and "evaluated" in response to outbreaks of typhoid fever, the etiologic agent of which was subsequently shown in human volunteer studies to have a high infectious dose (an ID_{50} of about 10^5 organisms)(Hornick et al., 1970). The second is that the choice of the coliform indicators and the specific limits on their levels were arbitrary with no epidemiological data to support them. The third is that the studies recommended by

Kehr et al. have never been conducted even after typhoid fever was replaced by infectious hepatitis and then acute gastroenteritis as the most frequently reported shellfishborne disease in the U.S.

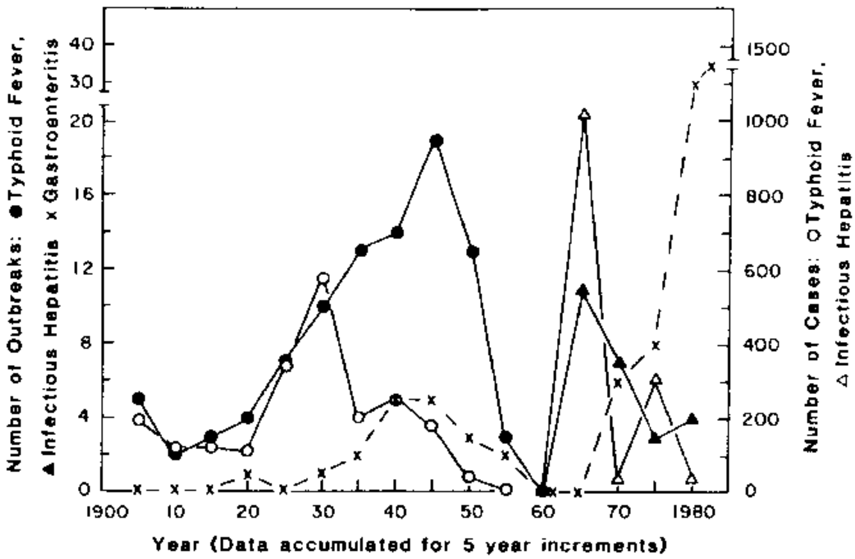


Figure 2. Shellfish-associated outbreaks and cases of typhoid fever, hepatitis A, and gastroenteritis reported from the United States in the period 1900 through 1980. Numbers accumulated for five year intervals. Data from Verber (1984).

It can be seen from Figure 2 that the number of reported shellfishborne outbreaks of typhoid fever peaked in the five years between 1940 and 1945 and sharply decreased in the ensuing decade. The combination of a number of factors may have been responsible for the decrease, including better removal of solids and subsequent disinfection during sewage treatment, fewer individuals excreting salmonellae into sewage via fecal wastes, improvements in the requirements of the "Shellfish Sanitation Program" as recommended by the USPHS in 1946, and the implementation of these requirements by the states. The elimination of shellfishborne typhoid fever notwithstanding, once the link between the consumption of raw shellfish harvested from sewage polluted waters and hepatitis A was established in 1957, a number of shellfishborne outbreaks of this disease were reported. The number peaked in 1964 and decreased during the ensuing decade. The last major outbreak of about 295 cases took place in 1973 (Portnoy et al., 1975); and one explanation for its occurrence was that, following a prolonged and extensive run-off of polluted fresh water off the coast of New Orleans, the viral agent persisted in the shellfish long after the coliform indicators disappeared from the shellfish and even their overlying water. The decrease in the number of outbreaks and cases of this disease must be attributed, at least in part, to improvements in the Shellfish Sanitation Program, specifically to better classification of growing areas, since the standards had not been changed appreciably since 1946. Improvements in wastewater treatment and disposal probably also contributed to the decrease.

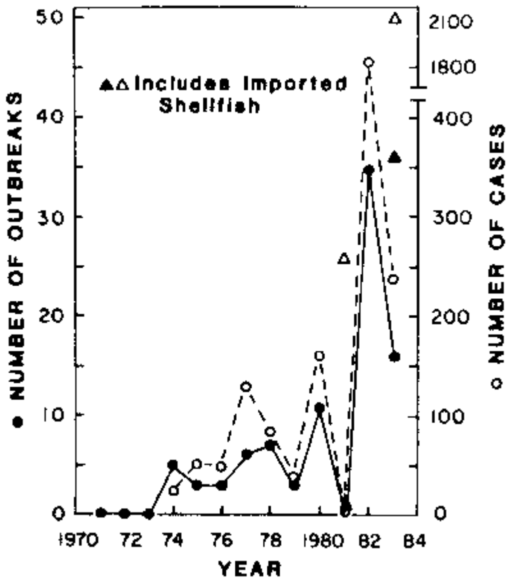


Figure 3. Reported shellfish-associated outbreaks and cases of gastroenteritis in the United States by year for the period 1970-1983. Data from Verber (1984) and internal reports of New York State outbreaks.

Coincident with or shortly following the increases in the more serious, reportable illnesses, there also were increases in the number of reported outbreaks of gastroenteritis. As noted earlier, this illness also is the one most commonly associated with other waterborne transmission routes. It can be seen from Figures 2 and 3, that there was a marked increase in the number of reported outbreaks and cases of shellfishborne acute gastroenteritis in the period from 1965 through 1980. This was undoubtedly due to a greater awareness of shellfish consumption as a transmission route for this relatively benign illness which, understandably, need not be reported to local or federal public health authorities.

More than 45 of the outbreaks shown in Figure 4 for the years 1982 and 1983 came from domestic clams sold in a single state, New York. They followed an outbreak in which two of 18 gastroenteritis cases subsequently developed hepatitis A

(Centers for Disease Control, 1982). Public awareness of the problem due to continuing media coverage of warnings issued by public health officials was largely responsible for the number of reported outbreaks. The extent to which such outbreaks go unreported can be seen from the observation that, although the clams could have come from New York, Massachusetts, Rhode Island and North Carolina, there were no reports of shellfish-associated disease from the other three states or from any of the other New England or Middle Atlantic states. In 1982, the attack rates for most of the gastroenteritis outbreaks exceeded 50 percent; there were 11 cases of hepatitis A; and the New York State Commissioner of Health issued a well-publicized warning against the eating of raw clams. Nevertheless, in 1983, there were 17 outbreaks and over 240 cases of gastroenteritis reported by New York State due to consumption of domestic clams and an additional 13 outbreaks and over 1850 cases reported by New York and New Jersey due to the consumption of imported clams. It would appear that a sizeable number of raw clam consumers believed that the 1982 outbreaks were local aberrations possibly due to bootlegging, did not feel that the warning was justified and/or decided that the risk of this illness was acceptable.

Two conclusions can be drawn from the analyses of the reported outbreaks of shellfishborne disease. The first is that the number of outbreaks and cases of shellfish-associated gastroenteritis reported markedly underestimate the actual number. The second is that the improvements in wastewater treatment and resource management which eliminated or markedly reduced typhoid fever and hepatitis A outbreaks, respectively, were considerably less effective in preventing shellfishborne gastroenteritis. Deficiencies in any of three components of the Shellfish Sanitation Program could have been responsible for the outbreaks -- the growing area standard, current practices whereby the standard is used in the classification of growing areas, and enforcement of area closures. There is ample justification to question both the growing area standard and area classification. Nevertheless, the last deficiency, the consequences of which would be the entrance of illicitly harvested, polluted shellfish into commercial channels, is all too frequently used to explain disease outbreaks. The first step to be taken in correcting this situation is an epidemiological evaluation of the microbiological growing-area standard and its replacement, if necessary. This, in turn, will provide the information necessary to examine the effectiveness of wastewater treatment and disposal strategies relative to the potential for shellfishborne infectious disease. The evaluation can only be conducted by a prospective epidemiological study analogous to the ones conducted with bathing waters. This is a critical research need. An epidemiological program jointly sponsored by the USEPA and NOAA has been initiated to obtain the needed data.

WATER QUALITY INDICATORS AND MONITORING STRATEGIES

Water Quality Indicators

The choice of the microbiological indicator generally affects the reliability of the water quality standard, and the numerical limits govern its restrictiveness. As noted in the previous section, the existing recreational and, by inference, shellfish fecal coliform water quality guidelines and standards are not restrictive enough to reduce the risk of swimming or shellfish-associated gastroenteritis to epidemiologically undetectable levels (Cabelli et al., 1982; Cabelli, 1982). As seen from the correlation coefficients for the indicator levels in the bathing water against the rates of swimming-associated gastroenteritis obtained in the bathing beach epidemiological studies (Cabelli et al., 1983), fecal coliforms are unreliable indicators and the enterococci are superior to them in this regard. Two requisites of an indicator bear upon its reliability. They are the extent to which its survival characteristics and sources correspond to those of the pathogens of most concern, the viral agents of acute gastroenteritis and hepatitis A. The enterococci are better than the fecal coliforms in both these regards (Fattal et al., 1983; Dufour, 1977; Levin et al., 1975), although there was an implication from one of the bathing beach epidemiological studies that, under conditions of prolonged residence in the aquatic environment, the viral pathogens survive better than the enterococcus indicators (Cabelli, 1981). Some field data recently obtained in our laboratory are consistent with this suggestion (data to be published). In these studies, certain bacterial viruses (F male-specific bacteriophages) consistently found in sewage in large numbers were used as simulants for the viral agents in examining the survival of the bacterial indicators relative to the viral agents and to the conservative tracer, *C. perfringens* spores. The survival of the phages was much better than that of the coliforms and enterococci and approached that of the *C. perfringens* spores.

Indicators of fecal contamination of the water rather than the pathogens themselves have been used to index the potential for infectious disease from wastewater discharges. There is ample justification for doing so, although, as noted by the author (Cabelli, 1978), there are limitations to the use of fecal indicators and the guidelines and standards based upon them. Except for situations where there is direct contamination of the environment with human feces, microorganisms which are not necessarily fecal but are exclusively and consistently present in wastewater discharges, which do not multiply or propagate in the receiving water, and which meet the other requirements, (e.g. their survival characteristics) could also be used as health effects water quality indicators. The male-specific bacteriophages may be a case in point.

There is yet another problem associated with the source specificity of the indicator, and it has implications with regard to the treatment and impact of both rural and urban stormwater run-off. To the best of our knowledge, the viral agents of the swimming- and shellfish-associated diseases of most concern are only found in human fecal wastes. Salmonellae have been recovered from the feces of a wide variety of lower animals and have been responsible for disease outbreaks via both transmission routes. From the history of such outbreaks, however, it is clear that they were due to gross contamination of the water with human fecal wastes; and there is no evidence that salmonellosis has occurred via either transmission route by contamination of the water with the feces of lower animals. Clearly, the risk of disease from this source is considerably less than that due to contamination of the water with human fecal wastes. This notwithstanding, even the most fecal specific indicators used, *E. coli* and enterococci, are found in the feces of lower animals as well as man, and, because of this, the risks of disease from the two types of sources are equated in the classification of recreational and shellfish-growing waters. The consequences are the loss of the resource or treatment and/or disinfection of sewered stormwater. The resolution of this problem requires (1) data from epidemiological studies in which the shellfish-growing or bathing water is subject to appreciable contamination from lower animal but not human fecal wastes, (2) a human fecal-specific indicator, and (3) some general guidelines on how monitoring data for waters thusly contaminated can be used in their classification and in selecting options for the disposal of sewered stormwater.

Classification of Resources

Hydrographic and meteorological factors, such as tidal conditions, currents, wind direction and velocity, and rainfall - especially, as it affects stormwater run-off and combined sewer overflows - generally produce considerable variability in microbial indicator levels. Historically, this has led to 90 percentile as well as median or geometric mean limits on the indicator levels in the water, a procedure which is consistent with good public health practice. In general the 90 percentile limits are more restrictive. All too often, however, the 90 percentile limits have been ignored, the number of samples used in area classification has been too small for obtaining reliable 90 percentile values, and statistical methods have not been used in arriving at the values. The solution to this problem is the abandonment of median or mean in favor of 90 percentile limits on indicator levels in the water.

Although it is the shellfish that are consumed, their sanitary quality is determined from indicator levels in the overlying water (USDHEW, 1965). Furthermore, while the molluscan shellfish in question reside on or in the bottom sediments from which the viral pathogens could be accumulated long after they are absent in the water column, surface water samples are generally taken for the classification of shellfish-growing areas. Aside from convenience, there are three reasons why this is done. Two of them are attributable to defects in the indicator used. First, at temperatures less than about 10 degrees C, the coliform indicators "disappear" from quahogs, even from those in highly polluted waters as seen from the coliform levels therein; yet, shellfish-associated outbreaks of hepatitis A and acute gastroenteritis have been reported from temperate zones during the winter (Cabelli and Heffernan, 1970). Marked decreases in the feeding activities of the animals (clamming-up, as it were) followed by differential die-off of the fecal coliform indicators relative to the viral agent is the best explanation for this observation. A similar effect from marked decreases in salinity due to excessive freshwater inputs was probably responsible for the 1973 outbreak of shellfishborne hepatitis A (Portnoy et al., 1975). Second, total coliforms, fecal coliforms and, under certain conditions, even E. coli can multiply in the shellfish, even in the interval between landing of the shellfish and their delivery to the shipper (S.A. Furfari, personal communication). The results of studies comparing fecal coliform and fecal streptococci levels in the shellfish to those in the overlying water indicate better survival of fecal streptococci in the shellfish (Plusquellec, et al., 1984). Data from our laboratory not only confirmed these observations but also showed that the virus levels in the shellfish — in this case the F male-specific bacteriphages — relative to those in the water markedly exceeded those of both the coliforms and enterococci. In addition, the phage levels in the shellfish exceeded those of the enterococci and fecal coliforms by about one and two orders of magnitude, respectively, although the levels of the former in prechlorinated sewage effluents were considerably less than those of the latter two (Table 1). Greater accumulation as well as survival of the viruses in the shellfish relative to the bacterial indicators is probably needed to explain the data. Any conclusions to be drawn from these results, however, presuppose that the accumulation, survival and elimination of the phages in shellfish simulates that of the specific viral agents that cause acute gastroenteritis and hepatitis A. There are no hard data that address this question. The acquisition of such data is another critical research need since the levels of these specific bacteriophages in the shellfish themselves could be used in the classification of shellfish growing waters.

The third reason for examining the overlying water rather than the shellfish in the classification of growing areas is that different species of shellfish with differing abilities to accumulate bacteria from their environment can be found in the same growing area. If an indicator whose survival characteristics better mimics those of the viral pathogens is used, the solution to these problems could be the use of the indicator levels in the shellfish themselves in the classification of growing waters. This would decrease the variability due to tidal conditions, at least, and probably reduce the number of samples needed for classification. The system could be further standardized by conducting the assays on caged shellfish of a single species (e.g. mussels) placed on the bottom sediments for periods of 5-7 days. In any event, there is a need for research on the sampling parameters as well as the indicators to be assayed in order to best index the potential for shellfishborne hepatitis A and acute gastroenteritis.

EFFECTIVENESS OF WASTEWATER CHLORINATION

It has been known for some time that, in general, animal viruses survive chlorination better than do the coliform indicators (Scarpino, 1972). It has also been shown that those bacterial viruses, including the Fd and f-2 like male-specific bacteriophages, which can be assayed on the *E. coli* strain K12 Hfr are present in chlorinated wastewater effluents at appreciable levels when the coliform and even enterococcus levels in the effluents are no longer detectable or markedly reduced (Lupo, 1979). Subsequent studies showed that this was due primarily to the survival of the F male-specific phages in the sewage, especially the single-stranded DNA phages like Fd (McBride, 1979). Data recently collected in our laboratory from the examination of effluents from a number of sewage treatment plants in Rhode Island, Connecticut and Massachusetts confirmed that the survival of the male-specific phages following chlorination to total residual levels between 0.4 and 4.0 mg/Liter was markedly better than that of the coliforms or the enterococci and was not significantly different from that for *Clostridium perfringens* (Table 2).

If the hepatitis A and gastroenteritis viruses are similar to the male-specific phages in their resistance to chlorination as practiced at wastewater treatment plants (the answer to this question is a critical research need), radical changes will be required in the strategies which are employed for the treatment and disposal of wastewater discharged into estuaries and embayments. Some very important information that addresses this need has recently become available. From the results of human volunteer, feeding studies, Keswick et al. (1985) reported that the Norwalk virus was extremely resistant to chlorination, even when the hypochorous acid was applied to distilled water suspensions of the virus preparation and total chlorine residuals of 1.5-20 mg/L were obtained after a contact time of 30 minutes. "Inactivation"

Stn[b]	Date[c]	FC/100ml[d] in SW	Indicator level/100g of shellfish: C. perfringens	F phage	Enterococci	FC	Ratio of shellfish/BW indicator levels[e]: C. perfringens	F phage	Enterococci	FC
B	1/27,28	31.5	80.0	225	ND	4.0	0.83	11.3	ND	0.20
	2/10,11	6.1	9.4	87.5	2.0	2.0	0.26	6.3	1.0	0.30
	2/17,18	22.0	31.6	150	2.0	2.0	0.37	21.8	0.63	0.14
	3/2,3	1.0	40.0	37.6	2.0	2.0	0.76	6.2	1.4	0.27
C	1/27,28	26.8	40.0	38.0	ND	2.0	0.23	3.0	ND	0.15
	2/10,11	2.0	30.0	87.5	2.0	2.0	0.75	2.0	1.8	0.31
	2/17,18	6.9	31.6	12.5	2.0	2.0	0.49	2.9	2.2	0.36
	3/2,3	1.3	20.0	62.5	2.0	2.0	0.37	4.8	1.1	0.07
$\bar{X}$							0.51	7.3*	1.4	0.23**
Pre-chlorinated effluents[f]			2.9×10^4	1.0×10^4	9.1×10^4	4.0×10^5				

[a]Area below permanent closure line whose management is based on rainfall (see text)
[b]Sampling station: B and C, one-third and two-thirds of distance into conditional area, respectively
[c]Status of conditional area: Closed 1/27-28; open 2/10-11, 2/17-18, 3/2-3 for shellfish harvesting, respectively
[d]Fecal coliforms (FC) in the surface water samples; geometric means of levels determined in 2-4 samples collected in the 36 hours prior to collection of the shellfish samples
[e]Denominator was geometric mean of levels in 2-4 bottom water (BW samples collected in 36 hrs prior to collection of shellfish samples)
[f]Geometric mean levels/100 ml in five samples each from the Providence and East Providence sewage treatment plants
*,**Significantly different from C. perfringens at the 95 + 99 CL, respectively

of the Norwalk virus was achieved, however, when total and free chlorine residuals of 8-10 and 5-6 mg/L, respectively, were maintained. As noted earlier, the Norwalk virus is the agent most frequently implicated in outbreaks of acute gastroenteritis. Two other findings are noteworthy. Poliovirus levels were reduced from 10^4 to 0, while those of the f-2 bacteriophage only decreased from 10^6 to 10^4. This phage is the least chlorine resistant (McBride, 1979) of the two groups of F male-specific bacteriophages for which data were given in Tables 1 and 2.

The good correlation obtained in the epidemiological studies between enterococcus levels in the bathing water and swimming-associated gastroenteritis rates (Figure 1) implies that the chlorine resistance of the enterococci is reasonably similar to that of the gastroenteritis viruses. This may be so; however, in most of these studies there were significant inputs of unchlorinated sewage into the water. <u>Thus, another critical research need is to conduct both shellfish and bathing beach epidemiological studies under circumstances where only chlorinated wastewater discharge impacts on the resource and where chlorination of the discharge is necessary for the resource to meet the existing water quality guidelines or standards.</u> The need for such studies is urgent because of large sums of money being spent on disposal strategies which depend on chlorination to achieve acceptable indicator levels in the receiving waters.

<u>Alternative Wastewater Disposal Strategies</u>

There are several wastewater treatment/disposal and resource management options which could be used as alternatives to the chlorination of wastewater effluents; and all of them have been exercised, if only in a research mode. The abandonment of those resources where the risk of water-related illness would not be acceptable without chlorination of wastewater inputs would probably not be a satisfactory option in most cases. Wastewater treatment systems which physically remove the viral agents as an alternative to chlorination are possible but expensive. The use of alternative disinfectants, such as ozone, which are more effective against viruses (Katzenelson et al., 1974) is a second option. A third possibility, which could be exercised when only a shellfish resource is affected, is relaying or "raftine-depuration" of the shellfish resource. Since the F male-specific bacteriophages are present in sewage at relatively high levels and are easily, accurately and inexpensively enumerated, they could be used as a model to evaluate the effectiveness of the alternative disposal strategies with regard to the elimination of the viral agents. <u>Rafting-depuration studies using the phages as simulants for the viral agents also are needed, as are studies on the physical removal of the simulants from wastewater discharges and their inactivation by alternate disinfectants.</u>

Reductions in F male-specific bacteriophage (viral simulant), *C. perfringens* spore, enterococcus, and *E. coli* levels following chlorination of wastewater effluents at sewage treatment plants.

Plant type[a]	N	Cl_2 resid.[b] (mg/L)	$\bar{x}$ for ($\log_{10}$pre-Cl_2 −$\log_{10}$post-Cl_2) levels *C. perfringens*	Phage[c]	Enterococci	*E. coli*
Pri,EA	5	3.7	2.034	0.959**	3.618**	3.672**
Sec,AS	5	3.6	0.525	0.447	3.600**	3.097**
Sec,AS	2	3.1	0.491	0.611	–[d]	–
Pri	2	2.9	−0.026	0.264	–	–
Pri	1	2.9	0.616	0.690	–	–
Pri	2	2.4	0.286	0.466	–	–
Sec,AS	5	2.0	0.735	0.479	2.456*	2.415*
Sec,AS	1	2.0	0.171	1.265	–	–
Sec,TF	2	1.8	0.864	0.936	–	–
	2	1.4	0.717	0.750	–	–
Sec,AS	5	1.0	0.640	0.406	4.044**	4.027**
Sec,AS	2	0.85	0.050	0.049	3.162**	4.689**
Sec,AS	6	0.45	0.079	−0.089	1.915**	2.340**
Sec,AS	1	0.00	−0.422	−0.451	−0.155	−0.070
			0.592	0.601		

- primary; Sec - secondary; AS - activated sludge; TF - trickling aeration.

l.

riophages.

s) with Cl_2 residuals > 0.45 mg/L.
ferent from *C. perfringens* at 95 and 99% CL, respectively.

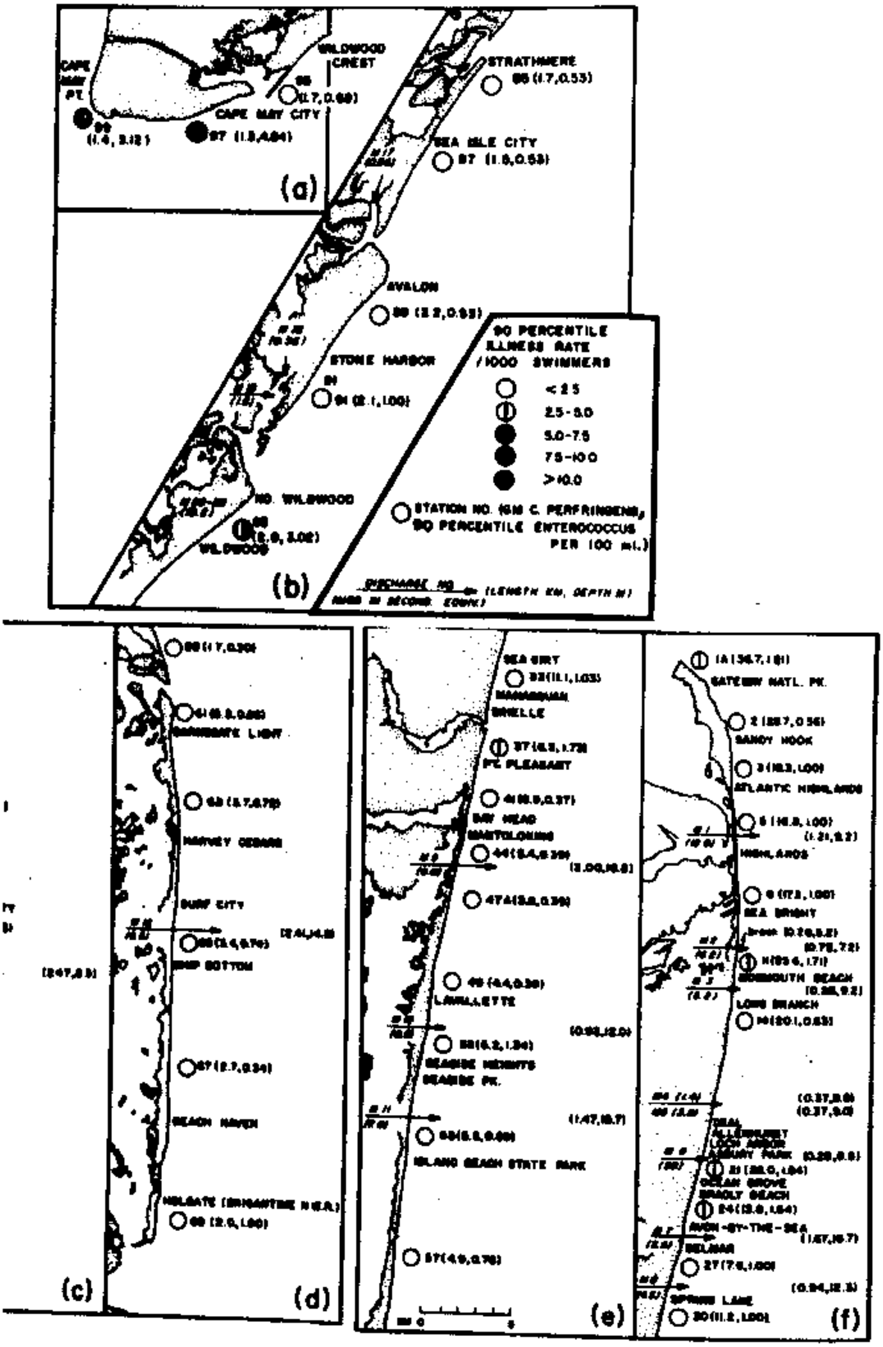
Another alternative to the chlorination, if chlorination is found to be ineffective against the viral agents, is disposal of the effluents through relatively long deep ocean outfalls. Although the excellent results obtained along the Pacific coast can not be expected along the Atlantic coast because of the differences in the length of the continental shelf, the requirement for disinfection can be minimized, if not eliminated. The effectiveness of this option can be seen from the data for the ocean outfalls (M-9 to M-13) along the central coast of New Jersey as given in Figure 4*. Although all the discharges are chlorinated, it can be seen from the comparison of _C. perfringens_ spore densities at the shoreline stations below and above the outfalls that disinfection of the secondary treated effluents would be minimally, if at all, required.

Ocean outfalls are a particularly attractive disposal option in those situations where combined sewer overflows into estuaries and embayments are a problem and there is not a critical need for the water resource present in the sewage. The use of this option does not diminish the need for pretreatment of industrial discharges to reduce their content of anthropogenic chemicals to levels that are acceptable both ecologically and with regard to public health. However, because coastal waters generally are better able than the estuaries to "assimilate" nutrients and anthropogenic chemicals in the wastewater discharges, the reductions required may not be as great. There are a number of estuaries along the Atlantic and Gulf coasts where this alternative to chlorination would be applicable.

The choice of the particular wastewater treatment and disposal strategy to be employed depends on a number of factors which vary both geographically and temporally. These include the need for using the water and possibly the nutrients in the wastewater as resources, the availability of terrestrial disposal sites, the extent to which the wastewater is contaminated with industrially derived anthropogenic chemicals, and, above all, the availability of funds. The following four case studies serve as examples of the importance of these factors. It is assumed in these studies that wastewater disinfection as practiced is only marginally effective.

Case Study - Hudson River Estuary. The problems which confront New York City in the disposal of its public wastes aptly illustrate the importance of those considerations which influence the selection of disposal options and suggest some additional research needs for the next decade. The problems

*The data given in Figure 4 were obtained in 1980-81; the characteristics and locations of the discharge derived from information obtained in 1980. There have been a number of changes since that time.



4. Enterococcus and C. perfringens spore levels in
water samples collected in 1980-81 from shoreline
beyond the surf zone) sampling stations along the New
coast in relation to wastewater discharges. Number of
occus values, 15-25; number of C. perfringens, 4.

include the requirement for disposal of wastes from a very large urban population located on an estuary along the Atlantic coast, the urgent need for the recreational resources located near the mouth of the estuary, the presence of combined sewer systems, significant industrial inputs into the sewerage system, and the scarcity of land for the disposal of solid wastes or sewage sludge. There also are chronic water shortages which probably derive in part, at least, from unmetered water use, a situation which probably will be corrected in the near future. During "dry years," however, the water shortages become acute.

The current program for upgrading sewage treatment facilities to the secondary level as mandated by law coupled with pretreatment of industrial effluents as needed will continue to improve the quality of water in the estuary, although a significant potential for swimming and shellfish-associated infectious disease will remain if the effluents are not adequately disinfected. Moreover, the problem of combined sewer overflows is not resolved by such treatment, and the quantity of sludge generated will increase.

Two alternative strategies are available; in both, the effluents are physically removed from the estuary. The first requires pretreatment of industrial inputs and sufficient, advanced treatment of the wastewater to produce an effluent whose quality is suitable for land disposal and compatible with immediate recreational use and subsequent use as a raw source of drinking water. It would be the ideal option with regard to ecological and public health considerations since the discharge would be removed from the estuary and the quality of the effluents would be such that the free chlorine residuals needed for efficient destruction of the viruses could be achieved. Considerable progress has been made in the development of the required treatment systems since the Santee Project (see Shuval, 1977). However, during the next decade, at least, the problem of combined sewer overflows probably will not be solved; and the capital and operating costs of such a water reuse system probably could not be justified, especially if there is to be enough reserve capacity to accommodate combined sewer overflows. Thus, it must be assumed that, in this case, such treatment systems are not yet cost-effective enough for competition with other wastewater disposal options. This could be altered by two changes; a critical need for the water resource or marked increases in the restrictiveness of the standards for the receiving waters. Since this is unlikely during the next decade, it is doubtful that this option will be exercised. Nevertheless, this may be the disposal option of choice during the early part of the next century; and research need to be continued during the next decade towards making such treatment systems more cost-effective using the tools of sanitary, biological and genetic engineering.

Another disposal option is the gathering of the secondary treated effluents, or possibly even those treated to the primary level under a 301(h) waiver, from the various treatment plants by a system of underground sewerage lines for disposal via a submarine outfall extending some 16-20 km out from Coney Island to the Christiaensen Basin. From there, the solids, at least, would be carried out the Hudson Shelf Valley towards the continental shelf (Cabelli et al., 1984). This option would resolve the problem of combined sewer overflows, reduce the degree of treatment required, virtually eliminate the risk of swimming-associated enteric disease without recourse to chlorination, and reclaim shellfish and recreational resources in and around New York City. Three other sources of pollution into the estuary would also have to be eliminated or markedly reduced with this and the preceding option. They are wastewater and/or industrial discharges into the estuary north of New York City, those discharges from communities on the west bank of the estuary, and those due to small pleasure and commercial boats and large sea-going vessels. The technology for this disposal option is available. The cost would be high but possibly justifiable when all the benefits are considered, especially the elimination of the costs of advanced or even secondary wastewater treatment and disinfection. Furthermore, this author would have allowed sludge dumping to continue at the 12 mile New York Bight sewage disposal site if the additional funds to be spent on transport to the 106-mile site during the next decade or so were used to implement this wastewater disposal option.

Case Study - Narragansett Bay Estuary. Many of the same problems which complicate wastewater disposal in the Hudson River Estuary also pertain to the Narragansett Bay Estuary. There are two notable exceptions. First, in the latter estuary, the size of the population whose wastes are discharged therein is much smaller. Second, there are very valuable and economically important recreational, shellfish and boating resources within the estuary. Since there is not a critical need for the water resource, the cost of the advanced wastewater treatment systems needed to achieve this end probably can not be justified. However, because of the importance of the estuarine resources, the removal of the wastewater discharges from the estuary via a relatively long ocean outfall may be a feasible disposal option. A strategy accepted by the shellfish sanitation program to allow the harvesting of shellfish when the coliform limits are exceeded only because of rainfall-induced combined sewer overflows is the establishment of a "conditional harvesting area." In Narragansett Bay, the conditional area contains much of the available shellfish resource. Utilization of this resource is managed by temporarily closing the area to shellfish harvesting for at least a week following a rainfall in excess of 1.27 cm. This interval of time is sufficient for the fecal coliform levels in the water and the shellfish to return to base-line conditions; and it is assumed that this is also true of the pathogens. Recent information obtained in our laboratory using the bacteriophages as simulants for the viral pathogens suggest that, with regard to virus levels in the shellfish, the assumption may be in error, at least during the winter when the feeding activity of the animals is minimal (see Table 1). As noted above, the costs of constructing the sewerage lines and submarine outfall would be high; but the economic benefits would be relatively greater than those in the Hudson River Estuary. In addition, the same inducements, 301(h) waivers, could be made available, at least for a decade or so. Finally, it would seem that a society that is willing to spend billions of dollars on roads to transport individuals in, around and out of our urban and suburban centers could also afford to treat and transport the wastes we produce to locations where they will be less harmful and better assimilated. <u>To the extent that the costs of constructing such sewerage systems and submarine systems can be reduced, this is a technological requirement</u>.

<u>Case Study - Southern California Bight</u>. Because of the short continental shelf, discharge through long, deep ocean outfalls is an attractive, relatively inexpensive, ecologically reasonable and acceptably "safe" option for the disposal of secondary and even primary treated effluents generated by coastal communities in Southern California. Southern California, however, has a chronic shortage of fresh water for domestic consumption, industry, agriculture and recreation. Thus, the hundreds of million gallons of wastewater effluent discharged daily into the ocean is a much needed but lost resource. <u>It would seem that the costs and control measures, notably those on the input of industrially derived anthropogenic chemicals, needed to produce effluents of sufficient quality for land application can not as yet be justified economically. More research on cost-efficient systems to accomplish this end is needed</u>. Coupled with this public health requirement is a second one, <u>the need for methods which would reduce the levels of infectious agents in the effluents to those which would not preclude their immediate agricultural and recreational use</u>.

<u>Case Study - Sludge Disposal</u>. The three marine sewage sludge disposal areas in the United States studied most extensively are the New York Bight (12 mile), Philadelphia and Southern California sites. In the first two instances, the sludge is barged to locations some 25 and 85 kilometers offshore into the Pacific Ocean. By court order, sludge disposal was terminated at the Philadelphia site several years ago and will be terminated at the New York Bight disposal area shortly. There is no epidemiological evidence that sludge disposal at

any of the three sites has produced human disease. The potential for swimming-associated disease at beaches along the New York Bight referrable to sludge disposal at the 12-mile site was examined with negative findings (Cabelli et al., 1976; Cabelli, 1982). When the C. perfringens spore levels in the bottom sediments were used to examine the movement of fecal pollution from the dumpsite towards the shore, it was found that this possible source was dwarfed by those emanating from the shoreline, notably the Hudson River Estuary (Cabelli and Pedersen, 1982; Cabelli et al., 1984).

It must be concluded, therefore, that the decision to close the two dumpsites derived from concerns over ecological effects, aesthetics and lost resources, including surf clams and ocean quahogs which could not be harvested because of public health-based shellfish area closures. The benefits to be derived are unmistakable and would seem to justify the decision. There are, however, two factors to be considerd, both of which have public health implications.

Most of the sludge formerly disposed at the Philadelphia site is treated further (e.g. composting) and applied to land in one way or another. To the extent that the sludges contain anthropogenic chemical inputs, this option may not be preferable to more widespread, and presumably less concentrated, dispersal in the ocean unless other means are found for the disposal of sludges which are unacceptably contaminated. On the other hand, a long-term benefit would be obtained if the decision to terminate disposal at this site leads to more extensive pretreatment of industrial inputs because of more rather than less stringent public health requirements on the "quality" sludges applied to land.

There is considerable logic beyond that which is immediately obvious to recommend the opening of the sludge disposal site off the continental shelf (the 106-mile site) concurrent with the termination of disposal at the New York Bight site. The availability of the 106-mile site for the disposal of unacceptably contaminated sewage sludges from the megalopolis between Boston, MA and Washington, D.C. should allow more stringent regulations on the quality of the sludges applied to land. On the other hand, the increased transportation costs for disposal at the 106-mile site should provide an incentive for pretreatment of industrial inputs to wastewater treatment and disposal systems so that the sludges produced would be acceptable for land application. Thus, if the decision to terminate sludge disposal at the 12-mile site is to be criticized with regard to environmental concerns, it could be done only with regard to priorities, timing and strategy. Considering the very intractable problem of combined sewer overflows in the Hudson River Estuary, would the additional funds needed for transport of the sludge to the 106-mile site over the next decade or so be better spent on a strategy which would remove all wastewater discharges from the estuary as noted above? This would seem to illustrate two administrative needs. The first is better communication between the various arms of the regulatory agencies. The second is a mandate to these agencies of a mandate to effect trade-offs such as this when a net environmental benefit would be obtained. The availability of wastewater treatment systems which are less expensive than conventional secondary treatment, yet effective enough for ocean disposal and the granting of 301(h) waivers in such cases would provide added financial incentives for removal of effluents from the estuaries.

THE ATTAINMENT OF FISHABLE AND SWIMMABLE WATERS

There is no question that all our estuarine and coastal waters ultimately could be made both swimmable and fishable. There are two more relevant questions, however. What risk of unwanted health and ecological effects would allow the waters to be designated as such, and how much is the public willing to pay both monetarily and in convenience (e.g. the land disposal of small boat wastes) to obtain incremental decreases in these risks? Once a decision has been made as to what risk of water-related infectious disease is acceptable, it becomes necessary to allocate the corresponding wasteload as measured by the appropriate water quality indicator among the potential sources of contamination and to define the appropriate treatment, disinfection and/or disposal strategy for each source (see Cabelli, 1983b). The most cost-effective way to accomplish this is through a site-specific transport model which considers both the biological decay of the indicator and its sedimentation along with the dilution parameters. The model must also be able to accommodate multiple contamination sources and be sensitive enough to budget wasteloads against water quality standards which are rather exacting in their regulatory application (e.g. the existing 90 percentile shellfish-growing water standard of 43 fecal coliforms/100 ml).

The author's rather limited experience has been that predictive models which satisfy all these requirements are unavailable, although considerable research has been done towards their development. A frequently used alternative to the use of such models has been the employment of "stream standards" as the effluent standards. The reduction in the indicator level needed to meet this requirement is achieved by chlorination of the wastewater effluents. This is consistent with good public health practice but should, in theory at least, lead to the unnecessary loss of valuable resources. In practice, the reductions in the downstream indicator levels predicted from those in the post-chlorinated effluents are not obtained at times because of so-called "regrowth" of the indicator. It would appear that this is largely due to environmental repair of chlorine-damaged cells. Field data being prepared for publication, in which *C. perfringens* spores were used as a conservative tracer, suggest that the effectiveness of wastewater chlorination on the enteroccus and coliform indicators is overstated by the use of methods which do not recover the damaged cells from the post-chlorination wastewater effluents. Better methods for the enumeration of bacterial indicators in chlorinated sewage effluents is another research need.

One of the problems with the models is the absence of reliable, site-specific biological decay coefficients for the indicators (see Chamberlin and Mitchell, 1978). The development of technology for obtaining them is another research need. There is a distinct possibility that, because of the vicissitudes of transport as they affect the microbial indicators, predictive models which satisfy the requirements for microbial wasteload allocation will not be available in the foreseeable future, although their development is another research need. Moreover, the hedge, as it were, provided by wastewater chlorination may not be as effective as thought. The needed response seems to be flexible treatment and disposal strategies which permit incremental increases in their effectiveness as required.

As noted earlier, it has been more than a decade since the Clean Water Act was enacted to accelerate the improvement of our environment. The cost-effectiveness of what has been accomplished in achieving the goals of the legislation with regard to public health and improvements still required need to be examined. The research needs associated with this requirement are some monitoring parameters, which are indexed so that they can be easily understood and which are amenable to cost-benefit analysis (O'Connor and Dewling, 1986), and estuarine and coastal monitoring and assessment programs which employ them.

REFERENCES

Archer, D.L., and Krenberg, J.E., "Incidence and Cost of Foodborne Diarrheal Disease in the United States," Journal of Food Protection, vol. 48, pp. 887-894, 1985.

Cabelli, V.J., "New Standards for Enteric Bacteria," in Water Pollution Microbiology, ed. R. Mitchell, pp. 233-271, Wiley, New York, 1978.

Cabelli, V.J., Health Effects Criteria for Marine Recreational Waters, EPA 600/1-80-031, US Environmental Protection Agency, Washington, DC, September 1980.

Cabelli, V.J., "Epidemiology of Enteric Viral Infections," in Viruses and Wastewater Treatment, eds. M. Goddard and M. Butler, pp. 291-304, Pergamon Press, Oxford, England, 1981.

Cabelli, V.J., "Predicted Swimming-Associated Gastroenteritis at New York Bight Bathing Beaches," in Marine Pollution Papers, Oceans '82, pp. 1000-1004, Office of Marine Pollution Assessment, NOAA, Rockville, MD, 1982.

Cabelli, V.J., "Waterborne Viral Infections," in Viruses and Disinfection of Water and Wastewater, eds. M. Butler, A.R. Medlen, and R. Morris, pp. 107-130, University of Surrey Print Unit, Surrey, England, 1983.

Cabelli, V.J., "Requirements for Wastewater Disinfection as Seen from the Results of Epidemiological-Microbiological Studies," in Municipal Wastewater Disinfection, eds. A.D. Vanosa, and E.W. Akin, pp. 83-98, EPA 600/9-83-009, US Environmental Protection Agency, Cincinnati, OH, July 1983b.

Cabelli, V.J., Dufour, A.P. Levin, M.A., and Haberman, P.W., "The Impact of Pollution on Marine Bathing Beaches: An Epidemiological Study," in Middle Atlantic Continental Shelf and the New York Bight, Limnology and Oceanography, ed. G. Gross, special symposium, vol. 2, pp. 424-432, 1976.

Cabelli, V.J., Dufour, A.P., McCabe, L.J., and Levin, M.A., "Swimming-Associated Gastroenteritis and Water Quality," American Journal of Epidemiology, vol. 115, pp. 606-616, 1982.

Cabelli, V.J., Dufour, A.P., McCabe, L.J., and Levin, M.A., "A Marine Recreational Water Quality Criterion Consistent with Indicator Concepts and Risk Analysis," Journal of the Water Pollution Control Federation, vol. 55, pp. 1306-1314, 1983.

Cabelli, V.J., Duncanson, R.A., Gaines, J.A., Pedersen, D., Sims, R.R., and Shokes, R.F., "Chemical and Microbiological Tracers for the Movement of Sewage Particulates," in Conference Record, Oceans '84, pp. 272-277, Marine Technology Society, IEEE Ocean Engineering Society, Washington, DC, 1984.

Cabelli, V.J., and Heffernan, W.P., "Seasonal Factors Relevant to Fecal Coliform Levels in the Northern Quahaug," Proceedings National Shellfisheries Association, vol. 61, pp. 95-101, 1970.

Cabelli, V.J., and Pedersen, D., "The Movement of Sewage Sludge from the New York Bight Dumpsite as Seen from Clostridium perfringens Spore Densities," in Marine Pollution Papers, Oceans, 82, pp. 995-1000, Office of Marine Pollution Assessment, NOAA, Washington, DC, 1982.

Centers for Disease Control, "Enteric Illness Associated with Raw Clam Consumption," Morbidity and Mortality Weekly Reports, vol. 31, pp. 450-452, 1982.

Chamberlin, C.E., and Mitchell, R., "A Decay Model for Enteric Bacteria in Natural Waters," in Water Pollution Microbiology, ed. R. Mitchell, vol. 2, pp. 325- , Wiley, New York, 1978.

Dufour, A.P., "E. coli: The Fecal Coliform," in Bacterial Indicators/Health Hazards Associated with Water, eds. A.W. Hoadley, and B.J. Dutka, special technical publication 635, pp. 48-58, American Society for Testing and Materials, Philadelphia, 1977.

Fattal, Badri, Vasl, R.J., Katzenelson, E., and Shuval, H.I., "Survival of Bacterial Indicator Organisms and Enteric Viruses in the Mediterranean Coastal Waters off Tel-Aviv," Water Research, vol. 17, pp. 397-402, 1983.

Frost, W.H., "Report of Committee on Sanitary Control of the Shellfish Industry in the United States," Public Health Reports, vol. 53, pp. 1-53, 1925.

Furfari, S.A., History of the 70 MPN/100 ml Standard, Internal Report, Northeast Marine Health Science Laboratory, US Public Health Service, Davisville, RI, April 1968.

Henderson, M.J., "Enteric Disease Criteria for Recreational Waters," Journal of the Sanitary Engineering Divison, vol. 94, pp. 1253-1276, 1968.

Hornick, R.B., Greisman, S.E., Woodward, T.D., Dupont, H.L., Dawkin, H.T., and Snyder, M.J., "Typhoid Fever: Pathogeneria and Immunologic Control," New England Journal of Medicine, vol. 283, pp. 686-691, 1970.

Hunt, D.A., and Springer, J., "Preliminary Report on a Comparison of Total Coliform and Fecal Coliform Values in Shellfish-Growing Area Waters and a Proposal for a Fecal Coliform Growing Area Standard," in Proceedings 8th National Shellfish Sanitation Workshop, p. 97, Department of Health, Education and Welfare, Washington, DC, 1974.

Katzenelson, E., Kletter, B., and Shuval, H.I., "Inactivation Kinetics of Viruses and Bacteria in Water by Use of Ozone," Journal of American Water Works Association, vol. 66, pp. 725-729, 1974.

Kehr, R.L., Levine, B.S., Butterfield, C.T., and Miller, A.P., A Report on the Public Health Aspects of Clamming in Raritan Bay, Public Health Report, 1941, Reissued by Division of Sanitary Engineering Services, Public Health Service, Department of Health, Education and Welfare, Washington, DC, June 1954.

Keswick, B.H., Satterwhite, T.K., Johnson, P.C., Du Pont, H.L., Secor, S.L., Bitsura, J.A., Gary, N., and Hoff, J. "Inactivation of Norwalk Virus in Drinking Water by Chlorine," Applied and Environmental Microbiology, Vol. 50, pp. 261-264, 1985.

Levin, M.A., Fischer, J.R., and Cabelli, V.J., "Membrane Filter Technique for Enumeration of Enterococci in Marine Waters," Applied Microbiology, vol. 30, pp. 66-71, 1975.

Lupo, L.B., Bacteriophage as Indicators of Fecal Pollution, MS Thesis, Department of Microbiology, University of Rhode Island, Kingston, 1979.

McBride, G., A Bacteriophage Simulant for Enteric Virus Behavior in Water Systems, MS Thesis, Department of Microbiology, University of Rhode Island, Kingston, 1979.

Miescier, J.J., and Cabelli, V.J., "Enterococcus and Other Microbial Indicators in Municipal Sewage Effluents," Journal of the Water Pollution Control Federation, vol. 54, pp. 1599-1606, 1982.

Moore, B., "The Case Against Microbial Standards for Bathing Beaches," in Discharge of Sewage from Sea Outfalls, ed. A.L.H. Gameson, pp. 103-109, Pergamon, London, 1975.

NTAC, Water Quality Criteria, Report on the National Technical Advisory Committee, p. 7, Federal Water Pollution Control Administration, Washington, DC, 1968.

O'Connor, J.S., and Dewling, R.T., "Indices of Marine Degredation, Their Utility," Environmental Management, (in press), 1986.

Plusquellec, A., Beucher, M., and LeGal, Y., "Valeur des Bivalves Comme Indicateurs de la Contamination Bacterienne du Littoral," 2nd International Colloquim of Marine Bacteriology, Brest, France, (in press), October 1984.

Portnoy, B.L., Mackowiak, P.A., and Caraway, C.T., "Oyster Associated Hepatitis. Failure of Shellfish Certification Programs to Prevent Outbreaks," Journal of the American Medical Association, vol. 233, pp. 1065-1072, 1975.

Scarpino, P.V., Berg, G., Chang, S.L., Dahling, D., and Lucas, M., "A Comparative Study of the Inactivation of Viruses in Water by Chlorine," Water Research, vol. 6, pp. 959-965, 1972.

Shuval, H.I., Water Renovation and Reuse, ed. H.I. Shuval, Academic Press, New York, 1977.

US Environmental Protection Agency (USEPA), Quality Criteria for Water, pp. 42-50, USEPA, Washington, DC, July 1976.

US Department of Health, Education and Welfare (USDHEW), National Shellfish Sanitation Program Manual of Operation, Government Printing Office, Washington, DC, 1965.

United States Public Health Service, Manual of Recommended Practice for Sanitary Control of the Shellfish Industry, Public Health Bulletin 295, US Government Printing Office, Washington, DC, 1946.

Verber, J.L., Shellfishborne Disease Outbreaks, Internal Report, Northeast Technical Service Unit, US Food and Drug Administration, Davisville, RI, January 1984.
Watkins, W.D., and Cabelli, V.J., "Effect of Fecal Pollution on Vibrio Parahaemolyticus Densitites in an Estuarine Environment," Applied and Environmental Microbiology, vol. 49, pp. 1307-1313, 1985.

ACKNOWLEDGEMENTS

The data in Tables 1 and 2 will be published at a later date. The author thanks Dr. Robert Armon, James DeBartolomeis, Charles Clavet and Dale Saad, his colleagues and students, for allowing him to use the data prior to their publication.

Gaps in our Knowledge of the Biogeochemical Processes Governing the Fates of Toxic Compounds Discharged to Coastal Seas

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I. INTRODUCTION

The goal of this report is to list the major processes governing the fate of waste chemicals in coastal regions and identify what I think are the major gaps in our knowledge regarding these processes. These areas of ignorance are sufficiently obvious now to recognize but are "difficult" enough so that I don't see solutions appearing soon: hence my attempt to identify research needs which we must be attacking by the year 2000.

Let me briefly set the scene as I see it: (1) it is the year 2000, and (2) mankind has wisely removed large quantities of industrially-derived toxic substances from our waste streams by simply not ever pouring these compounds down sinks and sewers. Boston and other "recalcitrant" municipalities have upgraded their treatment facilities, thereby reducing the hazardous load of our waste streams still further. However, we will likely have extended the points of discharge further and further from shore, and as a result directly affected more and more of the coastal zone. Since populations will be greater, it is likely that larger volumes of material will be released. In spite of our progress with respect to managing hazardous wastes, organic toxicants will still be present: polycyclic aromatic hydrocarbons

(PAH) from combustion and worn auto tires, haloforms from chlorination side reactions, biocides aimed at our never-ending battles with cockroaches, vermin, etc.

Given this scenario, wastewater discharges may continue to be the source of a spectrum of concerns illustrated as a rising plume in Figure 1. On the first level, we can imagine several broad issues. For example, by releasing incompletely decomposed organic matter we will be fertilizing coastal seas. This addition of nutrients is not necessarily bad, but may lead to a degree of eutrophication. Thus non-toxic inputs may have undesirable effects too. Another broad issue involves the addition of quite massive loads of water and solids to small areas, thereby potentially physically modifying these locations. One of the most dramatic examples of this type of problem is the mound of cohesive muds accumulating in the New York Bight as the result of barge releases. These deposits have clearly altered the benthic habitat in this region. Finally, we will still need to assess the toxic effects on the biota arising from the discharge of hazardous substances which we have not been able to eliminate from our waste streams.

The broad issue of toxic effects gives rise to numerous, more focussed concerns, identified in Figure 1, level 2 by keywords acute, chronic, behavioral, synergistic, and ecosystem. Clearly the effects may be important on divergent timescales (acute vs. chronic) and widely different biological scales (individuals vs. combinations vs. ecosystem). This is certainly an important set of issues, and I doubt anyone would debate the need to combat our ignorance here. However, as a prerequisite to estimating the hazards posed by chemical substances, we must be able to quantify the exposures of organisms to these toxicants.

This exposure prediction task leads us to the need for modelling the fate of chemicals introduced into coastal seas. Fate modelling attempts to estimate the concentrations of chemicals at various places and at different times after discharge. Tremendous progress in the area has been made in the last decade or so, and these advances have come chiefly as a direct result of distinguishing the individual processes which cause a chemical to move about or disappear altogether from an aquatic environment: volatilization, sorption, bioaccumulation, thermochemical transformations, photochemical reactions, and biologically mediated degradations (Figure 1, level 3). Each of these processes can be viewed as operating in tandem with the rest; by describing them individually as a function of chemical and environmental properties. We can eventually combine them to calculate their effects on the fate of pollutants. While this systematic approach has enabled us to direct our scientific efforts to understand each of these individual processes, three gaps in our knowledge may prevent us from accurately applying this modelling framework to estimate toxic effects of coastal discharges.

1. We need to know what ingredients or agents of the environment govern the intensity of several of the unit processes; these must be quantified and the factors causing their

abundance to vary must be understood.

2. We need to include the impact of temporal inhomogeneity - or episodic events - even CATASTROPHIC events (e.g., hurricanes) in our thinking and modelling; and

3. We need to consider unique aspects of coastal regions, and not to expect we can fully rely on our knowledge of processes from lakes, rivers, and open ocean sites.

For the remainder of this discussion, I hope to exemplify these needs - yet even as I point to our ignorance, I do not want to diminish the amazing breadth of knowledge regarding biogeochemical processes which we already have. Let me re-emphasize, a systematic approach to fate modelling is rapidly evolving in which scientifically based descriptions of individual processes affecting chemicals in aquatic environments are used. This modelling approach is particularly flexible in that it can be readily applied to new chemicals and new environments as the need arises. Generally we know the identities of the important individual processes, we know much about how the properties of the chemicals can be used, and we can commonly solve the "steady state" or "regularly varying" governing model expressions that result.

II. TYPICAL GAPS IN OUR KNOWLEDGE OF FATE MODELLING IN COASTAL SEAS

#1: Active Agents

Insofar as organic compounds are involved in reactions in coastal seawater, we often have very poor knowledge of the actual reagents present in the environment which instigate these transformations. Presently three unit processes are particularly difficult in this regard: redox reactions, indirect photochemical transformations, and biologically mediated degradations.

We know that many organic toxicants, which are stable under oxic conditions, are rapidly transformed under reducing situations. "Who done it?" is a very critical issue if we're to predict where, when, and how much these transformations will occur in new circumstances. A parallel problem exists in the area of indirect photochemical transformations. A great deal of light energy is soaked up by the sea, resulting in the formation of numerous highly reactive species (e.g., singlet oxygen, hydrogen peroxide); currently the abundances and variations of these environmental reagents are not well known. Finally, it is clear that microorganisms transform many organic materials, but which organisms or combinations of organisms remains unclear. Furthermore, accurate approaches to quantify this activity are currently unavailable. Thus, prediction is difficult, at best.

Allow me to briefly develop these observations. It has long been observed that toxic chlorinated compounds such as lindane, DDT, aldrin, or toxaphene are degraded in heterogenous reducing conditions (e.g., Figure 2; Hill and McCarty, 1967; Parr and Smith, 1976). Generally the conclusion has been that anaerobic microorganisms are involved, yet there are doubts since poisoned controls (e.g., lindane and aldrin in Figure 2a) also contain some activity. Several recent observations suggest these transformations may be through abiotic chemical reactions occurring as a consequence of the biologically lowered media E_h (e.g., note correlations in toxaphene losses in Figure 2b). Tratnyek and Macalady (1984) have shown that many compounds are rapidly transformed by supplying electrode reducing power; Roberts (personal communication) has accomplished much the same thing with reduced metals in our laboratory. Which environmental chemical agents are sufficiently reactive to interact with organic pollutants and how abundant these reduced species are in anoxic coastal muds remains unknown. I must know this to predict new reactions of related organic compounds and their rates.

When we turn to reactions which occur due to the instigation of the energy of light, we have a similar difficulty identifying and quantifying the environmental reagents involved. Gelbstoff (or yellow organic stuff) has long been recognized as an ingredient in natural waters. Clearly some of this macromolecular organic matter (labelled "?" in Figure 3) absorbs light energy and to some extent passes this energy to other substances: including singlet oxygen ($^1O_2$), peroxide (H_2O_2), and hydroxyl radical (OH) (Zafiriou et al., 1984). All of these, and likely many others, are very good oxidants - hence, they typically react with organic solutes. However, the important photochemically-derived reactants in seawater are poorly characterized (and hence their breadth of reactivities). The concentrations of the species which have been studied can only be estimated in terms of orders of magnitude; this is not surprising when you realize that OH radical is present at roughly 10 parts in a billion billions!

Finally, quantitative treatments of biodegrading capabilities are presently poorly known. Measures of organism abundance and activity serve inadequately as predictors. Bartholomew and Pfaender (1983) assessed chlorobenzene degradation in water samples from an estuary and coastal region. If biological removal occurs and can be modelled using Michaelis-Menten kinetics, the maximum rate (V_{max}) of chlorobenzene removal should correspond to the numbers of organisms present. Plotting these observed V_{max} results vs. microbial abundance determined by acridine orange direct counts (AODC) and colony forming unit (CFU) approaches (Figure 4a), they could not see any strong relation. Although the data is not particularly abundant, more recent observations don't improve the correlations. One may argue that AODC or CFU approaches do not quantify the abilities of the microbes living in these waters, but activity measures of these microorganisms have also proven not too informative, as shown in data from Wakeham et al. (1984) on toluene degradation (Figure 4b). We need some way to enumerate the biological "reagents" which control the removal of toxic compounds such as chlorobenzene or toluene.

To summarize here - "Who done it?" and "What factors control these agents?" in the processes of redox, indirect photochemical, and biochemical reactions, must be determined to allow predictive fate modelling.

#2: Episodic Events

Commonly, unit processes affecting organic pollutants can be understood in a predictive manner when the environmental reagent is known and its abundance remains essentially constant or regularly varying (e.g., diurnal light availability). Two examples of this are found in the transport unit processes: volatilization and sorption. When we set up a laboratory experiment with constant wind blowing across a beaker or flume of water containing volatile organic compounds, we can already do a pretty good job of estimating the air-water exchange rate which will remove the chemicals from the water. Similarly, if I set up a laboratory experiment in which I mix a known quantity of sediment with seawater containing a polycyclic aromatic hydrocarbon (PAH), I can do a good job of predicting how much PAH will sorb. In both of the laboratory instances the key ingredient which is often missing in the real world is the constancy of conditions, whether it be a steady wind or a constant solids-water mixing ratio.

Unfortunately, from the point of view of modelling, the world is not "steady" or even "regular". The first example of the "episodic" nature of the real world involves the wind - a critical environmental condition controlling volatilization in many instances and one which any of us knows varies in time and space. How does this variation effect volatilization?

Peng and his co-workers in the GEOSECS program (1979) have some real-world data to address this point. By measuring Radon-222 in the sea, they could deduce the rate volatilization of this inert gas or the thickness of the boundary later inhibiting exchange at various sites in the ocean. From our theories and constant-conditions laboratory experiments, we know that at steady state the rate of removal should be directly related, and the boundary layer thickness must be inversely related, to the wind speed. Comparing the predicted (from wind speed observations) and observed (from Radon-222 depletion) data, we see the piston velocity (a measure of volatilization rate) scatters widely as a function of wind (Figure 5a) and the boundary layer thickness controlling air-water exchange doesn't change nearly as extensively as would be predicted from the wind data (Figure 5b).

The problem doesn't really lie in the theory, but rather in the assumption of constant wind at any site. By monitoring wind at a single oceanic station for almost two months, one can see it varies greatly on a timescale of days (Figure 5b). The volatilization timescale removing chemicals from the mixed surface layer of the sea is roughly a month. Consequently, using wind data we find that the predicted volatilization effect is much more scattered than the observed effect. This relatively episodic nature of wind and the ability of the water column to react obviously must be considered before accurate air-water exchange predictions can be made.

I think an even more dramatic example of non-steady events governing pollutant fate involves the combination: cohesive sediments and violent storms. Due to their affinity for natural organic matter, many undesirable organic pollutants such as PAH and PCB's end up...more or less stored in muddy silts. If we take some of that sediment into the lab and suspend it in water, we can generally predict the release of pollutants to the water. However, in the real world cohesive sediments are not consistently exposed to the water column. The question is when, if ever, will they be resuspended?

Some recent data of Sternberg and Larsen (1975) exemplify my concern on predicting resuspension. By monitoring factors such as current velocity and wave height at a shelf station over a course of about 2 weeks, they hoped to correlate these physical forcing factors, which common sense indicates must cause movements of bottom sediments, with resuspension events. Their data (Figure 6) shows how difficult it will be to quantitatively predict resuspension at that site! Periods with the same wave heights do not result in the same water column turbidities, and the variations in current velocities bear no relation at all to resuspension. The bottom line is no one knows quantitatively what it takes to resuspend natural cohesive sediments...yet we do know that when they are lifted from the bottom and held in suspension a few hours to days, tremendous releases of hydrophobic pollutants should occur. Thus the incorporation of rare or catastrophic events, as well as short-term episodic changes, should enter our thinking and predictions of the fate of organic pollutants.

#3: Special Characteristics of Coastal Regions

Many people, including myself, focus closely on the individual unit processes with the belief that these processes work more or less the same in all aquatic environments be they groundwater, lakes, rivers, estuaries, or oceans. We assume that to the extent processes such as volatilization differ in various aquatic environments, we can tune the process descriptions for the environment of interest. However, this is not always correct - indeed sometimes things may work very differently in coastal regions.

The most obvious thing that works differently at coastal sites is public opinion . While relatively little attention and interest is focussed on the processes in the open ocean, coastal issues are frequently in the headlines (e.g., Boston Globe headlines regarding pollution of Boston Harbor and siting of future sewage discharges).

On a more scientific theme, there are some obvious and possibly not so obvious reasons to think the coastal ecosystem response to pollutants may be markedly different than other aquatic environments. Regarding the process of volatilization, it is unclear whether forcing by wind (assumed controlling in the ocean), turbulence due to current/shallow depth interactions (assumed controlling the streams and rivers), or breaking waves and the resultant bubble injections (generally neglected) is the most important factor driving gas exchange in coastal regions. Insofar as transport of pollutants in association with particles is concerned, the high ionic strength of seawater leads to contact interactions between particles and their consequent coagulation and sedimentation, unlike the situation in freshwater environments where particle surface charges prevent collisions and therefore flocculation. Additionally the shallow nature of coastal regions allows ready contact between the entire water column and the sediments, is obviously distinguishing from open ocean sites, and therefore particles must play a more important role nearshore.

Some reaction process differences may result from the unique organisms such as macroalgae living in coastal regions. For example, we know that harvesting of light by chromophores is the primary process driving photochemical transformations. Although it is true we can not uniquely define all the chemicals responsible for light absorption in any natural waters, we do know there are major compositional differences between freshwater and open ocean seawater (Figure 7), and this should result in different light absorption characteristics. Presumably the coastal zone contains a mix of these materials AND light absorbing substances released by macroalgae, such as polyphenols. Carlson and Carlson (1984) recently provided data suggesting that these phenols contribute a great deal to the dissolved organic load of seawater in the Gulf of Maine. Since these compounds are excellent light absorbers and are very reactive, their presence could have important ramifications to coastal zone biochemistry.

In another example, we have recently demonstrated (Gschwend et al. 1985) that temperate macroalgae release a variety of halogenated organic chemicals to the coastal seawater here in New England (Figure 8). One hypothesis that arises from this finding is that marine microorganisms, long exposed to these halogenated metabolites, may have evolved enzymatic apparatus to degrade these and similar compounds! Thus the biochemical capabilities of the coastal marine biota may be significantly different than those found in freshwater and open ocean waters.

The bottom line is: due to special characteristics and combinations of characteristics, coastal seawater needs to be thought about in terms of its unique nature.

tY

:n conclusion, let me reiterate the three AREAS requiring :ular attention before we can accurately describe ›chemical processes and their effect on chemicals discharged ıstal seawater:

.. Who done it? How do we quantitatively describe the :als or organisms affecting the processes with which we are :ned?

:. How can we incorporate episodic processes which control ›nmental rates on the important timescales?

ǀ. Have we considered all the special qualities of the ıl seas especially those resulting from the unique ıations of chemicals produced and consumed there?

:hese issues will require new approaches to quantify some of ıgents" determining the fate of organic compounds. We will :o develop non-steady state models and improve our knowledge ı physical processes in natural waters respond. We will :o use common sense and a continuing sense of exploration re think about coastal seas.

/e have made a good beginning, but more work must be done.

REFERENCES

Bartholomew, G.W. and F.K. Pfaender. 1983. Influence of spatial and temporal variations on organic pollutant biodegradation rates in an estuarine environment. Applied Environ. Microbiol. 45, 103-109.

Burreson, B.J., R.E. Moore, and P.P. Roller. 1976. Volatile halogen compounds in the alga Asparagopsis taxiformis (Rhodophyta). J. Agric. Food Chem. 24, 856-861.

Carlson, D.J. and M.L. Carlson. 1984. Reassessment of exudation by fucoid macroalgae. Limnol. Oceanogr. 29, 1077-1087.

Gschwend, P.M., J.K. MacFarlane, and K.A. Newman. 1985. Volatile halogenated organic compounds released to seawater from temperate marine macroalgae. Science 227, 1033-1035.

Higa, T. 1981. Ch. 3 Phenolic substances. In: P.J. Schener, (Ed.), Marine Natural Products. Academic Press, New York, pp. 93-145.

Hill, D.W. and P.L. McCarty. 1967. Anaerobic degradation of selected chlorinated hydrocarbon pesticides. J. Water Pollution Control F. 39, 1259-1277.

Hurst, H.M. and N. Burges. 1967. Ch. 11 Lignin and humic acids. In: A.D. McLaren and G.H. Peterson (Eds.), Soil Biogeochemistry. Marcel Dekker, New York.

Parr, J.F. and S. Smith. 1976. Soil Science 121, 52.

Peng, T.-H., W.S. Broecker, G.G. Mathieu, and Y.-H. Li. 1979. Radon evasion rates in the Atlantic and Pacific Oceans as determined during the GEOSECS Program. J. Geophys. Res. 84, 2471-2486.

Sternberg, R.W. and L.H. Larsen. 1975. Threshold of sediment movement by open ocean waves: observations. Deep-Sea Research 22, 299-309.

Stuermer, D.H. 1975. The Characterization of Humic Substances from Seawater. Ph.D. Thesis, MIT/WHOI Joint Program, 188 pp.

Tratnyek, P.G. and D.L. Macalady. 1984. Poising redox potential for laboratory studies of abiotic degradation of anthropogenic organic chemicals. Symposium on Mechanisms of Transformations of Organics in Soils, Sediments, and Groundwater. 1984 International Chemical Congress of Pacific Basin Societies. Honolulu, Hawaii, December 16-21.

;., E.A. Canuel, P.H. Doering, J.E. Hobbie, J.V.K. :h, and G. Lough. 1985. The biogeochemistry of : in coastal seawater: Radiotracer experiments in lled ecosystems. Biogeochemistry (in press).

,C., J. Joussot-Dubien, R.G. Zepp, and R.G. Zika. Photochemistry of natural waters. Environ. Sci. l. 18, 358A-371A.

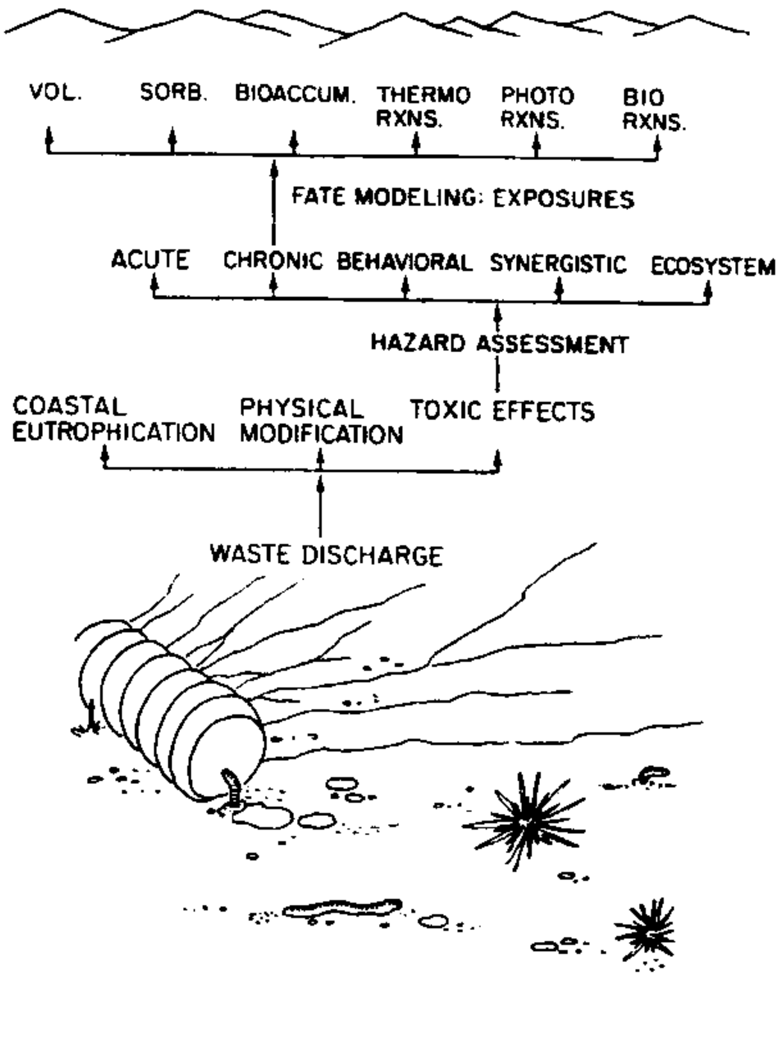


Figure 1 A rising "plume" of concerns derived from waste discharged to coastal seawater; upper levels are limited to problems involving assessing effects of toxic compounds.

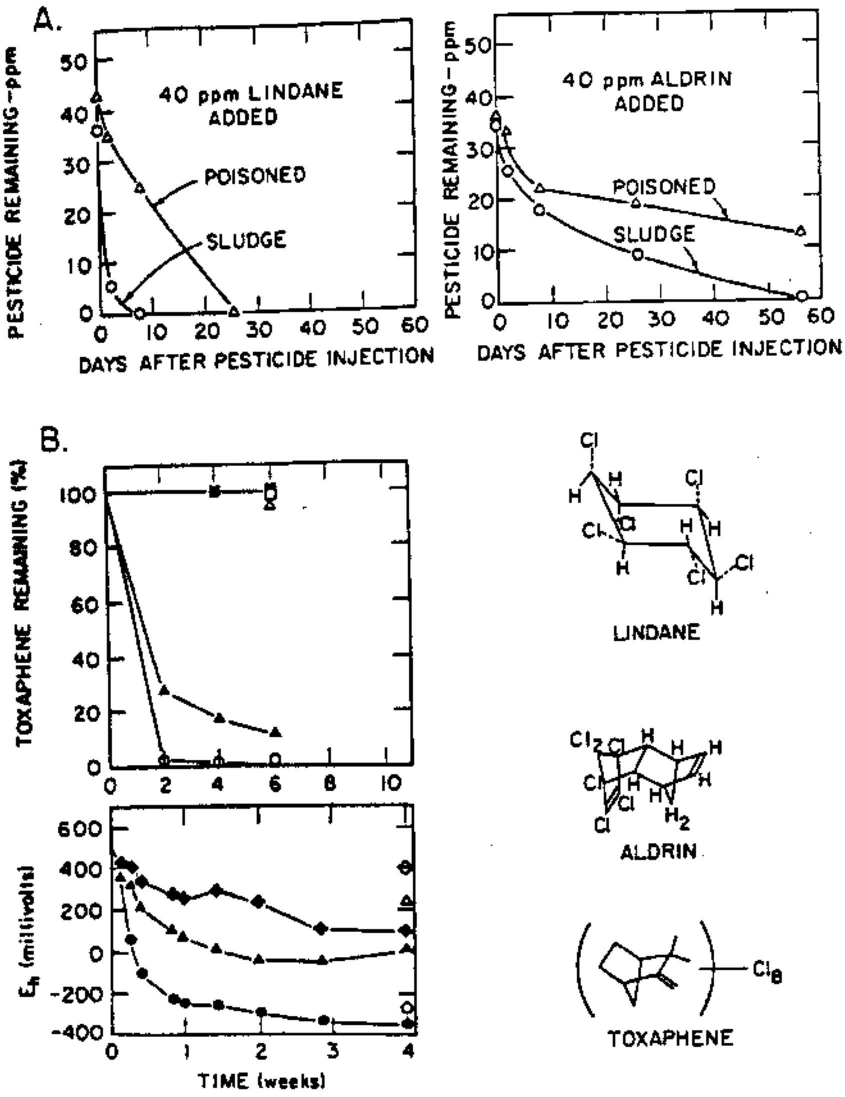


Figure 2 Removal of some chlorinated organic compounds (structures shown) in reducing conditions. A: Timecourse of lindane and aldrin removal in anaerobic sewage including a poisoned control (from Hill and McCarty, 1967). B: Timecourse of toxaphene loss in reducing soils simultaneously monitored for E_h (from Parr and Smith, 1976).

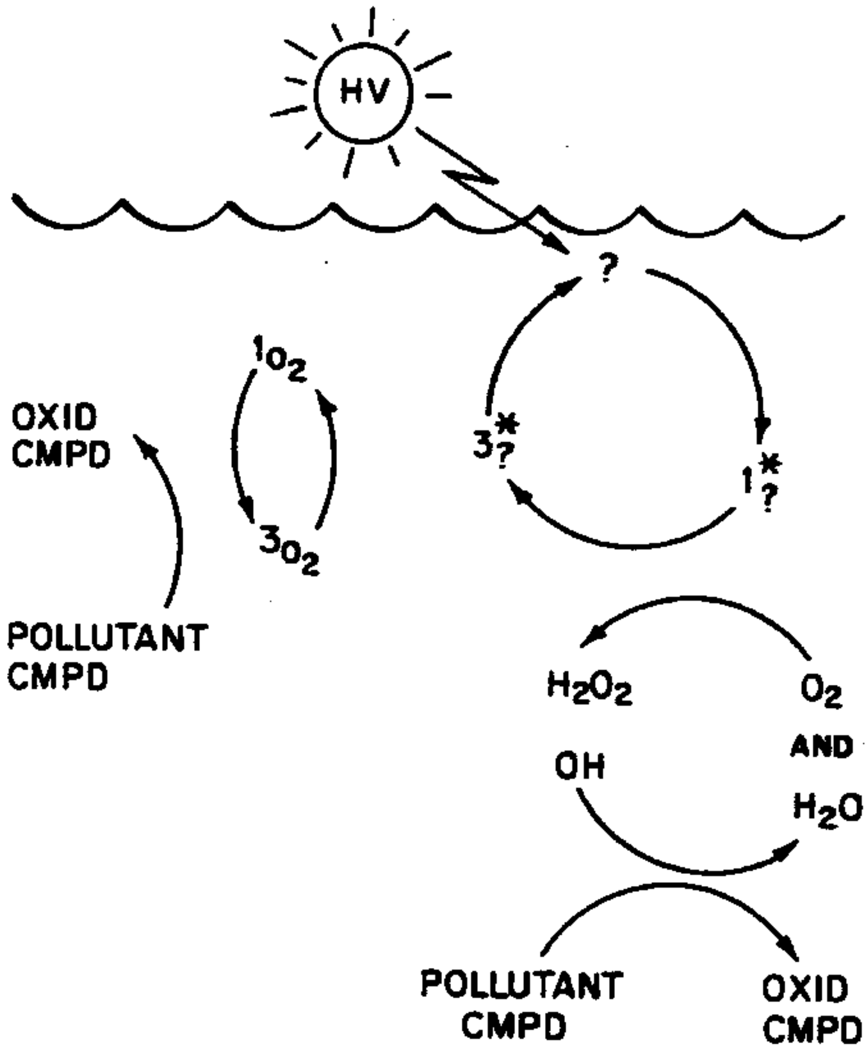
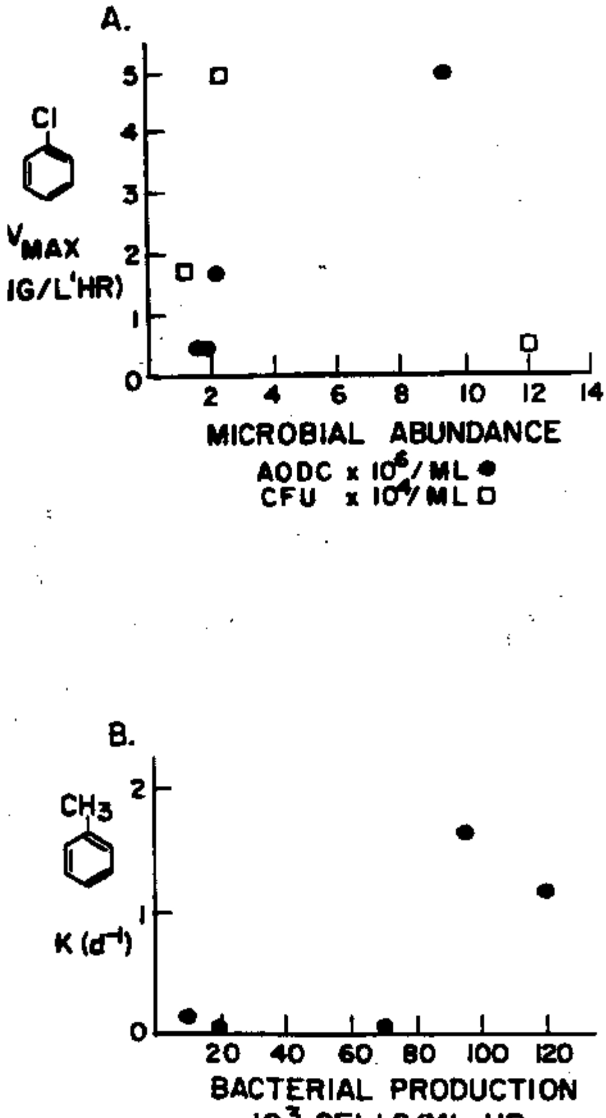


Figure 3 Simplified scheme illustrating the use of light energy to activate environmental reagents such as molecular oxygen to reactive species such as singlet oxygen ($^1O_2$), hydrogen peroxide (H_2O_2), and hydroxyl radical ($\cdot OH$), which subsequently may react with organic pollutants (modified from Zafiriou et al., 1984).



re 4 Attempts to quantitatively estimate microbial degradation rates of organic toxicants. A: Maximum rate of removal (V_{max}) of chlorobenzene vs. measures of microbial abundance (Bartholomew and Pfaender, 1983). B: First order degradation constant of toluene vs. microbial growth rate (Wakeham et al., 1984).

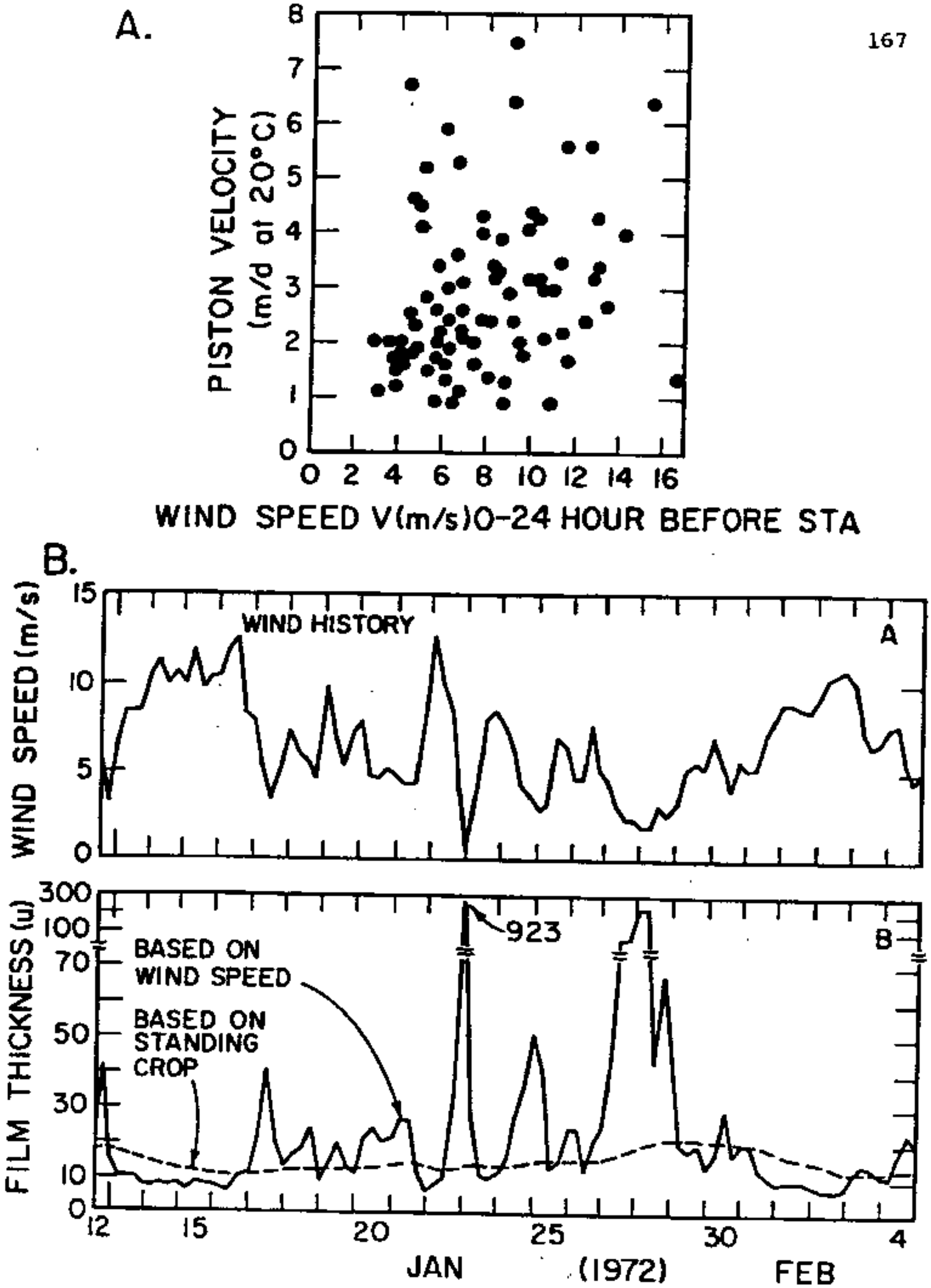


Figure 5 Field observation on the effect of wind on volatilization of Radon-222 from the sea (Peng et al., 1979). A: Composite plot of wind velocities measured for 24 hours at sampling sites vs. piston velocities calculated from observed Radon-222 deficiencies in the water column. B: The wind speed variations at a single station monitored for 3 weeks, the stagnant film thickness predicted from this wind data, and the film thickness calculated from the abundance of Radon-222 at that site.

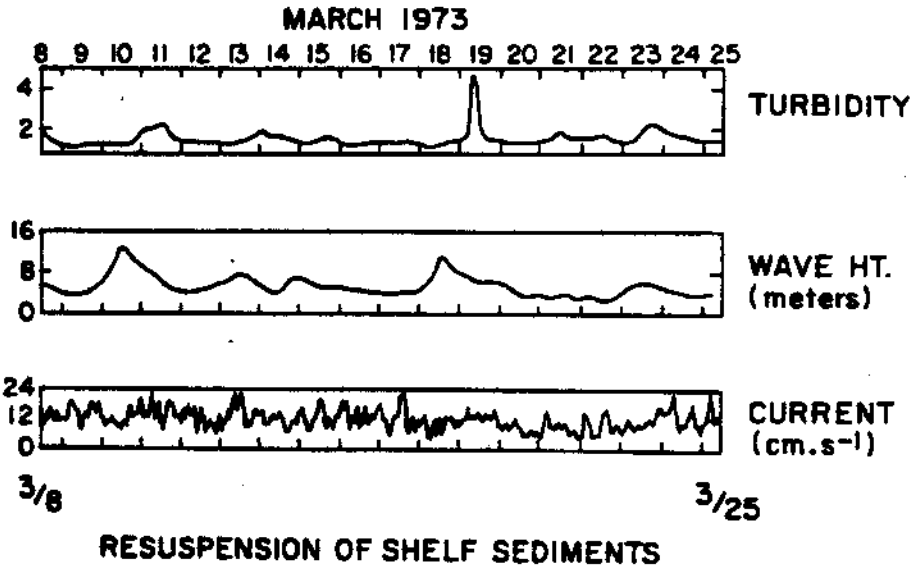
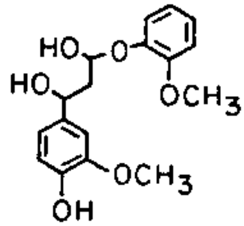


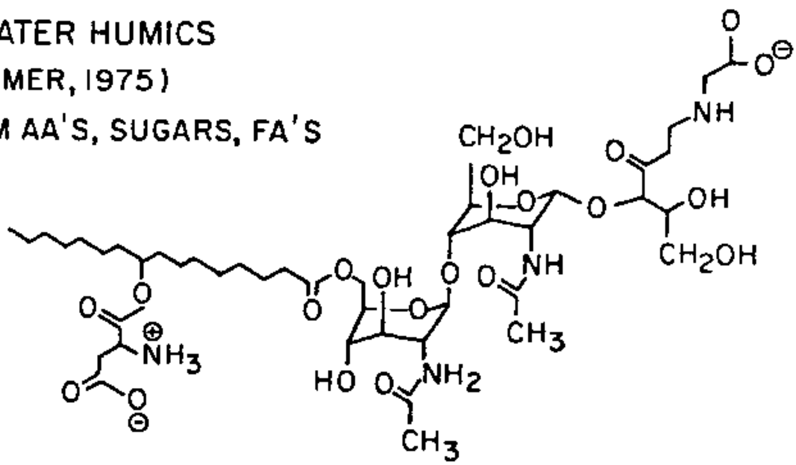
Figure 6 Timecourse observations of Sternberg and Larsen (1975) on the continental shelf off northwest United States seeking relations between wave heights or current velocities with the resuspension of the cohesive bottom sands as reflected by water column turbidity.

CHROMOPHORES LEADING TO INDIRECT PHOTOCHEMICAL OXIDATIONS

LAKE WATER HUMICS
(HURST AND BURGESS, 1967)
FROM LIGNIN



SEA WATER HUMICS
(STUERMER, 1975)
FROM AA'S, SUGARS, FA'S



POLYPHENOLS FROM
MARINE MACROALGAE
(IN HIGA, 1981)

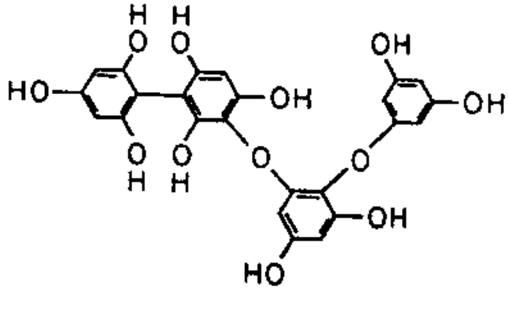


Figure 7 Structures of organic substances which may contribute to light absorption in various natural waters (Hurst and Burges, 1967; Stuermer, 1975; Higa, 1981).

FROM UTILITIES / INDUSTRIES		FROM ALGAE	REF.
CHLOROFORM	$CHCL_3$		
	$CHBRCL_2$		
	$CHBR_2CL$	$CHBR_2CL$	GSCHWEND ET AL 1985
BROMOFORM	$CHBR_3$	$CHBR_3$	
TRICHLOROETHYLENE			
	$CL_2C=CHCL$	$BR_2C=CHCHCL_2$	BURRESON ET AL 1974
CARBON TETRACHLORIDE			
	CCL_4	CBR_4	

Figure 8 A comparison of volatile halogenated compounds emitted by anthropogenic activities with some natural products found in marine macroalgae (Burreson et al., 1976; Gschwend et al., 1985).

Innovative Approaches to the Detection and Measurement of Marine Pollutant Impacts

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1. INTRODUCTION

Recent estimates place the direct input of municipal wastes to U.S. coastal waters at approximately 3.6×10^{12} liters yr^{-1} (Myers, 1983). If current trends continue, increasing amounts of these and other effluents can be expected to enter the nearshore marine environment in the future. Accordingly, sensitive and accurate methods will be needed to evaluate the efficacy of waste disposal vis-à-vis the protection of important marine resources.

Because many of the most toxic constituents of municipal effluents are strongly associated with particulate matter, the problem is largely one of dispersal. Methods for the detection and measurement of waste impacts should, therefore, be capable of providing information not only on the initial dilution at a discharge site, but also the spatial dispersion of waste materials in the far field. Under ideal circumstances, wastes would be rapidly diluted and advected away by currents such that no accumulation in the underlying sediments could occur. This behavior is rarely, if ever, observed for large coastal discharges. Instead, the sediments act as a sink for waste particulates as evidenced by the buildup of contaminants near many outfalls and dumpsites (cf. Figure 1).

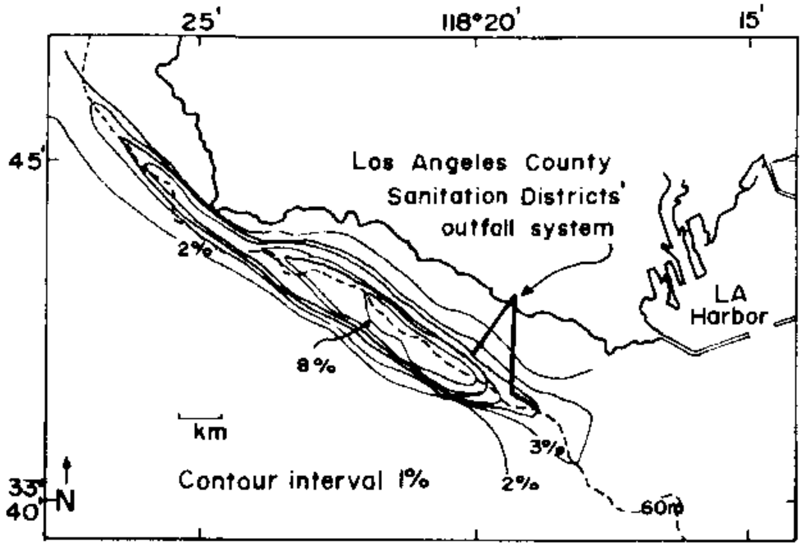

Figure 1. Distribution of organic carbon in surface sediments of the San Pedro Shelf, California (after Sweeney et al., 1978).

Aside from their obvious value as a monitoring medium, the sediments are of interest for other reasons. In particular, the presence of large amounts of organic matter and various toxicants has an important consequence. It imposes rather severe limitations on our ability to predict and verify the fate of wastes at sea.

Because municipal wastes result from man's activities, their disposal on a large scale usually takes place in the rivers, estuaries and coastal waters adjoining major urban centers. This introduces a second problem - that of differentiating environmental contamination due to sewage from contributions made by other sources.

For example, toxic substances such as the PCBs (polychlorinated biphenyls), petroleum hydrocarbons and chlorinated pesticides all enter the ocean via several routes other than by discharge of municipal wastes (e.g. urban runoff, rivers, discrete industrial releases and atmospheric transport). Compositional differences between inputs may or may not exist before mixing occurs. However, once scrambling of the various inputs on a molecular level has occurred, the resultant mixture presents a formidable challenge to the analyst, particularly if his goal is to determine the contribution made by one input to the sediments. The problem is further aggravated by compositional alterations that attend environmental exposure as a result of numerous physical and chemical processes. Important indicator compounds may be removed and source-related homolog and isomer patterns destroyed. In the face of these difficulties, the organic geochemist has but two options: 1) to carry out a detailed input assessment or 2) use waste-specific tracers.

In the former approach, concentrations of targeted pollutant species must be determined in all significant inputs over time periods spanning the discharge history of each compound. As illustrated by Figure 2, in cases where multiple inputs exist, input assessment is complicated by the fact that numerous spatial and temporal parameters intrinsic to each input must be considered. Thus, extensive sampling on various time scales would be necessary. Otherwise, numerous assumptions of dubious validity would have to be invoked. Generally speaking, information of this type and quantity is simply not available, and the cost of obtaining it on a continuous basis would doubtless be prohibitive. Furthermore, establishing a connection between the mass emission rates of a specific pollutant from various inputs and its abundance in sediments would be difficult. In short, input assessment is an indirect approach that is both expensive and fraught with uncertainties.

The alternative, that of using waste-specific tracers, is direct and simple in concept. A substance or chemical characteristic of a given effluent is used as a passive chemical tag. Measurement of this substance (or characteristic) in the environment permits one to infer, and hopefully quantify, the presence of waste material in the sample. In order for the full potential of this approach to be realized, however, a tracer must satisfy several rather stringent criteria.

may have adverse consequences for the benthic organisms that live in direct contact with them. Pelagic species can also be impacted by absorption of dissolved substances released from sediments, by direct ingestion of suspended sewage particles and/or by transfer of waste-derived toxicants through one or more trophic levels. In any case, the monitoring methods of the future will have to be capable of detecting and quantifying the accumulation of wastes and associated pollutants in both sediments and biota.

This paper assesses the advantages and limitations of several promising approaches to the problems described above. These methodologies are in the earliest stages of development and have yet to be tested carefully in the field. Although the technology for evaluating their utility has existed for some time, progress toward implementation has been slow. As will be shown, this is largely due to the fact that the organic chemistry of wastes and the fate of ocean-discharged sewage have received scant attention. The aim of this paper is to stimulate discussion on these and other innovative monitoring methodologies. While the focus here is clearly on organic constituents of municipal wastes, the general principles should apply equally to inorganic species and industrial wastes.

2. DETECTING THE PRESENCE OF WASTES AT SEA

Overview of the Problem

As mentioned above, several problems have impeded the development of methods for detecting the presence of wastes in the environment. Not the least of these is our limited understanding of the composition of sewage. Municipal effluents comprise an exceedingly complex assortment of substances derived from a multitude of human activities and products. In addition, certain compounds such as volatile halogenated organics, can be generated *in situ* during the treatment process itself.

Although it has been variously estimated that ca. 10-15% of the organic constituents have been identified, this estimate should probably be regarded as optimistic. Most of the identified substances are species readily amenable to separation by high resolution gas chromatography and structural elucidation by mass spectrometry and various ancillary spectroscopic techniques. (Many organic pollutants present in wastes at trace levels, for example, fall into this category). However, the vast majority of the organics in sewage are either high molecular weight substances having complex structures (e.g. humic substances) and/or they are non-volatile (Rebhun and Manka, 1971). In either case, they are difficult to separate and analyze. Thus, the complexity and analytical intractability of wastes may largely account for the lack of data on their organic chemistry. The fact that virtually nothing is known about effluent-to-effluent and temporal variations in the composition of sewage

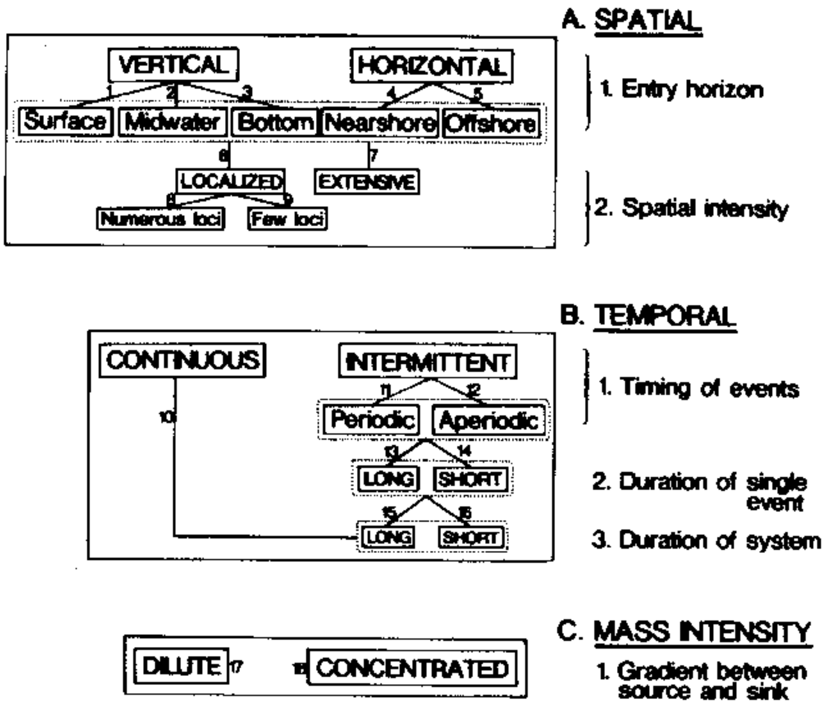

Figure 2. A schematic representation of the main elements involved in the delivery of organic matter to the ocean.

Requirements of a Waste-Specific Tracer

It is useful to distinguish between the terms "marker" and "tracer" as these are commonly used in the literature. Both suggest a direct link between source material (e.g. sewage) and sink (e.g. sediment); however, when observed in a sample, the former can be used only to infer the presence of the source material. A "marker" is, thus, a qualitative tool. A "tracer", as defined here, should be applicable in a quantitative sense. The measured abundance of a tracer in a sample can, in principle, be used to directly estimate the amount of the source material present.

Regardless of the source material or the parameter used, all tracers must satisfy three fundamental requirements. First and foremost, they must be source-specific. This means the tracer must be present in (or characteristic of) only one source material or it must be vastly more abundant in one source material than any others likely to be of quantitative importance. Because quantitation is desired, the tracer should not be produced in situ following introduction of the source material to the environment. Presumably this could occur by specific transformations of one or more precursors to the tracer itself.

Second, the tracer should be detectable and, more importantly, quantifiable in environmental samples. At a specified level of dilution, the detectability or quantifiability of a tracer will depend on three factors: 1) its concentration in the source material, 2) the extent of removal by various natural processes, and 3) the detection and quantitation limits of the methods used for determination of the tracer.

Finally, the tracer's environmental behavior must be known. As all organic substances are to some extent degradable by heterotrophic organisms, their transformation and removal is essentially a question of kinetics. Microbial community structure and activity are controlled by environmental factors such as temperature, pH, oxygen concentration as well as the availability of suitable substrates and nutrients. Consequently, these variables will exert an influence on the rate at which a specific tracer is removed or altered. In addition, photochemical, chemical and physical processes can play significant roles in the geochemical fate of a tracer through oxidation, polymerization, condensation and other reactions as well as phase transfer processes (e.g. evaporation, sorption, precipitation). It is also important that the behavior of a tracer be known within the context of the specific physical setting where discharge occurs. For example, water depth, current velocities, temperature and salinity will all affect the sedimentation rates of particles and, hence, the time available to the various processes responsible for tracer removal and alteration in the water column.

One final point should be emphasized. The stability and general behavior of the tracer must be known in relation to that of the waste co-constituents of interest. This suggests two classes of tracers: generic and specific. A generic organic tracer should be characteristic of the organic matter as a whole. The behavior of a specific tracer, on the other hand, will be dictated by its individual physico-chemical properties and susceptibility to degradation. To the extent that these are unique (i.e. unlike those of many of the substances likely to be present in the source material), the behavior of a specific tracer is more likely to be representative of a restricted class of compounds than the bulk organic matter.

Elemental and Isotopic Approaches

As illustrated in Figure 1, elemental abundance (i.e. organic carbon and nitrogen) can be a useful generic tracer of waste contamination. This is possible because the particulates in primary effluent and sludge contain approximately 10-30 times the amount of carbon and nitrogen found in uncontaminated coastal sediments (Eganhouse and Kaplan, 1984). The waste-specificity of organic carbon and nitrogen is obviously limited by the fact that any other sources of organic matter will contain these elements. Consequently, their utility as tracers is restricted to sediments receiving a large input of waste organic matter relative to other sources. Because of this requirement, however, detectability is not a problem. Even pristine continental shelf sediments can be easily analyzed for their carbon and nitrogen content.

Studies of the decomposition of sludge in seawater and seawater/sediment systems have been reported (Myers, 1974; Grunseich and Duedall, 1978). The kinetics of degradation observed at various temperatures both with and without oxygenation suggest that over periods of up to several months as much as 25-30% of the particulate organic carbon can be removed. Unfortunately, the actual alterations taking place on a molecular level and the nature of the resultant refractory organic matter have not been elaborated. It is equally unclear how the rates of mineralization of other sedimentary source materials such as planktonic debris or runoff compare with those of sludge or primary effluent. Because of these uncertainties, the abundances of organic carbon or nitrogen alone are probably of limited value as tracers. When used in conjunction with stable isotopic ratios, however, they can provide an independently verifiable means of estimating the contributions of waste organic matter to sediments.

Stable isotope ratios of the elements carbon ($^{13}C/^{12}C$) and nitrogen ($^{15}N/^{14}N$) have frequently been used to determine terrestrial and marine contributions of organic matter to estuarine and nearshore coastal sediments (e.g. Schultz and Calder, 1976; Hedges and Parker, 1976; Tan and Strain, 1979, *inter alia*). This application is based on a difference in isotopic composition of the two endmembers arising from differences in the stable isotope ratios of the inorganic precursors from which organic carbon and nitrogen are formed (Peters *et al*., 1978). Mixtures of marine and terrestrial organic matter, thus, have isotope ratios intermediate between the endmember values. Extension of this concept to tracking wastes at sea follows directly when it is realized that waste organic matter is largely terrestrial in origin.

With respect to its waste-specificity, stable isotopic abundance is limited by two facts: 1) other terrestrial inputs having isotopic ratios similar to that of sewage may exist (e.g. aeolian and river or runoff transported terrigenous debris) and 2) the range in endmember values can be large relative to the difference between the mean values of the endmembers. This relationship is important as it essentially defines the degree of resolution attainable. Since the individual molecular constituents of organic matter have measurable differences in isotopic composition (Degens, 1969), variations in endmember molecular composition will affect the range of endmember isotope ratios. Another source of isotopic variation for the marine endmember may be the influence of specific environmental factors which vary seasonally such as phytoplankton species composition, temperature and the availability of CO_2 and NO_3^- (Gearing et al, 1984). Ultimately, it is important to recognize that endmember values are likely to be both site specific and somewhat time-variant. As in the case of elemental abundance of carbon and nitrogen, isotopic composition is determined by and, thus, representative of the total organic matter. Consequently, both of these parameters can be considered generic tracers.

The practical quantitation limits equate to the amounts of organic carbon or nitrogen needed for reliable measurement of the isotope ratios. Acceptable analytical precision can be achieved with as little as 100 nanomoles of CO_2 or N_2. For a sediment sample of 10 mg this corresponds to organic carbon and nitrogen contents of 0.01% and 0.03%, assuming complete recovery. Because their use for tracking waste organic matter is restricted to nearfield sediments, isotope ratios can be easily determined using conventional combustion methods (Minagawa and Kaplan, 1984; Eganhouse and Kaplan, 1984).

The alteration of isotope ratios (i.e. fractionation) during and after sedimentation must also be considered. Two principal mechanisms for fractionation of organic carbon and nitrogen are recognized as possibilities. One is due to a normal kinetic isotope effect. This results in the preferential removal of atoms of the lighter isotope due to the slightly faster rates at which bonds to them are broken. The result is enrichment of the heavy isotope in the residual organic matter. The second mechanism involves selective removal of organic constituents whose isotopic compositions differ significantly from that of the bulk organic matter. For a noticeable fractionation to occur by this mechanism, either a large intermolecular isotope difference must exist and/or a large fraction of the organic matter must be removed during this process.

Little is known about the effect of decomposition processes on the isotopic composition of sludge. Myers (1974) observed a small but significant fractionation (i.e. ^{13}C-depletion) during sludge/seawater experiments suggesting that changes in endmember values could occur during the sedimentation process. No analogous experiments for nitrogen isotopes have yet been reported. Other investigators, noting systematic variations in the isotope ratios of suspended particulate organic carbon and nitrogen with depth in the ocean, have attributed these trends to biologically-mediated alterations occurring during settling (Libes, 1983; Jeffrey et al, 1983). The situation during early diagenesis, however, is unclear as no consistent trends have been observed. In sediments receiving compositionally uniform source materials, the post-depositional changes in isotopic abundance, if any, tend to be small relative to the differences between endmember values (Libes, 1983; Sweeney et al, 1978; Degens, 1969). This suggests that early diagenesis probably does not dramatically affect the original isotopic signature of the sedimentary organic matter.

These potential problems notwithstanding, efforts have been made to use isotope ratios as waste-specific tracers. For example, Burnett and Schaeffer (1980) estimated the fraction of sludge-derived organic carbon in sediments of the New York Bight Apex. Myers (1974) and Sweeney, *et al.*,(1978) in separate studies of a waste outfall system in California combined the elemental abundance and isotopic composition of carbon and nitrogen to develop a simple two source mixing model. The model assumes that elemental and isotopic abundances of the endmembers are known, constant and significantly different and that no alterations occur after sedimentation. Although simplistic, these assumptions allow two independent estimates of the contribution of waste organic matter to be made if measurements of %C and $^{13}C/^{12}C$ (or %N, $^{15}N/^{14}N$) in the sediments are performed. Figure 3 shows an ideal mixing curve for carbon along with data for sediments from a core

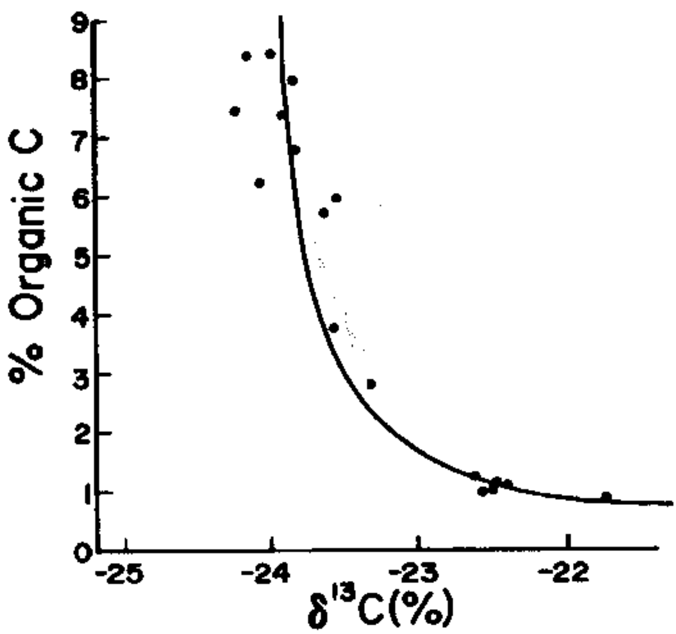

Figure 3. Per cent organic carbon versus $\delta^{13}C(^0/_{00})$ values for sediments taken on San Pedro Shelf, California. Line depicts ideal mixing between sewage and marine endmembers (cf. Eganhouse and Kaplan, 1984).

collected about 6 km downstream from a major outfall system in Southern California (Eganhouse and Kaplan, 1984). The fit is reasonably good, suggesting that either parameter can be used to estimate the fraction of waste organic matter in the sediments (Figure 4). Similar results have been obtained with nitrogen (Eganhouse and Kaplan, 1984). Thus, under appropriate conditions in nearfield sites, the history of waste deposition can

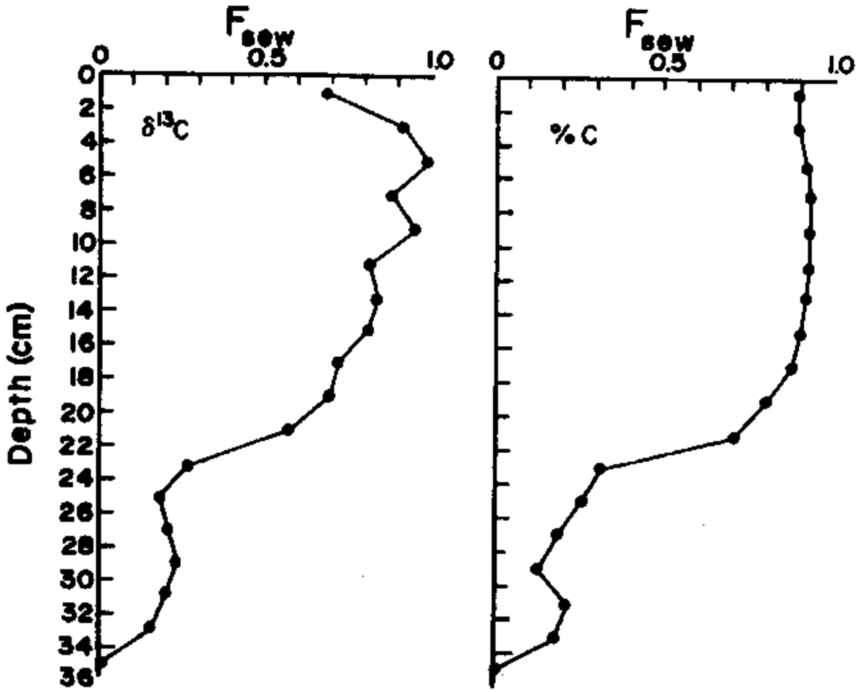

Figure 4. Vertical profile of the fraction of sedimentary organic carbon due to sewage. Values calculated using two source ideal mixing model (cf. Eganhouse and Kaplan, 1984).

be reconstructed, and the sedimentary mass accumulation of waste-derived organic matter can be estimated.

<u>Molecular Tracers</u>

Molecular tracers have one property in common-a high degree of source specificity. Information is stored in the particular spatial arrangement (rather than the abudance or nuclear composition) of its constituent atoms. It is immediately obvious, therefore, that molecular tracers have a significant advantage over elemental abundance and isotopic ratios in that they can, in principle, be used to detect waste impacts in the far field and at very high dilutions. On the other hand, being specific, rather than generic, tracers, they are less likely to be representative of the total organic matter.

Three compounds which appear to have potential as molecular tracers of municipal wastes are: coprostanol, the linear alkylbenzenes (LABs) and vitamin E acetate (cf. Figure 5). The use of coprostanol as an indicator of fecal contamination dates back to the 1960s (Walker, et al., 1982). This sterol is the major product of microbially-mediated, stereospecific hydrogenation of cholesterol in the intestinal tracts of mammals. It is rarely found in uncontaminated sediments except in trace amounts. Instead, the thermodynamically more stable 5α(H) epimer, cholestanol, is the stanol normally encountered. As marine mammals represent a trivially small fraction of the total biomass of the oceans, the identification of coprostanol in coastal sediments is taken as evidence of sewage contamination. However, a remaining unanswered question with respect to the waste-specificity of coprostanol is the importance of its in situ generation from cholesterol and other precursors. Studies involving the incubation of radiolabelled cholesterol in enrichment cultures (Taylor, et al., 1981), sediments and sludge (Gaskell and Eglinton, 1975) demonstrate that this transformation can occur naturally. However, the kinetics of coprostanol generation and decomposition have not been directly compared.

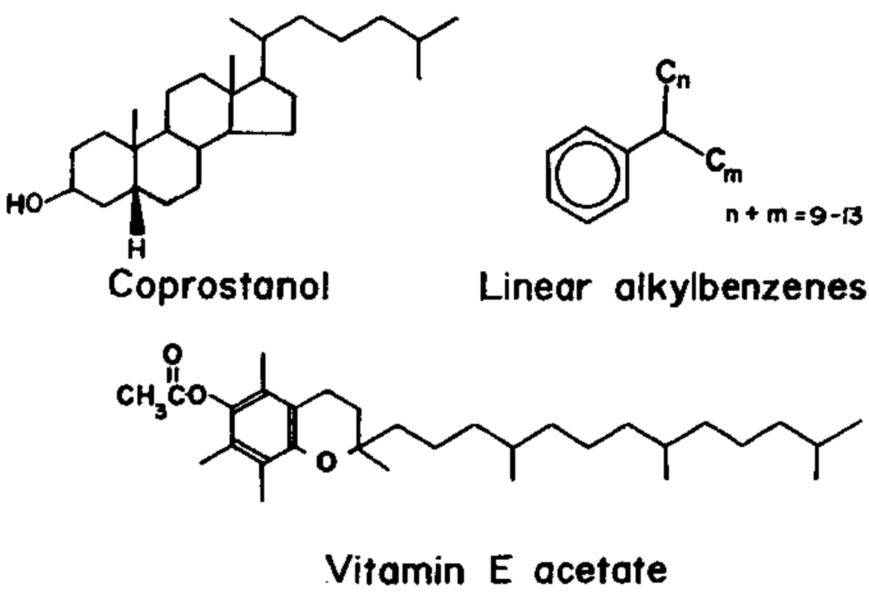

Figure 5. Structures of three potential molecular tracers of municipal wastes.

The linear alkyl benzenes comprise a group of 26 individual secondary phenylalkanes with alkyl side chains having 10-14 carbons. These compounds are synthesized for the production of linear alkylbenzenesulfonates, the anionic surfactants most commonly used in commercial detergents. The presence of LABs in the municipal effluents of southern California was attributed to the disposal of detergents (Eganhouse and Kaplan, 1982b). Because these products are so widely used, the LABs are expected to be present in virtually all effluents having a significant domestic component. Their accumulation in sediments and suspended matter in the vicinity of a major outfall system has been presented as unequivocal evidence of waste contamination (Eganhouse, et al., 1983).

Vitamin E acetate is an industrial product formed by acetylation of either natural or synthetic Vitamin E. The acetate is not known to be formed naturally, and its presence in uncontaminated sediments has never been reported. Recently it was identified as a constituent of municipal effluents where it is believed to arise from disposal of a variety of products such as vitamin supplements, cosmetics and certain foods (Eganhouse and Kaplan, 1985). Apparently, this molecule is sufficiently stable to have survived the earliest stages of diagenesis as it was also found in sediments near a major waste outfall system. Its widespread occurrence in municipal effluents has not been confirmed.

The detectability of molecular tracers is likely to be highly variable. Sensitive analytical techniques such as gas chromatography-mass spectrometry are capable of detecting these compounds at the picogram level. However, the crucial data needed to define detectability, namely, their concentration in waste effluents and their removal rates in the ocean, are not available.

As a simple exercise, the volumes of seawater needed for quantitation of the three potential tracers at several dilutions were computed (Table 1). The calculations are based on several assumptions and a rather meager data base. The first assumption is that the removal of each compound is 50%. While this figure will certainly vary from one compound to the next for a given set of conditions, the value chosen is probably an overestimate for short term exposures typical of those expected during initial dilution (Hatcher, et al., 1981). Second, the limit of detection (LOD) is set at approximately 20 picograms assuming flame ionization is used. GC/MS in the selected ion monitoring mode should be able to provide detection down to an order of magnitude lower. For the purposes of this calculation, the limit of quantitation was set at ten times the LOD or 200 picograms. This is most likely an overestimate. Finally, concentrations of the potential tracers in sewage sludge are taken from literature sources and unpublished data for the Hyperion 7-mile effluent in Los Angeles. This sludge is diluted with secondary effluent to facilitate discharge through an outfall. Thus, it contains approximately five times less suspended solids and coprostanol than the sludge dumped at the 12 mile site in the New York Bight (cf. Eganhouse and Kaplan, 1982a; Hatcher, et al., 1981).

Table 1. Volumes of seawater-diluted sludge needed for determination of coprostanol, the linear alkylbenzenes and vitamin E acetate.

		Volumes needed for analysis (ml) Dilutions[b]			
Compound	Concentration in sludge (ng l^{-1})[a]	10^2:1	10^4:1	10^6:1	10^7:1
coprostanol	30×10^6	.0013	0.13	13	130
linear alkylbenzenes	5.8×10^6	0.2	20	2000	2×10^4
vitamin E acetate	1.1×10^6	0.04	4	400	4000

[a]Data from Eganhouse and Kaplan, 1985 and Eganhouse, unpublished results.

[b]Volumes for the LABs computed as that necessary for an individual component where the average concentration in seawater for an individual component = 0.11 ng l^{-1}.

These calculations indicate that determination of the tracers should be feasible at dilutions exceeding 10^6. Recent modelling efforts (O'Connor, et al., 1983) predict that under low dispersion conditions, sludge dilutions in the upper mixed layer of the ocean at the 106 mile dumpsite should be about 5×10^5 within 20 km of the discharge point. Thus, the tracers should be useful for determining sludge dilutions at deepwater disposal sites on the continental slope. Conclusive experimental confirmation of these predictions needs to be made both in the laboratory and the field.

The key to full implementation of these and other potential molecular tracers clearly rests on their physico-chemical behavior and resistance to degradation. Although information in this area is sparse, a few clues exist. It is very likely that all of the compounds mentioned here will strongly associate with particulate matter. Both the LABs and vitamin E acetate are hydrophobic. Likewise, coprostanol should exhibit a low aqueous solubility due to its largely polycyclic aliphatic structure (Walker, et al., 1982). In fact, these substances do appear to be highly enriched in the particulate phase of sewage effluents (Eganhouse, unpublished results). Consequently, they would be expected to accumulate in sediments assuming the kinetics of their decomposition were moderate. In this regard, it is reassuring that all of these compounds have been observed in waste-impacted sediments.

Little can be said about the resistance of the three potential tracers to degradation as no studies specifically aimed at answering this question have been undertaken. They appear to be enriched in the particulate matter of anaerobically digested sludge when compared with primary effluent, suggesting some capacity for preservation, at least under anoxic conditions. In the absence of actual data, any further statements would be speculative at this time.

3. FUTURE DIRECTIONS/REALISTIC EXPECTATIONS

It is clear that a good deal more work is needed before the methods discussed here can be implemented as quantitative tools. Following is a list of the areas requiring further study toward this end.

1) The organic composition of waste effluents and its variation with time must be more firmly established. The concentrations of species identified as potential tracers should be determined, and efforts should be made to identify new potential tracers. This will increase the number of parameters available for independent corroboration of field results.
2) The physicochemical properties and partitioning behavior of potential tracers should be investigated and compared with those of relevant pollutant species.
3) Studies of the decomposition of waste organic matter in the ocean should be undertaken. We desperately need information on the kinetics of degradation, possible isotopic fractionations and the intermediates formed. The significance, if any, of _in situ_ production of certain tracers should be examined and the influence of salient environmental parameters must be established.
4) Data from the studies described above should be used to construct models of the transport and geochemical fate of tracers and pollutants introduced to the ocean. Model results can then be compared with physical oceanographic and tracer data taken at other existing and newly designated disposal sites.

It is useful to summarize some of the advantages and limitations of the methods discussed here. Elemental abundance and stable isotopic composition are generic tracers. They will probably be of greatest value when used in conjunction near discharges occurring in estuaries and nearshore coastal areas. Here the waters are shallow and currents cannot effectively disperse the wastes, leading to substantial accumulation of organic matter in the sediments. With increasing distance from such discharges or in the case of deepwater dumping on the continental slope, these parameters will be of little use because of the large dilutions and the importance of other interferring inputs.

Under these circumstances, molecular tracers of high source-specificity are needed. As analytical detection limits in the picogram range are attainable, the crucial limitations for the use of molecular tracers in the far field will be their concentrations in the effluent and the kinetics of removal. At present, neither of these are known with accuracy. A preliminary conservative calculation indicates that three compounds, coprostanol, the linear alkylbenzenes and vitamin E acetate, should be measurable at seawater dilutions up to or greater than 10^6:1. Thus, one immediate application of these potential tracers is as a means of determining the magnitude of waste dilution both initially and with distance from the point of discharge. Other uses include the evaluation of the spatial dispersion of sewage by analysis of sediments and the estimation of waste accumulation rates in sediments. The implementation of these methodologies is clearly predicated on a sustained effort to elucidate the biogeochemistry of both tracers and pollutants alike.

4. REFERENCES

Burnett, W.D., and Schaeffer, O.A., "Effect of Ocean Dumping on the $^{13}C/^{12}C$ ratios in Marine Sediments from the New York Bight," Estuarine and Coastal Marine Science, vol 11, pp. 605-611, 1980.

Degens, E.T., "Biogeochemistry of Stable Carbon Isotopes," in Organic Geochemistry, eds. G. Eglinton and M.T.J. Murphy, pp. 304-329, Springer-Verlag, Berlin, 1969.

Eganhouse, R.P., and Kaplan, I.R., "Extractable Organic Matter in Municipal Wastewaters. 1. Petroleum Hydrocarbons: Temporal Variations and Mass emission Rates to the Ocean," Environmental Science and Technology, vol 16, pp. 180-186, 1982a.

Eganhouse, R.P., and Kaplan, I.R., "Extractable Organic Matter in Municipal Wastewaters. 2. Hydrocarbons: Molecular Characterization," Environmental Science and Technology, vol 16, pp. 541-551, 1982b.

Eganhouse, R.P., Blumfield, D.L., and Kaplan, I.R., "Long-chain Alkylbenzenes as Molecular Tracers of Domestic Wastes in the Marine Environment," Environmental Science and Technology, vol 17, pp. 523-530, 1983.

Eganhouse, R.P., and Kaplan, I.R., "Application of Organic Geochemical Techniques to the Study of Waste-Impacted Marine Sediments," To appear in Proceedings of the 5th International Ocean Disposal Symposium, Corvallis, Oregon, September 10-14, 1984.

Eganhouse, R.P., and Kaplan, I.R., "α-Tocopheryl Acetate as an Indicator of Municipal Waste Contamination in the Environment," Environmental Science and Technology, vol 19, pp. 282-285, 1985.

Gaskell, S.J., and Eglinton, G., "Rapid Hydrogenation of Sterols in a Contemporary Lacustrine Sediment," Nature, vol 254, pp. 209-211, 1975.

Gearing, J.N., Gearing, P.J., Rudnick, D.T., Requejo, A.G., and Hutchins, M.J., "Isotopic Variability of Organic Carbon in a Phytoplankton-Based, Temperate Estuary," Geochimica et Cosmochimica Acta, vol 48, pp. 1089-1098, 1984.

Grunseich, G.S., and Duedall, I.W., "The Decomposition of Sewage Sludge in Seawater," Water Research, vol 12, pp. 535-545, 1978.

Hatcher, P.G., Berberian, G.A., Cantillo, A.Y., McGillivary, P.A., Hanson, P., and West, R.H., "Chemical and Physical Processes in a Dispersing Sewage Sludge Plume," in Ocean Dumping of Industrial Wastes, eds. B.H. Ketchum, D.R. Kester and P.K. Park, pp. 347-378, Plenum Press, New York, 1981.

Hedges, J.I., and Parker, P.L., "Land-Derived Organic Matter in Surface Sediments from the Gulf of Mexico," Geochimica et Cosmochimica Acta, vol 40, pp. 1019-1029, 1976.

Jeffrey, A.W.A., Pflaum, R.C., Brooks, J.M., and Sackett, W.M., "Vertical Trends in Particulate Organic Carbon $^{13}C/^{12}C$ Ratios in the Upper Water Column," Deep-Sea Research, vol 30, pp. 971-983, 1983.

Libes, S.M., "Stable Isotope Geochemistry of Nitrogen in Marine Particulates," Ph.D. Thesis, 288 p., MIT/Woods Hole Oceanographic Institution Joint Program, 1983.

Minagawa, M., Winter, D.A., and Kaplan, I.R., "Comparison of Kjeldahl and Combustion Methods for Measurement of Nitrogen Isotope Ratios in Organic Matter," Analytical Chemistry, vol 56, pp. 1859-1861, 1984.

Myers, E.P., "The Concentration and Isotopic Composition of Carbon in Marine Sediments Affected by a Sewage Discharge," Ph.D. Thesis, California Institute of Technology, Pasadena, 179 p., 1974.

Myers, E.P., "Introduction," in Ocean Disposal of Municipal Wastewaters: Impacts on the Coastal Environment," ed. E.P. Myers, pp. 3-36, MIT Sea Grant Publications, Cambridge, Massachusetts, 1983.

O'Connor, T.P., Okubo, A., Champ, M.A., and Park, P.K., "Projected Consequences of Dumping Sewage Sludge at a Deep Ocean Site Near New York Bight," Canadian Journal of Fisheries and Aquatic Sciences, vol 40, pp. 228-241, 1983.

Peters, K.E., Sweeney, R.E., and Kaplan, I.R., "Correlation of Carbon and Nitrogen Stable Isotope Ratios in Sedimentary Organic Matter," Limnology and Oceanography, vol 23, pp. 598-604, 1978.

Rebhun, M., and Manka, J., "Classification of Organics in Secondary Effluents," Environmental Science and Technology, vol 5, pp. 606-609, 1971.

Schultz, D.J., and Calder, J.A., "Organic Carbon $^{13}C/^{12}C$ Variations in Estuarine Sediments," Geochimica et Cosmochimica Acta, vol 40, pp. 381-385, 1976.

Sweeney, R.E., Liu, K.K., and Kaplan, I.R., "Oceanic Nitrogen Isotopes and Their Uses in Determining the Source of Sedimentary Nitrogen," in Stable Isotopes in Earth Sciences, ed. B.W. Robinson, DSIR Bulletin, vol 220, pp. 9-26, 1978.

Tan, F.C., and Strain, P.M., "Organic Carbon Isotope Ratios in Recent Sediments in the St. Lawrence Estuary and Gulf of St. Lawrence," Estuarine and Coastal Marine Science, vol 8, pp. 213-225, 1979.

Taylor, C.D., Smith, S.O., and Gagosian, R.B., "Use of Microbial Enrichments for the Study of the Anaerobic Degradation of Cholesterol," Geochimica et Cosmochimica Acta, vol 45, pp. 2161-2168, 1981.

Walker, R.W., Wun, C.K., and Litsky, W., "Coprostanol as an Indicator of Fecal Pollution," CRC Critical Reviews in Environmental Control, vol 10, pp. 91-112, 1982.

Design of Monitoring Studies to Assess Waste Disposal Effects on Regional to Site Specific Scales

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1. INTRODUCTION

Marine pollution monitoring has been defined as "the continual systematic time series observation of pre-determined pollutants or pertinent components of the marine ecosystem for a period of time sufficient to determine the 1) existing level, 2) trend, 3) natural variations of measured components, water column, sediments, or biota" (NOAA, 1979). The purpose of marine pollution monitoring was defined as "to obtain time series, data sets that can be used to detect SIGNIFICANT changes in the environment and to use this information to provide timely warning and other advice to management so appropriate actions may be taken" (NOAA, 1979, emphasis added). The key concept within this definition of the purpose of marine pollution monitoring is that the changes that are to be addressed through monitoring must be SIGNIFICANT to the marine environment and, therefore, to management needs. However, most programs intended to monitor the effects of ocean waste disposal have ignored this concept and have established monitoring designs that are based upon several implicit concepts, including:

A) The concentration of every contaminant for which concern has been expressed in any part of the global environment should be monitored in the waste material and in the water column, sediments, and biota at or near the discharge or dumping site in order to detect ANY increase in contaminant concentrations caused by the waste disposal. This monitoring should be done even if, based upon knowledge of the composition of the waste and the fate of the waste in the ocean, it is known that it is not possible for a measurable and significant increase in contaminant concentrations to occur.

B) The biota at or near the site should be monitored such that ANY changes in the abundance of ANY species which are caused by waste disposal can be identified.

These precepts reflect admirable, but unrealistic, impossible, and unnecessary goals, since the number of samples analyzed and the number of parameters that would need to be measured on each sample and at each site in order to approach attainment of these goals would be prohibitive. Therefore, trade-offs are made to bring the monitoring program within the bounds of practicability. Unfortunately, there is always a reluctance to drop the measurement of any particular contaminant or community of organisms, and, therefore, it is the intensity of sampling for each parameter which is invariably reduced. Consequently, each parameter is not measured with sufficient frequency in time and/or space to enable even substantial changes in the monitored parameter to be identified in a statistically-valid manner. Therefore, the monitoring exercise is futile, since a series of measurements for each parameter is obtained whose variations cannot be interpreted to identify trends unless many years, or even decades, of data are available. In addition, monitoring programs attempting to meet the goals stated above can never be successful, since, if the amount of change which is considered significant is not predetermined, any finite data set will always leave open the possibility that some extremely small change may have occurred that is below the resolution of the measurement system or smaller than the natural change. Accordingly, it is impossible to prove that absolutely no change has occurred due to any activity in the ocean, including waste disposal.

The deficiencies in marine waste disposal monitoring design are largely responsible for the controversial nature of all marine waste disposal management decisions. Since no guidelines have been agreed upon between policy-maker, manager, and scientist concerning the level of environmental change that is considered to delineate the demarkation between acceptability and unacceptability (referred to throughout this paper as the maximum acceptable effect level - MAEL), the scientific data usually support both those who wish to use the ocean for waste management and those who do not. On the one hand, the data do not show statistically-significant changes due to the waste disposal; on the other hand, there remains an unquantified possibility of a change having occurred which is below the resolution of the monitoring data and which may, or may not, be environmentally significant. For this reason, it is important that waste disposal monitoring programs be designed on the basis of i) a systematic evaluation of the management information needs, ii) the hypothetical impacts associated with management concerns, and iii) the feasibility of monitoring these hypothetical effects in such a manner that statistically-valid identification of the existence, or absence, of a specified level of effects can be achieved.

2. MONITORING OBJECTIVES, HYPOTHESES, AND STATISTICAL DESIGN

The most important step in designing waste disposal monitoring programs is definition of the objectives of the program. The broad objective of such programs is to provide the ocean use manager with information that can enable him to i) ensure that human health is not threatened, ii) ensure that unacceptable harm is not done to the marine ecosystem or marine resources, and iii) make informed decisions concerning continued or expanded use of the ocean for waste disposal. This broad objective must be further defined by the manager before monitoring program planning can proceed since, in order to design a useful program, answers must be obtained for questions such as: i) what level of incremental risk to human health from waste disposal is considered to be a threat to human health, ii) what is the type and degree of community structure change and what is the minimum area of ocean in which such a change must occur before it is regarded as unacceptable, iii) what ocean resources are at risk from the waste disposal, what type of adverse effects on these resources is possible, and how large must such an effect be before it is considered unacceptable? Rarely are these questions properly answered, particularly the questions which address the degree of change that is acceptable/unacceptable. The blame for this deficiency lies not solely with the politician or manager, but also with the scientist. All of these parties have failed to understand or acknowledge that i) no waste management strategy is without concomitant effects on man and the environment, ii) some degree of effect on the marine environment or man from ocean waste disposal is acceptable, and iii) no monitoring program can ever demonstrate that a waste disposal activity has ZERO effect on a particular resource.

Objectives of marine waste disposal monitoring programs must be carefully defined in accordance with realistic management needs. At present, most program objectives are vague and imply that the program should identify and measure any and all effects of the waste disposal operation on the ecosystem and man. Instead, the program objectives should be to demonstrate that specific adverse effects do not occur above specific predetermined levels. The predetermined effect levels will be somewhat lower than the MAEL's to provide a reasonable safety margin which will ensure that effects are detected before they reach unacceptable levels. In addition, there must be a significant probability that the hypothesized effect could occur at or near the MAEL.

One consequence of the need to define specific objectives for ocean waste disposal monitoring programs is that separate and distinct objectives can and should be established for monitoring the effects of waste disposal at and immediately adjacent to each point source (e.g., dumpsite, outfall location, estuary mouth) and for monitoring the broader-scale regional effects that may occur as a result of multiple point sources of waste disposal and other contamination.

Based upon the preceding considerations, it can be concluded that a successful ocean waste disposal monitoring program requires that, prior to the design of the program itself, a series of program objectives be established each of which defines:

A) The specific effect to be monitored (e.g., change in contaminant abundance in biota or sediments, reduction in benthic species diversity, lowered dissolved oxygen).

B) The level of effect that is to be considered to demark the acceptable/unacceptable boundary (the MAEL). This level must specify the area over which the effect occurs, the degree of change (e.g., percentage increase in sediment contaminant concentration, exceedance of specified infaunal trophic index value), and sometimes the rate of change (e.g., sediment contaminant concentration not to increase by specified percentage per year). Only when specific objectives such as these have been established can the scientific method of establishing and testing hypotheses be used to design monitoring programs.

Properly managed ocean waste disposal is expected to result in no unacceptable adverse environmental effects. Therefore, the appropriate approach for monitoring such disposal is the testing of null hypotheses. Appropriate null hypotheses are similar regardless of the hypothesized effect and take the general form:

> The waste disposal activity has not caused (or will not cause) a specific attribute of the ocean ecosystem (e.g., species diversity, coliform concentration, fish population) to change by a specified amount (magnitude and geographical extent of change usually must both be specified).

Since many attributes of the ocean ecosystem which may be affected by waste disposal may also be affected by natural changes and by other anthropogenic influences, this hypothesis usually must be restated as two-component, dependent hypotheses of the form:

> Component 1. A specific attribute of the ocean ecosystem has not changed by a specified amount (or does not differ from defined background conditions).
>
> Component 2. Changes in the specific attribute above the specified amount are not related to the waste disposal.

The second, or dependent, hypothesis need not be tested and, in fact, is untestable unless the first is tested and disproven (i.e., some change has occurred). Since establishment of cause-and-effect relationships between ocean ecosystem changes and waste disposal or other anthropogenic activities is often extremely difficult, monitoring programs which address the first component hypothesis alone are much simpler and more likely to succeed. In those instances where change beyond the acceptable limits is observed, specific research studies and/or monitoring studies to address the second component hypothesis can be performed. However, in most cases of properly managed waste disposal this will not be necessary or will only be required for one or two of the several hypothesized effects addressed by the monitoring program.

It is often argued that the cause-and-effect hypothesis must be addressed in waste disposal monitoring programs, because waste disposal will always result in some degree of environmental change which must be identified and measured. However, environmental change of some small degree due to waste disposal will be acceptable and of no concern to the manager. Therefore, it is not necessary for a monitoring program either to detect such small changes or to describe a cause-and-effect relationship. When the MAEL is conservatively defined and the first component hypothesis is proven for a level of change significantly below the MAEL and with acceptable statistical certainty (i.e., no unacceptable level of change has occurred), the waste disposal monitoring program has totally fulfilled its purpose. Certainly, in specific situations, there is a need to perform research to study waste disposal induced environmental effects that may be substantially smaller than the MAEL. However, such research should not be included in the monitoring program, since such research would almost inevitably compromise the monitoring program's effectiveness, usually because appropriate MAEL's are not established.

The three essential elements of a null hypothesis are the space and time scales on which it is desired to observe differences in ocean attributes, and the magnitude of the smallest change or difference that the program must observe and statistically verify (i.e., the "resolution" required). While many programs do not adequately define the space or time scales to be addressed, the third essential element, the required resolution, is almost invariably ignored until sampling and analysis are completed and the data are to be analyzed. Consequently, resources are wasted because either i) the sampling program is inadequate and does not provide the resolution required (i.e., the data show change was not observed or verified, although the minimum observable change was so high that a change whose magnitude exceeds the MAEL might have existed), or ii) the sampling program is intensive and identifies a change or difference that is so small as to be of no environmental significance, particularly to the pollution manager.

Establishment of the required resolution is not a simple task since the selected resolution must be i) scientifically attainable, ii) attainable through a sampling and analysis program which can be accomplished with available resources, and iii) environmentally significant. The selected resolution is not environmentally significant if it is i) so small that there is no potential threat of environmental damage (i.e., so much smaller than the MAEL as to be insignificant), ii) so large that there is no credible scenario in which it would occur, or iii) so large that, once observed, substantial environmental damage would already have occurred (i.e., greater than the MAEL). Selection of the appropriate MAEL and resolution for a given program hypothesis requires a dialogue between scientists and managers and a substantial amount of information concerning anthropogenic inputs, toxicity of specific contaminants, the manner in which marine ecosystems respond to stresses of different types and magnitudes, and the environmental concentrations of contaminants and biological population structures and their variabilities. Although some of these data are usually available, it will be necessary in most cases to obtain additional data concerning the environmental variance in the contaminant concentrations or in biological populations in the specific study location in order to determine the appropriate resolutions.

In many cases, the design study will result in a conclusion that an environmentally-significant resolution cannot be achieved because of technical limitations (e.g., extreme spatial and/or temporal environmental variability) and/or resource limitations which prohibit sufficiently intensive sampling and analysis. In such instances, testing of this hypothesis should not be performed since it can only result in equivocal results and an alternate null hypothesis addressing some other indicator parameter or effect should be used. In other cases, a properly performed design study will result in a less expensive program since the desired resolution can be successfully achieved through a more limited monitoring effort. In all cases, the establishment of a required resolution and the design of a program to achieve this resolution will prevent the otherwise-inevitable and unrealistic expectations of the monitoring program's ability to identify subtle environmental effects: expectations which are not, and cannot be, fulfilled by an improperly designed program. In addition, interpretation of the data resulting from waste disposal monitoring programs which address null hypotheses with predetermined resolutions will be simpler for both scientist and manager.

Once the specific program objective(s) and null hypotheses are defined, other program details must be decided, including the number and location of sampling sites, number of samples per site, frequency of sampling, number of replicate analyses per sample, and whether to time and space bulk. These choices are made to achieve the required resolution with the smallest possible commitment of available resources. Since these available resources (e.g., analytical capacity, and shiptime cost and availability) and the monitored environment vary, each program will be designed differently. However, a successful design for virtually all program hypotheses must achieve the same end: the mean value of the measured parameter within a chosen spatial element and the variance of that mean must be established such that, if any two such means for different spatial elements (or time periods within the same spatial element) differ by the required resolution, the variances of the two means are sufficiently small to provide a specified (usually 90%) probability that the two means are different.

Sources of variance in measured values of environmental contaminant concentrations or biological population characteristics are many and include: i) spatial variability on geographical scales smaller than those of interest; ii) temporal variability on time scales shorter than those of interest, including seasonal variations; iii) sample handling variability due to irreproducibility in obtaining, storing, and preparing the sample for analysis; and iv) measurement variability. The nature and extent of each of these variances must be understood before a sampling program can be properly designed. While some sources of variance are fairly well-characterized and are almost invariable between different monitoring programs, other sources must be assessed in the specific study area, either through historical data or preliminary field studies designed for this purpose. Once the magnitude of each source of variance is known, optimum sampling and analysis programs can be designed through fairly simple and routine statistical calculations (e.g., Gordon et al., 1980).

The selected sampling strategy may need to vary for different hypothesized effects or even for hypotheses addressing potential increases in concentrations of different contaminants within the same program. For example, Satsmadjis and Voutsinou-Taliadouri (1983) in their study of contaminants in bio-indicators observed that analytical variance was large compared to small-scale spatial variance for organic compounds, but that analytical variance was much smaller than small-scale spatial variance for trace metals. In such a situation, an optimum sampling strategy in which the total number of analyses is limited may dictate that a similar number of individuals should be taken from each site for analysis of both contaminant classes. The samples for metals should consist of a relatively large number of bulked, or even single individual, samples, each of which is analyzed once. In contrast, for organics, it might be preferable to utilize a single bulked sample and subject it to multiple replicate analyses. The variance of the estimated means for metals and organics will be different. However, in each case, it will be as low as possible within analytical cost constraints.

Statistical design of marine waste disposal monitoring programs is currently not sufficiently tested or well-documented in the literature. Nevertheless, the use of statistically-valid designs is imperative to the success and efficiency of such programs. Clearly, more experience is needed with statistically-designed programs before an optimum statistical design approach is identified.

3. SITE-SPECIFIC OCEAN WASTE MONITORING

All current marine waste disposal activites take place at defined locations within the ocean environment: either at outfalls, dumpsites, or the discharges at mouths of estuaries. Therefore, most ocean waste disposal monitoring programs are designed to ensure that any effects of discharging a specified material at a specified location are within acceptable limits. In most cases, the potential adverse effects of concern can be stated in general terms, as follows:

A) Contaminants in the waste material may be transported to beaches where they may threaten human health or aesthetic values.

B) Contaminants in the waste material may enter the marine food chain and be responsible for significant increases in human health risk due to consumption of seafood with elevated levels of these contaminants.

C) The waste may cause unacceptable damage to the marine biological communities in the area affected by the disposal.

Other concerns are often stated when establishing program objectives (EPA, 1983, Segar et al., 1984), such as a concern that toxic contaminant concentrations may increase in sediments. However, this and other similar concerns are not different in substance from the three basic concerns stated above. They are, in fact, only indicative of conditions which may be associated with the three basic effects of concern, or causative factors which lead to one of the three basic effects. Although marine waste disposal monitoring programs are ostensibly intended to address the three basic concerns listed above, little thought is usually given to the means by which these concerns can be most efficiently and effectively addressed. Too often the program simply follows historical precedents set by other such programs and consists of a grid of stations surrounding the disposal site at which measurements are made of i) water column chemistry, including nutrients, trace metals, and often coliforms or other microbiological indicators; ii) hydrographic measurements in the water column and a limited number of current measurements; iii) sediment chemistry, including organic matter, trace metals, synthetic organic compounds, and often coliforms or other microbiological indicators; iv) benthic infaunal community structure; v) fish populations; and vi) phytoplankton and zooplankton populations. In some programs, additional measurements, such as surface slick sampling and determination of trace metals and synthetic organics in fish and benthic infauna and epifauna, are also made. There is usually no clear concept or understanding of how the data obtained from such studies are to be used to address the managers' concerns.

One monitoring program designed in this manner is the compliance monitoring program performed at the 12-mile municipal sludge dumpsite in the New York Bight. Because the principal fear in the New York Bight was that beach contamination could result from disposal at this site, the station grid selected for this monitoring program covered only areas inshore from the site, despite the fact that it was known when the program was established that the normal transport was offshore. This program, which cost in excess of $300,000/annum over a period of almost a decade, resulted in data which were, with minor exceptions, useless in the assessment of the effects of disposal at this site on human health and the marine environment (Ecological Analysts and SEAMOcean, 1983a, b).

Following the lack of success of programs such as the New York program, many other monitoring programs, including the developing 301h waiver monitoring program, have been modified. Unfortunately, these programs have been modified not by better fundamental design, but by simply adding more stations and more parameters to be measured in the belief that somehow the greater quantity of data will provide effective answers to program goals. The monitoring program now being developed for the Deepwater Municipal Sludge Dumpsite (DMSDS) off the edge of the northwest Atlantic continental shelf, as originally conceived by EPA (EPA, 1983), was among these poorly designed programs. However, this proposed program has been further studied and these studies provide the basis for design of a rational, effective monitoring program. O'Connor et al. (1983, 1985) have proposed approaches to monitoring certain aspects of the effects of the proposed disposal at the DMSDS which are based on calculations of the maximum possible effect from disposal of known quantities of municipal sludge at the site. Based on these calculations, these authors have demonstrated that many of the measurements included in the original concept of the EPA program are too insensitive to enable detection of even the largest possible potential effect level. However, even this study falls short of being satisfactory since no attempt was made to establish MAEL's and statistical considerations were not taken into account. Segar et al. (1984) have discussed the DMSDS monitoring program and have suggested approaches to the design of an effective program which are based on the scientific method, concerns expressed by EPA (1983), and the data and calculations included in O'Connor et al. (1983, 1985).

As a result of these and other studies, it may be concluded that the design of an effective, site-specific marine waste disposal monitoring program should be conducted as follows:

A) Establish general objectives (e.g., unacceptable degradation of benthic biota must be detected, if it occurs).

B) For each general objective, establish one or more specific objectives, including an MAEL. These specific objectives can be either the direct detection of the effect itself or detection of some other change in the marine environment whose relationship with the waste disposal and the monitored effect is known, preferably a change known to have a cause-and-effect relationship with the effect of concern (e.g., to demonstrate that the infaunal trophic index does not exceed a specified number outside the dumpsite or initial mixing zone, and/or to demonstrate that the concentration of cadmium in sediments does not increase by more than a specified amount at any location).

C) Based on the specific objectives, establish null hypotheses to be tested which specify the parameter to be monitored and the required resolution. This resolution must not be larger than a change in the parameter which is equivalent to or equal to the MAEL, and it should not be so much smaller than the MAEL that it represents an insignificant change.

D) Perform a program design study for each null hypothesis. This design study must:

i) Establish sources and magnitudes of variance associated with measurement of selected parameters, including analytical or other measurement procedural variance, natural geographical variance, and natural temporal variance at the site to be monitored.

ii) Determine whether the maximum potential change that the disposal could induce in the monitored parameter would approach or exceed the MAEL, based upon historical data and knowledge; known chemical, physical, and biological characteristics of the waste; projected quantities to be disposed; and knowledge of the physical and chemical oceanography of the disposal site area. If the maximum potential change is so small that it could not approach or exceed the MAEL, the null hypothesis is unsuitable and it should be eliminated from the program.

iii) Based upon the magnitudes of different sources of variance, establish the optimum sampling and analysis program to achieve the required resolution at the specified level of statistical certainty. This optimum program will specify number and location of stations, number of samples per station, and whether and how samples are to be pooled.

iv) Establish whether the optimum sampling and analysis program can be achieved within practicable limits of program resources. If it cannot, this null hypothesis should be ELIMINATED from the monitoring program since the usual response to this situation (i.e., reducing the number of stations, samples, or analyses) will result in data which cannot provide the manager with the information he needs.

This design process will eliminate from many waste disposal monitoring programs some of the parameters that have traditionally been measured as a matter of course. While marine managers may be reluctant to drop traditionally measured parameters, the advantages offered by effective program design should offset such reluctance. Focussing the monitoring program on those adverse effects that are LIKELY to occur and providing sufficient resources to unequivocally determine whether or not such effects occur at unacceptable levels clearly justify dropping parameters which can only be useless in a given monitoring program.

It is also important to realize that many parameters traditionally monitored are indicative of the same potential effect. For example, various sediment chemistry measurements, benthic infaunal population studies, benthic recruitment measurements, and benthic fauna biochemical stress measurements are all used in monitoring programs to address the same management question: whether the waste disposal has caused unacceptable damage to benthic communities. However, it is certainly not necessary or efficient to monitor all of these parameters if the manager's question regarding benthic community harm can be unequivocally answered by measuring just one or two parameters in a statistically-sound program.

The preceding statements could be misinterpreted as indicating that all monitoring programs should only measure changes in the potentially affected biological communities themselves. In contrast, it is often these community and stress measurements that fail to meet statistical requirements within practicable program constraints. If we have reasonable knowledge from research studies of the cause-and-effect relationships between biological community damage and, for example, sediment contaminant concentrations or sediment acute bioassay results, we can instead monitor those symptomatic parameters to provide a statistically-sound demonstration of acceptable/unacceptable change. Of course, uncertainty in the relationship between the measured parameter and any biological damage will remain. However, if the MAEL for the measured parameter is established conservatively to allow for this uncertainty, adequate environmental protection will be assured, and the unequivocal nature of the monitoring program results will facilitate management of individual waste disposal operations. Controversy regarding uncertainty in the relationship between the monitored parameters would be addressed largely at the more fundamental regulatory level where procedures for establishing specific MAEL's should be established.

While the procedure outlined above would lead to effective and efficient site-specific monitoring programs, several other considerations are important when outlining the basic requirements for design of such programs. Field measurements made at the disposal site in a marine waste disposal monitoring program are important, but data describing the composition and quantity of the disposed waste are undoubtedly more important. Without these data, it is not possible to properly interpret monitoring results or to perform the monitoring program design evaluation that should occur periodically during the lifetime of a waste disposal operation. Site-specific waste disposal monitoring programs should not be invariable over the lifetime of the waste disposal operation. They should be redesigned and modified periodically in response to any significant changes in waste quantity or quality.

Site-specific waste disposal monitoring programs should also be redesigned periodically on the basis of program results and relevant new research findings. If waste disposal is continuous and the waste characteristics and input rate do not change, then the monitored ecosystem will reach a dynamic steady-state. This steady-state will be reached in different time periods depending on the half-life of the waste in a given ecological compartment. After any part of the disposal ecosystem has reached equilibrium, it may be possible to substantially reduce or eliminate some measurements within this part of the ecosystem. For example, water column contaminant concentrations reach a dynamic steady-state within weeks after initiation of a pipeline discharge. If the maximum concentration of a given contaminant in the water column following initial dilution at the outfall is measured over a range of likely hydrographic conditions (usually over one seasonal cycle) and this concentration is always found to be substantially below the MAEL throughout the discharge zone, measurement of this parameter in the monitoring program can be discontinued unless the concentration of the contaminant increases in the waste or the quantity of waste discharged increases. In a similar manner, if a measured parameter is found to approach the MAEL more closely than had been predicted, it may be necessary to measure this parameter more frequently or over a greater area surrounding the discharge site than originally planned.

Monitoring measurements made at, and immediately surrounding a disposal site, combined with information concerning the waste quantities, the waste's physical, chemical, and biological properties, and the transport processes affecting the waste, should be used to determine on a continuing basis the size of the area around the site that should be monitored. In many cases, this on-going re-evaluation will prevent collection of unnecessary data. If a given potential effect does not exceed the MAEL within the zone immediately surrounding the discharge, it will often be possible to conclude that the waste disposal could not cause this effect at levels which exceed the MAEL at more remote locations. Any poorly flushed, accumulative zones remote from the disposal site for which this conclusion might not be valid could be identified from known physical features and currents, and these areas could be separately monitored, if appropriate. The proposed Deepwater Municipal Dumpsite monitoring program in the North Atlantic provides a good example of how this principle could be applied. EPA (1983), in its original monitoring plan for disposal at this dumpsite, proposed a monitoring program extending many hundreds of miles from the site on the basis that, within 100 days, some waste material could reach such locations in extremely small concentrations after extensive dilution. However, O'Connor et al. (1984, 1985) and Segar et al. (1984) have shown that dilution and dispersion of waste disposed at this site would be so great that the potentially toxic constituents of the waste could not be detected more than several miles from the site. Evidence obtained from other studies of disposal of this waste (municipal sludge) elsewhere in the ocean supports a conclusion that any biological effects are small and difficult to detect even where the waste is highly concentrated and readily detectable. Therefore, these authors have concluded that monitoring of the effects of this disposal should be restricted to the site and areas immediately downstream of the site. Further, they concluded that, if adverse effects are not observed and contaminant concentrations do not exceed predicted levels, monitoring for effects at more remote locations would be futile, since there would be no realistic probability of the occurrence of such effects at a measurable and significant (approaching an MAEL) level.

4. REGIONAL MARINE MONITORING PROGRAMS

Site-specific monitoring programs and appropriate research studies can provide most of the information needed to successfully and safely manage marine waste disposal. However, site-specific programs do not provide adequate information to assess whether adverse effects caused by long-term, slow accumulation of individual waste contaminants take place in a wide ocean area as a result of the input of wastes from diverse sources. Therefore, there is a need to develop regional monitoring programs which can effectively address this question.

The adverse effects that potentially could be caused by accumulative contamination of the marine environment are the same as the effects of concern at specific sites: beach contamination, seafood contamination which threatens human health, and unacceptable ecological damage. The first of these concerns, beach contamination, is currently intensively monitored by state and local governments wherever beaches are used extensively for recreation. Since standard monitoring techniques are used by almost all such local and statewide programs, the regional monitoring needed to effectively address this concern is already in place. All that is necessary is better coordination of data-reporting and better evaluation of the data on a regional basis (Segar, 1981). Therefore, we will not address this concern further.

The remaining concerns, human health risk due to seafood contamination and damage to marine ecology, are very different from one another. However, they can be considered together in a regional monitoring program since each effect can only be caused by the same mechanism: increased environmental concentrations of bioavailable toxic contaminants. Ecological damage can also be caused by changes in the physical environment as a result of waste disposal, but such changes are extremely unlikely to occur on regional scales and, if they did occur, would undoubtedly be accompanied by major adverse effects at the specific sites of waste disposal. Therefore, such potential changes in the physical environment due to waste disposal are best addressed by site-specific monitoring. While broad-scale ecological damage due to accumulative effects of waste disposal could conceptually be monitored by direct assessment of the status and health of marine ecological assemblages, this is generally not possible because any differences or changes caused by cumulative effects of waste disposal could not be discerned within the large changes or differences that occur in these communities due to natural factors.

On the basis of these considerations, the general objective of regional monitoring programs can be restated as a requirement that the program identify those broad areas of the coastal ocean where bioavailable contaminant concentrations are elevated or are increasing due to marine waste disposal. This objective can be addressed through component hypotheses as in site-specific monitoring programs: first, a hypothesis that bioavailable contaminant concentrations have changed (or will change); and second, a hypothesis that the change has been caused by waste disposal. The first component hypothesis is the basis for development of specific objectives for regional ocean waste disposal monitoring programs.

Toxic contaminants introduced into the ocean in disposed wastes are subject to complex physical and chemical changes which depend on the physical and chemical nature of the waste material itself and the physico-chemical state of the contaminant and its chemical reactivity. Each contaminant will be partitioned between the sediments, dissolved and particulate phases in the water column, and the biota. A specific contaminant in the solid or dissolved phases in the water column or in the sediments may be in a variety of physico-chemical states which may be biologically available (i.e., potentially toxic to biota or man through biological uptake), or unavailable. Since it is only the bioavailable fraction of the contaminants (including the fraction which may become bioavailable through physico-chemical changes) that is of interest to the marine waste disposal manager, the most direct approach to regional monitoring programs is the measurement of contaminants in marine biota. Since different species concentrate bioavailable contaminants by different mechanisms and with different concentration factors, monitoring of bioavailable contaminants must be performed through the use of selected bioindicator species whose biology, biochemistry, and contaminant uptake characteristics are well known. The same species is usually used throughout the monitored region but two or more species can be used if their characteristics are known and interspecies calibrations can be made where their ranges overlap. The use of caged organisms transplanted to the sampling site from a controlled population may also be useful for several reasons, including that sampling sites may then be chosen where no suitable bioindicator species occurs naturally.

Phillips and Segar (1985) have recently discussed the factors that must be considered in the design of bioindicator-based regional monitoring programs. This report identified two major, but fundamentally different, general objectives of bioindicator monitoring programs:

A) To delineate spatial variations in the abundance of bioavailable contaminants (i.e., to identify hot spots), and

B) To elucidate changes in the abundance of bioavailable contaminants with time in a given site or area.

The design of bioindicator monitoring programs to address these two objectives must be radically different if the programs are to meet the objectives. However, most bioindicator monitoring programs have attempted to address both objectives using a single sampling scheme and as a result, have been only partially successful, at best.

The design of bioindicator monitoring programs to address either of these two general objectives should be performed in the same general manner as design of site-specific programs. However, for certain contaminants whose natural concentrations are close to toxic levels in the marine environment or for extremely toxic synthetic organics, it may not be appropriate to establish a MAEL for the regional program. The required resolution for these components will often need to be defined by balancing the desire to detect the smallest possible change or difference with the practical constraints of natural variability and sampling and the practicability of the analytical program .

Several unique factors need to be considered in designing regional bioindicator monitoring programs (Phillips and Segar, 1985). These include careful selection of the bioindicator species. Selected species should not metabolically regulate the concentration of the contaminant independent of the environmental concentration and should have known kinetics for uptake and depuration of the contaminant. Selection of suitable species will ensure that the measured contaminant concentration in the bioindicator accurately reflects the concentration of bioavailable contaminant at the sampling site integrated over a known period of time. Site selection for bioindicator monitoring programs must also be carefully performed since small-scale geographical variability of bioindicator contaminant concentration can often be large. Bulking samples taken from several locations within an area can be a useful technique for reducing the number of analyses required to establish an accurate mean bioavailable contaminant concentration from a small geographical area (or site).

Sampling designs for bioindicator programs require particular care since substantial variance in the mean bioavailable contaminant abundance can be introduced through variations in the concentration factors (contaminant body burden/environmental concentration) among organisms of the same species due to differences in their size, age, sex, sexual condition, and physical habitat. Variance from all of these sources can be reduced substantially by a well-designed sampling program. Samples should consist of individuals selected from a defined, narrow range of sizes, ages, and habitats; from a single sex; and at a defined stage of sexual condition. Time bulking, that is bulking for analysis of several samples taken at different time periods during an annual cycle, can be extremely useful in reducing variance, particularly variance due to sexual condition. Without time bulking, sampling along a regional coastline must be carefully sequenced to sample each station when the population is in the same stage of its sexual cycle: a difficult or impossible logistic problem.

Effective regional marine monitoring programs are needed to determine the effects of multiple waste disposal activities and inadvertant contaminant releases to the ocean and to ensure that the assimilative capacity of the coastal oceans is not exceeded. However, as with site-specific monitoring programs, successful regional programs which support management needs must be designed to address testable hypotheses, and traditional ideas of what must be measured in a regional program might be substantially altered by this process.

5. SUMMARY AND CONCLUSIONS

We have attempted in this paper to identify the essential elements of the decision and design process which are required to develop both site-specific and regional monitoring programs capable of detecting SIGNIFICANT changes in the marine environment caused by waste disposal and providing timely warning and other advice required by MANAGEMENT in order that appropriate actions might be taken. The process that we have outlined is a lengthy one which requires substantial inputs from both marine managers/policy-makers and the scientific community. Inputs are required which include both subjective judgements concerning the acceptability of different types and levels of effects and rigorous scientific analysis and use of available information. Many of the necessary design steps are difficult and could be controversial. Perhaps for this reason, these steps have usually been omitted and most marine waste disposal monitoring programs have been developed through an unsatisfactory process. This unsatisfactory process usually entails adopting all of the parameters that can be measured within practical program constraints, or that have been measured in other such programs and sampling on a randomly- or poorly-selected grid of stations around the disposal point or region.

Several of the steps required for successful monitoring design involve analyses or decision processes which are currently not sufficiently tested or well-documented in the literature. Nevertheless, the performance of program design utilizing these steps is imperative to the success of marine monitoring. Optimum approaches to individual program design steps can be developed only through further study and experience with the design process. It is hoped that this paper will stimulate both managers and scientists to approach future monitoring program design as an important exercise requiring both careful definition of the degree and type of change due to the disposal activity that would be unacceptable, and the application of fundamental, scientific principles to development of a program which can demonstrate whether or not such changes occur.

REFERENCES

Ecological Analysts, Inc., and SEAMOcean, Inc., Technical Information to Support the Redesignation of the 12-Mile Site for the Ocean Disposal of Municipal Sewage Sludge, pp. 1-438 plus Appendices, Submitted by New York City to U.S. Environ. Protection Agency, 1983a.

Ecological Analysts, Inc., and SEAMOcean, Inc., A Special Permit Application for the Disposal of Sewage Sludge from Twelve New York City Water Pollution Control Plants at the 12-Mile Site, pp. 1-594 plus Appendices, Submitted by New York City to U.S. Environ. Protection Agency, 1983b.

EPA, Monitoring Plan for the 106-Mile Ocean Waste Dump Site, Unpublished Document contained in Public Document File Pertaining to Final Designation of Site, U.S. Federal Register, vol 49, pp. 19005-19012, 1983.

Gordon, M., Knauer, G.A., and Martin, J.H., "Mytilus californianus as a Bioindicator of Trace Metal Pollution: Variability and Statistical Considerations", Mar. Pollut. Bull. vol 11, pp. 195-198, 1980.

NOAA, Federal Plan for Ocean Pollution Research Development, and Monitoring, Fiscal Years 1979-83, National Oceanic and Atmospheric Administration, Washington, D.C., August 1979.

O'Connor, T.P., Okubo, A., Champ, M.A., and Park, P.K., "Projected Consequences of Dumping Sewage Sludge at a Deep Ocean Site near New York Bight," Can. J. Fish. Aquatic Sci., vol 40, (suppl 2), pp. 228-241, 1983.

O'Connor, T.P., Walker, H.A., Paul, J.H., and Bierman, V.J., "Monitoring of Contaminant Distributions Resulting from Proposed Sewage Sludge Disposal at the 106-Mile Ocean Disposal Site," Mar. Environ. Res., In Press, 1985.

Phillips, D.J.H., and Segar, D.A., "Use of Bioindicators in Monitoring Conservative Contaminants: Programme Design Imperatives," Submitted to Mar. Pollut. Bull., 1985.

Segar, D.A., An Assessment of Great Lakes and Ocean Pollution Monitoring in the United States, Working Paper No. 7, Federal Plan for Ocean Pollution Research, Development, and Monitoring, FY 1981-1985, National Oceanic and Atmospheric Administration, Boulder, Colorado, 1981.

Segar, D.A., Stamman, E., and Davis, P.G., "Development of a Monitoring Program for the North Atlantic Deepwater Municipal Sludge Dump Site," Proceedings, 5th International Ocean Disposal Symposium, Corvallis, Oregon, In Press, September 1984.

Satsmadjis, J., and Voutsinou-Taliadouri, F., "Mytilus galloprovincialis and Parapenaeus longirostris as Bioindicators of Heavy Metal and Organochlorine Pollution," Mar. Biol., vol 76, pp. 115-124, 1983.

Public/Private Sector Issues

Decisionmaking in Waste Management

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The average individual in the United States discards about 5 pounds of solid wastes every day. That same individual also flushes away his personal waste. This contribution, if treated in a conventional wastewater treatment plant, generates about 0.12 lb, dry weight, of sludge, allowing for normal percentage of removal, aerobic oxidation in an aerator, and anaerobic destruction in a digester.

It would appear that if we could solve the 5 lbs./day problem, we would have no problem to cope with the incremental 0.12 lbs. This logic certainly has not prevailed up to now. The past approach to waste management, including the acceptable disposal of residuals, has been one of intense preoccupation with technical details of individual unit processes, while sidestepping the more intractable problems of structuring total programs inflicting least degradation to the total environment.

This past approach, of course, was inevitable. No city (and ours is an increasingly urban civilization) started "from scratch" to simultaneously cope with water supply, solid wastes, wastewater collection, wastewater treatment, air quality control, receiving water quality control, underground water management protection, and land use, all with optimally minimized risk "trade offs" between air, land, and water. Each aspect of the environment was handled as a separate enterprise, separated by long periods of time. Typically, the provision of a public water supply distribution system proceeded wastewater treatment by a century or more. In the same way, the collection of waterborne wastes and their disposal in contiguous waterways took place about a century before treatment.

The current controversy on the ocean disposal of sludges is really a subset of the total problem of waste management. However, a discussion of ocean disposal sharply illustrates and illuminates the total problem.

I propose to take a brief retrospective of almost 50 years of personal daily involvement with water and waste management to describe the very gradual metamorphosis of decision making from an engineering process driven by economics to a public and political process slowly incorporating an element of credible scientific rationale.

The City of New York, unfairly castigated as the "whipping boy" for all urban problems, can justly take pride in its wastewater programs. The planning started with two formation of a Metropolitan Sewage Commission in 1906, only 8 years after the incorporation of the Greater New York City. The intent was to abate the nuisance conditions and restore some dissolved oxygen to the inner waterways of the harbor then receiving about 500 MGD of raw sewage. Execution of the plan began in 1928, by which time about 1 BGD was being discharged. An annual harbor survey had started in 1913.

The condition of the harbor was also degraded by major raw discharges from New Jersey, which shared the harbor. The various arms of the harbor had widely different water quality . There was no credible way to predict the response of the harbor to a specific degree of treatment to a given volume of sewage at a specific site. Facilities had to be built, however, since some parts of the harbor were outright nuisances. In another area the most densely populated bathing beach in the world, at Coney Island, had to be protected. On a hot summer weekend literally a million people would be packed on a mile of beach. With no outside funding, the City would have to set priorities, design and build, pay as you go, presumably well into the next century.

There was no state or Federal program. With remarkable prescience, the very small in-house team leading this program acquired the sites for future plants along the waterfront. What was possible in the 1930's at reasonable costs would have been impossible, even at exorbitant costs in the 1950's.

How were the decisions made as to degree of treatment? These decisions were made by engineers, with very limited budgets, with no sophisticated scientific input, and certainly were not widely debated in a public forum. With this Edisonian, pragmatic approach, they did reasonably well.

At that time the choices were between an ox-cart and a Cadillac, plain sedimentation and activated sludge, between 30% and 90% B.O.D. removal, between 60% and 90% suspended solids removal.

In the 1930's there was a resurgence of interest in chemical coagulation. The Coney Island Plant came on-line in 1935 as a primary sedimentation plant, but enhanced by chemical coagulation and effluent chlorination in the bathing season. With its outfall about 1 mile from the beach, bathing water quality was reasonably maintained.

The activated sludge process had developed between 1915 and 1920, as the then ultimate state of the art. The Wards Island Plant, in the East River, came on line in 1937. This high degree of treatment was provided in this inner waterway in proximity to the most degraded arm of the harbor, the Harlem River. The outright nuisance conditions were substantially abated.

The great 1939-1940 World's Fair was adjacent to Flushing Bay. Two new plants, Bowery Bay and Tallmans Island, in the proximity of the Fair, were built as activated sludge plants and came on-line in 1939.

After World War II there was a renewed burst of construction, and five more major plants came on-line. However, a new approach began to emerge in an elite small group of in-house civil and sanitary engineers, led first by Richard Gould and then by Wilbur N. Torpey.

They began to challenge the accepted practices in the field, and raised questions, such as:

Is the degree of treatment designed for effluents into fresh water equally applicable to a tidally-mixed esturay?

Are the relatively simple models for BOD satisfaction and oxygen sag really applicable to tidal waters?

Is the same degree of treatment warranted at all times of the year, and at all points in the harbor?

Is disinfection of treated effluents required uniformly? Required the year round?

Can the enormous capital, maintenance, and energy costs be reduced without impairment of the mission?

How can biological processes be effectively and consistently controlled by engineers?

How reliably can we predict the effect of this enormous investment?

Such questions were very pertinent indeed in the 1950's. New York City was committing itself to a billion dollar program for a high degree of treatment on a tidal estuary, when major metropolises along fresh water rivers and lakes were discharging raw sewage or, like St. Louis, planning primary sedimentation.

Most significantly, they were distinguishing between "purpose" and "process". Engineers were comfortable with "process", the steel, concrete, pumps, motors, and piping. The "purpose" was to first protect, and then enhance, the quality of the receiving waters. It is much easier to design a pipe gallery than it is to predict what will happen to marine biota. I cite this as a beginning awareness that scientific knowledge had to be coupled to engineering expertise, although the environmental movement was still a decade in the future.

In any event a ferment of in-house development ensued. One of the products was the development of a series of variations of the activated sludge process to provide a spectrum of energy and capital saving processes, which could be tailored to the needs of the receiving waters. These were quickly applied to existing plants. Other products were the definition of the true rates of biological anaerobic sludge digestion and optimum ratios of food to biological mass in the aeration process. These findings profoundly affected design and operation. However, these were "process" improvements. The lack of a firm basis in scientific rationale was still evident in the "purpose" area. The 220 MGD North River Plant was to be built on the Hudson River. With missionary fervor I challenge the Chief Engineer to change it from a primary sedimentation plant to a "modified aeration" plant, providing at least an intermediate degree of secondary treatment. A distinquished professor, who had developed a first generation mathematical model, criticized me for this before the ASCE, for advocating treatment beyond plain settling at this site.

In effect, those of us either making or attempting to influence decisions on a project, that will now eventually come in at about one billion dollars, and the Federal government, which made the final decision, could all agree on the desired quality to be attained in the Hudson River, and differed by 300% in the BOD removal target to attain that quality. When the Federal government displaced the State and City of New York as the major funding source, they effectively eliminated any consideration of matching water quality goals to cost-effective treatment. By administrative fiat, they demanded the highest degree of treatment attainable by conventional activated sludge. They were practicing the "Golden Rule". They had the gold, and they made the rules.

Several years before the North River debate, I sought permission to attempt to operate the 26th Ward Plant at its design intent of full activated sludge. This plant discharged into the virtually landlocked Jamaica Bay, and I proposed to afford a higher degree of treatment and improved bacteriological disinfection during the summer recreational season. I did so, successfully. There was a dramatic change in water quality in the receiving canal and the adjacent area of the bay. However, the extra operating cost was decried, and this seasonal enhancement of treatment was not attempted again for many years.

Other events in the 1960's and 1970's convinced me that many actions profoundly affecting water quality were being made by men of essentially good intentions who were being steered by the narrow interest of their particular agency, by limited budgets, and by expediency. What they characterized as pragmatic "common sense" often was just a visceral feeling. As the environmental movement gathered momentum, the judgment and credibility of engineers in waste management came under increasing challenge.

For example, 50 years ago city engineers promised to convert noisome, mosquito breeding swampland into verdant vistas of parkland. In the 1970's, as the City's population pushed out to the shoreline, instead of what would now be characterized as precious wetlands, they were faced by a barrier between them and the Bay of a 30 foot high, and still growing, sanitary fill with its inevitable leachate.

In the mid 1960's, fired up with missionary zeal to bring science into the process, I went to Washington, to the FWQA, the predecessor of EPA, with the request to create a credible and verified mathematical model of the harbor that could predict the response of the NY harbor to multiple permutations of inputs, including combined sewage, on even non-conservative parameters such as coliform concentration and dissolved oxygen.

I pointed out we were spending hundreds of millions to build facilities. Why not spend a minute fraction to assess and evaluate? My thesis was that the decisions of the 1960's and 1970's were largely made already, but that the decisions of the succeeding decades would have to be made on a credible scientific basis rather than a "gut" reaction, and that there now were techniques capable of doing this.

Of course they rejected my radical request for a total harbor model, but they did fund such a study for Jamaica Bay, some 15 square miles shallow bay of with only one narrow salt water inlet, and receiving the effluents from 3 major wastewater plants and the combined sewage overflows from about 1 million people. I groveled with gratitude, and mounted a team of marine biologists, physicist, oceanographers, mathematics modelers, and chemists, in addition to sanitary engineers.

Several years before that the Port Authority of New York and New Jersey had proposed to build a solid runway from LaGuardia Airport into Flushing Bay. I reasoned that this solid barrier would impede the tidal exchange in an already vulnerable part of the harbor, and opposed the construction. The Port Authority was incensed. They hired a professor, the same one who had espoused primary treatment on the Hudson River. The dispute demonstrated the sheer power of words. When I raised the specter of a "cul-de-sac of confined septic sewage", the Port Authority built the runway on a relieving platform.

Consider how the situation now changed. The Port Authority now wanted to build a solid runway from John F. Kennedy Airport out into Jamaica Bay to Jo Co Marsh. They stated that if they could not, the airport could not compete and international traffic would be directed to another city. The Transportation Administrator of the City contended that it was an absolute necessity, was environmentally benign, and that no visionary environmentalist should impede beneficial economic development. The issue was so acrimonious that the Mayor of the City stepped aside to let me battle it out with the Port Authority and their proponents.

This time there were facts about the very weak tidal exchange at the head of the bay. This time the model was at hand. This time a select scientific committee from the National Academies of Science and of Engineering was appointed to investigate. This time there was corroboration that the eastern half of Jamaica Bay would become an anaerobic nuisance of this barrier were created.

The solid runway was never built, and 12 years later, the airport is thriving, as is boating and fishing in the Bay.

Shortly thereafter the Corps of Engineers proposed to build a hurricane barrier with a series of tainter gates and narrow channels athwart the one entrance to Jamaica Bay. Their concrete model at Vicksburg purported to show no adverse affect on the Bay's mixing regime and oxygen even while sharply diminishing the tidal exchange with fresh ocean water. Our model showed gross deterioration of bay water quality. There was again a sharp conflict of opinions. I pointed out that the model has been verified by intensive sampling after storms. The model had been further verified in that certain unexpected currents it predicted had been verified in the field and by differential film aerial photography. I noted, with some asperity, that ideas should not be cast in concrete. The net result was that the barrier was not built, and the Corps increasingly and heavily relied on mathematical modeling.

By that time, ten years after Rachel Carson's "Silent Spring", with Love Canal, asbestos, dioxin emission, PCB, and toxic leachate yet to succeed each other as the Menace of the Year, there was already considerable erosion of the credibility of the engineering profession in Waste Management, with the public and their political representatives. There was still no general public acceptance that the ultimate sumps for all waste was the land, water, and air. There was no recognition that every environmental decision was a "trade-off" among these three, and the very phrase of "risk assessment" was abhorrent. At the same time, legislative bodies were attempting to impose "Zero risk" by legislative fiat.

A great opportunity was created by the Clean Waters Act of 1972, and its successive revisions. In its famous "Section 208", it provided funding for a total multidisciplinary overview of wastewater programs before hard commitment to construction. It certainly could have been applied to marine sludge disposal. Unfortunately the times were such that there was urgency on the part of Congress and the States to quickly translate the Act into tangible construction. Section 208 was not effectively used. Indeed, the EPA hierarchy were skeptical of the process. In New York City I fought for and obtained the largest such grant. By then the City program had almost all hardened into design and construction. Nevertheless, I used this grant to at long last create a working model of the inner harbor, which, like the Jamaica Bay model, will influence decisions for the next decades.

Where does sludge disposal fit into this trend? Twenty-five years ago it was apparent to me that ocean sludge disposal, then absolutely ignored by the public and legislators, would eventually become the focus of controversy. When NOAA was created, I began to campaign for my long-deferred hope for a total credibly predictive model for the criticial New York estuary from the end of the salt wedge up the Hudson River to the broad reaches of the New York Bight beyond the sludge disposal areas. Eventually the decision was made for a study, but limited to the inner Bight. As I predicted, the study, ably performed by NOAA over about five years, was heavily steered by the mounting sludge controversy.

In effect the overall scientific study I sought in 1965 of the most heavily populated and polluted estuaryin the Country, the focus of a wastewater construction expenditure of about three billion dollars in 15 years, was being performed belatedly in the 1970's and 1980's in disjointed increments, and after the major expenditures had been made or committed.

One can well speculate that if "study first, build later" had prevailed instead of the converse, EPA might have been persuaded to forego its simplistic "check list" approach and concede that the City Engineers of the 40's and 50's were right in devising different treatments for different local conditions, including seasonal variation in treatment and disinfection.

Before discussing sludge disposal, permit me to state that some engineers and scientists, as well as self-anointed and self-appointed prophets (and prophetesses) of the environment, also have their pet prejudices, preconceptions, and quirks, like to bask in the limelight, and sometimes substitute inflammatory rhetoric for professional objectivity.

The last bargeload of garbage from New York City to be dumped at sea was towed out in 1924. Today there are still scattered references in the press to New York City still dumping garbage at sea. During my tour of duty as Sanitation Commissioner, fifty years after that last load, I still had to make statements on it.

I cite this not for whimsy or frivolity. It is a fact that the simple refutation always limps lamely behind the glittering allegation. It is a fact that public perception, hence political perception, does not have a Pavlovian reaction to clear statements of fact. On the other hand, when I whimsically made some humorous allusions to alligators in the sewers, when I was Commissioner of Water Resources, I was astonished at how literally it was accepted and embellished as it was retold.

This is directly pertinent to the ultimate solutions for sludge disposal. My thesis is simple:

> <u>Scientific rationale, however impeccable, will not yield optimum waste management decisions, unless it is buttressed with effective and credible public communication</u>.

The decision to barge sludge to sea was made by the City of New York in the 1930's when it commissioned the design and construction of three self-propelled sludge vessels, each with a capacity of 35,000 cubic feet. When they started operation in 1937 there was no public concern or interest. The Wards Island Plant had no digesters, so they were hauling some raw sludge to the 12-mile disposal point. There was no special significance to this point on the continental shelf, except that is was 10 nautical miles equidistant between New York and New Jersey, and amenable to monitoring by the Coast Guard. The Coney Island Plant hauled its digested sludge in a small barge for off-pumping to storage lagoons a mile away for synthetic topsoil on what would eventually be parkland. During World War II, during U-boat depredations, dumping was permitted closer to shore. After the war, the wave of new plants all produced digested sludge. This was lagooned for air drying at 26th Ward, Oakwood Beach, Coney Island, and for a time at Jamaica. I WAS then hauled away for surfacing completed landfills, and in fact created the synthetic topsoil for a municipal golf course. Some was pumped on to a small sandy island in Jamaica Bay to encourage vegetation. Several attempts were made for direct spraying on sandy parkland. The potential disposal sites were limited, the cost could not compete with ocean disposal, but land disposal of digested sludge was practiced wherever feasible. Undoubtedly, when some landfills are completed and sand covered, it will be used again. It was never considered for public distribution and use. It had never undergone heat treatment for sterilization. At best, in no year did such land disposal amount to over 15% of the total volume of sludge generated.

There was effective and continuing engineering preoccupation with marine disposal. It was all directed toward increasing dry tonnage, reducing the volume, increasing cargo capacity, increasing crew productivity and vessel reliability, reducing fuel costs, getting two trips per day per vessel, and improving docks to minimize hull damage. Typically, I put on an air mask and went through the cargo holds of a sludge vessel to measure the volume of grit trapped against the bulkheads, to take measures to optimize cargo capacity. It was at this time that NY City led the profession in improvement of gravity sludge thickening. This effort to hold down the unit costs of disposal led to selling off of the older vessels as two 65,000 cubic feet vessels were built in the 1950's, and then two 100,000 cubic feet vessels in the next decade. These last two were designed with pilot house engine control, variable pitch propellers, and a bow thrustor. Two years ago the now upgraded Wards Island Plant was digesting its sludge, so raw sludge is no longer routinely dumped. Incidentally the city, fifty years ago, began to practice sludge digestion, not to manufacture fuel gas, but to avoid odors in storing raw sludge. Of course, the City exploits this gas. At the Newton Creek Plant, for example, over 1 million cubic feet per day is generated for use by the plants diesel engine-generator sets.

In the 1960's we began to cost out alternatives like sludge incineration, but no technology could compete economically with our efforts to hold down unit costs of ocean disposal. Furthermore, there were no reliable techniques then to predict air pollution impacts, even at a time when we didn't worry about mercury of dioxins.

An era came to an end in the late 1960's. Public apathy changed overnight to public apprehension. An image was created in the public press and in purple political rhetoric of a foul black pulsating amoeboid mass in the outer harbor pushing its slimy tentacles ashore. This was literally believed. When abnormal wind and current conditions persisted for a month in the summer of 1976, masses of floatable debris were washed ashore. These had no relevance to sludge disposal, but they were cited by the vociferous critics as proof that the "ooze is upon us".

The issue was perfect for politicians. You could portray yourself as a champion of the environment, which is good, and you could castigate New York City, which is even better. One U.S. Senator reveled in this. One could talk learnedly about acceptable alternates like composting, RDF, purolysis, land application, and incineration, as though each one were economical, environmentally innocuous, and amenable to instant applicability.

Some pseudo science began to mingle with the mythical pseudopods of the great amoeba. One investigator kept a fish in a tank full of sludge and then displayed the creature as typical of the marine life in the Bight. The phrase "Dead Sea" caught on. I took the press and TV crews out to the dump site on our sludge vessel. "Where is the Dead Sea'?, they asked disappointedly. I really believed they expected a scene like that in the

Rime of the Ancient Mariner, "and yea slimy things did crawl with legs upon a slimy sea".

Where was the truth? The NOAA team of quiet scientists keep making their surveys and writing their monographs. Meanwhile the Federal EPA made an incredible error. It ordered cessation of ocean dumping on the assumption that other environmentally acceptable options were readily at hand. A judge knew better. The public does not read scientific monographs. They know there is a Dead Sea out there. NOAA has quantified the impact of sludge disposal, but sees no impending disaster. New York City would like to continue at 12 miles. The Feds would accept 106 miles. There is an intermediate position which recognizes the difference between kinds of sludges, those with varying industrial contaminant burdens, and the possibility of different disposal points. Recent studies have gone into the logistics of combinations of "mother" ships and towed barges. While these parties are milling around for the next episode in this continuing series, let's leave them and consider the mechanism for an ultimate decision.

Our history shows that a well informed public eventually, however glacially, makes reasonable decisions. A key phrase is "well informed." The key thought is that the public makes the decision.

In the JFK runway extension controversy the recommendations of the City engineers, the Transportation Administration specialists, the Port Authority management, and local politicans were all well circulated in the press and on the air, but all were correctly perceived by the public as partisan advocacy of special interests. However, the multidisciplinary committee of the joint National Academies was composed of engineers, oceangraphers, biologists, doctors, and economists, who had no preconceived position, and who took six months to investigate and deliberate. They were perceived as credible.

In the case of residuals disposal and waste management there is indeed reason for the public, whose environmental consciousness has been well heightened, to be skeptical of the assurance of any party, engineer, government agency, utility manager, or entrepreneur, who has a vested interest. The public can cite Three Mile Island, the Bhopal Methyl Isocyanate release, Love Canal, dioxin in Italy and Missouri, nuclear "near misses" in Idaho and Detroit, PCB in fish flesh, asbestos in schools, mercury and cadium poisoning in Japan, dam and bridge failures, toxic contamination of potable water supplies, all to rebut the validity of prior "expert" assurance.

In the case of sludge disposal, it is obvious that the ultimate decision making is no longer the personal "turf" of the cities which generate the sludge or the State and Federal regulatory agencies. In the 21st century, the public will increasingly rely on multidisciplinary and impartial scientific expertise, which, in turn, must be translated into clear language, comprehensible to the general public. Such expertise recognizes that waste management decisions are frequently "trade-offs" between sea, land, and air impacts, and that zero risk may not be attainable. It is interesting to note that scientists are more prone to concede "I don't know" about possible impacts, than are engineers.

By now the consequences of the dwindling landfills have given many communities a forced draft accelerated education on the hard realities of residual disposal options. They know the days of cheap "tipping fees" for garbage are over. Increasingly, communities are more willing to sacrifice some local political control, to participate in a regional plan or regional facility. Sludge disposal is very much a part of this. As communities inexorably move to resource recovery and various techniques of combustion and energy generation combinations, consideration for the incorporation of sludge solids is inevitable. As a better informed public understands the need for objective balancing of <u>all</u> trade-offs, environmental and economical, appropriate applications of ocean disposal will continue to receive consideration.

Essentially, waste management decision making is moving from visceral judgements based on economic stringency and political expediency, to a scientific ratonale approach, but only when coupled with effective public communication. As all residual management alternatives are weighed, ocean sludge disposal will eventually be assessed not by anecdotes and allegations, but by coolly objective scientific evaluation.

Regulatory Innovations for Sound Waste Management

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INTRODUCTION

How can we improve ocean disposal decision making in the future? The decision to dispose public waste in the ocean as well as the subsidiary decisions of how much waste, at what location and for what duration are influenced by--indeed often dictated by--laws and legal process. How well does this legal regime--the substantive and procedural laws-- promote sound ocean dumping decision making? Sound in terms of minimizing environmental disruption, equity, and efficiency. How can we change or supplement the existing legal regime to further engender sound decision making? This paper proposes and evaluates several innovative mechanisms to improve ocean disposal decision making.

Part one examines the existing legal regime pertaining to ocean disposal of waste. What restrictions do the relevant federal statutes and the case law impose on ocean disposal? Part two considers other obstacles to sound decision making such as economic pressures. Part three proposes several reforms to improve the legal regime and evaluates some of the benefits and limitations of these mechanisms.

EXISTING LEGAL REGIME

Prior to proposing legal and institutional changes to improve ocean dumping decision making in the 1990's it is necessary to understand the existing legal regime pertaining to ocean dumping of public waste. For instance, do federal laws as currently written and interpreted by the federal courts allow for continued ocean dumping? Contrary to widespread perception, the answer is yes. This answer derives from a re-examination of the key federal statute controlling ocean discharge of public waste, the Marine Protection Research and Sanctuaries Act of 1972 (hereinafter called the Ocean Dumping Act) as well as the relevant judicial opinions.

The Ocean Dumping Act

The Ocean Dumping Act is the principal law pertaining to ocean dumping in the United States. No dumping of any material is allowed without a permit. The U.S. Army Corps of Engineers is responsible for issuing permits for dredged spoil dumping and the Environmental Protection Agency (hereinafter the EPA) issues permits for all other waste dumping.

Prior to the issuance of a permit the ODA requires the permitting agency to determine "that such dumping will not unreasonably degrade or endanger human health, welfare or amenities or the marine environment, ecological systems or economic potentialities." The Act directs the EPA to establish regulations for making this determination. The EPA is required to consider nine factors in developing and revising the decision criteria. While six of the criteria pertain to effects of dumping on human health and the marine environment two of the criteria relate to the need for ocean disposal _vis-a-vis_ land based alternatives. The final factor pertains to the designation of appropriate dumpsites.

The ODA was amended in 1977 to specially address ocean dumping of municipal sewage sludge. The amendment was short but not definitive. EPA was directed to "end the dumping of sewage sludge. . . as soon as possible. . . but in no case may (EPA) issue any permit. . . which authorizes any dumping after

December 31, 1981." The amendment then defined sewage sludge as material which would "unreasonably degrade or endanger human health, welfare, amenities, or the marine environment, ecological systems or economic potentialities." Thus, despite widespread perception, the 1977 amendment did not absolutely prohibit sludge dumping after 1981. Rather, only sludge which the EPA found to have violated its regulations established pursuant to the nine statutory factors was absolutely prohibited.

The observation that the ODA imposes no absolute prohibitions against dumping of public wastes is further supported by the Act's treatment of warfare agents and high-level radioactive waste. The Act expressly prevents EPA from issuing any permit for ocean dumping of warfare agents or high level radioactive waste. In otherwords, when Congress desires to ban ocean dumping of a particular waste it does so expressly.

London Dumping Convention

In 1972 the United States and twenty-six other countries signed the International Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter (hereinafter the London Dumping Convention). In 1974 Congress amended the Ocean Dumping Act to include the provisions of the LDC not reflected in the ODA. Thus the LDC must be read as part and parcel of the ODA. The LDC requires its signatories to establish national programs to control ocean dumping. Like the ODA, the LDC imposes few absolute prohibitions. The LDC "prohibits" the dumping of certain substances listed in Annex I such as cadmium and mercury. Even Annex I substances can be dumped, however, if they are either rapidly rendered harmless by ocean processes or are present only as "trace contaminants."

Judicial Opinions

There have been a number of lawsuits involving the implementation and interpretation of the ODA. While the legal claims in each of these suits differed, there is a fundemental similarity among the outcomes of these cases. The courts in each of the six recent cases involving the ODA were unwilling to foreclose ocean dumping as a waste disposal option.

The first federal court decision principally involving the ODA was in 1980 and is known as National Wildlife Federation v. Costle. (U.S. Circuit Court, 1980). The National Wildlife Federation (hereinafter NWF) sued EPA in the late 1970s alleging that EPA's ocean dumping regulations violated the LDC and the ODA. NWF asserted that EPA violated the LDC because the agency did not use the LDC criteria in case-by-case application review. NWF further asserted that both the ODA and the LDC required site designation studies prior to the use of a dumpsite; EPA's interim site designation based on historical use, argued NWF, was therefore illegal. The Federal Circuit Court disagreed with NWF on both counts.

In 1981 a Federal District Court in New York issued a controversial opinion in the lawsuit City of New York v. EPA (U.S. District Court, 1981). In 1980 EPA denied New York City's application to dump sewage sludge at the 12 mile site, which it had been doing for decades. EPA's decision was based on its ocean dumping regulations which established a conclusive presumption that a substance which violated certain numerical standards would unreasonably degrade the marine environment. New York City sued EPA alleging that EPA's regulations violated the ODA because they precluded the consideration of the need to ocean dump, a factor specified in the statute. The court agreed with the City and concluded that EPA's regulations were arbitrary and capricious because they restricted ocean dumping beyond what was intended by the ODA. The court directed EPA to reconsider New York's application and develop new ocean dumping regulations. (Lahey, 1981)

In 1981 the United States Supreme Court rendered an opinion Middlesex County v. National Sea Clammers (U.S. Supreme Court, 1981), involving the ODA. Shellfishermen from New York and New Jersey sought monetary compensation for damage to their fishery from ocean dumping at the 12 mile site. The fisherman argued that the ODA gave them a right to sue for private damages caused by ocean dumping. The Supreme Court disagreed with the fishermen by finding that ocean dumpers were not liable for private damages via the ODA. The high court chose not to impose a new restriction to ocean dumping by refusing to expose ocean dumpers to financial liability.

In 1983 a Federal District Court prevented the State of Delaware from joining a lawsuit against the municipalities, other than New York City, which were dumping at the 12 mile site. In NWF v. Ruckeshaus (U.S. District Court, 1983) Deleware sought to join the National Wildlife Federation lawsuit in which the environmental organization challenged dumping by New Jersey and New York municipalities after the December 31, 1981 deadline. The Court concluded that Delaware's interest in the lawsuit was insufficient to allow the state to participate in the proceedings.

Finally, in 1984 a Federal Circuit Court upheld EPA's agreement with municipal dumpers, aside from New York City, which allowed dumping to continue at the 12 mile site subsequent to the December 31, 1981 deadline. In a case called National Wildlife Federation v. Gorsuch, the National Wildlife Federation sued EPA seeking to prevent the agency from allowing any further dumping beyond the statutory deadline. (U.S. Circuit Court, 1984) The environmental group argued that EPA's authorization subsequent to the end of 1981 violated the ODA and the London Dumping Convention. The Federal Circuit Court disagreed and upheld EPA's continued authorization of dumping. The appellant Court relied extensively on the opinion in City of New York v. EPA.

Federal statues and international agreements pose few absolute bars to ocean dumping in the future. Moreover, federal courts have generally been reluctant to interprete these laws in a way which would foreclose or even restrict this waste disposal option. Notwithstanding statutory amendments ocean dumping will continue to be a legal disposal method. The task therefore becomes one of how to make ocean disposal decision making safe and efficient.

OBSTACLES TO SOUND DECISIONMAKING

If ocean dumping is permissible in ten years, which appears likely given the current legal regime, what other barriers or obstacles might there be to making sound ocean dumping decisions? It seems fair to assume that the factors which currently distort decision making or render the process inefficient will continue to be present ten years hence unless steps are taken to reform the process. What follows is a quick summary of some of the forces which can be obstacles to sound decision making.

Public Opposition

Public opposition to waste disposal proposals has become almost commonplace. Local opposition to waste disposal facilities have delayed or prevented the siting of many waste disposal facilities. There are two types of public opposition which affect ocean dumping decision making.

First, local opposition to land based disposal options can result in increased use of the ocean because it is often the path of least political resistance. For instance, Nassau County on Long Island constructed a 14 million dollar state-of-the-art composting facility to solve their sewage sludge disposal problem. County officials, however, were dissuaded from ever operating it because of intense local opposition to spreading the resulting compost on land. Nassau County now ocean dumps its sludge instead.

National Wildlife Federation Attorney Ken Kamlet views the problem as one of disenfrachisement. Since fish do not vote, notes Kamlet, there is an inadequate political constituency to protect the oceans from overuse from ocean dumping. (Kamlet, 1981)

On the other hand, ocean dumping can inspire intense and widespread public opposition. This opposition is capable of delaying or preventing this disposal options. One illustration of this public opposition is from Texas. In 1983 EPA tentatively decided to grant permits to incinerate chemical wastes in the Gulf of Mexico. Over six thousand people attended the public hearing in Brownsville, Texas -- the largest public hearing ever held by EPA -- to express their opposition to EPA's decision. After the hearing the Director of the ocean incineration program at EPA, Steven Schatzow, was quoted as saying: "the definition of a backyard has gotten much larger." (Keller, 1984). Subsequently EPA withdrew its tentative approval for ocean incineration in the Gulf of Mexico.

Do these competing forces neutralize each other thus allowing ocean dumping decision making to proceed on the merits? Not likely. In most cases either the anti-land disposal or the anti-ocean disposal voices will be louder; rarely will there be a coincidential equalization of opposition. Even if there were, the tactics often employed by either side divert attention from the merits of the issue.

Economic Obstacles

Although ocean dumping has long been more expensive than land based alternatives several factors have caused marked increases in the cost differential. The dwindling availability of suitable disposal sites close to urban centers has made landfilling more expensive. Expanding local, state and federal regulations to protect groundwater and air quality restrict the availability of land based disposal sites and result in increased compliance expenditures. Rising land costs as well as public opposition from local residents, moreover, are forcing many municipalities to transport their wastes to isolated sites further and further away. This too results in major cost increases. Ocean disposal for Orange County, California, for instance, is approximately one-fourth the cost of land disposal options (Lahey and Connor, 1983).

At risk of stating the obvious, the larger the disparity between land disposal and ocean disposal costs, the more pressure there will be to ocean dump. This increased presure to ocean dump based on economics is likly to cause overuse of the oceans at particular sites.

Inadequate Sites Studies

Another impediment to sound ocean dumping decision making is the dearthof site designation studies. Areas of the ocean differ dramatically in terms of their physical, biological and chemical characterists. (Vaccaro, et al., 1981). Areas of the ocean with deep water and dynamic currents have a much greater capacity to dilute wastes. EPA, however, has tended to designate ocean dump sites based on historical usage rather than the characteristics of the marine environment. Of the some 150 dumpsites around the country only a handful have been thoroughly studied and formally designated. The high cost of these studies, estimated at a half a million dollars a site, is one reason for the minimal progress (Lahey, 1984). Recent and proposed federal cutbacks for marine science and monitoring programs cast doubt on future expansions of site designation studies.

MECHANISMS FOR IMPROVED DECISION MAKING

There are a number of legal and institutional mechanisms which could improve decision making pertaining to ocean disposal. Some of these reforms, such as fee systems, have been in use in other contexts for many years. Three proposals are offered to improve the ocean dumping regulatory program.

Fees

One way to improve the ocean dumping regulatory program is to institute a sliding fee based on the amount dumped, the type of contaminants in the waste and the location of disposal. Such a system would have beneficial regulatory and revenue-generating effects.

Controlling pollution via economic charges is hardly a novel idea. It has been discussed, under its may aliases, in the economic literature for over two decades. Environmental levies have been extensively used by European Countries. (Organization for

Economic Cooperation and Development, 1980). Moreover, a number of states have recently adopted environmental regulatory programs which rely on fee systems (Lahey, 1984). This theoretical assessment and real-world experience provide a solid foundation for the development of a national ocean dumping charge system.

The fee system most suited to ocean dumping is one designed to accomplish a dual objective: revenue generation and pollution reduction. France has designed a complex levy system for waste discharge which exemplifies a dual purpose charge. (Harrison and Sewell, 1980). The fee serves a revenue generating function by channelling funds to water basin commissions and to water improvement projects. Indeed, the fee levels are set to cover the cost of regional water quality management projects. The fee varies according to factors such such as the efficacy of the water pollution control technologies used and the conditions of the receiving waters. Discharges into waters with high dispersive characteristics, for instance, are charged less than discharges into shellfishing areas. These variable fees, therefore, encourage water pollution control.

As Supreme Court Justice Brandeis noted, states serve as important laboratories for legislative experiments. This point is illustrated in the context of dual purpose pollution charges. To help control hazardous wastes a number of states have passed statutes to generate funds to finance waste cleanup as well as to encourage environmentally sound disposal practices. (National Conference of State Legislatures, 1982).

The ocean, especially beyond the territorial sea, is the paradigmatic common property resource. Efficient allocation in the normal market sense is thus impossible. One manifestation of inefficient allocation is overuse of certain dumpsites, exceeding the ocean's assimilative capacity. An ocean dumping fee calculated in light of alternative waste disposal costs could reduce the volume of wastes dumped into the ocean to a level within the ocean's assimilative capacity.

A charge system would also counterbalance the growing political and economic pressure to dispose of wastes into the ocean. As mentioned, the ocean, unlike our backyards and local groundwater supplies, does not have strong political constituencies. Thus, pressure mounts to dispose of our wastes in the ocean because it is the path of least political resistance, not because of scientific or technical considerations. Further, oceans are also the path or least economic resistance. The failure of the market exchange system to attach a price for the use of oceans makes dumping an economically attractive disposal option. In economic terms, waste producers use the ocean because the internal costs of ocean dumping are much less than costs of alternative disposal methods. A charge system could bring ocean waste disposal into political and economic parity with other disposal methods. Consequently, waste disposal decisions would likely be based on environmental and public health risks, rather than political or economic expediency.

Two recent decisions by the EPA will have a similar effect of minimizing the cost differential between ocean dumping and land-based disposal. EPA's decision to close the 12-mile dumpsite and require sewage sludge dumpers to dump at the 106-mile site will substantially increase the cost of ocean dumping. Also, EPA's 1985 draft ocean incineration regulations propose significant cost increases for ocean incineration in the form of high liability insurance requirements. (EPA, 1985). EPA is considering requiring between 50 and 500 million dollars worth of liability insurance coverage as a condition for an ocean incineration license. This would impose sizable insurance premiums on ocean incinerators.

A fee would also create an incentive to develop alternative waste disposal techniques such as waste reduction and recycling. Dumpers would consider disposal alternatives which may become cheaper as the technique is improved. Recycling, for instance, could become less expensive or even profitable as the technique is refined.

A charge system could be designed to reduce the amount of certain harmful substances dumped in the ocean. The fee could vary according to the types and concentrations of contaminants in the waste. Thus, innocuous materials, such as cannery wastes, would be ascribed low dumping fees, if any, since uncontaminated organic materials pose few threats to the marine environment. Graduating the fee based on contaminant concentrations would create an incentive for dumpers to either reduce the volume of waste dumped or implement methods to reduce the contaminant concentrations. The variable fee would encourage the reduction of contaminants in sewage sludge through increased reliance on pretreatment.

Another regulatory end that a variable fee system could encourage is the use of appropriate ocean dumpsites. Dumpers using sites having heightened capacities to assimilate waste would be charged less than those dumping at sites with limited pollution tolerance. A variable fee structure could, moreover, help reverse the existing incentive to dump at less appropriate sites. In many cases the sites with heightened assimilative capacity are further from shore than less dispersive sites. Because of transportation costs, dumpers prefer those sites closer to shore. Tailoring a variable fee system to reduce dumping in overused sites could therefore reduce the net environmental effects of ocean dumping without curtailing the volume of materials dumped.

Revenue generated by an ocean dumping fee could be used for a number of important functions. First, the revenue could help finance ocean monitoring programs. Since our knowledge of the marine environment is still rudimentary, any effective system of ocean waste disposal must be accompanied by an ongoing monitoring program. The problem, however, is that while dumping is increasing, no corresponding increases are expected in monitoring efforts. In fact, cutbacks in federal funding will significantly reduce the amount of scientific work conducted in this area. Revenue generated by an ocean dumping user fee, therefore, could present a vital source of funding for these monitoring programs.

Second, revenue from dumping fees could help finance site designation studies. As discussed earlier, the deleterious effects of ocean dumping could be reduced merely by dumping at sites with appropriate characteristics. EPA has failed to make significant progress in conducting the studies necessary todetermine appropriate dumpsites. One of the major stumbling blocks has been the cost of the site designation studies. In an era of declining federal expenditures for environmental programs, it seems unlikely that adequate federal funding will be available for these studies. On the other hand, it is equitable to require those who make the studies necessary in the first place -- the dumpers -- to incur the cost.

Fee revenues could also be used to fund research and development of alternative waste disposal techniques. Many wastes which are currently dumped into the ocean could, through the application of certain scientific or technical processes, be transformed into valuable resources. Municipal sewage sludge, for instance, can be a safe and useful fertilizer when properly managed. Many alternative uses of sludge have been developed as a result of federal research money. Significant engineering advances in sludge processing technology, moreover, have occurred recently. These two factors suggest that cutbacks in federal research and development money could retard the implementation of safe and useful alternatives to disposal of wastes by ocean dumping. Nonetheless, cutbacks in federal funding of research and development have occurred recently and can be expected to continue in the future. Furthermore, scientific research on alternative disposal methods would help mitigate public opposition to certain land based disposal techniques.

Alternative Dispute Resolution

Another type of mechanism which could improve ocean dumping decision making is alternative dispute resolution. Alternative dispute resolution is a generic term used to describe a range of non- adversarial approaches to conflicts. These consensual approaches have proven to be very effective in resolving complex environmental disputes. (Talbot, 1983) These approaches are especially well-suited to multi-party, multi-issue disputes such as ocean dumping (Susskind & McCreary, 1985).

There are a number of potential benefits from using non-adversarialapproaches to environmental disputes. First, lengthy and divisive court battles can often be avoided thereby saving time and resources. Second, the outcomes are more likely to secure joint gains among all the parties rather than merely selecting winners and losers (Fisher and Ury, 1981). Third, interests groups are less likely to shift the dispute to another forum because they are given a stake in the process and the outcome. The National Wildlife Federation, for instance, sought to restrict ocean dumping by seeking redress from three separate courts; this could be avoided if NWF participated in a negotitated process which they felt was legitimate. Fourth, a consensual approach utilizes all parties' energies and ideas to develop solutions to complex problems. This increases the likelihood of a creative and effective final solution.

There are a variety of approaches to dispute resolution which can be effective alternatives to traditional judicial approaches. Three of these approaches which have been employed in the context of marine disputes are: unassisted negotiation, facilitated dialogue and mediation (Susskind and McCreary, 1985). Unassisted negotiation can be used when the parties are willing and able to meet face-to-face to work toward joint solutions. Facilitated dialogue involves nonpartisan facilitators who structure meetings or manage conversations for disputing parties. This role often involves identification of the interested parties and focusing the issues. Mediators are often brought in when the conflict has reached an impasse. Their role is to establish a dialogue and steer the parties toward mutual gain, often serving as a go-between.

Two examples of where alternative dispute methods could be used in the ocean dumping context are: designating new dumpsites and developing the permit terms for dumping at the 106-mile site. The designation of new dumpsites has frequently been a controversial decision. The extreme illustration of this type of dispute, as already mentioned, is the controversy surrounding EPA's tentative decision to allow ocean incineration in the Gulf of Mexico. While less newsworthy, the Army Corps of Engineers is also running into a lot of public opposition and controversy involving the designation of dredge spoil dump sites.

Designation of an ocean dumpsite usually involves a number of interest groups all of which could oppose by political or legal means and hence delay or derail a designation decision they disagreed with. It could be efficient, therefore, to bring these interested parties together and have them negotiate face-to-face. These negotiating sessions would focus on issues such as the geographic parameters of the site, the level and duration of the environmental monitoring and the type of waste that can be dumped at the site.

Dumping large volumes of municipal sludge at the 106-mile site will be controversial. It is very possible that some party will use legal means to limit or terminate this dumping.

For instance, the State of Delaware has already attempted legal action to prevent the use of the 106-mile site.

Negotiation among interested parties over the terms of EPA's permits to dump at the deepwater site could serve to avoid lengthy judicial procedures and could result in better permits. A mediator could help representatives of the interested parties reach cooperative solutions on permit issues such the amount of waste allowed to be dumped, the toxic limits, barge monitoring and the amount of the dumping fee to be charged.

Tradeable Permits

Tradeable permits is another regulatory reform which could improve ocean dumping decision making in the future. Marketable emission permits have been increasingly used by EPA since 1977 to control air pollution. (de Calvoy Gonzalez, 1981) Tradeable permits may be an effective way to allocate dumping capacity at ocean dump sites.

Two example of the use of tradeable air emission permits are offsets and bubbles. (Hahn and Noll, 1983) Offsets allow the permitee to sell his emission right to another company in the same region. EPA uses the bubble concept to allow an owner of a plant to increase emissions from one source if there is a corresponding decrease in emissions from another source at that plant. Creating a market for a regulatory permit is not new, most cities regulate the number of taxis with a medallion system. A fixed number of medallions, licenses if you will, are issued and these are salable.

There are a number of benefits to allocating fungible regulatory licenses in this way. First, it decentralizes decisionmaking. (Schelling, 1983) The regulated parties which need the permits the most will pay a higher price. EPA, for instance, would not have to get into the business of evaluating a city's need to ocean dump. The market system would identify which municipality has the least available disposal alternatives. Second, entry into and exit from the regulatory process could be much faster than with the traditional licensing process. Third, this system offers more predictability and reliability for regulated parties. This is important, for instance, for municipalities who need to plan their long term sludge disposal strategy. While the cost of a permit to dump a certain amount of sludge at the 106-mile site will fluctuate according to demand, cities can deal with this uncertainty much easier than the questions associated with the standard regualtory process.

Under this approach EPA would calculate the dumping capacity of each ocean dump site. In other words, calculate the assimilative capacity of the site. (National Oceanic and Atmospheric Administration, 1979) once EPA calculated the amount of annual pollutant loading they would permit at a particular site they could auction off authorizations to dump certain amounts of substances at that site for a fixed period. Any municipality desiring to dump waste at that site would be eligible to bid on the permits to dump. They could use these permits or sell them after a certain amount of time to another municipality. This dumpsite allocation system is more equitable and efficient than the existing process. The money generated from the auction, moreover, could be used for monitoring studies or other useful activities.

CONCLUSION

The existing legal regime poses few absolute prohibitions against future ocean dumping. The ocean dumping regulatory framework, however, fails to promote optimal ocean disposal decisions. Unmitigated economic and political pressures as well as inadequate scientific information has, and will continue to, thwart sound regulatory decisions. There are a number of ways the ocean dumping regulatoru program could be reformed to overcome these obstacles. Ocean dumping fees could acheive a number of important regulatory and revenue goals. Alternative methods of dispute resolution could reduce political opposition and encourage creative and more stable solutions. Finally, tradeable permits are a viable mechanism for efficiently allocating capacity at ocean dump sites.

REFERENCES

de Calvoy Gonzalez, "Markets in Air: Problems and Prospects of Controlled Trading," Harvard Environmental Law Review, vol. 5, pp. 377, 1981.

Environmental Protection Agency, "Proposed Ocean Incineration Regulations," Federal Register, vol. 50, pp. 8222-8288, February 28, 1985.

Fisher, R. and Ury W., Getting to Yes: Negotiating Without Giving In, Houghton Mifflin: Boston, 1981.

Hahn, R. and Noll, R., "Barriers to Implementing Tradeable Air Pollution Permits," Yale Journal on Regulation, vol. 1, pp. 63-91, 1983.

Harrison and Sewell, "Water Pollution Control: The French System of Contracts," Natural Resources Journal, vol. 20, 1980.

Kamlet, K.S., "The Oceans as Waste Space: A rebuttal," Oceanus, vol. 24, 1981.

Keller, B., "Decision on Incinerating Waste at Sea Postponed by EPA," New York Times, pp. A16, April 24, 1984.

Lahey, W.L., "Ocean Dumping of Sewage Sludge: The Tide Turns From Protection to Management," Harvard Environmental Law Review, vol. 6, pp. 395-431, 1982.

Lahey, W. and Connor, M., "The Case for Ocean Waste Disposal," Technology Review, vol. 86, pp. 61-68, 1983.

Lahey, W.L., "Economic Charges for Environmental Protection: Ocean Dumping Fees," Ecology Law Quarterly, vol. 11, pp. 305-342, 1984.

National Conference of State Legislatures, Hazardous Waste Management: A Survey of State Legislation, Denver, CO. 1982.

National Oceanic and Atmospheric Administration, Assimilative Capacity of the U.S. Coastal Waters for Pollutants. 1979.

Organization for Economic Cooperation and Development, Pollution Charges in Practice, 1980.

Schelling, T., "Prices as Regulatory Instruments," in Incentives for Environmental Proection, ed T. Schelling, pp. 1-40, MIT Press, Cambridge, MA, 1983.

Susskind, L. and McCreary, S., "Techniques for Resolving Coastal Resource Management Disputes," Paper presented at MIT Sea Grant Conference. 1984.

Talbot, A., Settling Things: Six Case Studies in Environmental Mediation, Conservation Foundation Press: Washington, D.C., 1983.

U.S. Circuit Court, National Wildlife Federation v. Costle, v. Environmental Reporter, vol. 14, pp. 1680-1695, 1980.

U.S. Circuti Court, National Wildlife Federation v. Gorsuch Environmental Reporter, vol. 21, pp. 1769-1775, 1984.

U.S. District Court, City of New York v. EPA, Federal Supplement, vol. 543, pp. 1084. 1981.

U.S. District Court, National Wildlife Federation v. Ruckelshaus, v. Environmental Reporter, vol. 21, pp. 1776, 1984.

U.S. Supreme Court, Middlesex County v. National Sea Clammers v. Environmental Reporter, vol.16, pp. 1118-1132, 1982.

Vaccaro, et al., "The Oceans and U.S. sewage Sludge Disposal Strategy," Oceanus, Spring, 1981.

Institutional Barriers to Technological Innovation in Municipal Wastewater and Sludge Management Practices

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ABSTRACT

Although researchers can repeatedly demonstrate the technological feasibility of innovative methods for managing municipal wastewater and sludge, it is often difficult to gain an acceptance of such innovations so that they can be used by operating facilities. Public opposition and a variety of institutional barriers in the private and public sectors often lead to lengthy delays in undertaking technological advancements in wastewater and sludge management process technologies and in reuse and disposal practices. This paper will review some of the institutional barriers that have been faced in regard to innovative practices of managing wastewater and sludge on land and at sea, and will offer suggestions on what might be done to help ease the burden of gaining acceptance of technological innovations in the future.

INTRODUCTION

Technological innovations in municipal wastewater and sludge management are often difficult "to sell" and are frequently questioned, not only by the general public but also by government policymakers and the scientific community. This has been true not only of reuse of wastewater in agriculture, ocean disposal of sludge, and other practices which were anticipated to generate some controversy, but of almost any change in common practice -- the use of new analytic and modeling techniques for tracing the actual fate of pollutants released into the environment, or the demonstration of improved treatment efficiency and cost-effectiveness of new equipment or management techniques. The reluctance to accept the results of technological innovation directly influences the numerous political, regulatory, and financial policy hurdles that wastewater and sludge management practices must clear to be declared desirable or acceptable. The skepticism and at times even violent opposition expressed toward proposed changes in the status quo of current practices and policies include legitimate concerns over protecting public health, the environment and the taxpayer's pocketbook as well as special interests based on personal preferences, past experience, or anticipation of financial, political and professional losses. Frequently even the most convincing display of scientific evidence about the safety and cost-effectiveness of a technological innovation will not be adequate to overcome the many institutional barriers facing its acceptance by scientists, policymakers and the general public.

Legislative Background

Concerns over environmental problems led Congress to pass many laws which gave the U.S. Environmental Protection Agency (EPA) responsibilities for dealing with a wide variety of waste management problems, including the control of municipal wastewater treatment and sludge management facilities and practices (see Appendix). These laws required EPA to identify and control the potential public health and environmental problems associated with land-based and ocean disposal practices, but also to encourage the beneficial reuse of municipal wastewater and sludge. As a result, in developing guidelines and regulations for waste management practices, the Agency has actively encouraged the maximum recycling of wastes as a resource within acceptable levels of risk.

The Federal Water Pollution Control Act Amendments of 1972 (P.L. 92-500) authorized a major Federal funding program to help abate water pollution from municipal wastewater treatment facilities. The language of P.L. 92-500 sent a clear message that use of the Federal grant funds authorized by this law should encourage both cost-effective and alternative technologies in municipal wastewater treatment and sludge management. In reviewing the progress made toward increased use of alternative technologies, Congress chose in the Clean Water Act of 1977 (CWA) to define and authorize a specific ''innovative and alternative (I/A) technology program'' as a part of the overall Federal construction grants program. This law spelled out and strengthened the Congressional mandate that Federal funds should encourage use of innovative and alternative technologies to conserve and reuse resources. It called for I/A set-aside reserves in Federal funds for wastewater treatment construction grants; a ten percent bonus grant for projects which met certain criteria; and authority to use Federal funds to correct failures, in an effort to encourage use of relatively unproven or unfamiliar technologies. The 1981 amendments to the CWA strengthened the statutory mandate to encourage use of innovative and alternative technologies by including provisions for field testing of innovative technologies. In effect, since 1972 there has been a consistent statutory trend to direct Federal funds to the implementation of innovations and comparatively unknown alternatives in wastewater treatment and sludge management programs.

Status of EPA's Construction Grants Program

As a result of its overall commitment to address water pollution, during the past 12 years Congress has appropriated over $40 billion toward the construction of municipal wastewater treatment and sludge management facilities. Over the years EPA and its predecessor agencies also have conducted a major wastewater treatment and sludge management research and development program, which since 1972 has been funded at as much as $9-10 million per year. The major goal of these programs has been to solve priority water quality and public health problems through the construction of cost-effective, energy-efficient, and environmentally-sound wastewater treatment and sludge management facilities.

A major accomplishment of these programs has been to provide improved municipal wastewater treatment to an increasing number of people, while noticeably improving water quality nationwide (ASIWPCA, 1984). Between 1972 and 1984, the population served by secondary treatment increased by over 57 million to a total of 142 million, while the number of people requiring but not receiving public sewage collection and treatment dropped from 21 million to 14 million. Another result of these and other efforts to improve municipal wastewater treatment has been a doubling in the production of sewage sludge, which is currently estimated to be about 7 million dry tons per year -- an amount which is expected to nearly double again by the year 2000, as wastewater volumes and the degree of treatment provided increase.

The response to the increasingly stronger Congressional mandates to direct Federal funds to encourage greater innovation at the local level has resulted in over 2,900 grant awards for I/A technolgies from inception of the program on October 1, 1978 through March of 1984. There is every indication that the national response to the program will encourage Congress to continue strong legislative support when it considers further authorization of the program (EPA, 1984). However, the General Accounting Office (GAO) has been critical of what had been approved as ''innovative'' by EPA. The GAO reported the program has ''had limited success,'' because not enough incentives are provided for consulting engineers and states to take the risk or incur the additional cost of developing innovative projects (GAO, 1984).

BASIS FOR INSTITUTIONAL BARRIERS FACING INNOVATION IN MUNICIPAL WASTEWATER TREATMENT AND SLUDGE MANAGEMENT

Personal perceptions and attitudes as well as formal rules dictate the acceptability of wastewater treatment and sludge management practices. Social groupings, or institutions, which control or reflect individual actions and attitudes include families, neighbors, businesses, trade and labor organizations, governmental bureaucracies, and professional groups. Some of the concerns voiced by individuals and groups over wastewater and sludge management practices are legitimate and concrete -- potential public health and environmental risks; nuisances; questions about performance, reliability, or cost-effectiveness; and competing uses for disposal sites. Other factors such as historic precedents, cultural influences, and individual special interests or personal preferences can greatly influence personal perceptions and attitudes.

Less concrete, and therefore more difficult to overcome, is the general negative psychological reaction by many individuals to waste management problems, or ''filth'' as they perceive it. This generalized avoidance has contributed greatly to the ''out-of-sight/out-of-mind'', ''not-in-my-backyard'' attitudes toward waste disposal which have developed during recent years in this country. Further complicating this situation is the tendency of some individuals to lump all wastes into general categories such as ''hazardous'' or ''toxic'' and to assume that the problems associated with clearly dangerous materials and disposal practices apply to all.

The influence of personal attitudes (and the factors which influence them) in gaining acceptance of technological innovation, especially in municipal wastewater treatment and sludge management, should not be underestimated. Such attitudes play an important (although often ''unofficial'') role not only in gaining public acceptance, but also in establishing the policies of political, regulatory, financial and research institutions -- often overriding what some may consider to be clear-cut scientific findings about the performance, safety and cost-effectiveness of a technological innovation.

RESULTS OF EVALUATIONS OF BARRIERS TO INNOVATION IN WASTEWATER TREATMENT/SLUDGE MANAGEMENT

Over the years there have been a number of experiments and studies aimed at evaluating the acceptance of innovative technology by State and local governments. These have included an Urban Technology System project in the late 1970s, conducted by Public Technology Inc. and supported by the National Science Foundation (NSF) and the National Aeronautics and Space Administration (NASA). In this study on-site contractor staff were placed in state and local government organizations to monitor and encourage the use of selected technologies by the participating organizations. An NSF-supported study by Syracuse Research Corporation included coverage of developing innovative wastewater treatment practices (Wileman, 1979; Mercer and Phillips, 1981). Some of the key findings of these experiments and studies and of a study conducted for EPA by Booz-Allen & Hamilton in 1976 on barriers to innovation in wastewater treatment suggest the following:

o Implementation of new technology proceeds much faster and most effectively when user involvement in the technology development/selection process has been high.

o Suppliers of State and local technology generally perceive a market that is not interested in technological innovation, and they perceive a market that is highly disaggregated.

o The political and institutional environment in State and local government is highly adverse to accepting risk (e.g., risk of environmental damage, risky use of public funds), although highly interested in acquiring scientific and technological expertise.

o Often the multiple objectives of Federal programs work at cross purposes (e.g., emphasis on enforcement of wastewater treatment and sludge disposal requirements on the one hand and encouragement of experimentation and innovation on the other).

o As compared to other industry sectors, the municipal wastewater treatment/sludge management industry is characterized by relatively high competition and low profit. Equipment suppliers face barriers to new product development which include:
- difficulties in obtaining patent rights;
- lack of information on market potential for new technology;
- procurement hurdles such as bid shopping and non-restrictive clause provisions;
- high costs of appropriately-sized demonstration projects.

o The risks associated with innovation are generally viewed as outweighing the benefits by nearly all the key ''actors'' in the wastewater treatment and sludge management decisionmaking process -- consulting engineers, local government officials, and State/Federal regulators.

o Consulting engineers must face a number of problems and realities when considering innovative solutions to wastewater treatment and sludge management problems that will provide municipalities with workable, reliable systems, including:
- adequate data for the evaluation of innovative technologies are frequently lacking;
- project deadlines often preclude taking the time that would be necessary to adequately check out (both economically and technically) an innovative technology before recommending it;
- low operator skill levels available to a community often preclude serious consideration of ''high tech'' innovative technologies.

o Local municipal officials generally are discouraged by the concerns and unknowns typically associated with the operation and maintenance costs and performance of innovative technologies.

o State/Federal regulators generally are discouraged by the lack of detailed performance data and experience with innovative technologies.

More recently, the findings of a 1983 workshop conducted for EPA by the American Public Works Association's Research Foundation noted similar constraints on more active private sector involvement in funding the development of new products in the environmental control area (APWA, 1984). The constraints included concerns over market size and limited profit potential, uncertainty about future governmental regulations, lack of clear-cut performance requirements, and the need for short payback periods.

<u>Lessons Learned Concerning Institutional Barriers to Land Application of Municipal Wastewater and Sludge</u>

Over the past decade, considerable attention has been given to land application practices for treating and recycling or disposing of municipal wastewater and sludge. These practices cover a wide range of alternatives -- e.g., application of treated wastes to farmland and forests, their use in land reclamation efforts, the distribution of composted or otherwise processed sludge products for use as a soil amendment or organic-based fertilizer, and the operation of dedicated surface disposal facilities which do not involve recycling. The experience gained from addressing the institutional barriers that face these alternatives may provide some insights into dealing with the institutional barriers facing further technological innovation in ocean dispoal of municipal wastes or in other waste management practices.

The concept of land application of municipal wastewater and sludge is certainly not new; the use of human wastes to fertilize the land dates back many centuries in Europe and Asia, and was used extensively by Americans in the 19th century (Jewell and Seabrook, 1979; Tarr, 1981) . However, the recycling of treated human wastes, although actively encouraged by EPA as mandated by Congress, is not always readily accepted by the general public, local treatment authorities, regulatory officials, politicians, scientists and researchers. Although some land application projects have proceeded virtually unopposed, most have generated some level of controversy at least at the local level (Deese et al., 1981).

In response to the mandates by Congress to encourage recycling of wastes, EPA and other Federal agencies (e.g., U.S. Department of Agriculture, U.S. Department of the Interior, Army Corps of Engineers) funded extensive research and demonstration projects in order to obtain better information on potential public health and environmental effects and to learn how to avoid or at least mitigate such problems. Early in these efforts, which were also expected to help gain better acceptance of recycling practices, the importance of political and institutional barriers was recognized. One of the first major U.S. conferences on land application technologies noted that ''unless political and institutional constraints on the land application of effluents and sludges are recognized, identified, and resolved, [land application] projects will likely fail, regardless of their technical, scientific and economic feasibility'' (NASULGC, 1973). Conference participants identified a variety of social, legal and regulatory ''institutional'' constraints on the acceptance of land application as a viable wastewater treatment and sludge management technology. Among them were unfavorable public attitudes toward land-spreading activities, the number of local and State regulations that apply to those activities, and the sometimes-ambiguous nature of the laws and regulations within which sludge and effluent land application projects must be developed and operated.

During the 1970s and early 1980s, many millions of dollars were spent on research and demonstration efforts related to land application of municipal wastewater and sludge, which resulted in a dramatic improvement in the understanding of such practices and in the design and operation of land application projects. Research and extension reports, technology assessments, technical conference proceedings, design manuals, guidance documents, regulations, policy statements, documentary films and other educational materials were issued. At the same time a number of surveys and studies which focused on public acceptance and institutional barriers were undertaken (Christensen, 1982; Deese et al., 1980; Donnermeyer, 1977; Forster & Southgate, 1983; Musselman et al., 1980; Olson & Bruvold, 1980; Peyton et al., 1983; and Stitzlein, 1980). Some of their findings and recommendations were the following:

o Women are generally more reluctant than men to accept land application practices, because of concerns over possible health risks.

o Age is negatively correlated with acceptance of land application, as with most technological innovations.

o Formal education (at least through high school) is positively correlated with acceptance of land application.

o Experience with or exposure to the concepts involved in land application increases acceptance of these practices.

o Public opposition is highly correlated with land use intensity near land application sites.

o Although farmers and other landowners exhibit strong interest in the potential for short-term economic gain and in the stewardship of the land for future generations, they tend to oppose laws or regulations which impose land use controls.

o The individualism, conservatism, and agrarian ideology present in many rural areas often leads to the view that municipal waste is an urban problem that should be solved in the city by those creating the waste and not by pushing it off on rural communities.

o The use of land application practices has greatly expanded since 1973 due to technological and economic forces, and their use is expected to further expand; however, Federal, State and local government intervention and controls over these practices has also proliferated.

o While Federal legislation has encouraged land application, some State and local regulations have hampered it. Land use controls, health codes and nuisance laws have often been used effectively at the local level by project opponents.

o State governments currently set most of the rules governing land application practices. There is a mixed reaction among the many communities currently applying wastes to land concerning the need for Federal regulation of land application practices, although the understanding of current Federal standards is highly imperfect.

o Programs designed to win public acceptance differ across the country, since they must be geared to cultural influences, historic precedents and public concerns which change from community to community.

o ''Conventional wisdom'' has stressed the need for mounting educational programs or public relations campaigns in order to win public acceptance of land application programs, and a majority of communities have used them; however, some successful programs have found this unnecessary. Demonstration programs were identified as among the more effective educational methods.

o Institutional constraints have not proved to be a significant obstacle to most communities wishing to expand existing land application programs. High transportation costs and lack of land application sites were the reasons given most often for decisions to downscale or to avoid increasing land application programs - although institutional constraints may have been underlying these two ''technical'' or ''economic'' reasons (e.g., local opposition forcing reliance on more distant application sites).

o Officials usually underestimate a community's acceptance of a proposed land application program, because they are more likely to be confronted by those expressing negative views than those favoring the program.

Clearly, there are a host of institutional barriers which can inhibit the acceptance of land application practices. Although, by their very nature, institutions may be slow and difficult to change, most of these barriers can be modified if the attitudes of the individuals involved can be changed, overcome, or accomodated. The techniques most commonly cited as effective for land application practices include:

- o providing for adequate involvement in the decision-making process of those initially opposed;
- o clarification of the benefits associated with the practice;
- o resolution of concerns associated with possible nuisance problems, costs, public health or environmental impacts;
- o the use of project advisory groups, aggressive educational programs;
- o adequate contingency planning to assure that problems that might occur can be quickly dealt with;
- o providing for responsible project management to avoid unnecessary problems from occurring in the first place.

IMPLICATIONS FOR THE FUTURE OF OCEAN DISPOSAL ALTERNATIVES

As with land application practices, there are many social, legal and regulatory barriers to the acceptance of ocean disposal as a viable municipal wastewater and sludge management technology. While it may appear that ocean disposal has been more favorably accepted by the public as a whole than land-based disposal alternatives, try telling that to a coastal community faced with newspaper headlines announcing beach closures due to raw wastewater diversions or to those groups attempting to gain political support for modifying the current laws and regulations that control ocean disposal.

The problems of public attitudes toward ocean disposal are compounded by the ambiguity in the laws and regulations that affect ocean disposal practices. The control of ocean disposal practices, unlike land application practices, has involved more regulation by the Federal government with much less direct involvement by most States or local government agencies. Many of the Federal controls issued to date under the authorities of the Clean Water Act (CWA) and the Marine Protection, Research and Sanctuaries Act (MPRSA) were not primarily concerned with the disposal of municipal wastes into the marine environment. Furthermore, the implementation of government regulations has been complicated by court decisions, economic factors, Federal guidance, and the availability of funding for municipal wastewater treatment and sludge management facilities.

Both the CWA and MPRSA require that wastes disposed of in the oceans cause no ''unreasonable degradation of the marine environment.'' However, the legislation as enacted provides little guidance as to how to determine such an endpoint, what indices should be used, or within what time frame to operate. At the same time, MPRSA outlines factors that must be considered in establishing criteria for reviewing and evaluating ocean dumping permits.

The major factors which have affected both the laws and policies currently controlling ocean disposal practices include the following: 1) lack of adequate information on the environmental effects of ocean disposal practices, 2) the assumption that a reduction in ocean disposal activities would reduce harm to the environment, 3) the assumption that wastes do not offer a significant resource value to desirable ocean productivity, and 4) the assumption that policies regulating waste disposal must be considered separately for each medium (air, land, and water).

Studies issued in the 1970s strongly supported the need for better information concerning the environmental impacts of ocean disposal practices (Ryther & Dunstan, 1971; NAS, 1975, 76, & 77; Gross, 1977). However, the research needed to evaluate both the fate and effects of pollutants placed in the ocean as a result of ocean disposal of municipal wastes has lagged in comparison to that undertaken on land application over the past decade, likely as a result of the pending Congressional deadlines for phasing out ocean dumping practices.

Clearly the potential effects of ocean disposal can be severe (e.g., beach closures, contamination of marine ogranisms destined for human consumption, toxic effects on the survival and reproduction of marine ecosyatems, degradation of benthic habitats, and anoxic episodes resulting in fish kills). Such concerns are similar to those raised as possible impacts associated with land-based alternatives (e.g., surface and groundwater contamination affecting drinking water supplies or freshwater fisheries, uptake of pollutants by human food-chain crops, adverse effects on wildlife). However, a major difference lies in the fact that in spite of extensive efforts to document their occurrence, there is very little evidence demonstrating that land application of municipal wastewater or sludge, as currently practiced, has resulted in any significant public health or environmental impacts (Page et al., 1983). Partly as a result of these ''lack-of-a-problem'' findings, it appears that further studies in this area will no longer be given the priority for funding they once received.

Although some of the more recent studies and assessments concerning the status of the marine environment and the impacts of ocean disposal practices (Duke, 1981; NACOA, 1981; NAS, 1984a & b) are far more optimistic than those issued in the 1970s, their conclusions continue to be clouded by inadequate supporting data. Yet these more positive scientific assessments have helped serve as a basis for a major change in future ocean dumping of sewage sludge in the North Atlantic -- namely movement from the shallow 12 Mile Site to the much deeper 106 Mile Site which is located just off the continental shelf, although it has yet to be determined whether this will be a long-term or short-term solution to the ocean dumping controversy. Still, unlike the research and development efforts that have supported improvements in land application practices, relatively little effort has gone into addressing the questions of how to dispose of municipal wastes in the ocean properly and how clean these wastes must be to be compatible with the marine environment. Instead, the work has continued to be focused on monitoring the effects of existing ocean disposal practices that have already been determined to be unacceptable on political and institutional grounds.

If, however, the marine scientists as a group are ready to agree that ocean disposal is beneficial to the marine environment or at least a tolerable insult, this message needs to be presented in a more convincing manner to the regulatory agencies, Congress and the general public. Surely those groups interested in the competing, ''more traditional'' uses of the ocean (e.g., commercial fisheries and recreation) will continue to mount strong opposition to anything they view as adversely affecting their uses of the ocean. The reactions to proposed off-shore oil drilling leases, ocean incineration, low-level rad waste disposal, and the proposals by coastal communities for permits to discharge less-than-secondary effluent into marine waters serve as good examples of the concerns and influence of these groups. Techniques similar to those identified as effective for gaining better acceptance of land application practices might be useful in getting across a new message regarding the acceptability of impacts associated with well-sited and well-managed ocean disposal activities (i.e., providing adequate involvement in decisionmaking by those initially opposed; clarifying the benefits as well as risks; developing aggressive educational programs; providing adequate contingency plans; and providing for responsible project management).

It may still be worth considering the ocean as another medium for recycling municipal wastes as a resource. Unlike countries such as Japan, little conscious effort has been made in this country to manage or stimulate beneficial effects, such as increased productivity of desirable species, as a result of waste disposal/management in the marine environment. For example, many of Japan's most productive shellfish culturing areas are in waters that would be closed for commercial harvest in the United States because of nearby domestic wastewater discharges. Careful management practices, including extensive depuration procedures, allow the Japanese to take advantage of these highly productive waters. This lack of past U.S. activity or the concerns over competition by the open-ocean fishermen should perhaps not preclude further consideration of the potential for stimulating increased productivity of desirable species. Engineering criteria and management practices could be developed by which wastes with acceptable characteristics could be added to carefully selected ocean sites (e.g., areas with nutrient- or carbon-limited productivity). This might lead to developing as strong a basis for recycling municipal wastes in the ocean as exists for their recycling by land application.

REFERENCES

American Public Works Association (APWA) Research Foundation, Proceedings of the August 11-13, 1983, Airlie House, VA Workshop Defining the Role of Private Sector Activities in Solving Municipal Environmental Problems, prepared by APWA for the U.S. Environmental Protection Agency, Washington, D.C., March 1984.

Association of State and Interstate Water Pollution Control Administrators (ASIWPCA), America's Clean Water: The State's Evaluation of Progress 1972-1982, ASIWPCA, Washington, D.C., February 1984.

Booz, Allen and Hamilton, Inc., A Survey of the Use of Innovative Technology in Municipal Wastewater Treatment, prepared for the U.S. Environmental Protection Agency, Washington, D.C., 1976.

Christensen, L.A., Irrigating with Municipal Effluent: A Socioeconomic Study of Community Experiences, ERS-672, U.S. Department of Agriculture, Washington, D.C., January 1982.

Deese, P.L., Miyares, J.R. and Fogel, S., Institutional Constraints and Public Acceptance Barriers to Utilization of Municipal Wastewater and Sludge for Land Reclamation and Biomass Production, EPA 430/9-81-013, U.S. Environmental Protection Agency, Washington, D.C., July 1981.

Donnermeyer, J., ''Socio-cultural Factors Associated with the Utilization of Municipal Waste on Farmland for Agricultural Purposes,'' in Wastewater Management in Rural Communities: A Socio-economic Perspective, eds. C.E. Young and D.J. Epp, p. 154, Institute for Research on Land and Water Resources, The Pennsylvania State University, University Park, PA., 1977.

Duke, T.W. (ed), Impact of Man on the Coastal Environment, EPA 600/8-82-021, U.S. Environmental Protection Agency, Gulf Breeze, FL., 1982.

Foster, D.L. and Southgate, D.D., ''Institutions Constraining the Utilization of Municipal Wastewater and Sludge on Land,'' in Proceedings of Workshop on Utilization of Municipal Wastewater and Sludge on Land, eds. A.L. Page et al., pp. 29-45, University of California, Riverside, CA., 1983.

Gross, M.G. (ed), Proceedings of the Symposium at the American Museum of Natural History, New York City, November 3-5, 1975, American Soc. of Limnology and Oceanography, Special Symposium Volume No. 2, Allen Press, Lawrence, KA., 1977.

Government Accounting Office (GAO), EPA's Innovative Technology Program for Waste Water Treatment Needs Better Controls, GAO/RECD-84-79, August 1984.

Jewell, W.J. and Seabrook, B.L., A History of Land Application as a Treatment Alternative, U.S. Environmental Protection Agency, EPA 430/9-79-012, Washington, D.C., April 1979.

Mercer, J.L., and Philips, R.J., Public Technology - Key to Improved Government Productivity, Amocon Press, New York, NY., 1981.

Musselman, N.M., Welling, L.G., Newman, S.C. and Sharp, O.A., Information Programs Affect Attitudes Toward Sewage Sludge Use in Agriculture, EPA 600-/2-80-103, U.S. Environmental Protection Agency, Cincinnati, OH., July 1980.

National Association of State Universities and Land Grant Colleges (NASULGC), Proceedings of the Joint Conference on Recycling Municipal Sludges and Effluents on Land, NASULGC, Washington, D.C., July 1973.

National Academy of Sciences (NAS), Assessing Potential Ocean Pollutants, Commission on Natural Resources, National Academy of Sciences, Washington, D.C., 1975.

National Academy of Sciences (NAS), Disposal in the Marine Environment: An Oceanographic Assessment, Commission on Natural Resources, National Academy of Sciences, Washington, D.C., 1976.

National Academy of Sciences (NAS), Multimedium Management of Municipal Sludge, Commission on Natural Resources, National Academy of Sciences, Washington, D.C., 1977.

National Academy of Sciences (NAS), Disposal of Industrial and Domestic Wastes: Land and Sea Alternatives, National Academy Press, Washington, D.C., 1984a.

National Academy of Sciences (NAS), Ocean Disposal Systems for Sewage Sludge and Effluents, National Academy Press, Washington, D.C., 1984b.

National Advisory Committee on Oceans and Atmosphere (NACOA), The Role of the Ocean in a Waste Management Strategy: A Special Report to the President and the Congress, NACOA, Washington, D.C., January 1981.

Olson, B.H. and Bruvold, W., ''Influences of Social Factors in Public Acceptance of Renovated Wastewater,'' in Water Reuse, ed. E.J. Middlebrooks, pp. 55-73, Ann Arbor Science Publ., Ann Arbor, MI., 1982.

Peyton, R.B., Gigliotti, L. and T. Logerstrom, A Survey of Public Perception and Acceptance of Sludge Application to State Forest Lands in Michigan, Phase I Summary Report, Dept. of Fisheries & Wildlife, Michigan State University, Lansing, MI., 1983.

Ryther, J.H. and Dunsten, W.M., ''Nitrogen, Phosphorus and Eutrophication of the Coastal Marine Environment,'' Science, 171:1008-1013, 1971.

Stitzlein, J.N., ''Public Acceptance of Land Application of Sewage Sludge,'' in Utilization of Wastes on Land: Emphasis on Municipal Sewage, U.S. Department of Agriculture, Washington, D.C., 1980.

Tarr, J.A., ''City Sewage and the American Farmer,'' Biocycle, 22(5):36-38.

U.S. Environmental Protection Agency, Innovative and Alternative Technology Projects: A Progress Report, Office of Water Program Operations, Washington, D.C., September 1984.

Wileman, D.L., The Role of Industry in Urban Technology Transfer: A Comparative Profile, prepared for the National Science Foundation (Contract NSF-PRA-77-23746), November 1979.

APPENDIX

Major Federal Legislation Concerning Municipal Wastewater Treatment and Sludge Management

FEDERAL WATER POLLUTION CONTROL ACT (FWPCA), as amended in 1972 (PL 92-500), 1977 (PL 95-217; the Clean Water Act), and 1981 (PL 97-117; the Municipal Wastewater Treatment Construction Grant Amendments), focus on the restoration and maintenance of the chemical, physical and biological integrity of the Nation's waters. Research, standards and enforcement, water quality planning and construction grants program authorities are included. They cover control of both point and non-point sources of water pollution. These laws authorize Federal funding for the planning, design and construction of publicly-owned wastewater treatment works (POTWs), including sludge management facilities. They also authorize the issuance of comprehensive sewage sludge management guidelines and regulations, the issuance of National Pollution Discharge Elimination System (NPDES) permits for point source discharges, and the development of areawide waste treatment management plans including best management practices (BMPs) for non-point sources of water pollution, and require the development and implementation of pretreatment standards for industrial discharges into POTWs. The 1977 and 1981 amendments added several important waste management provisions, including special incentives for greater use of innovative and alternative waste treatment technologies and methodologies, broad authority to regulate sewage sludge management practices, pretreatment credits for industrial dischargers to POTWs, and the opportunity for coastal communities to apply for modified discharge permits which would allow for less than secondary treatment for discharges into marine waters.

THE SOLID WASTE DISPOSAL ACT, as amended in 1976 (PL 94-580; the Resource Conservation and Recovery Act [RCRA]) and 1984 (PL 98-616; the Hazardous and Solid Waste Amendments), focuses on the regulation of solid waste management practices to protect human health and the environment while promoting the conservation and recovery of resources from solid wastes. Technical and financial assistance, training grants, solid waste planning, resource recovery demonstration assistance and hazardous waste regulatory program authorities are included. The key aspect of RCRA is the comprehensive regulatory system to ensure the proper management of hazardous waste. RCRA also provides for technical and financial assistance to State, local and interstate agencies for the development of solid waste agencies and solid waste management plans. In addition, it prohibits open dumping of wastes; promotes a national R&D program for improving solid waste management practices; and calls for a cooperative effort among Federal, State, and local governments and private enterprise to recover valuable materials and energy from solid wastes.

THE CLEAN AIR ACT AMENDMENTS (CAA) of 1970 (PL 91-604) and 1977 (PL 95-95) focus on the protection and enhancement of the quality of the Nation's air resources in order to protect public health and welfare and the productive capacity of the country. A national R&D program, technical and financial assistance, emission standards, and air quality planning assistance program authorities are included. CAA provides for technical and financial assistance to State and local governments for the development and execution of their air pollution control programs, encourages and assists the development and operation of regional air pollution control programs, and initiates an accelerated national R&D program to achieve the prevention and control of air pollution. It authorizes the development of State implementation plans (SIPs) for the purpose of meeting minimum Federal ambient air quality standards. It also authorizes issuance of regulations to control hazardous air pollutants and new source performance standards (i.e., emission standards).

THE MARINE PROTECTION, RESEARCH AND SANCTUARIES ACT (MPRSA) of 1972 (PL 92-532) and its amendments provide for regulating the dumping of all types of materials into ocean waters and limiting the ocean dumping of materials which would adversely affect human health and welfare of the marine environment and its commercial values. The MPRSA prohibits ocean dumping and transportation from the U.S. for purposes of dumping except pursuant to permit, and permits are not to be issued where dumping would ''unreasonably degrade'' the marine environment. Permitting regulations, marine research, and provisions for establishment of marine sanctuaries are included. EPA is required to establish criteria for evaluating ocean dumping permit applications, applying specific statutory factors. A 1977 Amendment (PL 95-153) effectively established December 31, 1981, as the deadline for terminating ocean dumping of ''sewage sludges,'' defined as municipal waste ''which may unreasonably degrade or endanger human health, welfare, amenities, or the marine environment, ecological systems, or economic potentialities.'' A similar amendment was enacted in 1980 for ''industrial wastes.''

THE TOXIC SUBSTANCES CONTROL ACT (TSCA) of 1976 (PL 94-469) provides for the testing and premanufacture notification of chemical substances and mixtures, and the regulation of production or use of certain ones which present an unreasonable risk of injury to health or the environment. Along with its many regulatory, testing and reporting requirements, EPA is also required under Section 9 of TSCA to coordinate actions taken under TSCA with actions taken under other Federal laws. In addition, TSCA requires that EPA issue rules respecting the manufacturing, processing, distribution in commerce and disposal of PCBs.

Sludge Management and Ocean Disposal: A Discussion of Public vs Private Alternatives

Harvey Goldman
Arthur Young

INTRODUCTION

Congress passed the Clean Water Act in 1972, but the consequences of that action affect our efforts at sludge management even today, and will continue to do so in the future.

One of the most highly publicized parts of the Act is the Construction Grants Program. The United States Environmental Protection Agency, in carrying out the directives of Congress, issued regulations concerning the treatment and disposal of wastewater, and provided a Federal grant program to cover 75% of a community's cost of building the required facilities. But over the years, as funding assistance became less abundant, treatment requirements became more stringent. And as the degree of treatment increased, the amount of residual material which had to be disposed increased as well. The U.S. EPA estimates that the quantity of municipal sludge produced annually has almost doubled since 1972, with municipalities currently generating approximately 6.2 million dry metric tons of wastewater sludge per year.

It is clear that over time a very unique set of sludge management issues has developed. These issues include:

Where and how will this increasing amount of sludge be disposed?

What will be the cost of disposal?

Will the cost of disposal be financed publicly or privately?

Is there any potential value to the waste material, and if so, can that value be enhanced?

Ocean disposal is one of many options for sludge disposal. While it is not the intent here to point to any one technology as the best alternative, this paper will explore the financing options available for communities disposing of their sewage waste in ocean waters.

GOING PUBLIC OR PRIVATE?

In attempting to address the issues presented above, and particularly whether a program or facility should be financed publicly or privately, one must first consider the factors that affect decisions related to public/private partnerships for sludge management. These factors include technology, public relations, resource management, public health, environmental concerns, and economics.

While the costs of different treatment and disposal technologies obviously differ, if there is a potential for the project to be privately financed, a key factor is that some technologies and disposal methods are more suited to private investment than others. In addition to the actual business opportunity of performing the service, investors look to tax benefits to make the opportunity more attractive. For example, equipment intensive technologies provide the opportunity for a high percent of five-year depreciable property and investment tax credits. For comparative purposes, consider land application based treatment technologies. Since land is not depreciable, potential tax benefits are much lower.

A program of public relations is essential if waste disposal is to be financed privately, because the general public must be convinced that the activity, which is related to public health, is in the hands of responsible parties. Public support must be continually maintained at high levels if the private sector is to be involved.

Resource management is an important part of any business venture. In waste treatment and disposal, the resources to be managed include personnel and environmental and economic resources. A private firm will try to optimize the use of these resources. A major difference between public and private approaches to sludge management is as follows: what the public sector typically considers waste disposal, the private sector considers waste management. In other words, many private, profit oriented firms, especially those living with or looking for business opportunities within The Resource Conservation and Recovery Act (RCRA) regulations, consider the "waste" material and the opportunity for waste management as potential revenue generators. This topic will be discussed again before the end of this paper, as there may be important implications for the public sector.

Contractual specifications are the key to addressing any public health and environmental concerns that may arise as a result of turning over waste disposal responsibilities to a private sector firm. When a municipality contracts with a private firm for the provision of a service, the responsibility for the quality and timely delivery of that service remains with the local government. Through a comprehensive and carefully written contract, safety and health ensuring measures can be made part of the arrangement.

The issue of economics is relatively straight forward. Independent of the other issues that pertain to the decision to go with private financing, if the private sector firm can provide the same level of quality service at a lower cost than the municipality can by performing the service itself, then most would agree that under Reaganomics the service should be provided through the private sector. However, perspectives and concerns of others must be considered. While certainly important, economics is only one factor in the decision, and the economic analysis should recognize that the community must still contribute time and effort into implementation, monitoring and oversight activities.

The method of waste disposal is another factor which affects the decision to go public or private. The five most common forms of sludge disposal are: land application; distribution and marketing of a sludge derived product; landfilling; incineration; and ocean disposal.

Currently about 25 percent of the nation's sludge is applied to land. In some cases, such as in agricultural and forest application and land reclamation, sludge is used to improve agricultural productivity as well as serving as a sludge treatment system. In other instances sludge is applied to land dedicated solely to the purpose of sludge disposal.

Distribution and marketing of sludge products is perhaps the most business oriented disposal option. Sludge products sold for application to lawns, shrubs and other ornamental plants, orchards and nurseries take advantage of the cash value of sludge. One of the most well known products is Milorganite, a heat-dried sludge product sold as a soil conditioner.

Cities that distribute heat-dried sludge include Chicago, Illinois; Houston, Texas; Largo, Florida; Newport News, Virginia; and the Greater Atlanta, Georgia, area. Municipalities that distribute composted sludge include Philadelphia, Pennsylvania; the District of Columbia; Kittery, Maine; Topeka, Kansas; Salt Lake City, Utah; Columbus, Ohio; Missoula, Montana; Portland, Maine; Portland, Oregon; and the Greater Los Angeles, California, area.(1)

The three remaining methods, incineration, landfilling, and ocean disposal, as commonly practiced, have a common feature. Although some work is being done in the area of co-incineration, none of the three methods look to the potential value of the waste material, as would a private sector firm. In addition, the last two have the potential disadvantage of being short-term solutions. Environmental concerns over landfilling and ocean disposal have led to the outright banning of sludge landfilling in New Jersey and designation of a 106 mile site to replace the 12 mile site for ocean dumping. The majority of sludge in New Jersey, measured on a tonnage basis, is currently ocean disposed. If ocean disposal were to be eliminated, the next least expensive and most easily implemented alternative, which would be landfilling, has been previously removed from consideration.

PRIVATIZATION

Private sector involvement in the area of wastewater and sludge treatment and disposal is a new trend, commonly referred to as "Privatization." Privatization is a process through which local governments can capitalize on advantages unique to the private sector in owning, building and/or operating capital intensive facilities. The advantages of service delivery through privatization include potential construction cost savings and lower financing costs.(2) Construction savings result from simplified procurement processes and regulations, and minimal regulatory involvement. Because a privatization project is private sector work, the requirements of public procurement and construction do not hold. The ultimate cost to a community of a service provided through privatization may also be significantly lowered due to tax benefits, which are available to the private sector, but are not available to tax-exempt municipalities. Among these tax benefits are an investment tax credit, accelerated depreciation and deductibility of interest.

Privatization provides other benefits as well. Through the private sector, operational efficiencies have the potential to lower operation and maintenance costs. These operational efficiencies include: pooling of technical and management personnel for multiple facilities; bulk ordering of chemicals and supplies; and the existence of a profit motive, which encourages innovative operations and uses of treatment facilities.

A typical privatization transaction is shown in Figure 1.

From the governmental perspective, the government has a contractual relationship with a private owner/operator for the establishment and operation of a facility to meet the service needs of the area. In return, the community pays a service fee according to a predetermined rate structure, while the private owner/operator must meet contractually defined service standards to earn the fee.

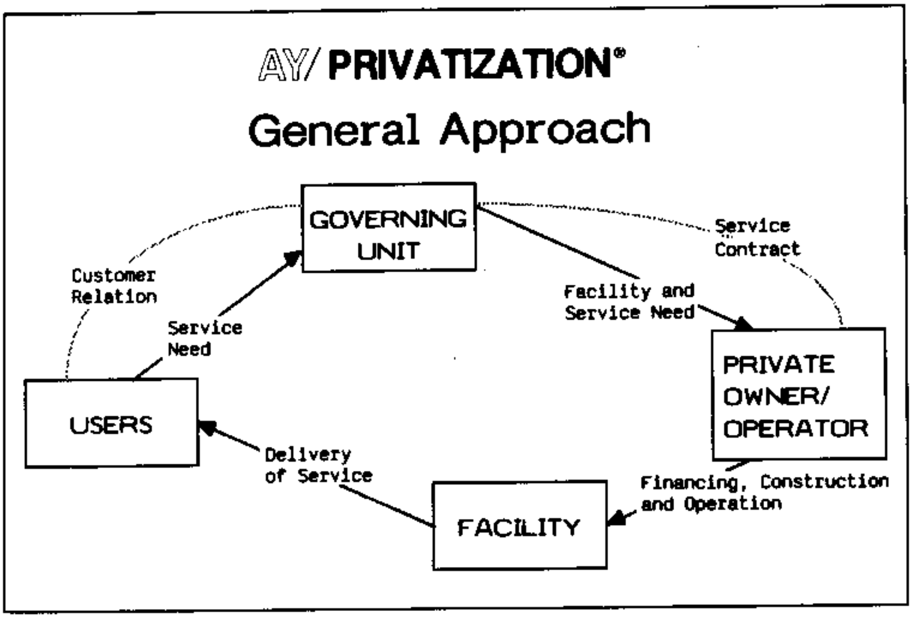

Figure 1

From the private owner/operator perspective, the owner/operator wins the business opportunity of an important service contract. In return for what is expected to be a reasonable profit, the owner/operator assumes most of the risks involved in financing, constructing and operating the facility in accordance with contract terms for the delivery of the service.

To the users of the service, the privatization transaction is invisible. The city sets the user charge rate and "retails" the treatment and/or disposal service, which is provided by the private sector "wholesaler". In addition to billing and collection from system users, the city typically will maintain reponsibility for system hookups and disconnections, customer correspondence and other customer interactions. The "privatizer" may be free to deal directly with non-municipal users of the system, and large commercial and industrial users within the local area.

One of the most recently completed privatization projects is in Auburn, Alabama. Construction is already underway and the facility is scheduled to go on line in October, 1986. Among the project's distinguishing features is that it is the first full service wastewater treatment privatization project to date in the United States. That is, there is one firm providing the financing, design, construction and operation of the facility. It is also the country's largest wastewater treatment privatization project to date. Through privatization, a 5.4 mgd plant will be built on the southside, a 1.6 mgd plant will be built on the northside and a total of 27 miles of interceptor sewer lines will be constructed. Savings due to privatization over the life of the 25-year project are estimated to exceed $25 million. Auburn considered other financing options, including local financing, a combination of local and grant funding, and privatization. Privatization resulted in the most economical alternative for Auburn. An economic comparison of the treatment options is presented in Figure 2.

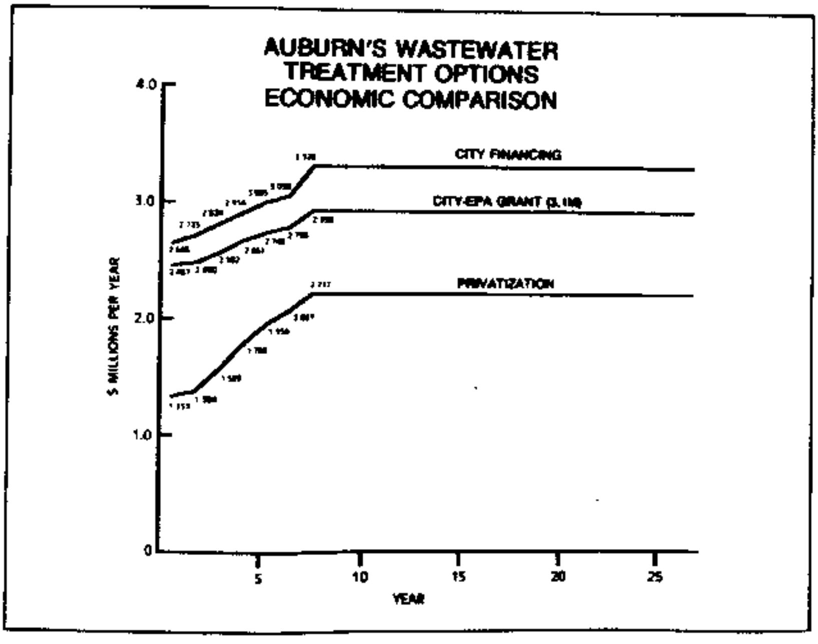

Figure 2

DETERMINING THE FEASBILITY OF PRIVATIZATION

Understanding why a community would want to undertake a privatization project, it is important to understand how a community would determine if privatization is suitable for its particular needs. Determining the feasibility of privatization, and once it is determined feasible, establishing an implementation program, can be described as a three-stage approach, which is diagrammed in Figure 3.

The first stage is a feasibility study in which 7 key areas are evaluated. The first step is to obtain a clear understanding of the community's needs. Many communities already have this information available. The need for a new facility may be so great that the facility has already been designed and the only problem to overcome is a lack of financing. A good source of information is likely to be the Mayor or City Manager, Public Works Director or Consulting Engineer. For example, the information for a wastewater treatment facility is likely to be obtainable from a 201 facilities plan, which may have been prepared as part of the U.S. EPA Construction Grants Program.

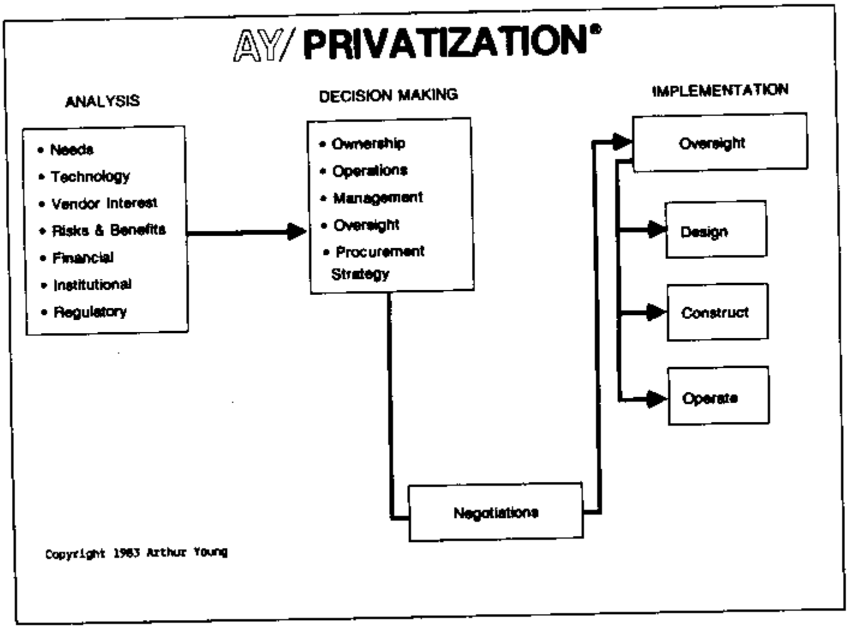

Figure 3

The second area is a study of technologies. The choice of technology will affect the economics of a privatization project in that the more equipment intensive technologies will result in greater tax benefits. Equipment intensive technologies offer a higher degree of 5-year depreciable property than do land-based technologies.

The third area for study is an assessment of vendor interest. The level of private sector interest in privatization of a proposed project must be gauged, and a strategy to cultivate private sector interest in the proposed project may often be appropriate. One approach to encourage interest and competition is to advertise the proposed project and to request preliminary expressions of interest and/or qualifications. As an example, before deciding to make the investment of time, effort and funds in preparing and issuing a request for privatization proposals, the Village of East Aurora, New York decided to solicit preliminary expressions of interest from potential privatizers of a needed treatment facility. The total project construction costs were estimated at approximately $3 million. Having received a strong demonstration of vendor interest in its project, the Village proceeded with the confidence that potential privatizers would be available.

The next area of analysis is that of risks and benefits. Impacts on a site specific basis must be addressed for each project from political, economic and social perspectives. These impacts are likely to include factors such as the impact upon economic growth and the control of future development. Equally important to consider is the strategy for protecting the interests of existing employees and local vendors, the potential of an increased tax base and the importance of educating the public and the local press about the concept.

The fifth area is related to performance of a financial alternatives survey. The economics of privatization are compared to other available financing alternatives. If privatization is found to be the best alternative, then the decision becomes which option should be chosen for stucturing the privatization transaction. The objective in putting together a financing package for privatization is to obtain the lowest annual charge for users of the facility, while at the same time obtaining an adequate return on investment to the private sector.

Privatization project financings are backed by project revenues, not municipal credit. These transactions can be financed with debt or they may involve some combination of debt and equity. Therefore, another objective in structuring the transaction is to be responsive to both debt and equity markets.

The sixth area in the feasibililty study addresses the institutional factors which may influence the privatization project. Generally, legal issues which must be addressed include any existing limits on a community's ability to enter into a long-term contract. Private sector firms involved in the operation and maintenance of facilities built through privatization want assurances that their contracts will last on the order of 20 to 25 years, so that they may take advantage of the full tax benefits available. If a community is constrained in its ability to enter into long-term contracts, the privatization project may not be attractive to some private sector firms.

Limitations or constraints on the method of procurement allowable for communities is important, because oftentimes, a community finds its would prefer to negotiate a privatization contract with one or two firms, rather than go to a competitive bid process for privatization.

A third type of legal issue to be addressed is whether or not any constraints exist on the type of contract which will be entered into between the community and the private sector firm. In doing an analysis of institutional factors and legal issues, the situation often arises that enabling legislation needs to be passed to allow privatization to go forward. Such was the case in the State of Utah, before the State passed the Utah Privatization Act. A number of other states have either passed or are considering enacting privatization legislation. These states include New Jersey, Georgia, Tennessee, Connecticut, Florida, Washington and Pennsylvania.

The seventh area for analysis in the privatization feasibility study is that of the regulatory interfaces which may occur during the project. This area of study concerns the involvement of regulatory agencies at the federal and state levels. Representative issues which must be addressed during the feasibility study are the role that the state government will assume with regard to privatization, compliance with National Pollution Discharge, Elimination System permit requirements, enforcement issues, treatment program requirements, environmental aesthetics and other factors.

The feasiblity study, if properly prepared, will have identified a variety of acceptable procedures or alternatives for each of the study areas. A community must choose among them to establish its step-by-step implementation plan. In some cases, circumstances may warrant that additional matters be considered. In others, previous decisions will reduce the alternatives to be considered.

In choosing the alternatives, a number of key decisions will have to be made by the community. Decisions relating to ownership and operation of the facility are of special concern to both the local government and the private sector firm. For the private firm to be eligible for the tax benefits, it must have ownership of the facility. However, the public sector should have the right to a purchase option, due to various legal, financial and other matters. The purchase is an important leverage factor for the public sector. The purchase option and the conditions under which it can be exercised must be written into the privatization contract.

Other issues of ownership may involve the land on which the facility is located. It may be appropriate for the community to own the site and lease it to the private sector firm for a reasonable, annual fee. Decisions regarding management of the facility relate to the fact that a community is turning over provision of an essential service related to public health. The nature and importance of the service demands that the community monitor the private sector's performance. In addition, most privatization transactions will include provisions for potential ownership transfer of the facility to the public sector at some future time. Since the public must be assured of the quality and reliability of the service provided, and of the overall condition of the facility, the community should require that it be allowed to have significant quality control and oversight responsibility. Therefore, while direct management will most likely rest in the hands of the private sector firm, the municipality may want to stipulate minimum reporting and management system requirements for the private sector firm to meet.

Similarly, the community must decide the oversight responsibilties which it is willing to undertake. Oversight responsibilities should include activities relating to the design of the facility and should continue through construction and operation. Local officials must concern themselves with the service quality, cost, and its timely delivery. In the case where a new facility is designed and constructed, the community's engineering advisors should review design plans and play a pre-determined, on-going oversight role during construction and/or operations. Establishing a privatization schedule will require consideration of all aspects or privatization, including the immediacy of treatment needs and the means by which the services of the private sector will be obtained.

One of the major challenges facing the community is selection of an appropriately qualified privatizer or service provider. Generally, and if allowable under local and state law, a municipality has two options for procurement: negotiated or competitive. If a competitive procurement process is chosen, a request for qualifications or a request for privatization proposals can be requested from interested firms. Table 1 is an example of what a representative sludge management RFP might contain. Once proposals are received and evaluated and the most qualified firm chosen, the next step is to enter into contract negotiations. Key issues during negotiations include technical matters, legal matters, issues related to risk mangement, financial issues and the types of oversight and education programs presented by the proposer.

PRIVATIZATION OF SLUDGE MANAGEMENT FACILITIES

While privatization is applicable to complete wastewater treatment systems, it can also be used for sludge management facilities. A Baltimore area sewage authority is undertaking a privatization project for a reactor composting unit to process 30 dry tons per day, with the potential for revenue from sale of the processed material as fertilizer. The project is likely to be financed with approximately $10 million in Industrial Development Bonds. There are at least six other privatization projects currently underway, or under consideration, for sludge management facilities.

Camden County, New Jersey provides an example of how privatization can meet the specific needs of public agencies. Over the past decade the County has received and continues to receive significant assistance from U.S. EPA for the construction of required wastewater treatment facilities. Need of increased capacity in both plant and collection systems, more stringent treatment requirements, and decreasing availability of grant funds in the state, have left the County to finance a required sludge management facility with local funds. An analysis of potential user charges under various scenarios of grant funding was performed

Table 1

Representative Contents of
Request for Proposals for
Sludge Management Facilities

1. Letter of Welcome
   - o Purpose/objective of RFP
   - o Instructions
   - o Summary of Needs
   - o Summary of Project Design
   - o Procurement Process Summary
   - o Schedule

2. o Instructions
   - o Standardized Assumptions
   - o Risk Posture of the Authority
   - o Proposer's Qualifications
   - o Guarantees Required
   - o Other Requirements
   - o Standardized Table of Contents for Proposal

3. Technical Conditions and Assumptions
   - o Location and Description of Existing Facilities
   - o Flows and Wastewater Quality Data
   - o Sludge Quality and Quantity Data
   - o Definition of "Acceptable Sludge"
   - o Acceptable Technologies
   - o Engineering and Survey Documents Available
   - o Availability for Increases in Flow
   - o Required Technical Description of Technology
   - o Description of Proposer's Existing Facilities
   - o Marketing and Distribution Plan
   - o Contingency Plans

4. Legal Insurance and Related Matters
   - o Ownership/Use of Site
   - o Term and Termination of Service Agreement
   - o Insurance and Indemnification Requirements
   - o Permits
   - o Existing Employees
   - o Assignment of Contracts
   - o Force Majeure Provisions
   - o Compliance with Applicable Law/Regulations
   - o Penalties for Non Compliance/Non Performance

5. Financial Conditions and Assumptions
   - o Standardized Financial Assumptions
   - o Base Service Charge
   - o Revenue Sharing
   - o Purchase Option
   - o Repair and Replacement Funds
   - o State and Local Tax Information
   - o Financial Plan

6. Anticipated Interfaces
   - o Municipal Government
   - o Public Relations/Public Education
   - o Regulatory Agencies

7. Facility Management System Requirements
   - o O&M of Facilities
   - o Marketing and Sales
   - o Reporting Requirements
   - o Oversight Program

8. Proposal Evaluation Process and Award
   - o Evaluation Process
   - o Evaluation Guidelines
   - o Contract Negotiations and Award

by Arthur Young. Since the treatment and disposal of sludge could be segmented out operationally, privatization of that portion of the treatment facilities could be practically implemented. It appears that privatization could reduce user charges in comparison to local public financing without endangering the grant funding that is available to the County for other portions of the treatment facilities.

The approaches taken to the privatization of sludge management facilities have varied both in style of procurement and extent of privatization. In the Baltimore area project for example, the authority specified in its Request for Qualifications the type of technology desired, and the level of experience required by prospective proposers. This limited the number of firms which responded. The authority then chose two firms for concurrent competitive negotiations. The Authority used its own financial advisors to intensively review proposed financial plans.

Negotiations, which covered detailed technical matters, financing, risk sharing, revenue sharing and guarantees, were coducted over a period of 4 to 5 months.

In contrast, the City of Orlando requested proposals for privatization of its sludge management facilities, but did not restrict proposals with respect to a specific technology. The City screened proposals, each containing cost estimates for different technologies, and held oral interviews with several firms before selecting one for contract negotiations.

In a recent development in Utah, the Central Valley Water Reclamation Facility Board is exploring several levels of privatization. The Board has just issued a Request for Proposals for the construction of a sludge disposal facility. As in the Balitmore procurement, acceptable technology has been limited. In addition, the Board is requesting that the proposals address three types of arrangements: A "turnkey" approach in which the proposer would design, contruct and startup the facility; a "full service" approach in which the proposer would design, construct and operate the facility; and a "privatization" approach in which financing and ownership of the facility would also be the responsiblity of the proposer. By taking this approach the Board has left its options open so that the Board can review the various alternatives and choose the approach that best meets its needs.

OCEAN DISPOSAL: PRIVATIZATION?

Before discussing privatization of ocean disposal, it is necessary to review the factors that affect the economics of ocean disposal. It is clear that a number of factors come into play. Among these factors is sludge quantity; the technology for sludge treatment; community size; storage requirements; on land transportation requirements; and disposal monitoring costs.

The cost of ocean disposal is proportional to the amount of material to be barged and the distance that must be traveled. Whether the material is measured in volume of wet sludge or the mass of dry sludge, costs will increase as the quantity of sludge increases.

Certain sludge treatment processes will affect the economics of ocean disposal. Thickening and dewatering may reduce the quantity of sludge and thereby reduce barging costs. However, the potential savings must be compared to the cost of installing and operating the sludge treatment equipment.

Large communities are more likely to afford the costs associated with ocean disposal than are smaller ones. A portion of these costs relate to the permitting and monitoring programs required for ocean disposal. For small communities, the cost of these programs may only be affordable when ocean disposal is undertaken in conjunction with other communities. Such associations would also ensure that sludge hauling vessels would be filled frequently enough for efficient operation.

Part of the cost of ocean disposal is related to sludge storage requirements. Storage facilities, whether tanks, land capacity, or in some cases, spare barges, are necessary for periods of disruption in service, or more likely, for periods in which all available barges are either in transit or out of service. The elements which determine the storage requirements are the time duration of hauling trips, the capacity of existing barges and the rate of sludge production. Sludge treatment facilities are often adequate storage facilities.

For a shore community, relatively near navigable water, on land transportation costs will not be an important factor in the economics of sludge disposal. But for an inland community, or a community without a protected, navigable harbor, transportation to barge loading facilities could prove expensive depending on the distance to shore.

Two other cost factors are somewhat related. The greater the distance that must be traveled at sea, the higher the disposal cost. A concern often expressed about ocean dumping is that poor weather conditions and rough water provide incentive for "short dumping," a term which refers to the discharge of waste material prior to the barge reaching the designated site. To discourage such action, electronic monitoring equipment can be used, but it is expensive.

While the list of factors affecting the economics of ocean disposal is by no means all-inclusive, it does provide some insight into overall cost consideration. Another source of insight into the costs of ocean disposal is provided in a report recently issued by the National Academy of Sciences (3). The report presents a parametric cost analysis for surface vessel and pipeline waste transportation, and includes case studies of disposal sites.

One of the most highly publicized case studies in ocean disposal is the controversy over the "12-mile" site versus the "106-mile" site for ocean-dumping communities in the New York/New Jersey area.

12-MILE SITE VS 106-MILE SITE

In 1970 Congress passed the Marine Protection Research and Sanctuaries Act, designating the 12-mile site off the New York/New Jersey metropolitan area for sludge disposal. Although the designation was temporary, and expired in 1981, dumping has continued under court orders. The 106-mile site has been designated as a replacement, but regulations concerning the site have not yet been promulgated.

While ocean disposal is generally considered an economically feasible method for sludge disposal, costs arising from required permitting and monitoring, and the potential requirement for deepwater dumping significantly increase the cost of ocean disposal. The sludge disposal issue currently faced by New York City is commonly referred to as the 12-mile versus 106-mile controversy. According to information provided by the City's Division of Plant Services, switching from the 12-mile site to the 106-mile site will increase disposal costs from $0.08 to $0.28 per cubic foot (over a 300% increase) and increase hauling time from 5 hours to 48 hours for one round trip. The City operates tug towed barges. It is interesting to note that the $0.08 figure is half of what it has been in previous years, a result of competition among various private sector firms vying for the towing contract.

Competition is also cited by the Linden-Roselle Sewage Authority as a factor in lowering its present cost of ocean disposal. The Authority's present cost for disposal at the 12-mile site is $1.14 per wet ton. As recently as three years ago the price was $2.53 per wet ton. One reason given by the Authoriy for the price difference is that in the past there was only one vendor of the service. If required to dispose at the 106-mile site, the Authority estimates that the cost per wet ton will increase to $8.50. The Authority has been looking into other disposal alternatives and has been investigating different options involving private sector firms.

Many private sector firms consider the switch to the 106-mile site as a new business opportunity. They realize that communities are not equipped to handle the longer hauling distance. One dredging company undertook an intensive effort to poll ocean disposing communities to determine if it should enter into the sludge hauling and disposal business. The results led the firm to investigate the potential opportunity further.

PRIVATIZATION OF OCEAN DISPOSAL

From an economic perspective ocean disposal is already considered to be a legitimate and worthwhile business by quite a number of private sector firms; and privatization concepts can be applied to ocean disposal of sewage sludge. However, while privatization of ocean disposal is economically viable, from an overall perspective it is unlikely that privatization will be viable for ocean disposal. The economic viability is based partially on that quite a bit of the investment necessary for an ocean disposal project would be considered five-year depreciable property. For example, digestors, and digestors used as storage tanks, on-land transportation vehicles, barge loading equipment, barges and tugs would be considered five-year depreciable property. Terminal facilities would be considered structures and therefore would be depreciable over eighteen years. This potentially attractive tax posture, coupled with the overall business opportunity, could make ocean disposal a good economic candidate for privatization. However, the uncertainty with respect to environmental acceptibility associated with the disposal method makes the disposal method unattractive for privatization. The greatest uncertainty is associated with the remaining life of ocean dumping as a legal disposal alternative. The

United States Environmental Protection Agency is already on record as favoring an end to ocean disposal practices. It is unlikely that a private sector firm would be willing to commit the large amount of capital resources that would be necessary given this unfavorable climate. Even if the resources would be committed, the uncertainty also makes questionable the availability of long term service contracts, which are usually on the order of 20 to 25 years, and are necessary for the private firm to maximize the available economic benefits.

Privatization of ocean disposal is not likely to be undertaken if a firm will be required to make large capital investments. However, a firm that already owns the necessary equipment, but is not yet in the sludge hauling business, may take advantage of the business opportunity that could be offered through contract hauling arrangements.

A BUSINESSMAN'S PERSPECTIVE

Although privatization may not be applicable to ocean disposal of sewage sludge, there are still lessons to be learned from the approach that would likely be taken by a private sector firm.

A private sector or businessman's perspective looks to maximize the economic benefits of every resource. As in the case of sludge management facilities, where private sector firms are vying for the opportunity of using an end-product of sewage treatment as a revenue producing resource, a business man would not consider ultimate disposal until all possible value is exhausted.

Consider the approach, or turn around, experienced by Dupont. The firm had been dumping acid at the 106-mile site. Recognizing its value, the firm opted to recycle the acid into a saleable product.(4)

Based on economic benefits available, without judging its technical merits or the merits of other possible alternatives, ocean disposal of sewage sludge is a waste of economic and environmental resources.

Through sludge management techniques, it is possible for this "waste" product to be used for beneficial purposes. Among these uses are: as an organic fertilizer to enhance agricultural productivity; as a source of methane gas; and as land reclamation material.

As a closing point, it is interesting to note that a number of the most recent innovative technologies in sewage treatment consider sewage sludge a resource.

Technologies such as the production of sludge "bricks," the generation of electrical energy from a mixture of sludge and other fuels, and a new process known as "CCBA," which stands for Chemical and Coordinate Bonding and Absorption, both produce materials which have useful purposes. The CCBA process is a chemical treatment and recovery system. The end product is a ceramic pellet, which can be used as a filler in lightweight concrete, according to the process developer.(5)

In summary, future private sector approaches to financing waste disposal will consider a variety of possible uses rather than ultimate disposal.

NOTES

The author wishes to thank Ms. Sandra Mokuvos Bellush and Mr. Lanny Katz for their assistance in researching and contributing to this article.

(1) U.S. Environmental Protection Agency, Use and Disposal of Municipal Wastewater Sludge, September 1984.

(2) Goldman, Harvey and Mokuvos, Sandra, The Privatization Book, Arthur Young, 1984.

(3) Ocean Disposal Systems for Sewage Sludge and Effluent, Committee on Ocean Waste Transportation, National Research Council.

(4) Ocean Dumping, Procedings of a 1982 Ocean Dumping Conference organized by the New Jersey Sea Grant Extension Service in cooperation with The National Oceanic and Atmospheric Administration's Ocean Assessments Division.

(5) Dorn, Patrick, "Retired 3M scientist touts efficient sewage treatment," Rochester Minnesota Post-Bulletin, July 21, 1984.