Proceedings of

Non-point Pollution Abatement Symposium

Sponsors

U.S. Environmental Protection Agency
Great Lakes National Program Office

Wisconsin Department of
Natural Resources

Marquette University

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University of Wisconsin, Madison

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University of Wisconsin, Milwaukee

Metropolitan Milwaukee Sewerage
Commission

City of Milwaukee

Southeastern Wisconsin
Regional Planning Commission

German Marshall Fund of the
United States

Milwaukee, Wisconsin
April 23-25, 1985

NONPOINT POLLUTION ABATEMENT SYMPOSIUM

TECHNICAL, MANAGERIAL AND INSTITUTIONAL PROBLEMS AND SOLUTIONS

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Metropolitan Milwaukee Sewerage Commission
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German Marshall Fund of the United States

Symposium Organizers: Division of Continuing Education
Marquette University

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FOREWORD

Each year billions of dollars are spent in the U.S. and worldwide on sewers and municipal point source abatement. Yet, without limiting pollution inputs from both rural and urban non-point sources, the effect of the vast expenditures on water quality of streams, lakes and estuaries receiving point and non-point pollution loads may be only marginal.

Attacking the non-point pollution problem may not be an easy task. Sources are not clearly identifiable and the organization and institutional structure necessary to resolve the problem may not be yet in place.

The Nonpoint Pollution Abatement Symposium was held in Milwaukee, Wisconsin with the sponsorship of U.S. Environmental Protection Agency, Wisconsin Department of Natural Resources, Metropolitan Milwaukee Sewerage District, Water Resources Center of the University of Wisconsin and several other agencies. It attracted 145 participants from six countries (U.S.A., Belgium, Canada, Germany, Great Britian, and Italy) and provided a forum for the exchange of ideas among the scientists, engineers and policy makers on the problem of non-point pollution and its abatement.

The Symposium was divided into three programatic blocks: (1) State-of-the-Art presentations by invited leading scientists on the most important issues dealing with the identification of the non-point pollution problem and its solutions; (2) Panel discussions on technical, managerial, economical, and legal issues; and (3) Technical presentations and case studies. The proceedings contain all materials presented at the Symposium.

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WISCONSIN DEPARTMENT OF NATURAL RESOURCES
TO THE NON-POINT POLLUTION ABATEMENT SYMPOSIUM*

Welcome to Wisconsin on behalf of Governor Earl.

I apologize for the Governor's absence and convey his best wishes for a successful conference.

Governor Earl was my predecessor at the Wisconsin Department of Natural Resources (DNR) and as Secretary, he oversaw the beginning of Wisconin's non-point pollution abatement program. I commend his foresight and thank him for his continued budget support.

Wisconsin is a leader in non-point source pollution control, not only because of the financial and institutional commitments we have assigned to the task, but because we have tackled the problem head-on from an interdisciplinary standpoint.

The DNR is an integrated natural resources agency and brings to bear a variety of resources that are necessary to view the non-point issue in a total, holistic sense: fish, wildlife, water quality, toxics management, groundwater, forestry, water regulation, etc.

As important as non-point source control is, it is more important to view the issue from an entire environmental systems standpoint. The variety of disciplines represented at the conference and on the conference program is a good sign that the program's sponsors see the need for such integration.

It is also important to involve ecologists and others with "a broader view" in the non-point source decisionmaking process. This has not happened much in the past, with engineers, agriculturalists, and bureaucrats dominating program development and decisionmaking; but it is starting to happen. I want to encourage that.

It is appropriate that this significant non-point source pollution abatement conference is held in Milwaukee. Not only has Mayor Maier placed the issue high on his personal agenda, but the state is beginning a major rural-urban non-point source abatement program within the Milwaukee River Basin. Further, Milwaukee also has underway a major point source control program that, when meshed with the non-point program, will make the area an even more impressive showcase. Finally, numerous individuals have pointed to the important relationship between clean water and the region's and state's economic future.

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*Text of a speech delivered at the Non-Point Pollution Abatement Symposium.

This conference is taking place within sight of the Milwaukee River, which we hope will be a national model for non-point source pollution control within a rural-urban watershed. Why are the river and its clean-up important to Milwaukee, to Wisconsin and to the conference? The river should remind all within southeast Wisconsin that their economic, environmental and recreational fate is tied together. No one municipality or community can stand alone. The 830 square miles within the watershed reminds us all of the vastness and interconnectedness of our communities.

The river represents a means for people to come together, recognizing that divisive and unnecessarily bitter disputes in the past have cost the region time, money and lost opportunities.

The river--for that matter, water in general, as the Great Lakes Governors and premiers recently underscored with the signing of the Great Lakes Charter--represents significant, as yet not fully tapped, economic assets that can be enhanced and developed into more than a recreational asset. They can be an important selling point for the kinds of employees and employers who demand a quality working and living environment. They remind us of not only Lake Michigan, which we are protecting through our point source pollution abatement program; they remind us of the 84 smaller, inland lakes within the Milwaukee basin itself; all recreational assets.

For this conference, the River is important because it represents the complexity of issues facing us all in non-point source clean-up. Here are some important points that show how the River's clean-up so adequately represents the challenges we all face, no matter where we come from.

1. The benefits of the clean-up are difficult to quantify, yet we know they exist--in terms of better quality of life, increased economic opportunities and sustained agricultural productivity.

2. The technological challenges involved in the river clean-up are similar to challenges elsewhere. Rural, agricultural sources have been identified and must be addressed. Suburban sources exist and must be addressed. Urban sources have been identified and must be cleaned up.

3. The institutional challenges involved in the river's clean-up are as tough as you will find anywhere. They are rooted in years of disagreement over numerous non-environmental issues and recent disagreements over the sharing of the point source clean-up burden.

The vastness of the river basin and the multiplicity of the problems within the basin have pulled us apart as much as brought us together in the past. Examples of this are:

1. Non-point pollution blame has been heaped on upstream residents; while in fact all in the basin are responsible for the problem and all must be involved in the solution.

2. Many diverse communities live within the basin. There is suspicion and distrust among these communities, especially where there is a difference in economic standing.

3. Past political and economic wars--some involving litigation and harsh words--are carried over to the non-point pollution issues.

4. The misunderstanding and even mistrust that has often divided rural and urban, urban and suburban areas generally hangs over the non-point pollution issue, here in southeastern Wisconsin and elsewhere, too.

As difficult as the problem may now seem to Milwaukee and southeast Wisconsin there are ample signs to indicate that the job can be done.

In the point source pollution area, seemingly impossible tasks have been achieved. Some examples are:

1. The Wisconsin River in the central part of the state has experienced a greater than 90% reduction in biochemical oxygen demand in discharges from pulp and paper mills and public wastewater treatment facilities during the last 10 years.

2. Wisconsin has highlighted more than 50 water quality success stories in a special report to Congress on progress in the municipal construction grants program.

3. Wisconsin's most difficult point source pollution abatement program--right here in Milwaukee--is making good progress.

There has also been progress in the non-point pollution abatement program elsewhere in Wisconin.

1. In 1978, we became the first state to enact a statewide non-point pollution abatement program. A 25-year strategy was set to protect the 130 watersheds that need attention.

2. Since initiation, 26 watersheds have been identified as priority, and planning and protection funding is underway. In some watersheds, like the Hay River in northwest Wisconsin, we already are seeing progress in water quality improvement.

3. Complex intergovernmental and interpersonal hurdles have been cleared in not only identifying priority watersheds, but in implementing a complex watershed management strategy. Local government citizen advisors, federal agencies and other state agencies are involved.

4. While the Wisconsin program recognizes the individual responsibility of the landowner to be a respectful steward of the land, it also provides some financial assistance to private property owners so the rest of society can share in protecting a common public resource.

For Wisconsin, the nation, and the world, the need to wisely manage and protect our soil and water resources is a paramount issue.

There is no question that the non-point pollution abatement program is "water quality driven." It exists because of water quality laws and its

benefits are calculated in terms of cleaner water for fishing, swimming, general recreation and public use.

Another benefit of the non-point pollution abatement program is reduced soil erosion. Much has been said recently about the farmer's plight. If the farmers of our country lose their soil they won't be the only ones in trouble: we will be, too.

In Wisconsin total tons of soil lost from our fields has increased by 50% in the last 10 years. In 1982, some 67 million tons of Wisconsin soil was lost to erosion.

A third of Wisconsin's farmland is causing 85% of the soil loss. The places where soil conservation experts say the loss is the worst is in the southern part of the state: especially the hilly southwest, but also the gently rolling to flat southeast, including the Milwaukee River watershed.

The problem of agricultural soil loss is one that not only threatens Wisconsin, but the entire world and its ability to feed itself. Whether it be in Wisconsin, or on the Great Plains, or in drought-stricken Africa, soil erosion disaster is being courted by humankind.

Farmers are forced by the marketplace to make too many short-term economic decisions at a long-term cost to the land. Larger equipment and changes in farming practices have resulted in less over-winter cover, and a decrease in windbreaks and water retention practices. Row crops such as corn and soybeans are much more popular today than 10 to 15 years ago. Land planted to row crops is much more prone to soil erosion. Outside the United States, people in need of firewood comb the countryside for anything that will burn. The land is stripped of trees that hold soil and moisture in place, allowing deserts to advance.

Here at home, soil conservation at all levels has been starved for funding and regulations. The last great test of voluntary compliance is in the Chesapeake Bay region and I predict failure. (Not enough attention has been paid to the loss--in terms of food production--of our vital coastal wetlands and estuaries. This is another example of the potentially tragic, global food-production consequences if non-point source pollution is not brought under control.)

Each of us needs to speak out on society's obligation to curb soil erosion and the failure of local, state and national governments to engage the task simply because they are too short sighted economically or meek politically to do what is right.

The non-point source pollution abatement issue and soil erosion have some common problems and can draw upon some similar solutions in both rural and urban areas.

Politicians need to be the first to understand the economic as well as esthetic value of natural resource protection. In Wisconsin this is especially important because not only was our state built on its natural resources, but because our future may depend upon how well we steward and allocate them.

Politicians and all others must recognize that in natural resources, as in business, short-term gains or expendiency may lead to long-term disaster. American business has gotten into trouble by adopting a management strategy based on next quarter profit statements. The American people and especially Wisconsin people can get into a much more serious kind of trouble if they squander their land and water.

The watershed needs to be recognized and respected for what it is: a marvelous, interrelated environmental system that demands comprehensive management attention. Failure to understand the linkage among vegetation, soil, water, surface activity and human population pressure will delay or deny management success. The watershed must be viewed in a holistic fashion, with humans as a part of that whole.

You won't get something for nothing. And you can't build Rome--or clean up the Milwaukee River watershed--in a day. The degradation of America's land and water resources didn't happen overnight and those who have profited either directly or indirectly from the degradation have profited for many years. It will take time and money--your money and my money--to clean up the mess we have made for ourselves.

There are some of us who won't like others of us for getting the job done. In addition to spending funds to save our soil and clean up our water, we need regulations--standards of social behavior that--if violated--will result in unhappy consequences for the offender. The person--rural or urban, private individual or public official--who indulges in anti-social behavior should be called on it and pay the public price.

We need to be proud of our successes and brag to others about our accomplishments. Our DNR recently published a report on Wisconsin's natural resources, 15 years after the first Earth Day. It felt good to list our successes as an agency and as a people. The same should be true for all who take part in the effort to retain our soil and clean-up our non-point source water pollution. Why:

1. The taxpayers need to know that government works; in Wisconsin and elsehwere.

2. We need to praise and give recognition to those private property owners and other who pitched in and made personal and financial sacrifices.

3. We need to market the fact that Wisconsin has something that other states fall short on: good water, a quality living environment, and a farsightedness that tells investors, visitors and others that we're looking beyond the next quarter, we're looking to the next century.

I have several hopes for Milwaukee, for Wisconsin, and for the entire non-point pollution abatement effort, here in the United States and globally as well:

1. That through this conference and the actions of the State of Wisconsin we will be recognized nationally and internationally as a leader in non-point source pollution control and long-term wise natural resource management.

2. That each of you at this conference gains knowledge and renewed conviction that the challenge of non-point source pollution control is a challenge that goes far beyond improving the esthetics of our natural environment. It is a challenge that addresses the very ability of humankind to sustain itself through the wise protection and management of its life-giving soil and water.

3. That the federal government in Washington will take the issue of non-point source control as seriously as it took point source pollution control; and that all levels of government, as well as private landowners, immediately engage the serious issue of soil loss, especially on our prime agricultural lands in areas affecting coastal estuaries.

4. That the land ethic of Aldo Leopold, who last Saturday was inducted into the Wisconsin Conservation Hall of Fame, will be sought out and embraced by peoples of all lands at all levels.

In conclusion, non-point source pollution control--the caring for our soil and water and the respecting of the total interrelationship of all elements within our natural environment--is important to Milwaukee, Wisconsin, the United States and the world.

The world's attention has recently been focused on the human disaster in Africa where starving children have captured the hearts and moved millions of global citizens to assist in relief efforts. Yet the tragedy of Africa is as much an environmental tragedy as a human one and in that sense its enduring consequences will be felt long after the popular song--"We are the World, We are the Children"--has passed from the record charts.

The song goes like this: "There comes a time, when we heed a certain call ... when the world must come together as one. We can't go on pretending, day by day, that someone, somewhere, will make a change."

From the soil and water of the world comes food for the people of the world. To the extent that our soil and water are not wisely managed and are lost to humankind, our food and our survival are threatened.

History suggests that there are regions of the world, now deserts that once were gardens of plenty. For whatever reasons--natural and human-induced --the deserts are advancing, not only on other continents but here in North America. We must ask ourselves how long will we have abundant energy supplies to mask the consequences.

As the song goes: "There's a choice we're making, we're saving our own lives."

Keynote Reports

NONPOINT POLLUTION PROBLEMS FROM A PERSPECTIVE OF A MUNICIPAL GOVERNMENT*

Henry W. Maier, Mayor, City of Milwaukee

I am most pleased to welcome all of you to Milwaukee and this most excellent, high-level symposium on non-point pollution.

I want to thank Marquette University for sponsoring this event. And Dr. Novotny, thank you for your role and your efforts in planning and organizing this very important three-day symposium.

Last May another seminar on the same subject was held here in Milwaukee. It was co-sponsored by Dr. Edgington's University of Wisconsin-Milwaukee Center for Great Lakes Studies, the Milwaukee Metropolitan Sewerage District (MMSD) and the Mayor's Task Force on Sources of Pollution from Outside the MMSD.

I organized that symposium because I felt there was a pressing need to launch a network that would act as a springboard to get action on the problem. I was concerned that if significant steps were not taken to deal with non-point pollution coming into Milwaukee from up-river, that our own costly point-source pollution abatement program would amount to the civic folly of our times.

The MMSD has been operating under rigid mandates and timetables set down by the state, the federal government and the courts to clean up local water pollution. But, little or nothing was being done about up-river non-point pollution.

My concern was that after spending an estimated $2.5 billion on Milwaukee's point-source program that our waters would still be polluted.

Part of the problem is that non-point pollution is just not a very glamorous subject, particularly for the layman. I became doubly acquainted with that fact during the last presidential campaign. I appeared in New York with a group of mayors to interrogate the democratic candidates for president on their stands on various issues.

I posed to them a question on non-point pollution. Although the question had been stated in simple terms and the candidates and audience attending the forum had followed my line of questioning, reporters for the *New York Times* and *The Milwaukee Journal* later wrote that my question was complicated.

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*Text of a speech delivered at the Non-point Pollution Abatement Symposium.

Following this incident I decided that it was time to redefine non-point pollution. I came up with my own definition--unattended, up-river, up-stream pollution. Because, in terms of our own situation, that is what it has been--unattended.

It is unattended because significantly, it had not been attended to by anyone in the past. There were some faint efforts and little echoes related to the problem, but no prior significant efforts. It was unattended by the state, by the federal government, and the courts.

Not so for point-source pollution. Our own MMSD has had strict mandates and strict timetables to meet. It has amounted to a forced march on the problem at tremendous expense.

It is estimated that by 1996, the pollution abatement program of the MMSD will cost a staggering $2.5 billion, conservatively. City of Milwaukee taxpayers--in a city containing 30% of the state's poverty--will pay about $500 million of that cost. That is enough to fund 7 years of our entire capital budget. That includes schools, bridges, streets and alleys, sewer and water mains, etc.

So we asked: After all that money is spent, what will we have?

Will we have a sparkling, clean, unpolluted Milwaukee River? The answer is no.

Will the river be fishable and swimmable under standards called for by the Congress and the EPA? The answer is no.

Will we have finally ended any dangers of accompanying pollution related to Lake Michigan? The answer is no--we would not have done that sufficiently.

What we will have is pollution flowing down from the north, continuing to pollute the Milwaukee River; continuing to pollute the very waters we were spending $2.5 billion to clean up.

I saw the time had come to get attention to the problem. I had outlined to the Common Council in my Charter Message the previous year the extent of the problem. I warned that action had to be forthcoming from those with the power to mandate endorsement and set timetables against up-river polluters. My warning got little attention.

So I decided that a major Milwaukee symposium on unattended, up-river, up-stream pollution was necessary if we were going to get everyone into the act. I might add it appears that all you have to do to get hasty action out of the State of Wisconsin is to have the Mayor of Milwaukee call for a symposium on the subject. As quickly as you could say non-point pollution, the legislature drafted a bill designating the Milwaukee River Basin as a priority study area and ordering the development of a plan for action.

I had called on the state to develop a plan as far back as 1974, but that plea fell on deaf ears. So has my call for mandated action by the state on non-point pollution.

The prospectus which has been developed as a result of the recent state bill could provide a solid foundation for action on non-point pollution. It could, if it mandated clean-up. It could, if the state guaranteed funding. But, there are no mandates, no guarantees of funding. While the prospectus does set timetables for action paralleling those of the MMSD's point-source pollution abatement program the strategies for achieving non-point clean-up are strictly voluntary.

The voluntary approach simply is not good enough. Previous efforts to get voluntary compliance have had limited success. I will say more about that a little later. It is my contention that the problem will not be solved unless mandates and sanctions are imposed similar to those imposed upon Milwaukee's point-source program.

I do not suggest that those up-stream simply be penalized. Funds should be made available by the state to get the job done. We in Milwaukee are on a forced march. Why shouldn't those responsible for non-pont pollution be forced to clean up as well?

During our symposium last May, I suggested a way for the state to provide the needed funding. I said that the state should set up a special fund that would guarantee availability of the money to do the job. It would be a segregated fund like the Wisconsin Highway Fund. It would alleviate the problem of going back to the Legislature every bienneium and convincing legislators who might have more politically expedient uses for the money to provide the funds.

We are going to be working toward that end--to get legislation that will mandate action and provide sufficient funds to get the job done. I have no doubt that it's going to be a tough, uphill fight.

While my criticism of the prospectus is aimed at the voluntary clean-up approach recommended, the document is, as I said, sound in other ways.

And, I commend Kurt Bauer and his staff of the Southeastern Wisconsin Regional Planning Commission (SEWRPC), the Department of Natural Resources and its staff, and the members of the Advisory Committee which assisted them, for their efforts.

I might point out, to complete the study, they will need an additional $350,000 from the state between now and 1987. The State Legislature and the Governor should approve those funds. I urge them to do so.

In the meantime, our own city engineer Ed Laszewski, will continue to provide assistance as a member of the Advisory Committee to SEWRPC and the DNR.

I mentioned the symposium we held last year. That effort resulted in several very positive actions. A resolution was approved by the symposium participants that has led to establishment of a position for a District

Director for Non-Point Pollution by the MMSD. The new Director, Mr. Chris Potos, is attending this conference. He will be on board next month (June 1985) and will begin working to get action on the problem.

Secondly, at a meeting of the United States Conference of Mayors in Philadelphia last June, I sponsored a resolution calling for federal action that would mandate states to submit plans for non-point pollution control before the Senate and House which would achieve that. Further, the bills would provide $150 million a year for action programs. While that sum is nowhere near what is needed to get the job done, it would signify that the federal government recognizes the need. Unfortunately, the fate of those bills may be determined by the White House. It is my understanding that the Reagan Administration opposes funds for the program and will be working against the legislation. I would urge all of you attending this symposium to voice your support for the legislation and urge that the bills be strengthened.

The Conference of Mayors and a coalition of national organizations has taken a strong stand in support of federal legislation requiring state mandated clean-up. I have provided Dr. Novotny with copies of a February 21, 1985 letter from the coalition to Sen. John Chafee, chairman of the Senate Environmental Pollution Subcommittee. It outlines the coalition's position. I urge you to write to your senators and representatives in Washington supporting that position.

Let me discuss for a moment the state bill I mentioned before. That was the bill which led to the recent prospectus report on the Milwaukee River Basin. When the bill was passed, there was nothing in it addressing the need for timetables for unattended up-river pollution in areas where point-source pollution programs are underway or scheduled. I was pleased, however, to find that the prospectus does call for implementing non-point source control within five priority watersheds in the basin on a schedule matching that of the MMSD's point-source control program.

The state's designation of the Milwaukee River Basin as a priority area for the study leading to the prospectus was a direct outgrowth of a task force I appointed in 1982. The Task Force was charged with looking into the whole question of pollution entering the Milwaukee River from up-stream. It was to determine whether implementation of the massive point source water pollution abatement program by the MMSD, together with point source abatement programs, would produce streams and water courses in the greater Milwaukee area that were "fishable and swimmable." The Task Force concluded that completion of the MMSD's program would not by itself fully meet the objectives of producing "fishable/swimmable" surface waters.

As a result, the Task Force recommended the the MMSD petition the state DNR to designate the Milwaukee and Menomonee River watersheds as priority watersheds. It also urged the MMSD to seek to have the Governor and the Legislature provide the necessary funding for critical non-point pollution abatement projects, within a time frame matching the Milwaukee District's point-source program.

On May 8, 1984, just 3 days before the Milwaukee symposium, the Governor signed the bill into law designating the Milwaukee River Basin as a priority

watershed. As I said before, my primary concern with the bill and the study is that the action program contemplated is one based on voluntary, rather than mandatory compliance.

The state started a program of voluntary compliance in 1978 and its own audit of that program, released in January 1984, indicated that the program had only partial success in getting polluters to participate. It pointed out that in some watersheds participation levels of upstream polluters were very low, making it extremely difficult to achieve water quality improvements. And, even in those areas where participation levels were higher, the audit report noted that nonparticipation of just a few landowners in critical areas can negate the program which the majority is carrying out.

That is why I have been against the voluntary approach. It simply will not work. And again I repeat: It will be a tragedy to spend all the money that is being spent on the District's point-source pollution abatement efforts if up-river pollution is still flowing down to us after the District's program is completed.

Let me quote from the February 1985 issue of the *Journal of the Water Pollution Control Federation*:

"It is critical that the costs of not addressing non-point source pollution control be considered. Available data clearly show that the effects of non-point source pollution are diluting the return of the billions of dollars invested to date to control point sources ... non-point source pollution control is a substantial barrier to achievement of the goals of the Clean Water Act."

So, while we and others are paying through the nose to build, operate and maintain point-source systems, non-point pollution is negating our efforts. In our own case, up-river polluters have the nice option of "voluntarily" installing pollution abatement measures while we operate under rigid timetables and mandates.

And, if they volunteer, they will be doing it at an estimated 3% or less than the cost of our program, even though they're responsible for about 50% of the problem. Those estimates are based upon SEWRPC data contained in the Mayor's Task Force Report. The mismatch is as obvious as the inequity. There is no doubt that this kind of thing is being repeated in a number of metropolitan areas throughout the country. That is particularly true where cities under clean-up orders have pollution coming in from rivers and streams that cross jurisdictional boundaries.

What is needed is strong federal legislation requiring the states to force compliance with Clean Water Act provisions by up-river/up-stream polluters. Without it non-point pollution will not be adequately controlled or abated. In the meantime, those of us under orders, timetables and mandates will continue on the forced march to point-source clean-up at a great cost.

As for non-point controls, the big question is whether the legislature would be willing to commit the funds necessary to do the job, even on a voluntary basis. It has been roughly estimated that it could cost between $15 and $20 million. Frankly, I don't know if those are realistic figures.

Further, there are no funding guarantees. There is a draft plan. We will provide whatever support we can to those responsible for seeing it through.

In the meantime, the symposium should bring added attention to the problem and those dealing with it. And, I urge all of those attending to read Dr. Novotny's and Dr. Gordon Chesters' definitive work on the subject, *Handbook of Non-Point Pollution*. It is an excellent book. In it, they cite the magnitude of the problem, pointing out that nearly half the polluting sediments--nearly 2 billion tons--result from runoff from agricultural lands.

I might add that the head of the Conservation Foundation, a national organization that has been trying to persuade Congress to take action on non-point pollution, has stated that the economic costs of rural soil erosion totals between $2.5 and $3 billion a year.

And, the latest federal Soil Conservation Service report shows that in Wisconsin about one-third of the land in the state contributes 85% of the erosion. Our own studies have shown that tons of mud laced with pollutants flow down from up-stream and pollute the Milwaukee River and harbor estuary. Professors Novotny's and Chesters' book also contains some very thoughtful recommendations and strategies to deal with the problem. I am sure you will be discussing some, if not all, of them during this symposium.

Again, I want to commend Dr. Novotny and Marquette University for sponsoring this significant forum, and I wish you every success in your endeavors.

Magnitude and Variety of Problems of Non-Point Pollution from Agricultural Practices

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After about 20 years of exposure to the ideals of environmentalism, modern society now demands that soils be fertile and that waterbodies and waterways be oligotrophic, that is relatively infertile. In practice, these goals are incompatible. The thermodynamic forces of entropy in nature dictate that nutrient species present in or applied to land, being at a higher potential energy than lower-lying receiving waters, must gravitate toward the lower energy regimes of streams, rivers, lakes and groundwater.

When a chemist looks at chemical species and chemical reactions, she/he applies known physical constants to determine the amount of "free energy"--the potential chemical energy--locked up in a particular chemical compound. Whether this energy can and will be released is determined by the availability of chemical reactions to lead from this compound to products of lower free energy. Sometimes there is no way for such transitions to happen; sometimes there is only one route, sometimes there are many competing routes, sometimes there are many intermediate steps of progressively lower energies along any one reaction pathway. Often compounds of extremely high energy content do not lose their energy for lack of a convenient pathway to follow. In such cases, the chemist applies catalysts to achieve the desired conversion, either because she/he wants the reaction products or desires to use the chemical energy. The catalysts produce a reaction where none would otherwise occur or speed up otherwise slow reactions. An example of this is the oxidation of the high-energy aromatic hydrocarbon solvents benzene, toluene or xylene. Without catalysis, these compounds are stable and remain unchanged. In soils or water, they are slowly degraded by microorganisms using oxidizing enzymes as catalysts. Here the release of their energy is so slow and gradual that it is not particularly beneficial to the organisms which accomplish the reactions. They are concerned more with the destruction of materials which are toxic to them. However, when these compounds, as constituents of petroleum fuels, are burned in furnaces or internal combustion engines, the release of their energy is fast and furious and can be applied to useful purposes: heating of homes, generation of electricity or driving of engines.

Conversely some chemical reactions occur spontaneously to convert high-energy to low energy products because of the ready availability of convenient reaction pathways. Rusting of iron is an example. If undesired, such reactions can be retarded by use of anticatalysts. Good examples of this phenomenon are the use of nitrite to retard spoilage of meat and of antioxidants to retard perishing of rubber and spoilage of food. Car tires last much

longer nowadays since chemists recognized that ozone was the cause of cracking of rubber and manufacturers started putting antiozonants into rubber.

Pollution of water by contaminants from nonpoint sources follows the same laws of thermodynamics as chemical reactions. Of necessity, contaminants on upland sites have the potential to move to lower energy states in surface and groundwaters. The only aspect of this process with which man can interfere is in control of the kinetics--i.e., the rate of transport--by selecting agricultural practices which in effect act as anticatalysts to contaminant movement.

With this general physicochemical concept in mind, it is important for the soil conservationist/water quality expert to recognize this natural law and to combine forces to determine what are the "reactions" which are of greatest qualitative and quantitative concern for the preservation of highly productive fertile soils and clear, pristine surface waters or safe, uncontaminated groundwaters, and to decide what measures will function as good "anticatalysts" to retard spontaneous pollution reactions. In these deliberations, it is important to divorce scientific thought from the emotional, knee-jerk or gut-feeling approaches to water quality of the past. We must consider not only the unaesthetic appearance of eutrophication but less readily visible toxicological concerns. Some lakes--though pristine in appearance--may harbor greater threats to human health and the environment in terms of excess acidity or difficultly detectible but biomagnifiable concentrations of pernicious microcontaminants such as pathogenic bacteria and viruses; toxaphene and other pesticides; volatile organic compounds; polychlorinated biphenyls and dibenzofurans; or toxic metals, than lakes which look dirty and polluted. Abundance of algae blooms may be a sign of a healthy lake! Only by alloting appropriate emphasis to these more important emerging concerns can scientists expect to generate the social pressures that will lead to fresh infusions of funding to achieve desirable water quality goals.

Past efforts have largely curbed pollutant inputs from point sources and done much to reduce sediment and nutrient loadings from nonpoint sources to surface water and produced readily visible water quality improvements. We have made great progress toward ensuring that the nation's waters are becoming "fishable and swimmable." Concern must now be refocussed on whether it is safe to swim in these less murky waters or to consume the fish we harvest from them. The alliances that led to the first round of water quality improvements--those with groups or agencies that sponsored primarily control of sediment and phosphate loadings--must be broadened to encompass those that espouse techniques to control more insidious pollutants. Only in this way can we expect to effectively establish pollutant control priorities and set pollutant standards and criteria in a timely manner. When directed toward agriculture, nonpoint source pollutant control programs must couple cost-effective means of minimizing water pollution with least disruption to efficient farming practices and to production of high-yield, high-quality agricultural crops. This approach will necessitate close interaction among chemical industries, federal and land-grant university research, and state and county extension services.

Fortunately most of the infrastructure for this collaboration is already in place and is operating. Just as necessary and perhaps more difficult to achieve will be an open-minded receptiveness on the part of corporate and private farming enterprises to take steps to achieve improved water quality through modification of their farming practices, especially during this period of financial stress in the realm of agricultural production. It is up to the ingenuity of the agricultural research and agribusiness communities to develop ways to make both goals compatible. Great strides have already been taken in this direction, fortuitously dictated by other circumstances, such as the energy crisis of the early seventies: the increase in costs of tractor fuel and fertilizers resulting from the escalation of the prices of petroleum products provided a greater incentive to the implementation of conservation tillage than the loftier but more esoteric goal of water quality enhancement through sediment and nutrient control. Who cares? The fringe benefit of efforts to reduce energy inputs to agriculture was tillage practices that do such a good job of soil conservation and receiving water protection that they have become the cornerstone of "best management practices" for nonpoint pollution control.

Thus, efforts to protect water from nonpoint pollution from agriculture must adopt a forward looking attitude based not on the past, but on the current state of agricultural practices. To continue progress in water quality protection from nonpoint sources, we must try to assess where we stand at the moment and anticipate where we go to from here.

Agricultural pollutants from nonpoint sources enter the environment in a diffuse but pervasive manner. The sources can be land-based or airborne and the pollutants can be transported in particulate, dissolved or volatile forms. They can be naturally-produced or man-made components. Because of their diffuse distribution they are difficult to isolate and often appear in low concentration, making their control a complex task. The fluxes of contamination from land to water vary widely in space and time, depending primarily upon individual or sequences of meteorological events, so that pollutant loadings associated with isolated farming activities, derived from whole or individual portions of watersheds, or introduced by long-range transport from remote geographic areas become extremely difficult or impossible to quantify. Vast data bases of information collected from laboratory, field plot and field experiments and watershed monitoring studies have been assembled and a variety of clever mathematical models generated to evaluate such data to aid in describing pollutant delivery systems and predicting benefits to be derived from their manipulation. Yet the reliability of these efforts is still too questionable for them to convince the public and their elected representatives to subsidize measures which would convert scientists' fears, concern, and suspicions into action to support curtailment of continued water quality degradation.

In these days of massive federal government deficits, the movement to support water quality protection is running out of steam. Section 208 of the 1972 amendments of the Federal Water Pollution Control Act provided funding for the development of mananagement plans to curtail water pollution from nonpoint sources. While this planning effort was successfully completed the follow-up implementation languished for lack of federal funding. In only a

few cases, e.g., Wisconsin[1] and Maryland[2] did state government jump in to fill the void by creating state-supported nonpoint pollution control programs. Maryland understood better than most states through effective public relations and lobbying efforts to parlay their concerns into a $27 million federally funded program of research on the Chesapeake Bay estuary, which it followed by a $36 million state program to curb pollution from both point and nonpoint sources. In constrast, little has been done to utilize the findings of the PLUARG (Pollution from Land Use Activities References Group) studies and plans to implement measures to protect the Great Lakes. The time is ripe for the Great Lakes States to move forward in unison from the concerns of the past with regard to agricultural nonpoint pollution--mainly BOD, bacterial, sediment and phosphorus loadings--to those of the present and future, mainly toxic metals, acidic deposition, pesticides, and organic supertoxicants such as PCBs, PCDDs (dioxins), and PCDFs.

The literature is replete with quantitative and qualitative information on pollutant losses from agricultural activities on a variety of scales--from field plot experiments through subwatersheds and watersheds to regions and the nation. It would be futile to try to summarize this information here. A compact summary is already available in the literature[3]. The nature and quality of the data are highly variable, but in summary the state-of-the-art is convincing enough to specify certain problems and to reveal that some measures being implemented to mitigate nonpoint pollution are indeed working effectively. For example, conservation tillage has obviously reduced sediment loadings from agriculture. Rather than recite a litany of recognized pollutants, and the successes achieved in managing them, I feel it is more advantageous to examine what problems lie ahead. Most important questions are whether we want more of the same, or whether we should redirect our attention to new or emerging problems, and in either case, who is going to foot the bill for research, development, planning and mitigation programs. We must start to examine agricultural operations related to water quality from new perspectives in the light of more recent scientific findings. A good way to do this is be using a few of the prominent agricultural pollutants as examples.

Concern with soil loss preceded concern with water quality deterioration by sediment and any nutrients or pesticides associated with eroded particles. Measurement of erosion rates on small plots of cropland, although perhaps not a good model of overall erosion potential because they neglect the long-distance delivery ratio, do provide a means of pinpointing "hydrologically active areas," i.e. the areas most sensitive to constant rill and sheet erosion. It has been calculated that half of the soil loss in 1977 occurred by water-induced rill and sheet erosion from only 10% of the cropland. Such land was located mainly in Mississippi Delta states (AK, LA, MS), the Southeast (AL, GA, SC, FL), Appalachia (KY, TN, WV, NC) and the Corn Belt (IA, MO, IL, IN, OH)[4]. USDA National Resource inventories for 1977, as revised in 1980 and 1984[5] indicated that the national average erosion rate for row crop and small grain cropland was 5.4 tons/A/year. Wind erosion increases this average to 7 tons/A/year. The average rate of soil formation is placed at a mere 0.5 ton/A/year. The same data sources indicates that the "T value"--a widely accepted criterion of 5 tons/A/year considered as a desirable objective by the Soil Conservation Service and EPA--is exceeded on over 112 x 10^6 acres of cropland, including 33% of that in corn, 35% of that in cotton, 39% of that

in sorghum and 44% of that in soybeans. Clearly soil erosion is a national problem with emphasis in particular regions.

Using values for the bulk density of soils, it can be calculated that a soil loss of 5 tons/A/year leads to loss of 1 inch of topsoil in 30-40 years, depending mainly on topsoil texture. This may not seem like a significant rate to a farmer in Iowa growing corn on 20-60 feet of loess, but it is to farmers in the driftless area of Wisconsin or in Appalachia trying to eke out an existence on low fertility thin soils over high bedrock. These latter areas are those that would profit most from erosion control measures, but present funding to curb erosion is not allocated by prioritization. It is distributed equably among states and regions, rather than by need or the prospect of deriving the maximum benefit--i.e. the biggest bang for the buck. In any case, there is insufficient funding for adequate programs of soil conservation programs anywhere!

It will take imaginative new programs to provide appropriate funding for soil conservation research and implementation of conservation measures in a prioritized plan. One good way to achieve this would be to follow the model of the Comprehensive Environmental Response, Compensation and Liability Act of 1980 and create a sort of Soil Conservation Superfund. Although nominally it is present-day chemical manufacturers who pay for cleanup of orphaned chemical dumps through donations to Superfund, regardless of who the original dump operators were and which companies deposited chemical wastes in them, in fact it is the general public which is footing the bill indirectly through higher prices for the chemical, textile, petroleum and other products we use in everyday life. It seems appropriate that a Soil Conservation Superfund should be underwritten directly by the consumer, in the form of a small checkoff tax--say 1%--on all food products. Such a tax would be shared equably among the beneficiaries of agricultural production--everyone according to his/her means. Whoever buys choicer cuts of meat and better quality fruit and produce pays higher prices and consequently more Soils Superfund tax.

Regrettably there are barriers to implementation of such a scheme. While it would provide technical means to reduce soil loss and water pollution by land erosion, its social and political acceptance creates a "thermodynamic" barrier. Only interaction between physical and social scientists will develop the catalysts needed to provide political decision-makers with the means to provide the "activation energy" to make this reaction work.

Fortunately we are at a point in the recognition of serious water quality problems associated with materials other than sediment or phosphorus that may help this reaction to go in a series of steps.

Agriculturalists have some tendency to try to justify the erosion loadings from their land--or at least to place them into perspective--by comparing them with erosion from construction sites, estimated at about 50 tons/acre or 85 million tons annually from areas disturbed for home construction (80,000 A) combined with highway construction and other civil engineering activities (1.6 million A). This is actually small compared with 413 million acres of cropland delivering an average of 5 tons/A/year for an annual total of 2.65 billion tons annually. A bigger concern than the quantitative

comparison is the fact that the eroding cropland soils are more heavily laden with nutrients (mainly nitrogen and phosphorus), pesticides (mainly herbicides) and, if manure and sewage sludges have been applied to the soils, toxic metals and bacteria. Even in areas of relatively flat topography where little erosion normally occurs, or where wetlands act as sinks or sediment traps for these materials, flooding or violent storms can catalyze a violent reaction and transmit pent-up pollutants with a scouring velocity to receiving bodies.

The impact of sediment, or even of eutrophication-controlling phosphate may dim in importance compared to the effects of toxic contaminants from nonpoint sources. Sport fisheries in many inland lakes and the Great Lakes are now suffering because prize catches are unfit to eat. The EPA recently reduced the tolerance for PCBs, i.e. the concentration considered to be acceptable from a health safety standpoint, from 5 to 2 parts per million, defining that most Lake Michigan game fish are risky to consume. Mercury levels from unknown sources in some Wisconsin lakes are so high that anglers have been warned off. PCBs in lake trout from Lake Superior are above the standard, fish from the Saginaw River, Saginaw Bay, and parts of South Lake Huron are polluted with PCBs, PBBs and dioxin. A fish consumption advisory is in effect for Lake Ontario because of mirex content--although this came from industrial rather than nonpoint sources.

Being at the bottom of the thermodynamic progression, bottom sediments in the Great Lakes are apparently becoming the ultimate sink for many toxic metals and lipophilic ("fat loving") contaminants such as PCBs, TCDD, PCDFs, DDT and toxaphene, which tend to bioconcentrate in aquatic food chains. Although these compounds are extremely insoluble in water, their vapor pressures are high enough for them to enter lakes mainly through the atmosphere at a slow insidious rate. In lakes they become associated with the organic fraction of suspended particulate matter and microbiota (phytoplankton and zooplankton) consumed by higher life forms and ultimately reach fish and man. The best safeguard against long-term in-lake recycling of these compounds is their burial in deep bottom sediments. For this reason it would be unwise to strive to completely eliminate sediment loadings to lakes, curtailment is a cheaper and more prudent option from the standpoint of toxicant immobilization. Inactivation by sediment burial is the best strategy for in-lake control of toxic metals as well as of supertoxic organics.

The guiding principle must be selectivity in the sediments admitted to lakes--they should be those which will settle in the deepest sinks and which do not introduce higher loadings of priority pollutants than they scavenge. Refinement of these concepts will take further fundamental research. Luckily we now have good models for the transport of many pollutants. Acceptance by the public, regulatory agencies and decision-makers of the concept of a tolerable "beneficial" sediment loading will require a good selling job that physical scientists seldom perform.

Finally some deliberations about groundwater pollution. Nitrate pollution of groundwater is a natural phenomenon that is greatly exacerbated by modern use of synthetic fertilizers, especially in areas of high groundwater tables underlying coarse-textured soils or fragmented or creviced bedrock. Irrigation in such areas can compound the problem. Here use of a

tolerance--EPA's primary drinking water standard of 10 mg/L of nitrate nitrogen--has served to eliminate consequences, even if the risk *per se* remains undiminished. Where the standard is exceeded, parents are advised to provide water from nitrate-free sources to infants--the only population at risk. This strategy has been effective: cases of nitrate toxicity are so rare that they warrant publication in public health journals[6]. We have just conducted a survey of all pediatricians and general practitioners in Wisconsin. Only 20 of them reporated ever having seen a case of infant methemoglobinemia, none of them lately.

In this case, fundamental research coupled with toxicological data and regulatory guidelines have effectively eliminated the consequences of the health risk. The same approach can be used to cope with emerging problems of pesticides and volatile organic compounds in groundwater. Some basic principles are worth considering in this connection. The only pesticides that will leach through soils are those that have high water solubilities and low affinities for sorption onto soil particles. Most pesticides do not exhibit these properties. That is why there was formerly no requirement for leaching studies as part of the environmental fate information required by the EPA for pesticide registration. Pesticides were not expected to wash through the soil with percolating water; indeed the earlier generations of lipophilic pesticide compounds did not.

Only after the episodes of incursion of aldicarb residues into groundwater on Long Island and in Florida, Wisconsin and a few other states did information on pesticide leaching become part of the standard registration protocol. Actual leaching experiments need not be conducted nowadays; on the basis of chemical and physical properties of the active ingredients, accurate predictions can be made as to whether pesticides will leach or not under conditions of normal agricultural use. Advances in instrumentation for chemical analysis of pesticides in water have made it possible to detect extremely minute concentrations of pesticides in groundwater. Depending on the compounds concerned and the sophistication of the method and detector system used, levels as low as parts per billion or parts per trillion can be accurately quantitated. The EPA created short-lists of compounds they considered likeliest to leach. Recent surveys and monitoring of groundwater using reliable methods to look for these compounds have not revealed evidence of extensive contamination of groundwater or field drainage with significant levels of pesticides, other than aldicarb residues and minor traces of other compounds. To appreciate the significance of these findings, toxicological data must be considered.

Aldicarb residues in groundwater generated concern mainly because aldicarb has such a low LD_{50} (the dose lethal to 50% of laboratory animals in acute toxicity tests): about 1 mg/kg of body weight in rats. Fortunately aldicarb residue do not exhibit any mutagenic, teratogenic, carcinogenic or other chronic/subchronic toxic effects. A "tolerance" of 10 parts per billion has therefore been set for groundwater; this is a nonbinding "health advisory" suggested by EPA's Office of Drinking Water. New York adopted a level of 7 ppb based on a value suggested by a National Academy of Sciences[7] panel before the pesticide was actually found in groundwater.

Aldicarb is in effect one kind of worst-case situation: it has a high water solubility, is poorly sorbed by mineral soils and has a low LD_{50}. Fortunately, most of the pesticides used in agriculture do not share these features: they are less likely to leach, and their acute toxicities are weaker. Whether a pesticide is toxic to man or wildlife depends largely on its mode of biochemical action. Chemists try to design pesticides which are highly toxic to pests but not to desirable plant and animal species. They therefore exploit the differences in anatomy and physiology of the species of concern. Insects are different enough from vertebrates that many pesticides specifically toxic to insects but relatively innocuous to man and animal, bird and fish species can be created. The differences in physiology between higher animal species and plants or fungi and molds are even greater. Consequently in general herbicides and fungicides tend to be much less harmful toward man, animals and fish than insecticides.

Herbicides are the most commonly used pesticides: nationwide, farm use in 1980 was 445 million pounds versus 306 million pounds of insecticides, and 95 million pounds of fungicides, rodenticides, and other pesticides. Farm use was about 72% of total use. Expanded use of conservation tillage practices tends to increase use of herbicides and fungicides--the less hazardous pesticides. Expanded use of integrated pest management is tending to reduce all pesticide use, but especially that of insecticides, the most hazardous poisons. Banning of most of the persistent organochlorine insecticides, DDT, aldrin, dieldrin, hexachlorobenzene, lindane, etc. and some soil fumigants has stimulated industrial research for new, more selective, superior pesticides. Some herbicides are presently under development which promise to be effective at application rates which are only a fraction of those used with the current products. Thus, water quality problems with pesticides are expected to decrease. Only those with the organochlorine "antiques" remain a major continuing threat. Ironically, although banned in most western industrialized nations, the organochlorine pesticides, with DDT at the forefront, are still the most widely used insecticides worldwide. They are cheap and easy to manufacture and are therefore still popular in developing countries. Long-range atmospheric transport is therefore likely to continue to add organochlorine pesticide loadings to U.S. water bodies but at much lower rates than occurred during and following their use--and overuse--in the USA.

DDT was banned, not because of toxicity toward humans, but because of its chronic toxic effects on widlife, particularly bird reproduction. A strong stimulus for the reappraisal of the environmental effects of organochlorine pesticides such as DDT came from important books such as Rachel Carson's "Silent Spring." Reproductive problems in mink fed on mink farms with Great Lakes fish, once wrongfully ascribed to DDT contamination in the fish, led to recognition of the chronic toxicity of PCBs, hitherto considered harmless because of their low acute toxicity. High incidences of crossbill malformations in herring gulls, cormorants and terns around Green Bay appear to be associated with PCBs or other organic supertoxins (e.g. TCDD or TCDF).

Episodes like these have led to close scrutiny of the chronic toxic effects of pesticides. Continuing tests of chronic toxicity may lead to some surprises. One is already happening. It has been known since the middle 70's that traces of pesticides occurred in cropland runoff. For example,

measurements in Western Iowa[8], revealed average losses for the years 1973-75 of 0.21, and 0.15% of the organophosphorus insecticide fonofos applied at a rate of 2 lb/A in the sediment and water fractions of runoff under conventional, till-plant and ridge-plant tillage systems used for continuous corn. Corresponding losses of the triazine herbicides cyanazine and atrazine were 0.28 and 1.81%. Losses of alachlor carboxyanilide herbicide popular for use in corn was 0.19 and 0.77% in the sediment and water. In terms of the acute toxicities of these compounds, one need not be concerned with these minor amounts. Greater concern should be with the massive loss of topsoil (a silt loam from 12-18% slope) observed, equivalent to about 23 and 10 tons/A for the first two systems. But now continuing tests have shown for the first time that alachlor causes cancer in some strains of mice. Though alachlor has a solubility in water of only 33 mg per liter versus 33 g per liter for aldicarb, meantime alachlor has also been detected in groundwater at two sites in Wisconsin (probably not as a result of field use) and in samples of tile drainage in Ohio. Canada is presently taking action to ban alachlor use.

Obviously, in the light of this new information, the use of this pesticide must be reexamined. EPA is subjecting it to a "special review." Is it likely to go the way of DDT? Not necessarily. Better understanding of the mechanisms of carcinogenicity are leading to cooler appraisals of cancer risk, so that banning of useful products is not an automatic consequence of positive results in animal tests. The Delaney Clause, which once banned use of any food additive which produced any cancer in any test species did not hold up in the case of saccharin as a result of public pressure for its continued use. The case was made that the benefits of the compound far outweighed its minute cancer risk. A popular analysis of national policies in cancer research and prevention[9] forms an antipode to "Silent Spring" and documents justifications for the new regulatory approaches of risk assessment and risk management. The Office of Science and Technology Policy recently published its report on the guiding principles to be used by all federal government regulatory agencies in dealing with carcinogenic risk [10].

USA policy is therefore developing in a more objective scientifically considered manner independent of Canadian policy and political pressure. In the past, Canada has strongly influenced USA policy with regard to phosphate loadings to the Great Lakes. It is lobbying strongly to influence decisions concerning sulfur dioxide emissions in the USA. No doubt it would like to see its actions on alachlor duplicated in the USA. This may not happen. However, the alachlor situation does seem to justify more intense efforts to control nonpoint pollution from agriculture.

If nonpoint pollution from agricultural sources is to be further reduced, there has to be funding for the effort and this must be clearly justified. It will be necessary to reappraise the goals, and to return to my thermodynamic model to determine what compounds, what reactions and what reaction products are of interest, which processes need catalysts, and which ones need inhibitors. Priorities must be decided. Which loading is more important: 5 tons of sediment, versus 20 pounds of nitrogen, versus 1 pound of phosphorus, versus 0.1 pounds of herbicide, versus 0.05 pounds of insecticide, versus 0.01 pounds of cadmium, versus 0.001 pounds of PCB, versus 0.0001 pounds of dioxin, versus 10^6 bacteria, or 10^5 PFU (plaque-forming units--a measure of viruses) per

acre? Use of sediment and phosphate loadings as a management tool in policy-making was useful in the first round of water quality protection from nonpoint source contaminants. Now is the time to review and realign our efforts and substitute the uncertainties of the present by a new "chemistry" of the future--a coalition of physical, biological and social sciences that will make reactions happen.

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MAGNITUDE AND PROBLEMS OF NONPOINT POLLUTION FROM URBAN AND URBANIZING AREAS

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ABSTRACT

Pollution generated from urban and urbanizing area surfaces can be a significant portion of the total pollution entering a region's receiving waters. These nonpoint sources of pollution typically are extremely variable in character, as they are driven by rainfall and snowmelt events of variable magnitude. Pollutant deposition and erosion rates are similarly variable.

Sources of contaminants contributing to urban nonpoint pollution include street surface dust and dirt, atmospheric fallout, construction activities, vegetative matter, chemicals from various uses, and miscellaneous sources such as rooftops and channel erosion. These contaminants result in pollutant concentrations and loads that often exceed wastewater treatment plant outputs and recommended guidelines for good quality water. Receiving water biota can be severely impacted by urban runoff, and groundwater can be impacted by the introduction of chemicals at above-background levels. Another impact often overlooked is the physical destruction that results from increased amounts of urban runoff; such destruction includes increased channel scour and erosion, flooding, devegetation, sedimentation and debris deposition.

Efforts to properly address urban runoff must strive to adequately identify the problem and its causes before any management can be effective. Management efforts should focus first on developing areas where maximum pollution controls can be most easily implemented on sites where they are most needed.

Keywords: Urban runoff pollution, nonpoint pollution, stormwater runoff quality, construction erosion.

INTRODUCTION

Defining the problems resulting from urban runoff is an elusive task, requiring that an "impact" be seen. In general terms, an impact is seen if there is a loss or curtailment in the beneficial use of a water body. A beneficial use, in turn, is a local perception and could include such things as body contact, aesthetics, recreation, fishing, domestic or industrial consumption, or boating. The task is elusive because it requires describing a phenomenon that is a function of several extremely variable inputs (precipitation, surface area pollutant deposition, soil erosion, complex overland and channel hydrology, pollutant transport) that might have a measurable impact on a receiving water.

This paper is a cursory review of literature that has in some way defined the pollution character of urban and urbanizing area runoff. The effort relies quite heavily on the results of the recently completed U.S. EPA Nationwide Urban Runoff Program (NURP), on various NURP site studies and on water quality monitoring results from Metropolitan Council efforts in the Twin Cities region of Minnesota. The actual documentation of "problems" from urban runoff is scarce because it has only recently been recognized as a problem, meaning little good data exist; the effects are not well understood and are often masked by point source pollution; the term "impact" is not universally agreed upon; and receiving waters are not well defined (Heaney and Huber, 1984).

The best approach for addressing the topic of nonpoint pollution from urban runoff is to adopt a planning strategy that consists of defining the problem, determining the source of the problem, exploring methods for managing the problem, and evaluating the results of the management program to assure progress and make adjustments. The information that follows will assist water managers with the first two of these steps.

FACTORS CONTROLLING URBAN NONPOINT SOURCE POLLUTION

Runoff quantity

The mechanism for mobilization of pollutants in an urban environment involves the generation of runoff from rainfall or snowmelt at a level with sufficient energy to suspend or solubilize pollutants. The total load or mass of a pollutant in runoff is the product of the pollutant's concentration (mass per unit volume) times the flow volume carrying the pollutant. Research into the dynamics of urban nonpoint pollution loading has shown that typically the flow factor plays a dominant role in loading behavior because it is the driving force in the mobilization of pollutants (Bannerman et al., 1983; Colston, 1974; Konrad et al., 1978; Oberts, 1982; U.S. EPA, 1983). Concentration values might be as variable as flow, but loading seems to mimic the pattern established by flow, as shown in the load-hydrograph in Figure 1. This behavior points very clearly to the need to undertake comprehensive surface water management as a first step in urban/urbanizing area nonpoint pollution control.

The approach toward control of urban runoff has undergone a large change in the past 30 years. Formerly the method used to handle surface water runoff was to capture it and remove it as quickly as possible. This approach, however, led to many related problems downstream. A graphic prepared by Tourbier and Westmacott (1974) (Figure 2) shows how runoff becomes increasingly significant as the amount of urban development (that is, paved surfaces) increases. Some of the washoff from impervious, paved surfaces drains to pervious areas or to

areas of closed drainage that do not contribute to any receiving water. The most important factor in loading, then, has been found to be the total "hydraulically connected" or "effective impervious" area, rather than the total impervious area (Bannerman et al., 1983; Miller and Mattraw, 1982; Pitt and Bozeman, 1982; Sonzogni et al., 1980).

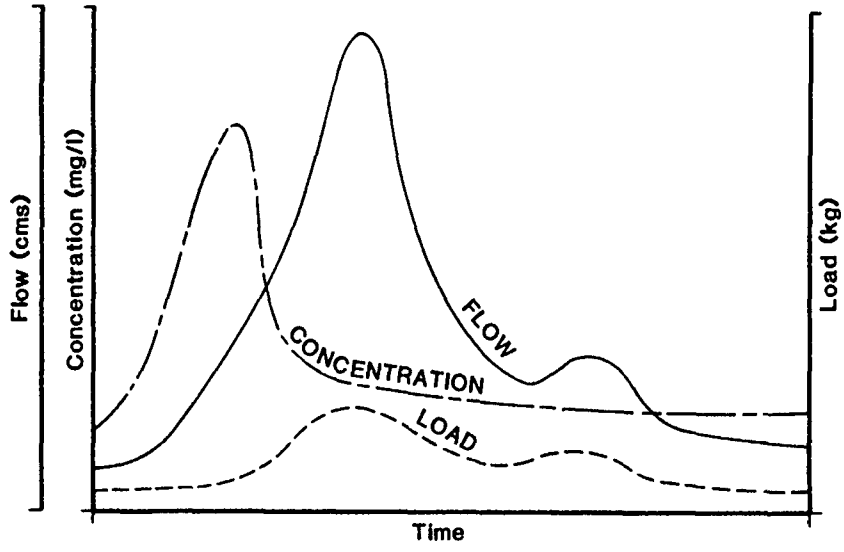

FIGURE 1. FLOW, CONCENTRATION AND LOAD BEHAVIOR DURING A RUNOFF EVENT

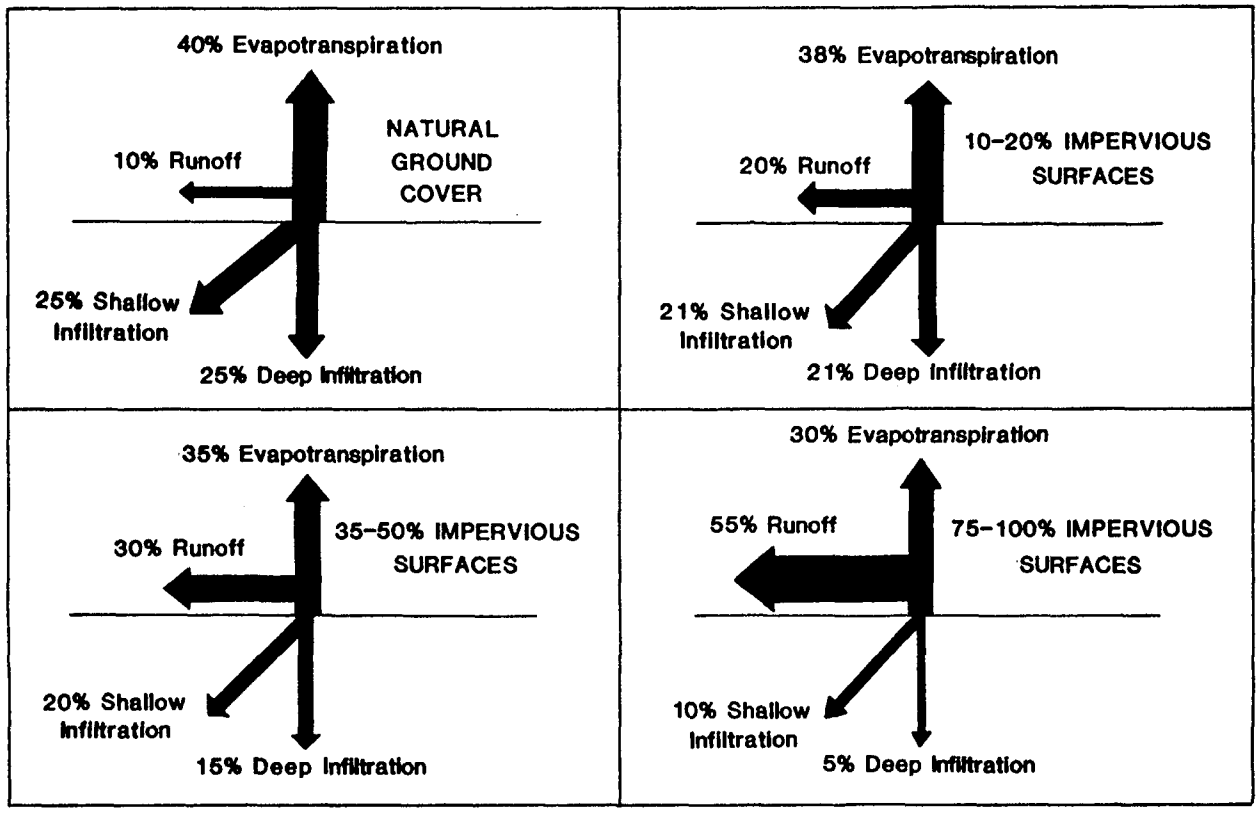

FIGURE 2. EFFECT OF GROUND COVER ON URBAN RUNOFF (from Tourbier and Westmacott, 1974).

The volume and rate of urban runoff have impacts other than transporting pollutants. Porcella and Sorensen (1980) report that flood stages can be increased by a factor of two to five as development occurs. This increased volume and the accelerated rate at which it can occur result in increased channel erosion and bottom scour, lower baseflow because of decreased groundwater recharge, and runoff exposure in areas previously unaffected.

The extreme variability in precipitation and snowmelt behavior add to the complexity of runoff characterization. Some generalization can, however, be made based on the fact that on a long-term basis, precipitation character can be statistically defined. Rainfall-runoff coefficients can be generalized based on actual and modeled runoff behavior. Konrad et al. (1978), Oberts (1982), and U.S. EPA (1983) have shown that the percentage of runoff increases with the percentage of effective impervious area, ranging from 80 to 90 percent for moderately large events in watersheds with over 90 percent effective impervious area, to less than 10 percent for small events in low effective impervious areas. Typically for urban areas, most runoff comes from the effective impervious areas with a minor amount contributed from pervious areas. In urbanizing areas and in developed areas during the period of frozen ground, runoff from pervious areas becomes a far more significant contributor to watershed runoff.

Runoff quality

Runoff picks up material on urban/urbanizing surfaces and moves it to receiving water. U.S. EPA (1984) in a report to Congress stated that urban nonpoint source water quality problems affect 20 percent of the nation's river miles and occur at some level in one-half of the nation's river basins. Six of 10 U.S. EPA regions and 36 of the 50 states have similarly identified urban/urbanizing area runoff as a major problem. The remainder of this paper focuses on the pollutants that cause the largest receiving water problems in these areas. Essentially everything that comprises the urban/urbanizing surface, or is used within these areas, can be expected to occur in some form in runoff. A list of those pollutants and the problems they cause follows:

- Solids--Inorganic (sediment) and organic debris can be moved by runoff either in dissolved or particulate form. The particulate, suspended solids are the most significant problem commonly associated with urban/urbanizing runoff. The problems caused by suspended solids in transit and after settling include turbidity, aquatic biota habitat destruction, transport of adsorbed pollutants, benthic accumulation, and direct adverse impact on aquatic organisms (Lager et al., 1977; Thronson, 1976).

- Nutrients--Of particular importance are phosphorus and nitrogen. These nutrients accelerate eutrophication in quiescent waters, exert toxic effects on aquatic organisms (ammonia), produce highly organic sediments, affect dissolved oxygen concentration, aid in the shift toward less desirable aquatic organisms and impact groundwater (nitrate) (Lager et al., 1977).

- Toxicants--This category includes heavy metals, pesticides and organic chemicals. U.S. EPA (1983) has listed 129 "priority pollutants", 77 of which were detected in EPA's recently completed NURP studies. The effects of these chemicals are related to ill-health of humans and aquatic life. The toxicity of the various chemicals depends on hardness of the water,

biological species being affected, degree of risk accepted and pollutant concentration. Accumulation of toxicity in the environment, for example, heavy metals in sediment, means that they remain available for extended periods of time.

- Oxygen-demanding substances--Organic and other oxidizeable material (ex., iron) exert a demand for oxygen in order to degrade. The measure of oxygen demand is termed biochemical oxygen demand (BOD) or chemical oxygen demand (COD): the latter is best suited to description of urban runoff impact. The effect of these pollutants is the depletion of dissolved oxygen, resulting in stress or death for aquatic organisms and generally degraded water quality (Abernathy, 1981; Keefer et al., 1979).

- Bacteria and virus--Numerous bacteria and virus strains occur in high con centrations in urban runoff, but most studies measure bacterial pollution through the fecal coliform count. Fecal coliforms are excreted from the intestinal tract of warm-blooded animals, and serve as a less-than-perfect indicator of human pathogens. The direct impacts of bacteria and virus are related to human health and disease (U.S. EPA, 1983).

- Chloride--The last major urban runoff pollutant of concern is chloride which results from the application of road salt. High chloride levels in groundwater present health risks. Elevated chloride levels can also inhibit the circulation of water in lakes (Lager et al., 1977).

Some of the above pollutants present a water quality problem in the short-term (toxicity, turbidity, oxygen depletion), while others take longer to become a threat (accumulation of metals, benthic oxygen depletion and nutrient enrichment). Specifics of pollutant levels will be covered in a later section.

SOURCES

The sources of the pollutants just itemized are numerous. The categories that follow are those best representing urban and urbanizing area activities that contribute nonpoint pollutants.

Street surfaces

Dust, dirt and debris accumulated on impervious surfaces contribute substantial amounts of pollution when runoff events wash these surfaces. Material deposited on street surfaces typically includes combustion emissions from motor vehicles and industries, wind blown dust and dirt, traffic deposits (tires, clutches, brakes, fluid, leaks, etc.), vegetation, material wash-in, roadway material, anti-skid grit and deicing chemicals, and litter (APWA, 1969; Novotny and Chesters, 1981; Pitt and Amy, 1973; Sartor and Boyd, 1972; Shaheen, 1975). Table 1 summarizes the data presented in the above referenced efforts relative to solids accumulation rates and associated pollutant concentations (Manning et al., 1977). This table reflects the material available for washoff from street surfaces. Bannerman et al. (1983) point out that a portion of this material is very likely to be washed-off during melt and rainfall events. They report in their Milwaukee NURP study that street loads of solids and toxics are at their peak in the snowmelt and early spring rainfall runoff. Pitt and Bozeman (1982) report after studying various sources of urban pollution that parking lots and street gutters contribute the greatest loads of pollutants. They also indicate that 75 percent of street surface pollutants and 50 percent of parking

TABLE 1. SUMMARY OF MEAN SOLIDS ACCUMULATION RATES AND MEAN ASSOCIATED POLLUTANT CONCENTRATIONS
(From Manning et al., 1977)

Pollutant	Single Family Residential	Multiple Family Residential	Transportation*	Commercial	Industrial
Solids (kg/curb km/day)	182	157	---	45	288
BOD_5	5,260	3,370	2,300	7,190	2,920
COD	39,250	41,970	54,000	61,730	25,080
Total nitrogen (TN)	460	550	235	420	430
Dissolved phosphorus (DP)	49	58	---	60	26
Cadmium (Cd)	3.3	2.7	---	2.9	3.6
Chromium (Cr)	200	180	80	140	240
Copper (Cu)	91	73	120	95	87
Nickel (Ni)	38	18	190	94	44
Lead (Pb)	1,570	1,980	12,000	2,330	1,590
Zinc (Zn)	310	280	1,500	690	280
Fecal coliform (geometric mean)	82,500	925	38,800	36,900	30,700

*Data from one study only (Shaheen, 1975).

lot pollutants are eventually delivered to catchment outfalls. As a result of this high delivery rate, they found that the two areas contribute 50 percent of the total solids, 85 percent of the TP, 80-100 percent of toxic metals, and close to 100 percent of the fecal coliforms delivered to urban catchment outfalls.

Most of the polluting material on street surfaces is associated with fine-grained deposits rather than with the coarse-grained sand and grit. Table 2 shows the results of a study by Sartor and Boyd (1972) on the percent of street surface pollutants by particle size. Similar work on particle size distribution from other areas has verified the general pollutant distribution, that is, concentration of pollutants in this finer-grained fraction (less than 246 microns). For example, 92.2 percent of the phosphates and 73.0 percent of all toxic metals are associated with 43.4 percent of the total solids (less than 246 microns). Bannerman et al. (1983) documented the phenomenon of fines-associated pollutant wash-off, showing that most pollutants of concern peak in concentrations in the .125-.250 micron range (fine sand, silt). These results are important to keep in mind during the development of management strategies for pollution reduction, such as street sweeping and leaf/litter removal.

TABLE 2. PERCENT OF STREET SURFACE POLLUTANTS BY PARTICLE SIZE
(From Sartor and Boyd, 1972)

Pollutant	Particle Size (microns) Less than 2,000	246-2,000	43-245	Less than 43
Total solids	24.4	32.2	37.5	5.9
Volatile solids	11.0	29.4	34.0	25.6
COD	2.4	17.5	57.4	22.7
BOD_5	7.4	35.8	32.5	24.3
TKN	9.9	31.6	39.8	18.7
Nitrates	8.6	14.4	45.1	31.9
Phosphates	0	7.8	36.0	56.2
All toxic metals	16.3	32.4	23.5	27.8
All pesticides	- - - - 27.0- - - -		- - - -73.0 - - -	
PCBs	- - - - 66.0- - - -		- - - -- 34.0 - - -	

The exact behavior of pollutant buildup is a much debated phenomenon. The best data have been presented by Sartor and Boyd (1972), who define an exponential buildup of pollutants, with the highest rate occurring two to four days after washoff. Lager et al. (1977) and Roesner (1982) present additional information in support of those exponential functions. Bannerman et al. (1983) studied the total pollutant loads (rather than time) versus daily accumulation rates and found no data to support diminishing accumulation rates with increasing street loads; that is, buildup was not affected by existing load. In short, accumulation of street surface pollutants is a well documented occurrence that occurs at an uncertain rate. Management strategies that hope to reduce urban area loadings must address the fine-grained materials occurring on impervious surfaces and do it in a way that reduces accumulation capacities.

Atmospheric

One contributing factor to street surface runoff pollution is atmospheric deposition. Such deposition, however, contributes also to all other urban area surfaces to some degree. Sources of pollution to atmospheric wetfall and dryfall are vehicular emissions, industrial emissions, soil erosion, improperly applied chemicals, wind-blown vegetative debris, and fuel combustion (ex., wood stoves). Pollutants resulting from these sources include nutrients, inorganic and organic solids, toxic metals and organics, pesticides and sulfur.

Dry deposition appears to be reflective of surrounding land use activities and tends to be seasonal in behavior (Bannerman et al., 1983; Oberts, 1982). The driving mechanisms for dry deposition are particulate and aerosol suspension, sorption, and particulate settling, all of which occur on a continual basis. Data on dryfall deposition rates are shown in Table 3. Ranges are given in Table 3 because of the extreme variability of dry deposition. It is importannt to establish some local calibration data to determine location within the range.

TABLE 3. RANGE OF CONTAMINANT DRY DEPOSITION RATES
(From Bannerman et al., 1983; Novotny and Chesters, 1981; Oberts, 1982)

Pollutant	Deposition Rate ($mg/m^2/day$)
Total solids	11 - 735
Total P	0.007 - 0.421
Total Pb	0.001 - 0.390
COD	1.2 - 86.7
Sulfate	0.52 - 8.67
TKN	0.52 - 1.58

Wet atmospheric deposition occurs as rain and snow scavenge pollutants from the atmosphere. Wet deposition seems to be more uniform in quality and does not reflect land use activities because of the larger geographic mobility of precipitation. Data on wetfall are given in Table 4. As with dryfall, concentration ranges are given because of the great variability in the data. Seasonality of wetfall loading is more a function or precipitation behavior than land use activities.

TABLE 4. RANGE OF CONTAMINANT WET DEPOSITION RATES
(From Bannerman et al., 1983; Konrad et al., 1978;
Manning et al., 1977; Oberts, 1982)

Pollutant	Concentration (mg/l)
Total solids	2 - 100
Suspended solids	0.5 - 58
Total P	0.001 - 0.162
Total Pb	0 - 0.110
COD	2.5 - 34.0
Sulfate	0.50 - 13.0
TKN	0.20 - 2.20
pH	3.0 - 6.4
PCB and pesticides (ng/l)	0.5 - 32.0

Together wetfall and dryfall can contribute a substantial portion of the total load from a drainage catchment. Contributed amounts of total nitrogen always seem to be quite high, often contributing over 75 percent of the total load, while total phosphorus is somewhat less, often ranging from 25 to 50 percent (Bannerman et al., 1983; Lager et al., 1977; Miller and Mattraw, 1982; Oberts, 1982; Reckhow et al., 1980). Pitt and Bozeman (1982) attribute generally less than 15 percent of any pollutant to precipitation in San Jose, Calif. This finding is consistent with Novotny and Chesters (1981) who point out that loading from precipitation is expectedly high in the more humid midwest and southeast portions of the U.S. They also note the increased pollutant scavenging efficiency of snow over rain because of the larger surface area of snowflakes.

The outlook for improved atmospheric loading is mixed. Much attention is being paid to reduction of industrial and automotive emissions, but reserves of low

cost, low sulfur fuel are being depleted and many homes are increasingly changing to wood burning furnaces and stoves, to name just two forces working against reduced pollution.

Construction

The highest per unit solids loading in urban/urbanizing areas comes from construction activities that disturb the landscape (Thronson, 1976). Construction of residential, commercial/industrial, transportation and utility projects presents challenges for those trying to minimize urban area nonpoint pollution.

The most obvious problem associated with construction activities is soil loss. Sonzogni et al. (1980) report that soil texture is the most important land form relating to erosion, but they also note significance for other soil characteristics (water holding capacity, erodibility, etc.), slope, topography, geology and drainage patterns. Thronson (1976) adds other nonland related factors including climatic conditions, activity at site, equipment use, length of soil exposure and management practices used to reduce soil loss. U.S. EPA (1984) reports that annually 1.6 million acres of land are developed. This construction activity accounts for about five percent of the nationwide sediment load coming from a very small, concentrated source area.

Erosion problems are caused by on-site operations such as clearing and grubbing, rough and final grading, foundation excavation, and site restoration/landscaping. Some reported soil losses from construction sites are (Heaney et al., 1975; Konrad et al., 1978; U.S. EPA, 1973; Yorke and Herb, 1976):

- o Maryland residential - 354-42,350 MT/km^2/yr
- o Washington D.C. residential - 16,800
- o Maryland residential, commercial - 1,000-100,000
- o Fairfax Co., Virginia highway - 12,600
- o Georgia highway - 17,500-52,500
- o Montgomery Co., Maryland residential - 8,770-42,000
- o Menomonee Basin, Wisconsin residential - 4,370

The on-site erosion noted above does translate into receiving water problems. Oberts (1982) notes that the loads at the mouths of two watersheds in which residential construction was occurring were 79 and 203 MT/km^2/yr, a substantial reduction from the on-site values but dramatically above similar watersheds with no construction. Konrad et al. (1978) report for the Menomonee River watershed that 37 percent of the TSS load and 48 percent of the TP load during the study period originated from 2.6 percent of the watershed under development, thus indicating primary management attention focused on developing areas.

Other pollutants associated with construction sites include nutrients, construction-related chemicals and biological contaminants. The level of nutrients associated with construction is a function of organic soil content, fertilizer use for vegetative establishment and vegetative debris washoff. As previously noted, a substantial TP load (8MT/km^2/yr) can result from construction activity (Konrad et al., 1978). Oberts (1982) reports that a high level of on-site nutrient washoff resulted in watershed outflow TP levels of about 0.2 MT/km^2/yr.

Construction-related chemical problems have been noted, but documentation of the problems is essentially nonexistent. U.S. EPA (1984) and Thronson (1976) report problems with pesticides for weed and pest control, petroleum products from vehicles, construction materials (paint, curing compounds, solvents, salt, asphalt, acids, mortar, dye, insulation) and general litter. Additionally, both sources note problems with human wastes from improper sanitation practices.

Construction activities clearly have the potential to cause major degradation of urbanizing area waters. Even a small amount of construction in a watershed (2.6 percent in the Menomonee River case) can cause substantial pollution problems. This fact points to the need for special effort to control construction activities.

Vegetation

Vegetative material contributes to the solids, organic and nutrient load from urban areas. Sources of this vegetative litter are leaves and twigs, grass clippings, seeds, and weeds. Problems are associated with both particulates and with dissolved leachates. Heaney et al. (1975) report that evergreen and deciduous leaves contain 0.04-0.28 percent (dry weight) TP, and 0.58-1.25 percent TN. They also report that burning leaves concentrates the nutrients. The loading values from these percentages translate into 0.6-4.5 kg/ha/yr for TP and 13.6-64.7 kg/ha/yr for TN. Grasses can contribute another 9-13 kg/ha of TP and 48-361 kg/ha of TN. Overall, Heaney et al. report that 3,000 kg/ha/yr of vegetative litter is typical of urban areas.

Pitt and Bozeman (1982) in their San Jose study identify vegetation as a major source of loading from lawn and landscaped areas, vacant lots, sidewalks, parking lots and street surfaces. The authors follow this source analysis with an indication that vegetative litter falling on impervious surfaces is more than likely to be washed off and contributed to watershed outfall loading. They report that vegetation is responsible for approximately 70 percent of the BOD loading, 30 percent of total solids and 10 percent of TKN.

Chemical use

As pointed out previously, every contaminant used in an urban environment will eventually be seen in urban runoff. In northern climates, the need to deice requires the use of salt, which can lead to chloride pollution problems. The snow and ice accumulate all pollutants for periods of time between melts. Oberts (1982) addresses the pollutants associated with snowmelt runoff and indicates that essentially all of the salt added to a roadway can be accounted for in runoff to drainage conduits and eventually to receiving waters. This poses a problem of groundwater and lake contamination. Similar problems have been noted in Wisconsin, Massachusetts, New York, Michigan and Canada (Manning et al., 1977). Total salt use seems to be decreasing, but when safety is concerned it can be assured that salt will always be used as the answer.

Pesticides and fertilizers are also chemicals commonly seen in runoff. Again, anything applied in residential areas, in public parks, and on hobby farms close to urbanized areas will eventually be seen in runoff. In Minnesota, for example, approximately one-half of all pesticides applied annually are applied by inexperienced urban household dwellers, who may or may not apply and dispose of the chemicals properly. Sonzogni et al. (1980) report that home fertilizer use can also be a major cause of nutrient impact to receiving waters and that

such nutrients are usually readily available to noxious organisms. They also indicated a wide spectrum of pesticides in runoff. Novotny and Chesters (1981) also point out that improperly applied chemicals can volatilize and migrate as aerosols, or can migrate directly as adsorbed particles.

Similar migration problems can be seen for PCBs, which have been detected in both Arctic and Antarctic ice (Novotny and Chesters, 1981). Such dangerous chemicals can be introduced into the environment by very routine, but ill-advised practices such as hosing down spill areas, dumping chemicals down household or street drains, or failing to quickly repair leaking chemical containers. A discussion in a later section shows the NURP data on the detection of priority pollutants in urban runoff.

Other sources

There are some other source areas of urban runoff pollutants that are not as obvious as those previously noted. Bannerman et al. (1983) found that galvanized steel eaves troughs and downspouts are major sources of Zn in rooftop runoff. They found, however, that precipitation was the major source of most contaminants to rooftop runoff. They conclude that the relative importance of rooftop runoff decreases with increased urban development because other sources become more important. Pitt and Bozeman (1982) report that residential, composition shingle roofs can be a potential significant source of oil and grease and other toxic materials that might compose the shingles or be blown onto them. They found, for example, that 1 to 20 percent of some metals (As, Zn, Cu) could come from rooftops.

Urban street catch basins can also be a little-suspected source of contamination. A Chicago study by APWA (1969) found that runoff water in catch basins becomes septic and could easily reach a BOD concentration of 60 mg/l. COD levels in the settled solids were 38,300 mg/kg before a wash-through and 24,500 mg/kg afterward. Sartor and Boyd (1972) studied catch basins in San Francisco and found COD levels as high as 143,000 mg/l; BOD up to 1,500 mg/l; TN up to 33.2 mg/l; and TP up to 0.3 mg/l. They reference similarly high values from Milwaukee and Baltimore.

Fecal coliform can get into urban drainage conduits from septic tanks, infiltration from sanitary sewers to storm sewers, leakage at manholes, interconnections and surcharges that cause overflows (U.S. EPA, 1983). Extremely high levels of fecal coliform in urban runoff are discussed in a subsequent section.

The final two sources of urban contamination, for which little documentation exists, are litter and collection system erosion/deterioration. Litter is a very visible, yet little documented contributor. Urban stream channel and bank erosion, and conduit scour contribute pollutants directly to the runoff that has already been captured and is in transit.

EFFECTS OF URBAN RUNOFF

The actual water quality effects of urban/urbanizing area runoff can be measured by pollutant concentration and loading, by biological impact, and by groundwater impact. The following sections address these effects.

Concentration levels

The most complete set of data on the concentration of pollutants from urban runoff was collected in conjunction with EPA's Nationwide Urban Runoff Program (NURP) (U.S. EPA, 1983). The NURP effort involved a great deal of attention to the technical methodology for data collection and analysis at 81 sites in 28 cities throughout the country. It truly represents state-of-the-art urban runoff documentation. As such, the majority of the conclusions relative to urban runoff presented in this paper are somehow derived from NURP.

Perhaps the most significant finding of NURP is that variability in flow and pollutant concentration are so extreme from site to site and from event to event that relying on specific land uses and geographic locations to base pollution projections upon is not statistically valid. Rather, NURP recommends at a minimum using NURP summary data for concentration and applying it to local runoff data to determine loading. This, of course, is in lieu of locally collected data.

NURP concentration results are reported as "event mean concentration" or EMC, which is defined as the total mass load for an event divided by the total volume of flow. This could also be termed the flow-weighted mean. The distribution of concentrations was found to be lognormal, that is, a logarithmic (base e) transformation is required to describe the data in linear statistics.

Table 5 shows the various land use EMC data and the summary data for all sites. The all sites entry is recommended by EPA "for planning level purposes as the best description of the characteristics of urban runoff." The land use EMC data are sufficiently different that they can be used to "fine tune" local pollutant concentration input to best fit local conditions.

TABLE 5. NURP SUMMARY WATER QUALITY CONCENTRATIONS FOR URBAN RUNOFF (US. EPA, 1983)

	Site Median EMC (Coeff. of Variation)*					
	All Sites					
Pollutant	Median	90th Percentile	Residential	Mixed Urban	Comm./ Ind.	Open/ Nonurban
TSS (mg/l)	100(1-2)	300	101(0.96)	67(1.14)	69(0.85)	70(2.92)
BOD_5 (mg/l)	9(0.5-1.0)	15	10(0.41)	7.8(0.52)	9.3(0.31)	---
COD^5(mg/l)	65(0.5-1.0)	140	73(0.55)	65(0.58)	57(0.39)	40(0.78)
Total P (ug/l)	330(0.5-1.0)	700	383(0.69)	263(0.75)	201(0.67)	121(1.66)
Soluble P (ug/l)	120(0.5-1.0)	210	143(0.46)	56(0.75)	80(0.71)	26(2.11)
TKN (ug/l)	1500(0.5-1.0)	3300	1900(0.73)	1288(0.50)	1179(0.43)	965(1.0)
NO_2-NO_3-N (ug/l)	680(0.5-1.0)	1750	736(0.83)	558(0.67)	572(0.48)	543(.091)
Total Cu (ug/l)	34(0.5-1.0)	93	33(0.99)	27(1.32)	29(0.81)	---
Total Pb (ug/l)	144(0.5-1.0)	350	144(0.75)	114(1.35)	104(0.68)	30(1.52)
Total Zn (ug/l)	160(0.5-1.0)	500	135(0.84)	154(0.78)	226(1.07)	195(0.66)
Fecal Coliform (warm weather, No./100 ml)	21,000(0.8)	---	---	---	---	---
Fecal Coliform (cold weather, No./100 ml)	1,000(0.7)	---	---	---	---	---

*Coefficient of variation = standard deviation divided by mean (higher value indicates more variable data).

The data in Table 5 are analyzed by U.S. EPA (1983) relative to importance in urban runoff. The TSS concentration in urban runoff is typically much greater than that from a well run secondary sanitary treatment plant (100 mg/l vs 25 mg/l). The urban runoff solids differ from treatment plant solids in that the urban runoff solids are more mineral (less organic) and have more pollutants adsorbed to them. EPA suggests that control of solids be pursued where water quality problems exist with solids and where sediment build up is a problem.

Oxygen-demanding substances (BOD and COD) are present in urban runoff at about the same concentration as from a secondary treatment plant. No NURP project area defined low dissolved oxygen (DO) as a water quality problem resulting from urban runoff. Other studies, however, have shown impacts. Keefer et al. (1979) looked at 104 sites nationwide and found a likely probability that a higher than average DO deficiency occurred at time of high flow or during periods of precipitation. They report that the diurnal cycle typical of DO behavior disappears during high flow, and that DO levels are depressed 1.0-1.5 mg/l below low flow levels for periods of 1 to 5 days. Similar DO behavior has been noted by Abernathy (1981), Manning et al. (1977), Mikalsen (1984) and Rimer et al. (1978). For these reasons it is possible that the "critical" period for DO in streams is not simply during dry periods of low flow, but rather during such periods when precipitation generates urban washoff.

Nutrients (N and P) are always present at elevated levels in urban runoff, but generally their contribution is less than point source discharges. For example, TP from an advanced treatment plant is still 1.0 mg/l and TN from a secondary plant can be as high as 20 mg/l. Oberts (1982), however, reports that even though nutrients from urban nonpoint sources might be less than point sources, quite often the receiving waters are urban lakes or quiescent streams where eutrophication or toxic ammonia impacts can occur.

Toxic contamination can also occur from heavy metals and organic chemicals. NURP sampled priority pollutant concentrations in urban runoff at 61 sites in 20 of the 28 study locations. A limited sample was taken for 120 priority pollutants, and supplemented by a special metals study; both of these were in addition to routine metals analyses. Of the 120 pollutants examined a total of 77 were detected at some level. The detected group included 13 metals, cyanide, and 63 organic chemicals. Table 6 lists the most frequently detected pollutants and shows the percentage of samples in which the pollutant was detected.

Of particular note is the 91-94 percent detection for Pb, Zn and Cu. Levels of these metals, other metals and many of the organics are above what can generally be called good quality water, violating U.S. EPA recommended or established criteria. The organics are seen at a lower frequency, but some type of a criteria violation (aquatic life, drinking water, carcinogen and/or mutagen) occurs for PCP, chlordane, bis (2-ethylhexyl) phthalate, gamma-BHC, alpha-endosulfan, phenanthrane, pyrene and chrysene (U.S. EPA, 1983). It must be noted that the NURP data are the best available even though they are limited in number with only two samples taken at each site. The results do clearly point to the fact that toxics are moving in urban runoff.

Coliform bacteria are present in very high levels in urban runoff, ranging in the NURP studies from 10's to 100,000's/100 ml. Every sample taken following a runoff event showed very high levels, far in excess of the criteria established by EPA (200/100 ml). Table 5 shows the differences in concentrations between warm and cold weather, a difference due to the dynamics of coliform and to

TABLE 6. NURP PRIORITY POLLUTANT STUDY RESULTS
(U.S. EPA, 1983)

Inorganics		Organics	
Pollutant	% Detected	Pollutant	% Detected
Lead	94	Bis (2-ethylhexyl) phthalate	22
Zinc	94	alpha-Hexachlorocyclohexane (BHC)	20
Copper	91	alpha-Endosulfan	19
Chromium	58	Pentachlorophenol (PCP)	19
Arsenic	52	Fluoranthene	16
Cadmium	48	gamma-Hexachlorocyclohexane (Lindane)	15
Nickel	42	Pyrene	15
Cyanides	23	Phenol	14
Antimony	13	Phenanthrene	12
Beryllium	12	Dichloromethane	11
Selenium	11	4-Nitrophenol	10
		Chrysene	10

activity of contributors in the watershed. The health implications of high coliform levels are hard to define because of the fact coliform is a poor indicator organism and people are generally not in contact with urban runoff water during and shortly after events.

The major NURP findings, other than expanding the data base, are that EMC's are not a function of geography, land use or watershed physiography, and that traditional point source oriented water quality criteria are not at all suited to describe the true impact of urban nonpoint pollution, particularly when EPA has not recommended criteria for many of the pollutants of concern. It should be noted that the factors listed as not influencing EMC do influence runoff volume and will, therefore, impact total load.

Much other data exist describing the nature of urban runoff pollutant concentrations. Table 7 is a summary of some other data. It must be noted that the methods of collection in these studies is in most cases not consistent with NURP. They are presented for comparative purposes for those who feel they could adjust their local situation. Where single values are presented, only one study was found that collected that data.

Some brief comments can be made about runoff concentration. Receiving water studies have shown that the high pollutant concentrations noted in Table 7 do translate into impacts on receiving water (Keefer et al., 1979; Konrad et al., 1978; Oberts, 1982; Pitt and Bozeman, 1982; U.S. EPA, 1983). Examples of impacts seen are DO depression, toxic contamination of the water column and sediments, channel degradation and eutrophication.

The "first flush" phenomenon of pollutant concentration likely does occur, but its importance in total loading is minimal for noncombined sewer areas (Konrad et al., 1978; Manning et al., 1977; Porcella and Sorensen, 1980; Rimer et al., 1978). The reason for this is that total load is more affected by flow character than by pollutant level. Quite often peak concentration and peak runoff rate are reached at the same time, yielding a maximum loading rate.

TABLE 7. URBAN RUNOFF CONCENTRATION RANGES OF REPORTED MEANS FROM NON-NURP STUDIES

Pollutant	Land Use Type				
	Residential*			Comm./	
	HDR	MDR	LDR	Ind.	Mixed
TSS (mg/l)	88-439	44-768	6-578	42-2052	13-5000
BOD_5 (mg/l)	11-38	6-37	2-63	8-76	3-160
COD (mg/l)	68-289	32-152	22-278	45-597	20-600
TP (ug/l)	400-2100	160-1390	60-410	100-1100	200-1100
Sol. P (ug/l)	100-2100	100-460	60-320	20-1140	190-1100
TKN (ug/l)	1630-4200	570-5000	500-1000	1000-2400	960-2500
NO_2-NO_3-N (ug/l)	420-820	70-790	400	530-700	50-800
Total Cu (ug/l)	10	53	---	4-64	1-150
Total Pb (ug/l)	190-780	20-400	80	146-2200	1-460
Total Zn (ug/l)	122-270	55-235	---	130-490	18-360
Fecal Coliform (1,000/100 ml)	14-130	5-31	22	0.1-18	4-126
Total Cd (ug/l)	4	---	---	2-4	5-10
Total Cr (ug/l)	25	5-255		31	2-251
Total Ni (ug/l)	---	16	---	33	17-150

*HDR-High-density residential; MDR-Medium-density residential; LDR-Low-density residential.

Seasonal differences in runoff quality occur and are especially pronounced in locales experiencing long periods of snow cover (Bannerman et al., 1983; Novotny and Chesters, 1981; Oberts, 1982). The highest unit loadings occur during snowmelt and early spring rainfall events when accumulated winter debris is washed off. Lowest values are of course, during periods of little rainfall, typically late summer and fall.

Loading levels

Another method to arrive at watershed loading is by application of normalized loading factors. NURP developed such factors, as have numerous other studies of urban runoff. The recommended reporting unit for loading is kg/ha/yr (lb/acre/yr). Table 8 reports the results of NURP, normalized for a 30-inch annual average precipitation. In addition to NURP values, Table 8 lists ranges of loading values, where they exist, from studies with conditions typical of urban runoff from an annual 30-inch precipitation.

The NURP data in Table 8 show high relative loadings for commercial/industrial areas, and near-equal loading for residential and mixed areas. It should be noted that the NURP data base contains only light industry. The non-NURP data show a very wide variability and do not necessarily reflect highest loadings from commercial/industrial areas. Again, no data base is nearly as complete as NURP, and use should be judged accordingly. Also, construction areas are not directly reflected in the data, but might be contributing to the higher loads noted for TSS and the nutrients.

TABLE 8. NURP (U.S. EPA, 1983) ANNUAL URBAN RUNOFF LOADS IN KG/HA/YR (Non-NURP Values*)

Pollutant	Residential	Commercial/ Industrial	Mixed
TSS	412 (11-3650)	1095 (50-5100)	480 (16-8030)
BOD_5	27 (0.64-9.7)	74 (2-97)	32 (2.6-560)
COD	188 (60-178)	500 (297)	219 (41-320)
TP	0.98 (0.04-2.8)	2.6 (0.10-4.5)	1.1 (0.08-10)
Sol. P	0.38 (0.20)	0.90 (0.30)	0.38 (0.05-1.3)
TKN	4.4 (2.1-8.0)	11.6 (12.7)	5.0 (1.2-7.3)
NO_2-NO_3-N	2.0 (0.30-2.8)	5.2 (1.8)	2.7 (0.96-6.1)
Total Cu	0.10 (0.03-0.80)	0.26 (0.29-8.6)	0.11 (0.02-1.9)
Total Pb	0.41 (0.01-5.6)	1.11 (0.17-13.2)	0.49 (0.14-6.5)
Total Zn	0.46 (0.02-1.7)	1.23 (2.9-12)	0.54 (0.10-2.8)

*References: Bannerman et al., 1983; Konrad et al., 1978; Lager et al., 1977; Novotny and Chesters, 1981; Oberts, 1982; Pitt and Bozeman, 1982; Porcella and Sorensen, 1980; Sonzogni et al., 1980; Whipple et al., 1976.

The primary factor in load determination is runoff. Control of loading, therefore, should focus on curtailing the movement of water from impervious areas. Novotny and Chesters (1981) recommend focusing on critical areas where large volumes of runoff are generated. The dominance of point source vs nonpoint source loading is usually variable by location, indicating the need to define this relationship before moving ahead with expensive corrective measures.

Biological impact

The biological impact of urban runoff is an area only recently receiving much attention. The best documented biological impact is that of solids, which was referenced in a previous section. Heavy metals also exert a strain on aquatic organisms. Morrison et al. (1984) report that toxic forms of metals account for 63, 66 and 73 percent of Zn, Pb and Cd, respectively, in urban runoff. Toxic forms of Cu account for a relatively lower 32 percent. The Pb in runoff is mostly associated with the particulates whereas the other metals cited are mostly dissolved or evenly divided. What this means is that Pb readily settles and accumulates in areas receiving urban runoff, and that several of the other metals do the same or become available in dissolved form in the water column. Disruption to the benthic community as streams flow through urban areas has been documented by Mikalsen (1984) and Pitt and Bozeman (1982), and Whipple et al. (1976). The runoff impacts are lowered organism diversity and total number, less desirable (pollutant tolerant) species, and decreased growth rates. NURP (U.S. EPA, 1983) results from two studies showed that many toxicants are moving in urban runoff, but impairment from these pollutants is not evident; that is, no acute or chronic effects can be documented. EPA does, however, recognize that impulse loads of toxicants can have drastic results, and recommends that some criteria be developed to address diffuse source loading. The NURP studies did point out the impact that physical channel degradation can have on aquatic life, and suggest that in certain instances this can be more severe than chemical impacts.

Groundwater impact

A limited number of studies have shown that urban runoff can impact groundwater, but usually not to a large extent. Konrad et al. (1978) report that even though some chloride problems exist with groundwater in the Menomonee River Basin, the total groundwater pollution load to the river is insignificant. Similar high chloride levels in urban/traffic areas are noted by Sonzogni et al. (1980). U.S. EPA (1983) showed in two NURP studies that deliberate infiltration of urban runoff does not cause toxic introduction problems to groundwater in the short-term. However, long-term impacts are unknown. The same studies showed that chloride is not attenuated by infiltration, that attenuation of essentially all toxicants occurs in just a few meters of soil, and that vegetation helps increase infiltration rates but has no effect on pollutants. Lager et al. (1977) report that some attention should be paid to nutrient inflow to lakes from groundwater, since it could represent substantial loading.

SUMMARY

Research efforts have shown that washoff of pollutants accumulated on urban and urbanizing surfaces can adversely affect water quality. In fact, some researchers suggest that the critical period for water quality in a stream is not a dry weather, low flow period, but rather a low flow period in which an urban runoff event occurs.

Data have been presented to show that urban/urbanizing area runoff can easily equal and exceed pollution loading from wastewater treatment plants, especially for solids, toxics and oxygen-demanding substances. Because of this nature of urban runoff loading, it tends to accumulate and exert its negative influence over long time spans, resulting in such problems as benthic accumulation of toxics and eutrophication of lakes and streams.

Characterization of runoff quality is a difficult task since the variability in quality seems to be independent of geographic locale, land use and watershed character. The best method for characterization, if local data are not available, is to apply NURP concentration data to locally developed runoff volume data. The impact of this runoff depends on several things, including climate, local receiving water physical and biological conditions, local perceptions of water quality, desired use of receiving waters, and regulations in effect. Once the relative impact of urban runoff is determined, cost-beneficial implementation measures can proceed addressing both point and nonpoint sources.

It is apparent that short-term efforts for management should be focused on high runoff, high pollutant concentration areas, particularly commercial/industrial and construction areas. Programs that grossly generalize management techniques should be avoided in favor of site by site or watershed by watershed evaluation. Urban runoff controls can be very expensive, so their use should be reserved for occasions when they are definitely needed.

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IMPACT OF NONPOINT POLLUTION ON RECEIVING WATERS

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INTRODUCTION

There is no doubt that the quality of our waterways has improved considerably since the first permanent Water Pollution Control act was passed in 1956. However, the major thrust of water pollution control efforts has been towards the easily controllable pollutants emanating from point sources. In many instances, the major contributions to a water body are diffuse in origin and the dimunition of point sources has not solved water quality problems. In many cases, more stringent controls on the point sources is not rational inasmuch as spending large amounts of money on what might be a fraction of the total pollutant load reaching a waterway is not in the public interest.

It is now time to put our water quality problems in perspective, define the real effects of various sources of pollutants on water quality and direct our control efforts towards those measures that will result in definitive improvement in the quality of the receiving water. An engineer is taught that 90% of solving a problem is defining the problem. This concept is particularly applicable to the management of multi-origin pollutants.

Many of the questions concerned with source-receptor relationships and their effects on water quality can only be answered by adequate data. While much data is collected on a nationwide basis, they are not collected to solve problems and for the most part, their only value is to indicate "trends" in water quality. In fact, the nation-wide improvement in water quality noted previously is mostly based on observations made in rivers and not in lakes where major nonpoint problems exist. Other major problems with the data include a lack of adequate quality control, a lack of comparable methodologies in collection, storage and analysis and inadequate interpretation. It is obvious to any user of these data that the entire collection, storage and retrieval system needs scrutiny.

Inasmuch as the impact of pollutants on receiving waters is essentially the same, whether they originate from point or nonpoint sources (ignoring distribution), it is important to determine their real significance. For example, one might find 3 mg/l of lead in urban runoff, however, if the flow only occurs for a few moments, the effects on the receiving water would probably be insignificant. Thus, determinations of mass loadings are paramount to determining impacts. The bottom line is whether or not the resulting stream concentrations are sufficiently high to cause adverse effects. Of course, possible long term sorption by sediments should be considered.

SOURCES

The EPA report to Congress on Nonpoint Source Pollution (1) identified five source categories of nonpoint pollution as follows:

Agriculture
Urban Runoff
Silviculture
Mining
Construction

One should probably add atmosphere sources to this list although the

relative contributions are probably not significant in many cases.

Agriculture contributes pollutants from cropland, grasslands and livestock operations and is probably the most pervasive diffuse source. Pollutants originating from agricultural practices include sediment, microorganisms, pesticides, nutrients, organic materials. It has been estimated that agriculture contributes more than 50% of the sediment delivered to the nations waterways. The role of agriculture practices in contributing nitrogen, phosophorus and pesticides to our waterways is also significant. Figure 1 demonstrates the relative concentrations of nutrients contributed by agriculture to other land uses (2).

Urban runoff also may play a significant role in contributing to water quality problems. Urban rainwater may yield heavy metals, microorganisms, sediment nutrients and organic materials. While urban runoff appears to have the potential for resulting in serious impacts on receiving waters as shown by Figures 2 (2) and 3 (3), the International Joint Commission (IJC) concluded that pollution from urban runoff does not have a major impact on water quality in the Great Lakes (4). It should be noted, however, that urban runoff's contributions of heavy metals, PCB's, etc. over a long term basis must be considered. The potential for these and other priority pollutants to accumulate to significant levels over long periods of time is a viable possibility. In addition, significant adverse water quality impacts do occur on a localized basis.

In silviculture, similar to agriculture, the major pollutant by volume is sediment. Other pollutants that might be contributed by silvicultural practice include nutrients, organics and pesticides. In addition, concern has been expressed over the resulting increase in river temperature when trees are cut along river banks.

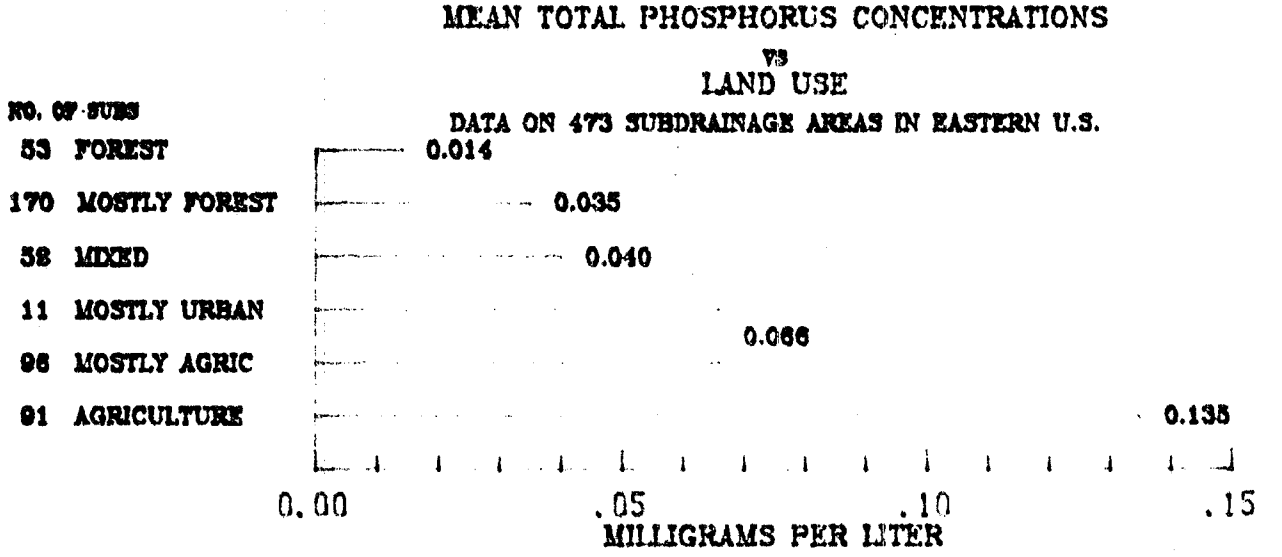
Mining activities, while not as widespread as other diffuse sources, can result in serious degradation of not only waterways, but groundwater. Contaminants emanating from mines may consist of numerous substances including sediment, acid, salts, metals, radioactive material and organics. The problems from abandoned mines are local in nature, however, their multitude and seriousness makes them a nationwide problem. The problems are exacerbated in the West because of the paucity of water.

Construction activities, while localized in nature, may result in severe water quality impacts. The major construction site pollutant is sediment, however, its contribution is only 4 to 5% of the total sediment load to receiving waters (1). Other pollutants contributed by construction may include nutrients, pesticides, petroleum derivatives and debris.

As pointed out by Krenkel and Novotny (6), the atmospheric contribution to nonpoint pollution may include nutrients, organics, acid deposition, particulates and any materials that may be carried into the atmosphere and subsequently washed out by precipitation. While the concentrations appear relatively small, the areal distribution may be significant in some areas.

EFFECTS

The effects of nonpoint pollution are similar to those of identifiable



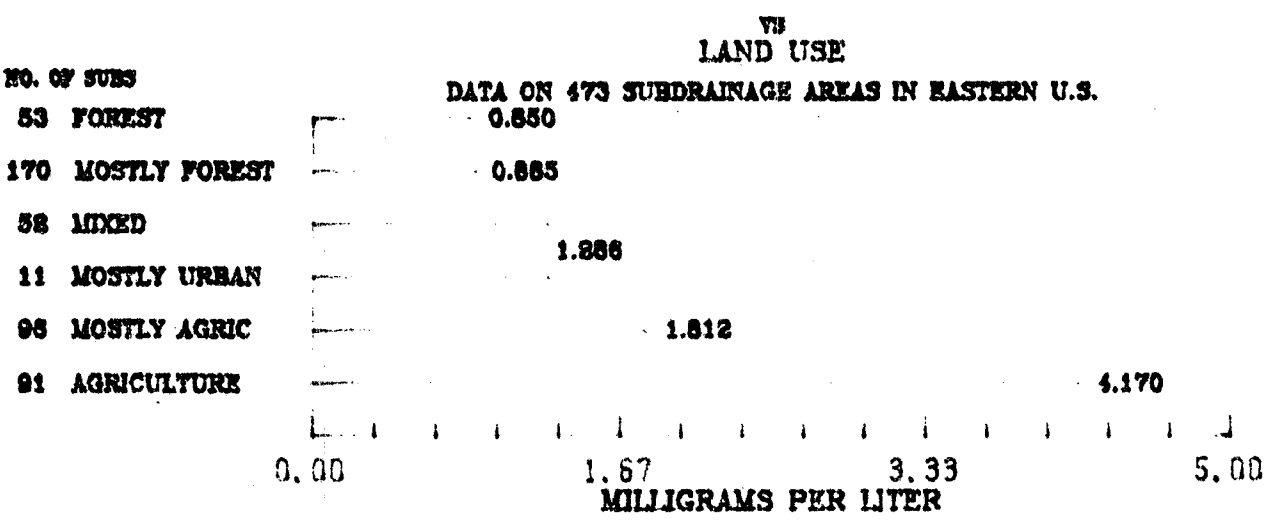


FIGURE 1 RELATIONSHIPS BETWEEN GENERAL LAND USE AND TOTAL PHOSPHORUS AND TOTAL NITROGEN CONCENTRATIONS IN STREAMS

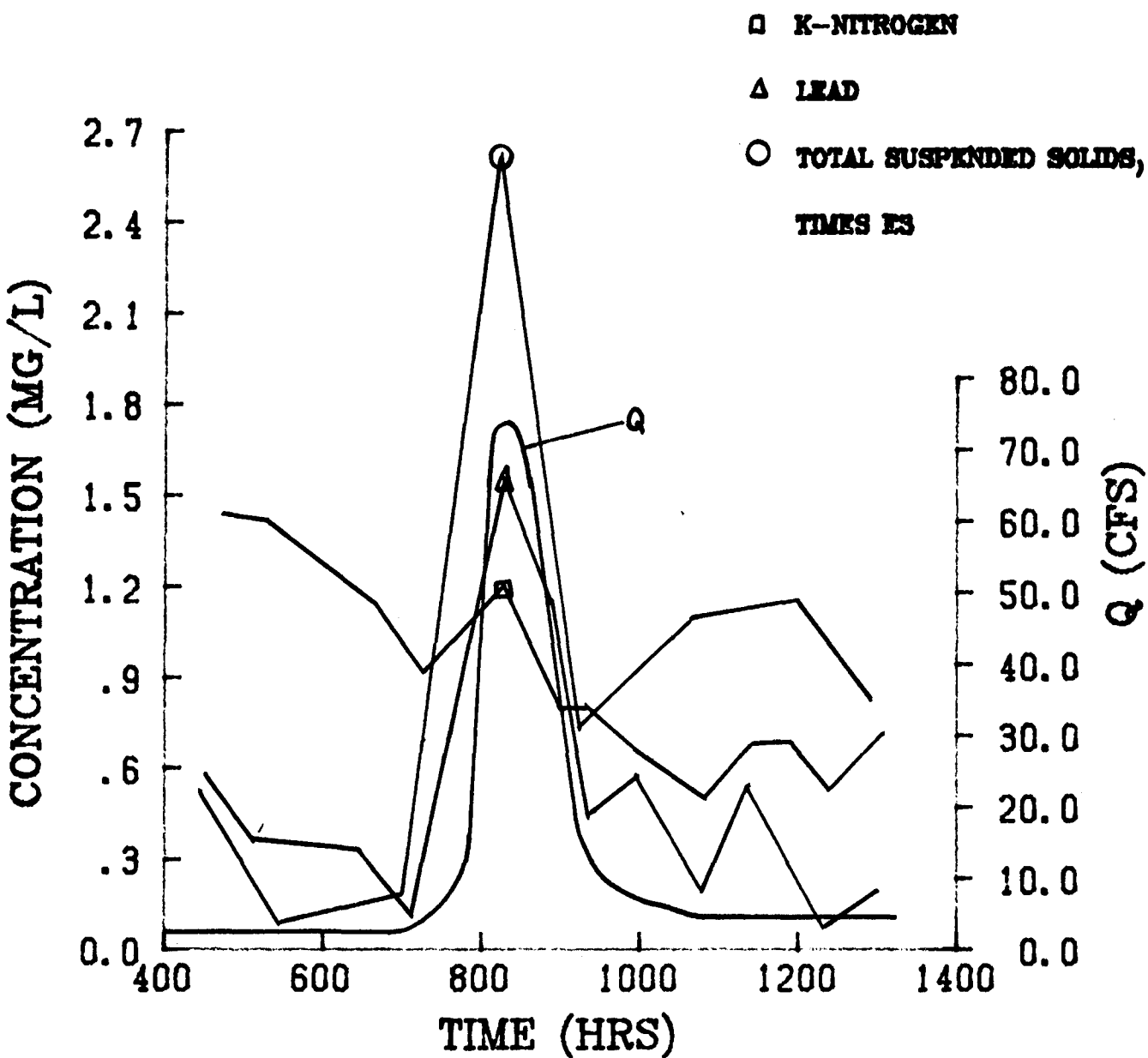


FIGURE 2 POLLUTANT VARIATIONS W/Q AND TIME
(REDRAWN AFTER COLSTON (5))

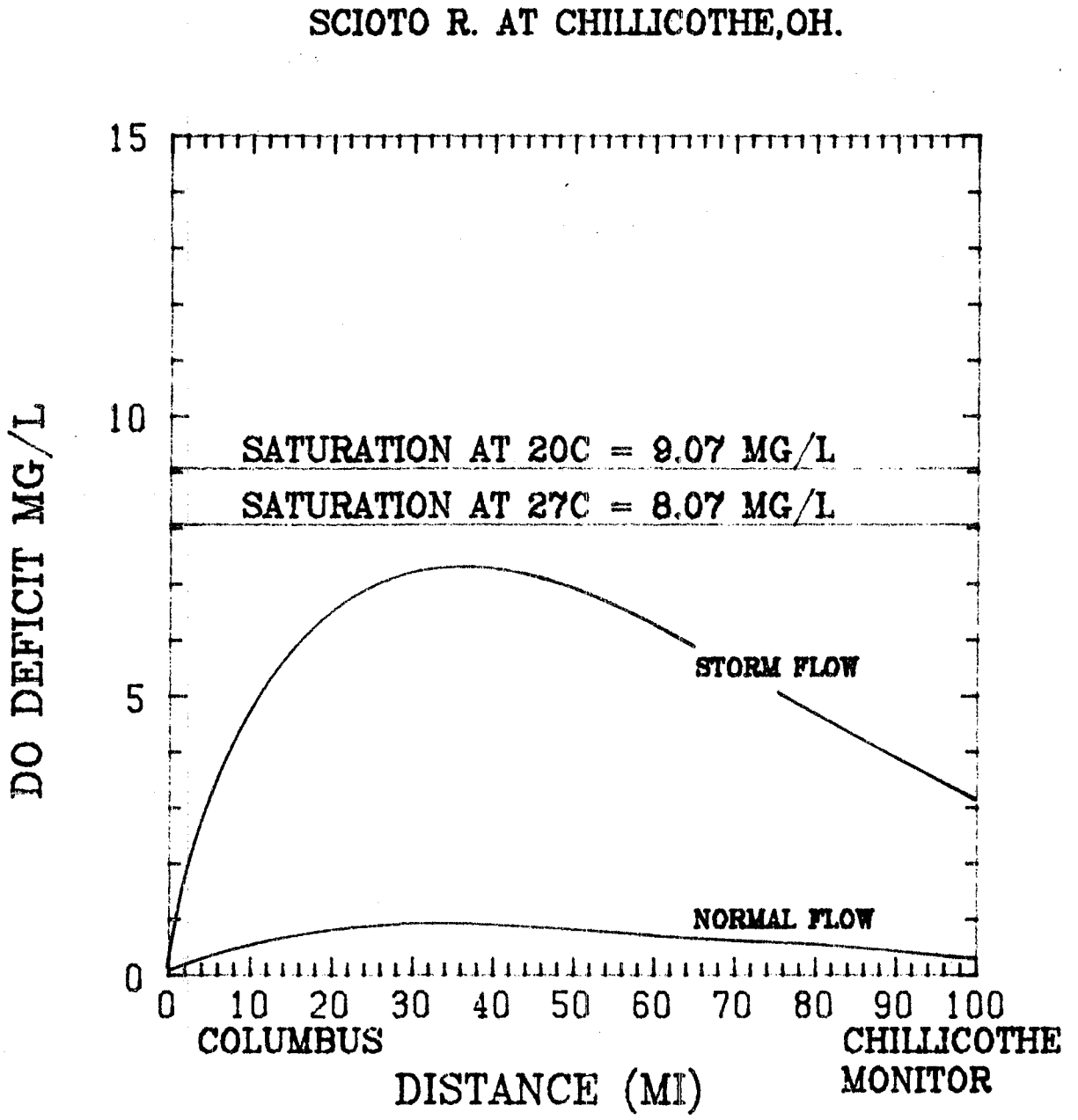


FIGURE 3 STREETER-PHELPS ANALYSIS RESULTS FOR SCIOTO RIVER AT CHILLICOTHE,OH.

point sources. However, their distribution with time and the manner in which they are introduced into a water body may differ. Therefore, the traditional analysis of diffuse sources of pollutants requires a somewhat different analytical treatment. Figure (4) illustrates the way in which the typical nonpoint pollutant enters a receiving water (7).

A discussion of the impacts of these pollutants is therefore in order.

Oxygen Demanding Substances

The contribution of biodegradable organics to a receiving water causes a depletion in oxygen resources. Should the concentration of oxygen be diminished below acceptable water quality standards, the lower dissolve oxygen levels will be inimical to aquatic life. As shown on Figure 3, nonpoint pollution may cause a dimunition of disolved oxygen subsequent to a storm event that may result in an oxygen deficiency. This can be particularly serious in the case of a small river or impoundment.

Microorganisms

The introduction of microorganisms into a receiving water is of concern because of the potential for the presence of pathogens. This may result in impaired recreational use of the water, an increase in water treatment costs and a potential human health hazard. The prime nonpoint source of these materials is from agriculture and urban runoff.

Metals

Metals in the aquatic environment may have a multitude of effects because of their toxicity. If the concentration is sufficiently high, they may cause death of aquatic life and lower levels may cause chronic toxicity resulting in adverse effects on propagation and life expectancy.

The metals may also be sorbed in the sediments and become toxic to benthic organisms, be biologically methylated and released to the water for fish accumulation and enter the food chain for subsequent biomagnification.

Metals have also caused economic problems by affecting commercial fishing, sport fishing and increased water treatments costs. The synergistic effects of metals with other metals and contaminants should also be considered.

Nutrients

Probably the contaminants from nonpoint sources causing the most problems are nitrogen and phosphorus. If one of these elements is growth limiting for either phytoplankton or macrophytes, their introduction into a lake may induce prolific growths of algae and/or attached aquatic plants. This hastens or accelerates the problems associated with eutrophication. It should be noted that most North American lakes are phosphorus limiting, thus making the control of phosphorus of significance.

Excessive algal growths and aquatic weeds may diminish the use of lakes for recreation, increase the costs of water treatment, increase hypolimnetic

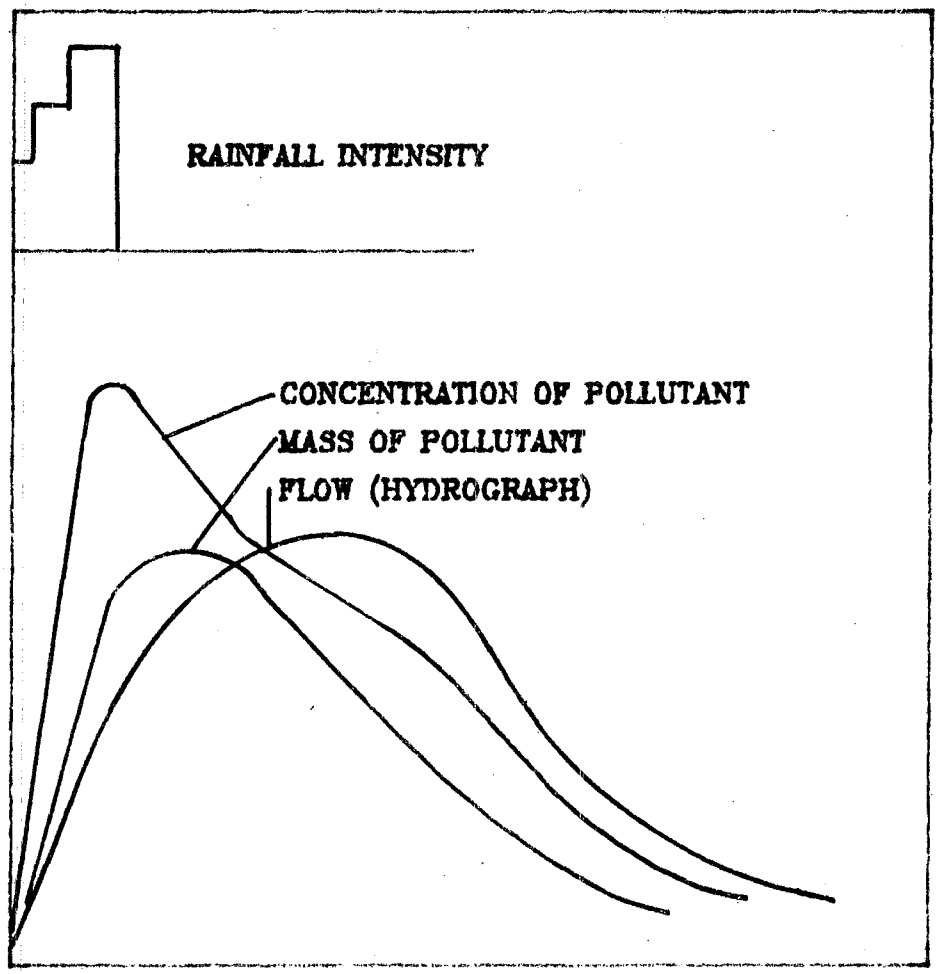


FIGURE 4 FLOW AND QUALITY HISTOGRAM FROM NONPOINT SOURCES

oxygen depletion and result in toxic effects on fish and humans. The appearance of the water may become aesthetically unacceptable thus causing major adverse public reaction. The major nonpoint source of nutrients in the U.S. is from agriculture.

Salts

The salinity of a water may have drastic effects, in particular in the arid parts of the country where nonpoint sources are the major culprit. For example, the sodium content of parts of the Colorado River are near 120 mg/l and the water is used for drinking purposes. Sodium also has an adverse effect on soils and salinity may affect crop yields and ultimately render the soil toxic to crops. The salt water intrusion along our coasts is a well known problem and could be considered as nonpoint pollution.

Herbicides and Pesticides

Herbicides and pesticides are obviously toxic materials and their presence in the aquatic environment can cause contamination of fish resulting in a public health hazard, sub-lethal effects on the entire food chain, biomagnification and adverse effects on reproduction. The major contribution of these materials is from agriculture.

Sediment

Sediment is the largest nonpoint problem, not only because of its magnitude, but because it may result in all of the previously described impacts due to the materials associated with the sediments.

In addition, sediment may cause decreased water transparency, thus interfering with photosynthesis, destroy spawning areas, clog fish gills, cause density problems and interfere with the benthic population. Turbidity also results in aesthetic problems and increases water treatment costs.

BIOLOGICAL AVAILABILITY

A major question with regards to some of the pollutants contributed from diffuse sources is their availability to be utilized by the microorganisms, for example, much of the phosphorus contributed via runoff is in the form of apatite, which for all practical purposes, is biologically unavailable (8).

This is an important concept inasmuch as it has been implied that one reason for placing emphasis on the control of point sources rather than nonpoint sources of phosphorus in the Great Lakes is because of the high bioavailability of phosphorus in wastewater and the large quantities of biologically unavailable phosphorus in many sediments (9).

The uncertainties caused by the lack of definitive resolution of the bioavailability question needs immediate attention. Every effort should be made to resolve this issue at an early date.

EXAMPLES OF THE EFFECTS OF NONPOINT POLLUTION

The effects of nonpoint pollution can best be illustrated by some

examples that will serve to show the magnitude of the problem.

The Great Lakes

The latest International Joint Commission report stated that "even if all the commitments with respect to phosphorus control are met and detergent phosphorus limitations are continued, the full extent of the phosphorus problem will still not have been addressed" (10). In addition, the GAO performed a study on the water quality improvement efforts on the Great Lakes and concluded that unless nonpoint sources are addressed, the Great Lakes water quality objective may not be achieved even if all other sources of pollution are completely controlled or eliminated (9).

Table I, taken from that report, demonstrates the problem. According to the GAO report, while 74% of the phosphorus load originates from diffuse sources, the major emphasis on control has been on point sources.

TABLE I

1976 Phosphorus Loadings by Source (Percent)

	Point			Nonpoint			
Lake	Municipal	Industrial	Total	Air	Upstream	Land	Total
Superior	7	3	10	37	0	54	91
Michigan	39	5	44	26	0	30	56
Huron	11	2	13	23	14	50	87
Erie	39	3	42	4	6	49	59
Ontario	24	2	26	4	41	31	76

Assuming that the relative contributions shown in Table I are correct, it is obvious that if the water quality in the Great Lakes is to be further improved, immediate attention must be given to the control of diffuse sources of phosphorus and other contaminants.

The situation existing in the Great Lakes is not unique. Duda and Johnson report that "the damage being caused by nonpoint sources across the country is staggering" (11). This conclusion is based on EPA and state studies. As pointed out by Carter, "virtually all of the 53 states and territories acknowledge that they have a severe nonpoint source pollution problem" (12).

Mining Problem

As previously noted, nonpoint sources from mining activities can cause serious local problems. An example of the impact of mining wastes to a waterway is shown on Figure 5 (13). As can be seen, the water upstream from the tailing ponds was relatively low in suspended solids. As the water passed the areas where the tailing ponds were being discharged into the river, it picked up considerable amounts of material resulting in concen-

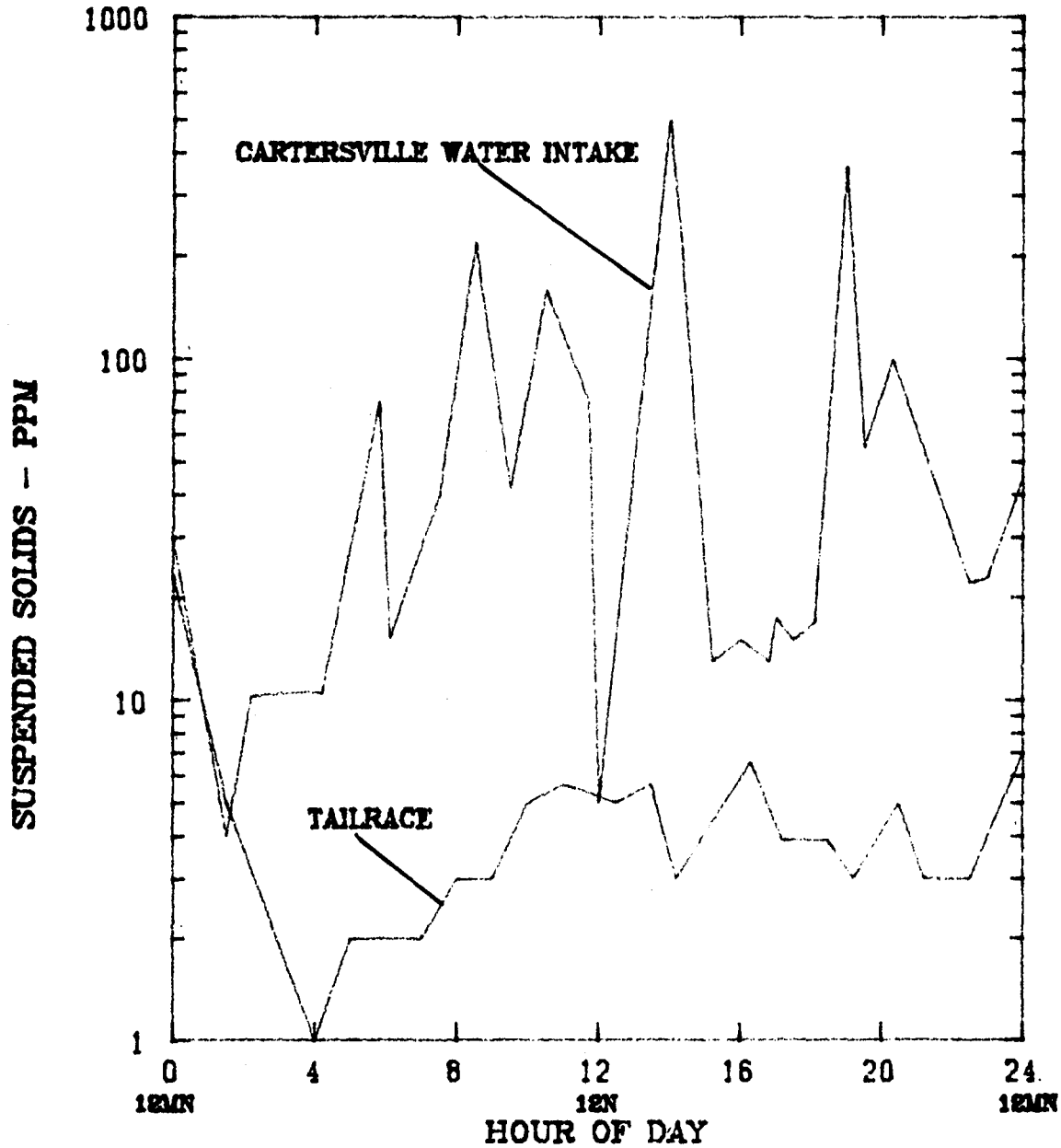


FIGURE 5 VARIATION OF SUSPENDED SOLIDS WITH TIME ON 20 AUG 64 AT ALLATOONA TAILRACE AND CARTERSVILLE INTAKE

trations varying from 4 to 500 mg/l s.s. within a few hours. The resulting impact on the water treatment plant is obvious. In addition, significant concentrations of iron and manganese were added with the solids causing problems in the water treatment plant, the distribution system and at the tap. Barium was also being added to the river via the sediments.

The concentration of sediment reached levels such that the regulatory agency stated that the receiving waters were unfit for reasonable uses including fish and wildlife development and recreation. The river discharges into a reservoir, thus compounding the sediment problems.

Lake Lahonton

An interesting study of the relative importance of nonpoint and point contributions has been the subject of much controversy in Northern Nevada. Severe phosphorus restrictions have been placed on the effluent from the Reno/Sparks Wastewater Plant for several reasons, one of them being because of the alleged impact on Lake Lahonton; which is east of Carson City. Obviously, the degree of phosphorus removal required will have a significant impact on the cost of wastewater treatment.

A comprehensive analysis of the phosphorus loading revealed that the contribution from the Reno Plant was a small percentage of the total load to the lake. A significant part of the phosphorus being utilized by the phytoplankton was found to be that released from the sediments. Other nonpoint sources were also found to be of significance. The water quality problems on this lake have become so bad that the lake was closed for recreation for a period of time.

The impacts of closing the lake are exacerbated by the paucity of recreational lakes in Nevada. Thus, nonpoint pollution control is of the utmost importance to the area.

SUMMARY

The problems of nonpoint pollution have been discussed as well as their sources and impacts on receiving waters. It was pointed out that the lack of adequate water quality data is a major problem. In addition, it was noted that until diffuse pollution is ameliorated, the attainment of the goals of PL 92-500 will not be achieved in many waterbodies.

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USE OF COMPUTER MODELS IN MANAGING NONPOINT POLLUTION FROM AGRICULTURE

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ABSTRACT

Concepts and philosophies of modeling are presented to provide a background for some of the more widely used models for assessing nonpoint pollution from agriculture. A few models are briefly reviewed, and suggestions given for model selection and application. Application of the CREAMS model is presented to demonstrate the use of computer models in evaluating management practices for control of nonpoint source pollution. Runoff, percolation through the root zone, sediment yield, pesticide runoff, and pesticide leaching are analyzed to estimate the effects of agricultural management practices on edge-of-field pollutant loads.

Keywords: Computer models, mathematical models, nonpoint pollution, runoff, erosion, percolation, groundwater, water quality, agricultural practices, management practices, pesticides, plant nutrients.

INTRODUCTION

The development and improvement of the digital computer during the past 30 years has made it possible for engineers and scientists to perform complex repetitive calculations at extremely high speeds. Also, improved storage capabilities now permit operation of complex computer programs, or mathematical models programmed for computer operation. With these developments, scientists have been formulating models to represent physical processes and their interactions in nature that relate to soil and water resources conservation and nonpoint source pollution. Numerous models have been developed during the past 20 years for various uses with later combining of some models as components to form more complex models for evaluating nonpoint source pollution. This paper describes the application of computer models for evaluating nonpoint pollution and developing management practices for its control. The specific model applied is a mathematical description of physical processes.

COMPUTER MODELS AND MODELING PHILOSOPHIES

Backgrounds of the symposium participants are quite varied, and perhaps it is desirable to describe and discuss computer models before reviewing some of the more common ones. What is a computer model? This question is not intended to insult anyone's intelligence; rather, it is intended to get all the participants thinking along the same lines. A computer model is the formulation into a computer language of the modeler's concepts of the physical system and processes. Models are simplifications of the real system, and the degree of simplification may be the result of the modeler's understanding of the processes or the desired degree of accuracy, or the purpose of the model itself. For example, a scientist conducting research on infiltration of rainfall into the soil might very precisely describe mathematically the infiltration process and generation of runoff volume, but might grossly simplify the movement of surface runoff rates and the erosion processes. Likewise, engineers familiar with runoff and erosion processes might disregard or oversimplify chemical adsorption to soil particles and the processes of chemical transport. If the end product from the model is estimation of chemical movement, a scientist may formulate the adsorption/desorption processes in considerable detail, but drastically simplify the runoff and erosion processes. Therefore, one can readily see that a computer model represents the modeler's concepts of the processes involved and the model is developed for a specific purpose.

There are two important factors in any model: space and time. How large (or how small) an area can be represented? What is the time step (increment) in the model? Before discussing these questions, some consideration should be given to the overall problem. To talk about nonpoint pollution generally implies water quality, but where is the interest or concern in water quality? In a lake? In a stream? In a pond? At the edge of a field? Water quality is measured at some receiving body of water, but not at the edge of a field unless that field drains into a pond. When management practices are applied to reduce nonpoint source pollution, where are those practices applied? Is a given practice recommended for every field on every farm in the drainage basin? Management practices are applied on individual fields within the basin, but it is necessary to know the impact that practice will have on the quality of water in the receiving body. The same practice on every field will not necessarily have the same impact on water quality downstream. Then water

quality specialists really need two models: one for field-size areas to examine alternate management strategies, and one for watershed or basin scale to estimate the aggregate affects of field treatment on in-stream processes and downstream loads. Thus, we will discuss models of two scales: relatively small field-size areas and basin scale.

Within a computer model, some time step or time interval of calculation is involved. The time step varies with component and with modeler as well as with scale. For example, a modeler conceptualizing the infiltration process may be concerned with time steps of 1 minute or 5 minutes on field size areas. An ecologist often thinks of a 1-year increment when concerned with nitrogen and phosphorus cycling in basin-size areas. A chemist may think of one day time intervals for pesticides. A one-year time increment would be meaningless for pesticides which have a half-life of 1 to 4 weeks. Another element of time that more nearly reflects scale is the routing interval within or through the drainage areas. Routing, if done on field-size areas, may require increments less than an hour, possibly 15-minute intervals. For basin-size areas, such as a 200-square kilometer drainage in the Coastal Plain or Coastal Flatwood physiographic areas, a 6- or 12-hour routing interval might be sufficient. A routing interval of one day may be adequate for an area of 1000 square kilometers. Differences in infiltration rates from one soil type to another probably has no significance when simulating for 1000-square kilometer drainage area. The impact of management practices become less significant in such large basins unless the treated areas are near the stream or drainage basin outlet. For example, a terrace system or a no-till practice on a 20-hectare field will have negligible effect on the pollutants leaving the basin unless the field is a critical source area adjacent to the outlet. This emphasizes the need for both field- and basin-scale models. The field-scale models are needed to develop management practices, and basin-scale models are needed to simulate the aggregate affects of treating some, or all, of the individual fields.

Another important factor to consider in computer models for nonpoint pollution is the time period for model application. In the design of road culverts or other drainage structures, a "design" type storm is generally used whereby the expected rainfall amount during some predetermined time span is estimated. The resulting runoff rate and volume generated with the model are used to determine the size of structure to be built or the sediment yield from the storm. These are practical applications of models. However, in the case of water quality, the design-type storm generally is not sufficient. If sediment is the pollutant of concern, then a model that simulates for a single design-type storm is satisfactory. If pesticides are suspected pollutants, a daily or continuous simulation model must be used to consider possible ranges in pollutant loads or concentrations of toxic compounds. Since pesticides degrade in time, storms that occur within a few days or weeks after pesticide application are potentially the most dangerous to the environment. If rainfall does not occur within two or three weeks after pesticide application, generally very little losses will occur even with large rainfall events.

Storms that produce small amounts of runoff on the day of pesticide application, or one day after, may result in the most hazardous toxic condition. Large runoff producing events on that same day may result in dilution (lower concentration), and a toxic condition may not exist. Therefore, all possible

conditions must be examined in order to assess potential pollution. Since weather conditions do not repeat exactly each year, it is not sufficient to make model application for a single year, or even two or three years. It is recommended that a representative climatic record be used, 20 years for example, in order that a range of conditions can be considered and risk analysis made to determine the probability of exceeding some potentially hazardous concentration.

Pollution may occur from nitrate leaching into groundwater supplies or leaching below the plant root zone and moving laterally into water bodies. Although potential problems may be greatest shortly after fertilizer application, such potentials may exist most any time due to continuing nitrogen cycling in the natural system. For example, in a climatic region where significant deep percolation occurs in the winter, high losses of nitrate leaching may occur even though several months elapsed since fertilizer application. Combinations of low nitrogen uptake by the previous crop, return of crop residue to the soil, nitrogen mineralization, and possible animal waste application can result in high soil nitrate levels and leaching below the root zone. Here again, long-term continuous simulation is needed to assess pollution potential.

It is impossible to formulate a computer model to exactly represent a physical process or processes. Some model developers think this inexactness can be overcome by using observed data to calibrate a model before making production or simulation runs. The modeler recommends setting some initial value of a few (five or less) parameters, run the model, and compare the model results with observed or measured data. Then parameter values would be changed, one at a time, until the best comparison is obtained between model simulation and observed data. If observed data are not available for a given site or problem area, then data from a similar site can be used (similar soil, similar climate, similar pesticide, etc.). If the model is physically based, that is, represents physical processes, and parameters represent physically measurable quantities, the calibration amounts to "fine tuning" those sensitive parameters. Calibration using optimization techniques with observed data is not required. A model that is not entirely physically based, and parameters are not physically measurable, require more detailed calibration. This results in additional burden, expense, and uncertainties for the model user. Data must be obtained and computer simulations made for calibration. Probably more important is the fact that data for calibration represents some given management practice. What happens to those calibrated parameter values when different management practices are represented in simulation? The user would not have data for alternate practices. Can the user be sure that the original calibration is valid? Probably so if the alternate practices merely represent changes in fertilizer or pesticide application rates and dates. If, on the other hand, the alternate practices involve different types of tillage such as chisel plow or no-till compared with moldboarding, these practices affect infiltration rates, crop residue incorporation, etc. Are the originally calibrated parameter values valid for the alternate practices? That is a decision the user must make. Every model should be validated with observed data where possible. Fine tuning of parameter values is desirable to improve the predictability of a model, but is not essential when considering relative differences between management practices.

A question that potential model users always ask is: "How accurate is the model?". Model developers may claim their model "predicts" within 75%, 80%, or or 90% of measured values of runoff, sediment, or chemical load. What they do not say is that the level of accuracy is only for one location where calibration was performed, the same management practice was used, and only a few storms were simulated. The problem of model accuracy touches on two aspects: the proposed use of the model and model validation. Before discussing these two points, it would be well to discuss measured or observed data.

How accurate are measured data? A model must contain some relationships for estimating runoff from rainfall and it generally will include a soil-water accounting procedure. Measured data really does not include measured rainfall and measured soil-water content. Rather, these are estimated from samples. Rainfall is measured by, at best, a few raingages. For example, there may be two or three raingages in a 10-hectare field. A standard raingage is 200 mm in diameter and approximately 31,400 square mm in area. Thus, rainfall is estimated from samples. Soil water content in that 10-hectare field may be "measured" at 10 "sample" sites using a 25 mm diameter tube. The soil is not absolutely uniform over the field, and there is a system of ridges and furrows. A sample depth of 150 mm on the ridge has a different elevation, and possibly soil texture and water content, than a sample depth of 150 mm in the furrow. Tractor wheels run in some furrows and the soil is more compact than the soil on the ridge. Then soil water content for the field is "measured" by weighting or averaging the content determined for the same depth from all locations. Measurement of rainfall depth and soil water content in samples may be quite accurate, but how well the samples represent the field is another matter altogether. Therefore, the true rainfall and soil water content are not known. Conversely, runoff from the field passes through the gaging station equipped with a time-continuous chart and hydraulic engineers say runoff can be measured $\pm$ 10 to 15% of the actual value.

Now consider sediment yield from the 10-hectare field. All of the sediment cannot be trapped and measured. The only feasible method is to collect samples of sediment-laden flow periodically during a storm event and precisely determine the concentration of sediment. Just as the soil in the field varies, eroded soil varies and the sediment concentration in the flow varies both across the flow and vertically through the flow. The sediment concentration may somewhat pulse in the flow just as we sometimes see runoff itself pulse past a gaging station. Measurement of sediment concentrations in samples may be very accurate. These measured sediment concentrations are applied over some time period to calculate the sediment load. Then sediment yield must be sample estimates rather than measured data.

The same water and sediment samples may be further processed for dissolved and adsorbed chemicals. The same degrees of uncertainty exist relative to samples versus measurements. When chemicals (fertilizers and pesticides) are applied on the 10-hectare field, concentrations of the chemicals can be "measured" in the soil periodically just as soil water content was measured, i.e. by sampling. It was stated above that a ridge and furrow system may be applied on the 10-hectare field and the point was made about elevation of the respective 150-mm sample depths. If a chemical was spread uniformly over the field, incorporated uniformly by tillage, and then the ridge-furrow system was formed,

where is the chemical? All in the ridge? Partially in the ridge? Where vertically? How is "average" concentration in a 150-mm layer "measured" from the samples? How many samples are needed to obtain the average concentration? If the soil surface is rough and cloddy, where is the reference level for samples? Are the clods part of the sample? In stony soils such as in glaciated areas, are the stones taken as part of the soil sample for water and chemical content? When samples are taken for chemical determination, does the sampling device scrape soil from the sides of the hole each time it is entered or withdrawn, thus contaminating the sample? Every one of these questions are very valid and have been asked by good researchers many times over. There is not an exact answer, but in defense of the research profession, techniques are continually improved and refined to the degree that research data are generally quite good if quality control is maintained

Reconsider the runoff samples which are used to "measure" sediment and chemicals. The samples are generally collected by electrically operated mechanical or electronic samplers. These samplers are not fail-safe, i.e. malfunctions will occur: power (battery) failure, intake plugged by sediment or debris, timer malfunction, or switching mechanism failure. In many instances one or two storms account for 90 to 95% of the annual runoff, sediment yield or chemical yield. When "measured" data are closely examined, it is not uncommon to find that equipment failure resulted in only two samples during the storm when as many as 20 to 30 samples should have been collected.

Now that everyone has been thoroughly dismayed by these discussions, how does a modeler define the accuracy of a model or calibrate model parameters? It has been stated by authoritative researchers that sometimes a model is better than the data being used for validation. The foregoing discussion does not imply that a modeler should take the attitude that all data are bad and he has a perfect model that should not be subjected to validation. It simply means that caution must be exercised in the way a model is tested and used in application. Rather than attempt to use models for absolute predictions, they should be used to examine relative response between alternate management practices. Obviously model developers and model users alike hope to achieve the most accurate results possible, but it may be much more desirable to determine if one practice results in lower pollutant loads than another practice with the same long-term climatic record.

REVIEW OF MODELS

Now that everyone thoroughly understands what computer models are and what they can and cannot do, and no one trusts models or modelers, we will review a few of the more commonly used models before demonstrating an application. The review will be brief since we will not consider details of models or comparisons of models.

The Federal Water Pollution Control Act Amendments of 1972, Public Law No. 92-500, was the legislation that attracted concern about pollution from diffuse agricultural and forestry sources, and provided the stimulus for model development. It was not considered initially that mathematical models would be developed and used; in other words, PL 92-500 did not state "Let there be models". There was considerable support for monitoring fields, watersheds or basins in order to evaluate the pollution and any improvement in water quality

resulting from improved management practices. But in practicality, monitoring is expensive and data from a specific practice on a particular soil, in a particular climatic region could not be extrapolated to another practice at the same location or the same practice on another soil type, or in another climatic region, without the use of models. The use of models is really the only way that research data can be transferred from the location of collection to another location.

Engineers have used models for many years to estimate storm runoff from rainfall. Kuichling (1889) developed the rational formula, $Q = CiA$, to calculate urban storm sewer discharge rate, Q, using a runoff coefficient, C, rainfall intensity, i, and drainage area, A. The rational formula is a model, and it can be programmed for a computer very easily. This model is being used today to make quick estimates of peak discharge rate for selected recurrence intervals. In fact, it is an accepted engineering practice supported by legal precedent.

In the 1950's, the U. S. Department of Agriculture, Soil Conservation Service (1972) developed a procedure, referred to as the SCS curve number method, to estimate direct surface runoff volume. Although the commonly used procedure is in the graphical form, mathematical relationships are available for use in computer models.

Wischmeier and Smith (1965) published the first comprehensive erosion equation. The Universal Soil Loss Equation (USLE), which estimates long-term average-annual gross erosion, has become the basis of many erosion models.

One of the earliest published hydrologic computer simulation models was the Stanford Watershed Model (Crawford and Linsley, 1962). Although the model was developed for large watersheds or basins, it was later adapted for field size areas.

Negev (1967) developed the first erosion/sediment yield counterpart of the Stanford Watershed Model. Negev utilized some of the basic concepts of Wischmeier and Smith (1965) on rill and interrill (sheet) erosion, but also included a component for channel erosion.

Glymph and Holtan (1969) developed an infiltration-based hydrologic model to determine the effects of management practices on streamflow. The computer model, called USDAHL (U. S. Department of Agriculture Hydrograph Laboratory), later became the hydrologic component for a nonpoint pollution model.

Other hydrology and erosion/sediment yield models have been developed, and have been used successfully for specific purposes. However, these aforementioned models are the more important ones that later were used in the development of nonpoint source pollutant models that include plant nutrients and pesticides as well as sediment

Under the terms of PL 92-500, the Environmental Protection Agency (EPA) had responsibility for agricultural chemical pollution and its reduction. Therefore, the EPA had a major role in research and development, both "inhouse" and contracting, relative to nonpoint pollutant models. The U. S. Department of Agriculture, Soil Conservation Service, was responsible for reducing sediment

as a pollutant and as a chemical pollutant carrier. These respective roles were later changed, but for the purposes of model development and review in this paper, the dual agency roles were important.

The first EPA-contracted model for nonpoint pollution was the Pesticide Transport and Runoff (PTR) model developed by Crawford and Donigian (1973) for field-size areas. The PTR model used modifications of the Stanford Watershed Model (Crawford and Linsley, 1966) and Negev's (1967) erosion/sediment yield model. Although the hydrology and sediment yield components were developed for basins, they were applied on field-size areas with pesticide transport incorporated. A real danger may exist when scaling a component up or down, and that is over sensitivity of some parameters, or distortion of the parameters out of the range for which they were originally developed and tested. Another common pitfall is that a component previously developed may not be consistent or provide the exact information needed in a developing component. It is more desirable and efficient to develop all components simultaneously.

Constraints of various kinds can result in not being able to develop a comprehensive model on the first trial, and this was no different for PTR. In testing and further development of PTR, Donigian and Crawford (1976a) incorporated, among other items, plant nutrient transport and produced the ARM (Agricultural Runoff Management) model.

During the development of ARM, it was realized that a simpler version was needed to operate in connection with urban models. Thus, Donigian and Crawford (1976b) developed the NPS (NonPoint Source) model. The hydrologic algorithms of NPS were used with the Hydrocomp Simulation Program (HSP) to simulate nutrient loadings in surface runoff from urban and agricultural watersheds.

The need for basin-scale models resulted in the natural evolvement under contract from EPA, and the development of the HSPF model (Johanson et al., 1980). HSPF is the principle basin-scale model in use today.

At the same time that some of the EPA contracting occurred, efforts were made in the U. S. Department of Agriculture to develop models that reflected expertise in hydrology and erosion/sedimentation. Frere, Onstad, and Holtan (1975) used the USDAHL model (Glymph and Holtan, 1969) and a modification of the USLE (Wischmeier and Smith, 1965) to develop the Agricultural Chemical Transport Model (ACTMO). ACTMO was a basin-scale model, but has been applied on field-size areas as well. Somewhat simultaneous with the ACTMO development, Bruce et al. (1975) developed a parametric model for water-sediment-chemical (WASCH) runoff for single storm events on field-size areas. The WASCH model required data for parameter calibration, and its chemical component only considered pesticides.

Lack of documentation and validation of ACTMO and the data requirement for calibration of the WASCH model resulted in little use of these two models. In order to fulfill their role in research to the Soil Conservation Service, the Agricultural Research Service established a national project that culminated in development of the CREAMS model (Knisel, 1980). CREAMS is a physically-based model that can be used, without calibration, to compare edge-of-field pollutant load response for alternate management practices. The SCS curve

number method (U. S. Department of Agriculture, 1972) is used as one option in CREAMS to simulate daily runoff from daily rainfall. A second option (Smith and Williams, 1980) uses a modified Green and Ampt (1911) infiltration equation to simulate runoff from time-intensity rainfall data. The erosion component is the Foster et al. (1977) modification of the USLE (Wischmeier and Smith, 1965) on a storm basis. CREAMS also includes plant nutrients (Frere et al., 1980) and pesticides (Leonard and Wauchope, 1980). A paper to be presented later in this symposium (Leonard et al., 1985) will describe modifications of CREAMS to consider pesticide leaching. CREAMS has been used widely, both nationally and internationally, to develop or select management practices to reduce nonpoint pollutants delivered to the edge of the field.

Steenhuis and Walter (1980) used the SCS curve number method (U. S. Department of Agriculture, 1972) and the USLE (Wischmeier and Smith, 1978) for the hydrology and the erosion components of their CPM (Cornell Pesticide Model). CPM actually was developed to make use of most any hydrology and erosion components that generated volumes of water and loads of sediment.

The above referenced models do not include groundwater components. Enfield et al. (1983) developed the PESTANS model to provide a one-dimensional projection of vertical chemical movement through the unsaturated zone. Although PESTANS treated very simplified conditions, it led to the later development of the Pesticide Root Zone Model (PRZM) by Carsel et al. (1984). PRZM contains the SCS curve number method (USDA-SCS, 1972) to partition rainfall into runoff and infiltration, and contains Williams and Berndt's (1977) modification of the USLE (Wischmeier and Smith, 1965) to calculate sediment yield. PRZM was developed to evaluate movement of pesticides within the root zone and the lower unsaturated/saturated zones.

Leonard et al. (1985) modified the CREAMS model (Knisel, 1980) to calculate pesticide movement within the active root zone. Although the modifications do not constitute a groundwater model, percolation of pesticides below the active root zone provides estimates of loadings to the vadose zone.

A number of models have been developed for erosion and sediment yield only, and do not include plant nutrients or pesticides. Beasley et al. (1977) developed the Areal Nonpoint Source Watershed Environment Response Simulation (ANSWERS) model to consider erosion/sedimentation in a basin for a design-type storm. ANSWERS has been used to identify critical areas of erosion for protective practices. Also, ANSWERS has been applied on field-size areas to examine variability of erosion and deposition within heterogeneous fields.

Simons et al. (1977) modified the USLE (Wischmeier and Smith, 1965) to develop a procedure for estimating source erosion, and developed a sediment routing component for calculating basin sediment yield. The model operates with a design-type storm.

Williams and Hann (1978) developed a computer model to analyze runoff, sediment, and plant nutrient losses from basin-size areas. Although the model was developed to consider economic factors, it provided information on sediment and nutrient losses from a basin.

The models herein reviewed obviously do not represent an exhaustive list, but they do represent those models most commonly used by action agencies today. Furthermore, it is easy to see what basic hydrology and erosion components are included. Due to this author's involvement and experience with CREAMS, this model was selected for demonstration of how a computer model can be used to develop management practices.

MODEL SELECTION AND APPLICATION

Model Selection

Model developers never envisioned that models, either field scale or basin scale, would be used to identify water quality problems. That is, it was not anticipated that a model such as, for example, HSPF (Johanson et al., 1980) would be applied on some basin for existing management practices, and be able to say "there is a problem of toxicity of pesticide 'X' in this water body". On the contrary, it was envisioned that after a water quality problem was identified, such as concentrations of pesticide "X" at toxic levels, models would be useful in analyzing alternative practices for alleviating the problem. Intuitively, a person can generally be more specific about the problem. For example, pesticide "X" is highly adsorbed to the soil, and erosion/sediment transport is essentially the only method of transport. Then management practices can be evaluated on the basis of sediment and pesticide delivered to the edge of the field or to the outlet of the basin. This sounds simple and straightforward, but caution must be exercised. If management practices are developed to reduce erosion and sediment yield, include conservation tillage or minimum tillage, a new spectrum of herbicides, insecticides, nematicides, or fungicides may be required. If any of the new pesticides are very mobile (in water), the improved management practices may actually result in more pollutant runoff. Another possibility may be that little of the mobile pesticide is lost in surface runoff, but significant loads may be transmitted to the aquifers. Therefore, the full range of management effects should be evaluated with models over some extended climatic record. These kinds of situations are common occurrences that must be considered totally.

A large number of models has been developed, and this is both good and bad. It is good in the sense that the model user has several potential tools from which to choose for their job. It is bad in that the burden is upon the user to decide which model is appropriate for the problem at hand. The user must learn the model concepts, assumptions, and limitations and whether or not it will adequately treat the problem of concern. The user also must carefully consider the input requirements, if required data are available, how much time is required to develop parameter sets, what is involved in changing parameters to reflect management changes, and what the computer requirements are relative to storage and run time. In essence, the model user must know as much as the developers of several models. A person really cannot learn a model until they actually run it, get familiar with it, and feel comfortable in what they are doing. In other words, a model user must become an expert before really understanding a model. This does not mean that no one should use a model before they become an expert. A potential model user oftentimes is in a dilemma as to which model or models should be used. Some thought has been given to establishing an independent unbiased group (council) to test and run available models to be able to recommend model A, model B, or model C for a specific

user's problem. This is not really feasible, but it points out the fact that decisions are not always easy to make.

Model Application

Details of the CREAMS model will not be given in this paper. Model concepts, assumptions, limitations, or specific physical processes are not important in demonstrating the model applicability. This information is fully documented by Knisel (1980), and readers are referred to that publication if interested. However, a schematic representation of the physical system addressed by CREAMS is shown in figure 1. The field (watershed) is depicted as being made up of some geology, soil, and topography which is acted upon by natural climatic input (precipitation, radiation, and temperature). The farmer superimposes his management input (crops, tillage practices, and pesticide and fertilizer applications), and the field responds to produce such output as surface runoff, erosion/sediment yield and associated dissolved and adsorbed chemicals. Some fraction of precipitation infiltrates into the soil, and a portion of the infiltration may go to percolation which also may contain dissolved chemicals. Thus, figure 1 is a flow chart of the system that was used as a guide in the development of CREAMS.

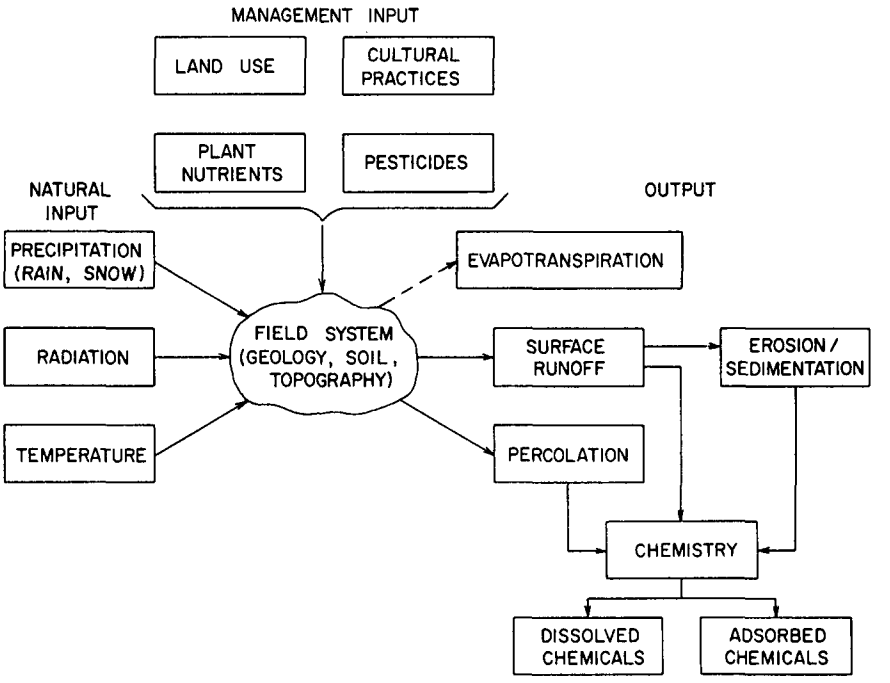

Figure 1. Flow chart of system for evaluating nonpoint source pollution.

A very simple scenario is used to demonstrate application of a computer model to select management practices for reduction of nonpoint pollution. In order to not compound the system response among crop rotation, year-to-year climatic changes, and fertilizer-pesticide-tillage interactions, a simple continuous crop was used. Continuous corn was simulated in a conventional tillage system. A 20-year climatic record at Tifton, Georgia was used for simulating runoff, percolation, sediment yield, pesticide runoff, and pesticide leaching on a loamy sand soil in the Coastal Plain. Rainfall at Tifton averages about

1200 mm per year. Several of the Coastal Plain soils are subject to development of plow pans which result from continued tillage at the same depth year after year. These dense compaction layers are similar to the clay pan soils that occur in Wisconsin. The layer restricts root growth and water movement at a depth of approximately 20 cm. This is really the effective rooting depth for a crop unless tillage practices are used to break the pan condition. The main difference between the Wisconsin clay pan and the Coastal Plain plow pan is that the clay pan is more restrictive to water movement. Also, chisel tillage is more effective in breaking the plow pan.

For demonstration purposes, three management practices were simulated:

(1) continuous corn, conventional moldboard tillage (no surface residue), 20 cm root depth;

(2) continuous corn, chisel 30 cm deep, residue management, 45 cm root depth;

(3) continuous corn, chisel 30 cm deep, residue management, 45 cm root depth, with a rye winter-cover crop.

In all three management practices, two generic pesticides were considered to be broadcast on the surface at planting time. The two pesticides have the same solubility (10 ppm) and same half-life (approximately three weeks). Pesticide A is transported mainly in the water, whereas pesticide B is more strongly bound to the soil particles and organic matter and thus move with sediment. These management practices are not recommendations, and the results would differ for other soils, other climatic regions, and other cropping systems. They are useful, however, to demonstrate the use of a model in managing nonpoint pollution from agriculture.

Simulation results of direct surface runoff for the three management practices are presented in figure 2. The largest accumulated runoff volume was simulated for management practice 1 representing a 20 cm root zone. The second highest accumulated runoff was simulated for management practice 2. The chisel tillage in practice 2 affected three parameters in the model: (a) increased the rooting depth, (b) increased the hydraulic conductivity of the root zone, and (c) reduced the evaporation potential due to crop residue left on the surface. These changes resulted in a significant reduction in runoff by about one third compared with practice 1. The rye winter cover of practice 3 resulted in some plant transpiration, but only about 7% additional reduction in runoff occurred as depicted by the bottom line in figure 2. These results represent the runoff response for the three management practices which has further implications relative to erosion and pesticide losses.

Results of simulation of percolation below the root zone, i.e. 20 cm for management practice 1 and 45 cm for management practices 2 and 3, are shown in figure 3. The same order differences as for runoff are apparent: more percolation for practice 1, intermediate for practice 2, and least for practice 3. This requires a little further interpretation. If runoff was less for practice 2 than practice 1, how can percolation be less, also? The explanation is that for practice 2, there is an increased soil reservoir (storage) and also more plant use of water. It should be noted that the difference in percola-

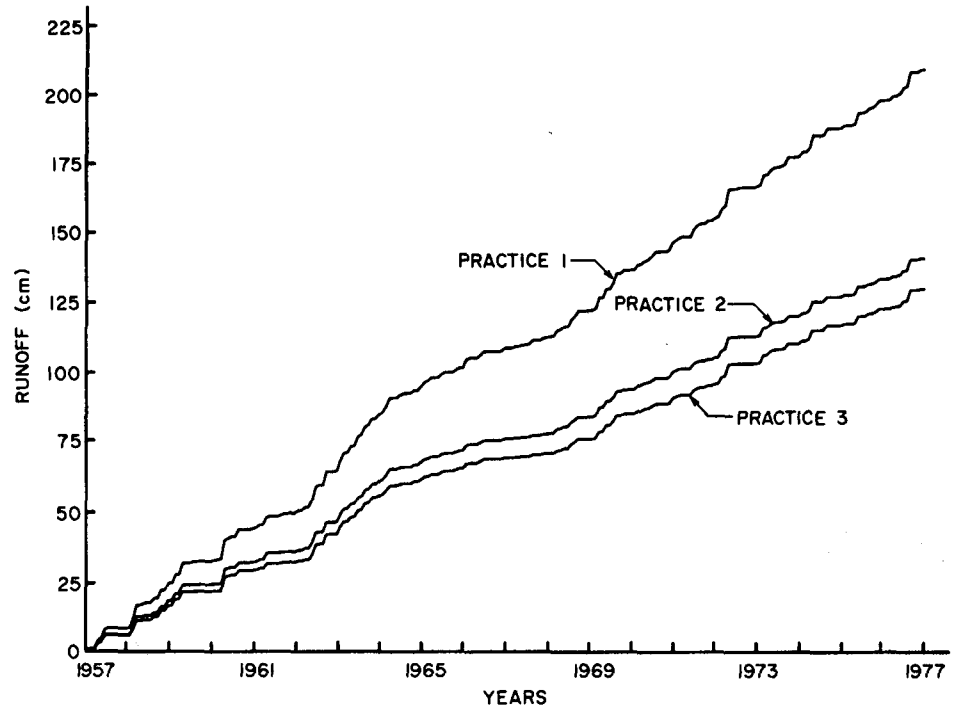

Figure 2. Accumulated runoff simulated with the CREAMS model, 1958-77.

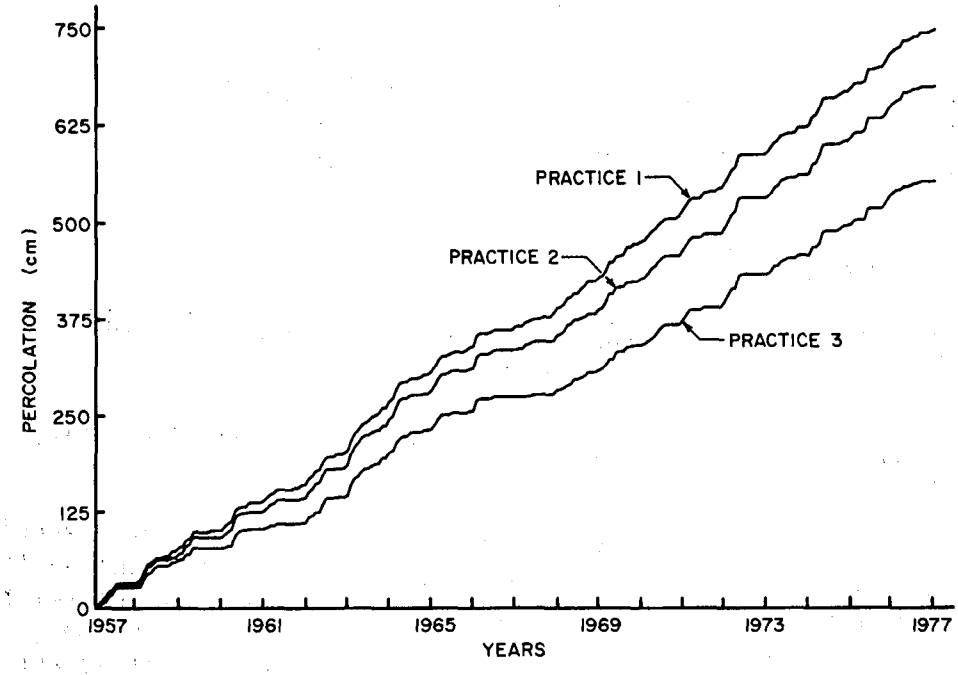

Figure 3. Accumulated percolation simulated with the CREAMS model, 1958-77.

tion between practice 1 and practice 2 is relatively small compared with the difference in runoff for the same practices (10% for percolation). The addition of a winter cover crop in practice 3 results in water uptake, thereby reducing soil water stored, during the time that significant amounts of rainfall and percolation normally occur. Thus, percolation for practice 3 (bottom line of figure 3) is reduced an additional 16%. Another important point about figures 2 and 3 are the relative magnitudes of runoff and percolation. The 20-year accumulated values show that percolation is almost four times greater than surface runoff. This has significant implications in nonpoint pollution as will be shown later.

Figure 4 shows the cumulative 20-year simulated sediment yield for the three management practices. Sediment yield for practice 1 is quite high (figure 4) when considering the low volume of runoff (figure 2). The reduced runoff and residue management from practice 2 resulted in a 43% reduction in sediment yield. An additional 10% reduction resulted from the winter cover crop for practice 3 (figure 4).

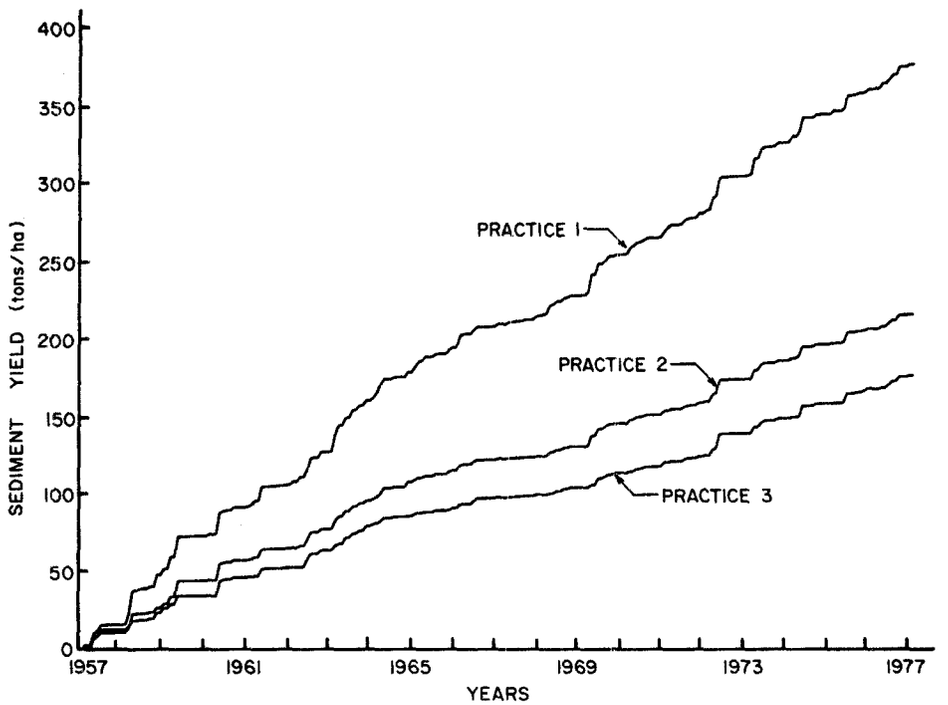

Figure 4. Accumulated sediment yield simulated with the CREAMS model, 1958-77.

Pesticide losses were simulated for the three practices. The two generic pesticides were surface applied on the same date each year (at planting time for corn), and applied at the same rate. Solubility (10 ppm) and half-life (approximately three weeks) were the same for both pesticides. The only difference was the adsorbtivity: pesticide A had a partitioning coefficient of 0.2 and pesticide B had 200. Runoff and sediment losses of pesticide A were negligible (less than 1 g/ha cumulative for 20 years). However, since pesticide A had a low adsorbtivity, a relatively large amount was lost due to percolation below the root zone, 26 kg/ha in 20 years for management practice 1

(figure 5). The 20-year percolation loss was 43% of the amount applied. Although percolation volume was reduced by about 10%, for management practice 2, it resulted in a 41% reduction in percolation loss of pesticide A. Since the pesticides were applied each year of simulation on the date of corn planting and the half-life was about three weeks, percolation reduction by the winter cover crop was not effective in further reduction of pesticide losses (management practice 3).

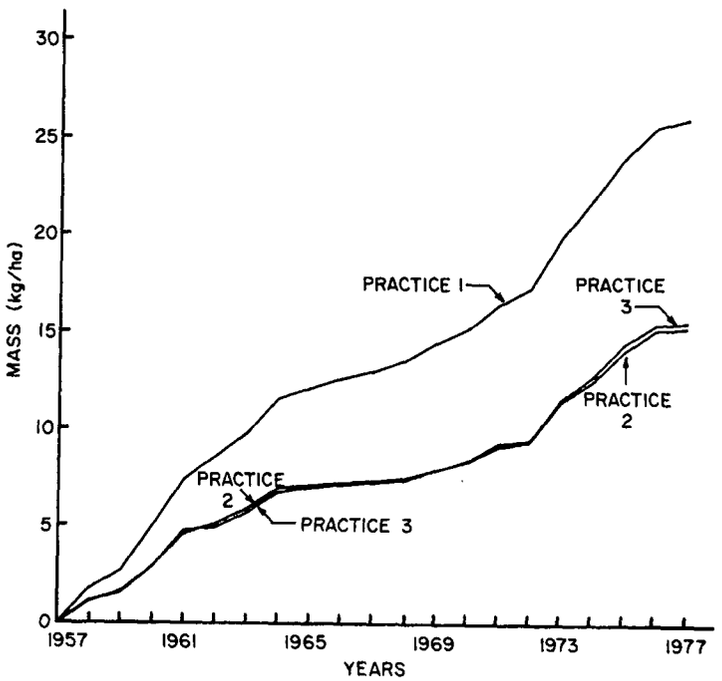

Figure 5. Accumulated leaching loss of pesticide A simulated with the CREAMS model, 1958-77.

The 20-year sediment-associated loss of pesticide B for management practice 1 amounted to less than 100 g/ha, whereas runoff loss was 1640 g/ha and percolation loss was 16,900 g/ha. Data from simulation of surface runoff losses of pesticide B are shown in figure 6. The 20-year runoff loss for practice 1 was 2.7% of the amount applied. Reduced runoff from management practice 2 resulted in a 37% reduction in the runoff loss. Although the runoff reduction for practice 3 compared with practices 1 or 2 was very small (figure 2), the pesticide runoff loss was significantly reduced by practice 3, a 53% reduction from that of practice 1.

Figure 6 shows another very important point made earlier in this paper. The flat portions of the curves between 1965 and 1972 differ drastically from the 1958-64 and 1974-77 periods. If a 5-year period between 1965 and 1972 had been selected, relative difference between practices would have been negligible. If the 5-year period (1960-64) had been selected for simulation, projected difference would have been exaggerated. It is important that longer, more representative climatic records are used. In the 1965-72 period, either runoff was low each year, or runoff producing rains did not occur for some time after pesticide application.

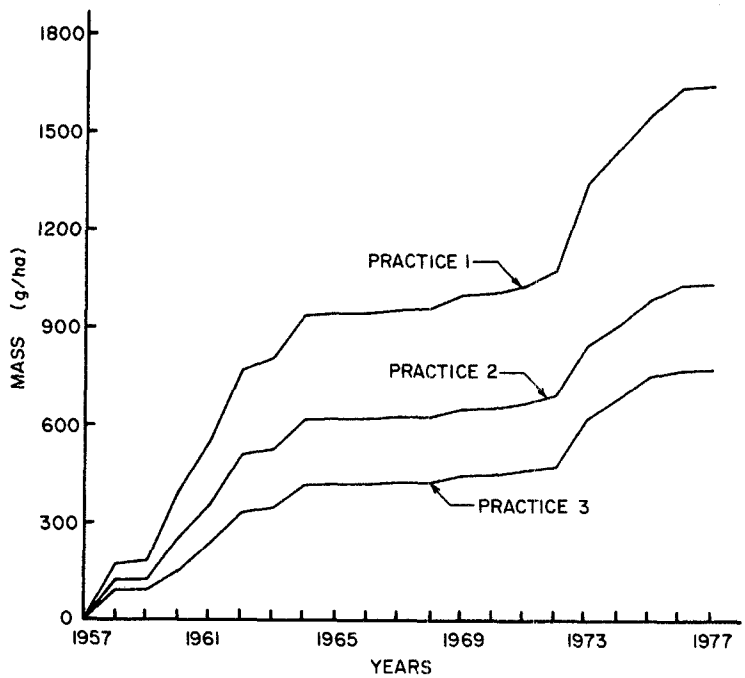

Figure 6. Accumulated surface runoff loss of pesticide B simulated with the CREAMS model, 1958-77.

The results of the several CREAMS model simulations do not show that there is or is not a water quality problem because CREAMS is not a water quality model. The simulations were not intended to be used to make such claims. They are merely edge-of-field and bottom-of-root zone losses for three different management practices. If there had been a problem in water quality, as determined at some off-site location, this kind of model application could be made to estimate response from existing and alternative management practices. Even then, the simulation results would not indicate an offsite water quality problem and its "cure". A basin-scale model would have to be applied for routing the aggregate field response to the receiving waters -- river, lake, estuary, or etc. Thus, one has to be aware of the problems, needs, tools, benefits, pitfalls, and other obstacles and opportunities to address nonpoint pollution problems.

SUMMARY

Computer models are valuable tools with which sound professional judgment can be used in evaluating management practices and treatments for nonpoint pollutant loads. The several models currently being used all have their strengths, weaknesses, assumptions, and limitations. None are predictive in terms of absolute quantities. They can be extremely useful to consider response from different management practices over a long representative climatic period. Also, they can be used to extrapolate research results from the specific research location to another location (soil, climate, management practice) of interest.

There are no water quality models. There is no model that this author knows about that can be used to say "these management practices cause eutrophication of this lake", or that "these practices caused pesticide concentrations that are toxic to fish (or man) in this stream". What the computer models can tell the user is that certain pollutant loads may occur over some period of time as a result of some practices. Models, used with sound professional judgment, can be valuable aids in making sound management decisions.

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ROLE OF MATHEMATICAL MODELS IN DESIGN AND SELECTION OF BEST MANAGEMENT PRACTICES FOR CONTROL OF POLLUTION FROM URBAN AND URBANIZING AREAS

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ABSTRACT

Mathematical models are an integral and indispensable part of any nonpoint pollution abatement project. There are two types of the models which have been developed: screening overview models, and detailed hydrological models. These models were divided into five levels according to their complexity.

The paper discusses the types of models and their usefulness in identification and abatement of nonpoint pollution from urban and urbanizing areas. The approach is demonstrated on a practical example of a use of a medium complexity hydrological model on a typical urban watershed.

Keywords: Complex models, hydrological models, urban runoff models, water quality models, nonpoint pollution, nonpoint pollution models, runoff, pollutant build-up, pollution by urban runoff, urban stormwater modeling, mathematical models.

INTRODUCTION

Hydrological mathematical models have become an integral part of urban drainage engineering. Recent advancement of small computer hardware technology and availability of relatively inexpensive microcomputers have meant that almost every city engineer's or consultant's office can now possess a computer capable of an advanced application of urban hydrological models.

There are now a large number of models available from various agencies, universities and software vendors that range from simple applications of basic hydrological procedures to more advanced and more complex urban runoff quantity and quality models. A user may have a problem of selecting a model that would be reliable, perform the tasks that are needed, would have adequate documentation and support, its acquisition would not be costly, and, finally, select a model that has been tested and verified.

Why Models Are Needed

One question model advocates must first answer is "why we need the models?" There are a number of reasons for and against the use of models, but the most obvious one is that models can go far beyond the results of any monitoring program. Monitoring programs are always time limited. A typical very expensive monitoring program (such as those conducted during the National Urban Runoff Project) may cover two years of monitoring on a few experimental small watersheds. Chances are that neither of the two years will reflect extreme or average conditions required for the design. In this case the loading estimates from the monitoring program may be misleading, considering the probabilistic hydrologic nature of the nonpoint pollution generation process. In a dry year, the loading will be low, in a wet year too high. In this case the model is a tool that can help to organize the data, and bring them in line with the average or extreme hydrological conditions.

Another extreme view is to rely solely on models and disregard the data. As it will be shown in this article, monitoring field data are needed to calibrate and verify the model. A model that has not been calibrated (tuned) loses its practical usefulness because the answers given by the model can be inaccurate. In an ideal and most common situation both monitoring and modeling programs should be a part of a project. Furthermore, the monitoring program should be designed to provide the necessary calibration and verification data.

TYPE AND BASIC STRUCTURE OF THE MODELS

Nonpoint pollution models are part of a category of "loading models," which represent and simulate generation and movement of water and its pollution content from the point of origin to watercourse. These models may interface with water quality models for receiving waters and provide input concentrations and flow rates (hydrographs and quality loading histograms),

Urban runoff and nonpoint pollution models fall generally into two categories that can be further subdivided, (Novotny and Chesters (1984); Novotny (1984); Sutherland (1980)). The first broad category includes more simple screening models and the second category covers more complex computerized hydrological assessment models. A good review of hydrologic fundamentals of urban runoff models was presented by Hall (1984) and by the APWA (1981).

Screening Models

Screening models rely on -- or evolved from established traditional urban drainage design procedures and/or statistical results of long term monitoring programs. They can be further subdivided into: I. Simple statistical (desktop) procedures and unit loads with no interaction among the processes, and II. Simplified procedures with some interaction among the processes.

The hydrological fundamentals of these models are not complete or even may not be considered and some components are missing (for example a procedure may simulate only runoff from impervious areas and neglect subsurface flow). These models do not commonly require calibration and verification with measured data.

Screening models and computerized procedures often require a determination of a design storm. The magnitude of the design storm and the frequency are usually selected by the designer according to local or state building codes and flood protection insurance regulations.

Examples of the procedures included in category I include the Rational Formula for estimating peak runoff rate and the British Transport and Road Research Laboratory (TRRL) Hydrograph Method (Watkins (1982), Terstriep and Stall (1969)). These simple models estimate design flows for sewer systems. Simple water quality estimation procedures for urban areas rely on estimated unit loadings of pollutants. A unit loading is a simple value or function expressing pollution generation per unit area and unit time for each typical urban land use. The units are expressed in kg per hectare per year or season. In some cases, the unit loads for developed urban areas are expressed as a mass of pollutants per unit of curb length. Then the unit load can be obtained by multiplying the curb loads by curb density (curb length/area). The magnitudes of unit loadings and simple procedures can be obtained from a publication by McElroy et al. (1976).

A number of Simplified Hydrological Computertized procedures (modeling category II) have been developed in the past fifteen years. The most known hydrological procedures include the Soil Conservation Service - runoff hydrograph method (SCS(1975)) in the U.S. or the Wallingford Procedure developed in England and described by Price (1984). The SCS Method provides a means of estimating surface runoff volume and hydrograph using a semiempirical relationship between the rainfall and runoff based on a runoff curve number that depends on land use and soil types, vegetation, and antecedent moisture conditions. The runoff can then be routed overland by a simple triangular unit hydrograph.

Water quality components in the category of screening models usually rely on the Universal Soil Loss Equation (Wischmeier and Smith (1965)) that estimates soil loss from pervious surfaces and use the pollutant build-up-washoff concept for impervious urban areas (Novotny and Chesters (1981)). The modeling methodologies have been published by Johnson et al. (1978), Sonzogni et al. (1980), and Haith and Tubbs (1980).

In spite of their questionable accuracy and reliability the simple screening procedures have found wide application in areawide pollution abatement planning efforts. One reason explains this popularity: these procedures provide a simple mechanism and quick answers to pollution problems of large areas where more complicated efforts would fail because of the enormous amount of information required. The screening concepts enable indentification of hydrologically active nonpoint pollution generation areas within a large watershed that should be subjected to futher studies and subsequent management.

The effect of various abatement measures on the nonpoint pollution loads using the simplified screening procedures can be estimated only very approximately by using simple rule-of-thumb factors and pollution removal efficiencies. For example, it may be assumed that an effective street sweeping program will reduce pollution loads from a high density residential urban area by 20 to 30 percent or that a properly designed wet retention pond may reduce particulate pollution loadings by 50 percent.

Hydrologic Simulation Models

The more complex hydrologic simulation models in the second group represent a description of the hydrologic rainfall-runoff transformation process with associated erosion, pollution build-up and washoff, and other quality components. These models estimate the pollution generation from an area based on the accumulation during a dry period and their subsequent transport by rainfall and convective water movement.

Similarly to the hydrologic models used for modeling nonpoint pollution from agriculture, the scope of such a model can cover a subdivision or a small section of a city. Application of these models to entire large urban watersheds is usually tedious requiring a break-down of the watershed and a step-by-step additive simulation. Most of the urban models are limited in their scope and in a single application they are limited to relatively small watersheds.

Urban Hydrological models have the following basic components (Figure 1):

1. The surface runoff generation component that describes the transformation of rainfall into excess rainfall and surface runoff. Most of the nonpoint pollution in urban areas originates from so called hydrologically active areas, that is areas from which surface runoff is generated. The purpose of this component is to locate these areas and to determine the magnitude of the surface runoff.
2. The soil and ground water movement component that is so important for agricultural models, is in most urban models represented by simple relationships and empirical equations or is not included.
3. Accumulation, removal and washoff of pollutants from impervious surfaces. This component balances the particulate pollutants that accumulate near

the curbs of paved roads during the dry days and estimates their washoff into the runoff during storms. The accumulation mass is a result of deposition of particulate pollutants from various sources (atmospheric dry deposition, litter deposition, traffic, salt application) and their removal from the curb storage by wind, traffic induced turbulence and street sweeping.

4. Soil erosion component estimates soil loss from pervious areas. The Universal Soil Loss Equation (USLE) is the most common model used to represent the process.
5. Pollutant adsorption components that in agricultural models is represented by complex adsorption-desorption relationships, is in most urban models replaced by factors that relate the mass of a particular pollutant to that of total suspended solids.
6. Runoff and pollution routing component transforms the excess rainfall and its pollution content into a inlet or stream hydrograph or quality

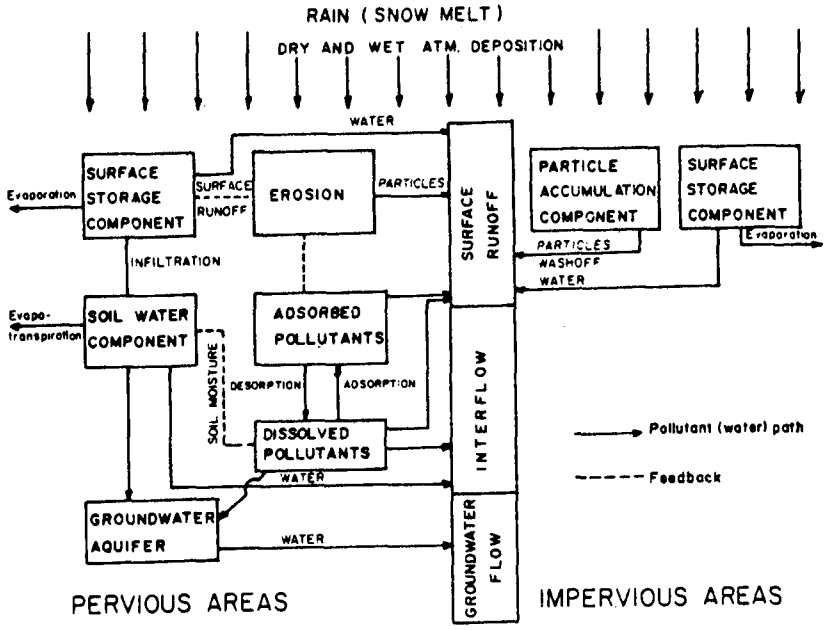


Figure 1 - Components of Watershed Hydrologic-Nonpoint Pollution Models

histogram. Common techniques for excess rainfall -- surface runoff routing (convolution) rely on a synthetic unit hydrograph. In the absence of a detailed sediment redeposition and pollution trapping description during the overland flow rate, delivery and enrichment factors are used to relate the sediment and pollutant yields in the receiving water body to those at the source.

Following Chow's (1972) definition of hydrological models, most of the urban runoff quantity-quality models can be classified as deterministic lumped parameter models. Deterministic means that for one set of inputs there is only one set of outputs. This differentiates them from so called stochastic hydrologic models whereby outputs from the model may vary depending on the probability distribution of the inputs and model parameters.

The lumped parameter models treat the watershed or a large portion of it as one unit. The various characteristics of the modeled areas are lumped together (for example slopes are represented by an weighted average slope) often with the use of weighting empirical equation.

Models have been designed to run on an event or continuous basis. Discrete event modeling simulates the response of a watershed to a major rainfall or snowfall. Continuous models simulate flows and pollutant loadings over an extended period of time that may range from a season to several years. The principle advantage of event modeling over continuous simulation is that it requires relatively little meteorological data and can be operated with a shorter computer run time. A principal disadvantage of event runoff modeling is that it requires specification of the design storm and the antecedent dry period, determining subsequently the amount of accumulated pollutants and soil moisture conditions. This a priori implies that the recurrence interval for the rainfall and runoff and pollution content are the same.

The principal advantage of continuous modeling is that it provides long time series of water and pollutant loadings that can be statistically analyzed as to their frequency. A principal disadvantage of continuous modeling is that it requires long simulation runs, this imposes restrictions on the amount of alternatives that can be investigated. It also requires historical data on precipitation often in less than hourly intervals, which may not always be available.

Classification of urban hydrological nonpoint pollution models

As to their complexity the hydrological urban runoff quantity-quality models can be divided into the following subgroups (Sutherland (1980)):

III. Simplified models - either event oriented or continuous. In these models some hydrologic components are missing some other are represented by a simple relationship. There will be some interactions among the processes but this is usually done in an approximate way. Water and pollutant routing may not be included.
IV. Sophisticated single event simulation models. These models provide for extensive interactions among the various processes that are important

in the simulation of stormwater quantity, including infiltration, overland flow, gutter and pipe flow, and flow routing. The quality components may be less sophisticated, relying on simple pollutant build-up-washoff concepts. Street sweeping effects can be included only during a predetermined relatively short antecedent dry period, and, in general, such models may not be suitable for evaluation of typical long term best management practices for controlling urban runoff pollution. Their primary use is in design and evaluation of urban drainage systems.

V. Sophisticated continuous models. These models provide for extensive interaction among the various processes that are important in the simulation of both stormwater quantity and quality. Routing of flow and quality are included. Modeling of intermittent dry periods enables to account for soil moisture, pollutant build-up and removal and other important processes occurring between the rainfalls.

Examples of simplified models in category III include the Corps of Engineers' (Anon. 1975) Storage-Treatment-Overflow-Runoff Model (STORM), Wisconsin Urban Runoff Model developed by this author (Novotny (1983)) and a number of proprietory models. Most of these models simulate runoff either by Modified Rational Formula (STORM) or by the SCS Runoff Curve Model (Wisconsin Urban Runoff Model). Most of these models simulate street drainage and storage-treatment systems.

The most known models in the category IV (event oriented models) are the EPA's Stormwater Management Model - SWMM (Huber et al. (1975)) and Illinois Urban Drainage Area Simulator-QUAL-ILLUDAS (Terstriep and Stall (1974)).

SWMM is a widely used stormwater runoff quantity-quality model which is readily available from the U.S. EPA laboratory in Athens, Georgia. It has an excellent documentation and its continuously maintained and updated. The model can analyze single storm events on multiple catchments. The model is primarily used for large urban projects where considerable data are available for calibration, verification and sensitivity analyses.

The ILLUDAS model is available from the Illinois State Water Survey. The model has been widely distributed in the U.S. and Canada as well as in many other countries. ILLUDAS uses a design storm hydrograph and physical basin characteristics to predict stormwater runoff hydrographs from both paved and unpaved areas. Its use for water quality prediction is limited.

The Fortran version of the HSP-F model developed by Hydrocomp, Inc. (Hydrocomp (1979)) is an example of the level V models capabable of simulating hydrologic time series of runoff quantity-quality events. This version of the model is again available from the U.S. EPA Environmental Laboratory in Athens, Georgia. The model had originally evolved from the Stanford Watershed Model. It is a large model and similar to SWMM, it requires a considerable effort when applied to a watershed. The model can be applied to urban areas in a fashion similar to a previous Hydrocomp model called Nonpoint Simulation Model - NPS (Donigian and Crawford (1976)).

RELIABILITY AND USEFULNESS OF THE MODELS

The mathematical models are only a rough approximation of the real world. All real world processes are more-or-less stochastic in nature, meaning that it is not possible to exactly predict the outcome of a series of events in the future. This is especially true for hydrologic processes such as flow and pollution generation. Ideally, such processes should be modeled using statistical techniques and models. Yet, almost all urban runoff quantity-quality models that have been developed recently are deterministic.

A common error made by many planners and users of the results of modeling is that they accept simulation results as true and accurate. In order to avoid disappointment and court challenges, users should be aware of the limited accuracy of the models.

Figure 2 shows schematically reliability of various types of models and relative magnitudes of errors. The most accurate models (± few percent) are hydrologic models simulating runoff from small, uniform impervious areas, the least reliable (on order of magnitude of more) are water quality models for large watersheds simulating pesticides and similar complex chemicals. The errors can be reduced if the model is properly calibrated and verified by actual measured data. It should be noted that in modeling pollutant transport, hydrologic (runoff) components must be calibrated first, followed by erosion components, and finally pollutant generation-transport components.

In spite of the errors involved in modeling complex systems such as nonpoint pollution from urban areas, the model as a planning tool cannot be replaced by any "rule-of thumb" approach. The use of models is beneficial and greatly enhances the planning process for the following reasons (Novotny and Chesters (1981)):

1. Models can provide a forecast of the impact (although only approximately) of planned actions on water quality and pollution loadings.
2. Models can provide an understanding of the processes involved in pollution generation from nonpoint sources.
3. The data base necessary to construct and calibrate the model is useful for other planning activities. Many problems will be solved and questions answered or become clearer just by evaluating the data and compiling them into an appropriate input format.
4. Critical processes and areas of concern can be delinated and detected by modeling.
5. Models can generate numerous alternatives according to the specifications of the users. Various strategies can be investigated, and the impact of remedial measures can be evaluated.
6. Although the absolute accuracy of the outputs from the model are limited and sometimes even small, a comparison and ranking of outputs for various alternative remedial measures are reliable and in most cases more than adequate.
7. Models can estimate and analyze trade-offs between planning objectives. A system providing the lowest pollution may not be optimal for other objectives such as development. If the environmental objectives are known the alternatives to achieve them can be measured in terms of economic and welfare efficiency by considering the willingness of those involved and affected by the nonpoint pollution abatement measures to pay for the consequences and the benefits.

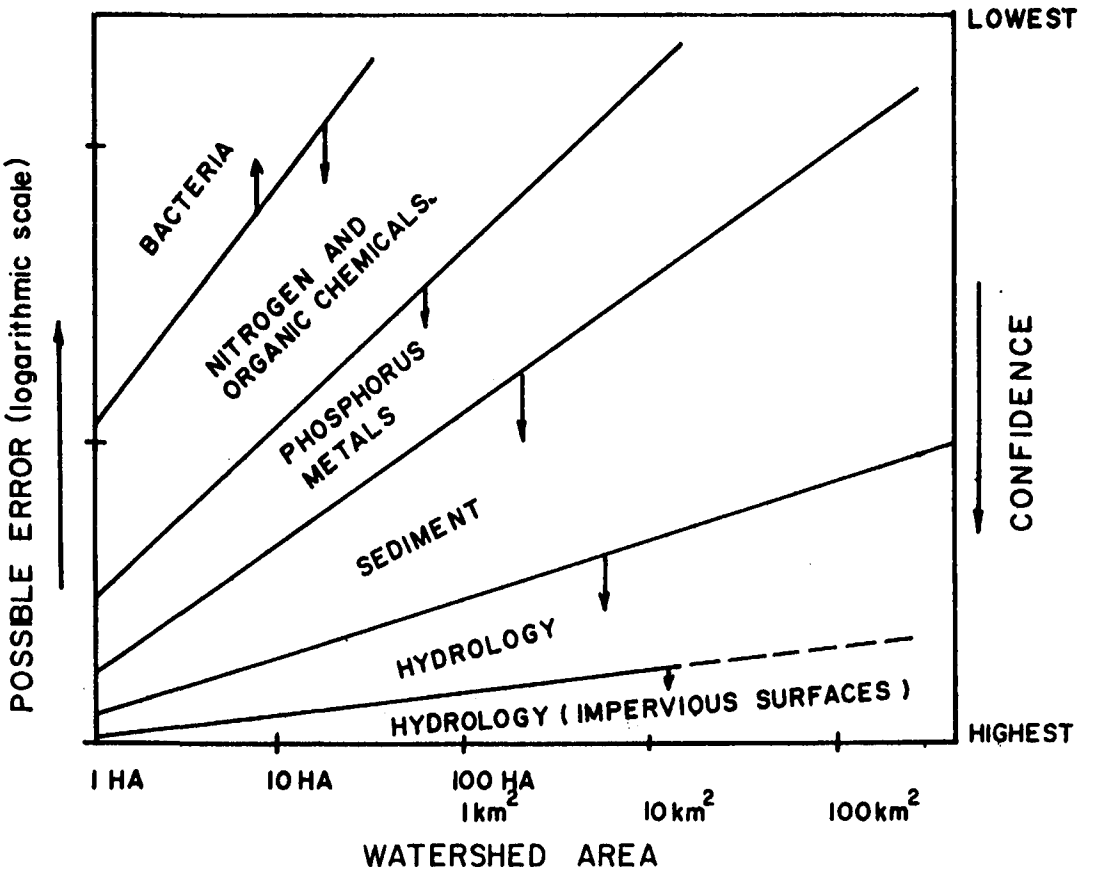


Figure 2 - Accuracy and Reliability of Urban Nonpoint Pollution Modeling

SELECTION OF THE MODEL

After the models and their capabilities have been introduced in the preceding sections, one question managers and urban planners may ask is which model to select and for what purpose. The ideas of an universal model that could do everything that was advocated in 1970's by some developers of the mathematical models, may not be as popular and feasible today. The progress and availability of small and relatively cheap micro- and minicomputers have revolutionized the model applications. A highly powerful (500K RAM memory) fast microcomputer can now be purchased for $1500 to $2500. These computers are comparable in their capacity and speed to medium size computers of the 1970's.

What this means is that a menu of smaller models may be more useful to a practicing urban engineer and manager than one large model. Thus, one should look at the model selection question from the point of view of what purpose and tasks the model should serve. The model itself should be looked upon as a tool to accomplish the task and objectives.

As stated in the preceding section, most of the hydrological models are limited in their areal extent. A typical size of the watershed best suited for detailed hydrologic modeling would be from a fraction to several km^2. For example the applications of ILLUDAS model should be limited to basin areas of less than 13 km^2. Thus, most of the applications have been limited to small experimental watersheds and techniques on how to expand the modeling to larger watersheds are still in evolution. Bearing in mind that commonly only a portion of a typical urban or urbanizing watershed is responsible for most of nonpoint pollution, overview or screening modeling techniques should be made compatible with the hydrological detailed models. In a typical watershed nonpoint pollution abatement process the modeling effort should proceed in the following phases:

(I) Overview screening models should identify the problem areas.
(II) Detailed continuous models (level III or V) could be used for obtaining more accurate nonpoint pollution loadings figures and for screening the various management practices and structural measures and their water quality impact. Such modeling activities must consider both the source strength and pollutant delivery.
(III) A detailed event oriented model (level IV) could be used to finalize the design of some technically complex structural and nonstructural measures selected and evaluated in Phase II. Such modeling may include design of storage for multiple overflows, various treatment schemes, and redesign of drainage systems.

A schematic diagram of the process is shown on Figure 3.

Calibration and verification with field measured data may be necessary for Phase II modeling if accurate loadings of pollutants are desired. Calibration is less important if the effect of various management practices on the loadings is studied.

This approach leads to a hierarchical modeling process whereby the knowledge and extensive testing associated with some large and complex urban runoff-pollution models can be extrapolated to the practical load management models and approaches used for selecting of hazardous critical areas within the

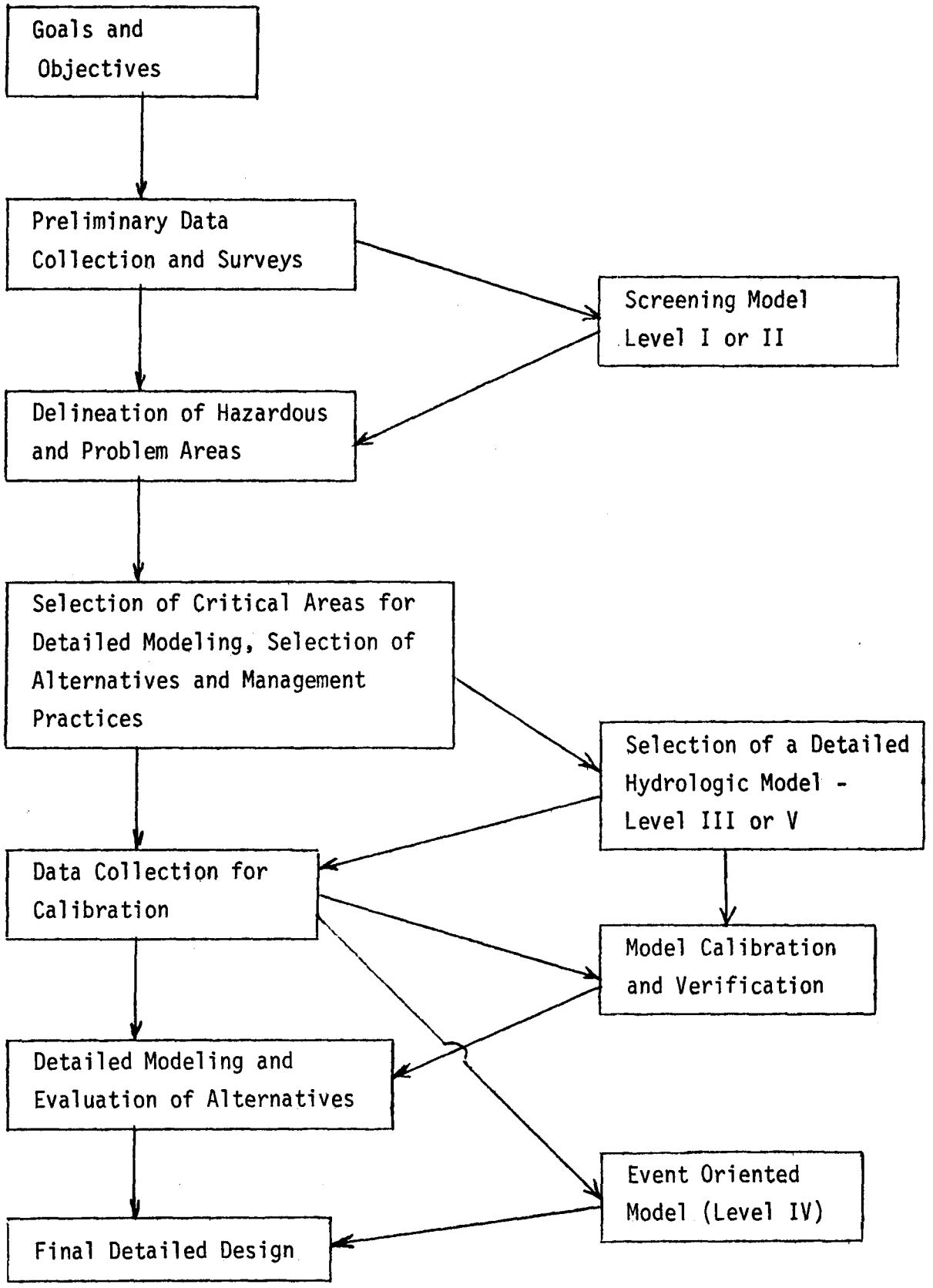


Figure 3

Schematic of the Modeling Process

watershed and for selection of appropriate management practices. A detailed hydrologic model calibrated and verified by small plot field studies can be used to estimate unit loads for a screening overview model as shown by Novotny and Bannerman (1980).

This concept has been used by the Wisconsin Department of Natural Resources to identify the problem areas within so called "priority watersheds" that is watersheds with a serious nonpoint pollution problem.

The Model Enhanced Unit Load concept (MEUL) enables selection of the priority watersheds and the problem areas within them. The units loadings were generated by a calibrated and verified hydrologic-nonpoint model. In this study loadings were generated for 1 km^2 (100 ha) areal units located on four hydrologically uniform soil types with a uniform slope of 6%. A time series summarizing several years of meteorological data was statistically analyzed to provide long term average loadings under different land use conditions. In a subsequent study that focused on urban areas only, a hydrological level III model, calibrated and verified by 3 year monitoring program encompasing 12 small urban watersheds, was used to extrapolate the findings into more general relationships (Novotny (1983, 1984), Novotny et al. (1985)). Figure 4 shows such a relationship whereby pollutant loadings from hypothetical unit urban areas (residential and commercial land use) are related to the percent imperviousness of the area soil type, and street sweeping practices.

USE OF THE MODELS FOR SELECTION OF CONTROL OF URBAN RUNOFF

There are a number of measures that can be used to control pollution by urban runoff. These methods will be presented in detail in a subsequent paper. From the modeling standpoint, urban runoff (quantity-quality) control measures can be categorized as follows:

(1) On-site source control measures that include control of atmospheric deposition, litter control programs, leaf and grass pick-up, street sweeping. These measures affect the input and accumulation of pollutants on urban impervious areas.
(2) Control of urban pervious areas, including mostly control of bare and unprotected soils within the watershed, especially those due to construction. This is modeled by adjustment of a few hydrological parameters (depression storage, runoff curve number, infiltration rate) and by the cover factor, C, in the Universal Soil Loss Equation.
(3) Hydrologic modification of urban watersheds. Many measures in this category of urban nonpoint pollution control are similar to those used for control of flooding. But not all flood control practices can control nonpoint pollution. Simple flow retardation, for example, in a dry detention pond or surface ponding on impervious surfaces in areas served by storm sewers may have no effect on reduction of pollution loads. Hydrologic modification for modeling purposes can be described as follows:
    (a) practices that increase permeability and enhance infiltration, such as the use of pervious pavements or vegetation infiltration strips;
    (b) practices that will increase hydrologic storage, such as rooftop storage on flat roofs, temporary surface ponding, and restriction of stormwater inlets;

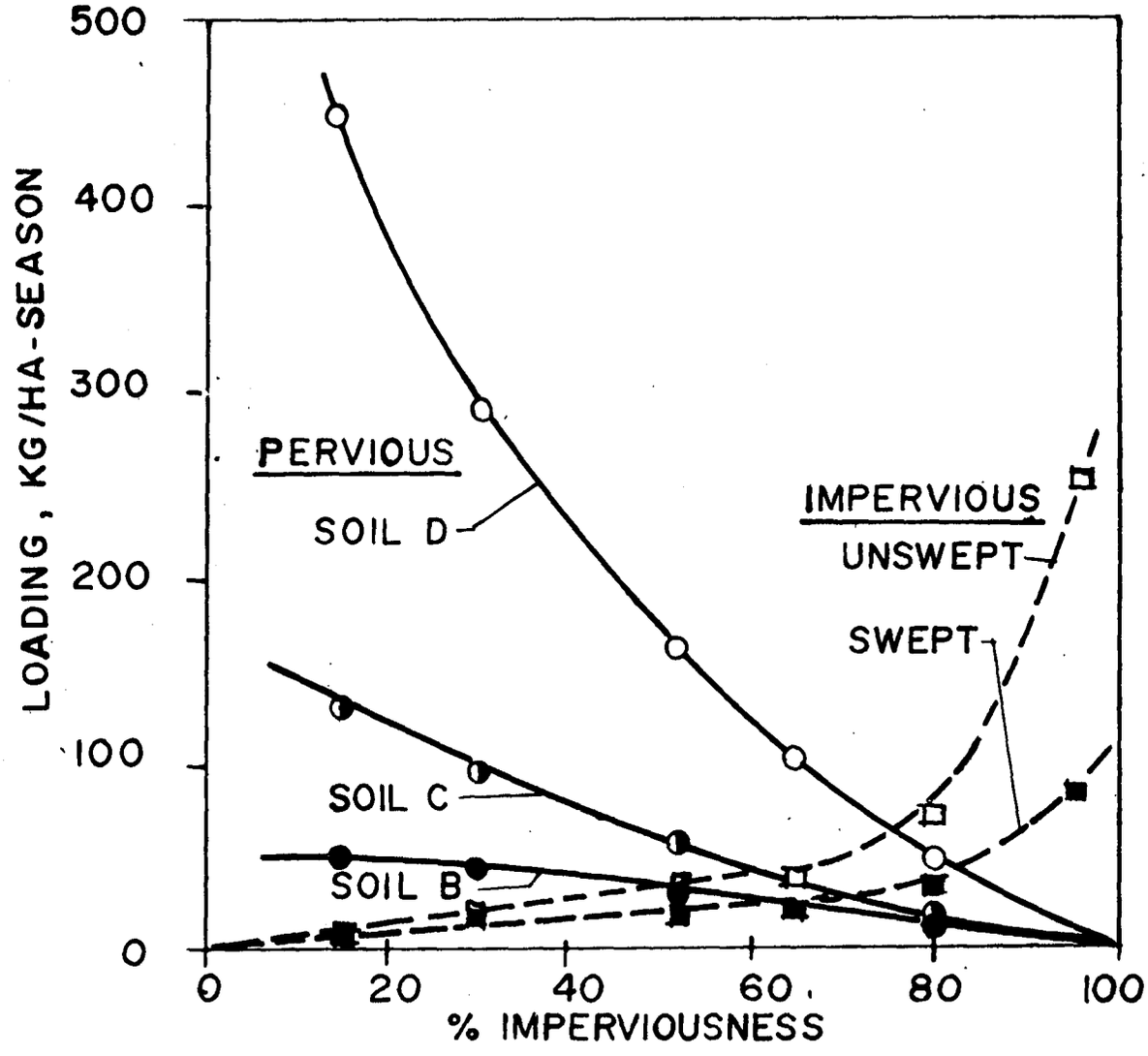


Figure 4 - A schematic representation of sources of urban particulate nonpoint pollution related to imperviousness, soil type and street sweeping. The loadings are expressed for 1 km^2 unit watersheds with 6% slope.

(c) practices that will reduce directly connected impervious areas, that include disconnnecting roof drains, letting surface runoff overflow on adjacent pervious areas, use of dry wells, and use of pervious pavements.

(4) Reduction of delivery of pollutants. After the particulate pollutants leave the source areas, they can settle or be removed on route from the sources to the receiving water body. A delivery ratio factor expresses, in the models, the ratio of the amount of pollutants delivered to a receiving water body (pollutant yield) to the amount of pollutant generated at the source.

Urban areas with a good storm drainage have a delivery ratio close to one, meaning that almost all pollution, after it leaves the source, is eventually delivered to the receiving water body.

Residential areas with a natural (swale) drainage have delivery ratios ranging from a few percent to about 50 percent. As a consequence, urban areas with a natural drainage produce much less pollution than areas with a good storm sewer system.

(5) Runoff detention-retention. The terms "detention" and "retention" have been used synonymously in most literature interpretations and in common usage. However, with recent advancements of the detention-retention technology, detention refers to a stormwater storage facility that is normally dry and is designed to temporarily hold stormwater during high-peak runoff events. Such basins have only a very limited efficiency in controlling urban runoff pollution. A retention facility always contains a substantial volume of water to serve other purposes such as recreation or aesthectics. The modeling of the effects of these facilities to control urban runoff is significant. The modeling of the effects of these facilities may vary in sophistication. The simple techniques will adjust the erosion control practice factor P of the Universal Soil Loss Equation using rule-of-thumb factor. Such factors have been developed by Driscoll (1983) and they can be used in combination with the Level II or III models.

A number of more complex models of retention facilities evolved in the last ten years. A summary of the models was presented by Prasad (1984). One of the most advanced retention facility models is SEDIMOT (Wilson et al. (1983)). Most of the Level IV and V models mentioned herein have a retention storage segment.

Example of the Model Application

The Level III Urban Runoff Model developed by the author (Novotny (1983) and Novotny et al. (1985)) was used to study the effect of several low cost management practices.

In the first phase of the research the model was calibrated and verified by monitoring data of eight small urban experimental watersheds gather by the Wisconsin Department of Natural Resources (Bannerman et al. (1984)).

For each watershed the monitoring program provided the following information:

(a) atmospheric deposition rates (wet and dry),
(b) pollutant accumulation in the curb storage,
(c) runoff quantity and quality,
(d) traffic count and land use characteristics.

The calibration process proceeded as shown on Figure 5:

(a) The generated street pollutant accumulation was matched against the monitored data (Fig. 6).
(b) When adequate fit of street accumulation data was obtained the calibration then proceded to matching runoff quantity for storm events with monitored data.
(c) Runoff quality calibration was the last step (Fig. 7).

Generally good to satisfactory fit of measured and computed data was obtained for most of the pertinent watersheds.

After the calibration precipitation data for 1981 and 1982 was used to generate seasonal and annual (excluding winter) loadings, the output from the model provided event runoff volume and quality for five constituents -- suspended solids, volatile suspended solids, nitrogen, phosphorus, and lead. These simulated loads and breakdown of sources of pollution are shown in Table 1.

This type of study provides the manager with information on magnitudes of nonpoint pollution and on the source strength. For example, in the Milwaukee study modeling indicated that atmospheric deposition accounted for less than 5% of the input, traffic became an important source of lead only in high traffic watershed, and effect of street sweeping was not significant.

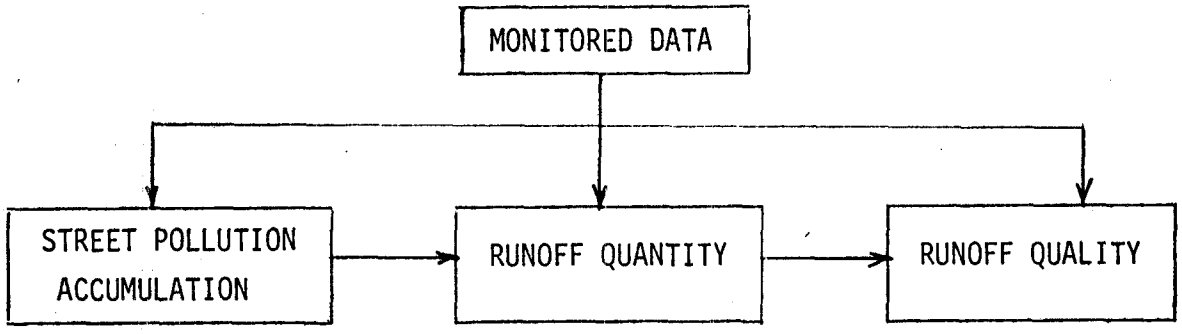
The effect of several management practices on water quality was then documented on a case of a "hypothetical" watershed that was set-up using the surface, soil, and land use data similar to those established for the study sites. The watershed consisted of the following land uses:

medium density residential zone	100	ha
low density residential zone	7.5	ha
open unused land (natural grass)	23.6	ha
commercial (retail)	12.7	ha
transportation (freeway)	14.5	ha
parks	19.50	ha

Figure 8 is a map of the hypothetical watershed. The soils were in silt and silty clay loam textural classes. Slopes ranged from 0 to 6 percent.

The following alternatives were studied:

Alternative 1 - No action.
Alternative 2 - Effective street sweeping.
Alternative 3 - Street sweeping and grass strips along the roads.
Alternative 4 - Street sweeping and two retention ponds providing 50% reduction of pollutants.
Alternative 5 - Street sweeping, two retention basins and grass strips along the roads.



Litter Deposition Rate

Litter Particulate Distribution

Breakdown Factor

Pervious Area - CN

Impervious Area - DS

Pollutants Conc. in Litter

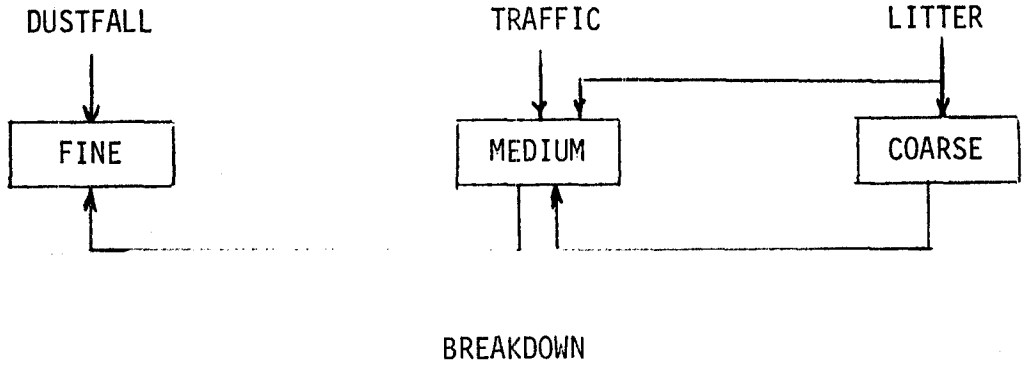


Figure 5 - Calibration Process Schematics

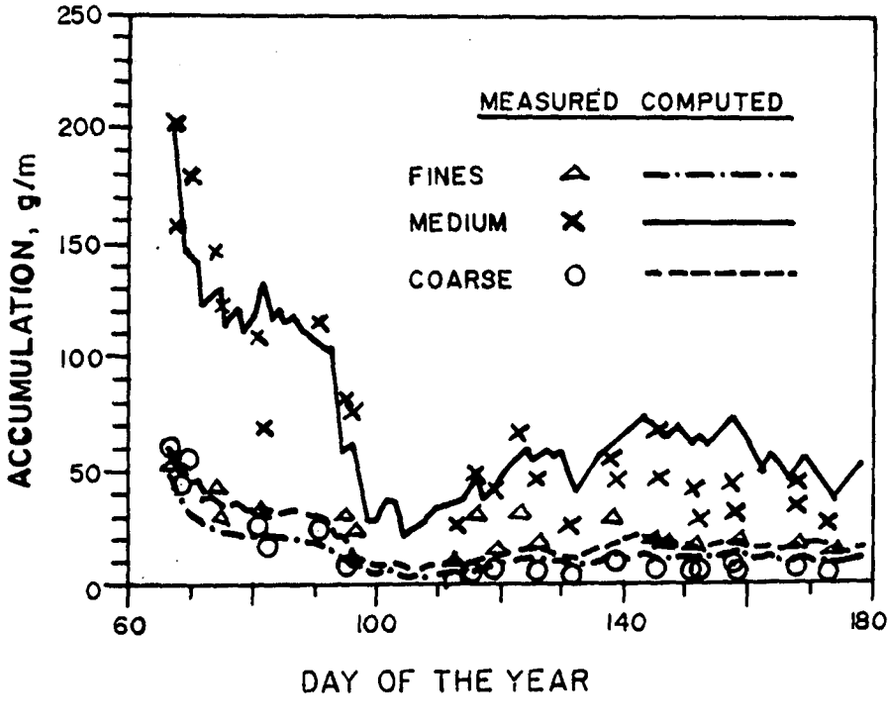


Figure 6 - Calibration Results of Street Solids Accumulation Process

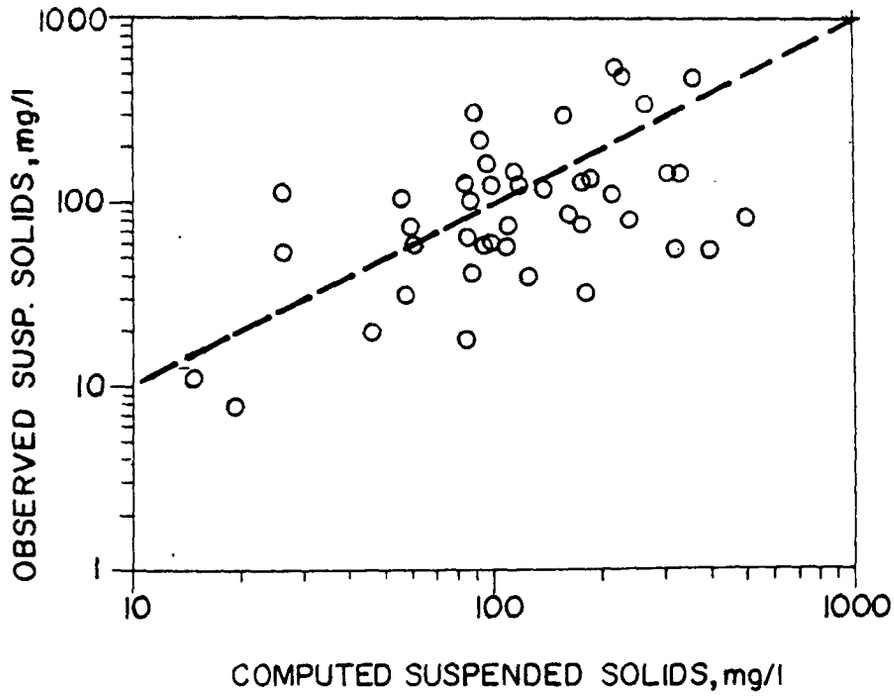


Figure 7 - Calibration Results of Runoff Quality

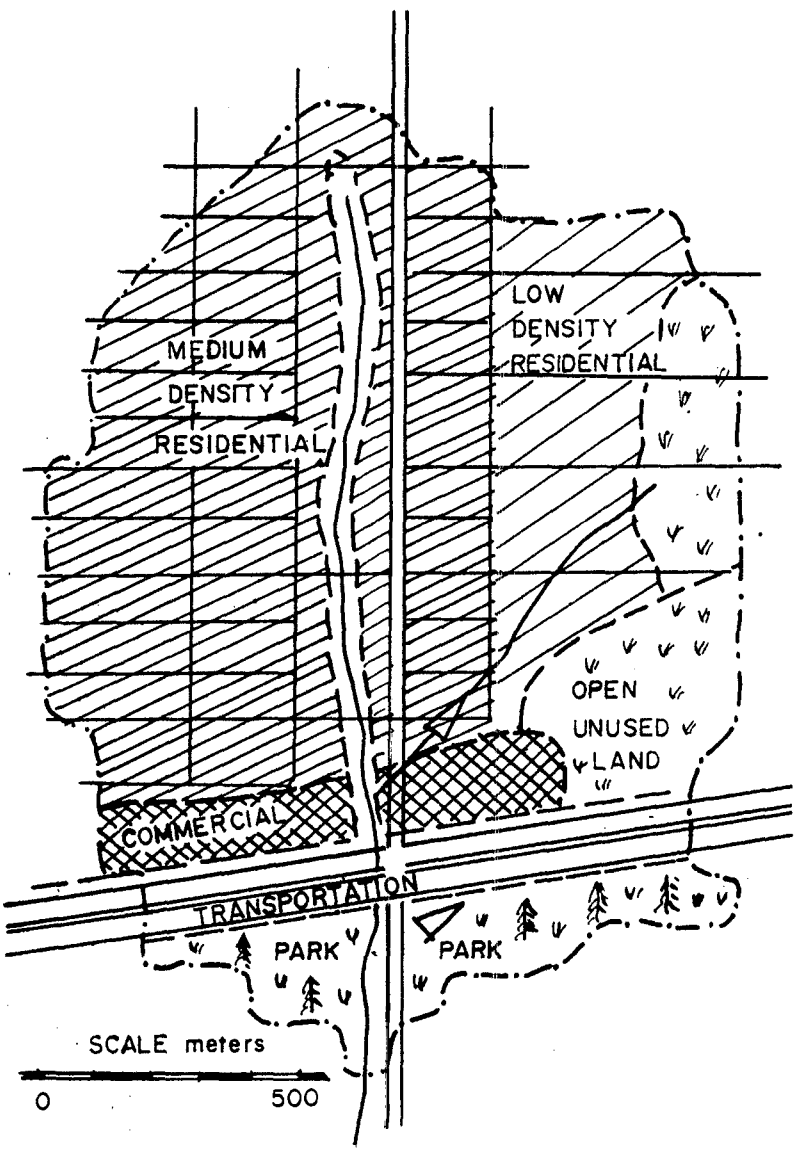


Figure 8 - The Experimental Watershed

Table 1

Computed Sources and Outputs of Pollutants
From an Urban Watershed

Overall Mass Balance

	Susp. Solids (ton)	Organics (kg)	Nitrogen (kg)	Phosphorus (kg)	Lead (kg)
Street Loads					
Initial	9.225	897.443	14.761	21.139	13.838
Final	6.641	529.347	8.715	12.645	24.615
Daily Inputs					
Dry Atm.	8.052	797.155	12.883	8.052	8.052
Litter	93.614	24527.625	149.813	257.288	140.444
Street Dirt	271.149	59919.688	362.217	631.968	960.308
Total	372.814	85244.438	524.913	897.308	1108.804
Event Inputs					
Wet Atm.	1.496	748.006	299.202	3.740	3.740
Soil	14.577	831.012	346.410	20.118	5.831
Total	16.073	1579.018	645.613	23.858	9.571
Outputs					
Blown Out	328.965	73871.187	465.394	801.693	961.323
Swept Out	.000	.000	.000	.000	.000
In Runoff	62.507	13320.262	711.177	127.966	146.275
Total	391.472	87191.437	1176.571	929.659	1107.597

Figure 9 shows the unit loads of solids for the no action alternative (Alt. 1). Such figures can aid managers and designers in their decision making to identify the problem areas and sources of pollution. In a typical mixed land use watershed identification of hazardous land is necessary since not all lands and land use segments require management. For example, established low density residential areas with natural drainage have loadings comparable to undisturbed natural prairies and woodlands.

Figure 10 shows the cummulative probability plot of suspended solids concentrations. The results indicated that grass strips along the roads (2 meter grass strips between the impervious road surface and the drainage system - sewer or swale) provided the best water quality benefit both in reducing the concentrations by 80% and runoff volumes by 50% for medium and by 25% for longer storms, respectively.

Street sweeping was not an effective pollution control measure in this sensitivity analysis, although it did reduce the pollutant outputs from highly impervious segments.

The suggested optimal best management practices for this hypothetical case study would include street sweeping in the commercial zone, two meter grass strips along the roads and streets, and retention ponds to intercept flows from open lands and residential areas located on high slopes and poor (silty clay loam) soils.

In addition, grass clipping control, leaf pick-up during the fall season and litter control ordinances should be implemented.

Conclusions

The urban runoff quantity-quality models are indispensable tools in the evaluation of sources of pollution from urban and urbanizing areas and the design of the abatement measures. Such models can:

(1) Estimate loadings of pollutants from nonpoint sources under different hydrological and meteorological conditions.
(2) Estimate source strengths of different contributing areas within urban and urbanizing watersheds.
(3) Evaluate relative efficiency of various management practices and structural measures.
(4) Enable design of nonpoint control measures.
(5) Aid in watershed planning.

Both screening and hydrological models should be used, however, not on the same scale and with the same detail. Screening models are useful for identification of problem areas within larger watersheds. Hydrological models are then subsequently used for detailed modeling of nonpoint pollution loadings and evaluation of the impact of abatement.

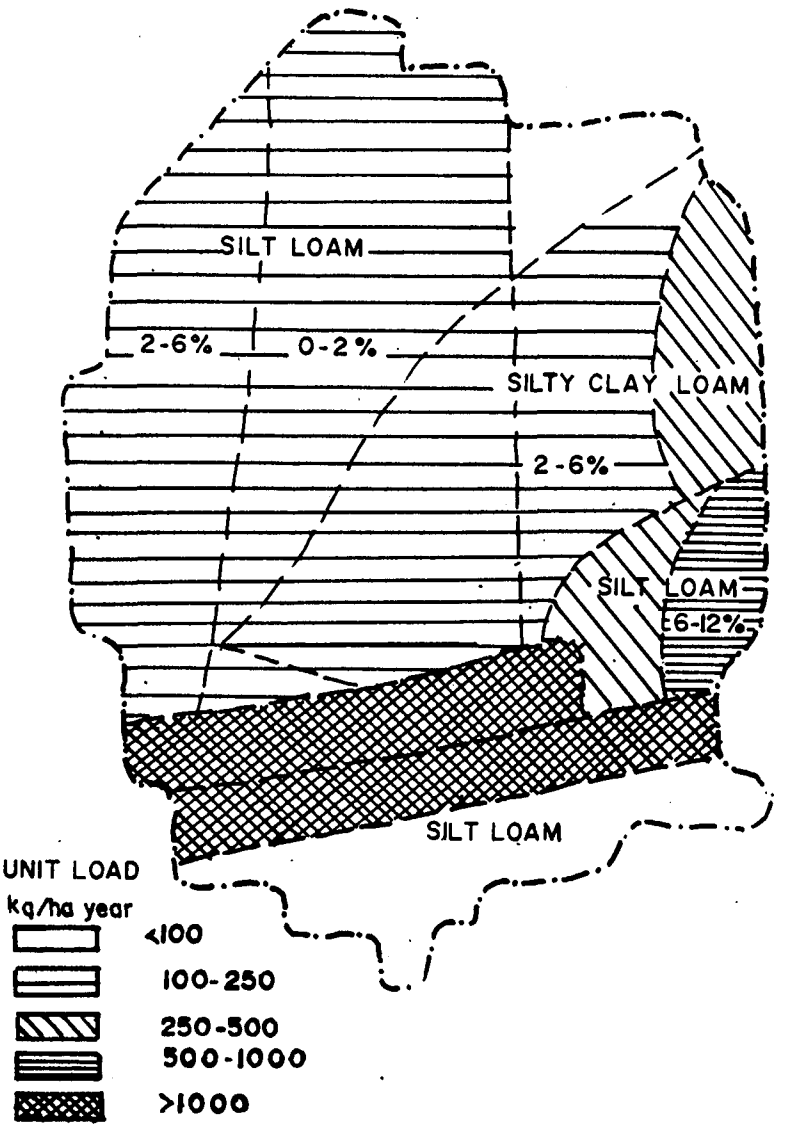


Figure 9 - Units Loadings of Solids from the Experimental Watershed for Alternative 1 (No Action)

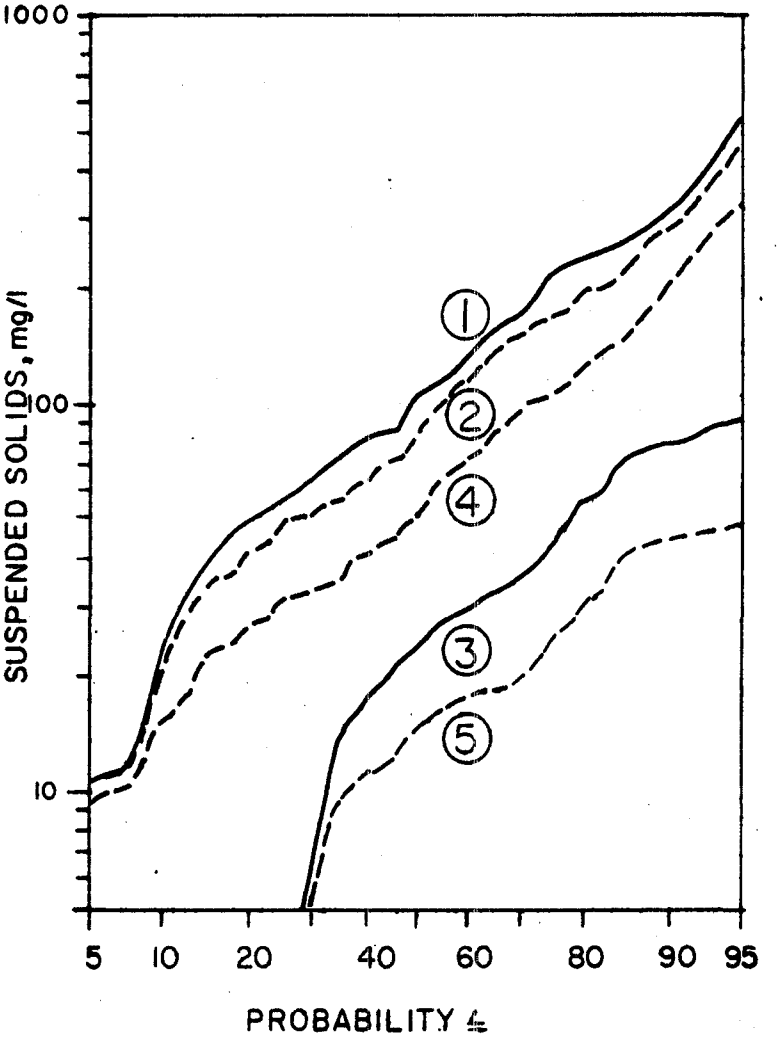


Figure 10 - Cumulative Probability Plot of Suspended Solids Concentrations in the Runoff for 5 Alternatives Measured by the Model

Alt. 1 - No action,
Alt. 2 - Street sweeping only,
Alt. 3 - Street sweeping in commericial zones plus grass strips along the roads in residential zones,
Alt. 4 - Street sweeping plus two retention ponds,
Alt. 5 - Same as Alt. 3 plus two retention ponds.

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MANAGEMENT AND CONTROL OF POLLUTION BY URBAN RUNOFF

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Abstract

The paper reviews progress on urban stormwater management and pollution control with emphasis on non- and low-structurally intensive techniques along with the total system approach encompassing control-treatment.

Many of the United States Environmental Protection Agency's demonstration-evaluation projects are presented to exemplify: Land Management Techniques, i.e., land use planning, best use of natural drainage, dual use of retention and drainage facilities required for flood control designed simultaneously or retrofitted for pollution control, porous pavement, surface sanitation, and chemical use control; Collection Systems Control, i.e., catchbasin cleaning, flow regulators (including swirl and helical blend devices), and the new concepts of elimination or reduction of unauthorized cross-connections in-channel/conduit storage and/or other forms of storage for bleed back to existing treatment plants; In-Receiving Water Storage; Treatment, i.e., physical/chemical, disinfection, and a treatment-control planning and design guidebook; Sludge and Solids Residue from Treatment; and Integrated Systems, i.e., storage/treatment dual-use wet-weather flow/dry-weather flow facilities, and reuse of stormwater for non-potable purposes.

Recommendations for the future in the areas of: control based on receiving water impacts, toxics characterization and their control, sewer system cross connections, integrated stormwater management, and institutional/sociological/economic conflicts are also presented.

Keywords: Storm sewers, Overflows--sewers, Drainage, Surface water runoff, Runoff, Hydrology, Combined sewers, Water pollution, Water quality, Wastewater, Sewage, Contaminants, Waste treatment, Storage tanks, Disinfection, Mathematical models, Remote control, Hydraulics

INTRODUCTION

The mission of the Storm and Combined Sewer Program (SCSP) of the U.S. Environmental Protection Agency (EPA) is to develop methods for controlling pollution from urban stormwater discharges and combined sewer overflows (CSO). This paper will emphasize research relevant to urban stormwater. This program has been involved for 20 years in the development of a diverse technology including CSO and stormwater control technology, instrumentation, problem assessments, best management practices (BMP) development and evaluation, stormwater management models, sludge handling and disposal methods, infiltration inflow (I/I) control, erosion control, and many others. Because the time allotted will not allow complete coverage, just the more important products will be highlighted with emphasis on some recent developments.

POLLUTION PROBLEM ASSESSMENT

Characterization

The quality and quantity characterization of urban stormwater discharges is necessary for problem assessment, planning, and design. Summaries of characterization from many research studies are available [1-3].

About 70 lb/ac/yr of biochemical oxygen demand (BOD) in urban stormwater runoff discharges contribute about 45% of the annual BOD load if secondary treatment is provided for the dry-weather flow (DWF).

Since 1974, EPA supported an urban rainfall-runoff-quality data base [4,5] for two important data requirements: characterization of urban storm runoff and calibration and verification of urban stormwater management models.

From an in-house project, preliminary screening of urban wet-weather discharges from 24 samples from nine urban areas found that approximately one-half of the 129 EPA's priority pollutants (Table 1). The heavy metals were consistently found in all samples. Polynuclear aromatic hydrocarbons from petroleum were the most frequently detected organics.

Table 1. Distribution of Priority Pollutants Frequently Detected

Pollutant	Frequency*		Range**	
	Liquid	Sediment	Liquid (µg/l)	Sediment (µg/kg)
Phenols	6	-	0.3-0.9	-
Polynuclear Aromatic Hydrocarbons (PAH's)				
Fluoranthene	5	3	0-98	1,200-4,000
Pyrene	4	3	0-76	1,200-3,000
Anthracene	-	3	-	375-7,000
1,2-Benzanthracene	-	2	-	1,100-4,100
Benzo(a)pyrene	-	2	-	600-25,000
3,4-Benzofluoranthene +	-	2	-	2,800-4,600
11,12-Benzofluoranthene	-	-	-	-
Chrysene	-	2	-	1,500-4,100
Fluorene	-	2	-	210-300
Phenanthrene	-	2	-	375-2,000
Esters				
Bis(2-ethylhexyl)phthalate	6	3	0-160	130-59,000
Di-n-butyl phthalate	6	3	0.5-3.4	300-25,000
Butylbenzyl phthalate	-	2	-	450-1,200
Heavy Metals				Sediment (mg/kg)
Arsenic	7	3	2-15	2.8-4.8
Beryllium	7	3	1-2	0.1-0.2
Cadmium	7	3	2-7	0.06-1
Chromium	7	3	9-36	8.10-28
Copper	7	3	11-49	7-140
Lead	7	3	30-400	24-370
Nickel	7	3	9-40	3-16
Silver	7	3	3-8	0.5-2.1
Zinc	7	3	90-330	37-380
Antimony	7	3	<20	3-3.3
Selenium	7	3	3-10	0.06-1.4
Thallium	7	3	2-10	<0.2
Mercury	7	3	0.2-0.56	2.1-2.4

*Reported when found four or more times out of seven liquid samples and two or more out of three sediment samples.

**These are mostly grab samples and therefore cannot be accurately related to pounds of pollutants discharged. Also, the mass emission of pollutants will vary for each rainfall event.

A project in Syracuse, NY using the Ames Test to evaluate urban runoff mutagenicity[6,7] found detectable responses for 22% of the samples. It is significant that some mutagenic substances are present with a potential for entering the food chain.

The discharge of pathogens from CSO and storm sewers is a major public health concern. Indicators such as fecal coliform have long been known to be present in stormwater discharges in densities sufficient to cause contravention of standards. A study in Baltimore, MD identified actual pathogens and enteroviruses in storm sewer discharges[8]. Cross-connections from sanitary sewers were strongly implicated as the major cause. This problem is not confined to Baltimore. The problem is likely to be widespread particularly where sewers are older and cross-connections are present.

Receiving Water Impacts

Knowledge of the receiving water impacts resulting from urban wet-weather discharges is a basis for determining the severity of problems and for justifying control. Problem studies of receiving water impacts are described in a proceedings from a national conference[9] and in a journal paper[10].

Under certain conditions, storm runoff can govern the quality of receiving waters regardless of the level of DWF treatment provided. Based on national annual mass balance determinations, urban wet-weather oxygen demand loads are greater than the dry-weather loads from the same areas[11,12]. Hence, control of runoff pollution can be a viable alternative for maintaining receiving water quality standards. However, documented case studies of urban runoff impacts are scarce.

The SCSP has had only partial success in finding urban storm flow generated receiving water impacts employing the conventional dissolved oxygen (DO) concentration criteria[10,13]. The problem appears to be in the application of conventional dry-weather monitoring techniques to the unsteady-state flow regimes caused by storms. Further studies should be undertaken to resolve the anomaly between actual runoff loadings and observed receiving water impacts. The SCSP has been more successful in sediment analysis than in water column analysis for DO depletions. Direct evidence has been obtained (from the Milwaukee River Project[14]) of how a disturbed benthos depletes DO from the overyling waters. Studies have also shown that storm and CSO

adversely affect sediment by toxics enrichment and resultant biological upsets[9,10,15-17]. Since particulate matter in untreated storm and CSO is larger, heavier, and in significant quantities when compared to treated sanitary effluent, more needs to be known about the fate and transport of settleable and separable materials. Hydrodynamic solids separation and sediment transport routines must be added to receiving water models to take care of the neglected or presently omitted significant particulate and bed flow fields.

A simplified continuous receiving water quality model has been developed for preliminary planning and screening of areawide wastewater treatment alternatives in terms of frequency of water quality violations[18,19]. Also, a general methodology has been developed for evaluating the impact of CSOs on receiving waters and for determining the abatement costs for achieving various water quality goals[20]. In addition, a methodology for defining criteria for wet-weather quality standards has been developed[21-23]. In recognition of an important gap in the developed methodologies, the duration of water quality standards vs. species survival was taken into consideration.

USER'S ASSISTANCE TOOLS

User's assistance tools include instrumentation, stormwater management models, manuals of practice (MOP), methodologies, compendiums, and state-of-the-art (SOTA) reports.

Instrumentation

Storm flow measurement is essential for process planning, design, control, evaluation, and enforcement. Sampling devices do not provide representative aliquots. Conventional flowmeters apply to steady-state flows and not to the highly varying storm flows.

Flowmeters have been developed to overcome these adverse storm conditions[24,25]. A prototype sampler for capturing representative solids in storm flow has also been developed and a design manual is available[26]. This gave manufacturers the incentive to perfect samplers by increasing intake velocities and in other ways. SOTAs are available for flow measurement and

sampling[27,28]. Because storm flow conditions are extremely adverse, the manuals and instruments developed are useful for monitoring all types of flow.

Simulation Models

The SCSP has fostered the development of models for assessment, planning, design, and control of urban stormwater pollution. The program's urban water management analysis approach involves four levels of evaluation ranging from simple to complex that can be worked together.

The various levels of the Storm Water Management Model (SWMM) are the most significant model products in terms of past SCSP resources, user acceptance, and overall popularity. SWMM is one of the most widely used urban models and its benefits for planning and design have been demonstrated. It has been employed by consulting engineers to design sewers and to analyze pollution control alternatives.

There are significant recent enhancements of SWMM. Probably the most significant is Version III[29,31] which includes: better documentation and improved stability of the dynamic flow routing model, EXTRAN (Extended Transport Block); documentation of continuous simulation and snowmelt capabilities; new flexible physically-based storage and treatment routines which can provide estimates of treatment (by settling) in storage basins; an addition of the physically-based Green Ampt infiltration model; and flexible water quality routines and other routines.

Documentation and user's manuals are also available for three continuous stormwater planning models. One is a detailed model, the Quantity-Quality Simulation model (QQS)[32,33]. The other two are macroscopic planning models, ABMAC[34,35] and EPAMAC[36,37].

Operational models which have been implemented in Detroit[38], Minneapolis[39], Seattle[40], and San Francisco produce control decisions during storm events.

SOLUTION METHODOLOGY

A major emphasis of the SCSP was solution methodology, implemented through development of SOTA reports, MOPs, and user manuals.

The SOTA texts, user's guide and the assessment of urban stormwater technology are recommended documents[41,42]. Separate engineering manuals are available for storm flowrate determination[43,44], porous pavement design[45-48], cost estimating[49-51], storm sewer design[52-54], planning and design guidance[55], and for conducting stormwater studies[56]. Seminar proceedings with themes of "modeling, design, operation, and costs" have been published.

The SOTA document on particle size and settling velocity[57] offers significant information for solids treatability and their settlement in receiving waters, important areas overlooked in planning and design. An excellent film is being distributed by the General Services Administration (GSA), National Audio Visual Center which covers the EPA CSO Research, Development, and Demonstration Program, and in particular full-scale control technologies[58].

A report entitled, "Urban Stormwater Management and Technology: Case Histories,"[59] presents 12 case histories which represent the most promising approaches to CSO and stormwater control. The case histories were developed by evaluating operational facilties that have significant information for future guidance.

Three illustrative methodologies for conducting stormwater facility planning have been published[60-62].

MANAGEMENT ALTERNATIVES

The next major SCSP area is management alternatives. First the choice of where to attack the problem; at the source by land management, in the collection system, or off-line by storage. Pollutants can be removed by treatment and by employing integrated systems combining control and treatment.

Land Management

Land management includes structural, semi-structural, and non-structural measures for reducing urban and construction site stormwater runoff and pollutants before they enter the downstream drainage system. Various concepts have been fostered by the SCSP including:

- Land use planning;
- Natural drainage which will reduce drainage costs and pollution, and enhance aesthetics, groundwater supplies, and flood protection; and
- Retention and drainage facilities, and other management techniques required for flood and erosion control which can be simultaneously designed for pollution control.

Retention on-site or upstream can provide for the multiple benefits of aesthetics, recreation, recharge, irrigation, or other uses.

Porous Pavement

Porous pavements provide storage, enhancing soil infiltration that can be used to reduce runoff and CSO. Porous asphalt-concrete pavements can be underlain by a gravel base course with whatever storage capacity is desired (Figure 1).

Results from a study in Rochester, New York, indicated that peak runoff rates were reduced as much as 83%[63] where porous pavement was used. The structural integrity of the porous pavement was not impaired by heavy load vehicles. Clogging did result from sediment from adjacent land areas during construction; however, it was relieved by cleaning (flushing). The construction cost of a porous pavement parking lot is about equal to that of a conventional lot with stormwater inlets and subsurface piping.

A project in Austin, Texas developed design criteria for porous pavements[46] and compared porous asphalt pavement to six other conventional and experimental pavements[47,48].

Surface Sanitation

Maintaining and cleaning urban areas can have a significant impact on the quantity of pollutants washed off by stormwater. Tests under real work conditions in San Jose, CA showed that street cleaning can remove up to 50% of the total solids and heavy metal yields in urban stormwater with once or twice a day cleaning[64]. Typical street cleaning programs of once or twice a month proved ineffective. Organics and nutrients could not be effectively controlled even with intensive cleaning.

In Bellevue, WA, street cleaning proved ineffective; however, a modified regenerative air Tymco™ street cleaner showed promise[65,66]. Street cleaning

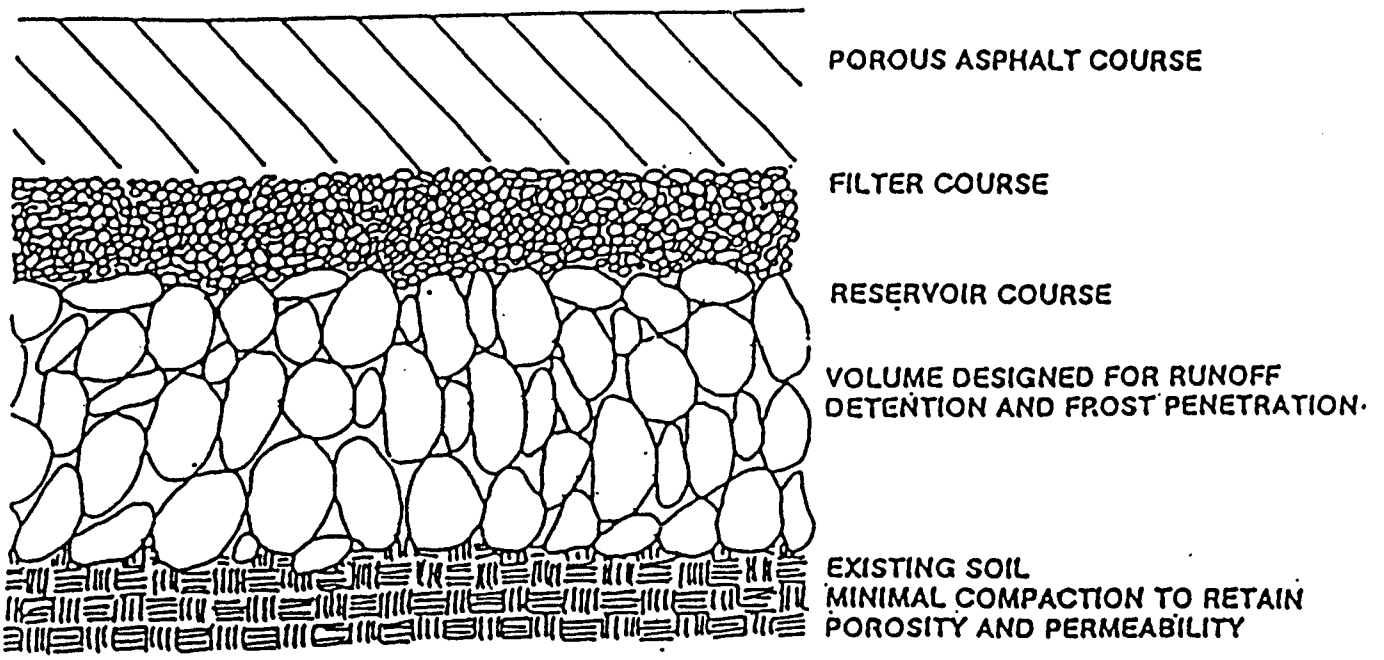

Figure 1. Porous Asphalt Paving Typical Section

is no panacea for stormwater pollution control (and is site specific dependent upon climatic conditions), but if integrated with other methods, could reduce city-wide costs.

Chemical Use Control

Until assessment in 1971[67], there had been limited research on highway deicing effects. A search to define alternatives[68] confirmed the need for an economic impact analysis and for identifying a hydrophobic substance to reduce ice adhesion. The economic analysis has been conducted[69] and hydrophobic substances have been identified and investigated[70,71]. Even though their costs appear greater than salt, when considering an estimated $3 billion annual damage to the environment, the costs are acceptable.

The 1973 assessment[72] identified the problems of sloppy salt storage practices and over application, resulting in manuals of practice[73,74] for improvement in these areas. These manuals were recognized as highly significant. The Federal Highway Administration reprinted them, and over 8,000 copies have been distributed.

SCSP work has prompted several states to enact legislation controlling salt application and storage. Also the SCSP's expert testimony helped Wisconsin to change its laws to contain controls for roadway deicing. And lastly, Program work has greatly assisted the Federal Highway Administration and the Army's Cold Climate Research Laboratory who are utilizing SCSP results and doing further work with hydrophobic substances.

Collection System Controls

The next overall SCSP category, collection system controls, pertains to management alternatives for stormwater interception and transport. These include: improved maintenance and design of catchbasins, sewers, in-pipe and in-channel storage, elimination of sanitary and industrial cross connections, and remote flow monitoring and control.

Catchbasins

In a project conducted in Boston, MA, catchbasins were shown to be potentially quite effective for solids reduction (60-97%)[75,76]. Removals of associated pollutants such as chemical oxygen demand (COD) and BOD, were also

significant (10-56% and 54-88%, respectively). To maintain the effectiveness of catchbasins for pollutant removal requires cleaning probably twice a year depending upon conditions.

Regulators and Hydrobrakes

At present, there is a strong need to develop and have a reserve of control hardware for urban runoff control and to effectively reduce the associated high cost implications for conventional storage tanks, etc. The dual functioning swirl flow regulator/solids concentrator has shown outstanding potential for simultaneous quality and quantity control[77]. A helical type regulator/separator has also been developed based on principles similar to the swirl. It is felt that the swirl/helical type regulators, previously applied only to CSO, can also be installed on separate storm drains before discharge and the resultant concentrate flow can be stored in relatively small tanks, since concentrate flow is only a few percent of the total flow. Stored concentrate can later be directed to the sanitary sewer for subsequent treatment during low-flow or dry-weather periods, or if capacity is available in the sanitary interceptor treatment system, the concentrate may be diverted to it without storage[78].

These methods of stormwater control may be more economical than building huge holding reservoirs for untreated runoff, and offer a feasible approach to the treatment of separately sewered urban stormwater.

A completed demonstration project in Boston, MA evaluated the swirl and helical bend for separate stormwater treatment. A complete swirl/helical design textbook has been published[79].

Hydrobrakes™ have been demonstrated by the SCSP in Rochester, NY[63] and Cleveland, OH[80,81]. Hydrobrakes have no moving parts, require no external energy source to operate and are not susceptible to blockages. They can be used as upstream off-line attenuators, and in-line flow back-up devices. Conceptually, they allow flow release without orifice movement or closure at a constant predetermined rate regardless of upstream head.

New Concepts for Stormwater Control

Research efforts have shown that sanitary and industrial contamination of separate storm sewers is a nationwide problem.

The SCSP has fostered a new concept in response to this problem which includes simple methods of checking for cross-connections. If found, they can be eliminated or the storm sewer can be dealt with as though it were a combined sewer whichever is more feasible. Other new concepts are using storm sewers and storm drainage systems for in-channel storage applying CSO in-sewer storage and routing technology and other storage facilities for ties into the existing sewage treatment system. Thus making better use of existing facilities for overall pollution control.

Storage

Because of the high volume and variability associated with stormwater, storage is considered a necessary control alternative. It is the SCSP's best documented abatement measure. Project results and theory indicate storage must be considered at all times in system planning, because it allows for maximum use of existing dry-weather and other treatment plant facilities and results in the lowest cost in terms of pollutant removal.

Storage facilities may have auxiliary functions, such as sedimentation treatment, flood protection, sewer relief, flow transmission, and dry-weather flow equalization.

Storage concepts investigated include the conventional concrete holding tanks and earthen basins, and the minimum land requirement concepts of: tunnels, underground and underwater containers, underground "silos," natural and mined under and above ground formations, and the use of abandoned facilities and existing sewer lines[82,83].

The in-receiving water flow balance method (Figure 2), is a recently developed storage alternative[84]. In-receiving water storage facilities contain stormwater between floating plastic curtains. After cessation of the overflow, pumps start automatically and the surrounding waterbody will enter the compartments and push the stormwater back towards the first compartment where it is pumped to the plant. Thus, the waterbody is used as a flow balance medium.

The storage method is low cost due to the employment of low cost materials (plastic and wood), the time required to install the unit (several days to months vs. months to a year for more structurally intensive

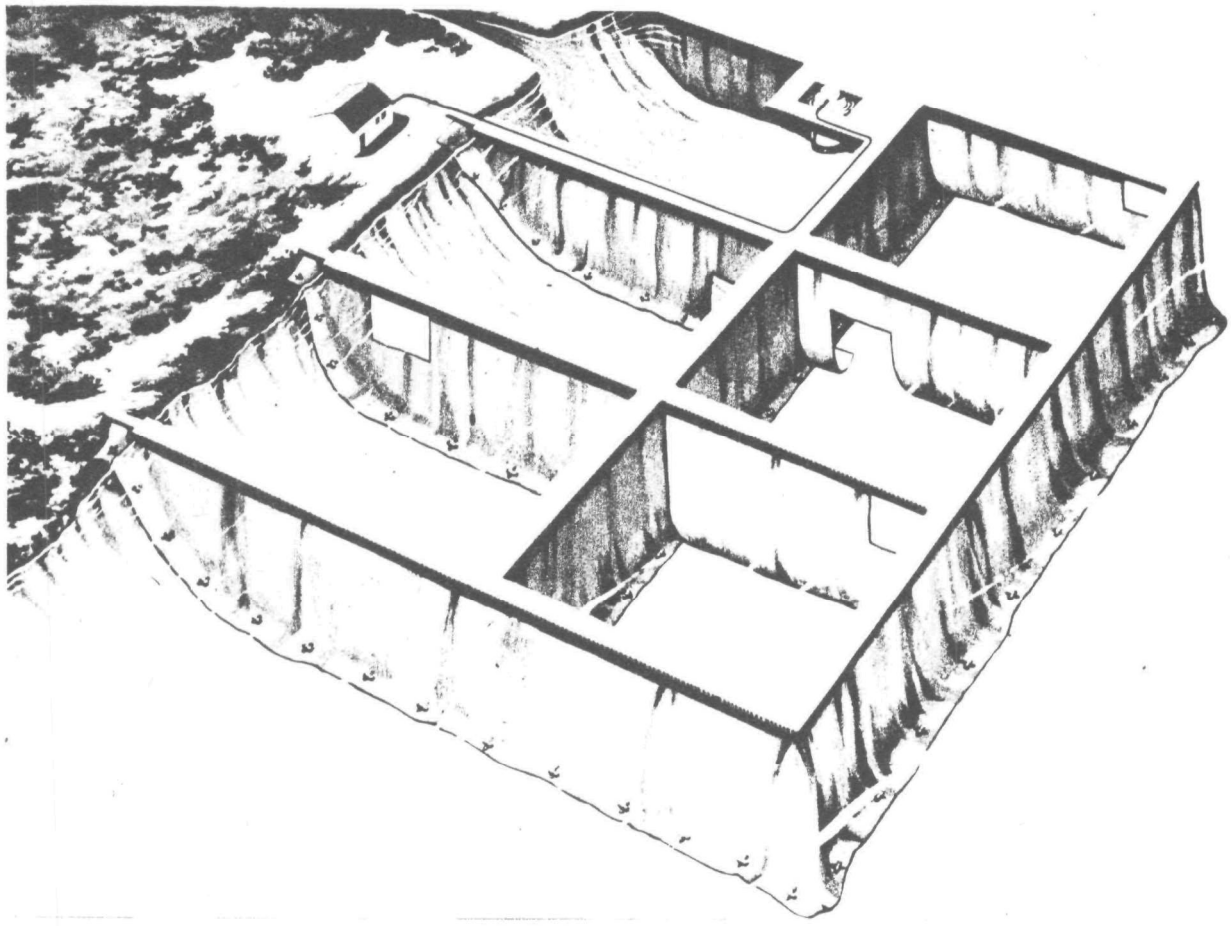

Figure 2. In-Receiving Water Flow -- Balance Method

alternatives), and the absence of land requirements. Studies show that costs could be about 5 to 15% of conventional concrete tank costs.

The facility which was tested at three locations in Sweden performed very satisfactorily, and was able to take ice and wind loads without adverse impact. It is desirable to demonstrate a facility in a harsh urban estuarine or marine site, such as the SCSP project will be doing shortly in Fresh Creek Basin in New York City, NY.

A storage/sedimentation planning/design manual[85] is being finalized which includes the following:

Major Issues

Applicability,
Integration into existing treatment systems,
Costs,
Effectiveness as treatment/control device.

Secondary Issues

Citing impacts,
O&M requirements,
Sludge production and handling.

Treatment

Due to adverse and intense flow conditions and unpredictable shock loading effects, it has been difficult to adapt existing treatment methods to storm-generated overflows, especially the microorganism dependent biological processes. Physical/chemical treatment techniques have shown more promise than biological processes in overcoming storm shock loading effects. To reduce capital investments, projects have been directed toward high-rate operations approaching maximum loading.

Storm-flow treatment methods demonstrated by the Program include physical, physical-chemical, wetlands, biological, and disinfection[86]. These processes, or combinations of these processes, can be adjuncts to the existing sanitary plant or serve as remote satellite facilities at the outfall.

Physical/Chemical Treatment

Physical processes/systems with or without chemicals, such as: fine mesh screening, swirl degritting, fine mesh screening/high-rate filtration (HRF),

sedimentation, and fine mesh screening/dissolved air flotation (DAF), have been successfully demonstrated. Physical processes have shown importance for storm-flow treatment because they are adaptable to automated operation, rapid startup and shutdown, high-rate operation, and resistance to shock loads. The high-rate processes/systems (DAF, fine mesh screening, and HRF) are felt to be ready for municipal installation.

The swirl has also been developed for grit removal. The small size, high efficiency and absence of moving parts offer economical and operational advantages over conventional degritting facilities. A full-scale demonstration of a (16 ft diameter/11 mgd design flowrate) swirl degritter has been completed in Tamworth, Australia[87,88]. Removal efficiencies confirmed laboratory results. Compared with a conventional grit chamber, construction costs are halved, and operation and maintenance costs are considerably lower.

Disinfection

Because disinfectant and contact demands are great for storm flows[89], research has centered on high-rate applications by static and mechanical mixing, higher disinfectant concentrations[90-94], and more rapid oxidants, i.e., chlorine dioxide[91-94], ozone[90], and ultraviolet (UV) light; and on-site generation[90,95,96]. Demonstrations in Rochester, NY[91]; Syracuse, NY[94]; East Chicago, IN[97]; and Philadelphia, PA[93], indicate that adequate reductions of fecal coliform can be obtained with contact times of 2 minutes or less by induced mixing and dosing with chlorine and/or chlorine dioxide. A pilot scale UV demonstration with a contact time of less than 10 seconds was conducted in New York City, NY.

The hypochlorite batching facility is still being used in New Orleans, LA to protect swimming beaches in Lake Ponchartrain[96]. The SCSP supported the development of a brine hypochlorite generator now being used in industry[95].

Treatment/Control Design Guidebook

A compilation of the SCSP's best research efforts in CSO treatment/control over the past 18 years has been published[86].

Treatment Process Performance

Treatment process costs and performance in terms of design influx rate (gpm/ft^2) and BOD_5 and SS removal efficiency is provided in Table 2. The

high-rate performance of the swirl, and the microstrainer, screening/HRF and screening/DAF systems, is apparent when compared to sedimentation.

Table 2. Wet Weather Treatment Plant Performance Data

Device	Control Alternatives	Design Loading Rate (gpm/ft^2)	Removal Efficiency (%) BOD_5	SS
Primary	Swirl concentrator	60	25-60	50
	Microstrainer	20	40-60	70
	High-rate filtration	24	60-80	90
	Dissolved air flotation	2.5	50-60	80
	Sedimentation	0.5	25-40	55
	Representative Performance		40	60
Secondary	Contact stabilization		75-88	90
	Physical-chemical		85-95	95
	Representative Performance		85	95

Sludge-Solids

Another SCSP area is the sludge and solids associated with storm flow treatment. Sludge handling and disposal must be considered an integral part of treatment because it significantly affects the efficiency and cost of the total waste treatment systems. A study quantified solids residue and sludge that would result from the treatment of separate urban stormwater on a nationwide basis[98,99].

INTEGRATED SYSTEMS

The most promising and common approach to urban storm flow management involves the integration of control and treatment. Integrated systems are divided into storage/treatment, dual-use wet-weather flow/dry-weather flow facilities, and control/treatment/reuse.

Storage/Treatment

When there is storage, there is treatment by settling, pump-back/bleed-back to municipal works, and sometimes disinfection. In any case, the break-

even economics of supplying storage must be evaluated when treatment is considered. The SCSP has demonstrated all of these storage/treatment concepts at full scale.

Dual-Use Wet Weather Flow/Dry Weather Flow Facilities

The concept of dual use is--maximum utilization of wet-weather facilities during non-storm periods and maximum utilization of dry-weather facilities during storm flows. The SCSP has demonstrated the full-scale dual-use of high-rate trickling filters[100], contact stabilization[101-103], HRF[104], and equalization basins[105]. Various municipalities are employing dual-use microscreening.

Control/Treatment/Reuse

"Control/Treatment/Reuse" is a "catch-all" for all integrated systems. A prime consideration should be the various nonstructural and land-management techniques. In Mt. Clemens, MI, a series of three "lakelets" have been incorporated into a CSO treatment/park development[106]. Treatment is being provided so that these lakes are aesthetically pleasing and allow for recreation and reuse for irrigation.

RECOMMENDATIONS FOR THE FUTURE

Receiving Water Impacts

Ties between receiving water quality and storm flow discharges must be clearly established and delineated. Quantification of the impairment of beneficial uses and water quality by such discharges is a major goal. Project results indicate the potential for significant impact of wet-weather flows on receiving waters. Control of runoff pollution can be a viable alternative for maintaining receiving water quality standards. However, the problems found seem to be site-specific in nature. Therefore, site-specific surveys are required that must consider the effects of larger materials and floatables near the outfalls. Based on results from these surveys, control may be warranted.

Toxics Characterization and Control/Treatment

Results from a limited in-house effort and the Nationwide Urban Runoff Program (NURP) indicate that urban stormwater runoff and CSO contains significant quantities of priority pollutants. Additional investigation of the significance of concentrations and quantities of toxic pollutants with regard to their health effects or potential health effects and ecosystem effects is required. A need exists to evaluate the removal capacity of alternative treatment technologies for these toxics and to compare their effectiveness with estimated removal needs to meet water quality goals.

Sewer System Cross Connections

Investigations have shown that sanitary and industrial contamination of separate storm sewers is an extensive nationwide problem. In practice, a significant number of separate stormwater drainage systems actually function as combined sewer systems. Therefore, a nationwide effort on both federal and local levels, to alleviate the pollution impacts from discharges of these systems is required. It is better to classify such cross-connected drainage systems as combined systems for pollution control priorities.

Integrated Stormwater Management

The most effective solution methodology for wet-weather pollution problems must consider: (1) control of wet-weather pollution impacts in lieu of continually and blindly upgrading existing municipal plants, (2) structural vs. non-structural techniques, (3) integrating dry- and wet-weather flow systems to make maximum use of the existing drainage during wet conditions and maximum use of wet-weather control/treatment facilities during dry-weather, and (4) the segment or blend on the percent pollutant control vs. cost curve in which incremental costs exceed incremental control efficiencies subject to load discharge or receiving water requirements.

Flood and erosion control technology must be integrated with pollution control, so that the retention and drainage facilities required for flood and erosion control can be simultaneously designed or retrofitted for pollution control. If land management and non-structural techniques are maximized and integrated costs for the extraction of pollutants from storm flows in the potentially more costly downstream plants will be reduced.

Institutional and Socio/Economic Conflicts

Some of the most promising opportunities for cost-effective environmental control are multipurpose in nature. However, there are institutional problems that hinder their implementation. First, the autonomous Federal and local agencies and professions involved in flood and erosion control, pollution control, and land management and environmental planning must be integrated at both the planning and operation levels. Multi-agency grant coverage must be adequate to stimulate such an approach. For example, the EPA would have to join with the Corps of Engineers, Soil Conservation Service, Department of Transportation, and perhaps other Federal agencies as well as departments of pollution control, sanitation, planning, and flood control at the local level.

Another problem is that construction grant (and other) incentives are geared towards structurally intensive projects which may counter research findings in the area of optimal solutions. Optimized wet-weather pollution involves a city-wide approach including the integration of structural as well as low-structural controls. The low-structural measures are more labor intensive. Construction grant funding does not presently address this expense and accordingly, municipalities are discouraged from using them.

CONCLUSIONS

In general, on a mass basis, toxic compounds, oxygen demanding, and suspended and visual matter in urban stormwater are significant. Ignoring the problem because it appears to be too costly or too difficult to solve, will not make the problem go away. The integrated approach to wet-weather pollution control is the only way which is going to be feasible, economical and, therefore, acceptable. Potentially tremendous savings can be derived from wet-weather pollution control research fostering integrated solutions. As you can see, the SCSP has investigated a problem, proven its significance, and developed a gamut of design and control techniques. Better advantage needs to be taken of proven technology.

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NONPOINT POLLUTION ABATEMENT PROGRAMS: THE MANAGER'S DILEMMA

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ABSTRACT

The passage of PL 92-500 in 1972 set governments at all levels--state, local and federal--on a collision course. The legislation changed national pollution abatement policy in that it directed the pollution control efforts be focused on treating pollution at its source rather than in receiving waters.

The challenge of dealing with non-point sources of pollution is that it interfaces land-use and water quality problems. In the country, historically, the federal government has dealt with water policy whereas responsibility for land-use decisions has been delegated to local units of government. As a result, there exists a tremendous institutional complexity which must be overcome if nonpoint problems are to be dealt with effectively. Implementation programs and mechanisms have evolved and nonpoint programs can, and should be put in place.

Keywords: Nonpoint source pollution, water policy, land-use policy, nonpoint source programs, PL-92-500, institutional arrangements, abatement program implementation

NonPoint Pollution Abatement Programs: the Manager's Dilemma

The passage of PL 92-500 in the early 1970's set governments at all levels--state, local and federal--on a collision course. This landmark legislation was the result of extensive deliberations by the Congress which focused its attention on the continuing deterioration of the nation's surface waters even after several years of fairly significant federal investment in the upgrading of municipal sewage treatment plants.

The legislation, of course, represented, in the final analysis, a significant change in policy for the nation in its struggle to control the continuing degradation of its waters. Pollution control efforts were now going to be focused on treating pollution at its source, rather than in receiving waters. Permits were to be issued to control any pollutants discharged from a point source and pollution arising from the wide array of man's land-use activities was also to be treated.

The term nonpoint source pollution was coined to cover basically all pollution arising from diffuse sources. It remains interesting that in the entire text of PL 92-500, the term nonpoint source pollution was never defined; it was basically taken to mean everything else.

Section 208 of that same legislation launched the nation on a massive planning effort designed to devise strategies for the control of both point and nonpoint sources of pollution. Unfortunately, the language in that section was also apparently vague and it took several years and many fits and starts before that program got underway.

The challenge of dealing with nonpoint sources of pollution is that it interfaces land use and water quality problems; the inevitable collision course that it puts units of government on is deeply rooted in the institutional and legal history of this country. Since the early days of the nation, the federal government has played a major role in water resources development as well as in water quality and policy. In 1824, the U.S. Supreme Court concluded that navigation and related water resource programs were within the purview of the Congress under its authority to regulate interstate commerce.

Over the years, the federal government has been directly involved in the construction of dams, canals, flood control projects, irrigation projects, and the like. A minimal responsibility for pollution control was placed in the U.S. Public Health Service in 1912 although the first abatement authority was probably given to the Corps of Engineers under authorities granted in the Refuse Act of 1899. Pollution abatement efforts, of course, have been strengthened enormously over the years with significant legislative overhauls in 1948, 1965, and 1972.

The states, basically, have exercised authority over water resources problems only in those areas clearly without federal jurisdiction. Since the 1950s, they have been involved in implementing federal water quality programs, all of which has been done with pretty strong federal policy guidelines. States do have the responsibility for the regulation of municipal and private use of water, water supply, groundwater, and industrial waste. It is noteworthy that these regulatory responsibilities

have been carried out primarily by the states themselves and that very little responsibility in these areas has been delegated to local units of government.

Policy governing land-use management has followed a very different course. The idea that a "man's home is his castle" is deeply rooted in our national tradition. Much of the impetus behind the westward movement of the pioneers in this country was the desire to own a piece of land to which an individual could do whatever he or she wanted, presumably without any interference from anybody. Although the states, primarily through their police powers, have the authority to regulate land use, this authority has traditionally been passed on to local units of government. States will often define areas of interest where land use activities affect more than local jurisdiction or will set limits to local land use activities but above and beyond this, the tendency has been to interfere as little as possible in land-use issues. The result is a very diffuse approach to land management questions which is in sharp contrast to the rather sharply focused, water-quality management programs.

The federal role in land-use plicy is very limited. They do, of course, administer public lands which does assume significance in some western states where those holdings make up a large percentage of the land area. Federal policies on soil and water conservation, however, are carried out at the local level by unique state-local institutions which were created at federal insistence, the Soil and Water Conservation Districts (SWCDs).

But PL 92-500 set the stage. It established the fact that nonpoint source pollution was a significant contributor to the degradation of surface water resources and that it was a problem that had to be dealt with. Dealing with it meant that land-use policies had to be reviewed, amended, updated, and in some instances, established in order to solve water quality problems.

PL 92-500 also stimulated a number of research projects to provide the technical backdrop for institutional, legal, and programmatic changes that the implementation of nonpoint source control programs would require. Many of the efforts examined and attempted to quantify, loadings from land-use activities and the impacts of those loadings on surface-water quality. Funds for some of this work came from Section 208 although perhaps the largest program was mounted by the Great Lakes National Program Office in Region V of the U.S. Environmental Protection Agency (EPA) with funds appropriated under Section 108 of PL 92-500. That program was linked closely to a massive effort by both U.S. and Canadian researchers under the auspices of the International Joint Commission.

Rural and urban nonpoint problems were identified and assessed. Best management practices were evaluated. Technical solutions to nonpoint problems were tested and a fairly good understanding of how to deal with the majority of the diffuse pollution problems emerged. Of all these projects, the Washington County Project in southeastern Wisconsin was unique because it was designed to examine the legal and institutional complexities associated with the implementation of nonpoint source control programs and even had, as one of its specific program objectives, the development of a sediment control ordinance. It was a bold and innovative effort funded by the Great Lakes National Program Office.

The project involved 11 cooperating agencies representing state, local, federal and regional units of government with the main actors being the Water Resources Center of the University of Wisconsin-Madison, the State Board of Soil and Water Conservation Districts, the Wisconsin Department of Natural Resources (DNR), and the Washington County Soil and Water Conservation District Supervisors. Project staff included a smattering of soil scientists, agricultural engineers, and water chemists but was dominated by lawyers, planners, sociologists, and economists. The challenge was fairly straight-forward, namely, to work through the legal-institutional maze which had evolved as a result of the differing approaches taken to land and water management in this country and to determine if the very heterogeneous mix of institutions could be woven into some kind of pattern that could deal effectively with the control of nonpoint source pollution problems.

At the inception of the Washingotn County Project, EPA's interest was twofold, i.e., they were committed to focusing responsibilities for implementation on a local unit of government, preferably the Soil and Water Conservation District, and they were committed to regulating sediment loss in rural areas.

In Wisconsin at that time (the mid-1970s) the set-up of Soil and Water Conservation Districts was somewhat unique in that the districts were coterminous with county boundaries and the SWCD supervisors were, by law, those members of the county board who served on the Agriculture and Extension Education Committee. A project sponsored study of the County Board in Washington County concluded that change--in this instance, perhaps, even radical change, namely sediment control--could probably best be accomplished at this government. The reasoning behind this was basically that county board supervisors were somewhat isolated from their constituents who generally have very little notion of what county government was all about. As a result of this, county supervisors tended to be re-elected time after time without much voter concern about what they were doing. The thinking was, then, that perhaps major change could be accomplished at this level of government if it were done carefully.

County government in Wisconsin is basically an arm of state government and really has only those authorities that the state grants it. A legal review suggested that much could be accomplished within the existing statutory framework but that the role of the SWCD could be strengthened by making it a standing committee of the County Board and by integrating its function with other county land use functions. This led to an extensive revision of state statutes which abolished Soil and Water Conservation Districts in Wisconsin and replaced them with a committee of the County Board called the Land Conservation Committee. Authorities of the old SWCDs were retained; new responsibilities for planning for erosion control and animal waste management have been added. Major funding for implementing these programs may be forthcoming from the current session of the Wisconsin legislature.

A sediment control ordinance was developed which set a performance standard for soil losses from an individual farm based on the Universal Soil Loss Equation. A farmer not meeting the standard could adjust his management practices in anyway he wished as long as the predicted soil loss rate met requirements established by the county. The Washington county SWCD

supervisors were prepared to take the proposal to a referendum vote in a part of the county when a certain amount of waffling on the question of sediment regulation occurred both in the state legislature in Madison and at EPA headquarters in Washington. That was the end of that proposal; the substance of the idea was adopted finally by the SWCD as a policy or position paper.

Efforts to regulate sediment losses from subdivisions in urban and urbanized areas proved to be more successful. In a simple sense, urban residents are much more accustomed to regulation than their rural counterparts and are required to obtain permits and to meet all kinds of requirements for almost everything they do. One more requirement one way or another didn't seem to make much difference. The major problem encountered involved convincing local officials that, in fact, a problem existed.

During this period, the Wisconsin Department of Natural Resources launched its Nonpoint Source Pollution Abatement Program. Five watersheds were selected for the program initially involving multiple governmental jurisdictions. Local government agencies with implementation authority--called Designated Management Agencies or DMAs--were identified to run the programs. One watershed, the Root River in southeastern Wisconsin, had 23 DMAs. The program established the pattern for nonpoint source implementation, namely, that local units of government with appropriate authorities would implement programs guided by firm state policy directives. The institutional complexities, however, remain.

This past year the EPA convened a Nonpoint Source Task Force to develop a National Nonpoint Source Policy to accelerate the implementation of nonpoint source control programs. The proposal policy urges continued federal leadership and coordination but goes on to emphasize that the states will have the lead in developing and implementing nonpoint source management strategies. Local governments are to be charged with developing nonpoint source strategies in coordination with the states and ultimately, with implementing those strategies.

All the pieces would seem to be in place, yet the question still remains: Can nonpoint control programs be implemented? The linkages between the water and land agencies have been forged although a great deal of institutional complexity remains. Federal agencies, particularly those in the U.S. Department of Agriculture, are falling prey to the budget cutter's ax.

This year is the 50th Anniversary of the establishment of the U.S. Soil Conservation Service. The agency was formed in response to the Dust Bowl crisis of the 1930s and under the leadership of the inspirational Hugh Hammond Bennett represented a bold and innovative effort to solve nagging soil erosion problems.

Now, 50 years later, we are again faced with many of the same problems and perhaps now it is time once again to look for innovative answers to the problems plaguing nonpoint control programs.

Authority for passing money through to states for technical and cost-sharing assistance has existed for some years now. Maybe the time is right for trying that approach.

Participation of landowners in rural nonpoint programs today is a problem much as it was with PL-566 Watershed Projects 20 and 30 years ago. Is regulation the answer? The jury seems to still be out on that question. More and more states are looking to regulation and the idea of establishing some realistic goal, like T by 2000, seems to have become very appealing. The established goal is to be reached in steps and most proposals call for a regulatory mechanism to kick in if interim goals are not met.

It goes without saying that as point sources of pollution are increasingly controlled, nonpoint sources assume even greater significance in the overall effort to control the continuing degradation of the nation's water. Many problems have been solved, some still remain. The where-with-all to mount a major program to control nonpoint source has been developed. The time has come to put that knowledge into action.

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INSTITUTIONAL ARRANGEMENTS FOR NONPOINT POLLUTION CONTROL PROGRAMS

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ABSTRACT

Institutional arrangements for nonpoint pollution control must be structured to provide maximum local acceptance, implementation, and enforcement. The more remote the institutional arrangements are from the local polluter and his understanding and trust, the more suspect and difficult implementation becomes. The ultimate in institutional arrangements is about 180 degrees from point-source pollution abatement arrangements.

Strong educational and technical assistance to the polluter, coupled with realistic subsidies for pollution abatement practices that are not cost-effective for the polluter, are the key to acceptance and adoption. Local institutional arrangements must prevail in these efforts. Enforcement must be established with a low profile and used as a last resort.

State and federal institutional involvements must carry a low profile as well, with realistic assistance to the local institutions. State and federal dictates will only slow implementation. Federal and state subsidies and implementation funds can be administered most efficiently at the local level and should be provided through grants.

The standard procedures for monitoring point-source discharges and establishing tolerances becomes nearly impossible and indefensible for nonpoint pollution, particularly from an economic point of view. This dilemma merits the alternative solution of preventing nonpoint pollution as the most acceptable means of control.

Local institutional arrangements must and will vary from state to state and community to community according to the forms of institutional authorities and responsibilities and the severity of nonpoint pollution problems. State and federal efforts to stereotype local institutions and programs will usually hinder local initiatives and implementation.

Coordination of local, state, and federal institutions for maximum effect and efficiency is necessary for success and for gaining of the public's confidence that the nonpoint pollution problem is being addressed in a realistic and productive manner. Rearranging authorities, responsibilities, and support programs of existing institutions can and must prevent the formation of additional institutions at any level of government.

Institutional arrangements adopted within a state must provide the capability for implementation if responsibilities are not carried out by the primary institution of authority, thus assuring the public of positive action.

INSTITUTIONAL ARRANGEMENTS FOR NONPOINT POLLUTION CONTROL PROGRAMS

Distinct differences exist between point-source and nonpoint-source pollution. These differences must be taken into account in developing acceptable institutional arrangements for the control of nonpoint-source pollution. Following are the major differences:

* Nonpoint-source pollutants are primarily diffuse in nature. Most are associated closely with surface runoff, usually following land-disturbing activities, in both rural and urban settings. In contrast, point-source pollutants are primarily associated with industrial and municipal discharges that emanate at a singular point.

* Point-source pollutants frequently are by-products of commercial activity with little or no use or value. Nonpoint-source pollutants, on the other hand, often are of economic value to the generating unit and, therefore, represent an economic loss. Pollution abatement efforts must consider the fact that the motivation to pollute or not to pollute involves these forces of opposing value.

* Nonpoint pollutants from agricultural operations mainly include sediment, fertilizers, pesticides, and animal wastes. All represent purchased or natural elements required for the production of food and fiber. The positive value of these elements seemingly would be sufficient motivation for agriculture to attempt to retain such pollutants on the land. Unfortunately, it is necessary to expose these potential pollutants, which are necessary for agricultural production, to rainfall and wind during land-disturbing activities.

* An individual's or firm's ability to recover point-source pollution abatement costs is possible--the individual or firm generally controls a product's supply and market price. Nonpoint polluters, particularly agricultural operations, have little, if any, ability to recover pollution abatement costs through supply control or price setting. Therefore, effective pollution abatement costs cannot be recovered in normal production and marketing activities.

* Monitoring and permitting point-source polluters is quite simple and precise. These activities usually generate the factual data, records, and analysis that are crucial to enforcement and prosecution if necessary. Because of their diffuse nature, the identification of sources of nonpoint pollution and the volumes of pollutants they contribute makes meaningful monitoring, control, and enforcement nearly impossible.

* Issuing permits to point-source polluters, a relatively easy process, usually is accomplished by establishing an acceptable level of pollutants to be discharged or issuing an order to the polluter requiring compliance with an established standard. The technical and financial burden for compliance becomes a production cost to the point-source polluter; this cost is programmed into the cost of producing a product and ultimately is passed along to consumers. The same procedure could be used to control and finance the elimination of municipal sewage pollution. However, 75 percent

of the technical and financial burden for constructing such facilities is assumed by the federal government in the name of public benefits, which alleviates a large portion of the municipality's direct cost. Increased federal taxation is necessary to pay for this federal subsidy. The higher tax becomes an added cost to the nonpolluter having no direct obligation.

Permitting nonpoint sources of pollutants has little, if any, support from the public or governmental institutions because of the impossibility and prohibitive costs of monitoring the discharges. Fluctuations in discharges, both volume and concentration, by virtue of variations in both rainfall intensity and soil characteristics create a nearly impossible condition for reliable and defensible monitoring. Enforcement under a permit system would be questionable at best.

A strategy for nonpoint pollution control

The differences between point-source and nonpoint-source pollution are the basis for development and execution of two drastically different abatement strategies.

Success in reducing point-source pollution through the sequence of permitting, monitoring, prosecution, and penalties has been effective because of governments' extensive authorities.

Success in reducing nonpoint pollution cannot make use of this same sequence of tactical actions by governments. Success can only be attained through education, demonstration, technical assistance, subsidization, and, as a last resort, enforcement by an institution capable of assessing a penalty.

Technological deficiencies are substantial in nonpoint pollution abatement. Identification of a pollutant's source, its concentration or volume, best control techniques, economic impacts, and the practicality of solutions are all matters that agricultural research has dealt with in only limited fashion. Agriculture instead has pursued profit-motivated research and development strategies generally acceptable within our free enterprise system. Little effort has been devoted to determining the environmental effects of new products or technologies.

Extensive investigation gradually produced a prevention strategy as the most acceptable and defensible approach to nonpoint-source pollution control. As a strategy, prevention represents a system of management practices designed to reduce, alter, or eliminate landowner and land operator actions that generate nonpoint pollutants. Other than sediment, the agricultural pollutants of primary concern are phosphates, nitrates, pesticides, and animal wastes. Phosphates and pesticides affix to soil particles upon application to the land. Nitrates pollute water by becoming an ingredient of surface water runoff or by percolating down through the soil into underground water supplies; the degree of pollution depends upon rainfall intensities, nitrate application rates, and timing of application. Animal wastes pollute water if applied to the land in excessive quantities or through the uncontrolled concentration or confinement of animal units without adequate animal waste management facilities. The focus of the prevention strategy has been to keep agricultural pollutants from entering streams and underground water supplies

through the reduction or elimination of soil movement and storm water runoff from the land into streams and lakes.

This prevention strategy focuses directly on farmers and their actions to produce food. Owners and developers of residential, industrial, and transportation sites are examples of nonagricultural land-disturbing activities causing erosion and sediment pollution of our waters. Nonpoint pollution by sediment itself contributes little chemical pollution, although it does produce extensive turbidity and damage to the biological environment.

The identification of major nonpoint pollutants and why they enter water supplies puts the prevention strategy squarely in front of millions of landowners and land operators. Neither owners or operators have an economic motivation to pollute, nor do they understand to what extent they may or may not be polluting.

Nonpoint Abatement Goal--A First Priority

Identification of nonpoint pollutants by volume and source provides the foundation for developing institutional arrangements for their control. Acceptable and attainable nonpoint pollution control goals dictate what institutions should be involved and the extent of that involvement.

Institutional structure must also recognize three facts: that landowners and land operators are independent, private entrepreneurs by choice; that most dislike regulation; and that resistance to regulation magnifies as higher levels of government become involved. Many landowners still harbor the position that a fee simple title grants complete management authority, regardless of the impact on the public's interest. With few exceptions, however, most will accept change if they understand the need for pollution control and if control efforts are both justified and fair.

For 10 years I was involved in the development and early implementation of Ohio's nonpoint pollution abatement program. The merits of that state's strategy deserve attention. Other states have structured similar strategies with minor differences due to varying forms of local and state governments.

Four major determinations influenced Ohio's final institutional arrangement:

1. All means were used to avoid the creation of a new institutional structure to address nonpoint pollution control.

2. It was determined which existing institutions might logically and willingly expand their responsibilities and authorities to implement a nonpoint pollution abatement program.

3. Consensus was secured among agricultural, governmental, and environmental interests regarding their desired institutional arrangement.

4. Legislation was written to authorize the selected institutions to produce the necessary standards, rules, and regulations and implementation procedures for an effective nonpoint pollution abatement program.

Among the important factors influencing the selection of a final institutional structure and implementation strategy were the following:

1. Existing federal and state agricultural agencies were unwilling to restructure their voluntary-service approach to include enforcement.

2. Controlling the largest volume of agricultural pollutants was attainable through application of soil erosion control and other land management practices recommended by existing local conservation institutions.

3. Good working relationships and a reasonably high level of trust and respect existed between local conservation institutions and farmers.

4. Farmers seemed relatively comfortable with the identification of pollution problems by local agencies and recommendations from those agencies for abating the problems.

5. Pollution abatement practices, enforcement procedures, penalties for noncompliance, and legal recourse for polluters were generally formulated and agreed to prior to legislative action granting state and local institutions the necessary authorities for implementation.

6. Institutional arrangements acceptable to the U.S. Environmental Protection Agency and, presumably, the public for point-source pollution control proved both inadequate and inappropriate for nonpoint-source pollution control programs.

Local Responsibilities

Conservation districts in Ohio accepted the authority to investigate and specify the necessary combination of land management practices required to reduce soil loss to the state-adopted limits. Districts provide technical assistance to landowners and land operators when necessary and recommend cost-sharing where applicable. Should enforcement procedures become necessary, a district's administrative board issues the first warning to a landowner or land operator. The board also establishes a time frame for compliance and offers the farmer any technical or financial assistance available. If the landowner or land operator complies within the designated time frame, the enforcement process ceases. If the landowner or land operator fails to comply, he or she is declared to be in violation of the rule, and the district board informs the individual that the designated state enforcement agency, the Division of Soil and Water Conservation within the Ohio Department of Natural Resources, is to be notified of the violation. The violator is also informed that he or she has a designated period of time within which he or she may appeal the local board's proposed action, if desired, prior to the board's forwarding of the citation to the state agency.

Public complaints about pollution problems can be registered with districts or the Division of Soil and Water Conservation. Complaints must be in writing, and they must be signed by the complainant. Complaints registered with the state agency are referred to the conservation district for investigation and recommendations. A district representative discusses the complaint with the accused violator. Technical and financial assistance are offered, if

appropriate, to facilitate installation of any needed management practices. This contact usually results in a cordial solution to the problem. Both the individual who filed the complaint and the accused are informed of the district's findings and disposition of the complaint.

This strategy for controlling agricultural pollution is premised on education and information, technical assistance, and financial subsidy, where appropriate. Enforcement is a last resort. But to view the strategy as weak and cumbersome is inappropriate because enforcement is administered, when necessary, to attain compliance.

State Responsibilities

Every state has an agency with administrative responsibility for soil and water conservation and conservation districts. In Ohio, this unit is the Division of Soil and Water Conservation, which also has been assigned certain responsibilities for nonpoint pollution abatement.

Specifically, the state legislature assigned the Division responsibility for developing and implementing a program for abatement of agricultural and urban sediment nonpoint pollution. The program consists of rules and regulations for the control of pollution from soil erosion, pesticides, fertilizer, animal wastes, and urban sediment generated by land-disturbing activities. Criteria were also established for subsidizing the installation of sediment control practices and animal waste control facilities that are not cost-effective to the landowner but were required for public benefits.

The Division disburses these subsidies and administers the enforcement procedure for conservation districts when recommended. It also grants the accused landowner or land operator a hearing prior to its final determination and issuance of a court order and penalty schedule if that becomes necessary.

Federal Responsibilities

The U.S. Environmental Protection Agency has responsibility for setting the parameters within which a state formulates its nonpoint pollution abatement plan for meeting requirements of the Clean Water Act. These parameters provide the needed flexibility to states for implementing a nonpoint pollution abatement program tailored to a state's particular problems and institutional arrangements. These institutional arrangements at the state and local levels are not and should not be a federal concern beyond the assurance that the chosen institutional structure has the capability to get the pollution control job done. Periodic progress reports to EPA are justified as a measure of a state's progress in meeting the nation's clean water goals.

EPA must assume leadership responsibility for other federal institutions in compliance with state-adopted program standards and local implementation procedures.

Urban Sediment Control Strategy

Institutional arrangements and strategies for controlling sediment pollution from land-disturbing activities in urban areas are basically the same as in

agricultural areas. The major responsibility for action must rest at the local level. County and city governmental units should be given the authority if they are not already authorized to enact and enforce sediment control ordinances. Both governmental units control development and building permits within their jurisdiction, and such permits should contain sediment and storm water management plans. No special permit need be required. In fact, additional permits are economically and politically frustrating to the permit seeker.

Conservation districts may encourage adoption of such sediment control ordinances and offer technical assistance, or even enter into working agreements for review, approval, and implementation services, as the city or county may desire.

Should cities and counties fail to accept these local responsibilities, state initiatives should be authorized to assure progress and compliance with state-adopted rules and regulations.

Enforcement should automatically fall to the legal arm of the enforcing unit of government, be it local or state. Also, local units of government should be given the authority and opportunity to address urban sediment pollution problems before involving state and federal institutions, beyond encouragement and guidance from those levels.

The state can play a vital role by developing and adopting urban sediment control rules, regulations, and a model ordinance as guides for local initiatives.

Logistical Support Strategy

Local institutions that assume the responsibilities for nonpoint-source pollution abatement have many state and federal agencies available to assist them in varying degrees and ways. It is not only logical but imperative that these agencies be used to the maximum extent. The structuring of new governmental units should be avoided.

The major areas of assistance needed and normally available include the following:

Education and Information. Education of the public and landowners specifically about the chemical and physical aspects of nonpoint pollution is essential to acceptance of an abatement program. The Cooperative Extension Service has a long-standing, highly respected reputation for public education in agricultural technology. This education is founded on research by state and federal agencies. Extension's participation in education should be sought. But information, of course, can be disseminated by any public agency as well as commercial interests and the public media. These educational and informational forces are extremely important to landowner acceptance and public support.

Research. Research must address the causes of and cures for nonpoint pollution. The Agricultural Research Service and state agricultural research

institutions must produce practical and economical pollution control solutions.

EPA has essentially been ineffective in its strategy of allocating research funds for nonpoint pollution control by requiring answers in three years. No defensible data or research finding can be produced in such a short time span when incorporating such unpredictable variables as rainfall quantity and intensity. A more realistic time frame would attract new research urgently needed and produce reputable findings.

Technical Assistance. Technical solutions to preventing nonpoint pollution are directly related to land and water management systems. Both are directly affected by soil and water conservation practices. The Soil Conservation Service, state conservation agencies, and conservation districts have specialized in planning and applying these practices in cooperation with landowners for about 50 years. These agencies can continue to provide the technical assistance needed to deal with nonpoint pollution. These same agencies must exercise caution in using the Universal Soil Loss Equation as a pollution abatement equation. The equation must be used only as a measurement of soil loss from a given site. Delivery of such sediment to a given stream or body of water requires the development of a delivery formula or model.

Local institutions must provide technical and administrative expertise above and beyond that received from state and federal institutions. Acceptable implementation rates for pollution abatement systems dictate that local institutional capabilities be strengthened.

Financial Assistance. Financial support is imperative for the successful implementation of a nonpoint-source pollution abatement program. All levels of government and the polluter must share the financial burden for pollution control.

Federal funding to help solve nonpoint pollution problems must be commensurate with the point-source pollution abatement funding relative to public benefits. To date, spending for nonpoint pollution has been insignificant from a federal point of view. Institutionally, federal funds for nonpoint pollution abatement should represent a realistic portion of the federal funding of pollution abatement efforts by the congressional public works committees to the U.S. Environmental Protection Agency. Such funds should not be provided through the congressional agricultural committees to agricultural agencies. It is logical and imperative that such funds be viewed by the public as water quality funds benefiting the public and not just agriculture as another subsidy. Federal funds need to be allocated to state and local institutions in the form of grants for implementation and subsidies. Experience dictates that this procedure will not only be more productive but more efficient per dollar spent. Any other system of federal financial support will be less effective and produce financial waste.

State funding should supplement federal funding to assure that local implementing institutions are adequately staffed and subsidy needs in a state or area are sufficiently met. State funds can be most effective if they are used by local institutions responsible for implementation.

Local funding should provide for basic facilities and support personnel. Special needs and cooperative efforts can best be funded locally to assure jurisdictional control and direction.

Subsidies to private landowners from any institutional or governmental level should be restricted to those practices required of the landowner that are not cost-effective to the individual but are required to achieve public benefits. Any subsidized landowner practice must be accompanied by a mandatory maintenance provision, and there must be a means for recovering public funds if a practice is not maintained. This mandatory maintenance provision is now used by many state and local institutions, but it has never captured the support of federal agricultural institutions administering subsidy programs for erosion control. The public has a right to require this protection of its investment.

Other Assistance and Structural Issues. Other assistance, such as legal support and private-sector support, must be built into institutional arrangements on the basis of various state and local jurisdictions and capabilities.

The most successful institutional arrangements for nonpoint pollution abatement must be acceptable to the local implementing institutions first, then to state and federal institutions, in that order. The nature and complexity of the problem dictates these priorities of acceptance.

Institutions implementing nonpoint pollution programs must accept the fact that nonpoint pollution cannot and will not be eliminated, but reduced by the best of institutional arrangements. Nature by itself produces significant nonpoint pollutants through natural processes.

The number of polluters essentially equals the number of landowners nationwide. This requires a strong, persuasive effort prior to enforcement action. It also requires a graduation of compliance, dealing with flagrant violators first, which will generate rapid public support.

Progress toward nonpoint-source program goals and clean water objectives must and will move at a somewhat slower pace than has been the case in point-source programs.

A benefit secondary to the immediate purpose of pollution abatement, but possibly an even greater benefit in the long run, is that pollution abatement practices will equally benefit the protection of the food-producing base of our nation. These co-benefits should generate public support politically and financially if properly implemented through realistic institutional arrangements.

WHO SHOULD PAY FOR NONPOINT POLLUTION ABATEMENT?

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ABSTRACT

Nonpoint pollution is an expensive social problem, but abatement of nonpoint costs society as well. Nonpoint policies involve implicit comparison of these two categories of cost and their distribution among policy participants. This paper assembles economic evidence of costs of nonpoint and cost of abatement, discusses distribution, and suggests policy direction.

Emerging policy in nonpoint abatement must place greater emphasis on national efforts rather than state or local. Because those who cause nonpoint are usually separated by both time and space from those damaged by pollution, institutions that bring these interests together are essential. Further, a "polluter pays" philosophy will become more prominent in nonpoint policy. Society will insist that the polluter bear a greater obligation for the impacts. Economic incentives that redistribute cost (a tax, control, or cross compliance) or benefit (benefit share) will improve chances for design of acceptable, workable policy.

Key Words: Nonpoint pollution policy, economics, institutions, distribution of cost.

Who Should Pay for Nonpoint Pollution Abatement

by
Lawrence W. Libby*

Introduction

Nonpoint pollution is an expensive social problem. It imposes enormous costs on water users, a cost recently estimated to total about $6 billion a year (Clark, 1985a). Sediment and pollutants ruin the ecosystem habitat for fish, waterfowl and other organisms that are valued by people for various reasons. Clogged harbors and channels raise the cost of water transport. Nutrient enriched lakes and streams are less attractive for recreation or as part of a living environment; damaged ground and surface water must be treated before use in municipal systems. The costs and consequences of nonpoint have been discussed and documented throughout this conference. But abatement of nonpoint costs society also. Nonpoint policies involve implicit comparison of this cost of abatement with the cost of not abating and the distribution of the two categories of cost. Nonpoint is the unfortunate yet predictable side effect of activities that are generally worthwhile -- food production, construction, and others. Abatement, then, may mean less of that valued activity, or can be direct financial outlay to reduce the water runoff.

This paper addresses the economics of abatement -- both the efficiency and distributional implications of those actions taken to reduce nonpoint damage.Primary emphasis is on agricultural nonpoint pollution. Effficiency consideration basically establish the technical parameters, the feasible set within an abatement strategy may be selected. Distributional impacts influence choice by discribing who pays and who gains from the options available. The most economically efficient abatement technique, whatever it may be, has important distributional consequences. Thus it is just one among many abatement options, evaluated by political actors in terms of what it gives them at what price.

Purpose of this paper is to assemble the economic evidence being generated and draw conclusions that may be useful for this conference and for any subsequent policy recommendations that might be developed. A further purpose is to help set the stage for more specific technical sessions to follow.

First, a series of assertions about "the nature of things" in the political economy of nonpoint will establish context.

1. "Efficient" solution of the nonpoint problem implies the greatest possible abatement for the money spent, achievement of a given level of abatement at least possible cost and/or most importantly, comparing the cost of an additional increment of abatement to the benefits from achieving that increment. The assumption is that when

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an increment costs more than it gains, whoever gains or loses, society has acquired enough reduction of nonpoint pollution. Economists love to repeat this basic rule, but it has real meaning in the social process of allocating public effort among many valued services, of which clean water is just one.

2. Benefit, cost and therefore efficiency in resource use are defined within the structure of property rights that expand or constrain the rights of individual water users. These rights are established and reinforced by public authority. Thus any solution to pollution is valid or efficient only to the extent granted by publicly acknowledged and protected property rights. There is nothing meta-physical about economic efficiency, in abatement or any other production process. It is a fabrication, a result of transactions among actors granted the priviledge of commerce, with publicly protected rights to impose a price for a productive factor. Those rights could change, thus altering the efficient solution.

3. As with most forms of pollution, nonpoint agricultural pollution involves impacts that are separated in time and space from those causing the problem. Pollution is an additional output of a production process generating a "good" of some kind. Property rights protect the producer, limiting his liability for the cost of that undesireable output in the production decisions that determine quantity and form of output. Thus, the benefits of abating nonpoint are distributed differently from the costs of abatement. While those benefiting from abatement might be willing the "bribe" those causing the problem through some sort of compensation program, the costs of organizing the transaction would be formidable.

4. Participants in the policy process form positions on options based on a comparison of separable benefits to separable costs. There is no inclination to pay or bear inconvenience if that action produces no obvious result of consequence to the actor. Some people will bear personal cost to create benefits for others out of a sense of community or altruism. But it is risky to construct nonpoint policy on the assumption that that will occur.

5. In nonpoint policy as in other areas, good policy is acceptable policy. Policy changes that may be technically correct but require too great a sacrifice of other valued services are irrelevant. "Non-degredation by 1985" is and has been such an irrelevant policy goal. Changes occur in incremental adjustments as competitors compare the relevant consequences of options and bargain on that basis. Policy goals and objectives may be stated and used as targets so long a participants do not expect to actually achieve them. Goals and objectives evolve in policy bargaining. They must be flexible enough to accomodate reality (Lindblom, 1979).

In nonpoint abatement, the efficiency questions involve comparing the consequences of pollution with the consequences of abatement. Policy participants must understand the cost of failing to reduce erosion and compare to the cost of cleaning up, to arrive at the marginal benefit/marginal cost comparisons discussed above.

The Evidence on Cost: A Brief Review

Various empirical studies have sought to establish the magnitudes of the two types of cost -- pollution and abatement. There is further development of methodology to improve the comprehensiveness and reliability of these estimates. Shortage of site specific water quality data continues to be a problem in seeking economic estimates of damage.

Cost of Nonpoint. The cost of nonpoint pollution represents a reasonable starting point for estimating benefits of abatement. Some non-user benefits might be omitted from the cost figures, thus cost of pollution may understate the actual benefits from abatement. But certain instream costs to the biological ecosystem are also omitted. There is an implicit property rights issue imbedded in this cost/benefit template of nonpoint pollution. Those hurt by pollution are basically trying to defend something they already have -- a clean water supply. The polluter, however, implicitly owns the right to convert that clean water into the combination of dirty water and a product,both of which are outputs of the production process in question. The payment that the downstream user would be willing to accept to part with clean water would likely be greater than what he would be willing to pay to acquire a new supply of clean water (Bishop and Heberlein, 1980). Initial allocation of property rights, therefore, will influence benefit estimates of nonpoint abatement. Magnitude of that difference would be a useful empirical question to pursue.

The most definitive national estimates of nonpoint costs have been generated recently by the Conservation Foundation(Clark, 1985b). Authors assemble the most useful empirical literature on the subject, impose a few defensible assumptions, and develop some estimates. While more specific case studies within particular watersheds could improve the quality of the national figures, the CF estimates are a significant contribution to policy development in the area. Pollutants include sediment, nutrients and chemicals. Instream damages are defined to include impacts on the aquatic environment, reduction in recreation benefits from fishing and other water sports, reduced water storage and holding capacity of reservoirs and lakes, increased cost of maintaining harbors and water ways, and loss of certain non-user values associated with clean water. Their estimate of these instream damages is $4.3 billion per year, of which agriculture causes $1.6 billion. Off-stream damages result from increased flooding associated with suspended sediment, less efficient water conveyance systems (canals), and more expensive water treatment systems for power generation. The total off-stream estimate is $1.9 billion of which agriculture accounts for $660 million. These are acknowledged to be "order of magnitude" estimates (Clark, 1985a, p. 22).

A recent case study of the Obion-Forked Deer River Basin in Tennessee estimated offsite damagaes from erosion totalling $74 million per year plus losses associated with flooding of productive hardwood forests (USDA, 1980). Impact of declining water quality on value of waterside property along St. Albans Bay on Lake Champlian was estimated using an hedonic model. The water quality variable constituted approximately 20% of property value or $4,500 per property difference attributed to quality of the adjacent waterbody. Total damage experienced by owners of 430 single family residences was $2 million (Young and Teti, 1984). Other case studies are needed to get a more accurate fix on economic burdens associated with nonpoint pollution.

Benefits of Abatement. Another perspective on the cost of pollution side of the nonpoint policy question may be gained by estimating benefits of cleaning up the water. Benefit is more than just damage avoided. It also includes the liklihood of continued availability of clean water and the impact of that liklihood on willingness to invest in waterside property or other services that require clean water.

The Economic Research Service of USDA is putting considerable effort into estimating benefits of abatement in the various rural clean water projects (RCWP) around the country. Work is still underway, but a few results are available. Improving the quality of water in St. Albans Bay in the northeastern part of Lake Champlain in Vermont would produce annual benefits estimated at $230,300 through a contingent valuation technique and about twice that using a travel cost approach (Ribaudo, 1984).

The travel cost model involved estimating the cost that a recreator has been willing to bear to enjoy the bay for various recreation experiences and the amount of travel expense that recreator would pay if the bay were cleaner. Number and costs of trips from various origins in the region were estimated. Relationship between the willingness to bear travel cost and distance from St. Albans, availability of alternative sites and income levels was calculated for current water quality and for the hypothetical situation of a cleaner bay. Mean difference in willingess to pay for recreation attributable to improved water quality was $123 per recreator, for a total recreation benefit of $530,700. The contingent valuation model involved interviews of prospective recreators who were asked to rate several recreation opportunities involving St. Albans Bay and then re-rate those options with the possibility of a cleaner bay. These indifference maps were then converted to monetary benefits associated with improved water quality in the bay.

Benefit estimates of several other clean water projects will be published soon. Each is unique, with land use, soil, and water characteristics that differ significantly from site to site. Generalization is impossible. Yet these point estimates can improve chances for public agencies and various political participants involved in clean water debates to draw conclusions that are economically sound. With reasonable estimates of abatement costs and their distribution, one may draw a conclusion about the efficacy of a particular abatement program.

Cost of Abatement. The greatest amount of published work in the economics of agricultural nonpoint pollution deals with the costs that selected management techniques could impose on the farmer. These studies generally assume initial distribution of property rights that gives the farmer the right to decide for or against a reduction in run-off. In a 1976 linear programming analysis, Kasal examined farm income consequences of imposing limits on fertilizer use and soil loss as measures to reduce nonpoint pollution. For the sample farms selected and alternative measures imposed, nonpoint abatement reduced farm incomes by from 10 to 36 percent (Kasal, 1976). A national LP model run by Wade and Heady made similar kinds of estimates of increased production costs associated with nonpoint measures, within the overall requirement of meeting projected food demands. The minimum sediment solution to the production cost minimization model was 42% higher than the unrestricted solution. In both cases there was considerable cropland adjustment within the model to retain sufficient production to meet output requirements (Wade and Heady, 1977).

Alt, et al (1979) examined the farm level costs implicit in reducing sediment delivery to a specific reservoir in central Iowa from the Iowa River watershed. A 10 ton acre limit on gross erosion from farms in that watershed would increase production costs 17% while reducing sediment delivery by 91%. White and Partenheimer (1979) examined income consequences of erosion reduction plans for a sample of Pennsylvania dairy farms. Impacts varied among the specific farms studied, with two farms experiencing increased revenues from the recommend soil conserving plan, four with income reductions of less than 5% and six with reduction from 7% to 30%. A no-till option was introduced with the result that all but one farm showed increased returns. The net on-farm/off-farm economic consequences of erosion reduction were examined for a particular river basin in northeast Texas. The general conclusion reached in this analysis was that the positive on-farm impacts of soil conservation overshadowed whatever revenue reduction might be associated with complete attention to reducing off-farm damages. Thus, authors conclude, it is in the farmer's interest to conserve soil and no regulation or subsidy is warrented assuming that farmers are economically rational (Reneau and Taylor, 1979).

Results of these and many other case studies of the economic costs of various erosion control techniques have provided important analytical back-up for recommending best management practices in specific state and local water quality plans. The whole approach to nonpoint reduction has been local solutions to local problems, with "best practices" a function of the physical, economic and institutional circumstances prevailing in that area. A systematic approach for considering the various factors involved in selecting best management practices for a particular political/economic/hydrologic sub-region has been suggested by Bailey and Waddell (1979). The general policy goal has been to establish an abatement program that avoids major cost impact on local farmers. Procudures for considering the most important firm level costs have been recommended by USDA and EPA (1975) for the development of those BMP's (14). Inter-regional and national consequences of these strategies have been estimated also. The more aggregated the analysis or modeling becomes, the more imprecise and unreliable the estimates because of obvious differences among areas.

A significant and widely acknowledged gap in the evidence on alternative nonpoint strategies is the link between technique and water quality improvement (Christensen, 1983). We know that nonpoint costs money; we know that abatement costs money; we even know that the two costs are distributed differently. But we need much better evidence on the aggregate water quality results of farm level practices that reduce run off, so that the estimates of economic benefit of those actions may be improved.

Reduced tillage alternatives show particular promise since they can, under some circumstances, lead to increased net farm revenue while reducing run-off. Pope, et al (1983) concluded from their study of Iowa agriculture that conservation tillage systems have less negative impact on farm income than is true for conventional systems. Study of alternative tillage systems on Ohio farms found that no-till options that reduced erosion to T or less also produced higher net revenue than the base solution of fall plowing on soybeans (USDA, 1983). In his study of cash grain farms in the Jackson Purchase Area of Kentucky, Kugler found that for nearly all farms there was a tillage conversion that would increase net returns while reducing erosion (1984). Black, et al (1984) concluded that since yield is not affected by tillage system for the sample of Michigan farms studied in the Saginaw Bay watershed, the net revenue advantage for conservation tillage comes from reduced machinery and labor costs. Determination as to whether or not reduced tillage makes sense as a best management practice for nonpoint abatement depends on the net effect of that practice on the environment. While conservation tillage often reduces soil movement and runoff, it may also result in increased chemical concentration in the run-off water that does leave the field. The plant residue left in the field as part of the reduced tillage system tends to increase insect problems and reduce the effectiveness of some herbicides (Baker, and Laflen, 1984). Any calculation of the cost of nonpoint abatement using reduced tillage or any other erosion reducing technique must consider offsite damage as well as income impacts for the farmer. Conservation tillage may reduce erosion at little direct economic cost to the farmer but if it also increases chemical contamination of rivers and streams because of increased use of pesticides, its viability as a best management practice is questionable.

The real difficulty for an individual or government in reaching an efficiency decision on nonpoint investment is that information comes in fragments. Cost and benefit are not felt at the same time or place. At the farm level, an action that reduces nonpoint may have advantages in net revenue. On the other hand, a practice that reduces erosion and increases the farmer's revenue, may worsen the water quality problem. As analysts we try to isolate factors, examine them one at a time for their

positive and negative consequences. In fact, however, the farmer makes decisions within a complex of incentives, risks, and options. It may be that an incentive necessary to attract socially rational investment in nonpoint abatement should be artifically linked to the action that would reduce runoff. That is the general idea behind cross-compliance in soil conservation policy. As long as the positve incentive is greater than any income burden from the resulting nonpoint technique, it can be a rational choice for the farmer (Dinehart and Libby, 1981). So long as the public cost of the incentive, including whatever unintended side impacts it may cause, in less than the incremental cost of pollution it can be an improvement.

Success of alternative policies to reduce nonpoint pollution will depend primarily on the distribution of both the cost of pollution and the cost of abatement. In policy we are more concerned with who pays than with the overall size of the bill.

Who Pays?

Distributing the burden for reducing agricultural nonpoint pollution, assuming there is general agreement that nonpoint should be reduced, is the central policy question in this area. The prevailing theories are 1) those who benefit from abatement should pay for it, and 2) those who cause the problem should pay for its solution. The former implies that the right to permit run-off remains with the polluter; those who want less pollution because they currently experience a pollution cost, must pay for abatement. The problem, then, is to decide who benefits and levy a tax accordingly. In large measure this is the current system. Farmers and other polluters retain the right to pollute; abatement is voluntary; cost sharing and technical assistance are paid for by taxpayers. State level cost sharing programs exist in many states, implying that benefits are received by all state taxpayers. As of 1983, sixteen states have their own cost share programs (Braden and Uchtmann, 1984). Federal cost sharing, through ASCS of the US Department of Agriculture, implies that all US taxpayers would benefit from reduced pollution and should therefore pay for it. In practice, state programs actually supplement this federal support.

The latter approach to distributing the cost of pollution implies that the polluter has full liability for his actions and that any cost of abatement must be included in the cost of production for the commodity or service that creates to pollution. Thus far, the only method for implementing this approach to distributing abatement cost has been through regulation. County soil conservation districts in most states have the authority to impose regulation against excessive erosion, though few have exercised that authority. Five states have sediment control ordinances that include agriculture. Iowa has the most aggressive and comprehensive regulations against excess erosion. Illinois has a newly enacted law designed to "jawbone" farmers toward acceptable erosion levels by the year 2000. Pennsylvania has an innovative permit system functioning through the state's Clean Stream Law (Holmes, 1979, p. 63-93).

Other mandatory approaches being examined for soil conservation programs include various cross compliance measures that would require soil conservation as pre-condition for eligibililty for price supports and other income support programs. While these have not really been designed as nonpoint abatement measures, they could help meet water quality goals.

The matter of "who <u>should</u> pay" for nonpoint is obviously a matter of opinion. Conclusions presented here are based not on perception of right and wrong (though some element of that may creep into the discussion) but on a judgement of what is likely to

happen. The "benefits received" approach is well received in the agricultural community including, incidentally, most of the Land Grant system. The obvious problem is, however, that it has not worked. Policy development for the next decade will shift more generally toward the "polluter pays" approach reflecting concern for cost effectiveness in nonpoint and other natural resource policy as well as a growing impatience among the pollution control professionals about progress toward reducing nonpoint. The magnitude of the estimated national cost of nonpoint pollution and agriculture's portion of that total has the attention of the Soil Conservation Service. Estimated annual costs imposed by agricultural nonpoint pollution are several orders of magnitude higher than annual cost of soil productivity lost to erosion. SCS must give greater attention to off-site consequences of erosion in these days of national budget stringency for agriculture natural resources and for SCS in particular.

Policy Conclusions. The following conclusions on the general trends in policy and cost distribution are based on a review of the economic evidence on nonpoint summarized briefly above, review of policy literature concerned both with soil conservation and water pollution, and a general "testing of the wind." Following these brief conclusion, is a list of suggested innovations for policy in this area.

1. Abatement of nonpoint must be a national program, not just the sum of state programs. Despite the fact that 208 water quality planning, the Model Implementation Program and the Rural Clean Water Program emphasize state action, and the Reagan administration prefers decentralization, water pollution is just as is national as national defense. The implicit assumption that beneficiaries of abatement are only the first line pollutees who directly feel the results of dirty water, and that they reside in the same state as those causing the problem cannot be sustained. In fact, benefits of a clean water supply go well beyond immediate users. They include various non-user benefits from continious availability of water when and if it might be used. They include various multiplier impacts on the overall economic environment of a region where the farms, cities and countryside become attractive places to live and work. Clean water is part of a broad sense of well being or security that will encourage investment and general popular support for change.

A recent article in the Journal of Soil and Water Conservation was entitled "Saving the Chesapeake: Maryland's Agricultural Education Program," (Magette, 1985). Even if the title were changed to refer to "Maryland's Ironclad Regulation Against Erosion," it would be wishful thinking. There is just no way that the problems of the Chesapeake Bay can be handled separately by Pennsylvania and Maryland. Even when the pollution problem is reasonably well confined as in Saginaw Bay, Michigan or Green Bay, Wisconsin benefits of abatement extend well beyond. There has been important institutional innovation in states, and that must be supported. The Wisconsin nonpoint program is an exceptional example of a balanced and integrated program with all components from problem identification to implementation (Konrad, et al, 1985). RCWP's have become testing grounds for coordination among federal and state agencies. They have been strongly endorsed by EPA and UDSA (Groszyk, 1979; Unger, 1979). But beyond these intermediate purposes, reducing nonpoint to acceptable levels will require federal action. It is unreasonable to expect an agricultural state to enact abatement measures sufficient to produce adequate water quality benefits for downstream users in other states. To a large extent we have allowed the traditions of state and local prerogative on land use management to run the nonpoint program. We begin with the irrevocable presumption that locals know best when it comes to land use. Perhaps that is true, but it will not solve the nonpoint problem. If land use controls are the only solution to nonpoint, and land control must be a local government function, then nonpoint will be with us for a long time.

A national program for nonpoint abatement means that the cost will be shared among payers of national income tax. State programs may supplement but never replace the national focus. Taxpayers in agricultural states will pay more than taxpayers elsewhere since farmers in those states will likely have sufficient political clout toshift the abatement cost and get additional cost sharing programs installed.

2. Farmers and others causing nonpoint pollution will pay an increasing portion of the abatement cost. Most of this cost will be indirect, in the form of fewer property rights to use land in ways that cause nonpoint pollution and through any reduced revenue associated with less erosive farming systems that might reduce output. There is simply no possible way that continued reliance on technical assistance, cost sharing and voluntary action can satisfactorily cope with the nonpoint problem. Reduced tillage has really been the only hope for this softer approach and recent evidence on the pollutant concentration in run-off raises serious question about the viability of these techniques. This should not suggest that the voluntary approach with or without the other elements is inadequate in every case. There have been important successes. But in a national nonpoint program, these measures are simply inadequate.

There is nothing earthshaking about this conclusion. Resource policy professionals have known for somtime that only by changes in farmer behavior will agricultural nonpoint be reduced, and current incentives do not favor the necessary changes. The RCWP era in water quality policy has been one of support building and institutional design within the prevailing balance of power and responsibility among levels of government. It has been necessary. But there will be pronounced shift in philosophy toward the "polluter pays" approach. Those who stand to gain from abatement, a broadly defined community of water users, will be unwilling to leave all of the land use discretion in the hands of those who cause the problem. Epp and Shortle suggest that given this inevitable policy shift, the water quality agencies and research community should focus on design of effective and economically efficient mandatory programs (Epp and Shortle, 1985, p.111). That is sound advice.

Distribution of burden within agriculture deserves policy attention as well. Farmers on erosive land will find it particularly painful to meet mandatory limits. They suffer the initial disadvantage of less productive land and then the higher cost of compliance. The inevitable result will be further concentration of agricultural production on the most responsive and easily protected lands of the country. Additional people and areas will be displaced from agriculture. There must be special attention to these poeple to facilitate transition and limit the hardship involved. But attempting to mask the inherent disadvantage of farming these lands through special credit or disaster payments would be an expensive and apparently inappropriate policy response. With mandated erosion and run-off standards, the inherent advantage of some lands would likely be reflected in land price. Technology may reduce the differences between erosive and non-erosive lands, however.

Directions for Policy Innovation

The major needs in nonpoint involve, first, new organizational strategies for getting those who cause the problem together with those who gain from its reduction. As noted, costs of pollution and costs abatement are distributed differently. Only by creating a political environment conducive to negotiations between these categories of participants can lasting policy change be accomplished. The second major need is for financing schemes that take advantage of the economic facts of pollution and abatement. Economists have long been reviled for seeking social purpose through inherent selfishness of the individual. That is an unfair assertion, of course. The more positive perspective is

to facilitate expression of the true consequences of alternative actions so that private choices will yield results that are socially responsible.

1. The first suggestion is not new at all but entails giving new life and new missions to an old institution -- the river basin commission. There is logic in hydrologic units! As water resource professionals have known for many years, the river basin unit can overcome much of the transaction cost associated with bargaining between gainers and losers from a pattern of water use. This capacity can be particularly important in nonpoint abatement where the source of the problem is diffuse, benefits widely distributed in small increments. Some of these inter-state linkages have been established for the Chesapeake. Experience in the Deleware and Susquehanna River Basins has been well documented (Libby, 1970). Bay area regional commissions with multi-state memberships could be established or focused more clearly on nonpoint problems. Purpose of the water shed, or more generally "problem-shed", is to establish the setting for bargaining over the terms of nonpoint policy. There should be no claims of designing the optimal water management program. Instead, interested parties could iterate toward acceptable levels of pollution and abatement, and a distribution of burden that is tolerable. With focus clearly on nonpoint, voting and membership rules would have to encompass the essential parties involved. A different organization might be appropriate for other resource problems in the basin.

Hydrologic units are awkward to adminster. They cross many jurisdictional boundaries and lack political legitimacy. The administrative overhead can be substantial. The problems are not much different from those of multidisciplinary research is an acedmic institution. But there can be no lasting reduction of nonpoint pollution without some mechanism of forcing confrontation among the interested parties. Provision must be made for beneficiaries outside the basin and some basic understanding of problem source and benefit of improvement. Water quality standards would help establish targets. Even with mandatory controls on erosion, the mechanism for bargaining would be valuable.

2. Within a decision or bargaining unit such as a river basin, a mechanism for redistributing the benefits of abatement could improve chances for incremental resolution of the nonpoint problem. Such an innovation has been analyzed by Park and Shabman (1981) for the Occoquan River Basin in Virginia. Under such a scheme, those who gain from abatement could spread the benefits sufficiently to generate support necessary for a compensation scheme that will bribe the polluter not to pollute. Alternatively, this system might produce the plurality to enact mandatory controls. Areas receiving major measurable benefit from abatement could compensate areas that would benefit less to help finance installation of BMP's with the basin commission acting as banker. As long as net benefits to all parties benefits are positive, it is in their interest to share to build support. The basin commission's role is to reduce transaction cost associated with bargaining both among beneficiaries and between those who cause and those hurt by nonpoint pollution.

The Chesapeake Bay situation would seem to lend itself to the type of innovation suggested by Park and Shabman. Any such system would require improved information on water quality benefits and impacts of alternative abatement techniques.

3. Another device for employing an economic incentive to encourage private actions with a social purpose is installation of a tax in pollution. Economists like this idea particularly well because it forces the producer to internalize social costs of the production process. The tax also creates the incentive to reduce pollution to avoid the tax. While there apparently is no real experience with a tax on nonpoint pollution, Seitz

et al used a linear programming model of hypothetical farms representing western Illinois agriculture to conclude that a soil loss tax would be more efficient than an erosion restriction in reducing nonpoint (Seitz, et al, 1979, p. 375-376). Under a tax scheme, the farmer would seek the least costly means for reducing run-off, unique to the land conditions on that farm. The erosion standard or restriction, on the other hand, might impose a more costly technique than is necessary.

The obvious difficulty with tax schemes is absense of specific information on performance of practices, cost of pollution and other essential variables for a given farm. It may be that trial and error would be a reasonable research design in this area of building institutions. The theory of tax schemes is well developed, but implementation is lacking. Selected pilot studies could be a reasonable use of scarce research funds.

4. Marketable or tradeable rights to pollute constitute a third type of innovation for marshalling the forces of greed in the interest of society. The sediment and waste assimilative capacity of a stream or lake might be allocated among farmers on the basis of some readily determinable index such as frontage on water course or acreage adjusted by an indicator of accessibility to water. Then the farmer would have the right to sell or trade those rights when their value exceeds the cost of reducing run-off. Implementation of such a scheme would require monitering to determine compliance for a particular farm. There might also be a problem of localized pollution, where farmers decide to acquire pollution rights and let the pollution occur. Those farming areas that can reduce run-off most readily will have much cleaner water. The average , or overall assimilative capacity, could hide extreme situations that impose local hardship. There would be overhead cost involved for the basin commission or other managing unit in defining terms of trade and upholding the result.

After thorough review of the economics and institutional literature related to nonpoint pollution one could easily conclude that there is really very little that is new. There are few revolutionary new ideas. In fact, there is a fair amount of repetition at the conceptual level. What is lacking is experience, empirical observation of various means of coping with disparate patterns of distribution of the cost of pollution and cost of abatement. There are limits on acceptable means for redistributing those burdens, but the limits are less severe than traditionally assumed. Acceptability is a function of prevailing attitudes about who can or should shoulder more of the burden for meeting a widely held social objective for clean water.

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Panel Reports

SUMMARY
Panel I
Planning Processes for Nonpoint Pollution Abatement Programs

PANEL MEMBERS

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INTRODUCTION

Panel discussion and audience participation for Panel I involved planning processes for Nonpoint Pollution Abatement programs. Although well documented scientific data exist to support the fact that pollution other than from "end-of-pipe" or point sources exists, defining what is considered nonpoint pollution and a plan to remedy its consequences is still largely undefined. Identifying problems and goals, preparing a technical and economic attack, and drawing together conclusions to determine the effectiveness of the program, are essential to the overall success. The engineering, scientific and planning communities are dealing with such immense pollution problems taking place in broad, complex ecosystems, that no one person or approach can completely solve them. This panel has quickly concluded that general planning steps (see attached flow sheet) are necessary to implement a regional nonpoint pollution abatement program. However, one must be aware that program needs will vary greatly from one region to the next and, therefore, a wide variation in application and achievement in different programs can be expected.

FINDINGS

Many extremely important issues were raised concerning proper and effective

program planning and implementation. The following issues, aspects and conclusions were found to be the most significant:

1. Widely accepted means of defining nonpoint pollution problems are missing. Water quality standards, per se, such as are employed in point source work, do not exist and probably will not exist for some time to come. Criteria based on receiving water, beneficial use preservation, may represent the best strategy but more understanding of the effects of nonpoint discharge is required to implement it.

2. Substantial nonpoint source research findings are already available to partially support the development of problem assessment methods, although little effort has been made to do so. Further progress in this area will require much additional study. Success in these forthcoming studies will require programs in which great attention has been given to definition of general and specific objectives, and prioritization of program elements to achieve the maximum gain in knowledge for the resources expected.

3. In the nonpoint source water pollution field, unlike many other scientific and technical disciplines, there is little standardization of investigation methods. Greater standardization should be promoted to allow more interchange among investigators and reduce at least some of the duplication of effort now occurring.

4. Dealing with superimposed point and nonpoint loadings to remedy the cumulative, concurrent pollution burden in receiving waters must be addressed.

5. Nonpoint pollution assessment via the use of mathematical models is an indispensible tool, but is rendered useless without well organized sampling and data collection efforts. In addition to the site monitoring that takes place preceding model development and calibration, a continued long-term monitoring effort must be made to verify the model and evaluate any control techniques that are implemented. The time lag between project implementation and expected results is politically unattractive and can be a very sensitive issue.

6. Effective, well-defined legislation regulations are imperative to research project goals, but the effort is merely rote if proper enforcement does not exist.

7. The effects of nonpoint pollution loadings to groundwater must be assessed. This investigation must be concerned with trade-offs between surface water improvement and increased groundwater contamination resulting from strategies promoting infiltration of nonpoint runoff. More research is needed to determine the extent of chemical and biological pollutant reduction processes occurring within the reduction processes occurring within the soil layer. If these mechanisms are overlooked, erroneous conclusions may be reached regarding hydraulic and pollutant loading rates that can be permitted on land.

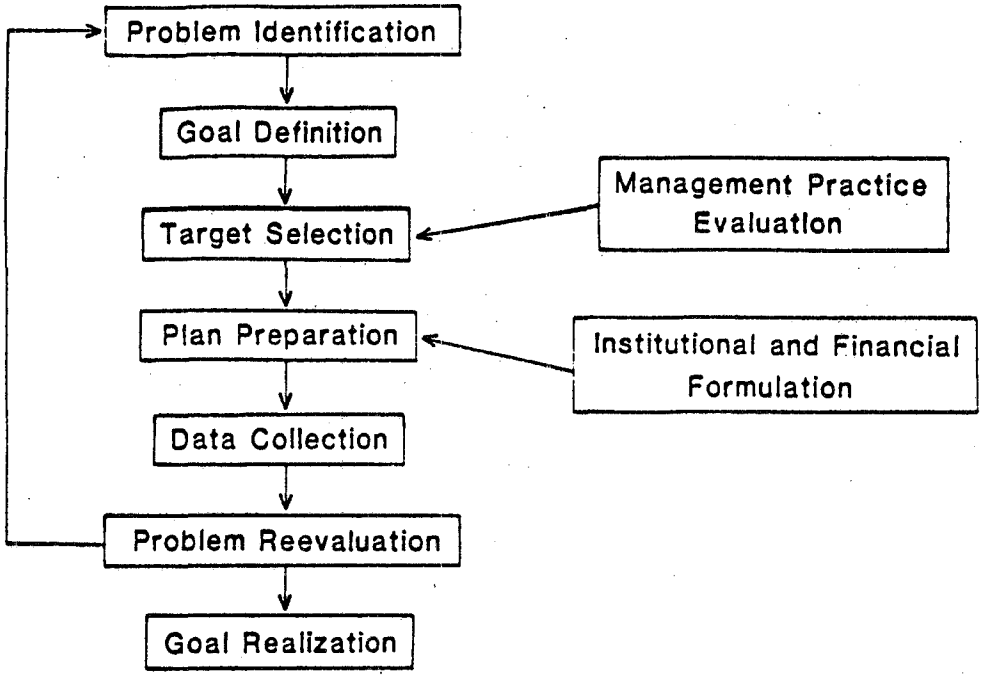
8. The impact of nonpoint loadings from the atmosphere appears to be an area in which little research exists and which poses a great problem to nonpoint water pollution managers. The political and institutional implications are many and could be extremely complex.

9. Our society works on an action/reaction basis. The professional community must determine what action(s) must be undertaken in order to make the public aware of the extent of nonpoint water pollution and to stimulate the reaction that is needed, i.e., support for study and abatement programs.

10. It appears at this point that the engineering and technological hurdles are lower than those encountered by planners. Identifying and researching nonpoint pollution problems in "critical areas" is essential, but projects based in these areas are doomed to fail if public participation and cooperation are not acquired.

CONCLUSIONS

It is the consensus of the panel that, although researchers are well aware of nonpoint pollution problems and many of their implications, very little planning and program implementation has taken place. Solving our point source pollution problems has been difficult but the challenge, politically, technically, and economically, to improve the overall quality of our environment has just begun. Public awareness and cooperation is essential and the overall challenge will stem more from sociological than technical feasibility problems. Essentially, the nature of the problem stems from the fact that water moves. In its movement, it does not recognize municipal, state or any other political boundaries. Truly, what happens in Minneapolis affects those living hundreds of miles away in New Orleans. What programs will be implemented to halt or reduce these problems, who will pay and how much and what level of political authority will take precedence in assuring continued environmental regulation, compliance and enforcement, are all issues that were examined in our panel discussion and must continue to be addressed.



Planning Process Steps For Abatement Of Nonpoint Pollution, From "Fundamentals of Watershed Management", Presented by Gary Oberts at the International Symposium on Lake and Reservoir Management, North American Lake Management Society, Oct.16-19, 1984, McAfee, N.J.

IMPROVEMENT OF ENVIRONMENTAL IMPACT ASSESSMENT OF NONPOINT SOURCES OF WATER POLLUTION

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ABSTRACT

Abatement of nonpoint source water pollution problems relies upon the ability to assess the dimensions and effects of those problems. This ability is not highly developed, and advancing it is hindered by difficult problems. Nevertheless, progress can be made by systematically building on a research foundation to develop impact assessment strategies providing the necessary basis for decision-making. This paper discusses examples of such strategies from our research. Included are techniques to design cost-effective monitoring programs to gather nonpoint source data and to carry out stormwater runoff sampling. Also discussed is a stepwise guide to assess quantitatively the aquatic impacts of operating highways and to allocate resources to their mitigation. These approaches offer means of overcoming a number of the problems impeding nonpoint source impact assessment and have potential application in other settings.

INTRODUCTION

The passage of federal and state water pollution control legislation in the past 15 years has spurred widespread actions to improve surface water quality in the United States. For much of this period the majority of the attention has been directed toward control of point sources of water pollution. Continuing water pollution problems in cases where some success has been achieved in reducing point source contributions has suggested, however, that more distributed sources also are major factors in surface water quality. These distributed contributors have been termed nonpoint sources and are associated with stormwater runoff from land surfaces put to various uses.

A prerequisite of acting effectively to solve nonpoint source water pollution problems is to be able to assess their dimensions and their influence on affected ecosystems. This process of environmental impact assessment has a fifteen-year history stemming from the passage of the National Environmental Policy Act (NEPA). Despite this rather lengthy period, little agreement has developed concerning how to conduct assessments in nonpoint source water pollution cases. The underlying legal and regulatory framework provides little distinct guidance, and practitioners have not had discussions in the breadth and depth needed to adopt and disseminate proven techniques and to promote standards of quality. As a result, nonpoint source impact assessments, much too frequently, cannot provide meaningful, objective problem analyses that can support effective decision-making. In fairness, it should be noted that our field of interest in this symposium is certainly not alone in this state.

Much of the applied environmental research performed since the adoption of NEPA has had as an implicit objective the enhancement of abilities to conduct environmental impact assessments. While substantial data bases and many useful methods have resulted from these efforts, there have been few attempts to apply the knowledge gained to developing comprehensive assessment protocols. Even more rare has been implementation of these research results in the practices of organizations that must prepare or evaluate environmental impact statements.

It is the central premise of this paper that environmental impact assessment of nonpoint sources of water pollution can be improved by developing systematic methods firmly rooted in research results. Further, the research community should take the lead in formulating these methods and should work closely with practitioners and regulators to implement them. At the University of Washington we have been addressing these issues in our nonpoint source work. This paper will discuss the problems we have recognized, the steps we have taken to solve them, and where we see the need for more widespread efforts. We believe our general approaches have potential application in other locations, to different nonpoint problems, and in many other situations.

PROBLEMS IN NONPOINT SOURCE WATER POLLUTION IMPACT ASSESSMENT

Developing means of assessing environmental impact necessitates some degree

of generalization, whereby knowledge gained in one instance can be applied with acceptable reliability at another time and, often, at a different place. The diffuse nature of the nonpoint source water pollution problem creates numerous and significant difficulties in meeting this necessity in the presence of the limitations on financial resources that always exist. These difficulties arise at every step in the process, from data collection, through data analysis and interpretation, to implementation of results in usable form.

Probably, some progress could be made in systematizing nonpoint assessment by applying methods already developed and reconsidering existing data with this objective in mind. Great progress, however, will require substantial new data collection. Planning effective programs for this purpose is the first obstacle. In any sizable nonpoint source water pollution problem, the potential subjects for monitoring are so extensive that full coverage is impossible. In the past, there has been little consideration of how to select scientifically among all the possible tasks and how to allocate the available resources. Further, there is little standardization in the nonpoint source water pollution field of specific sampling and data analysis techniques. Therefore, results frequently are not transferrable from one time and place to another.

Natural conditions encountered in nonpoint source water pollution work reinforce and add to these problems. The extreme variability documented (Wanielista, 1978; Horner and Mar, 1982) points out that nonpoint water pollutant concentrations and mass loadings are very site- and time-specific. Part of this reported disparity undoubtedly is due to method inconsistency, but variations in topography, meteorology, climatology, soils, geology, specific features of land use, and other factors seem to make great dispersion in the results of nonpoint monitoring an inherent characteristic. The result is that storm runoff usually has a greater diversity in contaminants than industrial and municipal sewage effluents, with less predictability of their levels. In addition to the measurement of physical and chemical constituents of runoff is the issue of whether receiving water biota should be observed in situ or in vitro. All of these considerations expand the list of operative variables beyond a measurable number.

Intertwined with all of the problems noted above are the issues of long-term versus short-term impacts of nonpoint water pollution sources and the cumulative effects of a number of activities in a watershed. As in other endeavors in the environmental sciences, most of the existing understanding is based on short-term observations of the acute effects on aquatic biota of high levels of single contaminants. Low-level, longer-duration exposures, synergistic effects of multiple pollutants, responses of communities of species, and how human actions influence the accumulating burden on receptors are all subjects that have received little study.

Assuming that these problems of data collection and analysis can be overcome, putting results to use in impact assessment is still a formidable problem. Not all occurrences will have significant impacts, and analysts should have means of distinguishing those that almost certainly will not from those that may. Most of the analytical effort then could be directed toward the latter. In this task, analysts need techniques that can produce

an objective and fully justifiable assessment in a convenient fashion. The limited resources available for impact mitigation should be allocated to actual problems identified through such a process.

Although the difficulties with nonpoint assessment described are daunting, some strategies are developing to make them manageable. The next section will present our major contributions to this progress and the remaining needs that we believe exist.

APPROACHES TO IMPROVING NONPOINT SOURCE WATER POLLUTION IMPACT ASSESSMENT

Monitoring of Nonpoint Sources

We have just begun the development of a methodology to guide the design of monitoring programs for assessing nonpoint source water pollution problems throughout Washington State. This effort is an extension of work we have completed for the electric power industry (Mar et al., 1985). We contemplate creating a conceptual framework to assist a user in identifying the potential causes of impact and subjects for monitoring and in formulating hypotheses to test in monitoring. A central feature of the framework will be a procedure (based on eigenvalue mathematics) to rank monitoring subjects and hypotheses with respect to designated criteria, so that the many possible variables can be reduced to those most likely to yield the needed information. Another feature will be a computational algorithm to determine the optimum sampling program design (number of stations, frequency, and replication) for a cost constraint or, alternatively, a constraint on statistical power. The product of this research is expected to focus Washington State nonpoint source monitoring programs on definite objectives and to improve their cost-effectiveness. We believe similar approaches are possible and advisable elsewhere.

In the absence of widely agreed upon specific nonpoint sampling procedures, we have given considerable attention to this subject. Upgrading the quality of the methods used and promotion of interchangeability of results will not occur until a standard-setting body outlines and promulgates standards that all will adopt. Increasing the standardization of experimental practices is a major prerequisite to the improvement of nonpoint source impact assessment.

The most common means of sampling stormwater runoff has been to collect a number of samples at discrete points throughout the runoff period using automatic equipment. This strategy produces a relatively complete record but is costly in both sampling equipment and laboratory analysis. We developed an inexpensive technique to fractionate and composite runoff from an entire storm, thus reducing both costs greatly (Clark et al., 1981). Our strategy is to perform enough discrete sampling through storms to understand maximum concentrations, followed by storm composite sampling to assemble a large whole-storm data base. The first data set allows assessment of intense, acute effects, which we also investigate with laboratory bioassays (Portele, 1982). The large latter set assists in resolving the problems of temporal and spatial variability discussed earlier and permits both deterministic modeling and statistical analysis, as described below.

Protocols for Impact Assessment

We conducted a five-year study of storm runoff from operating highways in Washington State that concluded with the development of a stepwise guide for assessment of aquatic impacts (Mar et al., 1982; Horner and Mar, 1982; 1983). This guide has several features that illustrate our approach to systematic nonpoint impact assessment based on research results. First, its organization is hierarchical, so that cases having differing potentials for significantly impacting aquatic ecosystems can be distinguished and proportionately allocated resources for problem assessment and solution. An initial screening process, based on specific criteria, removes those cases highly likely to create an insignificant impact from further consideration. For those cases analyzed further, the basic premise of the assessment is that the highway impact on the receiving water can be assessed most realistically in the context of the aggregate burden created by all activities in the watershed. This approach attempts to deal with the issue of cumulative effects, although much more insight is necessary before this phenomenon can be represented adequately.

The large storm composite data base resulting from the research was employed to develop a model of accumulated pollutant loadings delivered by highway runoff to the receiving water over an extended period (Asplund et al., 1982; Chui et al., 1982). Variability among individual storm events was too great for deterministic modeling, but the data were applied successfully to develop a probabilistic method for assessing short-term effects (Little et al., 1983). This approach exploits the log-normal distribution of the individual storm data and permits the impact analyst to estimate the frequency with which a given pollutant concentration, such as a water quality criterion, would be exceeded in a receiving water as a result of highway runoff. A similar technique was developed by the U.S. Environmental Protection Agency (1982) for assessing the effects of urban runoff. It also was recommended by Loftis et al. (1983) in a general review of statistical models that might be applied in water quality regulation. Modeling of cumulative pollutant generation and probabilistic evaluation of individual events allows estimation of both long- and short-term impacts.

The guide offers an opportunity to forecast potential aquatic impacts of highway projects at an early stage of development and to allocate resources for impact mitigation on the basis of need. This advance improves the cost-effectiveness of stormwater runoff management. We have worked with the Washington State Department of Transportation to implement the use of the impact assessment guide in its practices.

SUMMARY AND CONCLUSIONS

Assessment of the impacts of nonpoint sources of water pollution is rather poorly developed, and formidable problems impede its advancement. We believe these problems can be overcome by systematically applying research results to formulate assessment strategies that will yield objective, decision-making criteria. Such strategies are needed in the areas of both data collection and its application to analyze potential impacts. In the former category, we are developing an instrument to identify all of the possible elements of Washington State nonpoint monitoring programs and to

select the priority ones. In previous work we devised a strategy and tools to collect representative stormwater runoff data more cost-effectively than with traditional techniques. Greater standardization of methods is a definite need in the nonpoint source monitoring field. With regard to application of research data to formulate impact assessment protocols, we have developed a guide allowing Washington State highway designers to identify and scope actual environmental problem areas and to apply abatement resources accordingly. This guide reflects concern for cumulative and both short- and long-term impacts to the extent supported by current knowledge. We recommend implementation of similar strategies in other locations and in other nonpoint source problem areas.

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QUANTIFICATION OF NONPOINT SOURCE POLLUTION

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ABSTRACT

Storm runoff pollution can be characterized, in magnitude and in concentration of pollutans, as intermittent and impulsive-type discharges into receiving waters, causing shock loading problems to the ecosystem of these water bodies. The classical approach, using critical low flow as the design criterion for water quality management schemes, must include the effect of storm runoff. This paper presents state-of-the-art methodologies for quantifying storm runoff loads. Four categories of assessment methods are presented: these include zero-order, rational, statistical and simulation methods, for achieving different levels of prediction, i.e. (i) average annual storm load, (ii) storm load per event and (iii) storm load distribution within events.

QUANTIFICATION OF NONPOINT SOURCE POLLUTION

Jy S. Wu, Ph.D., P.E.

INTRODUCTION

In recent years, pollution from nonpoint sources has become an increasingly important consideration in water quality planning and management. Recognizing that point sources are obvious in nature and relatively easy to identify and characterize, the remaining problem is the evaluation of storm-related or nonpoint source pollution. The question naturally arises: what techniques of varying cost, accuracy, and sophistication are available for achieving a successful assessment program ?

Characteristics of storm runoff from various sources have been reported extensively in the literature (McElroy and Bell, 1974; Wu and Ahlert, 1976 ; Novotny and Chester, 1981). In genreal, storm runoff pollution can be characterized, both in magnitude and in concentration of pollutants, as intermittent and impulse-type discharges into receiving water bodies. The classical approach, using critical low flow as the criterion for water quality management, must take the effect of storm runoff into account. This paper presents state-of-the-art methodologies for defining and quantifying storm runoff pollution.

LEVELS OF PREDICTION

Various levels of refinement can be employed to define storm runoff pollution. They range from a relatively simple yearly loading to quite detailed descriptions that introduce temporal variation into the analysis. Three levels of detail can be defined (U.S. EPA, 1976; Wu and Ahlert, 1978a and 1978b).

Level 1: Average Annual Storm Load

This level of detail defines the average annual storm-generated loads as if they occur continuously, during both wet and dry periods. It is used to assess the cumulative long-term effects, e.g. sediment depostion in reservoirs, and may be used to make comparisons with continuous municipal and industrial point-source loadings and describe the relative magnitudes from each source type.

Level 2: Storm Load per Event

This level of detail considers the actual temporal distribution of storm event, including variability of storm loads from one event to the other.

Level 3: Storm Load Distribution Within Events

This level of prediction describes the actual runoff loading rate, as a function of time, within each event. It allows the evaluation of transient and shock-loading impacts on receiving streams.

In general, level 1 is an integral form of level 2 and, in turn, level 2 is an integral form of level 3. These classifications provide useful guidelines for water quality planners to define the necessary levels of stormwater impact anlaysis, and to select appropriate assessment techniques. For example, in characterizing storm loads, an average annual loading rate is useful as a measure of the relative impact of storm runoff. In the case of sediment deposition, the cumulative effect of all storms is more significant than the effect of a single storm or a specific period within a storm. In the case of transient water quality impact analysis, e.g. oxygen depletion of receiving waters during storm conditions, the long-term average loading becomes much less important, because it does not provide enough information to determine how often stream water quality standards are violated.

ASSESSMENT TECHNIQUES

Numerous methods have been used to estimate the magnitudes of storm runoff loads, for both urban and nonurban areas and for long-term as well as short-term predictions. These methods can be categorized as follows.

Zero-Order Method

This is the most crude and least expensive method. Storm runoff loads from an area are estimated from data in the literature. The method is inaccurate in most cases and applicable only to long-term predictions. This is due to the questionable transferability of data from areas of different climate, topography and management practices, etc. The term "zero-order" indicates the relative flexibility of estimation from literature data.

Rational Method

The Rational equation, Q=CIA, for predicting peak runoff rate is known by every water resoucres engineer. The rational type of equation takes the form of a product of several independent variables. If storm runoff and pollutant concentrations are independent, the mean runoff loading rate, w, will equal to the product of a mean concentration, c, and the mean discharge, q :

$$w = q \cdot c \qquad (1)$$

Assuming "q" can be estimated from basic hydrology, "c" can then be calculated from:

(a) published data from a matched watershed;
(b) the computed flow-weighted average concentration, or a representative concentration, obtained over a period of grab sampling (Whipple et al., 1976);
(c) an equation which relates "c" to land use, population density, and street sweeping effectiveness (U.S. EPA, 1976). Street sweeping has been a subject of debate for its effectiveness in reducing nonpoint pollution (Novotny, 1983).

Typical examples of the rational type method include the Univeral Soil Loss Equation (Wischmeimer and Smith, 1965) and its modified form such as the loading function approach (McElroy et al., 1976). The latter approach can be used to predict sediment yield from a watershed, while other water quality parameters are related as fractions of the sediment load.

Statistical Methods

Statistical methods employ techniques such as regression, correlation, frequency analysis, etc., to predict storm runoff loads.

A simple multi-regression equation was employed to predict total pollutant loads for single events (Wells et al., 1971). The variables included in the regression equaiton are rainfall volume, storm duration, time between storms, volume and duration of last rainfall. Heaney et al., (1976) developed correaltion equations for predicting annual average loading rates for each pollutants, as a function of land use, precipitation and population density. This method is proposed as a desk-top procedure for estimating the quality of urban runoff from combined, storm, and unsewered portions of urban areas.

In most cases, it is important to consider high-flow pollutant laoding as a phenomenon separate from low-flow pollutant loading. Paterson (1977) developed the idea of a "concentration matrix" for estimating in-stream storm runoff loading. The matrix is a 4x4 square matrix having 16 blocks. These blocks contain a specific pollutant concentrations representative of the 4 seasons of the year and , also, the 4 flow ranges of low, medium, medium high and high. The representative concentration in each block is obtained by averaging a large number of water quality data (from STORET, for example) in that particular flow range at the corresponding season of the year. For each day, the storm runoff loads are determined by multiplying the daily flow data and the respective concentrations in the matrix. Daily loads can be summed to derive the seasonal or annual loads.

Wauchope and Leonard (1980) employed regression analysis to estimate the maximum edge-of-the-field concentration of pesticides from an individual event. The regression equation relates pesticide concentration to application rate, time of runoff since application and an availability factor describing the pesticide-soil interaction. The accuracy of prediction using regression equations is usually limited by the range of numerical values of the variables originally used for deriving the equations.

Simulation Methods

Simulation methods deal with deterministic rainfall-runoff mechanisms that affect the quality of runoff within a watershed.

Sartor and Boyd (1972) defined dust-and-dirt accumulation on street surfaces as a major source of pollution in urban runoff. Later developments involve the use of their results as basic guidelines for the prediction of

pollutant accumulation rates on street surfaces, prior to a storm event. Accumulated dust-and-dirt is subject to a first-order washoff mechanism by rainwater. Other pollutant concentrations are predicted, as fractions of the total solid loads, as described in the original version of SWMM (Metcalf & Eddy et.al., 1971). Sutherland and McCuen (1974) related the accumulation rate of each quality constituent to land use, pavement type and condition, traffic volume and length of time since rainfall or street sweeping. The rate constant of the first-order accumulation equation was correlated with pavement condition, for single family and muti-family residential areas, and with traffic volume, for industrial and commercial areas.

The use of homogeneous land use approach in conjunction with the concepts of pollutant accumulation and first-order washoff was demonstrated by Wu and Ahlert (1978a). The first-order washoff model is calibrated against the various types of homogeneous or single land-use subareas, to obtain estimates of the two parameters, i.e. initial loading and washoff rate constant. The calibrated model can then be applied to the mixed land-use areas. In this case, the initial loading and the washoff constant are evaluated as composite, weighted values representing the different percentages of land uses for the whole area. Output will be estimates of storm runoff loadings for single events and the time distribution of pollutants within events. This approach minimizes the drawbacks of dust-and-dirt concept and sediment-related loads; model parameters are estimated independently for each quality constituent. However, it requires a relatively intensive sampling effort to obtain a detailed time history of runoff flows and quality, from a number of single land-use subareas and for a number of storms.

Donigian and Crawford (1976) modeled sediment yield from land surfacess including the processes of net daily accumulation of sediment as dust-and-dirt, detachment of particles by raindrop impact on fine sediment and soil, and transport of sediment by overland flow. Other quality parameters are related as fractions of the sediment loads. Wu (1980) modified this approach and assumed the BOD load consists of two parts, a sediment-related part and a soluble part. The soluble BOD load is subject to the first-order washoff by surface runoff. A stormwater assessment model was developed to interface with the SCS TR-20 hydrology program.

A storm runoff model can also be constructed to perform continuous simulation. The use of continuous simulation is to supplement the limited capability of field sampling programs in assessing the watershed responses to hydrologic conditions and management practices, over an extended period of time. One of the difficulties encountered in continuous simulation is the lack of adequate data-base for calibration and verification. In many cases, the collected data contain missing records and thus the use of statistical technique will be needed to generate these missing records (Wu, 1983). Nevertheness, the use of continuous simulation will provide results that can be used to develop probabilistic pollutant loading to assess the cost effectiveness of various storm water management alternatives at an acceptable level of storm runoff load.

CONCLUSION

Various levels of detail and analysis of storm runoff pollution have been highlighted in this paper. However, as new problems will be emerged the assessment methodology will have to be refined. One such example is the uncontrolled discharge of landfill leachate that represents a new horizon of nonpoint source pollution (e.g. see Wu and Hilger, 1984). Future research on nonpoint source pollution should include the development of assessment programs for those nonpoint sources other than the conventional ones such as combined sewer overflow, agricultural or urban runoff etc.

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SUMMARY
Panel II
Legislative Means and Financing of
NPS Control Programs

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INTRODUCTION

The charge of this panel was to investigate the means of legislative treatment for nonpoint source pollution control, to suggest how to treat this with legislation, and to suggest how to finance the control of nonpoint source pollution. The discussions were wide ranging, expressing a broad spectrum of views. The spectrum was so broad, in fact, that distillation of the matters discussed into findings agreed to by all is impossible. Accordingly, this report must necessarily reflect only the ideas that received the most uniform approbation.

FINDINGS

General

1. The panel and the attendees uniformly reaffirmed that legislation is the means to achieving nonpoint source pollution control.

2. There is a not-entirely-understood interrelationship between groundwater protection and nonpoint source pollution control programs.

3. Legislation seems most appropriate on a national or federal basis to avoid regional disadvantages. EPA, however, eschews any national legislative policy, in favor of state and local control of the problem.

4. Substantial social benefits from nonpoint source control measures were perceived. Society would be positively benefited from the creation of incentives to control nonpoint source pollution.

5. Financial incentives for nonpoint source pollution control in voluntary programs are generally inadequate; some form of regulatory imperative is needed.

Legislative Means

6. Legislation requiring nonpoint source pollution control is the preferred means to generate a "community will" to achieve the desired levels of nonpoint source pollution control.

7. Legislation should avoid water quality standards as the medium for achieving nonpoint source pollution control because the benefits of any control program are quite difficult to identify in strictly water quality terms, and enforcement of a violated water quality standard against a particular source of nonpoint pollution is problematic and ineffective.

8. Legislation should either identify best management practices or authorize administrative determination of best management practices that are not related to performance standards. The authorizing legislation should also contain provisions requiring compliance.

9. Efforts should be made to attach nonpoint source pollution control onto legislative efforts aimed at other environmental problems. For example, research money could be garnered from toxic control legislation.

10. Legislation should require affirmative efforts toward nonpoint source pollution control as a condition of other environmental grant programs.

Financing

11. Funding of nonpoint source pollution control is logically part of the public support for a clean, safe and healthy environment. Accordingly, nonpoint source control programs should be attached to other environmental projects well received by the public for funding. For example, erosion control was added to park development on a recent Missouri Constitutional Amendment.

12. Use of sales tax money garnered from food purchases, at rates as low as .1%, can generate needed research and implementation moneys.

13. "No-cost" control programs can be found by merely reviewing present governmental practices. For example, in new residential areas, abandonment of curb and gutter requirements can alleviate storm

water run-off.

A Contrary View

14. A thought-provoking paper was presented which utilized a performance standard predicated upon achieving designated water quality standards, with a responsible group being required to pay taxes when the standards are not achieved and being provided bonuses or rewards when the standards are achieved. Although a majority of the panel and attendees did not concur, the approach illustrates that control of nonpoint source pollution may ultimately be found in creative use of economic reward and penalty.

CONCLUSION

The panel concludes, federal regulation which establishes best management practices is needed. Funding must come by "piggy-backing" nonpoint source pollution control onto other programs and having funds earmarked for nonpoint source pollution.

LEGAL/REGULATORY APPROACHES TO CONTROL OF NON-POINT SOURCES OF GROUNDWATER CONTAMINATION

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ABSTRACT

The U.S. Environmental Protection Agency issued its Groundwater Protection Strategy in August 1984. The strategy outlined the agency's proposed approach to addressing both point and non-point sources of groundwater contamination. Non-point sources receiving attention in the Strategy include leaking petroleum and chemical storage tanks and chemical use, handling, and storage (including pesticides) over certain groundwater formations. The strategy envisions restrictions or bans on such activities or substances particularly over groundwater that is "ecologically vital" or an "irreplaceable source of drinking water."

Because state groundwater quality programs will be required to reflect the provisions of EPA's Strategy to some, as yet undetermined extent, a survey was conducted of existing legal/regulatory approaches to groundwater quality management at the state level of government including programs to control potential non-point sources of groundwater contamination, primarily pesticide and other chemical handling and storage. Few state programs were identified which regulate such sources to a degree commensurate with the EPA Strategy's approach. Significant changes in state legal/regulatory approaches may be required for states to conform to this component of EPA's Strategy.

GROUNDWATER QUALITY MANAGEMENT CONTEXT

Groundwater quality management and protection has been described as one of the top two priorities on the environmental agenda for the 1980's and possibly beyond, the other top priority most often mentioned being acid precipitation control. The importance of the issue is justified by a battery of now-familiar statistics: over 50 percent of the U.S. population depends on groundwater as a drinking water supply, over 80 percent in many rural areas of the country. Groundwater also supplies about 40 percent of the nation's irrigation requirements and approximately 25 percent of the requirements for so-called "self-supplied" industrial purposes.

In addition to these characteristics of groundwater use, characteristics of "typical" groundwater contamination instances also give evidence as to the importance of groundwater quality protection. Groundwater contamination has often been discovered near heavily populated and not coincidentally industrialized areas. The cost of remedial actions to remove, replace, or renovate contaminated groundwater can be measured in the millions of dollars at a single site alone. The Congressional Office of Technology Assessment has sounded a further ominous note in its recent study of groundwater quality management by observing that what we know today about groundwater contamination problems nationally is at least as reflective of how we have discovered or looked for problems and what we have been looking for as it is representative of the major outlines of actual groundwater contamination problems across the country. Almost all interested parties agree that there is at present insufficient groundwater quality monitoring in most locations to truly characterize the nature and extent of groundwater quality contamination problems accurately or comprehensively. As monitoring programs increase and more data becomes available it is more than likely that more contamination problems may be found as well.

U.S. EPA'S GROUNDWATER PROTECTION STRATEGY

In this context of groundwater management needs and problems, the U.S. EPA issued its Groundwater Protection Strategy in August 1984. In it the Agency outlined initiatives in four areas designed to improve groundwater quality protection efforts:

- Strengthen state groundwater agencies and institutions
- Strengthen EPA's internal organization related to groundwater management
- Assess problems from unaddressed sources of contamination
- Issue guidelines for EPA decisions affecting groundwater protection and clean-up to promote consistency in decision making and regulatory requirements.

The subject of this conference, non-point source pollution, was addressed more than once by EPA's Groundwater Strategy. Under the category of unaddressed sources of contamination, the Agency proposed to increase efforts to protect

groundwater from pesticide and nitrate contamination. EPA committed itself to evaluate more precisely the danger of groundwater contamination from pesticide use, to use labelling restrictions to restrict pesticide use over certain types of aquifers, to provide more health advisories and technical assistance to states, and other types of studies and analyses to increase understanding of the problem and its possible solutions.

The guidelines the Agency proposed to increase the consistency of its own groundwater-related decisions are based on the concept of classifying groundwater formations according to their present quality, current or best use, and vulnerability to contamination. Depending on its classification (the EPA system has three classes) groundwaters would be protected to a greater or lesser degree, but all groundwater in the same class would theoretically, at least, be protected to the same degree. The strategy outlined, for example, how hazardous waste facility siting restrictions would differ according to what class of groundwater underlied a particular site. The degree of clean-up required under Superfund could also vary according to what class of groundwater was involved.

Non-point sources were also addressed in this groundwater classification portion of the strategy. The highest class of groundwater in EPA's classification system, "Special Groundwaters", include groundwaters that are "irreplaceable source(s) of drinking water" or "ecologically vital." The Agency proposes to look into developing additional restrictions on the use, disposal, or storage of potentially threatening chemicals over this class of groundwater. This could potentially include pesticides, fertilizers, road salts, storage tanks of various types, and other substances typically included on the roster of non-point sources of pollution.

While the groundwater classification guidelines were proposed to apply to EPA's own decision-making they have implications for states as well. States with EPA-delegated authorities in groundwater-related program areas (such as RCRA, Superfund, Safe Drinking Water, etc.) will be required to show that their programs are no less strigent than Federal requirements in those areas which will result from implementation of the Agency's groundwater strategy. EPA has said it will keep regulatory requirements for states general and performance-oriented to the extent possible in these areas.

The strategy, particularly in its groundwater classification components, has generated great interest, particularly among some segments of the regulated community, as to how states regulate groundwater quality now and how they might be required to change their approach to conform to EPA's Strategy. The General Accounting Office, the Office of Technology Assessment the Environmental Law Institute, and the American Petroleum Institute have all completed studies on state groundwater protection programs. EPA's Office of Groundwater has a similar study scheduled for completion in Spring 1985. Various task forces and commissions have been formed to address how best to protect groundwater quality, and national groundwater protection legislation is an almost sure topic for congressional debate in 1985.

A groundwater quality management study recently completed by the author compared existing state groundwater programs with the provisions of EPA's Groundwater Strategy. The purpose of the study was to identify the nature and

extent of changes that might be required of states in order to bring their programs in line with the Strategy. One of the topics of this study was the non-point source-related provisions of the strategy mentioned earlier in this paper. Framed as a research question, this portion of the study asked: Do any states presently regulate pesticide, fertilizer, or other chemical use, handling, storage, or disposal on an aquifer specific or groundwater-specific basis as proposed in EPA's Strategy? Legislation, regulations, policy statements and related program documents from all fifty states were reviewed and environmental agencies in all states were contacted for answers to this question. Some example approaches and general themes from the state-by-state regional analyses are presented in the next section.

STATE LEGAL/REGULATORY PROGRAMS

With regard to non-point source regulation, most states have not gone beyond the requirement of best management practices (BMPs) as part of their general water quality programs. As has been pointed out by EPA itself recently, BMPs designed primarily for the protection of surface water quality may actually have resulted in increased degradation of groundwater quality, e.g., percolation "catch basins" for stormwater runoff which may result in concentrated pollutant streams reaching underlying aquifers.

Many, if not most state agency contacts consulted in the course of the study reported the perceived political impossibility of generating necessary levels of support in their state legislatures for any regulation of non-point sources of groundwater contamination beyond BMPs. Agency contacts offered this assessment while at the same time reporting that their existing statutory authorities probably provided a sufficient basis for developing and implementing such programs.

Some states with otherwise comprehensive and inclusive programs to regulate point sources of potential groundwater contamination, explicitly exclude non-point sources from their regulatory requirements. Montana, as just one example, is a state which employs a classification system similar to that proposed by EPA's Strategy and requires permits for most point source discharges to groundwater: a combination of management approaches employed by well less than half of all the states. The state has a relatively strict non-degradation policy for groundwater quality, yet expressly excludes from degradation restrictions changes in groundwater quality due to non-point sources "where all reasonable land, soil and water conservation practices have been applied."

This is reflective of the comments received from most state agency contacts, that given the present state of knowledge with regard to groundwater contamination dynamics and the expected high costs of thoroughly and positively regulating diffuse non-point sources, recommended management practices and guidelines are the only feasible management option for non-point sources capable of being implemented on a broad scale. Some states and localities have begun to address at least in part the non-point concerns of chemical storage use, and handling as outlined in the EPA's Groundwater Strategy. Approximately fifteen states have enacted specific regulatory programs to control leaks from above and below ground tanks for storing petroleum products, chemical feedstocks, and other substances. Over half of

the states with such programs are in the Northeast region of the country. Most states have general authorities to respond to tank leaks and require clean-up. State agency contacts report that these authorities generally do not provide a sufficient basis for an effective, preventive, regulatory program. Some existing tank leak prevention programs at the state and local level consider the hydrogeological formation over which a tank will be sited; more often than not this is not an explicit consideration. Many agency contacts in states without tank regulation programs have reported that they are waiting to see what requirements will be forthcoming under EPA's new tank authorities from the 1984 RCRA amendments before proceeding further on program development of the state level.

Apart from storage tank programs, few states have taken steps to restrict chemical use, handling, or storage, over certain groundwater formations which could be susceptible to contamination from such sources. Some states' major facility siting laws perhaps come the closest to embodying the types of authorities and restrictions apparently envisioned for pesticide and other chemical handling and storage (at least for Class I aquifers) by EPA's Strategy. Even states which currently classify aquifers or groundwater include restrictions only on certain point sources, such as hazardous waste landfills, and even these are allowed to locate over aquifers if certain design and operating conditions are met. Seldom if ever in such state programs are non-point sources singled out or included for regulatory attention.

New Jersey is an example of a state which has explicity addressed chemical handling and storage as a possible source of groundwater contamination in its water quality regulations. Major chemical storage facilities (greater than 400,000 gallons capacity) must provide for secondary containment structures lined with impermeable materials wherever hazardous substances are handled. Leaks must be able to be detected by inventory controls, automatic leak detection devices or groundwater monitoring wells. Regulations are applied to above and below ground storage of petroleum and other hazardous substances, drum storage areas, transmission pipelines and in-plant piping.

In Massachusetts, many local communities have enacted recharge area zoning ordinances which regulate and restrict the handling, storage, use, etc. of chemicals over particular aquifers and types of hydrogeologic formations. The state also has an aquifer/land acquisition program through which localities are assisted to purchase and thereby remove or prevent all threatening land uses from property overlying their local groundwater supplies.

In Virginia, a county government has created a special Natural Resource Conservation Overlay zone in the recharge area of one of its important drinking water aquifers. This special zoning legislation is directly aimed at restricting the use of agricultural chemicals in the recharge area to protect groundwater quality

CONCLUSION

These are a few examples of initiatives at the state and local level to control one category of non-point sources with the potential to affect groundwater quality: chemical use, handling and storage. These approaches

are neither typical nor wide-spread among the states. The most prevalent approach to non-point source pollution of groundwater continues to be recommended practices and guidelines. Many states have requirements that pesticides be registered for specific uses and that applicators be certified. But few of these programs consider underlying groundwater explicitly as is proposed in EPA's Groundwater Protection Strategy.

The EPA Strategy's proposed initiatives relating to pesticide and other chemical handling, use, and storage has the least number of analogies among state programs of any of the initiatives proposed in the Strategy. The number of local government programs of this type is not surprising, since what is being proposed in the Strategy is a variety of restrictive zoning authority usually associated with traditional local government powers. It appears likely that any initiation of even a limited regulatory program under this component of EPA's Strategy will require significant changes in existing state approaches to protection of groundwater quality from these and other non-point sources of pollutants.

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POSITION PAPER

URBAN WISCONSIN SHOULD JOIN THE FIGHT AGAINST NONPOINT POLLUTION

Kathleen M. Falk - Wisconsin Public Intervenor

The Wisconsin Public Intervenor, an Assistant Attorney General in the Wisconsin Department of Justice, is expressly charged by statute with the duty of intervening where necessary to protect natural resources in the State of Wisconsin. The need for protection of these natural resources from pollution through nonpoint sources is now at a critical stage, yet there are still no comprehensive programs to either prevent or clean-up nonpoint pollution. Urban Wisconsin must do its part to ensure that the future will be different from the past by regulating urban nonpoint pollution. My intention is to summarize some of my observations as to why there are still no comprehensive programs to prevent or clean up nonpoint pollution and why legislation must be enacted to change this. Legislators must recognize that one of the most promising ways to prevent future nonpoint pollution is through the control of urban sprawl.

In the early 1970's, a rash of fundamental changes occurred in federal and state environmental laws, turning around the way the country was polluting its resources. The immediate focus of the new laws was on pollution that one observed spewing out of a smokestack into the air, or pouring out of an industrial pipe into a river. Very little attention and effort was directed at nonpoint pollution.

We are at the point, historically, where most agree that tremendous advances have occurred in meeting many of the goals and mandates contained in the 1970's laws to clean up our state and nation from point source pollution. It is imperative that we now turn our political and technical skills forward to solving the problem we let lie underneath the rug for the past decade. Instead of cities simply pointing their fingers at pollution caused by their rural neighbors, and rural Wisconsin citizens pointing their fingers at the city culprits of nonpoint pollution, and instead of debating who causes more pollution and whose pollution is the worse, it is time to join together to remedy and prevent future nonpoint pollution. Of the many sources of nonpoint pollution -- agriculture, urbanization and transportation, to name a few -- one specific cause of nonpoint pollution gets too little attention: urban sprawl.

In a broadly historical sense, urban sprawl is a relatively new problem. After World War II, suburban development was considered a goal, almost an ethic; paving over our countryside was equated with "progress" and "growth." Numerous government programs subsidized suburban sprawl - many still do.

Urban sprawl causes enormous environmental and economic problems for Wisconsin. One of the major environmental harms is

urban runoff, one kind of nonpoint pollution. A recent study of the Menominee River Watershed reveals that approximately 75% of the total pollutant loadings in the river comes from urban nonpoint pollution. Urban runoff contains almost every type of pollutant: suspended solids and toxic substances, particularly heavy metals, bacteria, nutrients, asbestos, oil and grease and other pollutants. One study in Washington, D.C., found that, in water from streets, the concentration of suspended solids was 104 times higher than effluent from a secondary sewage treatment plant, and the lead concentration was 1,015 times higher. (President's Council on Environmental Quality, 1980.)

A recent study by Bauman, Domanik and Konrad, 1980, identifies the two major types of nonpoint pollutants generated by urban land use: 1) nutrients and sediments, and 2) toxic materials.

Urban sprawl is a major nonpoint source of these two kinds of pollutants because of the widespread land development, increased traffic density and deficient wastewater treatment facilities that are associated with it.

The magnitude of the nonpoint pollution generated by new land development is reflected in the following statistic on the Menominee River Watershed, taken from the Bauman, Domanik and Konrad study. Developing urban areas represented only 2.6% of the total area of the watershed, but contributed 37% and 48%, respectively, of the suspended solids and total phosphorous. Subdivision construction was cited as a major culprit in the generation of nutrient and sediment pollutants.

To further underscore the excessive pollutant amounts contributed by land under development, it is helpful to compare it with industrial nonpoint pollution. In 1978, industry put an estimated 5,100 kg/ha/yr of suspended solids into the Menominee River Watershed. Land under development was the source of 43,700 kg/ha/yr. That means new urban development, such as that which makes up urban sprawl, generates roughly eight times more suspended solid pollutants than industry (Bauman, Domanik and Konrad, 1980). The need for regulation of development is apparent.

Another by-product of urban sprawl is an increase in nonpoint pollution generated by transportation. Greater commuting distances traveled by each car will influence the amount of pollutants emitted into the air.

Automobiles are the primary source of airborne lead in the environment (Carberry, 1980). As a toxic, lead is one of the two major types of urban nonpoint source pollution. A recent compilation of studies on the source of lead and its transport in urban run-off indicates that emission of lead into the air by automobiles leads to fall-out of the lead particule in rainwater and soil. One study measured lead concentrations in rainwater in

Pittsburgh, Pennsylvania, and found average concentrations of 41 ppb for urban samples. This was compared with 19 ppb for rural samples. The maximum allowable lead concentration in drinking water at the time of this study was set by the EPA at 50 ppb. The conclusion drawn by J.B. Carberry, the compiler of the studies, was that "rainwater analysis indicated that urban regions were more highly contaminated from automotive exhausts than rural regions." To increase the emission of airborne lead as a by-product of urban sprawl is not only senseless, it is a threat to health. In man, accumulation of lead is faulted with hemoglobin interference and kidney function impairment. The increase in traffic density caused by commuters from suburban areas can only add to the nonpoint pollutants already threatening the environment and human health.

Along with the sprawl of urban development into former rural areas comes the need for wastewater treatment facilities. The cost of wastewater treatment facilities borne by each household and subsidized through grants is much greater for these new facilities than for those in established urban areas. Regardless of these prohibitive costs, facilities must be built so that adequate treatment of waste can be achieved before it is discharged into the environment.

An example of the cost discrepancy between the old and the new facilities is the following statistic from a study of wastewater treatment costs done by James A. Hanlon (1980). Wastewater treatment costs in Greater Chicago, Illinois, were an estimated $45.00 per household/year in 1980. The treatment costs for the Village of Elwood, Illinois, with a population of 783 were an estimated $215 per household/year. Urban Wisconsin can no longer afford to allow new sprawling developments to consume private and public monetary resources in addition to compounding the nonpoint pollution problem.

The state can prevent multiple ills by preventing urban sprawl. Wisconsin should get to work by beginning to get its own house in order. Each day state agencies make many, many decisions which actually cause, not prevent, urban sprawl and nonpoint pollution. These agencies should redirect their efforts and prevent suburban development and control this major source of nonpoint pollution. Urban areas should start reducing nonpoint pollution by regulating urban sprawl. Legislation is long overdue.

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ECONOMIC INCENTIVES FOR CONTROL OF NONPOINT POLLUTION

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ABSTRACT

Attempts to control agricultural nonpoint pollution (NPP) have focused on the strong encouragement of the use of "best management practices (BMPs)", with cost sharing used as an inducement to participate. However, few farmers have chosen to implement approved BMPs, and it appears that these voluntary programs are unlikely to result in levels of participation that are sufficient to combat NPP adequately. In addition, imposing mandatory practices to reduce NPP is unlikely to result in the use of least cost abatement techniques since the effectiveness of alternative techniques will vary across farms.

An alternative to the use of mandatory farming practices is the use of economic incentives to induce pollution abatement. This paper suggests one possible incentive mechanism that could be used when the suspected sources of the pollutants can be identified (e.g., a group of farms bordering a waterway) but the polluting activity (e.g., pesticide application or runoff) cannot be directly observed and/or monitored. The general mechanism combines a system of rewards for water quality above a given standard with a system of penalties for sub-standard water. Both the advantages and disadvantages of this incentive scheme are discussed.

Although the discussion is in the context of agricultural pollution, a similar incentive mechanism could also be applied in cases of non-agricultural NPP of either surface or ground water.

Keywords: Nonpoint pollution, pollution abatement, agricultural runoff, economic incentives, pollution taxes.

INTRODUCTION

Twelve years ago the U.S. Congress passed the Federal Water Pollution Control Act Amendments (P.L. 92-500), which were intended, among other things, to address and control nonpoint water pollution. Yet, in its assessment of the success of existing pollution control programs, the General Accounting Office (1982) identified NPP as one area in which little progress could be claimed. Since agricultural runoff is one of the main sources of NPP, the problem is particularly acute in the largely agricultural North Central Region.

Attempts to control agricultural NPP have focused on the strong encouragement of the use of "best management practices (BMPs)," with federal or state cost-sharing used as an inducement to participate. However, few farmers have chosen to implement approved BMPs. For example, in Wisconsin, the Department of Natural Resources (DNR) set a goal of 70% participation after two years by landowners in priority management areas, while actual participation rates have ranged only between 11% and 38% (DNR, 1984).

There are at least two possible reasons for low participation rates. The first is that, even with cost-sharing, individual farmers might expect that the costs to them of implementing BMPs are not justified on the basis of their expected returns. For example, if the benefits of the use of BMPs result primarily from reduced downstream sedimentation and pollution rather than from increased productivity or reduced input costs, these benefits will not accrue directly to the farmer and it is unlikely that farmers will be willing to bear the associated costs voluntarily.

A second possible reason for low participation is that farmers tend to be very independent and object to anything they consider to be an infringement on their freedom to operate their farms as they choose. Since participation involves a commitment to certain operating practices, it limits a farmer's freedom to operate independently.

Given these disincentives to participation, it is not surprising that the U.S. Environmental Protection Agency (EPA) should conclude that "In such cases, sole reliance on voluntary programs is not likely to accomplish adequate reductions in pollutant loads and, as a result, other approaches may be needed (e.g., economic incentives or regulation)." (EPA, 1984, p. xv). The most likely form of regulation to reduce NPP would be some form of mandatory use of BMPs or other farming practices. This approach has two drawbacks. First, it does not in general provide sufficient flexibility to control NPP in a site and source-specific manner that guarantees the most efficient use of available funds. For practical reasons, direct regulation or control is generally applied across-the-board, without regard to differences between polluters in terms of either costs of abatement or damages imposed. In the case of NPP, since the effectiveness of alternative BMPs depends on conditions that vary across farms (i.e., any individual BMP is not necessarily "best" for all farms), mandating the use of these practices across-the-board is unlikely to result in use of least-cost abatement techniques. In other words, resources will be wasted unless site-specific controls can be imposed, since farmers would be forced to use practices that are inappropriate for their specific conditions.

Direct regulation has a second potential drawback as well, namely the difficulty of monitoring to ensure compliance with the regulations. Although the use of some practices (such as terracing) can be easily monitored, others (such as a modification of fertilizer or pesticide application techniques to reduce runoff) would be difficult to monitor on individual farms. Since farmers would often have little incentive to comply with regulations, voluntary compliance cannot be assured. Thus, in the absence of substantial enforcement efforts, the effectiveness of direct regulation may be further reduced.

As suggested by EPA, an alternative to direct regulation is the use of economic incentives to induce participation. This paper describes one possible incentive mechanism that could be used under certain conditions to reduce NPP. This mechanism combines a system of rewards for water quality above a given standard with a system of penalties for sub-standard water. The possible use of this incentive scheme and its advantages and disadvantages are the subject of this paper.

IMPORTANT CHARACTERISTICS OF NPP

An important characteristic of NPP that makes the standard solutions that have been successful in controlling point source problems unworkable for NPP is that, although the likely polluters can often be identified, it is generally not possible to identify a one-to-one relationship between the level of abatement or discharge and the damages from pollutants in the water system. The reason for this is twofold: (1) given any level of abatement, the resulting water quality effects are uncertain due to the contributing effects of stochastic variables, and (2) the level of abatement or discharge often cannot be directly monitored by the overseeing authority without excessive costs.

More specifically, the pollutant loadings that result from any given operating practice depend on a number of climatic and topographic conditions in a manner that cannot be predicted with certainty. For example, as noted by the EPA,

> The timing (intermittent discharge caused by rain or snow), concentration, and dilution of the pollution from nonpoint sources constitute only part of the picture when one considers the nature of associated water quality impacts; the transportation and ultimate fate of the pollutant constitute the other part.
>
> . . .The potential dilution of pollutants during high flow must be considered against the velocity with which pollutants are dislodged and transported. Thus, it is difficult to make generalizations about the concentration of loads from nonpoint sources. (EPA, 1984, p. 1-8)

The result is that associated with any given abatement practice or discharge level at any given time is a range of possible loadings for each pollutant. (More generally, there is a range of possible damages in terms of the impacts

on human health and welfare that depend not only on loadings but also on factors such as stream flow and exposure risks. Although conceptually the analysis could be applied to this broader range of impacts, for simplicity we focus here only on the range of possible loadings.) This range can be represented by a probability density function (p.d.f.) that is conditional on the abatement practice. The p.d.f. gives the probability that loadings of a given magnitude will occur at the specified time, where the probability depends on the abatement practices being used.

The objective of policies to reduce NPP is then to shift the distribution represented by the p.d.f. to the left, as illustrated in Figure 1, to increase the probability that actual loadings will fall below some tolerance level.

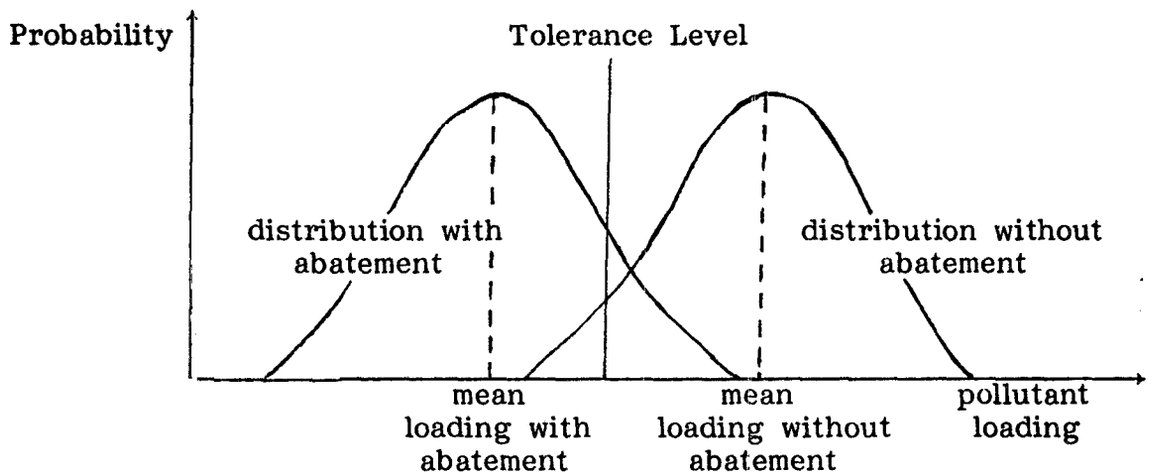

Figure 1: Distribution of Pollutant Loading With and Without Abatement

If direct monitoring of all farm operations were economically feasible or voluntary compliance with regulations were guaranteed, then the distribution could be shifted through mandatory abatement practices. However, as noted above, achieving this at least cost would in general require mandating site-specific practices. In addition, monitoring certain farm activities such as irrigation and pesticide or fertilizer use on a continual basis is generally impractical, and voluntary compliance with restrictions on use in the absence of monitoring is unlikely. Thus, the use of direct regulation to try to shift the distribution seems inappropriate; the use of a mechanism that provides an incentive (either positive or negative) for compliance is likely to be more effective.

In addition to shifting the p.d.f. of pollutant loadings, incentives can also result in risk-sharing. The existence of a distribution of possible loadings implies that there are certain risks associated with any given level of abatement. For example, if the actual level of abatement is set so that expected marginal costs equal expected marginal benefits, there is the risk that mitigating factors will be favorable and therefore that this level of abatement will have been higher than that necessary to achieve a target level of environmental quality, i.e., actual loadings will be even less than expected.

Conversely, under adverse conditions this level of abatement could turn out to be insufficient to meet the environmental quality goal. To date, U.S. pollution control policy has in general been structured so that polluters bear none of this risk associated with a given level of abatement and society bears the full risk. Since abatement requirements do not depend on the realized environmental conditions, i.e. the realized damages, the cost to the polluter is the same regardless of those actual conditions. His payoff from pollution abatement is nonstochastic. However, society's payoff in terms of reduced damages is a random variable depending on the realized environmental conditions. If mitigating factors are favorable, society will "win" the gamble in the sense that the actual damages will be lower than expected. Likewise adverse factors imply that society "loses" in that it suffers damages that are greater than expected. If society is adverse to taking risks, it can increase its expected well-being by shifting some of the risk to the polluters through the use of policy instruments that result in risk-sharing. This could benefit polluters as well if sharing risks induced victims to be content with more lenient target levels.

The use of incentive mechanisms and risk sharing to induce desired behavior has been studied by many previous authors, e.g. Stiglitz (1974), Holmstrom (1979), Shavell (1979), and Mookherjer (1984), in the context of what has come to be known in the economics literature as the "principal-agent" problem. A principal-agent problem exists when the welfare of one party, called the principal, depends directly on actions taken by another party, called the agent. The principal's challenge is to devise a payment schedule for the agent to induce the agent to take those actions that best serve the interests of the principal. However, the principal is unable to control the agent's actions directly and cannot even observe them. He can only observe his payoff that results from the agent's actions, which depends not only on those actions but also on stochastic conditions or events. The following question is then asked: what form of incentive mechanism should the principal use to induce the agent to take those actions that best serve the interests of the principal? The question can also be asked in the context of a more general model where there are many agents instead of just one.

If we interpret society as the principal and the farms generating NPP as the agents, the principal-agent problem describes the challenge faced by society to induce independent farmers to take steps that will contribute to improved water quality and thereby serve the interests of society. Viewing NPP in this way allows the insights that have been gained for solving general principal-agent problems to be applied in solving problems of NPP.

AN INCENTIVE MECHANISM

Single Polluter Problem

In devising an appropriate incentive scheme, the first point to note is that with regard to NPP we are interested in a reduction of loadings (or, more generally, the damages that result from them) rather than a reduction in, for example, soil erosion per se. This is emphasized by the EPA (1984), which notes that lands with high erosion rates are not necessarily those causing significant NPP problems, and vice versa. Thus, incentives designed solely to

reduce soil loss (such as soil loss taxes) are inappropriate for control of NPP. Instead, an incentive scheme designed to shift the distribution of loadings is needed.

Consider first the case of a single suspected polluter, e.g. a single farm whose land drains into a nearby stream. Let x be the level of actual loadings of a given pollutant in the stream, and let $\bar{x}$ be a specified target or tolerance level of loadings, which is set by authorities and could, for example, be adjusted seasonally. Actual loadings x will depend upon both the abatement actions taken by the polluter (e.g., the use of various BMPs) and random variables reflecting unpredictable weather and stream conditions, as illustrated in Figure 1.

A general incentive scheme designed to shift the distribution of actual loadings would take the form of automatic, required payments T(x) that depend upon the level of actual loadings as compared to the target level $\bar{x}$ and are given by

$$T(x) = \begin{cases} t(x - \bar{x}) + k & \text{if } x \geq \bar{x} \\ t(x - \bar{x}) & \text{if } x < \bar{x} \end{cases},$$

where t and k are constants that can be set by the regulating authority to ensure that the payment scheme provides the incentive necessary to induce the polluter to undertake the level of abatement that is deemed socially desirable. The optimal values of t and k will depend upon marginal abatement costs and benefits as well as how increased abatement shifts the distribution of loadings. (See Appendix A for details.)

The payment scheme is composed of two parts. The first, reflected by t, is a tax/subsidy payment that depends upon the extent to which x differs from $\bar{x}$. If actual loadings exceed the target level, the polluter pays a tax proportional to that excess, while actual loadings below the target level result in a subsidy or credit to the suspected polluter. Note that actual loadings may differ from target levels because of either the abatement actions of the polluter or the influence of the random variables. Thus, the polluter may be liable for tax payments that result from influences outside his control. Likewise, however, his liability may be reduced (and he may even receive subsidies if x falls below $\bar{x}$) due to favorable environmental conditions even if he has taken no action to control NPP. Thus, in choosing his level of abatement, he gambles on what his actual tax liability will be and weighs the additional cost of pollution abatement against the decrease in expected payments that results from increased abatement.

The same type of incentive is provided by the second component of the payment scheme, reflected in k, which is a fixed penalty imposed whenever loadings exceed the target. The amount of the penalty is independent of the extent to which the target is exceeded. Again, the polluters can weigh the cost of abatement against the decrease in the probability that x will exceed $\bar{x}$, i.e., that he will incur the penalty, that results from increased abatement. Note that the effect of this penalty scheme is different from that of penalties

applied to actions (or inactions) that are directly under the control of the polluter (e.g., penalties for point emissions in violation of standards). In the stochastic case, additional abatement will in general always decrease the expected penalty by decreasing the probability that x will exceed $\bar{x}$, whereas under penalties for emissions in excess of standards incentives exist to reduce emissions to the standard level but not below.

Either component of the incentive mechanism can be used by itself to induce a desired level of abatement, or they can be used in combination as given above. (See Appendix A.) Since a comparison of these alternatives is more relevant when there are many polluters, they are discussed below in that context.

Multiple Polluters Problem

In many cases of NPP, it is likely that several polluters, e.g., several farms, will be suspected of contributing to the loadings of a given waterway. An incentive scheme similar to the one introduced above can still be used, if t and k are allowed to vary across polluters, i.e., if tax payments of polluter i are given by

$$T_i(x) = \begin{cases} t_i(x-\bar{x}) + k_i & \text{if } x \geq \bar{x} \\ t_i(x-\bar{x}) & \text{if } x < \bar{x} \end{cases}$$

Again, t_i and k_i can be set to ensure optimal levels of abatement by each source.[1] (See Appendix A.) Polluters with high marginal abatement costs or whose abatement is likely to have a relatively small effect on the distribution of loadings will tend to have lower optimal payments than other polluters. Note, however, that each polluter's liability depends on loadings from the whole group, not just his individual contribution, since at any given time individual contributions are not known or observable.

As in the case of a single polluter, possible forms of the incentive scheme include (1) a pure tax/subsidy scheme where $k_i = 0$ and $t_i > 0$ for all i, (2) a pure penalty scheme where $t_i = 0$ and $k_i > 0$ for all i, and (3) a combined scheme where $t_i > 0$ and $k_i > 0$ for all i. Although the optimal abatement level can be induced using any one of the three alternatives, their implications in terms of total polluter/government payments are clearly different. For example, under the pure tax/subsidy scheme, combined subsidy payments to all polluters when $x < \bar{x}$ could far exceed the benefits of the reduced loadings, since each polluter would in some sense get credit for the entire reduction in loadings. In addition, it provides no way to reward or compensate polluters who abate "more than their share" and thereby create additional benefits for all other polluters. A pure penalty scheme has the advantage of requiring no government outlays for low loadings, but also suffers from the inability to compensate "good" polluters. The combined scheme avoids some of these problems. Under this scheme the t_i values are not constrained (see Appendix A) and thus can be chosen so that the sum of subsidy payments when $x < \bar{x}$ does not exceed the benefit of the reduced loadings. Although this still requires government outlays when $x < \bar{x}$, those outlays can theoretically be set as low as desired (as long as the k_i are adjusted to maintain proper incentives). In addition to choosing the sum of the t_i to avoid excessive

outlays, the regulatory authority can also set the individual t_i's to reward "good" polluters. Thus, the combined scheme allows the distributional effects of the incentive mechanism to be adjusted to satisfy other, non-efficiency objectives.

ADVANTAGES AND DISADVANTAGES

Regardless of which form of the incentive mechanism is chosen, the use of such a mechanism has several advantages.

First, it involves a minimum amount of government interference in daily farm operations, and farmers are free to choose the pollution abatement techniques that are least cost for their farms. Since individual farmers are in a better position to determine the abatement practices that will be most effective for their land (and will have an incentive to do so), their freedom to choose the techniques used provides the flexibility necessary to ensure that any given level of abatement is achieved at the lowest possible cost.

Secondly, once in place, the incentive mechanism can be easily administered since it does not require continual monitoring of farm practices or metering of runoff or soil loss. It instead requires that the regulatory authority monitor pollutant loadings regularly and calculate the necessary tax or subsidy payment. Accounts can be cumulated over time with payments made periodically. If, over the time period, tax liability exceeds subsidy payments, then no government outlays would be necessary even under the pure tax/subsidy or combined approaches. The subsidies would simply act as credits against tax liability.

Thirdly, the cost-sharing mechanisms of the existing NPP programs can be maintained to prevent placing excessive burdens on the agricultural sector, and other considerations regarding an appropriate distribution of costs can be accommodated, as long as the parameters of the payment scheme are adjusted accordingly to maintain proper incentives. As mentioned above, under the combined approach the t_i values can be chosen to reflect distributional considerations. In addition, federal or state cost-sharing to cover a portion of investment or operating expenses is consistent with use of the incentive scheme. The payment mechanism simply provides the incentive for participating in cost-sharing programs that seems to be missing under the current structure.

A fourth advantage of the incentive scheme is that it focuses on water quality rather than erosion or runoff, which is more appropriate for controlling NPP. In addition, to the extent that some of the fluctuations in pollutant loadings can be anticipated, there would be an incentive for farmers to try to offset peaks by, for example, avoiding heavy pesticide or fertilizer applications prior to anticipated rain or wind storms.

Finally, these incentives allow the risks associated with given abatement levels to be shared by victims and polluters alike, which differs from the typical approach to environmental policy in the U.S. under which only victims are forced to gamble on whether given abatement levels are sufficient. This could benefit both polluters and victims if sharing risks induced victims to be content with more lenient target levels.

The disadvantages of this incentive scheme include the information requirements that are necessary to set the levels of the t_i and k_i parameters initially to provide the correct incentive. (In general, this is a problem with any regulatory device seeking to achieve socially optimal outcomes.) The necessary information includes abatement cost estimates, estimates of damages from pollutant loadings, and estimates of how each polluter's abatement affects the distribution of those loadings, which would require the use of individual watershed models.

A second possible disadvantage of the mechanism is its implications with regard to distortionary taxation. It would have to be structured so that allowing the t_i and k_i parameters to vary across sources would not be considered to be distortionary taxation.

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APPENDIX A

This appendix describes how the parameters of the incentive scheme described in the text could be set to ensure proper abatement incentives. The scheme is similar to one suggested by Holmstrom (1982) in the context of optimal organizational structure.

Let the random variable $x(a)$ be actual loadings of a given pollutant which depends on the level of abatement of a single polluter, denoted a. Let $C(a)$ be the cost of abating to level a, and let $F(\bar{x},a)$ be the probability that $x(a)$ is less than a target level $\bar{x}$, given a. Finally, let $B(x(o)-x(a))$ be the benefit of increasing abatement from a base zero level to a. Then society seeks the level of abatement that maximizes $E[B(x(o)-x(a))] - C(a)$, where E is the expectation operator over the random variable x. The optimal level of abatement, denoted a^*, is implicitly defined by the first order condition $E[B' \cdot dx/da] + dC/da = 0$.

Given the payment scheme $T(x)$ in the text, the expected value of those payments, $E[T(x)]$, is given by $t \cdot E[x(a)] - t\bar{x} + k\,(1-F(\bar{x},a))$. The polluter thus chooses the level of abatement that minimizes $E[T(x)] + C(a)$. His choice, $\bar{a}$, is implicitly defined by $t \cdot E(dx/da) - k(\partial F/\partial a) + dC/da = 0$. A comparison of the optimality conditions for society and the polluter implies that the polluter will be induced to choose the level of abatement that is optimal for society, i.e., $\bar{a} = a^*$, if:

(1) $k = 0$ and $t = -(dC/da)/E[dx/da]$ evaluated at a^*,
(2) $t = 0$ and $k = (dC/da)/(\partial F/\partial a)$ evaluated at a^*, or
(3) t is arbitrary and $k = (dC/da + tE[dx/da])/(\partial F/\partial a)$ evaluated at a^*.

Note that under the pure tax/subsidy case (case 1), the optimal tax rate t is equal to marginal benefits B' if B' is a constant. Under a nonlinear benefit function, $t \neq E(B')$. However, $E(B')$ may be a sufficient local approximation to the optimal t, or serve as a guide in setting t.

If there are multiple polluters, let a_i be the abatement level of polluter i, let $C_i(a_i)$ be i's abatement cost, and interpret a in $x(a)$ and $F(\bar{x},a)$ as the vector $(a_1,\ldots,a_n)$ (where n is the number of polluters). Society then seeks the vector $a^* = (a_1^*,\ldots,a_n^*)$ that maximizes $E[B(x(o) - x(a))] - [C_1(a_1) + \ldots + C_n(a_n)]$. Individual polluters choose a_i to minimize $E[T_i(x)] + C_i(a_i)$ given a set of expectations about the actions of other polluters. An analysis similar to that for the single polluter case indicates that, in equilibrium where all expectations are realized, the possible optimal incentive schemes are given by:

(1) $k_i = 0$ and $t_i = -(dC_i/da_i)/E(\partial x/\partial a_i]$ evaluated at a^*,
(2) $t_i = 0$ and $k_i = (dC_i/da_i)/(\partial F/\partial a_i)$ evaluated at a^*, or
(3) t_i is arbitrary and $k_i = (dC_i/da_i + t_iE[\partial x/\partial a_i])/(\partial F/\partial a_i)$ evaluated at a^*.

LEGISLATION, NOT LITIGATION, IS THE SOLUTION TO NON POINT SOURCE POLLUTION

by

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PREMISE

Heightened recognition by society of problems arising from non-point source pollution have raised an increasingly large voice towards controlling such discharges. Often times, however, those adversely affected by non-point source pollution do not have available the tools or knowledge of the problem sufficient to obtain affirmative relief. Indeed, it is the premise of this paper that traditional legal remedies cannot adequately be relied upon to remedy non-point source pollution. Thus, it is important that society's interests be protected by the legislature, the source of our ordered form of government.

INTRODUCTION

Recourse to legislation is particularly apropos in dealing with non-point source pollution if for no other reason than non-point source pollution does not respect traditional boundaries. From any source of pollution, the effects pass on through the watershed. The resulting pollutant path, the pollution shed, effects many jurisdictions, transcending municipal and county lines and can, in the many cases, cross state lines providing a federal cast to the problem. Many of the traditional remedies provided through the courts are frustrated by this concept of pollution shed, and where not frustrated, the remedies are rendered so expensive that only the most well-heeled of litigants can realistically hope to finance a favorable result.

A further reason for recourse to legislation is the critical need to factor economic and societal values into the calculus of environmental rights. This analysis must of necessity require society's judgments of the future and the cost society is willing to bear today to ensure a sound environment in the future. As will be seen, this is particularly true when the need for a solution of non-point source pollution points toward a change in more or less traditonal uses of water resources to remove and pass along non-point source pollutants.

I.

TORT REMEDIES ARE INADEQUATE

The use of litigation as a vehicle to obtain relief from damaging non-point source pollution is the most familiar vehicle for people to look to for relief and yet at the same time it is probably the least adequate way to obtain the relief necessary. This inadequacy is a necessary component of the legal system from which the relief must be generated. Relief would be based upon a theory of tort violation; that is, an injury to a protected personal right caused by another's acting in an unreasonable fashion.

Two aspects of traditional tort litigation prove to be the most frustrating in the pursuit of relief from non-point source pollution: the need to establish a discharger's act was the cause of damage and that damage in fact occurred. These limitations in traditional tort litigation create significant problems with the allocation of responsibility for it is the plaintiffs burden in any litigation to identify a defendant whose conduct was the "but for" or "substantial factor" cause of the harm done to the plaintiff person injured.

With non-point source pollution, identifying a "but for" cause of damage is not easy. Perhaps a few statistics will illustrate the point. In the United States there is enough animal waste produced annually as would equal the waste generated by two billion people, nearly ten times the number of people currently living in the United States. Each day, a head of cattle produces six times the waste of the average human being. Erosion control in and of itself is a major cause of non-point source pollution but the costs of remedying it are astronomical and not entirely calculated. The Environmental Protection Agency has made various estimates in just three of those areas.

EPA has estimated that highway erosion control could cost anywhere from a $130 million to $7 billion; EPA has estimated that urban construction erosion control would cost anywhere from $140 million per year to $1.4 billion per year. EPA has estimated that sediment control in streams would cost approximately $200 million per year to $3 billion per year. These statistics illustrate that the problem of non-point source pollution is multi-faceted and its effect on water is pervasive. Indeed, with such diverse sources as animal waste, erosion, and sedimentation causing non-point source pollution of waters, it tends to make the allocation of responsibility in traditional tort litigation almost impossible.

A convenient starting point for review of the practical effectiveness of the tort system to treat of non-point source pollution is to review the source of the rights to be enforced. The most obvious rights against non-point source pollution are the rights of a riparian owner. A riparian owner is the owner of property bordering upon a water body. The riparian owner has a right to use of the water subject only to the use being reasonable and to the return of the used water in as to close to its original state as possible. A riparian owner obtains rights simply by virtue of the ownership of land bordering the water body. The owning of land bordering a water body brings with it a common law right to use the water in common with all other riparians.

An action by a riparian to protect his rights is limited to bringing an action against another riparian user. If a riparian owner seeks to obtain damages from a non-riparian, the riparian is limited to suit alleging nuisance.

In a suit to protect its riparian rights, the riparian need only show that another riparian has interferred with the suing riparian's use of the water. It need not show negligence nor any form of intentional invasion of a property interest. While this simplifies the burden of proof on the part of the riparian, it does not entirely answer the non-point source pollution problem since the polluter is subject to damages for its pollution only. Further, even a polluting riparian is allowed a reasonable use of the water which may include some reasonable amount of pollution caused by such use. This may be compounded by a downstream owner in effect granting an easement to pollute to an upstream riparian by acquiescing in a course of conduct which is known, and notorious.

The other major source of rights is found in the law of nuisance. A nuisance is an unreasonable interference with a person's enjoyment of land. In the case of a riparian owner, nuisance would be established if it could be shown that the defendant did an act that polluted the water in a material fashion and that the defendant's conduct was a proximate cause of that unreasonable interference.

It is the concept of proximate cause, however, which again poses the most substantial impediment in an action for nuisance. Proximate cause is at best difficult to establish when there are multiple discreet dischargers, making the tracing of water quality at best unachievable. It was the effective inability of government to trace a water quality violation from point source dischargers that led to the Federal Water Pollution Control Act Amendments of 1972. Tracing non-point source pollution dischargers presents an even tougher cause and effect problem.

The point of the above discussion is to show that traditional tort remedies for relief from non-point source pollution are simply inadequate to address the problem. The number of polluters are many, their pollution is diverse, and the cost of ascertaining proof to the requisite degree of certitude is almost prohibitive.

The law has attempted to find ways around these limitations resulting from a "but for" causation standard of damage by a defendant. Most of these attempts have developed in the field of products liability litigation. This alone makes use of these theories suspect when treating of non-point source pollution. There are grave conceptual problems in superimposing these doctrines upon non-point source pollution and it is accordingly not a recommendation that this be done. Nonetheless, it is important that we recognize what those theories are so that by understanding them we can appreciate why they should be rejected.

The four prevalent theories are: 1) alternative liability, 2) market share liability, 3) enterprise liability, and 4) concertive action liability. In the State of Wisconsin, only the latter, concertive action liability, is a recognized theory for allowing recovery in a products liability circumstance. A brief description of each of the four and its limitations:

1. Alternative Liability: If more than one defendant could have caused the liability, the defendants may all be joined in the litigation and the defendants must prove among themselves who is not liable. This theory of liability may not be used when a court does not have jurisdiction over all of the defendants. This poses a major problem with multiple jurisdiction pollution sheds. It also causes problems with large, undefined groups of defendants as are likely in non-point source pollution. Further, some of the defendants may be governmental bodies which are protected by dollar limits on their potential exposure.

2. Market Share Liability: This is based upon the theory that when the entire market is divided among a fixed group of defendants, all participants receive benefit from the existence of the market and should share in liabilities arising from marketing. All market participants are not needed in the litigation as a defendant, but any defendant in the litigation would be responsible for only its pro rata share of the liability in proportion to its share of the market. This is not readily available as an alternative for non-point source pollution because non-point source pollution is not a "market" oriented event similar to manufacture or distribution of a pharmaceutical product.

3. Enterprise Liability: This applies where there is an industry wide practice which created the unreasonable risk of harm, which risk is not capable of attribution to a single defendant. Like the alternative liability theory above, the defendants are required to prove among themselves who is not liable; unlike the alternative theory but like the market share theory the enterprise liability theory does not require that all defendants be in the suit. What is required, however, is that there must be a well defined industry and it requires an awareness of the risk and a joint liability to reduce the risk among all of the defendants. These characteristics remove non-point source pollution from the enterprise liability theory of recovery since there is no well defined industry nor is there a joint liability to reduce risk;

4. Concertive Action Liability: This theory of liability requires that there be a common design or agreement to act in a certain fashion. Under this theory, all participants in the concerted action are responsible for any harm that is created. The concerted action theory does no require any written agreement, and can be established by a showing of conduct estabhlishing action in concert. The fault with this theory in relationship to non-point source pollution is that the showing of concertive action across the broad cross sections of society that create non-point source pollution is virtually impossible.

In a sense the harm caused by non-point source pollution is a public harm, a public harm that cries for some form of intervention by the government. It is this no longer exclusively private harm which should attract the attention of the respective legislatures to conceive of means to address the need for a remedy to non-point source pollution.

The next problem, of course, is what form should that legislation take.

THE MANY FACES OF LEGISLATIVE SOLUTIONS

Recourse to legislation as a solution to the problems of non-point source pollution, while undoubtedly the preferrable means, nevertheless creates the problem of what form the legislation should take. It must borne in mind that the legislative forum has a number of limitations. Legislation often times requires political compromise and political compromise is

not susceptible to analytical review. Further, three important problems arise with legislation.

First, it must be borne in mind that fixing a problem is always more expensive than creating a problem. It is human nature that if something can be done cheaper but create a problem, it will have been done cheaper nevertheless. Second, it is hard to assess environmental problems in terms of unit costs. Thus, unlike fixing costs of pollution based upon production, the benefit of fixing an environmental problem or the harm caused by an environmental problem cannot be reduced to economically meaningful values that fluctuate in a predictable fashion. Third, environmental problems themselves are subject to change by merely redefining what the problem is. In the context of non-point source pollution, if stream erosion is eliminated from the definition non-point source pollution you have created an entirely different perspective on the problem. By like token, if urban non-point source pollution is eliminated from the legislation, you have a different problem than if you were dealing with all non-point source pollution.

Bearing in mind these limitations on the legislative process, there are basically four ways to achieve a legislative solution to environmental problems. These are: (1) change the legal status of the problem; (2) regulate the problem; (3) tax the problem; and (4) subsidize the solution.

Legislation that changes the legal status of an environmental problem is easiest to proclaim, but its ramifications are often the most pervasive and hence disruptive manner of dealing with the pollution problem. The Federal Water Pollution Control Act Amendments of 1972 are perhaps the best example of legislation which changed the legal status of a pollution problem. The Federal Water Pollution Control Act Amendments were predicated upon a change from water quality standards dictating what the effluent limitation should be to a system whereby a legislatively determined effluent limitation would be achieved. The legislation decreed a zero pollution goal. Notwithstanding the unreality of attaining a zero pollution goal, the idea behind it was essentially to remove the nation's waterways from the order of public goods free and available to all. In its stead, Congress created a prohibition against the use of those waterways as discharge points for pollutants.

The use of legislative change in legal status to address non-source point pollution may not correct the problem, however. Politicians are pragmatic people in many ways. It will require more than just a definitional change in legal status to work the full change in legal status that is necessary to control

non-point source pollution. The Federal Water Pollution Control Act Amendments illustrate this point. The Congress enacted a complete package of changes, including a construction grant program, a permit program, water quality standards and state implementation plans program, and a creation of effluent standards program. A change in legal status for non-point source pollution would require a similar, all pervasive response. I submit that such an all pervasive change in the status of non-point source pollution, while possible, is not yet politically expedient.

A second legislative form to deal with non-point source pollution would be to regulate the amount and means of non-point source pollution discharges. Regulation is favored in America for a number of reasons. Americans have a great love of quick solutions and are willing to abdicate their ability to attain greater societal change to a legal solution for a social problem rather than develope other practical ways of dealing with that problem. Many social problem solvers are also lawyers, resulting in a tendency on their part to create remedies with which they are familiar. Further, regulations alleviate the need for people who are relatively unsophisticated in economic and social theory to engage in intellectually painful analysis of how their acts economically effect society as a whole. Last, Americans favor regulation of environmental problems because it permits them to ease their consciences more easily. This is important for many because environmentalism has taken on virtually a religious fervor.

Regulation is, contemporaneous with its ease in being understood, economically quite troublesome. The relative ease in understanding regulation of non-point source pollution comes from the use in most regulatory shemes of the prescribed standard: one standard for all like discharges. It is this recourse to a uniform standard that gives rise to the problems, however, because this solution does not admit of any inefficiency in its implementation and because the standards are more or less rigid. Coupled with this is recognition that there is never a situation of governmental regulation without a cost of its own, if in no other way than salaries and the like for administration of the regulation, the creation of a bureaucracy. Government regulation creates distortions in economic efficiencies. This inefficieny is caused by the tendency to regulations to require "equally proportionate reductions" of pollutants from each source regardless of variance in control costs.* Although it is easy to

*Stewart and Trier, Environmental Law and Policy, 556 (Bobbs-Merrill, 1978)

prescribe a standard, it is not so easy to justify the standard. Thus, determination of standards can be quite difficult.

The third legislative avenue for dealing with non-point source point pollution is to create a tax on discharges. Essentially, a tax would take the form of an emission fee per unit of discharge. While taxing a unit of discharge is a ready shibboleth to the economic sloganeer it is not a concept that is free of significant problems.

Costs of abatement may vary per unit taxed, which is economically good. Taxes do provide an incentive for reduction and taxes are more adaptable and more easily administered. Nonetheless, the standards to be achieved tend to be quite static. Changing the standard to reflect varying efficiencies is not easily achieved.

The tax concept has many of the economic inefficiencies of the regulation except the tax furnishes an economic incentive to abate the pollution. Establishing a standard to be achieved is at best guesswork aided by political biases. The unit of detection, the pricing mechanism, and monitoring are even more problematic.

The last form of legislation that may be utilized to control non-point source pollution is the creation of a subsidy. Subsidies take essentially two forms. One is tax credits or increased depreciation for capital expenditures related to reduction in non-point source pollution discharges. Tax incentives and increased depreciation for capital expenditures create problems notwithstanding that the government in essence underwrites the efforts being made to relieve non-point source pollution because there is a cost to the operation which had been making the discharge. Depending upon the discharger, that cost may be serious enough to turn what is otherwise a profitable function into a nonprofitable function, primarily when these costs are not common across the competitive sphere. For example, where the government for any region less than the whole nation, such as the State of Wisconsin, requires its farmers to engage in various non-point source pollution elimination activities, including capital expenditures, no matter what tax incentives are given and no matter what increased depreciation is allowed, there will nonetheless be a capital expense to the farm operation which farm operations in other states or regions do not have. In a marketplace where dairy products, grain products, cattle products are essentially fungible, requiring but one group of competitors in that market, Wisconsin farmers, to take on the additional costs puts an undue hardship upon them and puts them at a distinct market

disadvantage.

A second form of subsidy is the concept of buying up units of non-point source pollution discharge. The legislature would establish an acceptable level of discharge that must be met and create a fund with which to eliminate discharge to that level. For every unit of historic discharge which is not discharged following the enactment of the subsidy, the discharger would receive a bounty from the government. Those dischargers who have relatively low costs in achieving discharge curtailment will then rapidly move into that field and will curtail their discharge and obtain the bounty money; those units with higher costs to curtail discharging will not be able to partake of the bounty. Nevertheless, this latter group would have the benefit of being able to use more of the acceptable limit for their specific discharges.

Both subsidiary ideas, however, tend to break down in the real world because the transference from theory to practice contains a cost for government administration which cannot be readily or accurately ascertained. The cost for ascertaining the historic level of non-point source pollution discharges is quite illusive and not readily calculable to the requisite degree of certitude.

The four forms of legislative response to the problems of non-point source pollution each have their own problems. When the public wants a solution to non-point source pollution, the legislature must weigh which is more likely to meet the public's demand. The outside influence of the citizen demand for action can force a choice.

III. PROPOSED SOLUTION

It seems that the most ready way to deal with non-point source pollution is to regulate the discharge of pollutants into the water. Although it has severe disadvantages, the use of regulation does have a simple, quick solution aspect to it which tends to be favored in American society. Regulation itself could establish preferred pollution avoidance practices which could be designated and enforced either by a state agency or as private rights.

Furthermore, once legislation takes up the subject matter of non-source point pollution it may remove it from local politics. Additionally, regulation need not address the whole problem at one time. Regulation would permit the legislature to address a major segment at a time. For example, it is not inconceivable for the legislature to designate the Department of Natural Resources or the Department Health, Social Services or the Department of Industry, Labor and Human Relations as the

authority to identify construction site pollution emission discharge limitations. The legislature could then delegate to the department the authority within another year to promulgate regulations on sediment control and yet further regulations on highway non point source pollution discharges. In this fashion the legislature would address the entire problem on a timetable.

The State of Florida has began such efforts. Its Department of Environmental Regulation has enacted a number of non-point source pollution elimination steps concerning highway erosion control which have subsequently lessened the discharge of non-point source pollution to affected water bodies. Such a role could be achieved in Wisconsin and in other areas nationwide.

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SUMMARY
Panel III
Efficiency and Feasibility of Best Management Practices
For Nonpoint Source Control

PANEL MEMBERS

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INTRODUCTION

The panel discussed the effectiveness and feasibility of both urban and rural BMPs, and addressed analytical techniques for evaluating the practices. The need to properly define nonpoint source pollution problems and evaluation criteria was also noted. It was acknowledged that problems are multi-dimensional, and that implementation of remedies can be greatly affected by social, cultural, religious, economic, and personal preferences.

It was questioned whether too much emphasis in evaluating nonpoint source control measures is placed on modeling, and whether more attention should be given to monitoring the water quality effects of BMPs. Panel members felt that modeling was most valuable for guidance in selecting individual, or an optimal mix of, BMPs. There should be good feedback between monitoring and modeling, with monitoring data being used to calibrate and verify models. The most beneficial use of models is in the insight they provide, rather than in the absolute quantification of variables. Models

also provide an economical means for assessing alternative conditions which could not realistically be evaluated with monitoring data alone. It was concluded that modeling and monitoring programs for nonpoint source control are not substitutes; rather, they complement each other.

RURAL NONPOINT SOURCE CONTROL

Concerns

1. Voluntary NPS control programs may not be effective and achieve successful implementation in areas with intensive agricultural production, particularly livestock raising.

2. BMPs may be applied for water quality improvement purposes without regard to cost-effectiveness and pollutant removal capability.

3. BMPs which effectively reduce soil loss by reducing surface runoff may not sufficiently control nitrate leaching from excessive fertilizer use of manure applications.

4. Manure storage facilities may simply encourage farmers to expand livestock operations.

5. Cost-effective waste control measures for unconfined livestock operations still need to be demonstrated.

Recommendations

1. Cost-effective soil conservation measures include contouring, vegetative cover, conservation tillage, and strip-cropping. Terraces are effective, but not relative cost-effective. Waterways and diversions cost less than terraces, but are also less effective.

2. Nitrate leaching into the groundwater can be controlled by reducing heavy applications of manure and fertilizers, (especially in highly permeable soils), and by increased education of farmers.

3. The most efficient water quality improvement programs would utilize the most cost-effective BMPs in critical NPS problem areas.

4. Confined livestock waste discharges should be collected and treated as a point source. Ultimate disposal of the animal waste should be land application or use as a feed source. Lagoon treatment and direct discharge to surface waters is unacceptable.

5. Unconfined livestock waste discharges should be treated with practices consistent with optimal forage production on a long-term basis. Congregations of animals in critical areas should be prevent. Grazing areas should be maximized and rotated.

URBAN NONPOINT POLLUTION CONTROL

Concerns

1. Proper selection of urban BMPs is difficult because of the wide range in the ability of the BMPs to remove various pollulants.

2. There is a lack of available BMPs which can achieve a reasonably high level of pollutant removal in fully developed urban areas.

3. Stormwater storage in wet basins, one of the few highly effective urban BMPs, cannot be used in areas with high slopes, low permeability, or limited space.

4. Because of the limited knowledge available concerning the design, operation, and effectiveness of urban BMPs, mandatory specifications of BMPs would be premature and would result in "action", but little progress.

Recommendations

1. Effective urban BMPs include wet basins, porous pavement, infiltration systems, and artificial wetlands. Dry basins, grassed swales, and street sweeping have not been found to be very effective.

2. Among stormwater storage measures, dry basins cost least; extended dry basins cost about 10 more, wet basins 20 to 45 percent more. Infiltration systems are cost-effective only in smaller residential areas.

3. Urban nonpoint source control programs are best implemented under the guidance of a comprehensive stormwater management plan which addresses both water quantity and quality concerns.

4. Screening procedures should continue to be developed to assist local developers in selecting those urban BMPs which are most appropriate within a given range of site characteristics.

5. Stringent inspection and maintenance programs must be developed to help ensure the continued effectiveness of urban BMPs especially storage facilities.

CONCLUSIONS

Following review of the urban and rural BMPs available, some panel members expressed concern about the inability to specify the relative cost-effectiveness, and to quantify the pollutant removal effectivenss of individual BMPs. The panel concluded, however, that many excellent studies and demonstration projects have been completed, and implementation of both urban and rural practices should proceed. More will be learned as these practices are implemented on a wide scale. The advancing state-of-the-art have provided a strong direction toward which types of BMPs work best

under certain conditions. The panel expressed confidence that the techniques are available, and the knowledge exists, to sufficiently control both urban and rural nonpoint source pollution in most areas. New and emerging technologies will likely "fine tune" the system, enhancing water quality improvements achievable with the present range of BMPs. Inability to guarantee that any BMP selected for implementation is the wisest choice should not be used as an excuse for inaction. Immediate prudent use of existing patently cost-effective BMPs should be encouraged because they are all steps in the right direction.

EVALUATING BMPs IN PENNSYLVANIA'S CONESTOGA HEADWATERS RURAL CLEAN WATER PROGRAM

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ABSTRACT

This paper discusses the relative cost effectiveness of best management practices (BMPs) for controlling agricultural nonpoint pollution. The discussion herein applies directly to the control of soil and nutrient losses resulting from crop production, and is derived from the economic evaluation of the Conestoga Headwaters Rural Clean Water Program (RCWP) project in southeastern Pennsylvania. The purpose of the project is to provide assistance for the installation and maintenance of BMPs to control nonpoint pollution and improve water quality.

Due to the current status of BMP implementation in the project, monitoring data are not available to evaluate the impacts of BMPs on water quality. Physiochemical and economic modeling of BMP impacts was undertaken to characterize the cost effectiveness and overall effectiveness of BMPs for controlling agricultural nonpoint pollution. CREAMS, a field scale model for Chemicals, Runoff, and Erosion from Agricultural Management Systems, is a continuous computer simulation model that was used to estimate field-edge losses of soil, surface runoff losses of N and P, and NO_3 leached out of the root zone.

Contouring and stripcropping (where the crop rotation makes it feasible) were the most cost-effective practices for controlling erosion and surface runoff losses of nutrients. Conservation tillage practices were also cost-effective alternatives. Terrace systems were highly-effective runoff control measures but were generally the least cost effective. Other structural practices, waterway systems and diversion systems, were found to be less effective than terraces for soil loss control, but were more cost effective.

NO_3 leaching, perhaps the project's most significant water quality problem, may not be alleviated and may increase when soil conservation practices are implemented. Sound nutrient management practices are necessary to control subsurface and surface deliveries of nutrients to water supplies. Nutrient inputs from animal wastes, commercial fertilizer, and crop residues must be managed in a way that prevents excessive inputs and limits movement off the field. Manure storage and application techniques should be consistent with maximizing the economic value of nutrients while minimizing field losses.

Keywords: Agricultural nonpoint pollution, best management practices, economic modeling, cost effectiveness, water quality, CREAMS.

INTRODUCTION

Increasingly intensive crop and animal production have characterized southeastern Pennsylvania's Conestoga Headwaters Watershed and the remainder of Lancaster County. This production intensity has been motivated by: (1) urban pressures on agricultural land, especially near the city of Lancaster; (2) Amish and Mennonite family farming traditions, which have resulted in the division of farmland into small parcels among children; and (3) proximity to large East Coast markets. The watershed is 1 of 5 experimental RCWP projects nationwide to be selected for comprehensive monitoring and evaluation. There are 44,516 hectares of land, of which 26,305 is in farms.

The "average" farm in the watershed has 21 hectares of land, almost entirely used to produce feeds for animal operations that usually exceed 5 animal units (AU) per hectare of farmland (USDA, 1982). Over a 13-year period from 1970 to 1983, the number of milking cows increased by 67 percent, swine production increased by 189 percent, and poultry production increased by 145 percent. During the same period, the percentage of total field and forage crops that was in corn grain and silage increased from 33 and 12 percent in 1970 to 36 and 24 percent in 1983, respectively (Pennsylvania Crop Reporting Service, 1970, 1983). Commercial N fertilizer purchases increased over the same period to meet the perceived needs of the additional corn.

Manure is applied at an average annual rate of 90 MT per hectare in the watershed. The average fertilization per hectare for corn, from animal manure and commercial fertilizers, is: 482 kg N, 304 kg P_2O_5, and 304 kg K_2O (USDA, 1983). These high nutrient inputs and the intensity of agricultural production have resulted in local water quality degradation.

WATER QUALITY PROBLEMS

Agricultural nonpoint pollution in the Conestoga Headwaters Watershed affects the water supplies of 175,000 people. The water quality parameters of greatest concern are: (1) high NO_3 concentrations in groundwater, especially in private wells; (2) fecal coliform (FC) bacteria in surface and groundwater supplies; (3) triazine pesticides in both surface and ground waters; and (4) sediment and turbidity in streams.

Most private wells exceed the Environmental Protection Agency (EPA) drinking water standard of 10 ppm NO_3-N, with many exceeding 30 ppm NO_3-N concentration during the year. Abnormally high levels of N, P, pesticides, and FC bacteria also have been found in streams and wells (USDA, 1983). Average annual soil losses are estimated to be 19.7 MT per hectare in the watershed, with annual losses from individual fields often exceeding 100 MT per hectare. EPA has identified southeastern Pennsylvania agriculture as a major contributor to the Chesapeake Bay of N (especially NO_3 during spring runoff events), toxic substances, sediment, and to a lesser extent P (EPA, 1983).

It is the goal of the RCWP project to substantially reduce the delivery of agricultural pollutants to receiving waters by implementing conservation and

other BMPs on farms. The purpose of this paper is to discuss the effectiveness and cost effectiveness of selected management practices for controlling field losses of soil and nutrients and ultimately for improving water quality. This discussion is directed toward an area characterized by erosion problems and high manure inputs.

BMP EFFECTIVENESS FOR CONTROLLING POLLUTION

Farmers in the RCWP project area currently are implementing contracted BMPs. Water quality is being monitored and data are available only for current management practices. It could be years before newly installed RCWP practices cause measurable changes in water quality. As a result, computer modeling was chosen to evaluate selected RCWP practices.

Modeling soil and nutrient losses

In order to isolate the effects of BMPs, both individually and in combination, the CREAMS model was used to obtain estimates of field-scale soil and nutrient losses. CREAMS is a computer simulation model, constructed by USDA's Agricultural Research Service to compare relative field losses of soil and selected chemicals among different management practices (Knisel, 1980). Computer modeling provides a method of estimating BMP effectiveness for pollution control without incurring the financial and time costs of BMP implementation and monitoring.

CREAMS estimates of field losses were used with economic optimization and budgeting techniques to evaluate the cost effectiveness of: (1) field BMPs; (2) manure storage and handling systems (and their effects on nutrient conservation/availability and field losses); and (3) manure transport (in order to reduce high rates of manure application). Surface runoff losses of soil, N, and P, and subsurface losses of NO_3 leached below the root zone, were estimated using CREAMS and published data (see USDA, 1984). Pesticide losses in surface runoff also can be modeled with CREAMS, but are not addressed in this paper. Deliveries of soil and nutrients to surface and ground waters can be assumed to be some fraction of the CREAMS estimates of field losses.

Soil losses

Table 1 shows the field BMPs that were modeled, their approximate costs, and the reductions in losses of soil and nutrients associated with the BMPs. The baseline soil loss was 24.3 MT per hectare per year. This represented the soil loss for conventional practices -- tillage up and down the slope with a moldboard plow followed by 2 passes with a harrow. A distinction is made between nonstructural management practices and structural practices.

Nonstructural management practices

Permanent vegetation, hay or pasture, is a highly-effective soil conservation practice. It also appears to be relatively cost effective, with cost effectiveness defined here as the percent reduction in field loss per dollar of expenditure. Permanent vegetation is applied with RCWP cost sharing only on severely erosive land that generally is adjacent to waterways. The opportunity cost of removing land from row crop production is prohibitive

without 50-percent government cost sharing, and therefore the practice has limited applicability.

Contour stripcropping involves performing tillage operations along the contours of the fields, and alternating row crops with close-seeded hay or small grain crops in strips (modeled to be about 37 meters in width using CREAMS). The effectiveness of stripcropping is primarily due to the reduction in average surface runoff caused by the strips of close-seeded crops, not accounted for by modeling the strip of corn in Table 1. Soil loss was reduced by 39 percent at no cost, making contour stripcropping a highly cost-effective practice for erosion control.

Conservation tillage practices -- reduced tillage (chisel plowing followed by harrowing once) and no-till -- were found to be both effective and cost effective for erosion control. No RCWP cost sharing is provided to convert tillage practices. This is consistent with findings by Crowder, et al. (1984) that conservation tillage practices reduce ownership and operating costs, resulting in greater profitability.

Winter cover and residue management are important components of any conservation tillage system. When combined with the zero-cost practices of conservation tillage and contour stripcropping, the overall management system provides significant erosion control for the cost of winter cover and residue management, $0 to $50 per hectare annually (depending on the crop residue).

Structural practices

The more important structural practices being implemented with RCWP cost sharing include: (1) pipe-outlet terraces; (2) diversion systems; and (3) sod waterway systems. The first column of Table 1 shows the approximate annual costs per hectare served for structural practices, amortized at 12 percent interest over 10 years.

Terraces are highly effective for erosion control, but at relatively high cost. If a 10-year useful life is assumed, the annual cost per hectare is $163.70. With 75-percent cost-sharing rates, however, terrace systems can be made attractive to farmers. Sod waterways and diversions may be more effective for preventing soil loss to streams than for reducing erosion. Given these circumstances, they are probably more effective for protecting water quality than soil productivity. Because they are considerably less expensive than terrace systems, sod waterways and diversions appear most cost effective than terraces for protecting water quality.

The last entry in Table 1 illustrates the reductions in pollution resulting from a typical combination of soil conservation practices. An 89 percent reduction in soil loss is possible at an approximate annual cost of $225 per hectare. The cheapest combination of management and structural practices can be chosen that will meet targeted soil loss reductions on a given field. In this way, soil losses can be reduced at minimum cost to both farmers and taxpayers.

Table 1. Annual costs and effectiveness of conservation practices for continuous corn silage, daily spread 45 metric tons manure per hectare per year, 5 percent slope, Duffield silt loam.

Conservation practice	Estimated annual cost per hectare	Percent reduction in field losses			
		Soil	N in surface runoff	Total N	Total P
Conventional practices	N/C	--	--	--	--
Permanent vegetative cover	$83*	95	87	45	91
Contour strip-cropping	N/C	39	30	18	33
Winter cover and residue management	$0-50	14	11	9	13
Chisel plowing/ reduced tillage	N/C**	44	34	20	37
No-till	N/C**	68	50	38	54
Terrace system	$163.70*	72	56	32	62
Diversion system with 6-meter wide sod filter strip	$30.09*	43	42	24	46
Sod-waterway system	$20.35*	64	49	29	52
Reduced tillage with terraces, contouring, stripcropping, residue management, and sod waterway	$225	89	78	43	82

*Permanent vegetation was amortized over 5 years at 12 percent interest, and is cost shared at rates up to 50 percent. Structural practices were amortized over 10 years at 12 percent interest. Cost-sharing rates of 75 percent on structural practices reduce farmers' annual before-tax costs to 25 percent of those shown.

**Cost of winter cover and residue management must be included for continuous corn.

Source: USDA (1984).

Nutrient losses

As part of the RCWP economic evaluation, 4 approaches to farm nutrient management and conservation were identified:

1. Use field conservation practices (BMPs) to control surface runoff losses of N and P.
2. Reduce animal manure production and/or commercial fertilizer applications.
3. Improve the allocation and application of manure and nutrients throughout the farm.
4. Export manure from farms with excess nutrient problems.

Field conservation practices (BMPs)

Nutrient losses in runoff can be reduced by implementing conservation practices which restrict surface runoff and soil loss. The last 3 columns of Table 1 show the reductions in nutrient losses for the BMPs discussed earlier. Annual losses for conventional practices were 58 kg N per hectare in surface runoff (of which almost 90 percent was sediment associated), 97 kg total N per hectare (39 kg of NO_3-N per hectare was leached through the root zone), and 28 kg total P per hectare in surface runoff.

The effectiveness and cost effectiveness of soil conservation practices for controlling surface runoff losses of nutrients parallel those for controlling soil loss. This follows because most nutrient losses in surface runoff were sediment-associated for the baseline conditions modeled. However, soil conservation practices have little effect on controlling NO_3 losses in percolate. In fact, such practices may actually increase the load of NO_3 lost by leaching because of the increased water percolation through the soil profile, caused by reductions in surface runoff. From the perspective of N mass, the gains in N control are significant although not proportionate to the reductions of soil and P losses. However much, and perhaps most, of the sediment-associated N will not become biologically active, so the efficiency of erosion control methods for controlling biologically active N losses is even less than projected in Table 1.

As for other water quality parameters discussed, permanent vegetative cover and the combination of BMPs (last entry in Table 1) are the most effective for controlling total N losses, followed by no-till, terraces, and sod waterways. Diversions, reduced tillage, and contour stripcropping estimates indicate that they can significantly reduce total N losses. Among individual practices (ignoring permanent vegetative cover due to its limited applicability and effects on farm forage production), contour stripcropping, no-till, and then reduced tillage are the most cost-effective nonstructural practices for controlling total N losses. In addition, nonstructural practices have the advantage of keeping soil and nutrients in place on farm fields. For structural practices, total N loss reduction per dollar of cost is greatest for sod waterways, then diversions, and last terraces -- the same cost-effectiveness ranking as for soil and P losses.

Decision makers should be aware that groundwater and stream pollution problems are not likely to be alleviated by the implementation of soil conservation practices alone if NO_3 is a critical water quality parameter. Improvements in

surface water quality from surface runoff control may be offset by increased NO_3 loads in groundwater and base flow. When groundwater NO_3 levels are a concern, as they are in the RCWP project area, nutrient management practices probably will be necessary to improve water quality substantially.

Reducing nutrient applications

The second way to reduce farm nutrient losses is to match manure and fertilizer nutrients more closely to crop needs. Available nutrients in manure and legumes should be credited toward the fertilizer needs established by soil test recommendations.

Part of the over-application of N could be prevented if farmers had inexpensive, accurate estimates of available soil N. Unfortunately, such estimates are not readily available, and the problem is further compounded because the N that becomes available from current and previous manure applications is not easily predicted. Due to these uncertainties, it makes economic sense to apply excess N commercial fertilizer as insurance to guarantee crop yields. Over-application of nutrients would be expected to decrease as the degree of uncertainty is reduced.

An example of the effects of over-application of nutrients, due to excess manure (which is responsible for much of the nutrient problem in the project area), is shown in Table 2. A manure application of 45 MT per hectare results in substantially less nutrient loss than the average watershed application of 90 MT per hectare. Fertilizer and manure management plans are being implemented through the RCWP program. Better timing of manure applications to enhance crop uptake and prevent field nutrient losses, and eliminating the large commercial N fertilizer applications that exceed crop requirements, can reduce the stream delivery of nutrients as well as other pollutants. Reducing fertilizer purchases also reduces farmers' costs. Reducing the application of manure nutrients is difficult when excess manure is produced on a farm. Storage, handling, and application systems which encourage N volatilization can be used, or the number of animal units on the farm can be reduced -- not an attractive option for farmers.

Allocation and application of animal manure

Significant reductions in farm nutrient losses may be obtained by applying manure evenly throughout farm fields, and by properly storing manure during crop growth periods and the winter months. For example, variation in soil NO_3 levels within a single field varied from 3.3 to 230 kg of NO_3-N per hectare. This was attributed primarily to "a relatively poor job of spreading nutrients from manure." It was predicted that most groundwater pollution results from those sites on fields with extremely high soil NO_3 levels (USDA, 1984). It is believed that significant nutrient conservation and pollution reductions are possible when manure nutrients are spread evenly on fields. Such a claim, however, is difficult to quantify.

Table 2 provides an illustration of the estimated nutrient losses for different manure storage periods and application methods. Losses shown are field losses only, and do not account for barnyard runoff losses (which usually are significantly greater on farms with unimproved barnyards, and on farms where manure is stacked and spread periodically -- a practice common

Table 2. Effects of storage period on nutrient losses (kg/ha) for corn silage following corn silage, 5 percent slope, Duffield silt loam, 45 and 90 metric tons of manure annually per hectare.*

Application	Nitrogen Losses			Phosphorus Losses
	Percolate	Surface	Total	Total
			45 MT/ha	
Daily spread on surface	41	13	54	5
Six-months storage with plowdown	45	12	57	5
Six-months storage with injection	45	13	58	5
Twelve-months storage with plowdown	33	13	46	5
Twelve-months storage with injection	20	15	35	5
			90 MT/ha	
Daily spread on surface	66	25	91	9
Six-months storage with plowdown	82	21	103	8
Six-months storage with injection	86	24	110	8
Twelve-months storage with plowdown	62	24	86	8
Twelve-months storage with injection	72	28	100	8

*Practices include reduced tillage, winter cover, contouring, stripcropping, with terraces and sod waterways installed.
Source: USDA (1984).

among farmers in the RCWP project area). Manure storage systems conserve nutrients, especially N, which is undesirable when plant-available N from animal manures alone exceeds county crop nutrient requirements by 35 percent. The P and K available from manures exceed crop nutrient requirements by 170 and 50 percent, respectively (USDA, 1983).

Field nutrient losses were similar for daily spreading and 6-month storage systems at an annual loading rate of 45 MT per hectare of manure (Table 2). Percolate NO_3-N and total N losses were substantially less for 12-month storage, an effective strategy for reducing nutrient loss. Also, nutrient conservation by manure storage can offset most, if not all, of the amortized costs of a storage structure. The economic modeling results indicate that, for a small 45-cow dairy with 24 hectares of tillable land (resulting in slightly less than 45 MT of manure per hectare applied to corn land), a 6-month earthen basin liquid storage system can increase annual net returns to the farmer by $750 (with no cost sharing for the structure) compared to a daily spread system. A 6-month solid storage system was found to decrease annual net returns by $700 compared to daily spread.

If, however, a farmer in the RCWP project area stores manure and applies it at the average rate of 90 MT per hectare, total N losses are about the same or substantially greater than the daily spread system (Table 2). If a farmer has excess manure nutrients, a storage structure no longer pays for itself because the marginal productivity of the conserved nutrients is no longer positive. Some reduction in total nutrient losses is possible by storing manure, but at substantially more cost than for farmers who do not have excess nutrient problems.

A critical concern for farmers is the effect of on-farm nutrient-loss abatement on net income. For the representative small dairy farm mentioned above, the results indicated that it was possible to reduce field losses of nutrients by storing manure, by uniform spreading of manure, and by growing less erosive crops or rotations on erosive fields. Further low-cost reductions in nutrient losses can be obtained by spreading manure evenly within the field borders, by performing conservation tillage operations along field contours, and other conservation management practices discussed before. These more cost-effective practices should be pursued to reduce field losses before those which significantly reduce income and cost the government large amounts of cost-sharing funds. In addition, if manure storage is planned on farms with high animal-to-land ratios, it should be designed not to conserve but to release unwanted nutrients by chemically degrading them to non-noxious or biologically unavailable compounds.

Off-farm disposal of manure

The last approach to reducing farm nutrient losses involves transporting excess manure to other locations, where its application will not be environmentally harmful and where its nutrients can be utilized more fully. With the volume of excess manure that exists in Lancaster County and surrounding areas, it is not anticipated that profitable manure marketing (for farmers with excess manure) is possible on a wide scale. The trend toward more intensive dairy and animal production will exacerbate the problem, and probably will require future manure hauling over longer distances than those required currently for safe disposal.

Off-site manure disposal is costly for a number of reasons. First, manure is a bulky material to handle and transport. Second, ensuring safe application to land, so as to minimize nutrient losses, limits the time period that manure can be applied. Manure applications must be compatible with crop growth cycles. Thus, manure must be stored for up to 10 months, to allow most of the manure to be applied prior to spring planting of row crops. Last, sufficient hauling and application equipment should be available to apply manure during this limited time period. Such equipment will stand idle during the remainder of the year. Therefore, manure hauling over long distances is not a low-cost method for reducing nutrient losses.

An option for hauling manure up to 40 miles, from the representative small dairy farm discussed earlier, was modeled using linear programming. It was assumed that farmers would pay for hauling and give the manure free to farmers receiving it in order to reduce manure applications on the originating farm. The receiving farmer would pay for field application. Using the CREAMS estimates of N loss, it was found that for a farmer using a daily spread system, income was reduced by about 20 percent when manure was hauled 40 miles to reduce total N losses by 30 percent. It was found under the assumptions employed, that if a farmer stored manure for 6 months in an uncovered solid storage structure, income was reduced by only about 8 percent while achieving the same 30 percent reduction in total N losses. Thus, manure storage structures may be necessary to cost-effectively institute off-farm manure disposal in the RCWP project area and elsewhere. Cost-sharing funds provided by RCWP and other programs for storage structures may make manure export economically viable for farmers with manageable manure problems. It is expected that manure export will impose substantial income penalties on farmers who have substantially more manure than can be used on their farms, unless government subsidies are provided.

Uses of manure other than fertilizer, most notably as fuel for power generation, usually require economies of scale at the farm level to be profitable. Refeeding manure to livestock causes health problems when done at significant levels. Ongoing research on these and other alternatives may provide hope for profitable off-farm manure utilization in the future.

CONCLUSIONS

It is apparent that while reductions in soil and P losses from farm fields can be achieved with soil conservation practices, the control of N losses (as well as FC bacteria) is more complicated. Reducing surface runoff and its constituents results in increased percolation of water through the soil. This potentially increases leaching of NO_3, pesticides, FC bacteria, and some small amount of P to groundwater. In short, solving one problem likely aggravates the other.

A comprehensive plan to promote sound management of manure, fertilizer, and pesticides is necessary to improve water quality in the RCWP project area. The rigorous application of manure management and soil conservation practices could impose severe financial hardship on farmers if government assistance for such practices is not continued. Even with the traditional forms of assistance provided through RCWP and other programs, farmer participation in

conservation planning has been poor in the project area due to the religious and cultural mores of the farm population.

A number of problems must be addressed to solve southeastern Pennsylvania's agricultural pollution problems. First, the level of pollution reductions necessary to meet water quality goals must be identified for critical water quality parameters. Second, those watersheds which have pollution problems, and the critical areas within the watersheds, should be identified. The most cost-efficient combination of practices should then be identified and selected to meet the pollution reductions. Third, implementation policies are required that will achieve the necessary farmer participation to efficiently and equitably obtain the desired pollution reductions. Finally, the trend toward increasing the intensity of land use should be constrained. Future expansions of animal operations should be conditional on the adoption of adequate nutrient control technologies. Gains made through better farm management could be negated if cropping intensity and animal intensity continue to increase without these controls.

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BEST MANAGEMENT PRACTICES FOR ANIMAL PRODUCTION

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ABSTRACT

The environmental effects of grazing animals and how to control these are presented. In general, pollutant yields from pasture and rangeland operations are very low. They are not directly related to the number of animals or amounts of wastes involved. Rather, they are intimately related to hydrogeological and management factors and are best described as the results of the erosion/sediment phenomenon. Control is related to following practices that are consistent with optimal forage production on a long-term, sustaining bases and regulating animal movement/congregational patterns to minimize development of highly impacted areas.

Keywords: Animal wastes, Pasture, Water pollution, Agricultural wastes, Agricultural land runoff, Nonpoint pollution, Rangeland, Livestock, Animal husbandry

INTRODUCTION

This report summarizes state-of-the-art knowledge related to the yield and control of water pollutants from animal production systems. Using findings from recently completed research projects, this report updates earlier work by the author that emphasized the character of the problem (Robbins, 1979). Herein, emphasis is on control techniques or best management practices (BMP's) aimed at keeping wastes from reaching surface waters.

By way of orientation, the difference between confined and unconfined animal production is that wastes generated in confined systems are subject to handling/diverting for conventional control or treatment, while those in unconfined systems cannot be handled and, thus, must be controlled through the management/operation scheme. Correspondingly, the contrast between confined and unconfined animal production systems is very similar to the differentiation between point and nonpoint sources of pollution, where a nonpoint source is one whose specific point of generation and exact point of entry into the environment cannot be defined. All grazing systems--where livestock have free access to pasture, range, woodland, or cropland and utilize the associated forage/residue as a major feed source--are unconfined systems.

About 40% or 360 x 10^6 ha of the land area of the United States is used for grazing livestock. Forages, which in the main are grazed, account for the production of more than 50% of the nation's milk, nearly 80% of the total feed for beef cattle, and 90% of the total feed for sheep. Less than 5% of swine feed and almost no poultry feed comes from grazing.

While approximately 50% of all livestock wastes in the United States are estimated to be produced in confinement, about 75% of cattle, 85% of sheep, and 10% of hogs are in unconfined systems at any given time. Table 1 lists the numbers of animal production systems and animals maintained in unconfined production systems. The vast numbers of individual units involved in unconfined livestock activities are reflected by the values given for numbers of production units (about one million). Although specialization, labor reduction, and improved efficiency have caused a trend toward confined production systems, unconfined production is expected to continue to predominate the beef and sheep industries.

IMPACT ON PASTURE AND RANGELAND

As grazing animals traverse pasture and rangeland, the stresses applied to the soil beneath their hooves often exceed the strength of the soil. Trampling may cause poorer water infiltration. Reductions in infiltration rates due to grazing correlate with increases in runoff as indicated by numerous studies. For example, White et al. (1982) found that where winter feeding of cattle occurred, the runoff volume was three times that of normal. Winter rotational grazing increased runoff by 75% and summer rotational grazing only, increased runoff by 17%. The increased runoff volumes were primarily associated with the dormat season, winter feeding and pasturing. Trimble et al. (1951) measured average infiltration rates of 52 and 11 cm/h on ungrazed and grazed pastures, respectively. In a New Mexico study of grazing intensity-infiltration relationships, infiltration rates were determined to be 10.5, 5.5, and 2.1 cm/h on undergrazed, overgrazed, and depleted ranges, respectively (Flory 1936).

TABLE 1. DISTRIBUTION OF LIVESTOCK (After Robbins, 1979)

Type Animal	No. of Production Units	No. of Animals	No. Confined	No. Un-Confined	Percent Unconfined
		Thousands			
Cattle and Calves		123 000	30 000	93 000	76
Beef Cattle	490	101 500	20 000	81 500	80
Dairy Cattle	220	21 000	10 000	11 000	52
Sheep and Lambs	50	12 700	2 000	10 700	84
Goats and Kids	10	1 300	200	1 100	85
Hogs and Pigs	260	55 000	50 000	5 000	9
Horses and Ponies	--	8 000	--	--	--

After a comprehensive review of over a dozen studies, Moore et al. (1979) were able to summarize that light and moderate intensities of grazing during unsaturated soil conditions will not decrease infiltration rates. However, a concentration of livestock is likely to produce significant decreases in infiltration. Adverse impacts on sheet and rill erosion and water quality constituents are an outcome of these changes. The phenomena of heavy trampling damage to infiltration rates may be an insignificant problem on fine-textured high clay content soils where undisturbed rates are less than 25 cm per hour. It is well documented that periods of non-grazing, however built into the grazing system, allow certain natural processes to progressively improve soil water infiltration rates to predisturbance levels.

A decrease in infiltration and increase in runoff on grazed lands may lead to more arid conditions than normal. For example, Hanson et al. (1970) found that the reduction in available moisture on heavily grazed South Dakota ranges averaged 2 cm a year, or about 8% of the annual precipitation available for plant production. A decrease in moisture available for forage production is usually accompanied by an increase in soil erosion.

Selective defoliation by grazing animals decreases the preferred forage species and allows the proliferation of less desirable species. Close grazing of desired plant species during critical stages of growth, such as before seed maturation, hinders their competitive capacity. Reduction of the leaf area of forages by close grazing reduces their photosynthetic abilities and may affect plant succession (Stoddart et al., 1975). Heady (1975) attributes the most prevalent cause of rangeland retrogression to overgrazing and other faulty management of livestock operations.

The general congregational and waste elimination behavior patterns of livestock follow ancestral instincts, but may be influenced by management practices. Behavior patterns and preferences of livestock species tend to result in certain areas showing intense environmental stresses. These "problem areas" are usually only a small fraction of the total unconfined production site and are caused by unregulated animal behavior that leads to selective--and over--grazing (of sites as well as plant species) and to animal congregation and waste elimination in or near water sources or other critical areas. While only a small portion of the total production area, these problem areas often contribute the major portion of pollutants and constitute the only sources that when controlled results in a discernable improvement in runoff quality.

BACKGROUND LEVELS OF POLLUTANTS

In assessing the environmental impact of unconfined animal production, the concept of "background (natural) levels" of pollutants is important. From a study of 12 agricultural watersheds, Robbins et al. (1971) found that distinguishing between pollutants from farm animal production units and natural pollutants in receiving streams is difficult or impossible. More recent studies designed to establish the contributions of pollutants from unconfined animal production to surface waters by Moore et al. (1979), Doran et al. (1982), White et al. (1982), Powell et al. (1982), and Saxton et al. (1982) all reemphasize that control of pollutants from unconfined animal production units may be to no avail unless other pollutant sources that naturally occur in the watershed are controlled as well. When management is directed to

optimizing forage production, pollutant yields from unconfined animal production systems are not more than would occur under native conditions. Only problem areas as noted earlier and areas used more to confine animals rather than to graze them would seem to be subject to corrective actions.

EROSION/SEDIMENT

Sediment is both a pollutant and a carrier of pollutants. As a first approximation, factors that govern erosion and sediment yields are the same factors that control pollutant yields from unconfined animal production systems. Numerous studies including Robbins et al. (1978), Smeins (1976), Moore et al. (1979), Saxon et al. (1982), White el al. (1982), and Powell et al. (1982) have pointed to the need for good soil and water conservation practices to minimize the movement of pollutants from animal production units into streams. Excellent, detailed and thorough reviews and discussions of erosion, erosion models, and sediment yields as related to agricultural non-point sources have been prepared by USEPA (1973), Stewart et al. (1975), and Sweeten and Reddell (1978). Sediment yield to streams and lakes exceeds 2 t $ha^{-1}yr^{-1}$ on the average. On-site erosion is estimated to be twice this value for an average of more than 4 t $ha^{-1}yr^{-1}$ or a total of 3.6 x 10^9 t/yr for the total U.S. land area. According to Froehlich (1976), the smallest levels of sediment yield are from certain undisturbed forestlands in the Rocky Mountains where sediment reaching high elevation streams may range up to 0.36 t $ha^{-1}yr^{-1}$. Other examples of background sediment yields are 0.59, 0.90, and 1.10 t $ha^{-1}yr^{-1}$ from three neighboring watersheds in western Oregon. Well stocked southern pine forests my yield 0.7-1.1 t $ha^{-1}yr^{-1}$. In southwestern forests with low precipitation, yet high intensity storms, sediment yields average 1.1-1.4 t $ha^{-1}yr^{-1}$.

Background sediment yields may be compared with yields from cropland. Cropland has been credited with responsibility for 50% or 0.9 x 10^9 t/yr of the 1.8 x 10^9 t/yr total sediment delivered to U.S. streams and lakes. As noted by Sweeten and Reddell (1978), 70% of the nation's cropland yields more than 6.7 t $ha^{-1}yr^{-1}$. Representative values of on-site erosion from several sources as reported by USEPA (1973) are given in Table 2. Here, grassland includes pasture and rangeland. While the erosion rate from grassland is 10 times that from forestland, it is considerably less than the average rate of 4 t $ha^{-1}yr^{-1}$ for all land; and it represents the average background or natural rate for grassland ecosystems.

Sediment yields are very erratic. The largest annual sediment loads for a given stream are often 20 times greater than the smallest sediment load and generally can be correlated with those years of greatest runoff. Large differences in sediment yield can exist on adjacent streams discharging at the same rate. And, even in the same stream, suspended sediment concentration can vary tenfold at a given discharge rate, depending on many factors (Froehlich, 1976).

Most sediment from a watershed may come from a relatively few small areas needing corrective attention. Stewart et al. (1975) listed conditions indicative of high sediment yield potential that usually can be identified by observation. Overgrazing and resultant loss of groundcover can increase greatly the erodibility of pasture and rangeland.

TABLE 2. REPRESENTATIVE RATES OF ON-SITE EROSION FROM VARIOUS LAND USES (USEPA 1973)

Land Use	Rate, t/ha yr	Relative Rate, Forest = 1	Total, t/yr	Relative Total Forest = 1
Forest	0.085	1	16.8	1
Grassland	0.85	10	185	11
Cropland	17	200	2840	168
Harvested Forest	42	500	187	11
Construction	170	2000	100	6

Results from four test sites that exemplify water quality effects resulting from suspected problem sites and unimpacted sites are given in Table 3. These results vividly emphasize the need to consider problem areas separately from the remainder of the unconfined production system and to design corrective actions to effectively control the impact of these. Yields of pollutants from heavily stocked pastures are exemplified in Table 4, along with that from other land uses. As indicated, heavily stocked pastureland contributes less than cultivated fields but more than permanent grasslands. Thus, improved pasture management may be called for to control the pollutants from such heavily used areas. The pollutant levels from other than problem areas and heavily stocked pastures are seldom discernible from background levels and, on an areal basis, are of the same magnitude as yields to land from rainwater and yields to surface waters from undisturbed lands.

Prediction of increases in pollutant yields to receiving waters due to unconfined animal production is not possible with existing technology. As measured in receiving streams, these increases usually are not only small in value but extremely erratic as well. As a first approximation, models for erosion/sediment yields may be adopted/used to predict pollutant yields from unconfined livestock systems.

BACTERIAL COUNTS

Bacterial indicators are reputed to be the most sensitive index of water quality impacts resulting from unconfined livestock. But the actual significance or meaning of elevated bacterial counts in this case is not clear. While the yield from pastured areas may be sometimes several fold that from control areas (White et al., 1982), the opposite is also not uncommon (Saxton et al., 1982). Their use in evaluating the effect of unconfined livestock production on runoff is certainly difficult, if not impossible. A growing consensus among researchers is that their use in evaluating nonpoint pollution is inappropriate. These organisms are properly used to designate the bacteriological safety of potable water. Their presence in water is accepted by public health authorities as an indication of fecal pollution. Whenever water is polluted with fecal material, it poses a potential health hazard if used untreated for drinking or body contact recreational purposes. There is a dearth of documented cases of health problems correlated with bacterial indicator counts in water related to unconfined animal production systems, but this may be due solely to the lack of intensive and systematic research in this area.

CONTROLS

While certain specific questions such as desired level of control and best methods to achieve water quality goals remain unresolved, the basic concepts, methods, processes, and procedures needed to reduce pollutant yields from unconfined animal production systems are available in principle. Since the pollutants are nonpoint in origin, control methodology is effected through changes in the production management system; i.e., by incorporating practices involved in controlled grazing, regulated animal congregation patterns, sustained forage production (establishment and protection of desired species), intensive erosion control, and proper land use. This may, under extremely adverse site conditions, call for complete exclusion of livestock.

TABLE 3. WATER QUALITY BY LAND USE (SEWELL AND ALPHIN 1972)

Land Use	BOD_5	DO	NO_3-N	PO_4-P	Total Coliform, cts/100 ml
	ppm				
Ungrazed Woodland	2.5	8.6	0.05	0.05	1500
Heavily-Grazed Pasture	13.8	6.1	4.5	7.1	330000
Farm Pond in Pasture	10.0	7.9	0.20	0.05	600
Farm Pond in Woodland	3.1	7.6	0.12	0.02	1500

TABLE 4. POLLUTANT YIELD BY LAND USE (DORNBUSH ET AL. 1974)

Pollutant	Yield, kg $ha^{-1}yr^{-1}$		
	Cultivated	Pasture	Grassland
Total Residue	334	58.2	32.4
Suspended Solids	286	11.8	4
Total Phosphorus	0.30	0.25	0.1
Nitrate-Nitrogen	0.37	0.40	0.24
Total Kjeldahl Nitrogen	0.91	1.12	0.73
Chemical Oxygen Demand	48	28	13

Pasture and rangeland management practices leading to optimal forage production on a long-term sustaining basis, such as that resulting from maintaining proper stocking rates and forage production practices with erosion control, are also those practices which minimize environmental impacts. On the other hand, when exploitive (poorly managed) production is followed, unnecessary detrimental effects on surface water quality are likely to occur. These systems may require major modification/adaptation to meet environmental goals as well as to achieve other objectives which govern pasture and rangeland use.

As detailed by Robbins (1978), when developing or modifying management practices for unconfined animal production systems, consideration should be given to the following concepts:

1. Multidisciplinary and multiagency review teams should be used to evaluate specific environmental problems resulting from unconfined animal production and to recommend changes in management practices to correct the problems. Their recommendations should be applied in concert with established regulatory procedures.

2. Regulatory programs should account for local (site specific) conditions. Regulated changes in management practices should be restricted to those documented to have measurable water quality benefits for receiving waters. Due to the low level of pollution associated with unconfined animal production, regulatory programs that would in general discourage or restrict livestock production on pasture and rangeland should be avoided.

3. Install and maintain an effective and complete program of soil erosion control.

4. Follow stocking rates and controlled utilization of forages (e.g., rotation, deferred, and seasonal grazing) that reduce erosion and minimize waste accumulation. Reduce stocking rates in problem areas and at critical times or seasons. Stocking rates and grazing programs should be tailored to the soil, vegetation, topography, hydrogeology, and microclimate of the particular site.

5. Avoid animal stocking rates and other practices that create holding areas rather than grazing areas. Promote necessary animal congregation in areas that are hydrologically remote from streams and other major drainage channels. Periodically move bedground, shelter, salting, feeding, and/or watering areas to control waste accumulation, soil compaction, and erodible paths and areas.

6. Maintain to the extent ecologically feasible, highly productive forage and dense ground cover on the land to decrease volume and rate of runoff, to entrap and hold animal wastes, to utilize fertilizer nutrients, and to prevent erosion. Increase herbaceous cover in proximity of stream banks, downslope from animal congregational areas, and on other critical areas.

7. Where the number of animals per unit area or the characteristics of the site present pollution problems, appropriate drastic management alternative/practices may include:

(a) Restrict animal access to critical areas. Use fencing to prevent livestock from entering highly erodible areas and critical stream or pond reaches and to prevent animals from wading in water. Provide summer shade (trees or artificial shelters) and insecticides to lessen the need for animals to enter water for relief from heat and insects. Block erodible paths with physical barriers and revegetate eroded paths. Move drinking facilities outside critical areas.

(b) Increase rate of fecal degradation and incorporation. Modify feed formulation and/or texture. Use tillage to break up, manipulate, and incorporate wastes in particular problem areas.

(c) Use land forming and diversions to modify drainage patterns.

Total stream fencing and other drastic controls will usually be both unnecessary and impractical except for a few problem areas within an operation.

CLOSING REMARKS

One major challenge still remaining is to demonstrate cost-effective routes toward achievement of various levels of pollution control for unconfined animal production systems. Whether resources should be used to reduce pollutant yields from a given production system really requires an on-site evaluation of whether the pollutants have a real adverse effect on the quality of the receiving water and whether the yields can be significantly reduced. In view of the many desirable characteristics of unconfined animal production compared with the problems associated with alternate livestock production techniques, adequate research, management, and educational resources should be allocated to assessing the importance and significance of increased levels of pollutants (indicator bacteria, plant nutrients, and oxygen demands) in surface waters (streams and impoundments), with consideration given to fate of the pollutants and uses of the waters. In cases where pollutants from livestock activity are not discernible from background levels, little or no change may be noted in stream quality even when significant efforts and resources are expended on control. When grazing is approached using well-planned management practices that include pollution control as an integral part, rather than a laissez-faire approach, the impact of unconfined animals will be well within acceptable limits and in keeping with water quality goals.

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COMPARATIVE POLLUTANT REMOVAL CAPABILITY, ECONOMICS AND PHYSICAL SUITABILITY OF URBAN BEST MANAGEMENT PRACTICES IN THE WASHINGTON D.C. METROPOLITAN AREA

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ABSTRACT

Recent field studies have assessed the comparative pollutant removal capability of six kinds of urban stormwater best management practices (BMPs) in the Washington metropolitan area. While some urban BMPs were found to be very effective in removing pollutants entrained in urban runoff (wet ponds, extended detention, porous pavement), the most frequently used stormwater BMPs, dry ponds and grassed swales, exhibited little or no pollutant removal capability. Local governments have recently sought to incorporate water quality objectives into existing stormwater management programs by encouraging BMPs with high pollutant removal capability. This strategy has not led to widespread implementation because of uncertainties about the physical, economic, and environmental constraints of these BMPs at the site level.

A systematic approach for screening urban BMP options is presented, based on a comparative review of the suitability of BMPs with respect to pollutant removal, cost, and site/environmental suitability for nine hypothetical development scenarios. A three level screening matrix is developed which provides rapid guidance to site planners on the suitability of seven urban BMP options. Major elements evaulated in the matrix include site restrictions such as soils, slope and site area, space requirements, pollutant removal, local and downstream habitat modification, landscaping and aesthetic value, total cost, and operation and maintenance requirements. The use of the screening matrix for developing effective, low cost urban nonpoint source control strategies is also discussed.

INTRODUCTION

Stormwater management programs have been in effect in the Washington, D.C. area since the early 1970s with the objective of controlling increases in post-development peak runoff rates. To date, nearly 3,500 structures have been constructed. At least 75% of all new regional development is now serviced by some kind of stormwater control, with dry ponds being applied in about two-thirds of all development situations (MWCOG,1984a).

As more information has become available on the impact of urban runoff loadings on important area receiving waters, Washington area local governments have actively sought to incorporate nonpoint pollutant removal into their existing stormwater management programs, chiefly through the use of alternative urban "Best Management Practices" (BMPs). These alternatives rely on either greater retention of stormwater (wet ponds, extended detention ponds), or enhanced infiltration (porous pavement, grassed swales, infiltration basins and trenches).

Despite strong local institutional and legislative commitments to improve urban stormwater quality, less than 20 percent of all stormwater control structures installed or designed during 1982 and 1983 had significant pollutant removal capability. (MWCOG,1984a) At the site planning level, the most frequently cited reason for the lack of widespread implementation has been uncertainty about the cost, efficiency and suitability of alternative BMP options.

In 1983, the Metropolitan Washington Council of Governments completed a four year BMP monitoring and cost survey under the EPA Nationwide Urban Runoff Program (NURP). The objective of the research was to provide a comparative assessment of the relative suitability of selected urban BMPs for nonpoint source control. Subsequent research has examined the degree to which urban BMPs can be applied over the range of site conditions encountered in the region. This paper summarizes the major findings of these efforts, and presents a systematic screening procedure for evaluating alternative BMP options at the development site level.

URBAN BMP ASSESSMENT RESEARCH

BMP Pollutant Removal Field Studies.

The pollutant removal capability of six different urban BMPs was monitored over a 12 to 18 month period in suburban Maryland and Virginia. BMP monitoring sites included dry ponds (1), extended detention dry ponds (1), wet ponds (3), grassed swales (3), infiltration pits and trenches (2) and porous pavement (1). Flow-activated automated water quality samplers were installed at the inlets and outlets of 11 BMP sites, which drained small (3 to 90 acre), single land use watersheds. Flow weighted composite samples of nutrients, sediments, COD and trace metals were collected for 20 to 50 storm events at each site. Pollutant removal efficiency was calculated in terms of total mass outflow load/total mass inflow load for volume BMPs, and outflow median loading rate/inflow median loading rate for infiltration BMPs. More detailed discussion of monitoring and data analysis methods employed in the study can be found in the final project report (MWCOG,1983a)

BMP Cost Surveys

Construction costs for pond BMPs were evaluated in an extensive local cost survey (MWCOG,1983b). Construction components and requirements were identified for 31 representative SWM ponds in the Washington metropolitan area (e.g. cubic yards of excavation, linear feet of pipe etc.). In-place unit cost estimates for each pond component or construction requirement were then independently solicited from 12 public and private construction

and engineering firms. By applying the unit cost estimates to each of the 31 ponds, 372 separate estimates of construction cost were generated. Subsequent statistical analysis of the dataset yielded a strong (r^2 =0.84), curvilinear relationship between pond storage volume and construction cost (Table 1).

No systematic cost surveys of infiltration BMPs could be developed since very few have been installed in the region. As an indirect approach, the basic geometry of infiltration BMPs was inferred from design standards for each site (MD DNR,1984). Construction requirements for each site were then estimated based on site geometry, and construction costs were estimated by multiplying construction requirements by unit cost data obtained in the pond cost study (Table 1).

Estimates of average O&M costs for pond BMPs were obtained in a survey of O&M budgets and component cost records of six public works agencies (MWCOG,1983b). Typical O&M activities reported included inspection, grass mowing, debris removal, bank stabilization, structural repair, nuisance control and sediment removal. Analysis of O&M budgets suggested that regular performance of these tasks would entail an annual cost equivalent to 3 to 5% of pond BMP construction cost. O&M requirements for infiltration BMPs are generally different in scope and frequency; the best available methods for evaluating these costs are shown below.

Table 1. METHODS USED TO CALCULATE COSTS FOR SELECTED URBAN BMPs IN THE WASHINGTON DC METROPOLITAN AREA

BMP TYPE	CONSTRUCTION COST (C)	CONTINGENCY COST (CC)	O & M COST (annual)	COMMENTS/NOTES
DRY POND	82.51Vs**0.51	C * 0.25	C * 0.05	Equations derived in Washington DC Stormwater Pond Cost Study (MWCOG,1983).
EXTENDED DETENTION DRY POND	82.51Vs**0.51	C * 0.25	C * 0.05	O&M Costs were computed assuming a conservative value of 5% Contingency costs are defined as costs incurred in planning, designing and overseeing construction of BMP
WET POND	82.51Vs**0.51	C * 0.25	C * 0.05	Vs = volume of pond up to the crest of the emergency spillway (in cubic feet)
INFILTRATION BASIN	66.23Vs**0.51	C * 0.25	C * 0.05	Equation derived by factoring out the costs of outlet works and structures from the pond cost equation. O&M requirements assumed to be similar to ponds.
INFILTRATION TRENCH	Construction Requirements Determined by Design Geometry	C * 0.25	$286 * SA of trench + Buffer SA in acres	Construction requirements include: excavation, fill of coarse stone or gravel, filter cloth, test wells, and seeding/mulching for buffer strip O&M based on cost/maintained acre in (MWCOG,1983)
POROUS PAVEMENT (PP#1) Not Space Limited	Incremental Cost over Conventional Pavement	C * 0.35 includes extra cost for addl. site surveys soil testing inspection	$0.003 * SA of PP#1 (commercial rate for vacuum street sweeping 4 times/yr	Incremental costs include: higher unit costs of porous asphalt relative to conventional asphalt, extra costs for filter cloth, and test wells. SA for PP#1 = 60 to 70% of total site area
POROUS PAVEMENT (PP#2) Space Limited	As above,+ cost of extra reservoir base under asphalt for runoff storage	C * 0.35	$0.003 * SA of PP#2 site (SA in sq.ft)	Depth of extra base course computed from State of Maryland Design Specifications. Associated extra cost for excavation and coarse stone fill derived from unit cost data contained in pond cost study. SA for PP#2 = impervious site area/2 (13 to 43% of total site area)
GRASSED ROADSIDE SWALES	Not Calculated	Not Calculated	Not Calculated	Grassed roadside swales generally assumed to be less expensive than the curb and gutter conveyance systems they replace.

TOTAL BMP COST is defined (C + CC) + OMpv; where OMpv is the present value of all future O&M payments, assuming a twenty year project life and an 8% discount rate. Land costs were assumed to be zero for all BMPs.

ANNUAL BMP PAYMENT was computed assuming a twenty year loan at 8% interest rate

Physical and Environmental Suitability Surveys

Physical or environmental conditions which might restrict the application of a particular BMP were evaluated in a review of local and state BMP design manuals and other relevant literature. Approximately thirty site planners, engineers and public works officials involved in BMP construction were also interviewed to obtain their perceptions and experiences about the limitations and attributes associated with each BMP option.

Development of BMP Screening Model

To arrive at a comparative assessment of BMP options at the site level, nine hypothetical development site scenarios were examined in detail. These scenarios included single family residential (2 DU/Ac--25% impervious), townhouse (8 DU/Ac--65% impervious), and commercial development (85% impervious) situated on 1, 10, and 25 acre sites. The TR-55 tabular method (SCS,1975) was used to determine the increase in post-development peak discharge and runoff volume associated with the two year design storm at each site. Curve Numbers (CN) of 70, 85 and 92 were assumed for the single family residential, townhouse, and commercial developments, respectively. All sites were assumed to have soils in hydrological group "B", and slopes of 3-5%. The stormwater management criteria for all sites was to maintain post-development peak discharges (pond BMPs) or increases in upland runoff volumes (infiltration BMPs) to pre-development levels for the two-year storm (defined as meadow,CN 58). Required runoff storage volume was determined for each BMP by either the short-cut routing method (pond BMPs) or State of Maryland infiltration BMP design standards (MD DNR, 1984).

The total cost of installing and maintaining each BMP on the nine development sites was computed using the methods outlined in Table 1. Physical constraints, such as space requirements, were evaluated assuming a standard geometrical design for each site. Environmental factors were assigned to each BMP site based on the consensus among site planners and engineers surveyed.

RESULTS OF THE COMPARATIVE URBAN BMP ASSESSMENT

Comparative Pollutant Removal

Long term pollutant removal observed at the BMP sites monitored is summarized in Table 2. As can be seen, the degree and kind of pollutant removal achieved varied widely among the BMPs studied. For example, the dry pond investigated was not effective in removing sediments, nutrients, trace metals or oxygen-demanding materials. Urban stormwaters passed through the dry pond too rapidly to achieve significant removal via settling. During some storms, the dry pond actually became a net source of pollutants, as pollutants were either scoured or resuspended from the pond bed.

Extending detention time in dry ponds by 6 to 12 hours proved to be effective in removing sediment, organic nutrients and trace metals. Removal of biologically available nutrients and other soluble pollutants, however, was negligible. Wet ponds were found to be the most effective pond BMP

studied. Removal of sediment, trace metals and organic nutrients was comparable to rates observed at the extended detention pond. In addition, biological processes occurring within the pond effectively removed the soluble nutrients most readily utilized in algal growth.

No statistically significant improvement in runoff quality could be attributed to grassed swales when compared to traditional curb and gutter conveyance systems. The poor performance of swales appeared to be due to the characteristics of the sites monitored--moderate slopes (4 to 6%), relatively permeable soils, and short grass. It is possible that better removal might have been achieved by swales with check dams on sandy, relatively flat sites. However, these conditions are not common in the Washington metropolitan area.

Monitoring difficulties and site limitations prevented a valid statistical assessment of pollutant removal by infiltration trenches. Trench BMPs did appear to be capable of removing sediments and trace metals, but did not show any capability for removing nutrients. It should be noted that the trenches monitored had outlet structures, and thus did not function as "true" infiltration BMPs (which infiltrate incoming runoff completely into the soil profile). If complete infiltration is accomplished, as is currently recommended, it is likely that infiltration BMPs would exhibit much higher removal efficiencies. This, in fact, was the case at the porous pavement site monitored. Although the site did have perforated underdrains (and therefore could be monitored), significant infiltration of runoff into the underlying soil was achieved. As a result, removal of most pollutants at the porous pavement site approached 90%.

Table 2. COMPARATIVE POLLUTANT REMOVAL OF URBAN BEST MANAGEMENT PRACTICES IN THE WASHINGTON D.C. METROPOLITAN AREA NURP FIELD MONITORING STUDY

	DRY POND	EXTENDED DETENTION DRY POND	WET POND	INFILTRATION BASIN	INFILTRATION TRENCH	POROUS PAVEMENT	GRASSED SWALES
PARAMETER	<------- PERCENT REMOVAL OF POLLUTANTS ---------->						
SEDIMENT	14	64	55	--	50	>95	NSD
TOTAL PHOSPHORUS	20	10	66	--	60	60	NSD
ORTHO-PHOSPHORUS	0	1	84	--	--	>50	< 0
TOTAL NITROGEN	10	24	28	--	-8	88	NSD
NITRATE & NITRITE	9	10	60	--	--	>70	NSD
TKN	11	15	15	--	--	>90	NSD
EXTRACTABLE LEAD (Pb)	-5	84	65	--	--	>95	NSD
EXTRACTABLE ZINC (Zn)	-10	57	51	--	48	>99	< 0
COD	0	40	28	--	37	83	< 0

Relative Cost of Urban BMPs

Figure 1 displays the computed total costs associated with seven urban BMP options for several land use and site area combinations. In general, conventional dry ponds represent the least cost BMP investment. Extended detention ponds involved an additional cost of 5-10% at most sites, whereas wet ponds and infiltration basins were 25 to 45% more expensive, depending on land use and area serviced. Infiltration trenches cost approximately 50% more than dry ponds on small residential sites, but were clearly uneconomical on larger or more intensively developed sites. Porous pavement was generally competitive with pond BMPs when space requirements were not limited. When available space was limited, however, porous pavement was only economical for small developments sites.

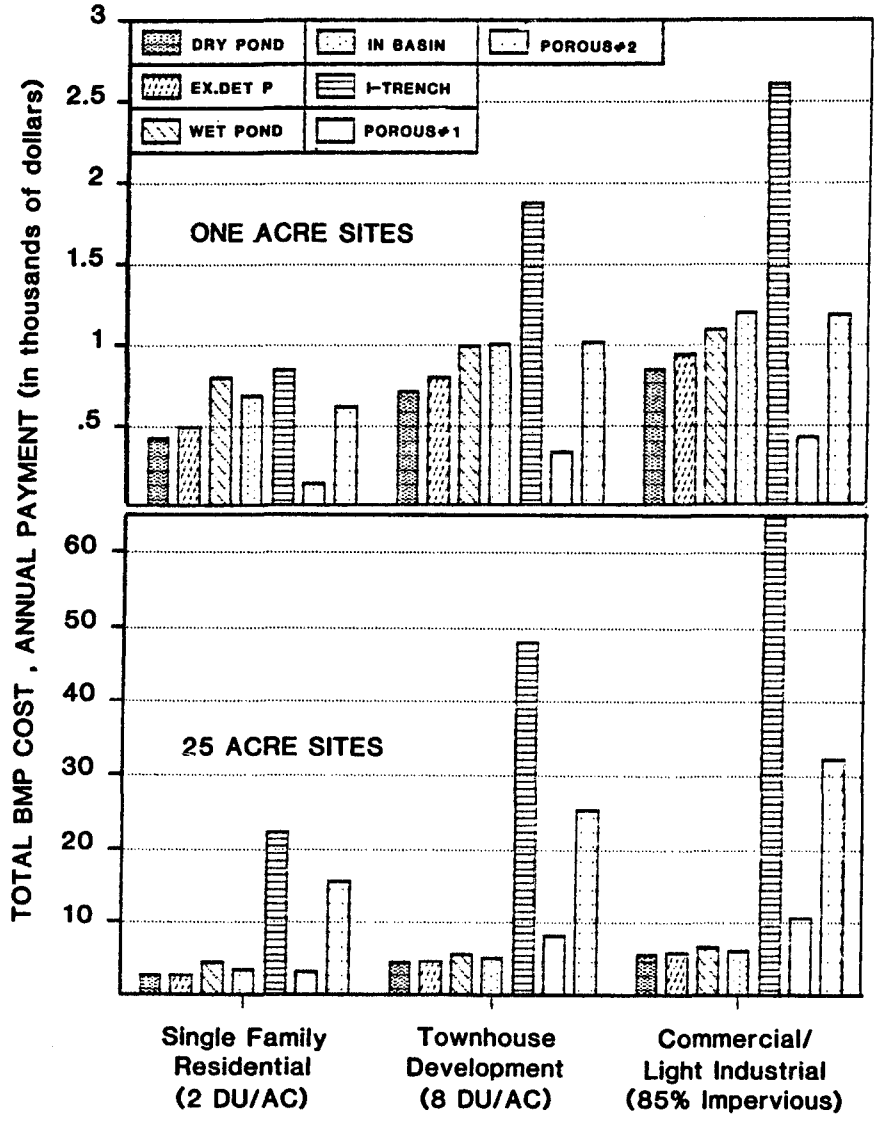

Figure 1. COMPARATIVE TOTAL COST OF SEVEN DIFFERENT URBAN BMP OPTIONS IN THE WASHINGTON METROPOLITAN AREA, 1984 DOLLARS. Top Panel: Cost Estimates for One Acre Sites. Bottom Panel: Cost Estimates for 25 Acre Sites. Total Cost Defined as Construction Cost + Contingenency Cost + Annual O&M Amortized at 8% for 20 years + Financing Cost.

The higher cost of infiltration BMPs can be attributed to two factors. First, unlike pond BMPs which only control the increase in peak discharge, infiltration BMPs must have sufficient volume to accommodate the entire increase in upland runoff volume. As a result, infiltration BMPs generally have greater excavation requirements because of the larger storage volume needed. In the case of infiltration trenches and porous pavement, even greater storage volumes are required to allow for non-void space in the gravel or coarse stone filter. The cost of filter materials can also be a major factor which drives up infiltration trench and porous pavement construction costs.

One advantage of infiltration BMPs is their lower cost for annual operations and maintenance, relative to pond BMPs. Because of their inherent design, many common pond O&M tasks, such as structural repair, and sediment and debris removal, are either unnecessary or impossible to perform on infiltration BMPs. Porous pavement, for example, was found to be cheaper to maintain than any other BMP on commercial sites. Similarly, infiltration trench maintenance typically was less than one-third as expensive as annual O&M for dry ponds on residential sites. As might be expected, dry ponds typically involved the lowest O&M cost for pond BMPs, ranging from $6 to $60/DU/yr on residential sites, and $62 to $232/Acre/yr on commercial sites. Extended detention pond O&M involved a modest additional O&M cost of 5 to 7%, whereas wet ponds and infiltration basins involved 16 to 70% more O&M cost when compared to dry ponds.

Physical Suitability of BMPs

The most frequently cited physical constraints to urban BMP application which emerged in the suitability surveys included topography, soil permeability, space requirements and site size. In general, infiltration BMPs were found to be subject to the greatest number of site restrictions. Current State of Maryland Infiltration Standards prohibit their use on sites with soil profiles of low to moderate permeability ("C" and "D" soils), slopes greater than 5%, and a depth to bedrock or groundwater of less than 2 feet. Also, infiltration trench and porous pavement BMPs may not be feasible on larger, more intensively developed sites, because of their significant space requirements (which may often exceed 25% of the total site area). However, the greater space requirements of porous pavement may not be restrictive on some sites, particularly if significant portions of the site are devoted to parking.

Application of extended detention ponds in the region is normally limited to sites with a minimum size of about 20 acres. In smaller sites, required orifice size in extended detention ponds becomes so small (<2 inches) that the outlet is susceptible to chronic clogging. Wet ponds can usually only be applied to development sites in excess of 10 acres, as a minimum watershed size is needed to maintain a permanent pool of water during extended periods of dry weather. Although pond drawdown does not adversely affect a wet pond's runoff control or pollutant removal properties, it may reduce or eliminate desired habitat, aesthetic or landscape benefits.

Environmental Considerations in BMP Selection

Natural Environment

As urban BMPs nearly always represent a modification to the urban landscape, it is often necessary to select or design BMPs to minimize their impacts on the environment. Recent research has shown, for example, that BMPs can have a substantial influence on the quality of wildlife habitat. A field survey conducted in Columbia, Md. (Adams et al, 1983) suggested that dry ponds had little habitat value. The same study also concluded that wet ponds enhanced both aquatic and terrestrial habitat value. A variety of waterfowl, marsh and shore birds, and terrestrial wildlife utilized wet ponds, presumably attracted by the food and cover supplied by aquatic vegetation. With proper design, it is likely that extended detention ponds can also function to create valuable wetland habitat. Infiltration trench and porous pavement probably have a negative influence on local habitat, since they consume more area than pond BMPs, and are normally unvegetated.

Downstream habitat is also influenced by urban BMPs. For example, infiltration BMPs often can sustain downstream flows during extremely dry periods because of their superior groundwater recharge, thus reducing seasonal stress on downstream aquatic life. Infiltration BMPs and extended detention ponds also function to reduce the loss of riparian habitat caused by downstream bank erosion (since these BMPs attenuate peak flows to a much greater degree than either wet ponds or dry ponds). Some concerns have been expressed that under some conditions wet ponds may produce thermal discharge which could impact trout fisheries in cold-water streams.

Human Environment

An often overlooked aspect of urban BMPs is their role in the human environment. Urban BMPs can function as a landscape amenity or as an unsightly and discordant community feature. Similarly, urban BMPs can support limited recreation use or be under-utilized and possibly dangerous empty space, depending on the type of BMP employed, and its subsequent O&M. As resident's perception of aesthetic value prominently figures in their acceptance of and support for BMPs, these considerations can be important, particularly if the same residents will be responsible for future maintenance.

In this respect, wet ponds appear to be superior to other BMPs, if properly designed and maintained. In a recent survey of resident attitudes towards stormwater ponds in Columbia, Md. (Adams et al,1983), 75% of residents surveyed preferred wet ponds over dry ponds. Most respondents felt that wet ponds provided a more pleasant and natural environment, as well more diverse recreational opportunities.

CONCLUSIONS

Comparative data on the cost, efficiency and physical/environmental feasibility of six urban BMPs has been integrated into a summary screening matrix shown in Table 3. The matrix is intended for use as a preliminary screening procedure for rapid evaluation of available urban BMP options at the site planning level.

Table 3. Generalized Screening Criteria Model For Selecting Urban Best Management Practices

	SCREENING CRITERIA	RESIDENTIAL BMPs (SINGLE FAMILY RESIDENTIAL → TOWNHOUSE)							COMMERCIAL BMPs					
		DP	XD	WP	IB	IT	PP	GS	DP	XD	WP	IB	IT	PP
PHYSICAL SUITABILITY OF SITE	**SITE SIZE**													
	1-5 ACRES	⊘	●	●	○	○	⊗	○	○	●	●	○	⊘	⊘
	10 ACRES	○	●	○	○	○	⊘	○	○	●	○	○	⊗	⊘
	> 25 ACRES	○	○	○	○	⊗	○	⊘	○	○	○	⊘	●	⊘
	SOILS													
	"A" SOILS	⊘	⊘	⊘	○	○	○	○	⊘	⊘	⊘	○	○	○
	"B" SOILS	○	○	○	⊘	⊘	⊘	⊘	○	○	⊘	⊘	⊘	⊘
	"C" or "D" SOILS	○	○	○	●	●	●	●	○	○	○	●	●	●
	SLOPE													
	0-5%	○	○	○	○	○	○	⊘	○	○	○	○	○	○
	6-20%	○	○	○	○	●	●	●	○	○	○	○	●	●
	>20%	○	○	○	●	●	●	●	○	○	○	●	●	●
	SPACE REQUIREMENTS													
	BMP AS % OF SITE AREA < 5 %	○	○	⊘	●	●	●	⊗	⊘	●	●	●	●	●
	BMP AS % OF SITE AREA 6-15%	—	—	○	○	○	●	⊘	○	○	○	⊘	●	●
	BMP AS % OF SITE AREA 16-25%	—	—	—	—	—	⊘	○	—	—	—	⊘	⊘	●
	BMP AS % OF SITE AREA 26%-75%	—	—	—	—	—	○	—	—	—	—	○	○	○
ENVIRONMENTAL FACTORS	**POLLUTANT REMOVAL**													
	NUTRIENTS	●	●	○	○	○	○	●	●	●	○	○	○	○
	ALGAL LIMITATION	●	●	○	○	○	○	●	●	●	○	○	○	○
	SEDIMENT	⊗	○	○	○	○	○	⊗	●	○	○	○	○	○
	TRACE METALS	●	○	○	○	○	○	⊗	●	○	○	○	○	○
	OXYGEN DEMAND	●	⊘	⊘	○	○	○	●	●	⊘	⊘	○	○	○
	OTHER FACTORS													
	GROUNDWATER RECHARGE	●	⊗	⊘	○	○	○	⊘	●	⊗	⊘	○	○	○
	WILDLIFE HABITAT	●	⊘	○	⊗	●	●	⊘	●	⊘	○	⊗	●	●
	RECREATION	⊘	●	○	⊘	●	●	⊗	⊘	●	○	⊘	●	●
	AESTHETICS/LANDSCAPE	⊗	⊘	○	⊘	⊘	⊗	○	⊗	⊘	○	⊘	⊗	⊗
	THERMAL DISCHARGE	○	⊘	●	○	○	○	○	○	⊘	●	○	○	○
	LOW FLOW MAINTENANCE	●	⊗	○	○	○	○	⊗	●	⊗	○	○	○	○
ECONOMIC CONSIDERATIONS	**TOTAL COST OF BMP***													
	within 10% of DP		○	●	●	●	●	○	—	○	●	●	●	⊗
	11 to 50% > DP		—	○	○	●	⊘	—	—	○	○	○	●	⊘
	51 to 100% > DP		—	—	—	●	⊘	—	—	—	—	—	●	⊘
	over 100% > DP		—	—	—	—	○	—	—	—	—	—	○	○
	O&M REQUIREMENTS*													
	less than DP		●	●	●	⊘	⊘	○	—	●	●	●	○	⊘
	0 to 10% > DP		○	●	●	○	○	—	—	○	●	●	—	⊘
	11 to 50% > DP		—	○	○	—	—	—	—	—	○	○	—	○
	51% > DP		—	—	—	—	—	—	—	—	—	—	—	—
	OTHER FACTORS													
	ECONOMIES OF SCALE	○	○	○	⊘	●	●	●	○	○	○	⊘	●	●
	LIABILITY/SAFETY	⊘	⊘	⊘	⊘	○	○	○	⊘	⊘	⊘	⊘	○	○

LEGEND			
○ APPROPRIATE	⊘ OFTEN APPROPRIATE W/DESIGN CHANGES	⊗ SELDOM APPROPRIATE	● NOT APPROPRIATE

NOTE: COMPARATIVE BMP COST DATA HAS BEEN COMPUTED USING DRY POND (DP) COSTS AS AN INDEX BASE

DP = DRY POND (DETENTION BASIN)
XD = EXTENDED DETENTION POND
WP = WET POND (RETENTION BASIN)
IB = INFILTRATION BASIN
IT = INFILTRATION TRENCH
PP = POROUS PAVEMENT
GS = GRASSED ROADSIDE SWALES

"A" SOILS = SANDS, LOAMY SANDS
"B" SOILS = SANDY LOAM, LOAMS
"C" SOILS = SILT LOAM, SANDY CLAY LOAMS
"D" SOILS = CLAY LOAMS TO CLAY

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Use of the Screening Matrix

As an example, suppose a site planner is reviewing a proposal for a ten acre shopping mall development situated on a site of moderate (>5%) slopes and relatively permeable "B" soils. Apart from controlling the peak discharge from the two year design storm, the planner is required by local regulations to install a BMP capable of effectively removing sediment and nutrients, and having, at a minimum, no adverse impact on the local environment. At the same time, the site planner wishes to minimize the costs and space requirements facing the builder.

Consulting the commercial development column in the screening matrix, the site planner can quickly eliminate the use of extended detention due to site constraints. Likewise, the dry pond and infiltration trench options can be rejected because of poor pollutant removal and prohibitive costs, respectively. If space is limited at the site, porous pavement can also be ruled out on the basis of large costs and space requirements. However, if the site will have a large area devoted to parking, porous pavement may be considered. Wet pond and infiltration basins are also indicated to be feasible BMP options. Both have similar cost and O&M requirements. However, the matrix indicates that wet ponds would have a smaller space requirement and greater overall habitat and amenity value. As this example illustrates, the screening matrix enables a site planner to select the best overall BMP option given diverse and often competing objectives.

Implications for Developing Effective Urban BMP Strategies

The matrix also offers several useful insights for developing low-cost, effective urban nonpoint source control strategies. For example, it is clearly evident that there are no single, "all-purpose" BMPs that can be applied over the entire range of site conditions. Rather, each urban BMP is only feasible and cost-effective in a rather narrow range of development situations. As a consequence, it would appear that stormwater management programs should be both flexible and site-specific, as broad prescriptive BMP policies will rarely be sufficient to cover the entire spectrum of development activities encountered in urbanizing areas.

For example, if high overall pollutant removal is a desired objective in a stormwater management program, wet ponds would appear to be the BMP of choice on most development sites greater than 10 acres in size. If nutrient control is not a major concern, then extended detention ponds may also be attractive on larger (20+ acre) sites. Infiltration BMPs generally provide the greatest overall mix of pollutant removal, cost and feasibility on smaller sites (1-10 acres). In particular, porous pavement appears most appropriate in small commercial development sites, whereas infiltration basins or trenches may be more suitable on small residential sites. When infiltration BMPs are not possible because of site constraints (a frequent occurrence in the Washington metropolitan area), there appear to be no other BMP options available which possess a significant nonpoint source control capability.

The screening matrix reveals that there are several other combinations of site conditions, environmental or cost restrictions for which no effective BMP can be applied for nonpoint source control. One implication is that

the existence of these situations could substantially curtail success in achieving basin-wide urban nonpoint source loading reductions. However, it is possible to achieve some measure of control at these sites by other methods. For example, some local jurisdictions have adopted off-site regional stormwater management programs which make it possible to pool urban runoff control needs from several small sites in a single, larger wet pond or extended detention pond further downstream. Developers who cannot build an effective BMP on-site are required contribute to the financing of the off-site facility.

A second approach might be to separate stormwater runoff control from water quality protection needs. Thus, low-cost detention BMPs could be used to control runoff from the two year design storm, while infiltration BMPs, designed to accommodate much smaller runoff volumes, could be applied to achieve pollutant removal for smaller and more frequent runoff events. Because of their reduced size, these infiltration BMPs would presumably be subject to fewer cost and site restrictions.

Future Directions

While the matrix can only evaluate a combination of nine general development scenarios at present, future research is being directed to develop an interactive software package capable of analyzing BMP options of any individual development site, given simple input parameters readily available from site plans. Concurrently, research is being conducted to better define the more uncertain elements of the matrix, particularly in regard to infiltration costs. By making better BMP screening tools available to site planners and developers, it is anticipated that wider implementation of effective urban BMPs can be achieved, and consequently, a greater degree of urban nonpoint source control can be attained.

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IMPLEMENTATION OF URBAN NONPOINT SOURCE WATER POLLUTION ABATEMENT PROGRAMS

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Abstract

Urban runoff management strategies have shifted from exclusive flooding and drainage concerns to an increased emphasis on water quality and comprehensive stormwater management. Recent studies, especially those conducted under the Nationwide Urban Runoff Program (NURP), have improved the technical basis for implementing urban nonpoint source pollution abatement programs. However, there are still questions regarding the practicality of providing adequate urban runoff pollutant control in existing urbanized areas. Implementation of an urban nonpoint source control program should ideally include the preparation of a stormwater management plan.

INTRODUCTION

Historically, most urban runoff concerns were related to flooding and drainage. The concept of the full-flowing storm drain designed by the Rational Method has been the mainstay of urban stormwater drainage practice in this country since the late 1800s. Its objective of the rapid removal of surface water did not fully consider water quality effects, as well as downstream water quantity effects, through the construction of culverts, storm sewers, and open channels designed to accommodate the increased flow rates from urban areas. The effect of this approach has been to increase the velocity of runoff, reduce the natural storage of water in the watershed, and transmit the increased runoff more quickly downstream. It is well known that the impervious surfaces of an urban area contribute to these increased flow rates: within a typical residential area, streets and parking areas may contribute from 45 to 85 percent of the runoff volume, with roof surfaces contributing an additional 10 percent.

Over the past 20 years, however, water quality has become an increasing concern. In the 1960s, studies by the U.S. Public Health Service and the Federal Water Quality Administration indicated that urban runoff caused water quality problems in receiving surface waters and posed a human health hazard. In the 1970s, water quality management, or 208, plans were prepared by regional or county planning agencies in most major urban centers. Many of the 208 studies concluded that urban stormwater runoff was a water quality problem, but the 208 studies lacked sufficient data on:

- Sources of pollutants
- Washoff and transport mechanisms
- Receiving water impacts
- Effectiveness and costs of urban nonpoint source control measures

In the 1980s, the EPA funded the Nationwide Urban Runoff Program (NURP) studies, and state and local nonpoint source planning efforts--such as a Milwaukee Harbor Estuary Study and the Milwaukee River Priority Watersheds Planning Program in Southeastern Wisconsin--also began to address control of urban nonpoint source pollution.

In order to understand how these recent studies have helped in the implementation of urban nonpoint source control measures, it is necessary to briefly review the technical findings of these studies.

CHARACTERIZATION OF URBAN STORMWATER RUNOFF

Compared to rural streams, streams draining urban areas frequently have similar concentrations of solids and nutrients, and higher concentrations of metals and bacteria. No only, as stated above, are runoff volumes higher from highly impervious land uses, such as commercial and industrial land areas, but pollutant concentrations are also higher from these land uses. Metals in urban runoff which often violate acute (short-term) and chronic (long-term) criteria include copper, zinc, lead, chromium, and cadmium. Within the Menomonee River in southeastern Wisconsin, studies have indicated that metal concentrations in the water column, as well as in the bottom sediments, increase substantially as the River flows into the Milwaukee metropolitan area.

URBAN NONPOINT SOURCES

Street surfaces are the predominant source of pollutants in urban runoff. In typical urban areas, street surfaces may account for 70 to 80 percent of the total solids load during average storm events, while pervious surfaces generally account for less than 20 percent. Pervious surface loadings become more significant during larger storm events. Traffic emissions and atmospheric loadings may each account for up to 10 percent of the total loading.

The NURP studies indicated that pollutant loadings on streets generally increase as the imperviousness of the land area increases. With respect to pollutant loading, the impervious surfaces which are directly connected to a storm sewer system are considered to be most important. The composition of the street loadings also varies for different land uses: transportation and industrial land uses generate higher concentrations of metals, whereas residential areas have the highest concentrations of fecal coliform.

URBAN NONPOINT SOURCE CONTROL MEASURES

Street Sweeping

In general, the NURP studies indicated that street sweeping is not an effective measure for achieving high levels of nonpoint source control. Street sweeping was not very effective because:

- Pollutant loading removals from the street surface by the sweeper were often less than 50 percent.
- Sweepers effectively remove the larger particles but do not generally remove the smaller particles--which contain higher pollutant concentrations.
- Street loads accumulate rapidly following sweeping or a storm event.
- There is too much variability related to wind, traffic, street, and rainfall conditions.

Nevertheless, street sweeping programs can be optimized, providing a modest--up to 20 percent--level of nonpoint source control, particularly in commercial and high density residential areas, by:

- Increased sweeping in spring and fall
- Increased sweeping of highly impervious areas
- Improved parking regulations

Stormwater Storage

The NURP studies indicated that stormwater storage could be an effective way to achieve relatively high pollutant removals. The studies indicated that:

- Properly designed wet basins can remove more than 50 percent of the incoming particulate loadings, whereas, dry basins generally do not remove significant amounts of pollutants.

- o Some dry basins may be modified to increase their detention time and pollutant removal effectiveness.

- o Off-site basins, draining large drainage areas of perhaps 100 to 700 acres (0.4 to 2.8 km^2), are more cost-effective than onsite basins draining 5 to 20 acres (0.02 to 0.08 km^2).

IMPLEMENTATION

The NURP and other related studies have provided the information on the technical basis for the development of implementation programs for urban nonpoint source control. It may be concluded from these studies that there is not a wide choice of urban control measures that can achieve relatively high levels of reduction in pollutant loadings. Generally, stormwater storage in properly designed wet basins appears to be the only practical method available which can achieve removal levels exceeding 30 percent for several pollutants. However, there are potential limitations and problems associated with the use of stormwater storage in many urban areas. These limitations include:

1. Maintenance problems related to:

   a. Lack of adequate access

   b. Clogging of outlets

   c. Litter accumulation

   d. Weed control

2. Safety problems related to:

   a. Stagnant pools in urban areas which attract mosquitoes, rodents, and other pests;

   b. High water velocities during storm events;

   c. Attraction to children.

3. Inadequate flood control, if hydraulic design and analysis is not considered. There frequently is a large difference between the volume of storage and outlet capacity needed to provide adequate flood control and that needed to provide optimal water quality protection.

4. Aesthetic problems, since these basins accumulate large amounts of pollutants, are frequently sediment-laden, and have high concentrations of nutrients which can support weed and algae growths.

5. Lack of suitable available open land sites. This is probably the most significant limitation for the use of stormwater storage basins, particularly in developed urban areas. Alternative storage facilities, such as subsurface storage and stormwater treatment systems, which may be more suitable in developed urban areas, are also very expensive.

While many of these limitations to the use of stormwater storage facilities can be mitigated by proper planning and design, and by the development of effective management and maintenance programs, these factors, nevertheless, do limit the wide application of stormwater storage as a water quality management measure.

Other urban nonpoint source control measure which can achieve limited effectiveness include:

1. Street sweeping, which as noted above can often be designed to remove an additional 10 to 20 percent of the loadings of various pollutants in some areas.

2. Construction site erosion control, which can effectively prevent high loadings of sediment.

3. Public education programs, which could particularly be helpful in voluntary nonpoint source control programs.

4. Pet waste control ordinances, which can help reduce high bacterial loadings especially in residential areas.

Finally, urban nonpoint source control programs, especially those that require extensive design and investment, are best implemented under the guidance of a comprehensive stormwater management plan. A stormwater management plan should address both water quantity and quality concerns and contain the following elements:

1. Hydrologic/hydraulic analysis of urban runoff quantity and quality under existing and future conditions.

2. Water quantity and quality objectives and design criteria.

3. Development and evaluation of the costs, hydraulic effects, and water quality impacts of alternative stormwater conveyance and storage measures.

4. Selection of a recommended plan which is economically feasible, which provides for the wise use of water resources, and which meets, where practical, the objectives of the plan.

NPS POLLUTION CONTROL: A CONSULTANT'S PERSPECTIVE[a]

by

Stuart G. Walesh[b]

INTRODUCTION

Since the early 1970's, much of the non-point source (NPS) pollution management activity in the U. S. has been driven primarily by government regulatory and funding programs. Examples of these programs, which have been at all levels of government from federal to local, are the federal 208 Program and local (county, city, village, and special district) erosion and sedimentation control ordinances. Government programs have apparently been based on the conviction that NPS problems exist or will develop.

In other words, NPS management activities to date have been primarily in a "top-down" mode, that is, government directing individuals and entities to mitigate or prevent NPS pollution problems. The historic "top-down" approach to NPR management is characterized by emphasis on research and development, large-scale studies, broad recommendations, and, in general, absence of specificity and little implementation.

The historic "top-down" approach contrasts with a "grass-roots" approach. With the latter, individuals and entities undertake NPS pollution management because they have NPS or NPS-related problems that must be solved now. Some consulting engineers have had opportunities to apply NPS pollution management measures on a problem resolution basis completely outside of, but consistent with the intent of, government programs.

PURPOSE

Using a case study approach, this paper briefly describes several NPS pollution control projects motivated largely by local need, not by government regulatory and funding programs. The case studies are intended to stimulate thinking concerning the value of encouraging and supporting a "grass roots" approach to NPS pollution management. In the absence of, and perhaps unfettered by, government controls, innovative, cost-effective NPS pollution control measures have been designed and are being implemented. Experience gained on these and similar "grass roots" projects is used to draw conclusions and make certain recommendations concerning NPS pollution control.

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[a]Presented at the Non-Point Pollution Symposium, Session on Efficiency and Feasibility of Best Management Practices for NPS Control, Milwaukee, Wisconsin, April 23-25, 1985.

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CASE STUDY 1: SEDIMENTATION BASINS IN SERIES WITH A DETENTION FACILITY

Physical Setting

The setting for this project, as shown in Figure 1, is a 640 acre almost completely urbanized watershed in the Madison, Wisconsin area. The watershed contains parts of three government entities--the City of Madison, the Town of Madison, and the Town of Fitchburg--plus the University of Wisconsin arboretum. The arboretum is a sanctuary for native plants and wildlife and is used for research and teaching.

Problems

Two growing problems were of concern to area residents several years ago at the beginning of the engineering phase of this project. First, flooding of increased frequency and severity was occurring in the peninsula of single-family residential development which protrudes into the arboretum from the east, as shown in Figure 1. Watershed runoff from the entire watershed passing through and over this area caused surface and basement flooding. Second, debris and sediment carried from the urbanizing watershed were being deposited in and causing damage to meadows, wetlands, and lagoons in the arboretum.

Solution

Working together, the three government entities and the Arboretum Committee agreed to a surface water management system consisting of two sedimentation basins and a sedimentation facility as shown in Figure 2. The sedimentation basins are located on the upstream side of the arboretum and trap objection-able sediment and attached pollutants, floatables, and other debris before it enters the arboretum. The detention facility, which is normally empty, temporarily impounds stormwater in the arboretum and thus provides flood protection to the downstream residential area.

The 2,200 foot long berm required for the detention facility was designed with a curvilinear alignment and irregular cross-section to minimize its visual impact on the aesthetically sensitive arboretum. The sedimentation basin-detention system was constructed in 1982 at a total of cost of $103,000 and is in operation (Donohue, 1981; Raasch, 1982).

CASE STUDY 2: SEDIMENTATION BASIN IN SERIES WITH RESTORED WETLAND

Physical Setting

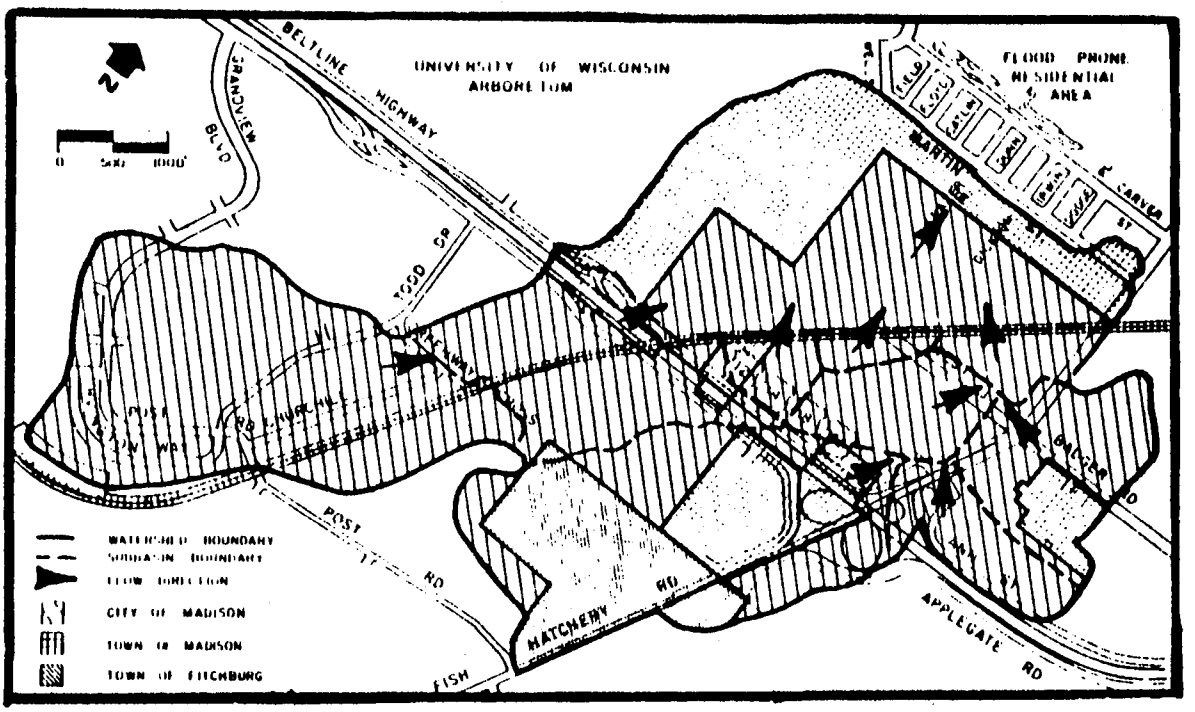
McCarron Lake in Roseville, Minnesota receives runoff from a fully urbanized 1.6 square mile watershed. The 72 acre lake is heavily used for fishing, swimming, and boating and provides a setting for a county park.

Problems

A one-year diagnostic study confirmed that McCarron Lake is eutrophic. Hypolimnetic oxygen depletion releases nutrients into the water column when the lake is stratified and there are abundant nutrients in the water column throughout the year. Surface water runoff into the lake is the principal contributor of the troublesome nutrients and of sediment.

FIGURE 1

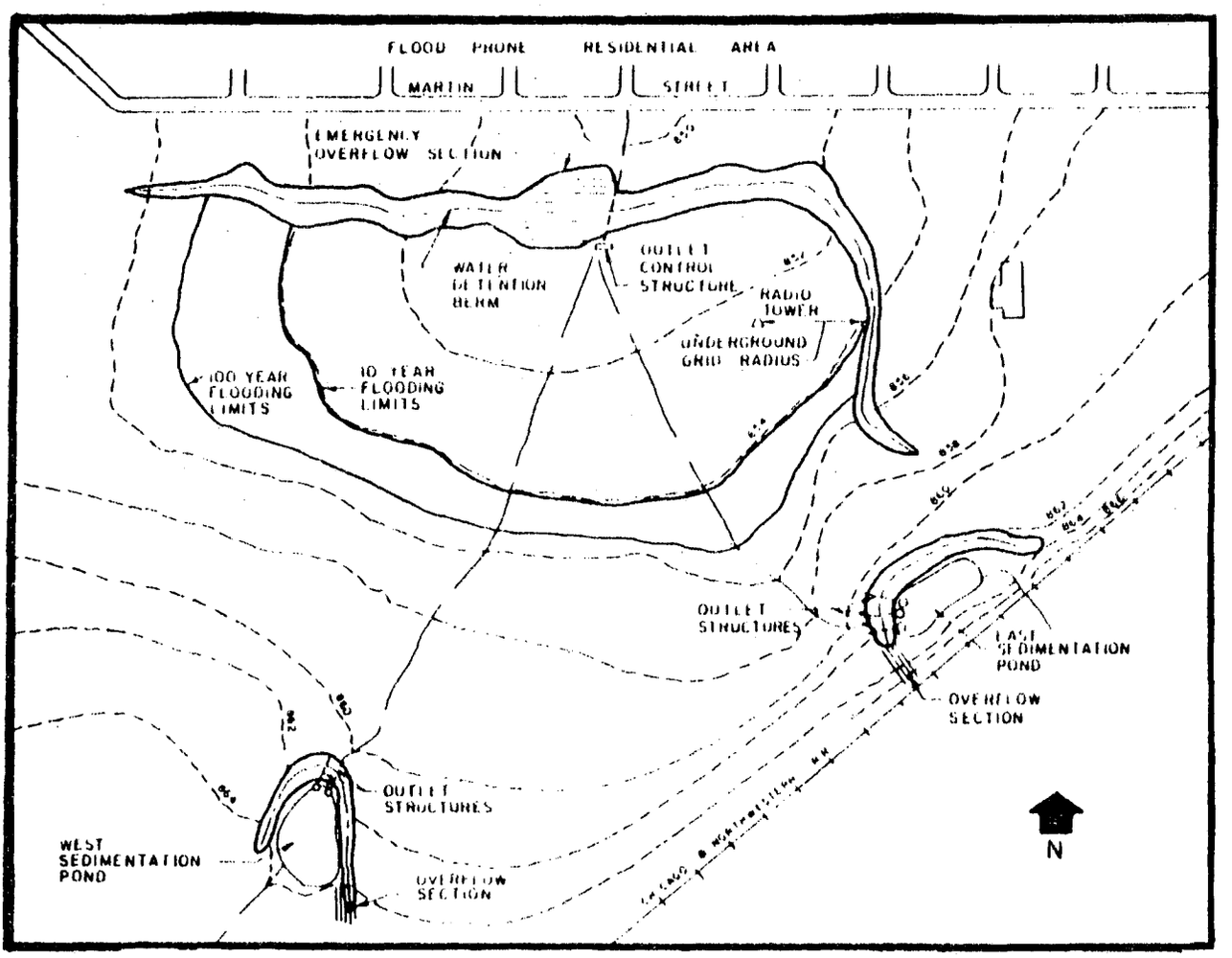
WATERSHED MAP



Source: Raasch, 1982

FIGURE 2

SEDIMENTATION BASINS-DETENTION FACILITY SYSTEM



Source: Raasch, 1982

The solution to the nutrient and sediment problem has two components. The first, as shown in Figure 3, is a sedimentation basin in series with a restored wetland on a drainageway that controls 80 percent of the runoff into the lake. The 1.5 acre sedimentation basin bounded on three sides by 8.5 acres of available, but normally dry, detention storage area is located at the upstream end of the system. The sedimentation basin and related detention area will trap sediment and control flow into the restored wetland. The wetland would be restored to a size of about 8.0 acres by building a series of low berms across the existing channel, as shown in Figures 3 and 4. Vegetation such as cattails, sedges, and willows will grow in the restored wetland. The sedimentation basin-restored wetland is expected to remove about 75 percent of incoming total phosphorus and suspended solids and have a total cost, including engineering and administration, of $250,000.

The second component of the NPS pollution control system is two large sumps to control large street runoff from the remaining 20 percent of the watershed. The circular sumps, one of which is shown in Figure 5, would be large--both are 10 feet deep and one is 18 feet in diameter and the other 24 feet in diameter. As suggested by Figure 6, the sumps are simple structures designed to remove the suspended solids and buoyant materials. The sumps, which are expected to achieve 65 percent reduction in suspended solids and a significant reduction in adsorbed or absorbed phosphorus, have a total cost of $75,000.

The sedimentation basin-restored wetland-sump system is under construction and will be completed by late summer 1985 (Donohue, 1983).

CASE STUDY 3: SURFACE AND SUBSURFACE DETENTION IN AN URBAN AREA

Physical Setting

The 1,200 acre Skokie, Illinois Howard Street Sewer District (HSSD), as shown in Figure 7, is a combined sewer service area. The HSSD is completely and densely developed and over 80 percent of the district is single-family residential with the rest being commercial and industrial. The long, narrow HSSD is very flat having an overall slope of only 0.2 percent.

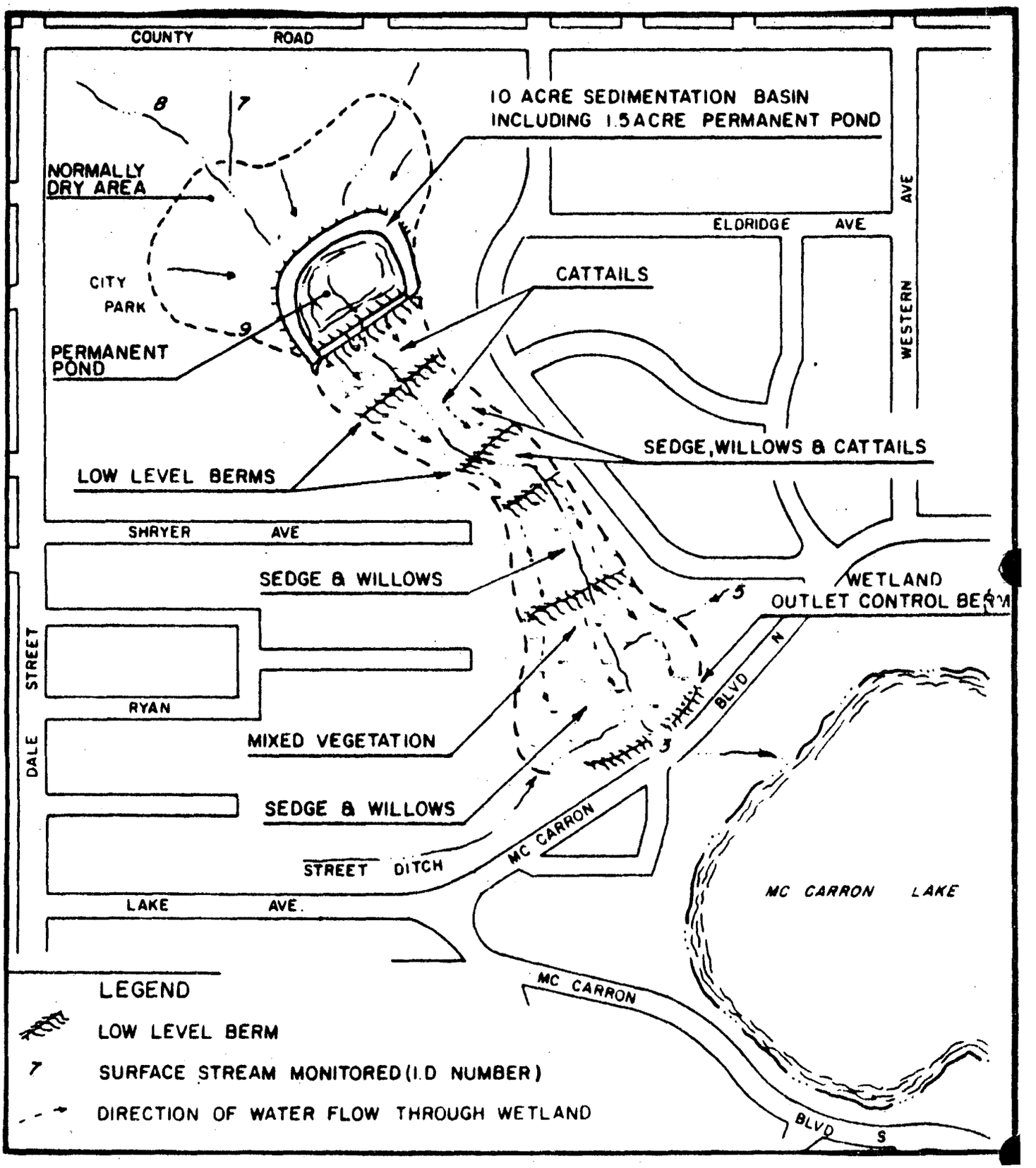
The combined sewer system serving the HSSD discharges at the east end of the district to an interceptor paralleling the North Shore Channel. The start-up of the Chicago area Tunnel And Reservoir Plan (TARP) in 1985 will mitigate wet weather overflow to the channel.

Problem

Athough solution of the combined sewer overflow problem is imminent, serious basement flooding will continue to occur in many of the over 5,000 single-family residences and other buildings in the HSSD. TARP will improve the outlet conditions for runoff events from the district, but it will not resolve deficient sewer capacities throughout most of the HSSD. Design, construction, and operation of a conveyance or detention system to take full advantage of TARP is necessary to solve the flooding problem and is the complete responsibility of the Village of Skokie. Although a relief sewer system, such as new separate storm sewers, is technically feasible, the cost is prohibitively high.

FIGURE 3

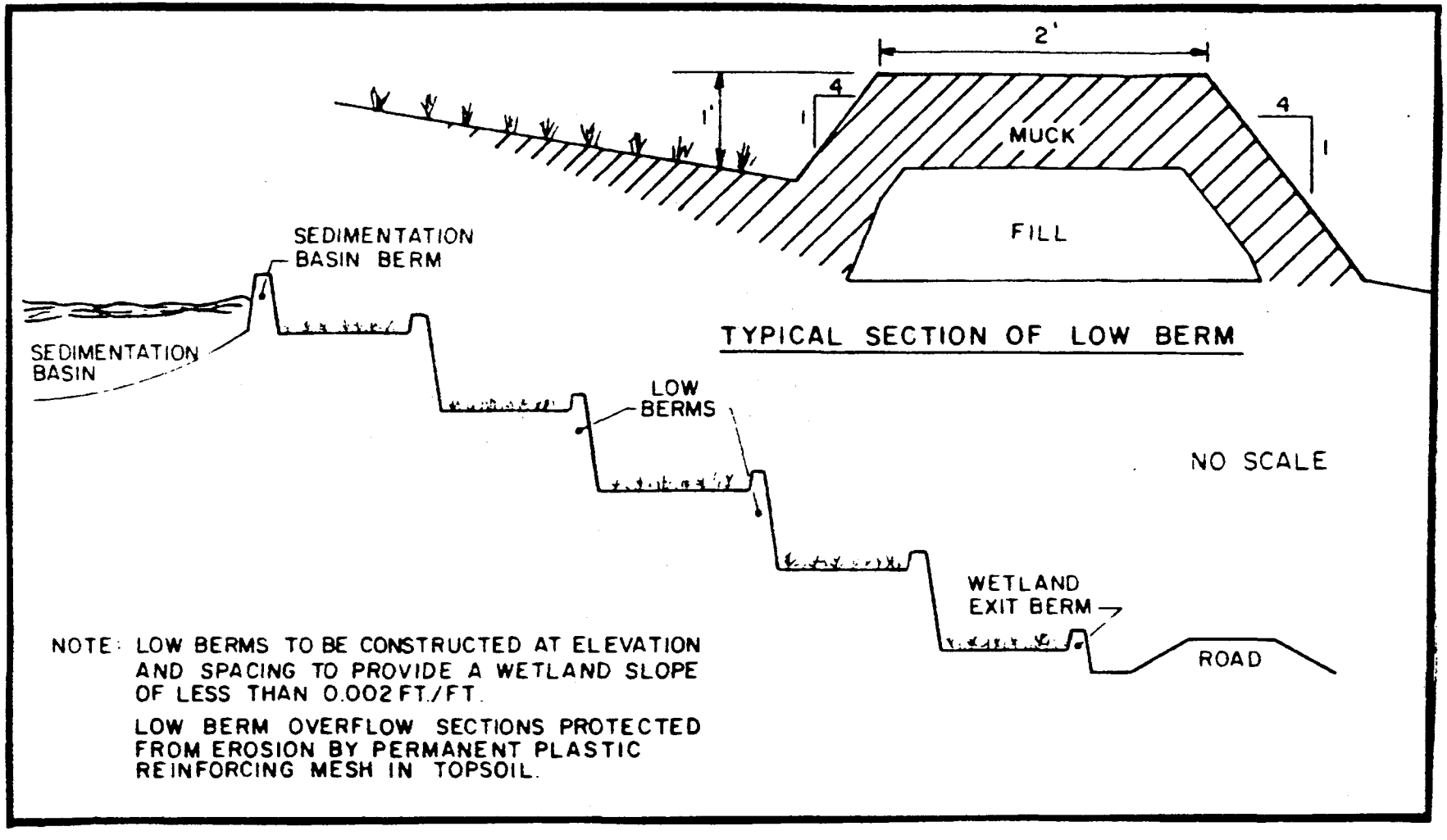
SEDIMENTATION BASIN IN SERIES WITH RESTORED WETLAND



Source: Donohue, 1983

FIGURE 4

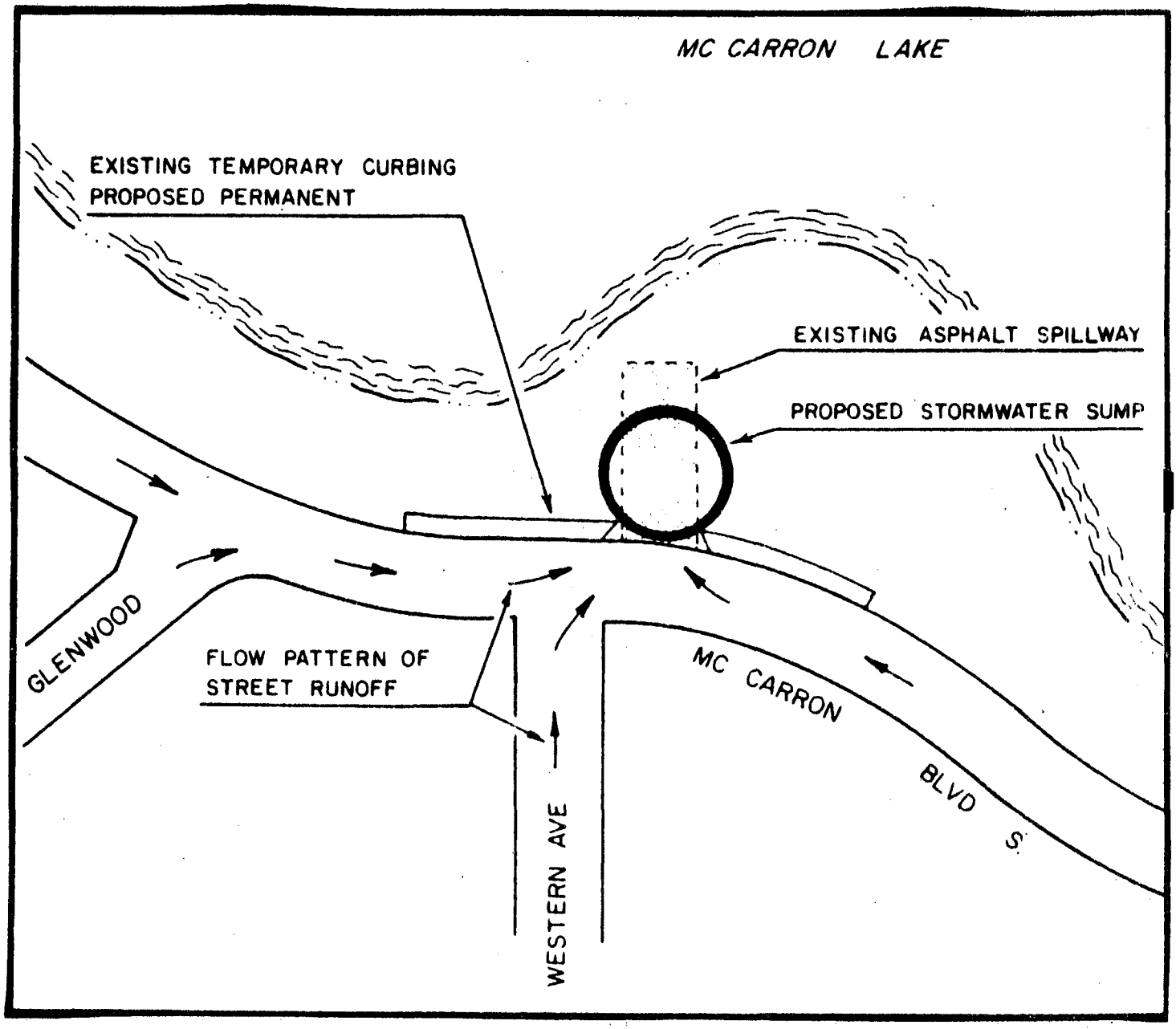
LONGITUDINAL SECTION THROUGH SEDIMENTATION BASIN AND RESTORED WETLAND



Source: Donohue, 1983

FIGURE 5

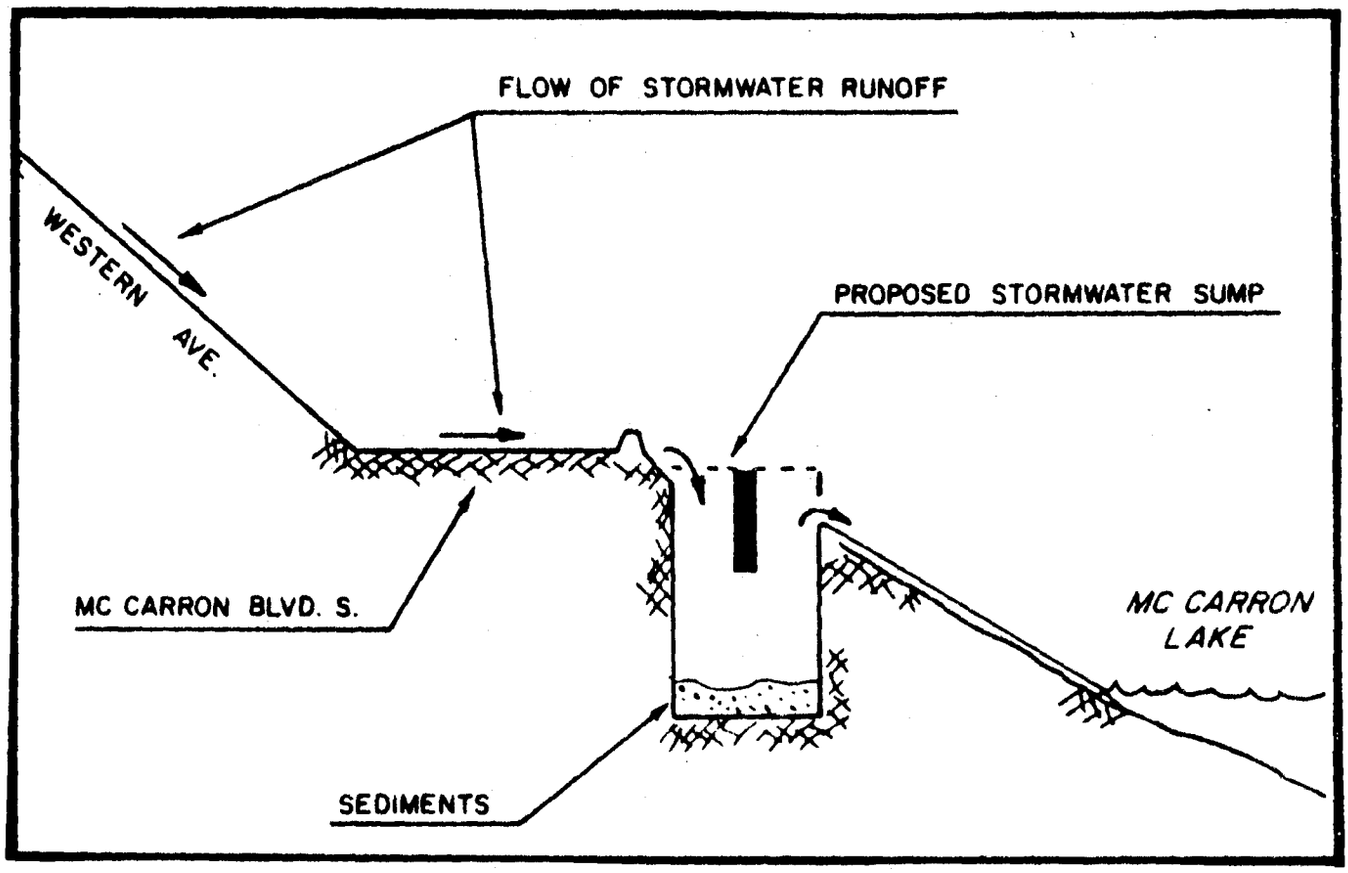
SUMP LOCATION



Source: Donohue, 1983

FIGURE 6

SECTION THROUGH SUMP



Source: Donohue, 1983

Solution

After extensive field and office investigations spanning several years, the Village of Skokie has begun implementation of a runoff control system (RCS) in which stormwater detention will be retrofitted into the existing stormwater drainage and combined sewer system (Donohue, 1982, 1984a, 1984b, 1984c; Walesh and Schoeffmann, 1984). The concept is shown in schematic form in Figure 8 for existing and proposed conditions.

Stormwater will be intentionally ponded in streets in a controlled fashion and then gradually released to the combined sewer system at a rate that can be conveyed by the combined sewer system without surcharging. On those streets where insufficient surface storage is available, supplemental subsurface storage tanks will be used. More specifically, the recommended RCS consists of:

- 431 flow regulators functioning in conjunction with 210 street berms.
- Eight subsurface storage tanks.
- 9,500 feet of 30-inch to 72-inch separate relief sewer for the commercial, downstream end of the HSSD.

The estimated 1985 cost for the recommended RCS, including engineering, legal, and administration fees, is about $8.5 million or less than 20 percent of the cost of conventional relief sewer system.

Most of the temporary street detention will be accomplished by installation of simple flow regulating devices in existing catch basins as shown in Figure 9. Each catch basin contains a sump which traps both settleable and buoyant material. In addition, each subsurface tank will, as illustrated in Figure 10, contain a large sump to trap soluble and buoyant material.

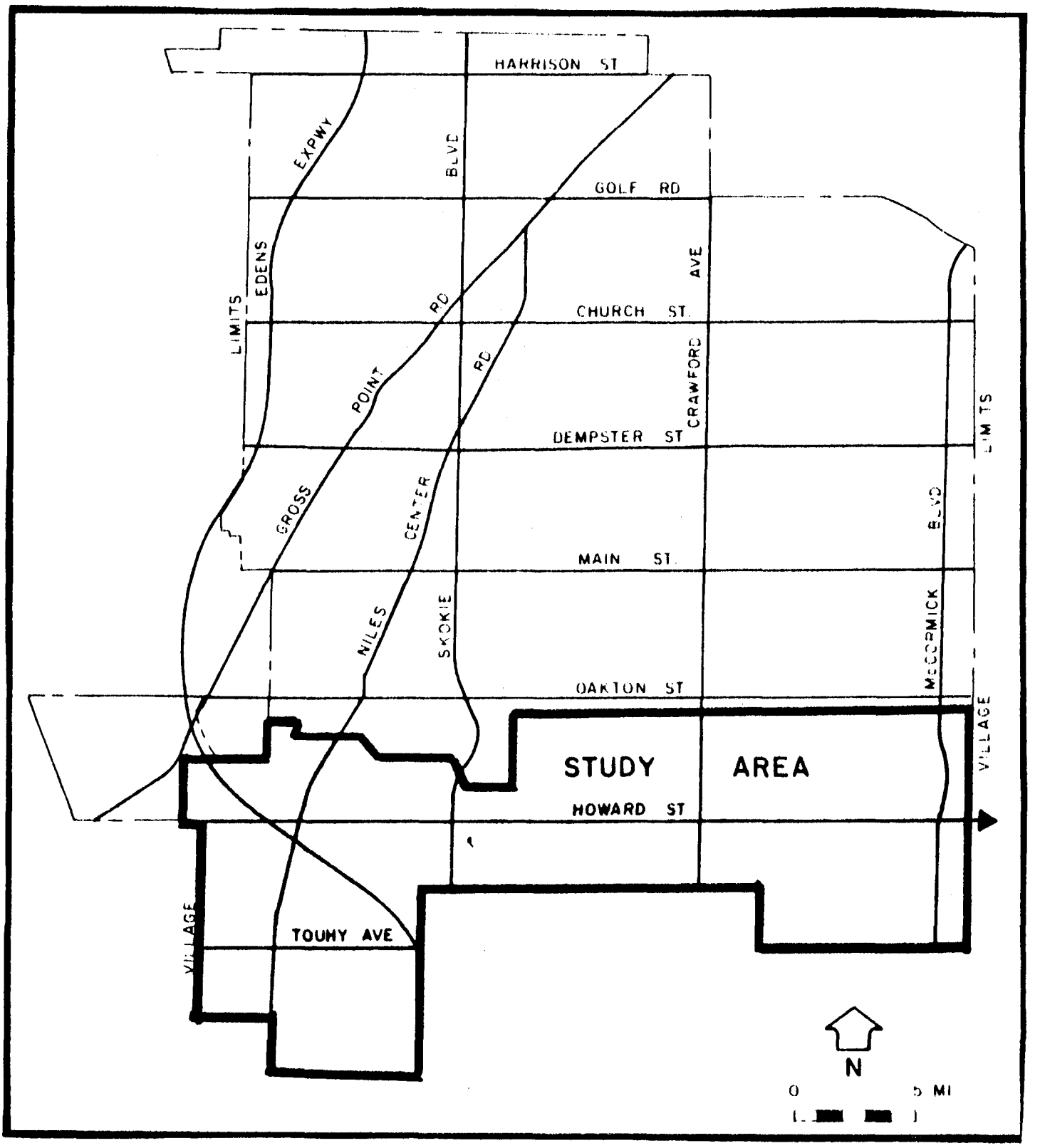
Therefore, essentially all stormwater will, while in route to the combined sewer, pass through and be detained in a sump where settleable and buoyant material, along with adsorbed and absorbed potential pollutants, will be removed. These sumps will be regularly cleaned and maintained by Village personnel.

Design of flow regulators, street berms, and tanks is almost complete. Flow regulators installation is underway and will be completed this year. Tank and berm construction will begin soon and be completed in 1985. Relief sewers will be designed in 1985 and are scheduled for construction in 1986.

Although control of suspended solids and other potential NPS pollutant is not a primary objective of the Skokie RCS, this system or a variation on it has the potential to control NPS pollutants near their source because of the widespread detention occurring and the presence of numerous sumps and traps. An engineered RCS built into an new separately sewered development or retrofitted into an existing separately sewered or combined sewered area could have the following water quality benefits:

FIGURE 7

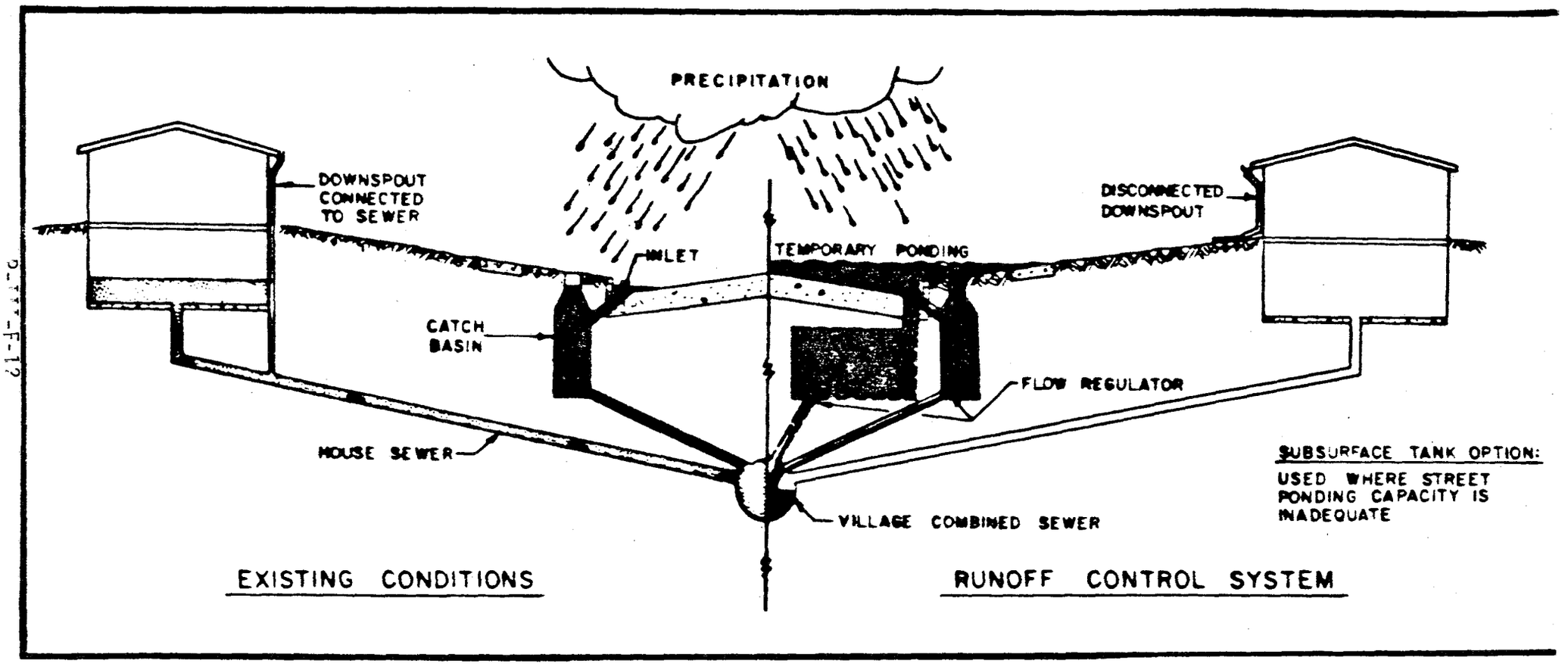
STUDY AREA: HOWARD STREET SEWER DISTRICT IN SKOKIE, ILLINOIS



Source: Walesh and Schoeffmann 1984

FIGURE 8

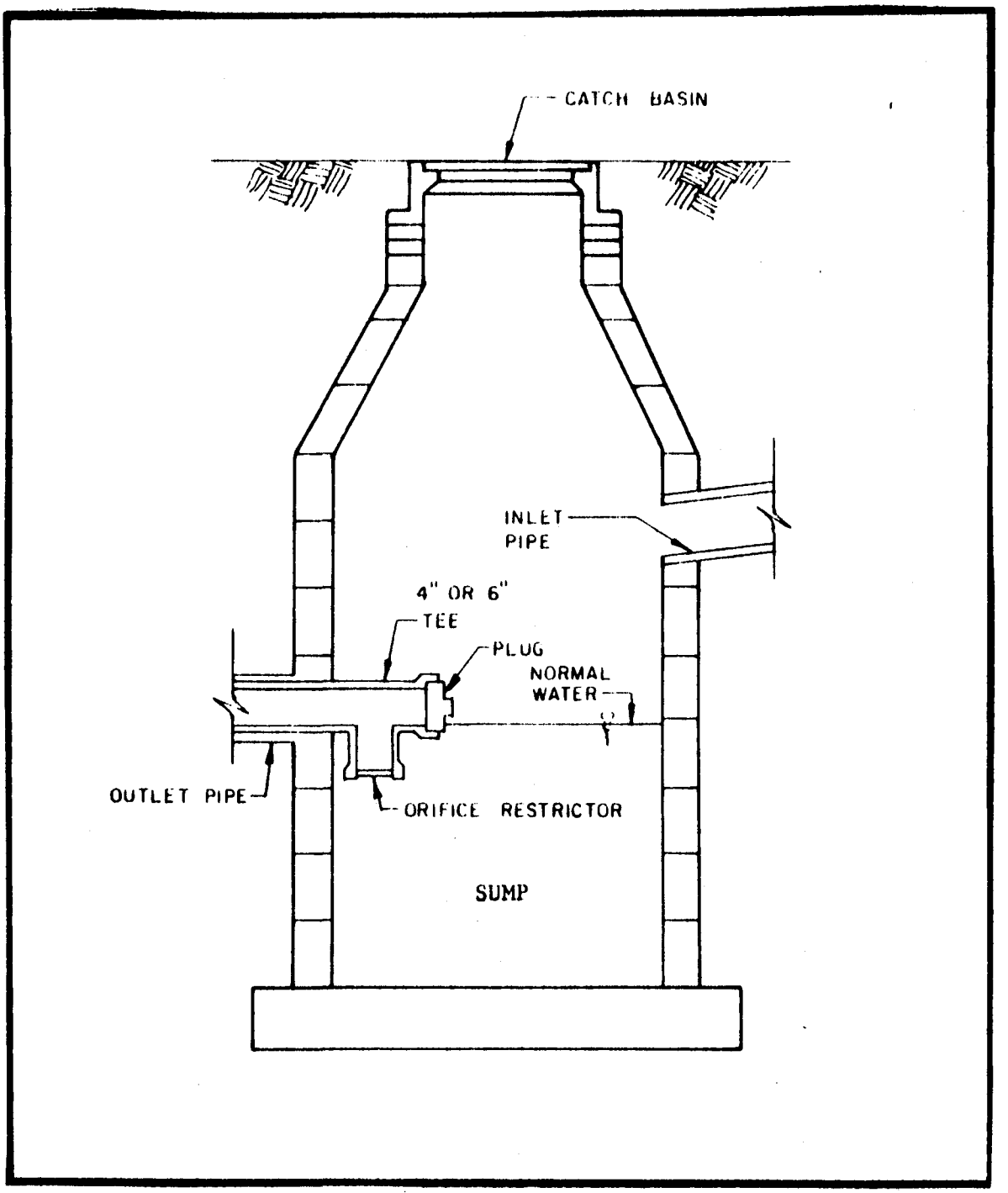
STREET PONDING, REGULATED CATCH BASIN AND UNDERGROUND TANK



Source: Walesh and Schoeffmann, 1984

FIGURE 9

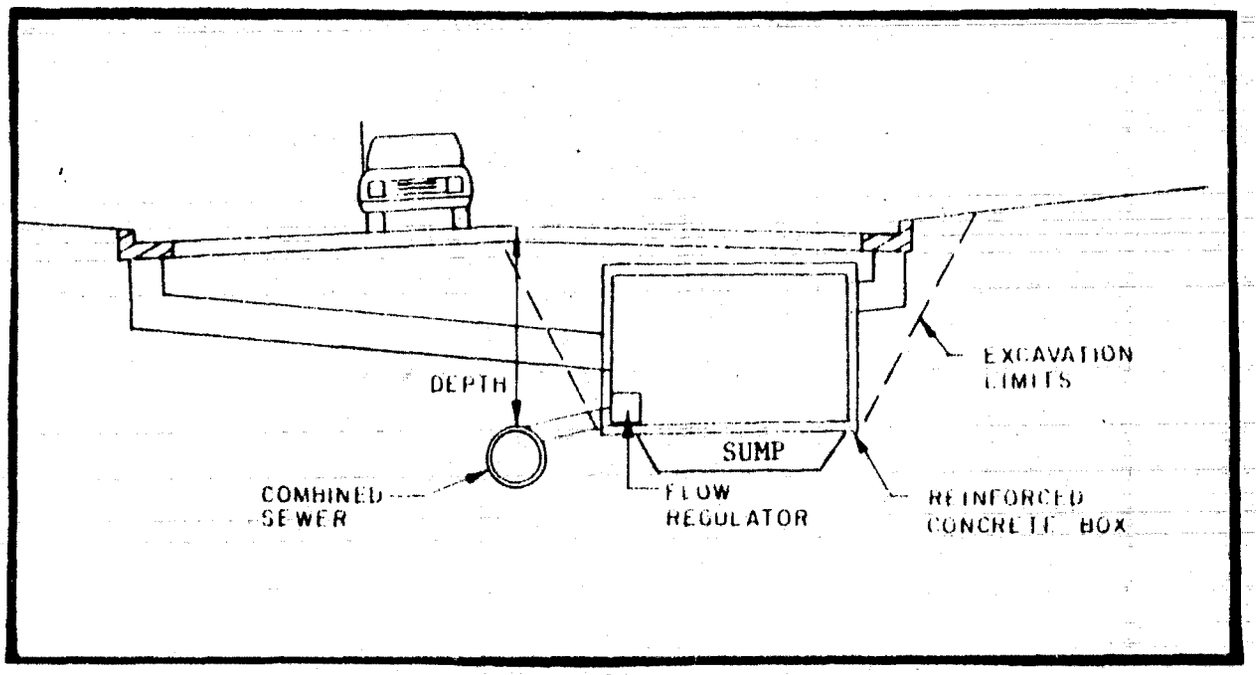
FLOW REGULATOR AND SUMP IN CATCH BASIN



Source: Walesh and Schoeffmann, 1984

FIGURE 10

UNDERGROUND TANK



Source: Walesh and Schoeffmann, 1984

- Reduced frequency and volume of combined sewer overflows to the surface water system.

- Control of NPS pollution near its source for subsequent removal as a part of sump cleaning and maintenance.

- Reduced peak flows at a wastewater treatment plant.

OBSERVATIONS

The following observations are made based on experience, such as the examples presented in this paper, with engineering of NPS controls at the "grass roots" level in response to particular environmental problems:

1. Suspended Sediment is Primary Target: In addition to being a potential pollutant, suspended sediment can be a tranporter of potential pollutants such as phosphorus, pesticides, heavy metals, bacteria, and oxygen-demanding materials. Therefore, the successful control of suspended solids should indirectly result in substantial control of other non-point source pollutants.

2. Small Incremental Expense Concept: In terms of land acquisition and construction costs, relatively little additional expenditures are likely to be required to add effective water quality control components to planned stormwater detention/retention facilities. That is, control of the quality of stormwater can often be achieved for a small incremental cost above that needed to control the quantity of stormwater.

3. Need for Design-Oriented Manual: Research results, performance data, and design tools and techniques should be integrated into a design-oriented manual. Such a manual could present tools and techniques to be used by engineers in the planning, design, and operation of temporary and permanent NPS pollution control facilities such as sedimentation basins, detention/retention facilities, enhanced wetlands and sumps.

4. Stringent Inspection and Maintenance Requirements: The principal purpose of NPS control measures is removal and concentration of potential pollutants from stormwater runoff. At minimum, sediment and other debris will be collected in NPS control facilities. These facilities must be regularly cleaned and otherwise maintained if they are to function as intended. Designers should clearly indicate the type and frequency of inspection and maintenance and provide cost estimates.

5. Resist Mandating Specific NPS Measures: Given the limited, but growing, knowledge concerning the design and operation of NPS pollution control measures, local, state, and federal governmental units should "go slow" in mandating specific means of controlling NPS pollutants. Such agencies should focus instead on statements of intent and performance. Premature rule-making and regulation is

likely to result in "action" but little progress. More is likely to be accomplished in advancing the state-of-the-art of achieving significant control of NPS pollutants by (a) funding additional research and development projects and pilot studies and (b) by encouraging the control of NPS pollutants but not dictating the means.

Acknowledgements

The author gratefully acknowledges the ideas and information provided by colleagues and drawn from project experience at Donohue & Associates, Inc. However, the author is solely responsible for the paper's content.

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SUMMARY
Panel IV
Organization and Institutional Arrangement of
Nonpoint Source Control Programs

PANEL MEMBERS

Moderator: Jim Bauman, Nonpoint Pollution Section, Wisconsin Department of Natural Resources, Madison, WI.

Assistant Moderator: Dean Massey, Economic Research Service, U.S. Department of Agriculture. Madison, WI.

Panelists:

Floyd Heft, Soil Conservation Society of America, Columbus, OH.
William Horwath, National Association of Conservation Districts, Stevens Point, WI
William Katz, Professor, Marquette University, Milwaukee, WI.
Fred Madison, Associate Professor, University of Wisconsin, Madison, WI.
Glen Stout, Professor, University of Illinois, Urbana, IL.

Recorder: Gary Korb, Southeastern Wisconsin Regional Planning Commission, Waukesha, WI.

INTRODUCTION

Sources of nonpoint pollution are numerous and diversified and their control would presently be the responsibility of a number of federal, state and local agencies and private landowners and developers. The present approach to nonpoint pollution control relies on voluntary compliance of polluters with little financial assistance available from federal and state sources. The oversight of the nonpoint pollution control programs are carried out by U.S. Department of Agriculture through its soil and water conservation programs; by the U.S. Environmental Protection Agency which has very limited resources to carry out any meaningful nonpoint pollution abatement programs, and by states that have enacted nonpoint pollution control and abatement laws. The questions this panel was faced with were as follows:

1. Is there an adequate organizational structure in place which would enable to carry out nonpoint pollution abatement programs?

2. If the present organizational structure could be used how could it be improved?

3. If the present organizational structure is not adequate what new organizations and institutions are needed to carry out the programs?

DISCUSSION

Panelists agreed that the present organizational structure and institutions are not adequate however the panel did not reach a conclusion whether a new organizational structure is needed or whether present organizations and institutions should be improved.

There are a number of drawbacks and gaps in the present organizational structure which were identified by one of the panelists as follows:

1. Inadequate funding.

2. Lack of joint planning for soil erosion and water quality control among the agencies.

3. Lack of public accountability due to large number of agencies involved.

4. Institutional goals are not clear.

5. Dilution and shallowness of educational programs due to voluntary approaches.

6. Uneven treatment and understanding of the nonpoint pollution problem across the state lines.

7. Targets and goals of the programs have not been clearly specified.

8. U.S.D.A. agency roles and roles of Soil and Water Conservation Districts have not been clearly defined.

Another member of the panel pointed out the legislative limits that have been imposed on present Soil and Water Conservation Districts. In 13 North Central states there are great differences in the regulatory powers of the SWCD's, and subsequently, there are substantial differences among the states as to:

1. Ease of adoption of ordinance.

2. Recommendations for soil conservation practices.

3. Administrative procedures.

4. Enforcement provisions.

Some states have so called permissive regulatory powers, other states have mandatory regulatory powers. States with permissive regulatory policies may have difficulties in getting necessary regulations adopted.

The availibilty of funds and funding for NPS control programs was also extensively discussed. Some panelists and discussors had reservations against subsidies since they could make the polluters wait for the subsidy and do essentially nothing. In general, financial aid (cost sharing) and subsidy should be given primarily to existing farms that have initiated a nonpoint pollution problem. New farms and farming entreprenuers should consider the NPS control as a part of their cost.

An innovative financial arrangement was suggested by one panelist for lake management districts whereby lake users (water quality beneficiaries) provide subsidies to upstream farmers to implement nonpoint pollution control programs.

CONCLUSIONS AND RECOMMENDATIONS

1. The panel concluded that the present lack of funding and somewhat inadequate organizational structure should not preclude the initiatives to limit nonpoint pollution inputs. The state-of-the-art of the technology and existing organizational structure are satisfactory for an initial effort to curb the nonpoint pollution in many watersheds.

2. Lack of state and, primarily, federal funding has been cited as the major obstacle for successful nonpoint pollution abatement programs.

3. The role of existing agencies should be clearly identified to avoid overlaping and unnecessary competition.

4. Cost sharing and subsidies should be used with caution in order for them not to become an obstacle and a factor causing slowdown of the efforts. Cost sharing and other subsidies should be limited to existing farms and based on their potential to degrade water quality. New farming operators and construction developers should incorporate the nonpoint pollution control in their cost.

5. New organizational structures may be needed if the present existing organizations and institutions are unable to accomplish the goals. The future most optimal organizational structure is not known.

6. The organizations and institutions should have some regulatory back-up if voluntary approaches fail.

ORGANIZATION AND INSTITUTIONAL ARRANGEMENTS OF NPS CONTROL PROGRAMS

William J. Horvath
North Central Regional Representative
National Association of Conservation Districts
Stevens Point, Wisconsin

ABSTRACT

Under S304(4)(1)(A,B,C) of Public Law 92-500, the Federal Water Pollution Control Act Amendments of 1972, EPA was directed, after consulting with appropriate state and Federal agencies and other interested persons to:

> "issue to appropriate Federal agencies, the States, water pollution control agencies, and agencies designated under S208. . . . .information including guide lines for identifying and evaluating the nature and extent of nonpoint sources of pollutants resulting from
>
> > (A) agricultural and silvicultural activities, including runoff from fields and crop and forest land;
> >
> > (B) mining activities, including runoff and siltation from new, currently operated and abandoned surface and underground mines;
> >
> > (C) all construction activity, including runoff from the facilities resulting from such construction. . ."

Both the Federal Government and state governments have struggled with the complexity of nonpoint source pollution control. Control raises complex legal problems by the very definition of it. Standards must be set and identification of source becomes a burden to the legal framework of enforcement.

More importantly--the number of actors necessary to control nonpoint pollution becomes difuse and often confusing. Institutional and legal remedies can not be wrapped up in neat packages. There are shortcomings that can only be addressed below the Federal level.

ORGANIZATION AND INSTITUTIONAL ARRANGEMENTS OF NPS CONTROL PROGRAMS

BACKGROUND

It is well established that until the Federal Water Pollution Control Act (FWPCA) was revised by PL 92-500 in 1972, water quality efforts dealing with nonpoint pollution were fairly limited to research. About the only programs that dealt with nonpoint sources did so on an indirect basis. Regular soil conservation programs and watershed protection activities of SCS, cost-sharing programs such as ACP administered by ASCS, and pesticide registration and labeling programs of ARS were the bulk of activities. State efforts were generally conducted as an adjunct to these efforts. Soil and water conservation districts and their associated agencies only indirectly dealt with the problem in what was thought of as best a bootleg operation.

PL 92-500 redirected the nation's efforts in water pollution abatement efforts. These new efforts focused attention on controlling both point and nonpoint pollutants at their source. Obviously, nonpoint took a secondary seat to the massive efforts undertaken to control point sources. The problem of point was more definable, the technology to a degree in place, and the institutional arrangements more clearly defined.

It was difficult to define nonpoint sources as well from the legal standpoint. The first consideration is the definition of pollution itself. Legislation is designed to maintain water quality by controlling pollution and that depends upon limiting contamination below some established standard.

Then what about identifying the pollutant itself? If you identify it does it become a source point and if you can't how can you legally impose sanctions on the polluter?

Such was the struggle in simplistic terms, but as the entire water quality program gained experience, it became clear that pollution contributed from nonpoint sources could easily equal or exceed efforts in point source control.

Nonpoint was never defined by statute but defined more by exclusion in generally three categories.

(1) It was not susceptible to treatment at the "end of a pipe" however controllable by other changes generally in land management;

(2) It was convened by a natural process such as a storm or seepage rather than through a deliberate controlled effort; and

(3) It was generally generated by a diffused set of man's activities and not identified with any particular one.

Refining was done to this identification for a number of reasons including the well known Natural Resources Defense Council v. Train which narrowed the definition of nonpoint sources. That called for regulation including the permit system to include smaller feedlots, irrigation return flows and storm sewers in urban areas and certain activities of silviculture such as rock crushing, gravel washing, log sorting and storage. A howl of protest and a flurry of activity arose over this ruling. Subsequent rules by EPA in 1976 softened some of the requirements for irrigators through the issuance of general permits if in conformance with 208 plans.

Farm and some closely aligned organizations like NACD spent a considerable amount of time trying to educate the actors to be involved. First to educate EPA and others aligned with them that nonpoint source had to be handled differently than point source. A permit and enforcement system just wasn't feasible for the bulk of nonpoint problems. It would not work to place a policeman on every farm in America.

Secondly, an educational process was needed in EPA as well as state water quality agencies that a mechanism was in place to handle a good portion of the probem since it became clear that soil movement was a major culprit in nonpoint sources. Nor was it as easy as establishing a single set of laws or ordinances to handle problems associated with agriculture, urban storm water, construction site run-off, mining run-off and silviculture.

NACD through a series of grants from EPA, conducted a number of nonpoint source pollution projects including a series of 45 state sediment control institutes, a series of nonpoint pollution control workshops and a number of publications including a series of 48 Nonpoint Notes published over a period of years designed to inform all who were involved in the nonpoint source areas. The Soil Conservation Service placed over 30 staff people on IPA's in Washington and EPA regional offices to assist them in establishing a nonpoint program. Cooperative Extension placed people on IPA in regional offices or contracted for conducting educational activities.

The Chicago office of EPA, because of the implications of nonpoint in the Great Lakes Basin, effectively utilized Section 108 of the Water Quality Act to fund several water basin projects to test certain approaches. The Red Clay Project along Lake Superior tested new methodology to combat sediment in red clay soils, the Washington County, Wisconsin project was designed to take a look at institutional arrangements, and the Black Creek, Indiana project was designed to look at a variety of best management practices (BMPs) and determine methods for measuring the results of BMPs against an improved water quality.

The appropriate roles of state and other management agencies received their most significant direction by the issuance of SAM-31, a program guidance memorandum establishing implementation criteria for EPA approval of NPS elements of state water quality managements. This guidance document

stated that the appropriate level of government and the form of a regulatory program depended on existing laws, programs, and governmental working relationships. Conditional approval could be given by EPA if the state could develop changes in legislative authorites or administrative capabilities.

Fully approved NPS regulatory programs had to include the following elements:

-enforcement authority
-a designated management agency to implement the program
-authority to use the control tools necessary to implement the program
-monitoring authority
-authority to require application of BMPs
-authority to control the activity, pollutant or geographical area to be assigned

Sam-31 also provided for approval of non-regulatory NPS control programs but such programs had to include

-a schedule for the program including implementation, monitoring, and progress evaluations
-annual progress reporting
-adequate technical and financial assistance
-effective educational programs
-identification of BMPs

REMAINING INSTITUTIONAL AND ORGANIZATIONAL GAPS

While a decade has passed since the public attention was first turned to nonpoint pollution, there remains a number of organizational and institutional impediments to completing the task.

Lack of reliable funding sources both at the state and Federal level.

Even though it is recognized that nonpoint source is the remaining significant problem in water quality, Congress has been unable to appropriate the funds necessary to attack the problems. At the state level it is a mixed bag primarily because major emphasis in the last several years has been on soil erosion control. Water quality and offsite benefits are secondary in concern.

Lack of combined planning and implementation for water quality and soil erosion.

Planning for soil erosion and water quality at the state level as well as the Federal level is done in different agencies which have different objectives. They are not necessarily coordinated at the state level and use a separate set of disciplines.

Lack of enforcement and standards leads to the lack of accountability.

With diffusion of best management agencies in the process including Federal agencies, state agencies and local agencies, the public has a difficult time of pin-pointing anyone accountable for program results or expenditure of funds.

The institutional role for water quality control agencies in nonpoint is not clear.

They must rely heavily on other conservation agencies that may be reluctant to play supporting roles in the overall effort.

A diffusion of educational effort combined with voluntary application is a problem.

Education of those necessarily involved in the process including those landowners who apply them are diverse and numerous. On a national scale this involves literally millions of people.

Unequal ability between states to adequately develop a state mechanism is still a problem.

A river basin such as the Great Lakes or Susquehanna River has a bevy of state capabilities which often do not mesh with the severity of the problems.

An over dependency on data and data that may be flawed still exists.

This unecessarily complicates implementation of water quality projects.

The external influcence of factors in controlling nonpoint are ever present.

Weather, changes in technology in farming, state budget problems, all are major factors in planning.

Conflict between old concepts of treating every problem equally with some effort rather than targeting the project is still a political problem where responsibility is diffuse.

The relationship between USDA agencies and conservation districts which most often provide technical assistance and coordination and state water quality agencies is not well defined.

The providers and implementers have different expectations.

Overall, the institutional arrangements are not as clear and concise as we might like because of the nature of the problem which is diffuse and requires a wide variety of approaches. However, the funding for water quality programs dealing with nonpoint supercede all problems. When society funds these programs most of the implementing problems will disappear.

NORTH CENTRAL STATES CONSERVATION DISTRICTS: EFFECTIVENESS OF REGULATORY POWERS

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ABSTRACT

Soil and water conservation districts are the only local agencies primarily responsible for controlling soil erosion and reducing sediment. The 13 North Central states have such districts or an equivalent. Districts in 9 of these states are authorized to regulate land use. Kentucky, Nebraska, North Dakota and Wisconsin provide for permissive regulatory powers; other states require more mandatory controls. Iowa and Ohio regulations are based on soil loss tolerances; Michigan's regulations are based on land-disturbing activities, and South Dakota's, on conservation standards. Illinois has both permissive regulations and mandatory standards for land-disturbing activities. A district's effective abatement of nonpoint pollutants depends on the ease of adopting regulations, conservation practices to be included, procedures for administering regulations, and enforcement methods.

Adopting land use regulations in states with permissive regulatory powers can be difficult, since proposed regulations must often be approved by a high percentage of voters. These states permit districts to include adequate conservation practices in erosion control regulations for agricultural land. None of these states specify how the regulations are to be administered or by whom, although most provide for enforcement power.

In states with more mandatory controls, state agencies have more control over nonpoint pollution. Regulations are handled in three ways: (1) districts are required to adopt regulations and have them approved by a state agency; (2) districts are required to adopt conservation standards and regulations based on state agencies' programs and guidelines; or (3) state agencies are required to adopt statewide control programs and regulations that are administered and enforced by state and local agencies.

Only Illinois, Iowa and South Dakota conservation districts administer land use regulations. Permits are required for land-disturbing activities in Michigan and South Dakota. Agricultural land in Illinois, Iowa and South Dakota is deemed to comply with regulations until a complaint is made; the districts then provide for enforcement. No enforcement provisions exist in Ohio under either agricultural or urban pollution control rules.

Keywords: Agricultural runoff pollution, conservation practices, land use regulations, nonpoint pollution, nonpoint pollution abatement, regulatory powers, soil erosion, soil conservation districts

INTRODUCTION

Nonpoint source water pollution is a major environmental problem in much of the United States. Sediment from erosion is the major nonpoint pollutant, and agriculture contributes more than half of this, both by mass and volume. Almost half of the agricultural sediment comes from cropland, and nearly half of all cultivated cropland consists of soils with a high potential for erosion.

Controlling sources of nonpoint pollutants is considered the best method of preventing sediment and plant nutrients from reaching water resources. Possible institutional tools for controlling sources of nonpoint pollutants include prohibiting certain land uses at particular locations and regulating uses on land at the source. Most of the responsibility for controlling nonpoint pollutants rests at the local level, with federal and state governments establishing water quality standards, guidelines, and soil loss limits, and providing technical and financial assistance. Soil and water conservation districts are the only local agencies whose primary responsibility is controlling soil erosion and reducing sediment and other resultant nonpoint source pollutants.

Section 208 of the Clean Water Act provides for the identification of agricultural, urban and construction-related nonpoint sources of pollution, and for the development of areawide water quality management plans containing regulatory procedures to control pollutants from these sources (Iowa L.R., 1977). (Citations to applicable federal and state statutes and state agency administrative regulations are provided in the references.) One or more state, regional or local agency must be designated to implement the areawide plans, and a possibility may exist for designating soil and water conservation districts as management agencies (Davey).

This paper deals solely with the 13 North Central states: Illinois, Iowa, Indiana, Kansas, Kentucky, Michigan, Minnesota, Missouri, Nebraska, North Dakota, Ohio, South Dakota and Wisconsin. It describes the regulatory powers of soil and water conservation districts in these states, and evaluates the effectiveness of these regulatory powers in abating nonpoint sources of pollution in both rural and urban areas; it finds that the effectiveness of regulations depends on ease of adoption, conservation practices that may be included, administrative procedures, and enforcement ability. District regulatory powers are also evaluated for their effectiveness in providing the local regulatory programs required to implement Section 208 areawide water quality management plans.

DISTRICT ORGANIZATION AND REGULATORY POWERS

All 13 of the North Central states have soil and water conservation districts or an equivalent. Nebraska eliminated conservation districts and created 24 natural resources districts that assumed their functions. Wisconsin abolished soil and water conservation districts and transferred their functions to county boards, requiring them to create land conservation committees to implement soil conservation programs.

Soil and water conservation districts and their equivalents, except in

Indiana, Kansas, Minnesota and Missouri, have some type of land use regulatory powers to control soil erosion and sediment. Illinois, Kentucky, Nebraska, North Dakota and Wisconsin give districts or counties permissive power to adopt land use regulations. More mandatory controls are established in Iowa, Michigan, Ohio and South Dakota, and in Illinois (under a second statute).

OPERATION OF LAND USE REGULATIONS

Adoption Procedures

District supervisors in four of the states with permissive regulatory powers--Illinois, Kentucky, Nebraska and North Dakota--may develop and adopt land use regulations after a public hearing and after the proposed regulations have been approved by the required number of voters on a referendum. Approval in the four states requires a favorable vote of between 75 and 90 percent of the qualified voters voting in the referendum. In Wisconsin, the county land use conservation committee develops the land use ordinance and submits it to the county board, who may adopt it after a public hearing. However, the ordinance adopted by the board does not become effective until approved by a majority of the voters on a referendum in the affected area.

Even though district supervisors or county boards in these five states are authorized to adopt land use regulations, the requirements for approval on a referendum as a prerequisite to adoption make adoption almost impossible in these states, except Wisconsin. Requiring a referendum and a favorable vote of more than 51 percent can block the enactment of regulations, especially if voting is limited to landowners within the district (Holmes). Because of such adoption procedures, these states cannot effectively provide the regulatory programs necessary for nonpoint pollution control under Section 208 of the Clean Water Act.

Adoption of the more mandatory regulations in Iowa, Michigan, Ohio and South Dakota, and under Illinois' second statute, involves a joint effort by state agencies and local governments and soil and water conservation districts (Davey; EPA, 1978). Illinois amended its original statute requiring the State Department of Agriculture to adopt guidelines, and requiring districts to adopt erosion control programs in accordance with state guidelines. Iowa district commissioners have classified land in the state's 100 districts and established and implemented soil loss limits for each classification.

Illinois and South Dakota require state agencies to prepare guidelines for erosion and sediment control, and require conservation districts to adopt regulations or standards based on the state guidelines (Holmes). The Illinois Department of Agriculture and South Dakota Conservation Commission have developed comprehensive statewide erosion and sediment control programs and guidelines to be implemented and administered by the conservation districts. Each district in Illinois had until April 1982 to develop and adopt soil erosion and sediment control programs and standards consistent with the state program and standards; the department reviewed and approved district programs and standards, and also developed appropriate programs and standards for districts that failed to develop their own. All except two conservation districts in South Dakota have developed and adopted conservation standards

in cooperation and consultation with counties and municipalities; the commission reviewed these standards prior to adoption.

State agencies adopt land use regulations that are enforced by state agencies and local governments in both Michigan and Ohio. The Michigan Department of Agriculture, with the assistance of the conservation districts, prepared and submitted a unified statewide soil erosion and sedimentation control program for approval by the Water Resources Commission. The commission, with the assistance of the Department of Agriculture, prepared and adopted rules for the unified program that became effective on January 1, 1975. The Ohio Division of Soil and Water Districts, with the approval of the Soil and Water Conservation Commission, adopted standards to abate soil erosion and degradation of state waters from agricultural, urban and livestock feeding activities (Holmes).

Conservation districts in the five states with more mandatory controls come closer, as adoption procedures are concerned, to meeting the program requirements of Section 208 water quality management plans. Illinois, Iowa and South Dakota give districts the most authority in land use regulation: districts in these states adopt their own soil erosion and sediment control programs, regulations or standards and have them approved by a state agency to assure compliance with statewide programs. Problems would exist, however, for districts in Michigan and Ohio, because rules and standards are adopted at the state level, with districts playing only an advisory role.

Conservation Practices to be Included in Regulations

Legislation should be sufficiently broad to enable state and district regulations to include a variety of soil management practices from which can be selected the most appropriate combination for each situation or condition (EPA, 1973). Criteria for selecting management practices include their ability to manage pollutants generated from nonpoint sources, compatability with water quality goals, effectiveness in preventing or reducing pollutants, practicability, and technical capability of preventing or reducing runoff.

Illinois, Kentucky, Nebraska, North Dakota and Wisconsin, with their permissive powers, do not require the inclusion of specific provisions in their land use regulations. Enabling legislation in these states, except in Wisconsin, generally provides that district regulations may contain any one or a combination of the following: (a) requirements for performing necessary engineering operations; (b) requirements for observing particular methods of cultivation; (c) specifications for cropping programs and tillage practices and (d) provisions for removing highly erodable areas from cultivation (Holmes).

Enabling legislation in these states is agriculturally oriented and applies to soil erosion control (Davey; EPA, 1973). Wisconsin's statute has the strongest provision prohibiting land uses and management practices that cause excessive erosion, sedimentation and nonpoint source pollutants. Land use regulations adopted by conservation districts in the other four states would probably be too narrow to fulfill the requirements of the Section 208 areawide management plans. Such regulations, if adopted, would adequately manage only nonpoint pollutants derived from agriculture.

Conservation practices that may be included in regulations of the five states with more mandatory programs vary. Iowa's program is based on soil loss limits, and Michigan's is based on land-disturbing activities. A combination of soil loss tolerances and standards for land disturbances forms the basis of the Illinois, Ohio and South Dakota programs.

Landowners in Iowa are required to employ soil conservation and erosion control practices on their land, but district commissioners may not specify particular practices to be employed as long as the owner voluntarily complies with established soil loss limits. Commissioners also may not require the employment of erosion control practices on land used in good faith for only agricultural purposes and may not prohibit operators from fall plowing.

The Michigan Water Resources Commission adopted rules based on land-disturbing activities, implementing the Department of Agriculture's recommended unified statewide program that included provisions for land use plans and permits. Under the rules, all persons engaged in earth changes that require a permit must design, implement and maintain acceptable erosion and sedimentation control plans that effectively reduce erosion. Permits are required when disturbing one or more acre of land, or if the disturbance is within 500 feet of a lake or stream. In addition, approved soil and water conservation district erosion and sedimentation control standards and specifications must be followed by those requiring a permit. Agricultural practices were included in the rules in 1979, and now agricultural earth disturbances, except for plowing and tilling, or unless the farmer has an agreement with the district, are subject to permit and plan requirements.

Conservation practices incorporated in land use regulations in Illinois, Ohio and South Dakota are based on a combination of soil loss tolerances and conservation standards for land disturbances. Under Illinois Department of Agriculture guidelines, all conservation systems and practices applied to agricultural land must seek to reduce soil losses over a period of years to levels at or below the soil loss tolerance ("T" value) established by the Soil Conservation Service. A "T" value is the average annual soil loss, in tons per acre, that a given soil may experience and still maintain its productivity over an extended period of time. Policies and specifications for various erosion and sediment control devices, structures and practices applied to agricultural land were also given in the guidelines. Programs and standards for various soils and land uses adopted by the districts contain criteria, guidelines, techniques and methods for controlling erosion and sediment from land-disturbing activities (Holmes).

District conservation standards or soil loss tolerance limits in South Dakota are to be consistent with the State Conservation Commission's guidelines. The guidelines contain recommended soil loss limits and suggested conservation practices and methods for controlling erosion and sediment (Holmes; EPA, 1978). Counties adopt ordinances incorporating district standards relating to agricultural and non-agricultural land-disturbing activities; non-agricultural activities subject to permit requirements must adhere to the district's soil loss standard and to a soil erosion and sediment control plan (Holmes).

The rules adopted by the Ohio Division of Soil and Water Districts to

abate agricultural nonpoint pollutants provide for the achievement over a period of years of the applicable soil loss tolerance or permitted soil loss values. Farmers are required to practice conservation and to follow a management system, so that under given cropping and management conditions, the predicted soil loss (using the Universal Soil Loss Equation) from erosion is equal to or less than the "T" value for that specific soil series, as specified in the Ohio Erosion Control Guide or the SCS Technical Guide. Farmers may not use earth-disturbing practices, including tillage, on land immediately adjacent to water, unless practices are constructed or implemented in accordance with proper engineering designs.

Conservation practices that may be included in the land use regulations of the five states with a more mandatory approach are generally broader than those of the states with permissive regulations. All are based on land-disturbing activities for both agricultural and non-agricultural activities, and all include sediment control in addition to erosion control. One drawback to the Iowa and South Dakota regulations is that they do not apply to agricultural land unless the erosion exceeds specified soil loss limits and a complaint is filed. However, considering the management practices and land-disturbing activities that may be included in the content of land use regulations, the statutes and regulations of these five states are probably broad enough to fulfill the regulatory requirements of Section 208 areawide water quality management plans.

Administrative Procedures

Nothing in the Illinois, Kentucky and North Dakota statutes states how land use regulations are to be administered or who administers them. Nebraska statutes provide only that districts have power to administer the rules and regulations; Wisconsin provides that the county board prescribe administrative procedures in the land use ordinance. In these five states, enabling legislation concerned with administrative procedures is probably inadequate to effectively provide the necessary regulatory programs for nonpoint pollution in the Section 208 areawide plans. Such legislation should provide who is to administer regulations, the method for administering, whether permits are to be issued, and, if so, who issues them, and if plans and specifications are to accompany permit applications. The powers and duties of the administrator should be specified along with the permit, plan, and specification requirements.

Of the five states with more mandatory regulatory powers, only Michigan and South Dakota provide much detail on administrative procedures. Regulations are administered by districts in Illinois and Iowa, by counties in Michigan, by local permit-issuing authorities in South Dakota, and by the State Division of Soil and Water Districts in Ohio.

Illinois does not require persons engaged in land-disturbing activities to apply to soil and water conservation districts for permits or to submit erosion and sediment control plans. Districts, upon request, make adequate information and technical assistance available to persons engaged in land-disturbing activities. Neither are permits or plans required for land-disturbing activities in Iowa.

The Ohio Division of Soil and Water Districts established procedures for the districts to administer agricultural and urban pollution abatement rules and to enforce animal waste management rules. The division also recommends criteria and procedures for approving urban sediment pollution abatement plans and issuing permits. Under the agricultural pollution rules, the division will inform the districts of state standards, criteria and procedures for pollution abatement; enter into agreements with districts for implementing agricultural pollution abatement programs; provide administrative guidance for implementing and administering such programs; and implement agricultural pollution abatement programs in districts failing to negotiate an agreement with the division. Permits, plans or specifications are required for land-disturbing activities in Ohio.

Counties in Michigan administer the state-adopted rules; county boards designate a particular agency responsible for enforcing the rules. Each county enforcing agency adopts the district's standards and specifications. Persons engaging in earth changes must obtain permits from appropriate county enforcing agencies, and must submit soil erosion and sedimentation control plans with permit applications. District standards and specifications that have been approved by county enforcing agencies must be followed and utilized as they apply to specified earth changes.

Conservation standards adopted by districts in South Dakota are administered by local permit-issuing authorities; neither the State Conservation Commission nor conservation districts may issue permits. Each permit-issuing authority within the district must include provisions in its permit procedure to ensure that any proposed action is in compliance with district conservation standards. Persons engaged in agricultural or minor land-disturbing activities need not prepare a conservation plan, secure a permit, or report their activities to the conservation district unless they violate the district standards.

Not only are Michigan and South Dakota alone in providing much detail on administrative procedures, they are also the only states requiring the issuance of permits for land-disturbing activities. Basically, only Michigan statutes provide sufficient detail on who administers the conservation program and how it is administered to be adequate for Section 208 water quality management plans. Administration in Michigan, however, is not by the soil conservation districts, but rather by counties.

Enforcement Procedures

A variety of methods are available in the states with permissive powers to enforce the land use regulations. Of these five states, only Nebraska does not provide for enforcement powers. Wisconsin goes further by providing that county ordinances should prescribe administrative and enforcement procedures. District supervisors have the authority in Illinois, Kentucky and North Dakota to enter upon any lands affected by the regulations to determine compliance.

Illinois and Kentucky districts may provide in their regulations that persons damaged by someone violating regulations may recover from the violator. Wisconsin ordinances may be enforced through civil forfeitures or by

injunctions in court actions initiated by the county board or the land conservation committee. There are no criminal penalties or forfeitures in Illinois, Kentucky or North Dakota. District supervisors in Illinois, Kentucky and North Dakota may initiate a court action compelling violators to perform the work, or permitting supervisors to perform the work themselves if the violators failed to perform in a specified time. Conservation districts or counties in Illinois, Kentucky, North Dakota and Wisconsin have adequate available enforcement powers over their land use regulations to permit regulation by such agencies in a Section 208 areawide management plan.

Enforcement authority also varies in the five states with a more mandatory approach. Agricultural land in Illinois, Iowa and South Dakota is deemed to comply with regulations unless a complaint is made; the conservation districts are then responsible for providing at least the initial phases of the enforcement process. Counties and a state agency provide the enforcement in Michigan, while there are no enforcement provisions for the Ohio agricultural or urban pollution rules.

Any person in Illinois engaging in land-disturbing activities is encouraged to comply with district standards for erosion and sediment control. Complaints may be filed with the conservation district by any person, the district, or the Department of Agriculture if it is believed that a serious problem exists. If the district finds that standards have been violated, it notifies the violator and seeks voluntary compliance, gives a deadline for compliance, and suggests practices enabling the person to comply with the standards.

Landowners in Iowa are required to employ either conservation or erosion control practices. An owner or occupier of land being damaged from sediment files a written complaint with district commissioners, charging that soil erosion is occurring in excess of the district limits. If the commissioners find sediment damages and excessive soil erosion, they are required to issue to the violator an administrative order stating the extent to which soil erosion exceeds the district's soil loss regulations; if the land is agricultural, the order states when the needed work is to be commenced and satisfactorily completed. Persons failing to comply with the order may be held in contempt and fined or imprisoned.

South Dakota does not require persons engaged in agricultural or minor land-disturbing activities to prepare a plan or obtain a permit unless the district determines that a particular land disturber is violating district standards. Once such a violation has been determined, the land disturber is ordered to prepare an erosion and sediment control plan, to have it approved by the district, and to implement the plan. Either the district or permit-issuing authority may initiate a court action for an injunction or other appropriate relief to enforce the order.

County or local enforcing agencies in Michigan are responsible for enforcing the state land use regulations. Persons who fail to get a required permit for earth-disturbing activities are guilty of a misdemeanor. County or local enforcing agencies notify the Water Resources Commission of all violations of the state regulations or local land use ordinances. If the

commission finds a violation has occurred, it proposes an appropriate correction agreement to the violator. If the violator does not agree to the proposal, an order requiring correction, enforceable by the court, is issued by the commission. The county or local enforcing agency may also issue cease and desist orders and revoke permits upon finding violations of regulations or approved soil erosion and sedimentation control plans.

The statutes giving the Ohio Division of Soil and Water Districts authority to adopt rules for agricultural pollution and urban sediment pollution abatement do not provide enforcement powers. If, however, the conservation districts receive complaints on violations of the standards, they will attempt to find a solution through voluntary cooperation.

Soil and water conservation districts are involved in enforcement in all these states except Michigan. However, soil erosion control in Illinois and Ohio is voluntary; there are no means of enforcing compliance with the standards, except through cooperative efforts. Only in Iowa and South Dakota do soil and water conservation districts meet the regulatory agency requirements of a Section 208 areawide water quality management plan from an enforcement standpoint.

CONCLUSIONS

All 13 North Central states have soil and water conservation districts or similar local governmental units. Districts in nine states either possess or are involved with some type of nonpoint source pollution regulatory power. Adopting land use regulations in the five states with permissive regulatory powers is a difficult process and involves stiff requirements for approval on a referendum. Enabling legislation in these states permits districts to include adequate conservation practices and measures in the land use regulations for controlling soil erosion on agricultural land. None of the permissive regulatory states specify who is to administer land use regulations or how they are to be administered. All the states except Nebraska provide for enforcement powers.

State administrative agencies in states with more mandatory regulatory powers--Illinois, Iowa, Michigan, Ohio and South Dakota--have more authority in the nonpoint source pollution control process than do state agencies in the states with permissive regulatory powers. Conservation practices and methods are based on soil loss limits, conservation standards and land-disturbing activities, and they apply to both agricultural and non-agricultural uses of land. Only in Illinois, Iowa and South Dakota are soil and water conservation districts responsible for administering land use regulations. Permits are used for land disturbances in Michigan and South Dakota. Agricultural land in Illinois, Iowa and South Dakota is deemed to comply with the regulations unless a complaint is made; the districts then provide for enforcement. Counties and a state agency provide enforcement in Michigan, and there are no enforcement provisions in Ohio for either agricultural or urban pollution control rules. Only districts in Illinois, Iowa and South Dakota have sufficient powers to be considered adequate management agencies for implementing the Section 208 areawide water quality management plans.

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EMERGING INSTITUTIONS FOR NPS CONTROL

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The management of our water and land resources require an interdisciplinary approach involving the many users of the resources. For fifty years, agriculture has been attempting to reduce nonpoint pollution, NPS, by various programs which had some success in the fifties, but, due to new technologies, the grass waterways approach has disappeared. New cultivation techniques are helping to reduce the NPS, but the users of the land are often unappreciative of the impact of the loss of soil and its consequences throughout the watershed and the subsequent deposition in lakes and deltas. We need a certain amount of erosion in order to maintain the coastal estuaries and delta regions. But, an excess amount is unnecessary for the natural ecosystem. The city dweller and the rural land manager has not been aware of the consequences of NPS as it affects the rest of the land resources, lakers and the ecosystem. Soil erosion fills in drainage ditches and causes lakes to lose their storage capacity and subsequent transformation into a wetland.

In the late 70s, residents of Mattoon approached the Water Resources Center for assistance in restoring their man-made lake which was rapidly filling with sedimentation. It was rather obvious that the problem was on the watershed and that there was a need for the lake owners to talk and communicate with the farmers in the area. This was a very difficult task since it involved a municipality, residents around the lake as well as the users of the land on the watershed. In fact, the city officials were reluctant to take any action because of the possible incresed taxes that might be invoked upon their city. Finally, after numerous meetings and publicity in the area, the region developed a concensus that there was a need to work together.

The city of Mattoon, who owned the lake, changed its attitude and began to support erosion control program on the Lake Paradise watershed. Farmers in the area received free water from the city, but, at the same time, were not aware that their soil was creating a water quality and supply problem for future users. Everyone was finally awakened to the issue and now there is a general concensus in the area that erosion is a problem for everyone. In order to maintain a high-quality water supply, the problem has to be resolved at the source.

The Water Resources Center has held three meetings on lake and watershed management (1, 2). Last fall, 200 people gathered in Springfield for a meeting which involved the Association of Soil and Water Conservation Districts, the EPA, owners and managers of lakes and scientists. As a result

of this meeting and other efforts in the state, there have been several new cooperative programs developed involving the integration of lake and watershed management.

Several years ago, David Daily, a student of soils at the Southern Illinois University at Edwardsville, undertook a personal project to determine the soil losses on the watershed of 4,000 acres for Holiday Lake, a nearby private lake, which was experiencing a heavy siltation (3). He went to the management of the lake and pointed out to them that the source of the problem was the watershed. The owners were receptive to his concerns, since they would have to spend $45,000 per year for dredging. Mr. Daily then explained the potential solutions to the landlords on the watershed. ASCS was approached for matching funds and the local Madison County Soil and Water Conservation District agreed to administer a sediment reduction plan regardless of the ASCS backing. As a result, the Holiday Lake Homeowners Association are providing funds to combat the erosion on critical portions of the watershed. This effort was initiated outside of government and is an excellent example of how people can work together in forming a local institution to solve local problems.

In October at the North American Lake Management Association meeting in Geneva, several papers are being presented which will describe additional recent examples of cooperative local efforts involving the Association of Soil and Water Conservation Districts, the Illinois EPA and the private sector.

The big issue in solving local problems is the compatibility of the individuals--everyone has to be unselfish and interested in the welfare of the entire community. Although there will be personal biases, these must be minimized in order to create an atmosphere whereby an institutional arrangement is established to solve local problems. Universities and state agencies can provide technical assistance, but they should not be involved in the legislation and demands of extravegant plans to resolve these problems. Many of the local erosion NPS programs could easily be solved by small units of government or individuals working together to protect the environment for the future of mankind.

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Technical Reports

APPLICATIONS OF GEOGRAPHICAL INFORMATION SYSTEM AND HYDROLOGIC MODELING TO AN AGRICULTURAL WATERSHED IN ILLINOIS

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ABSTRACT

The state-of-the-art appraisals of water quality impacts induced by non-point sources are difficult due to many factors. Lack of existing research data base, the diffusive characteristics of non-point source pollutants, and the technical difficulties relate the instream water quality to the topographic features, soil characteristics, land use, land cover, soil moisture conditions, and the management practices in the watersheds. In order to overcome these difficulties, the comprehensive monitoring and evaluation program was established in Highland Silver Lake Watershed.

The basic techniques may be divided into three categories: 1) field monitoring, 2) geographical information system, and 3) watershed modeling. These three components are mutually dependent. The field monitoring is described in an accompanying paper. The geographical information system consists of two basic types of data: cartographic coordinates which identify the points, lines, and polygons and attributes data which describe the characteristics of these features. The computer oriented GIS performs manipulation and analysis which include map overlay, buffer generation, dissolve boundaries (regroup and reclassify), tabular analysis and network analysis. The watershed modeling utilizes the data from field and GIS to evaluate the effects of watershed management practices. The MULTSED and AGNPS models were used. Results of the modeling effort are discussed.

Keywords: Nonpoint source pollution, geographical information system, watershed modeling, Best Management Practices, agricultural watershed, sediment load, runoff, water quality.

Introduction

Watershed modeling is a tool for hydrologic system synthesis, prediction, and optimization. Hydrologic modeling requires simplification or abstraction. There are two ways to pursue this abstraction: lumped or distributed systems (Chow, 1964; Haan et al., 1982). In lumped systems, non-uniform parameters such as rainfall, soils, land use, and topographic characteristics are averaged to obtain representative values for the entire drainage basin. On the other hand, a distributed system has the built-in capability of assembling the non-uniform parameters and simulating the interaction of these parameters. Thus, a distributed system should be more flexible for describing spatial-temporal dynamics of overland and channel flows. However, this type of model requires more detailed input data.

One way to overcome these difficulties is to use a geographical information system (GIS). A brief description of GIS is as follows.

Geographical Information System

In general, a geographical information system consists of two basic types of data: cartographic coordinates which identify the points, lines, and polygons; and attribute data, which describe the characteristics of these features. The computer-oriented GIS such as ARC/INFO (ESRI, 1984) can perform sophisticated manipulation and analyses which include map overlay, buffer generation, dissolve (regroup and reclassify), tabular analysis, and network analysis. The system contains a series of computer mapping and display capabilities for generating high quality cartographic displays. The user can specify size and scale to produce the desired map outputs.

Highland Silver Lake Project: An Example

The Highland Silver Lake Project is one of thirteen experimental projects for the National Rural Clean Water Program in the United States. The project has been in operation since September 1980. The project consists of field data collection and evaluation of different land management practices applied in the watershed. Since land management practices consist of different tillage practices and different crop sequences on different soil and topographical conditions, the collected field data alone will not be enough to determine the effects of management practices in the entire project area. Therefore, utilization of a watershed model is needed to evaluate the impacts of land management practices on water quality and on stream and reservoir sedimentation, as well as the costs and benefits of the program.

Data Base of Highland Silver Lake Watershed

In the Highland Silver Lake Project, the watershed data base consists of a hydrographic map (stream networks and water bodies), soil maps, land use data, and slope map. The hydrographic map was digitized from a standard U.S. Geological Survey 7-1/2 minute topographic map with the recent aerial photographs as supplemental data sources. The soil map was digitized from published county soil survey maps. The land use data were obtained from recent aerial photographs. The slope map was developed on the basis of topographic maps published by the U.S. Geological Survey. Watershed and sub-basin boundaries and locations of monitoring stations were also coded in the GIS.

Agricultural Non-Point Source Pollution Model (AGNPS)

This model was developed by the North Central Soil Conservation Research Laboratory, Agricultural Research Service, USDA (Onstad and Young, 1983). AGNPS is a single event, deterministic, and distributed model. It configures the watershed into small square subareas called "cells." It is at this level that all characteristics are established and calculations made.

Figure 1 shows a subwatershed of the Highland Silver Lake watershed. The model is composed of the following parts: (1) runoff component, (2) channel flow, (3) erosion, (4) sediment transport, (5) nutrient transport, and (6) feedlot.

The basic output from AGNPS includes a hydrology component which estimates both volume and peak runoff; a sediment component which estimates upland erosion, gully erosion, and sediment yield; and a nutrients component which estimates nitrogen and phosphorus yields in terms of concentration and load. The output can be examined on a cell basis or for an entire watershed. Sediment analyses are broken down into particle-size classes.

The second model is the Multiple Watershed Storm Water and Sediment Runoff Simulation Model (MULTSED) (Simons et al., 1978 and 1981).

MULTSED Model

This model is also a single event, distributed, deterministic model. The model contains two basic components: a hydrologic and hydraulic routing component and a sediment component. The calculations are conducted in three basic units: (1) two-plane and single channel, so called Wooding-plane subwatershed as shown in figure 2 (Wooding, 1965); (2) channel unit; and (3) a single-plane unit. The number and size of these units for a specific watershed can be selected by the users.

Use of GIS for Input Data Preparation

The two selected models that have been described represent two types of model configurations. AGNPS represents the grid-cell system. MULTSED represents the irregular configuration which reflects the hydrologic units such as Wooding-plane, single overland flow, and single channel.

The input data for the AGNPS model consist of: (1) cell identification number; (2) receiving cell identification number; (3) SCS curve number; (4) land slope in percent; (5) land slope shape factor; (6) field slope length in feet; (7) channel slope in percent; (8) channel side slope in percent; (9) Manning's roughness coefficient for the channel; (10) soil erodibility factor; (11) cropping factor; (12) practice factor; (13) surface condition constant; (14) aspect; (15) soil texture number; (16) fertilization level; (17) availability factor; (18) point source level; (19) gully source level; (20) chemical oxygen demand factor; and (21) impoundment factor.

In order to compile these input data from the GIS data base, the first step is to construct a "template" which delineates the watershed into sub-units. In the case of the AGNPS model, the grid-cell system is the template. The

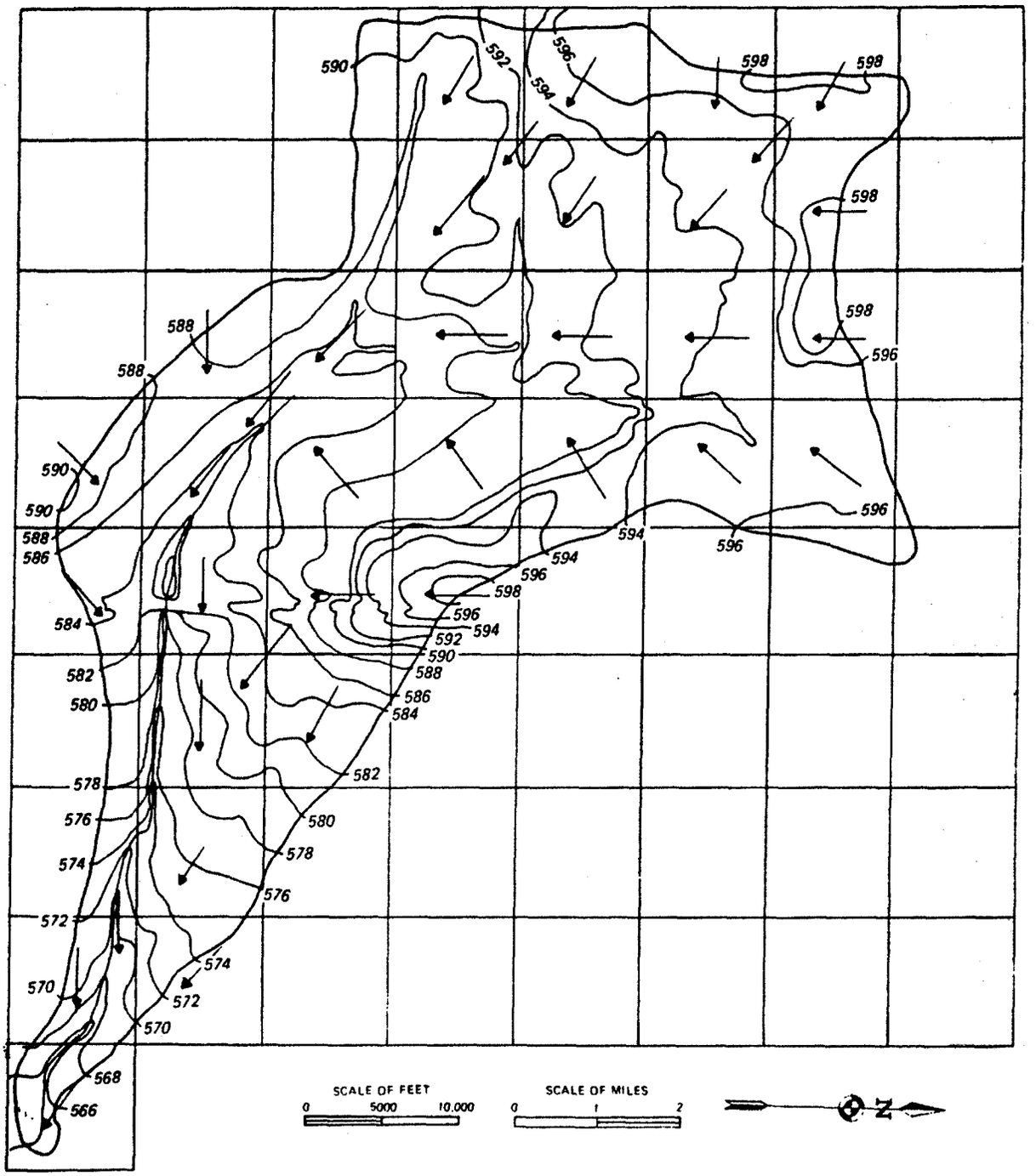
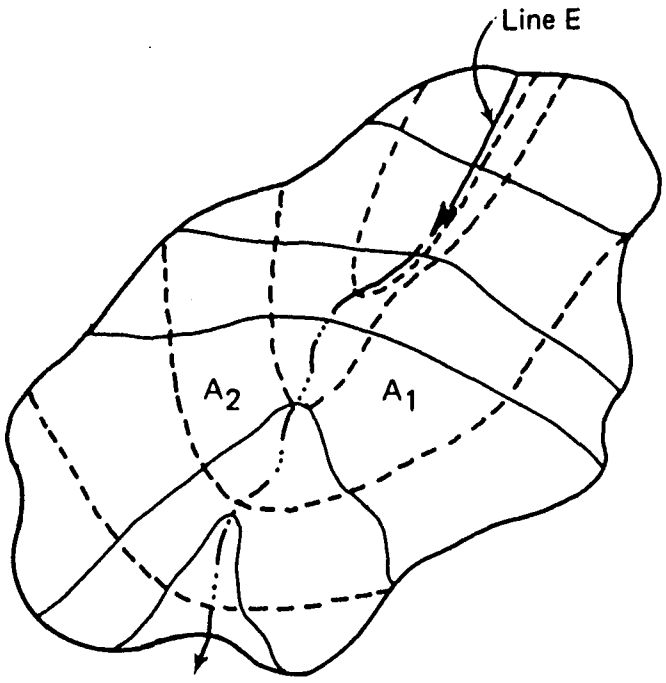
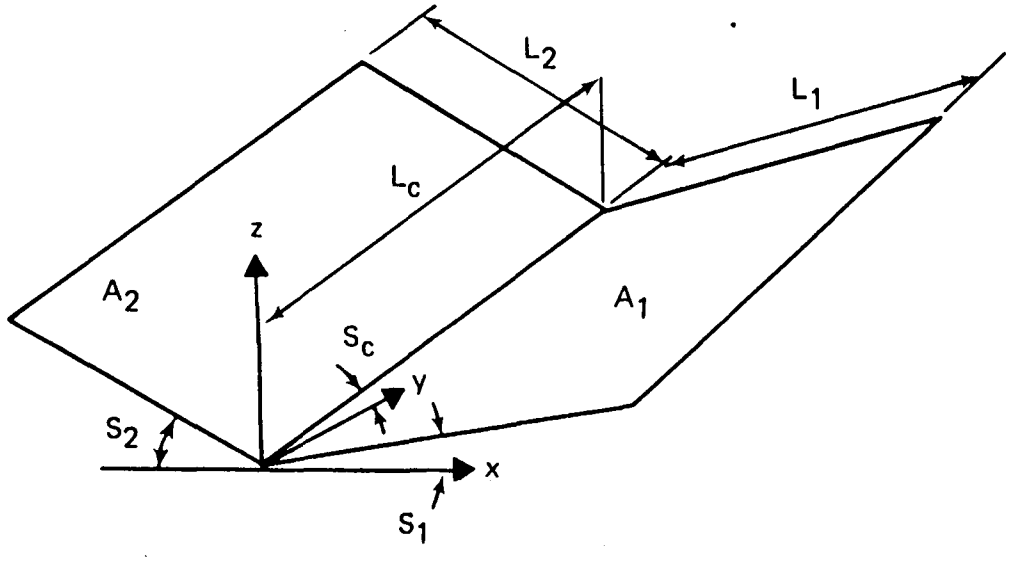


Figure 1. An example of grid-cell system of AGNPS model



(a) Original Subwatershed Topographic Map



(b) Wooding Plane Representation

Figure 2. Watershed configuration of MULTSED model

computer software is able to retrieve the attribute data and the sub-units from the data base. However, few items in the input data require field inspection to acquire the necessary data. Similarly, the data can be retrieved for the MULTSED model.

Results

AGNS was run for field site 5 of the Highland Silver Lake watershed using a 10-acre grid system as shown in figure 1. A 4.4-in. rainfall storm was considered which represents an extreme storm for this watershed in the last three years. Table 1 shows the results given by the AGNPS and MULTSED model for the same event.

Table 1. AGNPS and MULTSED Model Results for a 4.4-in. Rainfall Event in Field Site 5 of the Highland Silver Lake Watershed

	AGNPS	MULTSED
Runoff volume (in.)	2.8	3.6
Runoff ratio (%)	64.0	82.0
Peak runoff rate (cfs)	382.0	283.0
Sediment yield (tons/acre)	0.327	0.089

The primary results indicated that runoff volumes and runoff ratio show higher values by using MULTSED model, but peak runoff rate and sediment yield show higher values by using AGNPS model. Further verification with field observation is being conducted.

Three events during 1982 and 1983 were selected for the calibration of the MULTSED model. The events were selected so that they fairly represent the average characteristics of the events for these years. The events considered were: 1) May 31, 1982, with a rainfall of 0.88 in. and a duration of 4 hours; 2) June 4, 1982, with a rainfall of 0.4 in. and a duration of 11 hours; and 3) March 18, 1983, with a rainfall of 1.34 in. and a duration of 16.5 hours.

The parameters used in the calibration were: 1) the hydraulic conductivity, 2) initial soil moisture, 3) rainfall splash detachment coefficient, and 4) roughness coefficients. These parameters were adjusted by doing sensitivity analysis on their value ranges until the simulated and measured runoff and sediment concentration hydrographs were matched. Other parameters such as ground and canopy cover, porosity, average suction head, final soil moisture, interception by ground cover or canopy, and soil temperature were kept constant. Table 2 shows the results of the three selected events. A selected runoff and sediment hydrographs are shown in figure 4.

Conclusions

1. Application of the MULTSED model for a portion of the Highland Silver Lake watershed has been performed. Satisfactory simulation of runoff and sediment hydrographs was obtained. Very little is known on the relationships between the detachment coefficients and the land use and watershed management practices in the MULTSED model. An attempt to find these relations will be done when the entire basin is studied.

Table 2. Runoff and Sediment Yield for Field Site 5 and Gaging Station 3 of Highland Silver Lake Predicted by MULTSED Model

Event	Rainfall (in.)	Measured runoff (in.)	Computed runoff (in.)	Error (%)	Runoff ratio (%)	Measured sediment (tons)	Computed sediment (tons)	Error (%)
Subarea A								
May 31, 1982	0.88	0.198	0.204	3.04	22.5	N.A.	1.15	-
June 4, 1982	0.40	0.093	0.097	3.85	23.3	0.55	0.48	-13.5
March 18, 1983	1.34	0.352	0.355	0.64	26.3	2.44	2.23	-8.7
Subarea B								
May 31, 1982	0.88	0.213	0.222	3.99	24.2	38.32	27.10	-27.0
June 4, 1982	0.40	0.053	0.053	0.06	13.3	4.60	4.61	2.1
March 18, 1983	1.34	0.550	0.521	-5.2	41.0	80.37	80.58	0.25

N.A. = not available

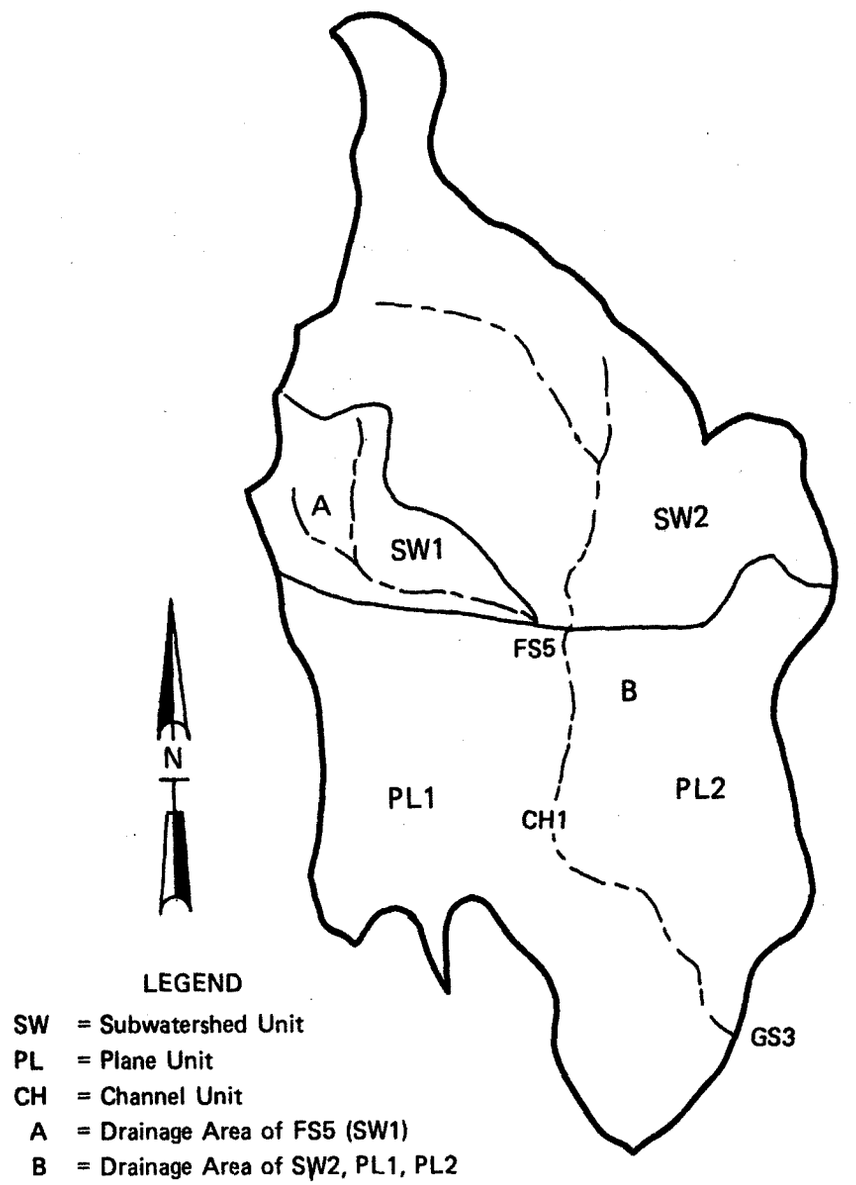


Figure 3. Gaging Station 3 Sub-watershed of Highland Silver Lake, Highland, Illinois

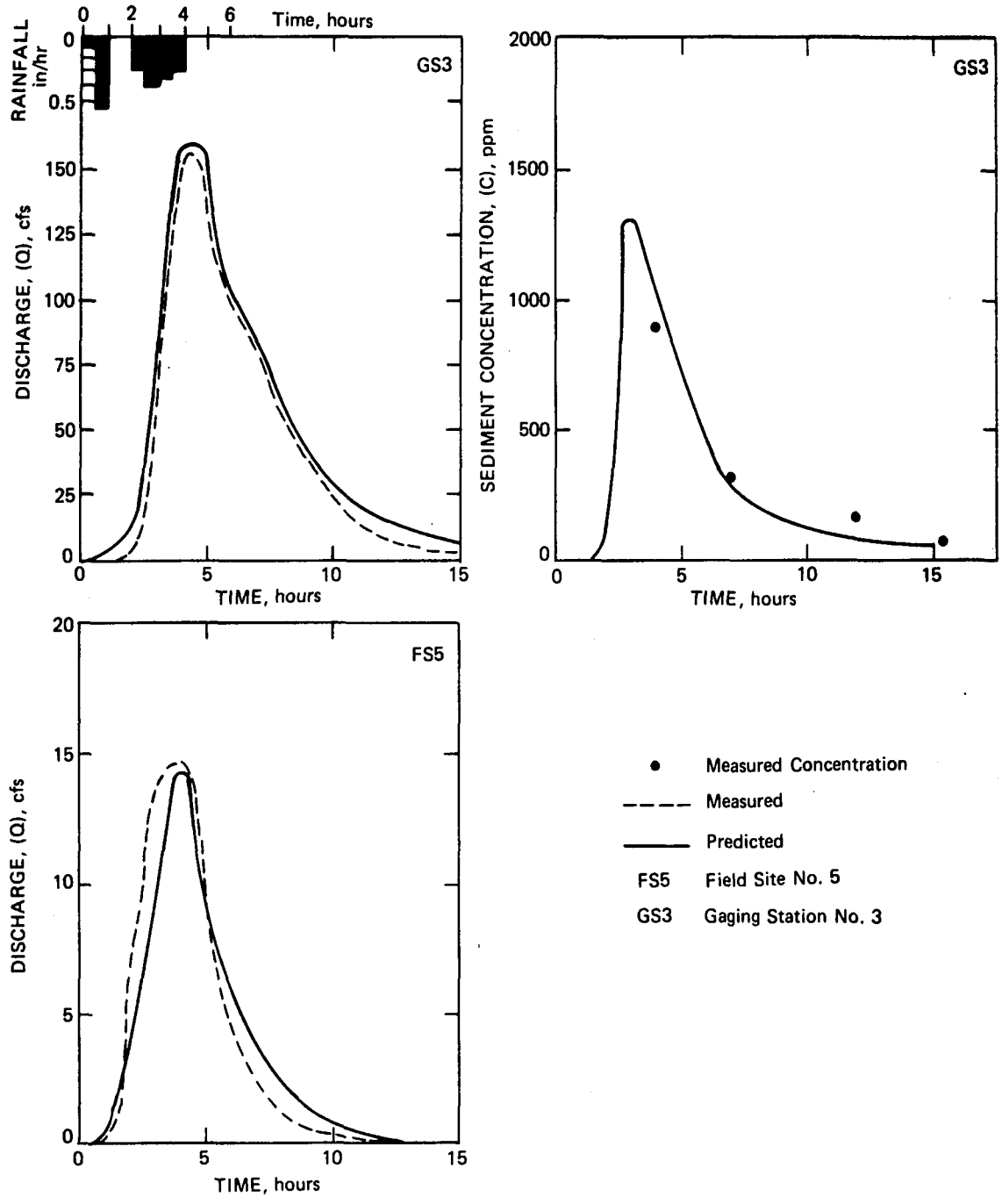


Figure 4. Runoff and Sediment Hydrographs of March 18, 1983 of Highland Silver Lake Watershed, Highland, Illinois

2. A methodology illustrates that a geographical information system can provide an efficient way to compile necessary input data for complex distributed hydrologic models.

3. The development of the data base in a geographical information system is independent of a specific model. The data can be retrieved from GIS to meet the requirement of different geometric configurations of watershed models.

4. The sensitivities of various geometric delineations and hydrologic processes in a watershed can be evaluated by utilizing a geographic information system.

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Delineating Sources of Agricultural Nonpoint Source Pollution

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INTRODUCTION

Fluvial sediment and associated nutrients and chemicals are now acknowledged to be a major cause of contamination in river and lake systems throughout North America (7, 8). A prime source of these contaminants is sheet and rill erosion from agricultural cropland (19). Such soil erosion is widely distributed over the landscape, but the rates at which soil erodes and yields sediment to streams can be highly variable from field to field and from watershed to watershed (15). Therefore, although soil erosion has been termed a nonpoint source of pollution, most basins have more or less specific locations which yield much of the sediment and associated contaminants. The design and selection of cost-effective non-point source pollution control strategies requires delineation of key source areas.

Recent erosion/sedimentation research studies at Guelph have led to the development of a method for estimating sources and rates of sediment yield in small agricultural watersheds (2, 6). Examples of the application of the method to two agricultural watersheds in Southern Ontario are presented in this paper, to reveal the utility of watershed modelling for delineating soil erosion and sediment yield source areas.

LITERATURE REVIEW

The notion of sediment contributing areas, i.e. that prime sources of sediment in agricultural regions constitute an area less than and in some cases much less than the total watershed area, has received considerable attention in recent years (3, 13, 16, 17). Much in the erosion/sedimentation literature, although not specifically focussed on the definition of sediment sources, provides useful background for the location of significant sediment sources. Some field-scale models provide estimates of annual erosion (22), and others predict annual sediment loads directly (9, 13, 22). Another category of models predicts annual amounts of erosion and sediment on the

basis of the Universal Soil Loss Equation (23) and a functional relationship for watershed delivery ratio or field transport factors (10, 14, 16, 17, 18, 21).

Single storm event and continuous erosion/sediment/chemical models have also been developed for both the field scale (11) and the watershed scale (1). This group is more descriptive of the physical and chemical processes involved and can be used to delineate key sources. However, the large volume of input data required for these models, some of which is not always readily available, is often a constraint on their use.

Therefore, although there is now an extensive literature on soil erosion and fluvial sedimentation which provides useful background and detailed distributed models, there has not been a method developed with the prime purpose of delineating key source areas. Further, although there has been considerable evidence that stream sediment loads are usually highly variable throughout the year (2, 5, 12, 19, 20), the variation of seasonal sediment sources has not been addressed. GAMES - the Guelph model for evaluating the impact of Agricultural Management systems on Erosion and Sedimentation - has been developed for the purpose of predicting on a seasonal basis the magnitude and location of soil loss by water erosion and subsequent sediment yield

METHODOLOGY

Several criteria have been kept in mind during the development of GAMES. The model was to take account of seasonal variations in soil loss and sediment transport mechanisms, but was not to become encumbered with the data requirements of an event on continuous model. The basic computational land units were to be field size, with the intent of applying the model to small agricultural watersheds. Although a predictive capacity was deemed to be desirable, the prime purpose of the model was to distribute sediment yields

throughout a watershed on a quantitative and objective basis.

The resulting model, GAMES, has been suitably constructed for the computation of seasonal potential soil loss (the soil loss component) and seasonal sediment yield (the delivery component) for field-size cells in small agricultural watersheds. The soil loss component is based on the Universal Soil Loss Equation (23) has been modified for seasonal application. That is,

$$A_S = R_S C_S K_S LS.P_S$$

where A_S = seasonal soil loss per unit area, R_S = seasonal rainfall factor, C_S = seasonal land use or cropping factor, K_S = seasonal soil erodibility factor, LS = slope gradient and slope length factor, and P = seasonal erosion control practice factor. The sediment yielded by each field unit to downstream field and stream units is determined from the product of the seasonal potential soil loss and the seasonal delivery ratio for the unit. The seasonal delivery ratio, or percentage of the potential soil loss which can be expected to be delivered downstream in a given season, is determined from the expression,

$$DR_S = \alpha \left[\frac{1}{n_S} . S . H_c . L \right]^{\beta}$$

where $0 \leq DR_S \leq 1$; n_S = seasonal surface roughness coefficient, as indexed by Manning's n; S = cell slope; H_{cs} = seasonal hydrologic coefficient, an index regarding the probability and depth of overland flow in a season; L_S = seasonal length of overland flow path; and α, β = calibrated parameters.

Application of the soil loss and delivery components initially involves the development of a composite overlay of land use, soils, and land slope to divide the watershed area into land units each of which is characterized by a single land use, a single soil type, and a single slope class. Then values of the various variables in the USLE (23) and DR (2) expressions are determined

and assigned to each unit. Determination of the α and β parameters for a watershed involves an optimization routine to compare the accumulated watershed sediment load calculated by means of GAMES for assigned values of and α and β with a measured sediment load or one estimated by another means. Output for a calibrated application is generated on a unit by unit basis for each season considered, and includes potential soil loss, potential soil loss per hectare, delivery ratio to the adjacent downstream unit, delivery ratio to the nearest stream, sediment yielded to the stream, and sediment yielded per hectare.

With the selection of soil loss and sediment yield tolerance levels, it is possible to extend the application of GAMES into the area of problem identification (4). The framework shown in Table 1 has been suggested on the basis of distinct definitions for erosion and sediment problems. An erosion problem is considered to be fundamentally a source problem, the loss of soil creating hardships such as decreases in agricultural productivity, loss of tillable land, damage to buildings, roads, equipment etc. A sediment problem involves downstream situations e.g. siltation of streams and reservoirs, pollution of fish spawning areas, increased water treatment costs. Areas of a watershed which fall into problem category I may be expected to exhibit both high soil erosion rates and high sediment yield rates. Areas in category II have high erosion rates, but contribute relatively little soil to downstream locations. Category III includes those areas which have relatively low soil loss rates, but from which most of the eroded soil moves into the stream channels contributing significantly to sediment problems. Category IV is the "no problem" category.

For the sake of illustrating the utility of a model for delineating soil loss and sediment yield source areas, GAMES has been applied to two small agricultural watersheds located in the Thames River Basin of Southern Ontario.

Table 1: Framework for soil erosion and sediment problem identification

Problem Category	Erosion* Problem	Sediment** Problem
I	x	x
II	x	-
III	-	x
IV	-	-

* Agricultural productivity is adversely affected

** Sediment yield and associated nutrient contribution to the stream is excessive.

The Stratford/Avon Watershed is comprised of 434 hectares of rolling upland soils, cropped predominantly in continuous corn with fall ploughing and few if any conservation practices. The Big Creek Watershed is 2141 hectares in extent on quite flat lowland soils. It is almost totally in row crops (e.g. continuous corn, beans, vegetables) which are fall ploughed.

SAMPLE RESULTS

Sample output maps from the GAMES application are presented in Figures 1 through 6. Figures 1 and 2 reveal the spatial pattern of spring potential soil loss quantities for the upland and lowland watersheds, respectively. The estimated average spring gross erosion per unit of watershed area is not large for either basin, being 2.5 tonnes per hectare for the lowland basin and 3.5 tonnes per hectare for the upland basin. However, the spring erosion estimates are quite variable in the upland Stratford/Avon Watershed, ranging from essentially zero to 25 tonnes per hectare. The range is much smaller across the lowland Big Creek Watershed, varying only from 0 to 3 tonnes per hectare. It is also evident from Figure 1 that a major portion of the eroded soil in the upland watershed moves within a small percentage of the basin. For example, 92 percent of the spring sheet and rill erosion volume is estimated to occur in 47 percent of the watershed area. The areal distribution of soil loss is more uniform in the lowland basin (Figure 2). About 56 percent of the landscape contributes 64 percent of the sheet and rill erosion.

Sample spatial distributions of sediment yield from the upland and lowland agricultural basins are presented in Figures 3 and 4. Similar to the erosion picture presented above, but more pronounced, the great majority of the spring sediment load leaving the upland watershed is estimated to emanate from a small percentage of the watershed area. For example, 83 percent of the load is generated in 14 percent of the basin. Indeed, such a watershed is

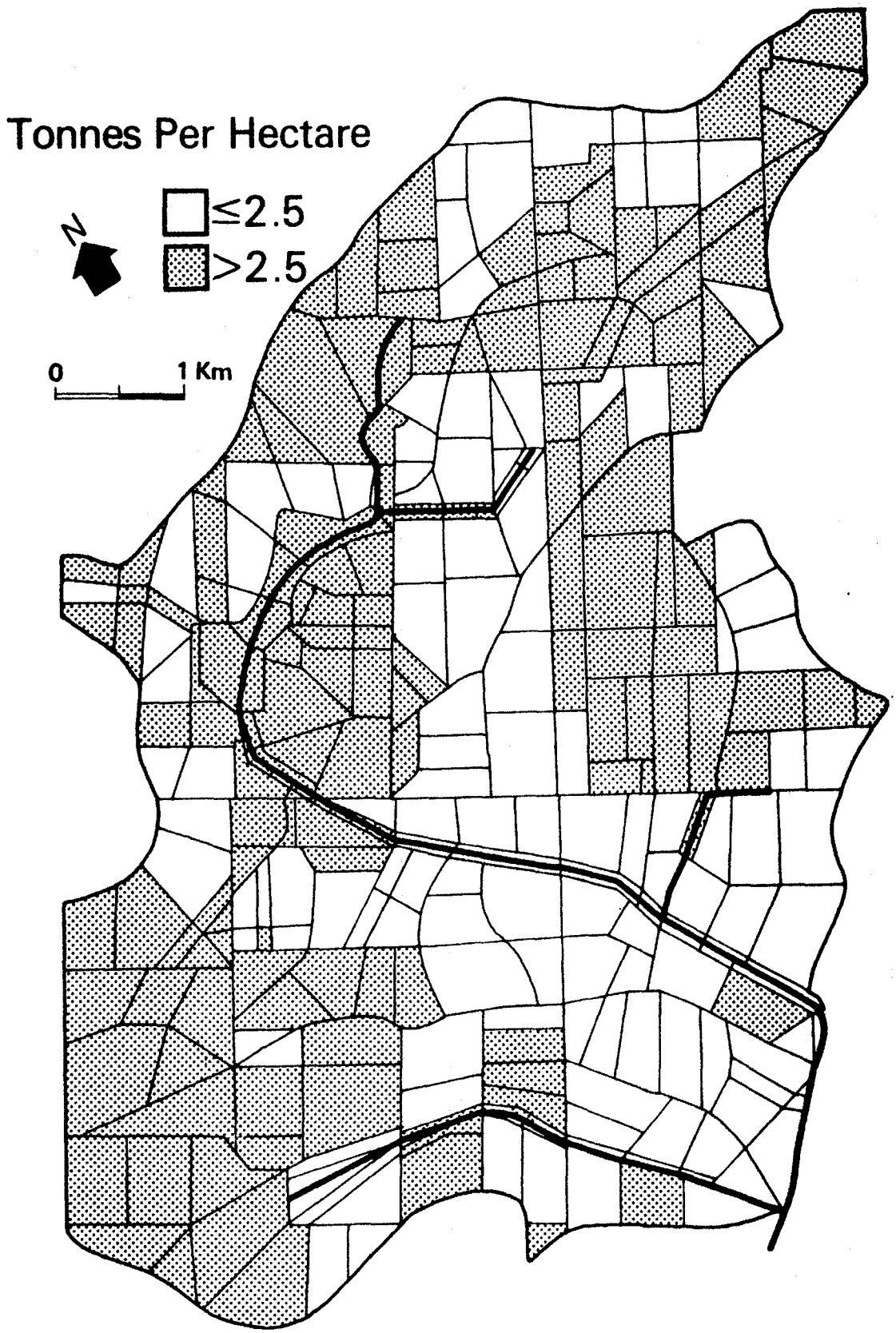

Figure 1. Spatial variation of soil erosion on upland watershed for spring conditions.

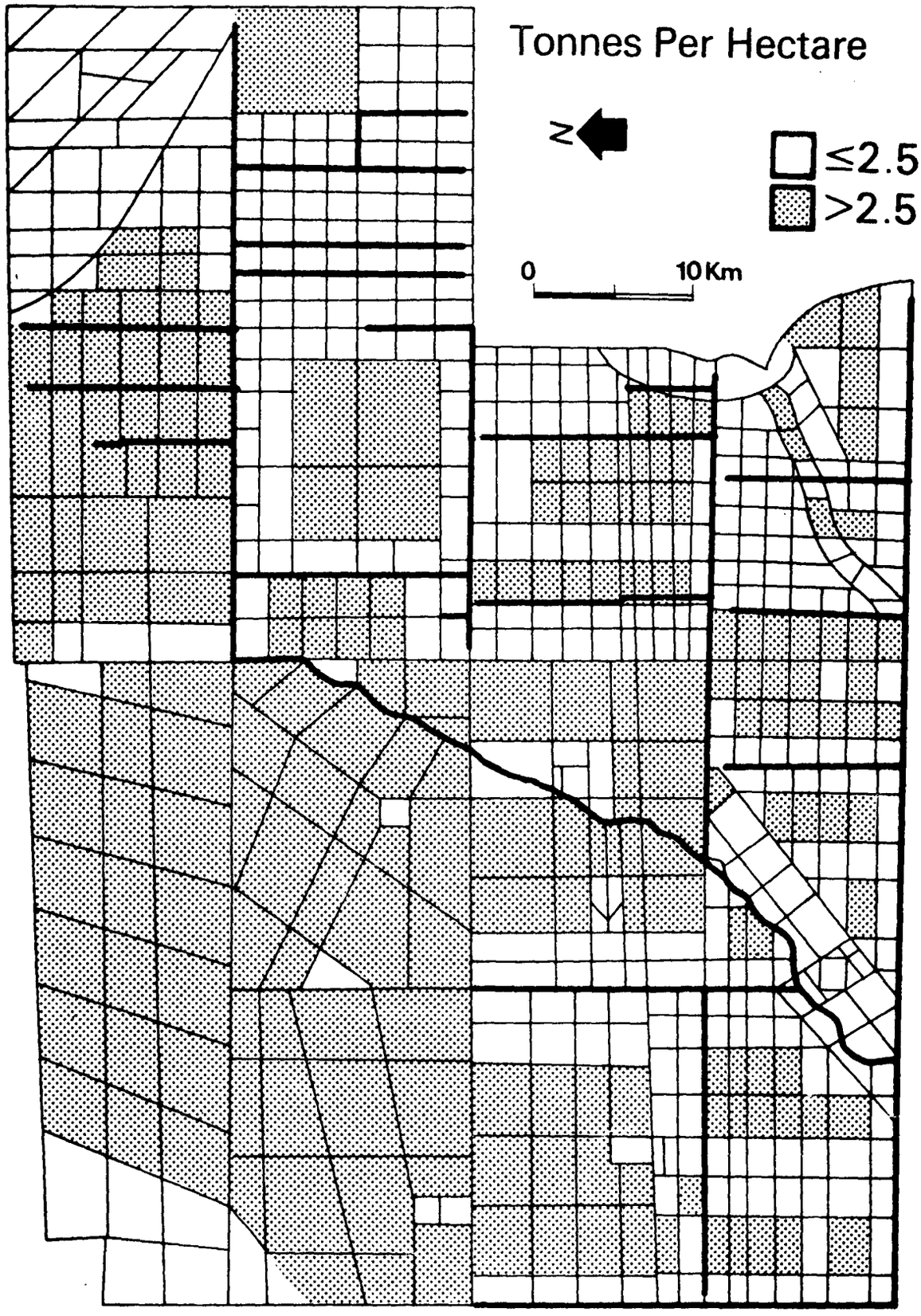

Figure 2. Spatial variation of soil erosion on lowland watershed for spring conditions.

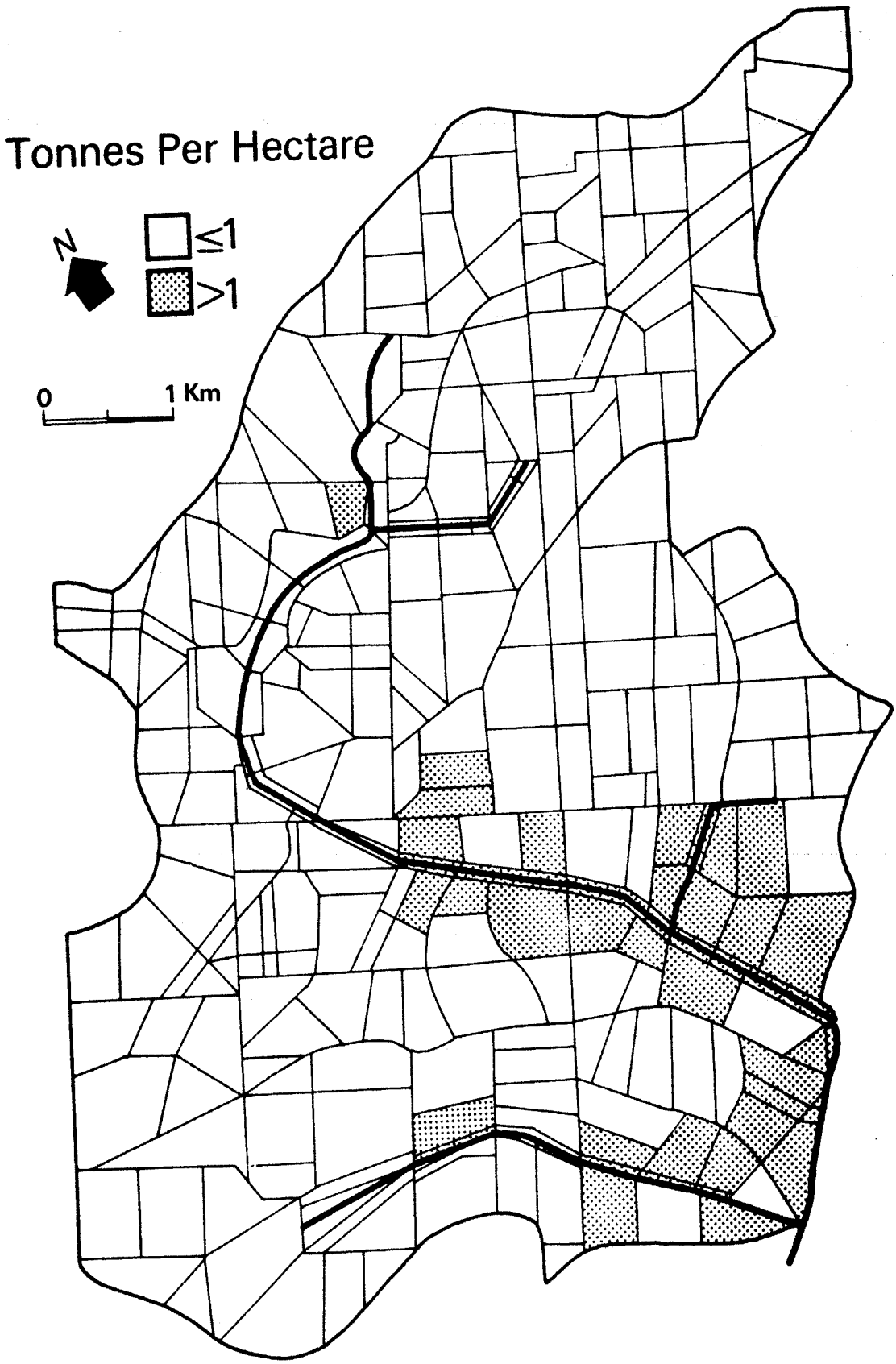

Figure 3. Spatial variation of sediment yield on upland watershed for spring conditions.

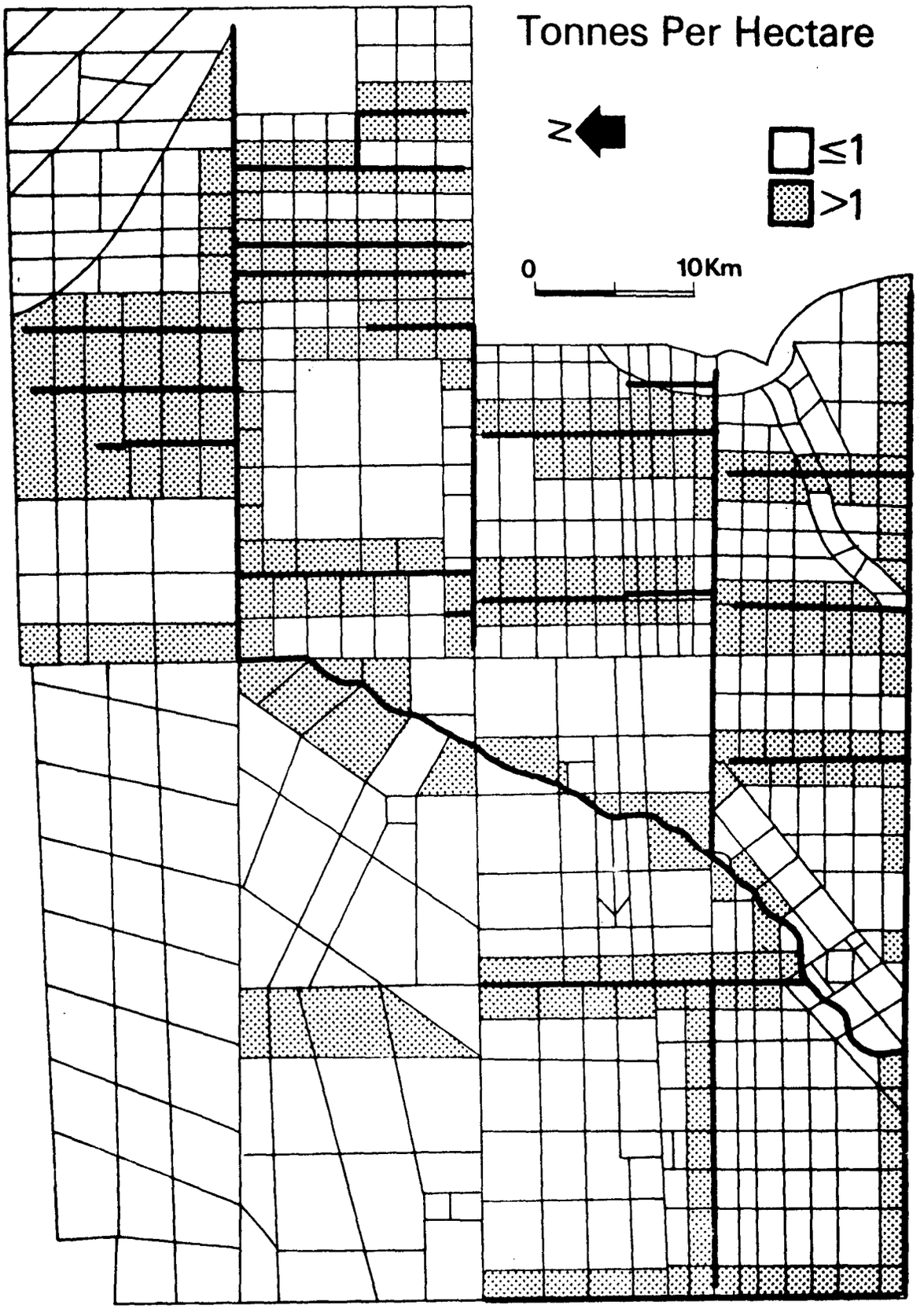

Figure 4. Spatial variation of sediment yield on lowland watershed for spring conditions.

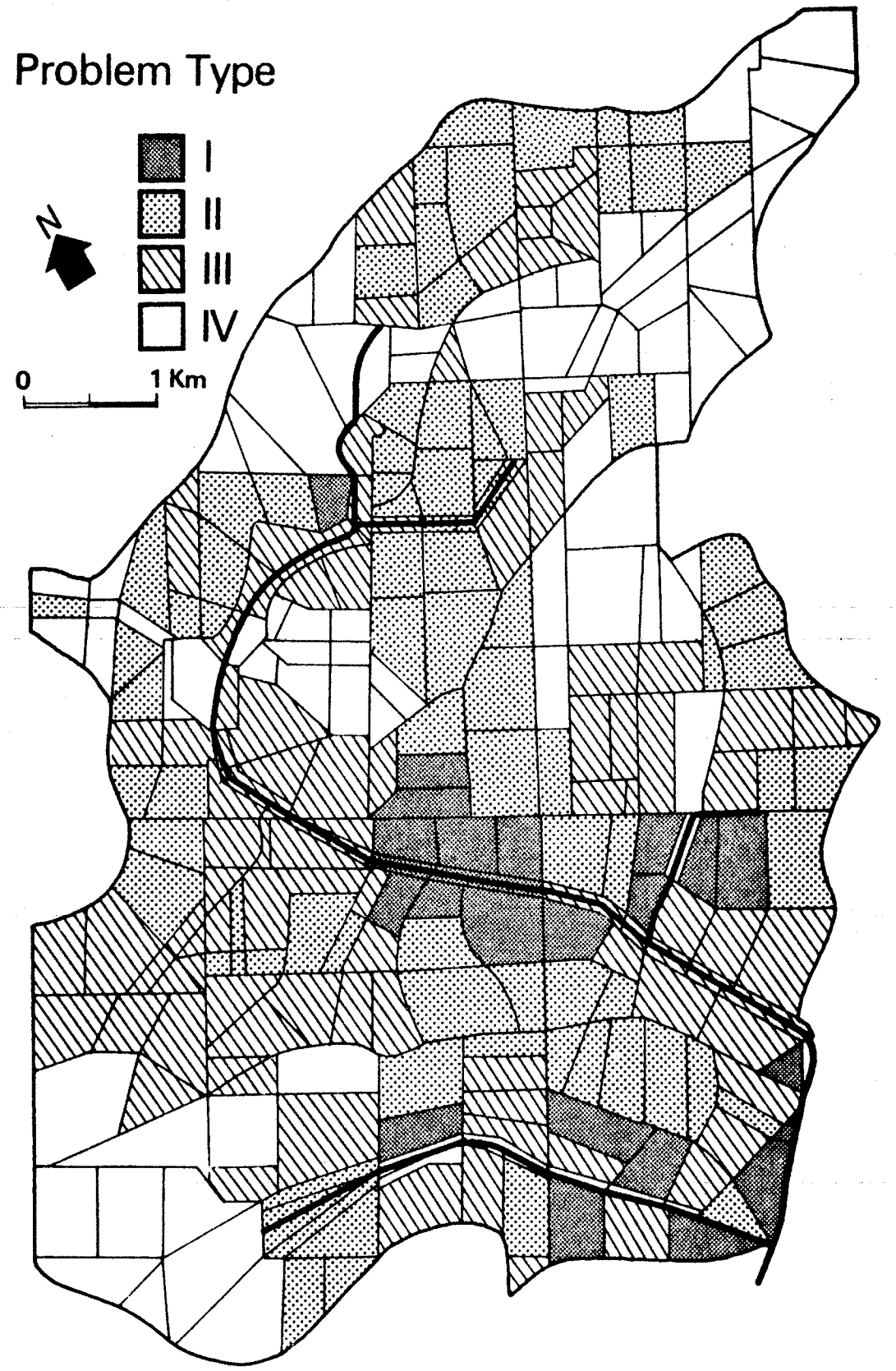

Figure 5. Spatial distribution of soil erosion and/or fluvial sediment areas on upland watershed for spring conditions.

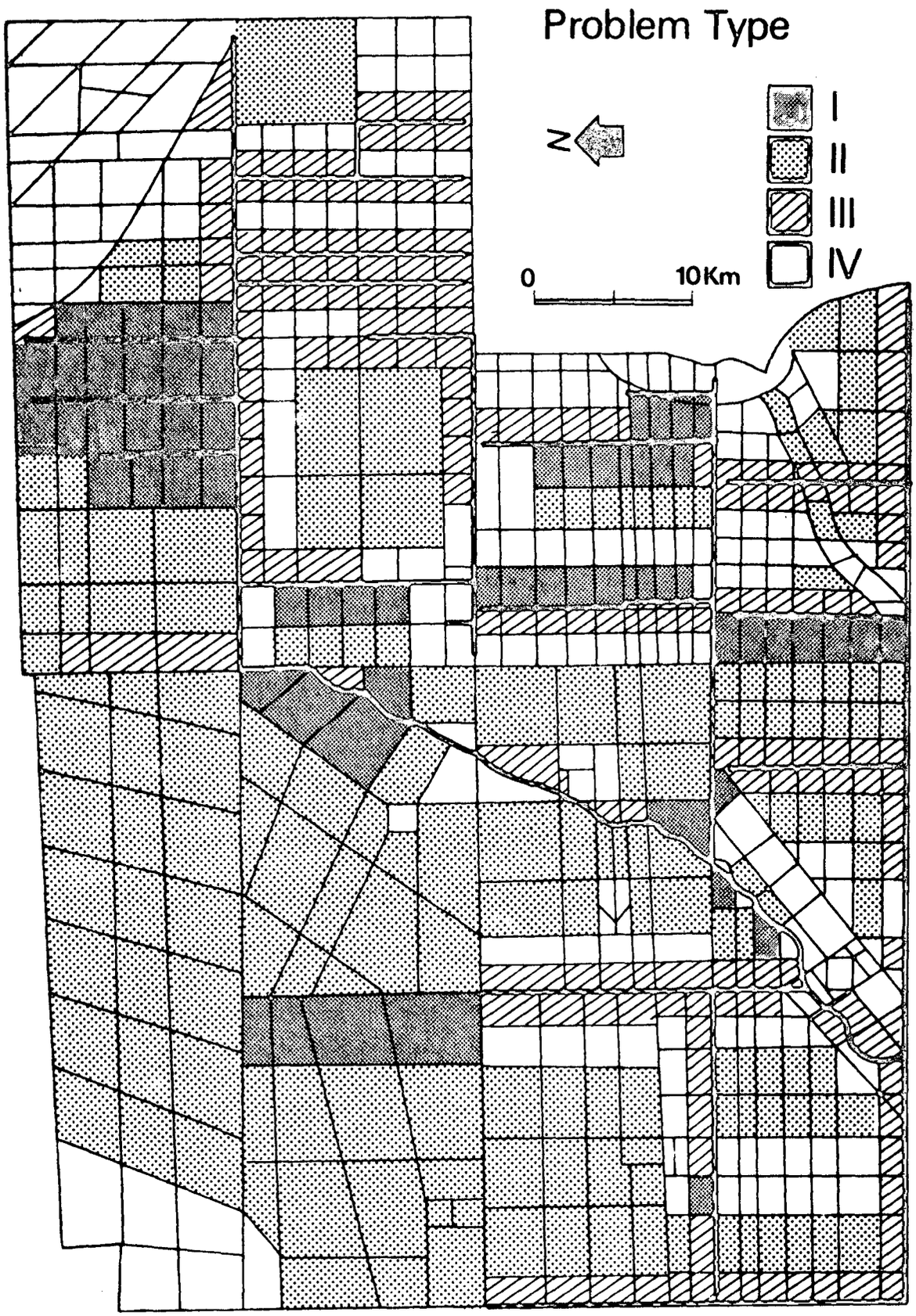

Figure 6. Spatial distribution of soil erosion and/or fluvial sediment areas on lowland watershed for spring conditions.

characterized by rather distinct sediment sources. On the other hand, the lowland watershed is seen to generate its sediment load rather uniformly across the basin, and sediment yield hotspots are not evident (Figure 4).

Figures 5 and 6 present maps of the four soil erosion and sediment problem categories on the basis of a spring erosion tolerance level of 2.5 tonnes per hectare and a spring sediment yield tolerance level of 1.0 tonnes per hectare. A quantitative summary relating to these figures is given in Table 2.

The sample results regarding erosion and sedimentation problem identification reinforce the results discussed previously, but also reveal additional significant points. It is clear from Table 2 that 47 percent of the upland watershed and 75 percent of the lowland watershed contribute significantly to erosion and/or sediment problems. In and upland Stratford/Avon watershed 14 percent of the basin exhibits almost half of the soil loss and is the source of more than three quarters of the sediment load. Only 47 percent of the basin is estimated to have spring erosion rates greater than the defined tolerance level, accounting for 92 percent of the total basin soil loss; while only 14 percent of the basin is estimated to serve as a prime sediment source, contributing 83 percent of the downstream loads. More than half of the landscape of this basin exhibits potential spring soil loss and sediment yield less than the defined erosion and sediment yield tolerance rates.

The lowland Big Creek watershed, typical of many level lowland watersheds in Southern Ontario, has more than half of its area in erosion problem categories. Fifty-six percent of the basin has spring erosion rates greater than the defined tolerance level. About one third of this basin is estimated

Table 2. Quantitative identification of soil erosion and sediment problems in selected agricultural watersheds

Problem Category	Comparative Percentages	Stratford/Avon Watershed	Big Creek Watershed
I	% of Area	14	12
	% of Soil Loss	45	13
	% of Sediment Load	83	18
II	% of Area	33	44
	% of Soil Loss	47	51
	% of Sediment Load	15	34
III	% of Area	0	19
	% of Soil Loss	0	17
	% of Sediment Load	0	30
IV	% of Area	53	25
	% of Soil Loss	8	19
	% of Sediment Load	2	18

to provide a prime source of sediments contributing to half of the downstream loads. It should be noted that 19 percent of the basin exhibits potential soil loss less than the defined erosion rate tolerance. Nonetheless, the same area contributes about one third of the downstream sediment load at sediment yield rates above the defined sediment yield rate limit.

CONCLUSIONS

A watershed soil erosion and fluvial sedimentation model, GAMES, has been shown to provide a very useful tool not only for evaluating the seasonal and spatial variability of field potential soil loss and sediment yield in an agricultural watershed but also for delineating spatially the distribution of erosion and sediment problems. For selected soil loss and sediment yield tolerances, the approach affords in map and summary statistics form information regarding the location and extent of areas contributing to (i) both erosion and stream sediment problems, (ii) only erosion problems, (iii) only sediment problems, and (iv) no problems. Such description and delineation of agricultural nonpoint sources is proving to be of great assistance in the planning and targetting of nonpoint pollution control programs in rural areas.

ACKNOWLEDGEMENTS

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COUPLING NONPOINT POLLUTION AND WATER QUALITY MODELS: AN EXAMPLE FOR THE GREEN BAY - FOX RIVER WATERSHED

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ABSTRACT

Green Bay is a major gulf located in the northwest corner of Lake Michigan which has been cited as one of the major water quality problem areas in the Great Lakes. Historically, interest in water quality degradation has been limited to the lower Fox River and extreme southern Green Bay, the site of municipal and industrial pollutant discharges. Recent surveys indicate dissolved oxygen depletion far out into the bay, beyond the immediate influence of the Fox River's organic loading. Attention is now focused upon nonpoint sources of phosphorus as stimulants for primary production and the ultimate cause of oxygen depletion problems. The Fox River contributes 78% of the annual tributary load of total phosphorus to Green Bay. Of this amount, only 3% is attributable to the Green Bay Metropolitan Sewerage Authority, the region's largest point source discharger of phosphorus.

Nonpoint source loads of total phosphorus are estimated using a microcomputer model which considers land use, soil texture, and the phosphorus loading associated with each location in the land use/soil texture matrix (UALs, unit area loads, kgP/ha·year). Land use classifications and soil textures are developed specifically for the Fox River watershed and UALs are derived from recent studies of nonpoint source loadings (PLUARG, 208 Areawide Water Quality Management Planning Program). Annual total phosphorus loads to Green Bay from the Fox River are calculated for baseline (= existing) conditions and six additional land use scenarios as the sum of the products of the land use/soil texture and unit area load matrices for each component sub-basin in the Fox River watershed.

Load estimates are input to a steady-state mass balance phosphorus model which calculates the phosphorus concentration along the major longitudinal axis of Green Bay. Trophic levels corresponding to points along the phosphorus gradient are established using a trophic index. A marked gradient in trophic conditions exists along the bay which is sensitive to variations in land use practices impacting nonpoint phosphorus loads. The hypothetical scenarios help demonstrate the utility and of interactive, coupled nonpoint pollution and water quality microcomputer models.

Keywords: Nonpoint source pollution model, water quality model, abatement of pollution from agriculture, phosphorus, eutrophication, Green Bay, Lake Michigan, Great Lakes

INTRODUCTION

Modeling approach

Microcomputer models for the estimating of nonpoint source pollutant loadings and receiving water quality response may be coupled and used to examine the impact of land use activities on water quality. This manuscript focuses on nonpoint source loads of phosphorus to Green Bay, Lake Michigan and their relation to the trophic state gradient within that system. Seven hypothetical land use managment scenarios are described to illustrate the application of interactive planning and management tool. The methodology is versatile and may be modified to address other substances and their impact through construction of pollutant-specific unit area load tables and quantification of sources and sinks for the parameter of interest.

Green Bay and the Fox River

Green Bay is a gulf approximately 160 km long and 22 km wide, located in the northwest corner of Lake Michigan (Figure 1). The bay has a mean depth of 15.8 m, a volume of 67 km^3 and a residence time of 6 years (Mortimer, 1978). Historically, industrial and municipal discharges of oxygen-demanding substances and plant nutrients have led to severe dissolved oxygen depletion during summer in the lower Fox River and extreme southern Green Bay. Recent evidence of hypolimnetic oxygen depletion over much greater portions of the bay has resulted in the current interest in nonpoint sources of algal nutrients, specifically phosphorus. The Fox River is the single greatest hydrologic and pollutant source for Green Bay, contributing 45% of the annual tributary flow and 78% of the annual tributary total phosphorus load (Roznowski and Auer, 1984). Strong longitudinal gradients in trophic status are set up in response to the Fox River phosphorus loading -- hypereutrophic conditions exist in the southernmost region near the river's mouth, while oligotrophic conditions prevail in the northern reaches near the junction with Lake Michigan.

NONPOINT SOURCE MICROCOMPUTER MODEL

Model Structure

The microcomputer model used to assess average annual total phosphorus loadings from nonpoint sources within a given drainage system is based on three critical assumptions:

1] The total phosphorus load generated by surface runoff from a given site may be represented by a unit area load (UAL) -- the mass of phosphorus contributed per unit area of land per unit time.

2] Under identical climatological conditions, the UAL for a given area is principally determined by the predominant land use and soil texture within that area.

3] UALs are derived to reflect nonpoint source loadings for an "average" year of wetness (total precipitation).

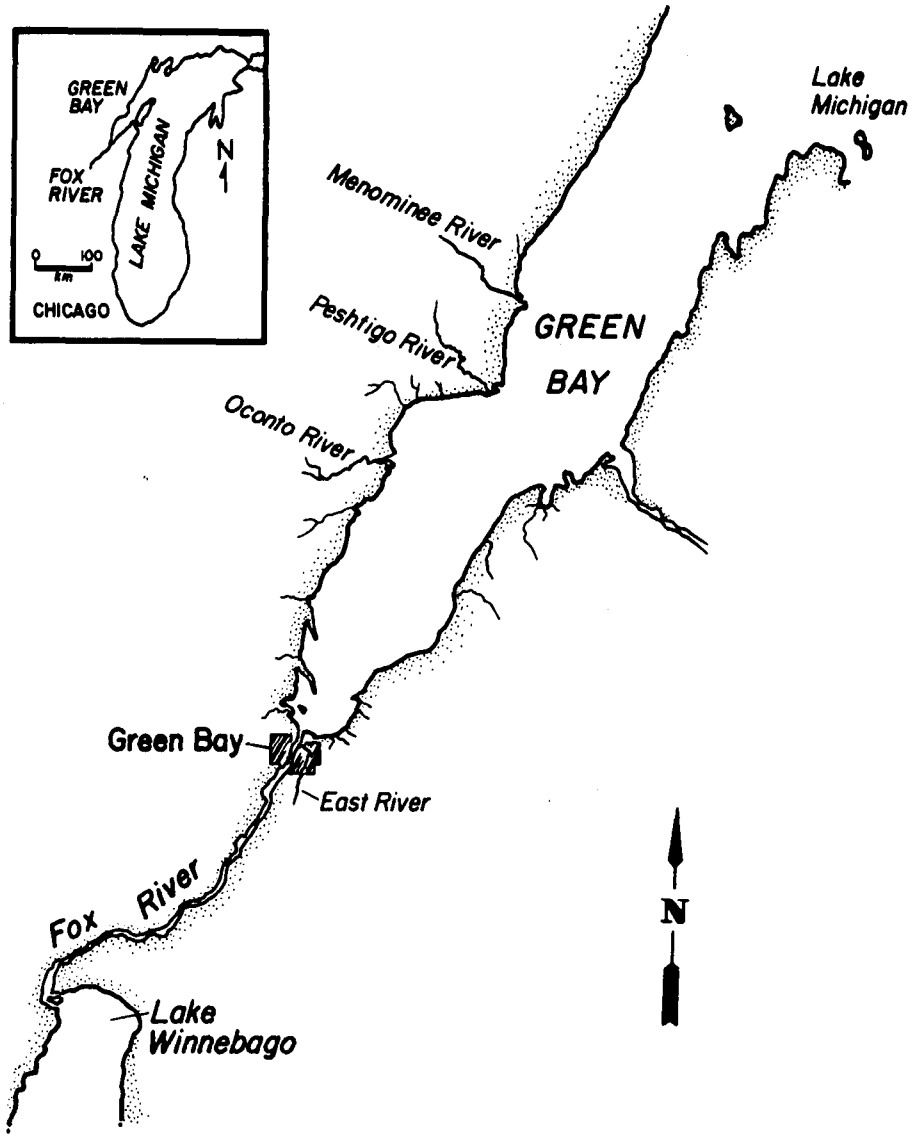

FIGURE 1. Location of Green Bay and major morphological features.

To evaluate the annual nonpoint source total phosphorus loading, the area under study is divided into a set of sub-basins or sub-watersheds representing the major hydrologic units within the system. Each sub-basin is then further divided into a set of cells which represent the area comprised of a specific land use/soil texture combination. The area of each cell is multiplied by its appropriate UAL and the products summed to arrive at the total annual nonpoint source load for any sub-basin. The cumulative load for the entire drainage system is simply obtained as the sum of the contributions for each of the major sub-basins as indicated by Equation 1 below:

$$TP = \sum_{i=1}^{n} TP_i = \sum_{i=1}^{n} \left[\sum_{j=1}^{m} \sum_{k=1}^{l} UAL_{j,k} \cdot A_{i,j,k} \right] \qquad (1)$$

where

TP: annual nonpoint source load for the entire drainage system

TP_i: annual nonpoint source load from sub-basin i ($i = 1, 2, \ldots, n$)

$UAL_{j,k}$: annual nonpoint source total phosphorus load per unit area of land use j and soil texture k ($j = 1, 2, \ldots, m$; $k = 1, 2, \ldots, l$)

$A_{i,j,k}$: area of land use j and soil texture k within sub-basin i

The microcomputer model allows a user to quickly conduct a preliminary macroscopic nonpoint pollution loading analysis on any drainage area of interest. The accuracy or goodness of the analysis is a direct function of the user's ability to accurately represent the land use, soil texture and hydrologic characteristics of the system under study.

The model is comprised of two major sub-programs: (1) SETUP, which prompts the user to define the land use and soil texture matrix for each sub-basin in the drainage system and to provide the UAL matrix for the pollutant of interest, and (2) LOAD, which integrates the data provided to SETUP with new information on land use changes or pollutant controls. Output from LOAD includes tabular and graphical displays of nonpoint source pollutant loadings for each major sub-basin and for any set of land use management/pollutant control scenarios under consideration.

Application to Fox River/Green Bay

Land Use/Soil Texture Description

The nonpoint source pollution model is applied in a preliminary assessment of annual total phosphorus loads to Green Bay from the Fox River drainage basin. Information on land use and soil classifications within the area were obtained from the Fox Valley Water Quality Planning Agency (FVWQPA) and Soil Conservation Service offices for the five counties comprising the Fox River watershed. For the purposes of this demonstration, soil classifications were grouped into three major soil texture categories -- coarse, medium, and fine; eight land use categories were considered in the analysis:

1] Residential
2] Commercial
3] Industrial/Transportation
4] Institutional/Government
5] High Tillage Cropland
6] Low Tillage Cropland
7] Pasture
8] Woodland/Residual

The Fox River watershed was divided into three sub-basins (Lower Fox River, Lake Winnebago, and River-Lakes) as shown in Figure 2. Acreages for each soil association in a given sub-basin were calculated from county soil survey maps. Soils in the River-Lakes (97% fine) and Lake Winnebago (99% fine) sub-basins are assumed to be comprised of 100% fine textured soils. The Lower Fox River sub-basin contains 35% coarse soils, 20% medium soils, and 45% fine soils.

Land use areas within the three sub-basins are dominated by cropland (64-69%), residual (17-25%), and residential (4-8%). The residual category includes woodlands, wetlands, and grasslands. Because UALs for any areal cell are dependent on the predominant land use and soil texture, it is necessary to apportion land use areas to the three soil texture classifications noted above. For this demonstration, the acreage of each land use corresponding to a given soil texture category is approximated using a simple weighting ratio of soil texture area to total sub-basin area. The acreage within each sub-basin corresponding to a specific soil texture and land use is defined by the resulting land use/soil texture matrix.

High Tillage vs. Low Tillage Cropland

The land use analysis conducted for this demonstration did not distinguish between low tillage and high tillage cropland areas. High tillage crop management practices are defined here as conventional or deep tillage methods consisting of complete soil inversion following crop harvest. Farming areas utilizing conservative tillage management systems, including minimum tillage (e.g., noninversion soil conditioning with chisel plows) and no till methods, have been grouped together and defined here as low tillage cropland. For the purposes of baseline calculations, it is assumed that the high tillage and low tillage cropland areas are equally represented in the three sub-basins.

Unit Area Load Matrix

Unit area loads used to estimate nonpoint source total phosphorus contributions within the Fox River watershed were derived from storm event monitoring data and literature values reflecting similar climatological, land use and soil texture conditions. Among the major information sources utilized in developing UALs were: a] surface washoff measurements from 21 test watersheds in northern Virginia (NVPDC/VPI, 1978), b] monitoring of 12 test watersheds in metropolitan Washington, DC (NVPDC, 1983), c] data from 11 test watersheds in the Chesapeake Bay drainage basin (Hartigan et al., 1983), d] pilot watershed measurements in the Great Lakes basin (PLUARG, 1978), e] methodologies of the water quality assessment procedure for toxic and conventional pollutants (Tetra Tech, 1982), and f] simulations of water quality and hydrology for the Occoquan Reservoir, Virginia (Office of Comprehensive Planning, Fairfax County, Virginia, 1982).

The UALs applied in this demonstration for the Fox River and Green Bay

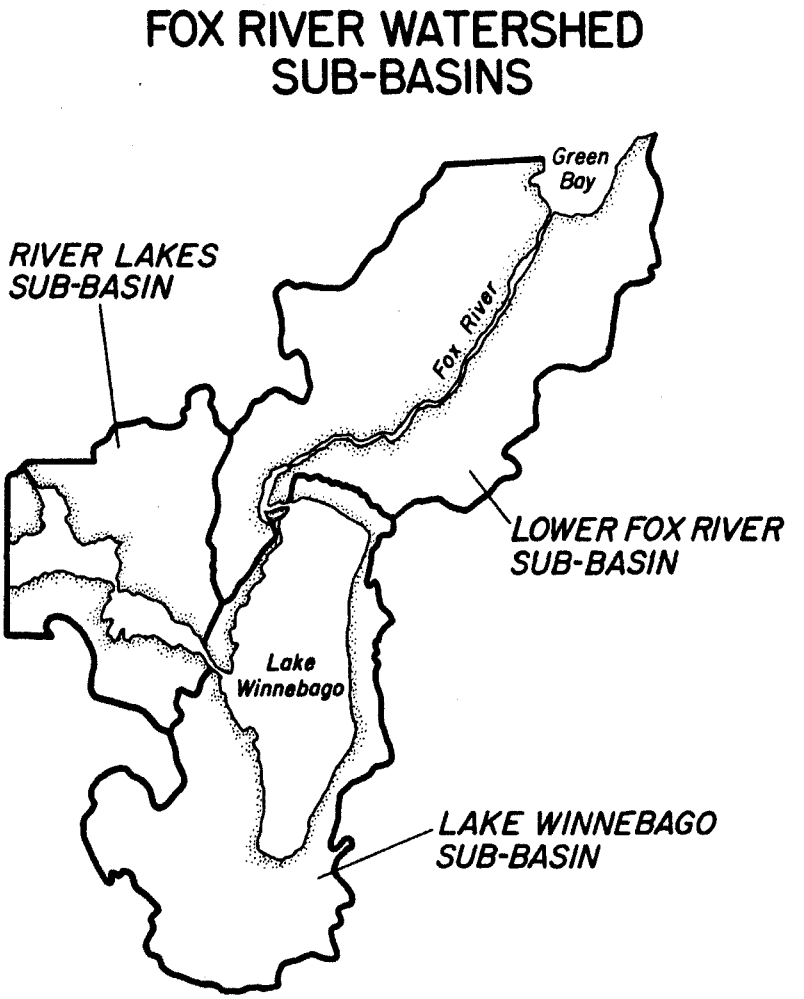

FIGURE 2. Component sub-basins of the Fox River watershed.

are presented in Table 1. Total phosphorus contributions to the Fox River and Green Bay from interflow/baseflow water are not included in the UAL values presented in Table 1; these loads are treated as constant at a rate of 0.018 kg/ha*year over the entire watershed. The UALs applied in this demonstration are intended to represent annual total phosphorus contributions from surface runoff events within the Fox River watershed during an "average" year of wetness or total precipitation. The current analysis has not attempted to quantify and compare climatological and physiographic factors among the areas from which UAL information was compiled.

Table 1. Total Phosphorus UAL Matrix (kg/ha*year)

Land Use	Soil Texture		
	Coarse	Medium	Fine
Residential	0.06	0.13	0.18
Commerical	0.28	0.28	0.28
Industrial/Transportation	0.18	0.37	0.55
Institutional/Government	0.09	0.18	0.28
High Tillage Cropland	0.55	0.74	0.92
Low Tillage Cropland	0.22	0.29	0.37
Pasture	0.06	0.07	0.09
Woodland/Residual	--	0.02	0.04

Average annual total phosphorus loads are calculated for each sub-basin as the product of the land use/soil texture matrix and the unit area load matrix.

Land Use Scenarios and Loading Estimates

The nonpoint source microcomputer model is used to generate estimates of average annual total phosphorus loads to the Fox River for seven hypothetical land use scenarios. These test cases are not intended to represent expected land use trends within the drainage area; rather, they demonstrate the value of the model in obtaining a rapid, macroscopic estimate of nonpoint loadings under a diversity of land use conditions. In addition, the test cases show how the model may be used to examine potential load reductions and concommitant water quality improvements resulting from implementation of remedial action within given portions of the watershed.

The seven scenarios include a baseline (=existing) condition, a "best-case" condition (all basin acreage in woodland), a "worst-case" condition (all basin acreage in high tillage), and three intermediate scenarios (baseline conditions plus combinations of high and low tillage). The estimated nonpoint source total phosphorus loadings to Green Bay from the Fox River are presented in Table 2.

Model Output and Analysis

An examination of model-estimated loadings points to the importance of the cropland contribution to the overall load -- an expected result considering the high UAL assigned to cropland areas and the high acreage of cropland present within eachof the Fox River sub-basins. The average annual total phosphorus load for Scenario 1 -- baseline conditions -- is estimated at

Table 2. Land Use Scenarios and Annual Phosphorus Loads

Scenario	Total Phosphorus Load (kgP/year)*	(kgP/day)*
1. Baseline (includes 50% High Till)	861,725	2361
2. Woodland	91,993	252
3. High Till Cropland	1,675,647	4591
4. Baseline/100% High Till	1,186,948	3252
5. Baseline/75% High Till	1,024,338	2806
6. Baseline/25% High Till	699,114	1915
7. Baseline/100% Lo Till	536,499	1470

* Watershed total; includes baseflow

861,725 kgP (2361 kgP/day). Almost 90% of that total is attributable to runoff contributions fom cropland. The influence of cropland on nonpoint loadings is further evidenced by the significant increases and decreases in the expected loads which accompany changes in the percent of cropland distributed between the high tillage and low tillage categories. The water quality implications of these hypothetical land use scenarios are now examined by coupling output from the nonpoint source model to a microcomputer-adapted trophic response model.

WATER QUALITY MICROCOMPUTER MODEL

Model Structure and Baseline Conditions

The water quality microcomputer model is a steady-state, 12-cell mass balance model for phosphorus; details of model development, calibration, and verification are provided elsewhere (Auer and Canale, 1985). The model considers sources (tributary loads) and sinks (net settling, mass transport) for total phosphorus in Green Bay and calculates the steady-state phosphorus profile for each of 12 model cells oriented along the major (NE/SW) axis of Green Bay. A comparison of model output with summer average total phosphorus concentrations for 1982 is provided as Figure 3. Trophic conditions for each model cell are evaluated using the methodology of Chapra and Dobson (1981); a total phosphorus concentration <11.0 ugP/L indicates oligotrophy, 11.0 - 21.7 ugP/L -- mesotrophy, and >21.7 ugP/L -- eutrophy. The gradient in trophic state corresponding to the phosphorus levels in Figure 3 (Baseline + 50% High Till) is presented as Figure 4. Under baseline conditions, the four southernmost model cells (approximately 20 km) are eutrophic, the mid-bay region is mesotrophic and the outer reaches are oligotrophic. It should be noted that significant oxygen depletion occurs in the hypolimnetic waters of the mid-bay region -- an area currently classified as mesotrophic.

Response to Management Scenarios

The microcomputer nonpoint source loading model estimates a baseline annual total phosphorus loading of 2361 kgP/day; this value compares well with the five-year average (2174 kgP/day, range 1240-2716 kgP/day) calculated from monitoring data and discharge permit reports by Roznowski and Auer (1984).

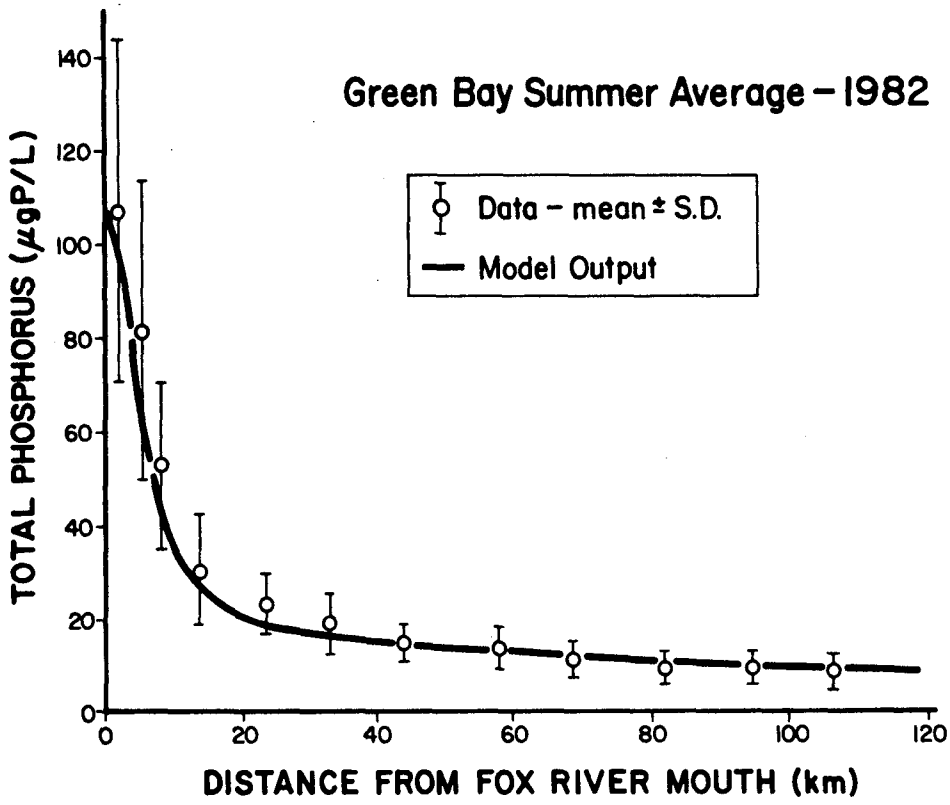

FIGURE 3. Steady-state model output compared with measured summer average total phosphorus concentration at 12 stations in Green Bay; bars (standard deviation, n = 18) indicate temporal variation at each station.

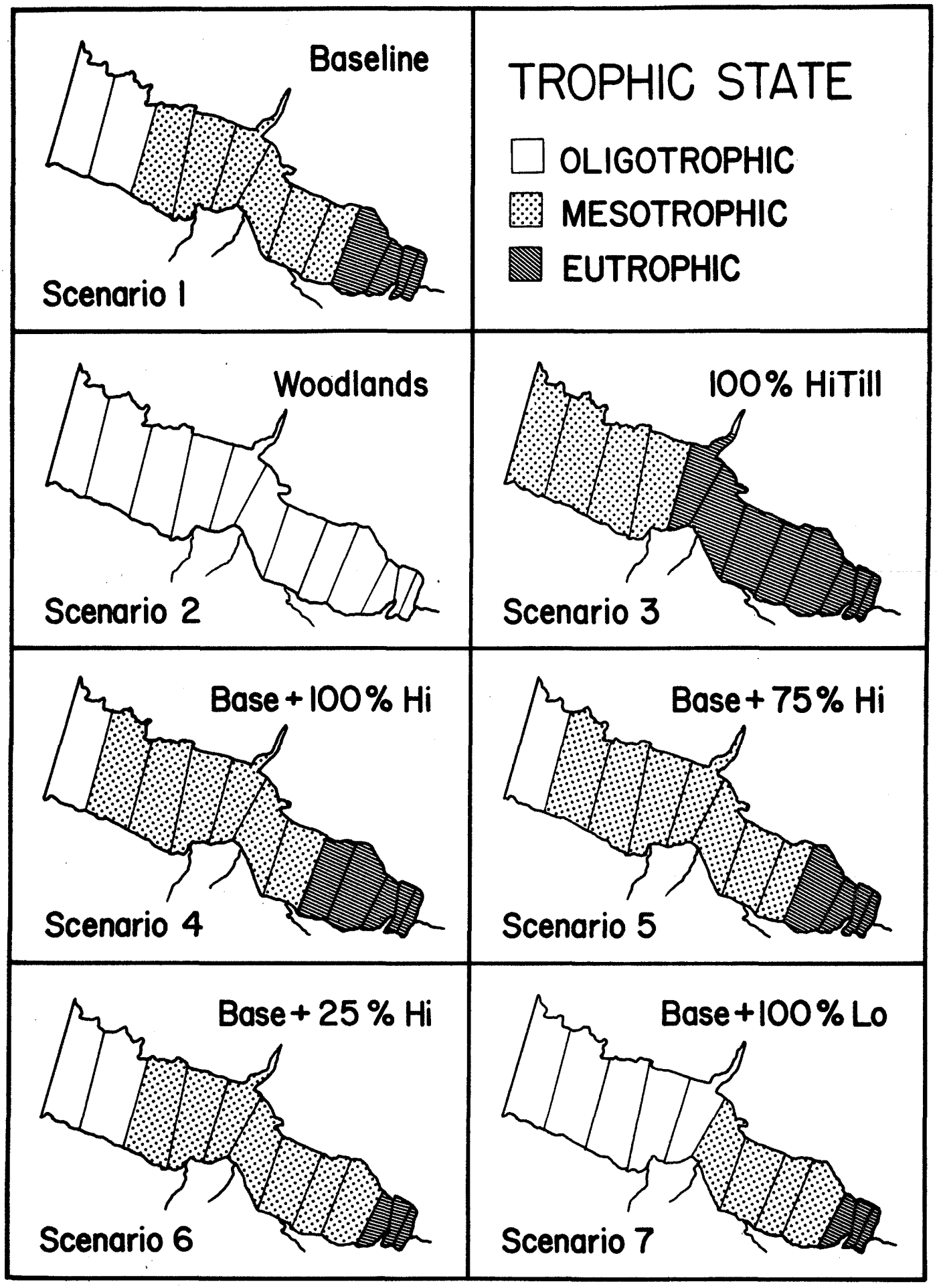

FIGURE 4. Trophic status for Green Bay under baseline conditions and 6 hypothetical land use scenarios.

The microcomputer water quality model calculates summer average total phosphorus concentrations based on the summer average tributary total phosphorus load. Scenario-specific summer average loads for the Fox River are calculated according to Equation 2:

$$W = (W_{scen} / W_{base}) * W_{sum} \quad (2)$$

where:

W = scenario-specific summer average total phosphorus load (kgP/day)

W_{scen} = scenario-specific annual total phosphorus load (kgP/day)

W_{base} = baseline annual total phosphorus load (kgP/day)

W_{sum} = baseline summer average total phosphorus load (kgP/day)

The baseline (1982) summer average total phosphorus loading to Green Bay from the Fox River is 959 kgP/day (Auer and Canale, 1985) and the baseline annual total phosphorus loading to Green Bay (Table 2, Scenario 1) is 2361 kgP/day -- thus the summer load accounts for approximately 41% of the annual load.

Illustrations of the gradient in trophic conditions for each land use scenario are presented in Figure 4. The most dramatic change in trophic status is that for the hypothetical "best case" condition (Scenario 2) corresponding to 100 percent woodland where oligotrophic conditions prevail over the entire length of the bay. The load yields a summer average Fox River total phosphorus concentration of 11.3 ugP/L, only slightly above the limit for oligotrophy, and approaching the Lake Michigan boundary condition of 8 ugP/L. Output from this scenario may reflect conditions prior to cultural development in the basin. Only slightly less dramatic is the hypothetical "worst case" condition (Scenario 3), where the entire basin is characterized by high tillage cropland. In this case, eutrophic conditions persist for approximately 65 km out the bay and overlay the regions most susceptible to hypolimnetic dissolved oxygen depletion. These results indicate that a wide range in water quality conditions is possible for Green Bay -- and that the current gradient in trophic state is strongly related to land use practices.

The "worst-" and "best case" scenarios demonstrate the sensitivity of the system to land use practice, but are not realistic or viable management options. Scenarios 4 - 7 examine the impact of variation in high tillage/low tillage ratios under the baseline scheme for land use. The output for Scenario 5 (Baseline + 75% High Till) is not appreciably different from that of the baseline condition. Expansion of high tillage practices to baseline plus 100% high till (Scenario 4) extends the region of eutrophy by approximately 10 km. Similarly, reduction of high tillage practices to baseline plus 25% high till (Scenario 6) or baseline plus 100% low till (Scenario 7) reduces the region of eutrophy to the southernmost 10-15 km of the bay. Implementation of 100% low tillage practices offers the additional advantage of extending the oligotrophic zone into the mid-bay region where oxygen depletion has been reported under baseline conditions. Specific water quality impacts may be evaluated for each scenario through the application of indices relating phosphorus and transparency or chlorophyll.

CONCLUSIONS

Microcomputer models for nonpoint source phosphorus loads and water quality are coupled to examine the impact of variation in land use practices on trophic state in Green Bay. Trophic response to various land use management scenarios is dramatic because of the large contribution (>85%) by cropland to the annual Fox River nonpoint total phosphorus load. Model output indicates that land use management, especially low tillage practices, offer an opportunity for improved water quality in the bay. The utility of this interactive planning and management tool has been demonstrated for several hypothetical land use strategies in the Fox River/Green Bay system.

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MODELING VERTICAL FLUX OF PESTICIDES WITH CREAMS

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ABSTRACT

Mathematical models to assess nonpoint source pollution and evaluate the effects of management practices are needed to respond to water quality legislation. The USDA model, Chemicals, Runoff, and Erosion from Agricultural Management Systems (CREAMS), which has been used successfully in modeling pesticides in surface runoff from field-sized areas was modified to simulate pesticide flux and persistence in the soil plant root zone. The modified version of CREAMS uses the daily rainfall option, and the soil erosion submodels in the original CREAMS. The modified version considers pesticide adsorption characteristics, degradation rates, and soil characteristics by horizon. Pesticide distribution in the root zone is computed with time and depth, as well as the quantities leached below the root zone. Pesticide concentrations in water and sediment fractions of surface runoff are also computed. Up to 10 different pesticides and multiple applications of each can be considered simultaneously. Preliminary tests comparing measured persistence and movement of the nematicide fenamiphos (Ethyl 3-methyl-4-(methylthio)phenyl(1-methylethyl)phosphoramidate) with simulation results appears promising.

Utility of the model in long-term simulation was demonstrated. Application of the model in comparing relative effects of management, climate, pesticide properties, and soil characteristics on potential movement of pesticides to groundwater is envisioned.

Keywords: Groundwater pollution, water quality, pesticide leaching, runoff, erosion

INTRODUCTION

Mathematical models to assess nonpoint source pollution and evaluate the effect of management practices are needed to adequately respond to the water quality legislation implemented since about 1970. Nonpoint source pollution from agricultural areas must be assessed, specific problems identified, and alternative practices developed to reduce or minimize problems. Sediment and chemical losses from surface runoff and chemical losses from the root zone by percolation must be addressed. Monitoring every field or farm to measure pollutant movement is impossible, and landowners and farm operators need to know benefits and costs before they implement alternative practices.

Although models are available for surface runoff and chemical transport, the burden is upon the model user to select from several potential models the one that will best represent the conditions, practices, and desired results for specific problems. The U. S. Environmental Protection Agency has sponsored development of several models; e.g. PRT (Crawford and Donigian, 1973), ARM (Donigian and Crawford, 1976a), NPS (Donigian and Crawford, 1976b), culminating in HSPF (Johanson et al. 1980) which is a comprehensive model with the capability of continuously simulating the dynamics of river basins and system response to implementing basin-scale water quality management decisions. Other field-scale pesticide models were described by Haith and Tubbs (1981) and Steenhuis and Walter (1980). In 1978, the U. S. Department of Agriculture, Agricultural Research Service, began a national project to develop mathematical models for evaluating nonpoint source pollution. The first development was the CREAMS model, published in 1980 (Knisel, 1980).

With reports of widespread contamination of groundwater (Pye and Patrick, 1983), concerns have mounted for the protection of this valuable resource. Agriculture may be an important contributor of organic contaminants such as pesticides (Cohen et al. 1984; Marti et al. 1985). In most of the reported incidences of groundwater contamination by pesticides, contamination has occurred because of combinations of several conditions and prerequisites conducive to pesticide transport. These may have been heavy applications of persistent and mobile compounds applied to soils with little adsorptive capacity, spills, shallow water tables, and high rainfall with annual aquifer recharge. As with surface water problems, models are needed to assess the potential for transport to groundwater and evaluate corrective measures. The PESTANS model (Enfield et al. 1982) was developed to provide a one-dimensional projection of vertical chemical movement through the unsaturated zone. The Pesticide Root Zone Model (PRZM) by Carsel et al. (1984) was developed for evaluating the movement of pesticides within the root zone and the lower unsaturated zones. This model incorporates a daily rainfall/runoff method similiar to that in CREAMS and a numerical solution for the chemical transport equation. The PRZM model is currently undergoing extensive verification and testing.

The USDA CREAMS model is widely used (Knisel and Svetlosanov, 1982; U. S. Soil Conservation Service, 1985) and has received rigorous testing and evaluation by the developers and other scientists (Leonard and Wauchope, 1980; Lorber and Mulkey, 1982; Nutter et al. 1984). Although CREAMS routed plant nutrients through the root zone, pesticides were not; leaching only below the surface 10 mm zone was considered.

Users of CREAMS have expressed considerable interest in an addition to CREAMS for pesticide leaching in the root zone. This paper describes such an addition presently called CREAMS 1.7R. This addition is in keeping with the original CREAMS structure and simplicity. A more comprehensive model, CREAMS2, is currently being tested and will also consider pesticide leaching through the solum (Leonard and Ferreira, 1985).

THE MODIFIED CREAMS MODEL

The original CREAMS model consists of three major components: hydrology, erosion/sedimentation, and chemistry. The hydrology component estimates runoff volume and peak rates, infiltration, evapotranspiration, soil water content, and percolation on a daily basis. If detailed precipitation data are available, infiltration may be estimated at histogram breakpoints. The erosion component estimates erosion and sediment yield, including particle-size distribution at the edge of the field. Storm loads and average concentrations of sediment-associated and dissolved chemicals in the runoff are also estimated as well as nitrate-nitrogen leached from the root zone.

The daily hydrology model of CREAMS accounts for soil water by dividing the root zone into seven soil layers and estimating flow through each layer using a storage-routing technique. CREAMS 1.7R uses this daily hydrology option exclusively and routes pesticides through the same seven layers using pass-files of water content and flux generated from the hydrology model. Plant nutrients are not simulated in CREAMS 1.7R to simplify input requirements. Also, the streamlined Soil Conservation Service version of pesticide input files and their crop rotation aspect (U. S. Soil Conservation Service, 1985) was incorporated into CREAMS 1.7R for user convenience. For long-term simulation runs, the irrigation option of DelVecchio et al. (1983) was added so that adequate soil moisture was maintained for crop growth without having to manually specify irrigation days and amounts.

A schematic of the pesticide submodel in CREAMS with the CREAMS 1.7R addition is given in Fig. 1. Pseudo first-order functions are assumed for pesticide dissipation from foliage and soil after pesticide application. Pesticide in the runoff active soil layer is partitioned between the solution phase and the soil phase by the following relationship:

$$(C_w Q) + (C_s M) = A\ C_p$$

and

$$C_s = K_d C_w$$

where C_w is pesticide concentration in runoff water; Q is the volume of water per unit volume of surface active layer; C_s is pesticide concentration in soil equilibrating with runoff; M is mass of soil per unit volume of active surface layer; A is an extraction ratio specifying the soil to water ratio in the extraction zone; C_p is the gravimetric concentration of pesticide residue in the soil; and K_d is the coefficient for partitioning the pesticide between sediment and water phases. Before runoff computation, a portion of the pesticide mass is moved below the surface 10 mm by the function:

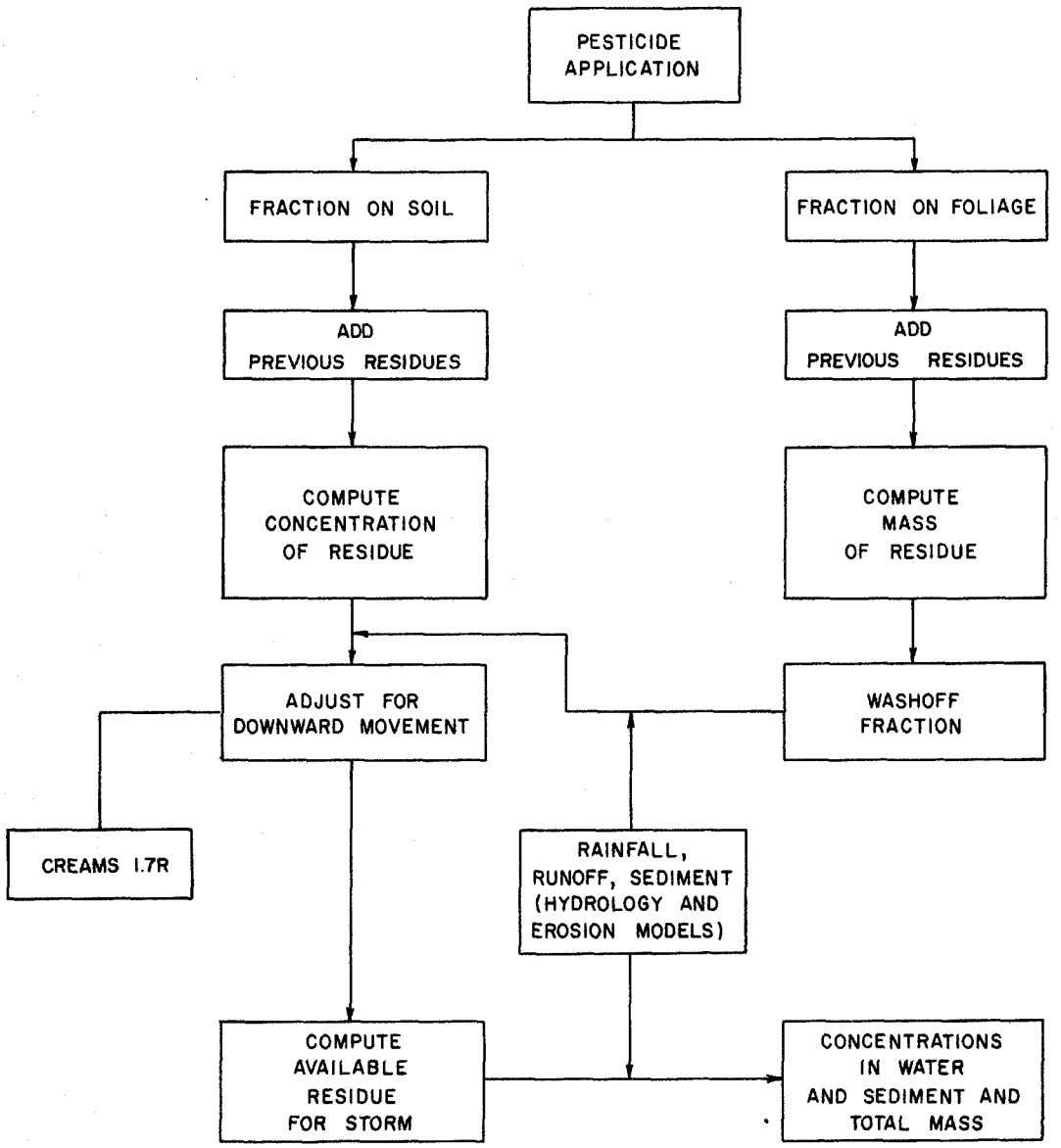

Fig. 1. Schematic of CREAMS 1.7R model

$$Z = Z_0 e^{-\left[\frac{RF-RO-S}{DK_d\,(1-p)+p}\right]}$$

where Z is the pesticide mass remaining in the surface layer after infiltration; Z_0, the initial pesticide mass; RF, rainfall; RO, runoff; S, initial abstraction; D, soil particle density; and p, soil porosity. In routing pesticide through the root zone, the mass (Z_0-Z) is added to the adjacent soil layer below, along with the volume of water percolate. Pesticide concentrations in the water and soil phases of this hypothetical reservoir are then computed using the relationship $C_s = K_d C_w$. In the model, K_d is calculated from $K_d = K_{OC} \cdot OC$, where OC is the fraction of organic carbon in each layer, and K_{OC} is the adsorption constant based on the organic carbon present. This layer is then allowed to drain to field capacity, the drainage volume with its pesticide mass being added to the next layer. This equilibration-storage routing procedure is repeated for each layer of the root zone or until the infiltration pulse is dissipated.

Model output for each rainfall period is pesticide content of the root zone by layer before and after rainfall and the mass leached below the bottommost layer. Soil-layer thickness, as well as total root zone thickness is specified by the user in input files. As in CREAMS, CREAMS 1.7R simulates up to 10 different pesticides simultaneously and multiple applications of each as needed.

Annual summaries are generated of the pesticide applications lost in runoff by water and sediment and the percent leached below the root zone.

MODEL EVALUATION

CREAMS 1.7R has received preliminary testing using data from an experiment we are conducting near Tifton, GA to investigate persistence and movement of several pesticides in sandy Coastal Plain soils. The experimental area is located on Bonifay sand (93.5% sand, 2.9% silt, 3.6% clay, <1.0% organic carbon). This soil has low fertility, low water holding capacity, and high leaching potential, but can be made productive using an intensive chemical/water management program. Several cm of supplemental irrigation are required for crop production, particularly in spring and summer months. The sand soil profile is underlain at 2-3 m by relatively impermeable sediments and a shallow water table is often perched to within 60-80 cm of the soil surface, particularly in winter and early spring.

Four main plots, each 12x12 m in size, were established, planted in sweet corn (Zea mays cv. Merit) and treated with selected pesticides. The overall design was such that paired main plots could be irrigated at different rates if desired. Wells for monitoring water-table levels and sampling for pesticide residues in groundwater were located within each plot at nominal depths of 0.9, 1.8, and 2.7 m. Well casings were 5-cm aluminum alloy tubes perforated for a distance of 30 cm from the lower end and covered with aluminum screen. Casings were sealed above the perforations by a mixture of clay and fine sand. Excess soil water was removed from the field by a perforated drain tile installed on the downslope side of the study area at a depth of about 1 m. Tile outflow was periodically sampled for pesticide residues. Each of the four main plots were subdivided into four replicate subplots for soil sampling.

Four replicate soil cores were removed from each plot to depths of 1 m. Each core was subdivided into 0-10, 10-20, 20-30, 30-40, 40-50, 50-75, and 75-100 cm depth intervals for pesticide analysis. Soil samples were removed at 5 time intervals, spanning a period of 1-40 days after pesticide application. Groundwater wells were sampled weekly after first determining water table depth and evacuating to remove stagnant water in the well casing. Rainfall and other meteorlogical data were available from a station located adjacent to the study area.

Several pesticides were included in this study, however, only data on fenamiphos in 1983 will be presented here. Fenamiphos (Ethyl 3-methyl-4-(methylthio)phenyl(1-methylethyl)phosphoramidate) is a systemic nematicide giving broad spectrum nematode control for a variety of crops. A granular formulation of fenamiphos was applied in 1983 at a rate of 6 kg/ha active ingredient in 30-cm bands in each row and incorporated 7.5 cm deep. The 6 kg/ha is broadcast equivalent, i.e. 6 kg/ha if broadcast uniformly, however, since only one third of the soil was treated (90-cm row spacing), actual mass applied was 2 kg/ha on an areal basis.

Soil samples (Ca. 300 g) were air dried, thoroughly mixed and stored under refrigeration pending analysis. Samples were extracted by shaking for two hours with 50% chloroethanol. After extraction, fenamiphos was converted to fenamiphos sulfone by oxidation with magnesium sulfate and potassium permanganate, partitioned into methylene chloride, and determined by gas chromatography using a thermionic detector operated in the nitrogen mode. Recoveries by this method were >95% with detection limits of 0.005 ug/g soil.

Fenamiphos concentrations found in the root zone are plotted in Fig. 2(a) with respect to time and depth. Points plotted are at midpoints of each sampling depth interval and are averages of four replicates. Movement of fenamiphos below the incorporation zone with time after application is indicated; however, concentrations rarely exceeded 0.005 ug/g below 50 cm. Apparently, most of the fenamiphos had dissipated by degradation in 34 days.

Using appropriate soils data and measured rainfall/irrigation inputs, fenamiphos degradation and movement was simulated as shown in Fig. 2(b). However, no model calibration to the specific site was done. A degradation constant based on a 14-day half-life was estimated from previous work of Johnson et al. (1982). A K_{OC} value of 160 was assigned by averaging measurements made in our laboratory on 20 Coastal Plain soils. Incorporation depth was set at 7.5 cm and all other parameter values estimated as recommended (Knisel, 1980). In the model runs, a root zone of 46 cm was assumed and divided into approximate increments of 0-1, 1-7, 7-15, 15-22, 22-30, 30-38, and 38-45 cm. Computed concentrations are plotted in Fig. 2(b) and are at the midpoint of each depth increment with respect to time. Comparing Fig. 2(a) and Fig. 2(b), a reasonable correspondence between measured and simulated results is measured considering that depth increments and times were not identical. Model output dates were chosen to correspond to rainfall events, whereas actual sampling dates were somewhat arbitrary. The estimated 14-day half-life provided good representation of observed fenamiphos persistence. Based on field-collected data, fenamiphos appeared to be more mobile than simulated by the model. Note concentration maxima at lower depths and broadening of the concentration profiles in the observed vs simulated. However, several variance factors should be

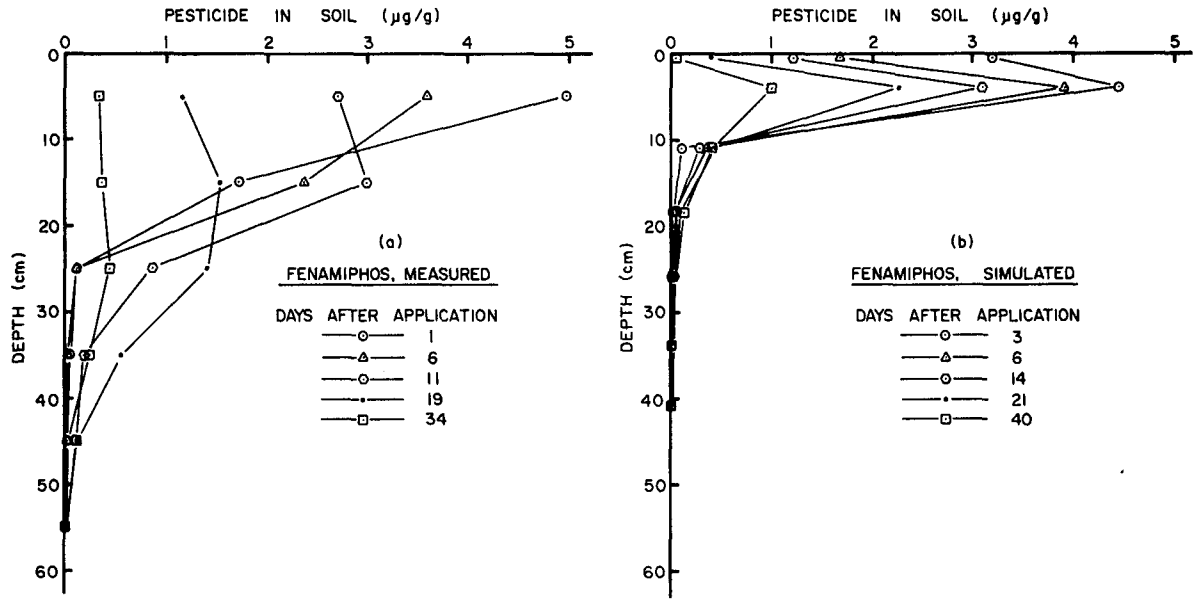

Fig. 2. Comparison of measured and simulated fenamiphos concentration in Bonifay sand

considered. Numerical dispersion may be created by the sampling and simulation depth increments chosen. The model assumes a constant 7.5 cm incorporation depth; however, in the field, the incorporation implement cannot be regulated exactly and the depth of incorporation may have varied considerably. Establishment of row beds at planting and soil subsidence with rainfall also creates surface roughness and distorts the surface reference.

In model application, a soil organic matter profile of 1.0, 1.0, 0.7, 0.5, 0.3, 0.2, and 0.1% was assumed for the seven depth increments. Organic matter contents were not available on the specific site, only an approximate value of $\leq$ 1% was available for the surface horizon. Lower organic matter content than assumed would have contributed to an underprediction of pesticide movement.

During the model simulation period depicted in Fig. 2(b), simulated loss below the 46 cm root zone was 10.2 g/ha. Computed percolation below the root zone during this period was 9.6 cm giving an average fenamiphos concentration of 0.011 mg/l in the percolate. At saturation, this solution concentration plus that adsorbed at equilibrium by the soil organic matter would give a concentration in soil of 0.007 ug/g on a dry weight basis. The practical limit of detection for fenamiphos by actual analysis was 0.005 ug/g, so it is obvious that comparisons are made in a range of uncertainty in the actual data. Using Fig. 2(a), an interpolated fenamiphos concentration at the 45-cm depth of 0.082 ug/g produces 0.2 mg/l in the soil solution at saturation. At this concentration, if 3.7 cm of water percolated through this plane (predicted by model simulation), 74.2 g/ha fenamiphos could have been transported. This is considerably greater than the 10.2 g/ha simulated. However, within the limits of detection no fenamiphos was transported below the 55 cm depth. Also, none was detected in sampling wells or tile outflow. For the entire simulation period, however, about 10 g/ha could have been leached and would not have been detected if concentrations remained below 0.005 ug/g soil. Clearly, we are in a concentration range that does not give a complete test of the model, and illustrates the difficulty in obtaining field test data for pesticides ranging from the nonadsorbed to moderately adsorbed (such as fenamiphos). For this and other reasons, models are best used to make relative comparisons and not absolute predictions.

An indication of model sensitivity to pesticide K_{OC} is provided for two hypothetical pesticides in Fig. 3. Model runs were performed using identical inputs as in simulations depicted in Fig. 2(b) except that in test A, K_{OC} = 500 and in test B, K_{OC} = 5 were assumed. A degradation half-life of 50 days for both and identical application rates of 1.7 kg/ha were also assumed. Comparing these simulations demonstrates that significant movement of very weakly adsorbed pesticides would have occurred below the root zone under rainfall patterns observed in 1983. Of the total amount applied, about 31% of pesticide B moved below the root zone compared to 0.01% of pesticide A.

In the fenamiphos simulation for 1983 (Fig. 2[b]), 0.74% of the application was estimated to move below the 46-cm root zone during the entire year. Annual losses will, however, be highly dependent on rainfall occurring within a few weeks after application. Field experiments are usually limited to 1 to 3 years and cannot assess the year-to-year variation expected in leaching losses. Using daily rainfall records for the period 1958-77 at Tifton, GA, a 20-year simulation of fenamiphos leaching at the experimental site was per-

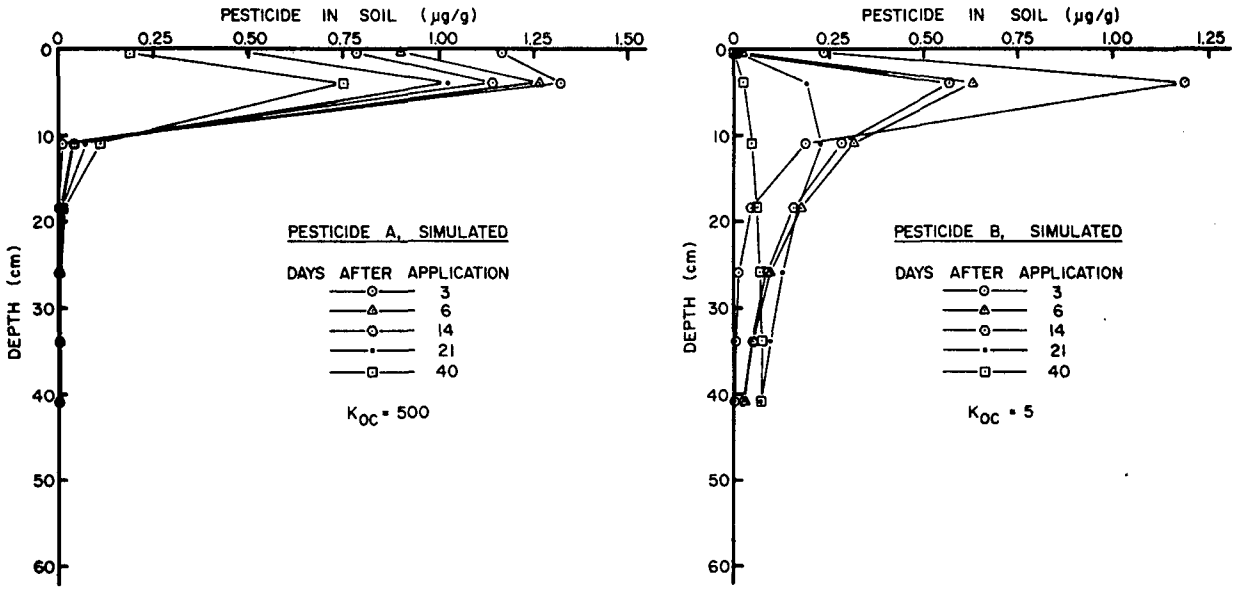

Fig. 3. Comparison of mobility in Bonifay sand of two hypothetical pesticides differing only in K_{OC}. Rainfall and other inputs same as used in simulation shown if Fig. 2.

formed, and results are presented in Fig. 4. Annual losses ranged from near zero to about 5%. The 5% loss occurred in 1973 when 20.3 cm of rainfall occurred within a 14-day period after application. These results demonstrate the value of long-term simulations in assessing expected leaching below the plant root zone and serves as a useful tool in extending field observations.

SUMMARY AND CONCLUSIONS

Based on preliminary evaluations, a modification of CREAMS which provides simulation of pesticide concentrations in the root zone, and masses leached below the root zone appears promising. Testing and verification efforts will continue. Model utility in long-term simulations was demonstrated. Model applications in evaluating relative effects of management decisions, differing soil and pesticide properties, and weather patterns on potential pesticide loading to groundwater are envisioned.

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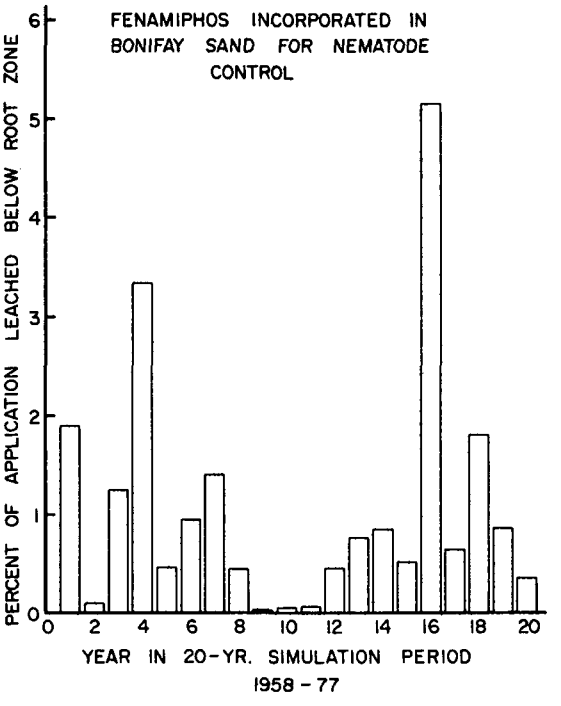

Fig. 4. Results of 20-year simulation of fenamiphos leaching below 46 cm root zone in Bonifay sand using rainfall record. 1958-77, at Tifton, GA

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NON-POINT POLLUTION ABATEMENT IN TAMPA BAY

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ABSTRACT

This paper presents the most recent evaluation of the effect of pollutional loadings upon Tampa Bay.

In the years between 1970 and 1984, the Tampa Bay Region has been the subject of 4 major pollution abatement programs. In each case urban runoff has been evaluated . In three of the programs all point and non-point sources were evaluated.. Loadings of pollution parameters as determined for the year 2000 in all of the programs displayed a remarkable similarity . The projections were prepared by divers groups of professionals over a long time span using totally different techniques. Thus loadings determined for the year 2000 for the wet and the dry seasons can be accepted as good estimates.

This report presents the methods used to evaluate the effect of the pollutants upon the water quality of Tampa Bay.

A data file of water quality parameters dating from 1969 through 1983 were subjected to multiple regression analysis.

An improved water quality model was developed calibrated and verified to more closely represent the nutrient storages and flows in Tampa Bay. In particular the effect of phytoplankton growth on light distributions in the bay were addressed. Field measurements were conducted to determine pertinent rate coefficients for the model. Benthic processes were found to be dominate in Hillsborough Bay and some portions of Upper Old Tampa Bay.

Keywords: Tampa Bay, Nonpoint pollution,point sources pollution, Statistical study, Ecological models,water quality, hydraulic models, benthic processes , eutrophcation

INTRODUCTION:

Tampa Bay is a shallow well-mixed bifurcated estuary on the west central coast of Florida-see Figure 1. It is beset by all of the problems attending the rapid development of the area. The University of South Florida (USF), Center for Mathematical Models has been involved in numerous studies relating to the water quality of Tampa Bay, dating from 1971 through the present. The 1984 Wasteload Allocation Program (WLA) sponsored by the Florida Department of Environmental Regulation (FDER) is the occasion of this paper.

From the first major study of a portion of Tampa Bay conducted by the Federal Water Pollution Control Administration [FWPCA 1969], to the present studies, the fundamental problems of the bay remain. The consistent problems are

A. Excessive nutrient levels which encourage algae growth,
B. Excessive benthic oxygen demand which produces oxygen depletion in the benthos and oxygen stratification in the water column.
C. The gradual disappearance of sea grasses.

The WLA program was designed to address each of these problems in attempting to allocate whatever resources the bay had to assimilate non-point and point source loadings. A program was devised to review all existing data, photographic and archival. Field tests were conducted to provide rates for the mathematical model. calibration and verification data were obtained. The existing data and the new data were subjected to statistical analysis.

This paper concentrates on the tasks performed by USF.

PERTINENT FACTS REGARDING TAMPA BAY

Tampa Bay is approximately 35 miles long (67.3 km), and 12 miles (15.5 km) wide at the widest point. The bottom area is 8.6 billion square feet (1.14 $x10^7$ km^2). The volume at an average high tide is116 billion cubic feet (5.6 $x10^9$ m^3). The normal tidal prism is 13.5 billion cubic feet (6.5 x 10^8 m^3). Tides are semi diurnal. Point source input to the bay is approximately 26.6 million cubic feet per day ($1.27 x10^6$ m^3). Wet weather nonpoint source input is approximately 260 million cubic feet per day ($1.27 x 10^9$ m^3). Pollutants enter Tampa Bay at the 39 locations shown in Figure 2. Pollutant loads for this study were arrived at by a statistical separation of low and

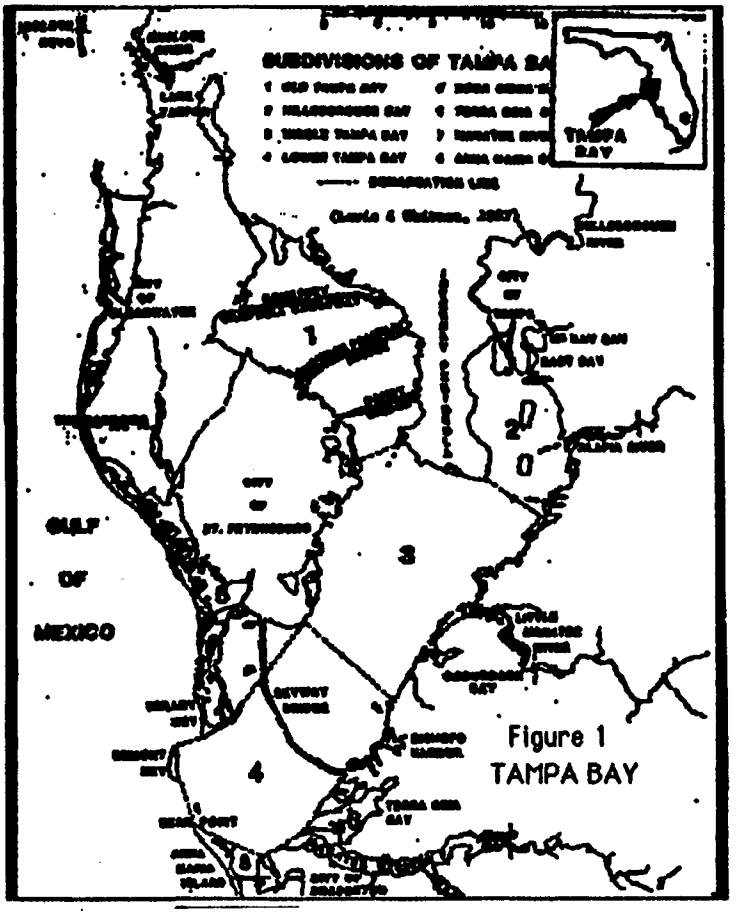


Figure 1
TAMPA BAY

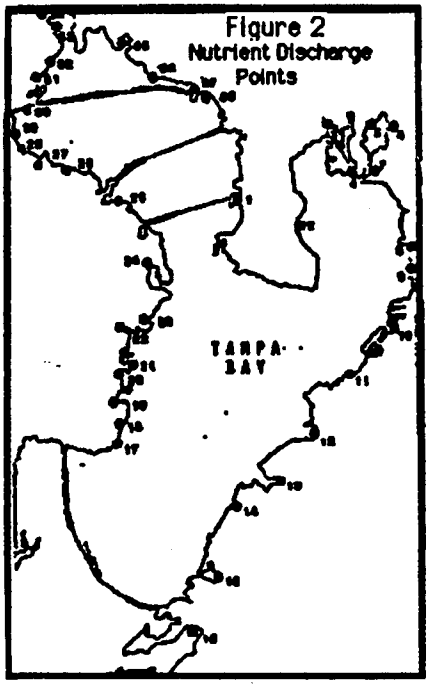


Figure 2
Nutrient Discharge Points

high flows by durations, CDM (1984). Runoff from ungaged areas were determined by use of U. S. Geological Survey empirical relations developed in the region ,USGS (1983). Parameter concentrations were determined by use of empirical relations from the same source. Yearly total loads determined in this study were within a few pounds of those determined in 1971 - for the year 2000, TBRPC (1973). Agreement of the year 2000 estimates for the 1976 determination and the present prediction was also within a total of 2200 pounds,TBRPC (1976). Natural phosphorus loadings are higher for the Tampa Bay region since this is an area of natural deposits and mining.

STATISTICAL ANALYSIS RESULTS

Samples of waters in Hillsborough Bay subjected to bioassay always indicate a nitrogen limitation to growth ,City of Tampa (1983). Statistical analysis of the 1980-1981 data shows no correlation of chlorophyl a and nitrogen in Hillsborough Bay but a positive correlation in other parts of the bay. Strong correlation existed between BODu and light extinction everwhere. A correlation between chlorophyl a and ortho phosphorus existed everywhere. The latter finding was surprising inasmuch as phosphorus concentrations are high in Tampa Bay .

In brief, the statistics showed that during the wet season planktonic growth for the most part is limited by some factor other than nitrogen or phosphorus, possibly light. The relation between chlorophyl a and phosphorus is the result of the spatial proximity of the two since concentrations of phosphorus were hundreds of times the half saturation concentrations required for chlorophyl a.

THE HYDRAULIC MODEL USED IN THE STUDY

The hydraulic model used in the study is the one developed at USF in 1970. The model comprises the vertically integrated equations of motion and continuity in two dimensions, Ross(1984). The basic equations used are given:

$$\frac{\partial U}{\partial t} = -\frac{1}{D}\frac{\partial U^2}{\partial x} - \frac{1}{D}\frac{\partial UV}{\partial y} + \Omega V - \frac{1}{\rho D}\frac{\partial P}{\partial x} - gD\frac{\partial H}{\partial x} + X - \frac{f|Q|U}{D2}$$

$$\frac{\partial V}{\partial t} = -\frac{1}{D}\frac{\partial V^2}{\partial y} - \frac{1}{D}\frac{\partial VU}{\partial x} - \Omega U - \frac{1}{\rho D}\frac{\partial P}{\partial y} - gD\frac{\partial H}{\partial y} + Y - \frac{f|Q|V}{D2}$$

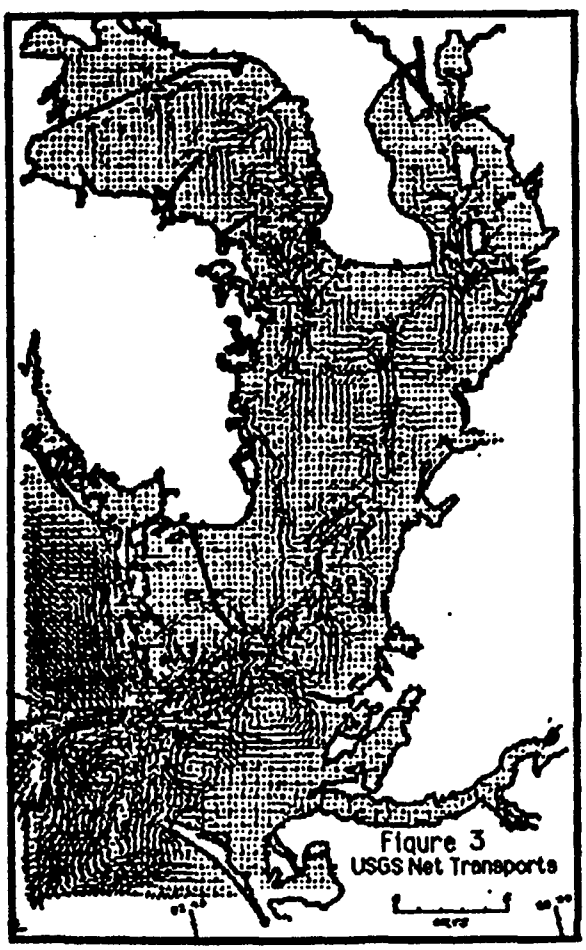


Figure 3
USGS Net Transports

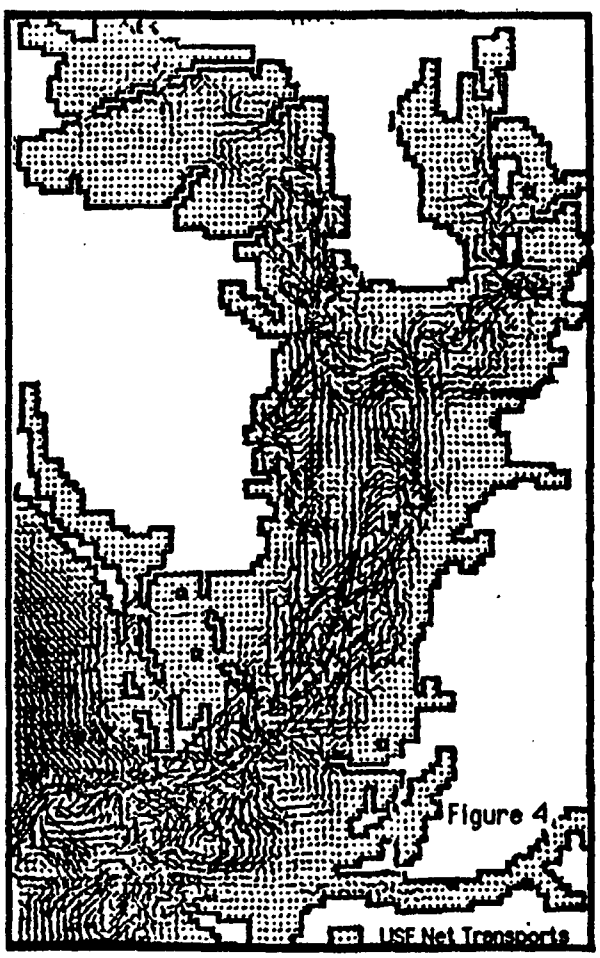
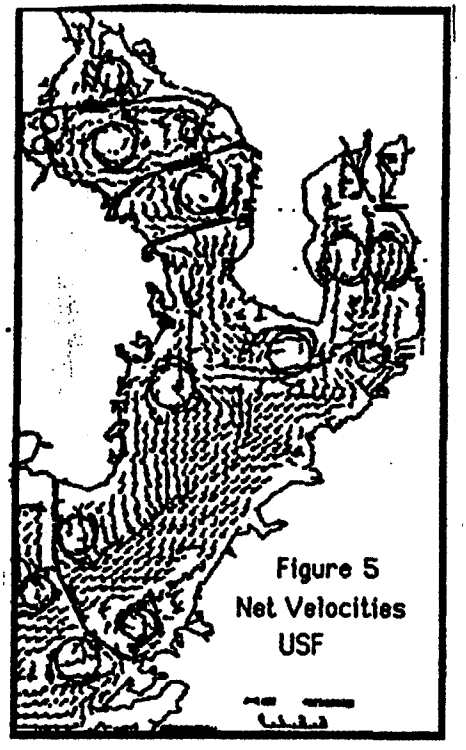
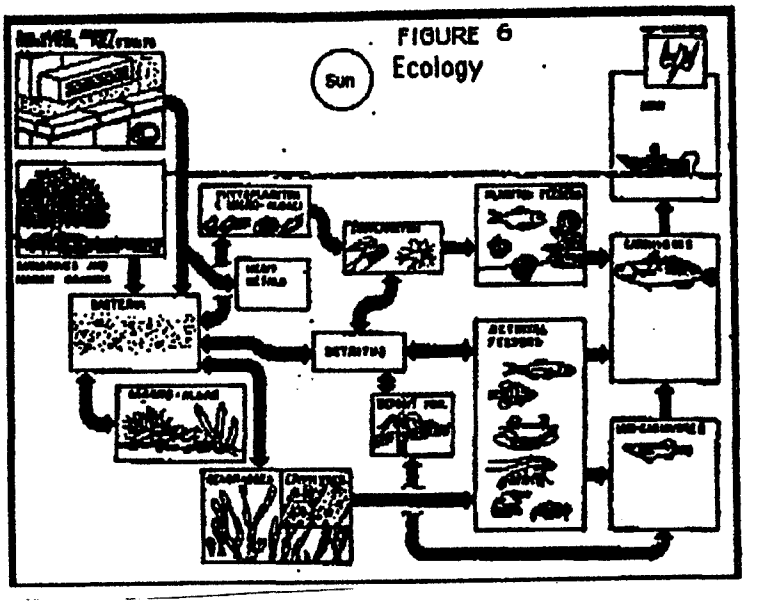
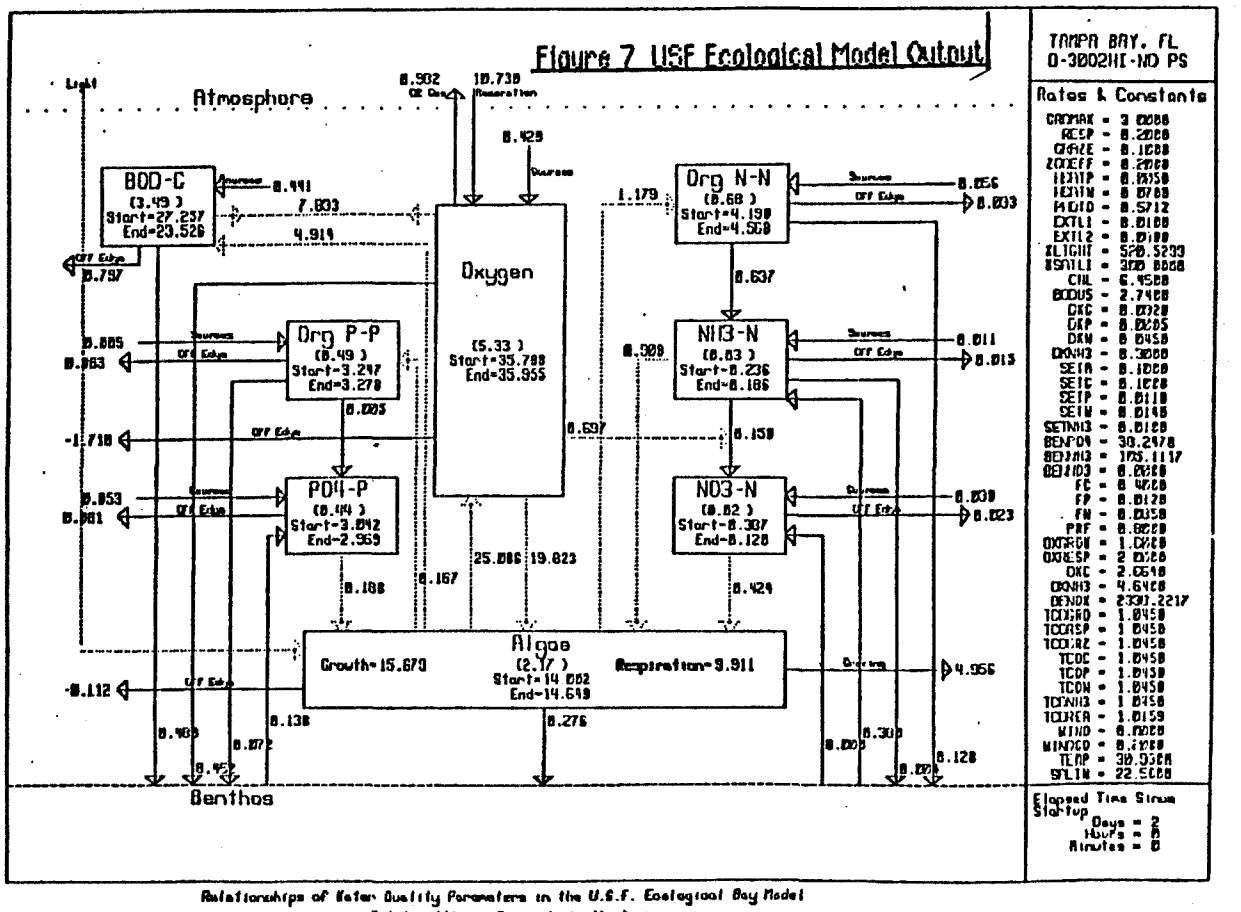


Figure 4
USF Net Transports







Relationships of Water Quality Parameters in the U.S.F. Ecological Bay Model
Total millions of pounds in the bay
(Average concentration in the bay (mg/l))

Table 1 Ecological Model Equations

Parameter	Decay Rate(K)	Source/Sink(R)
ALGAE(A)	$\rho_G + \rho + \frac{\sigma_A}{D} - \mu$	$\emptyset$
CBOD(C)	$f_1 K_C + \frac{\sigma_C}{D}$	R_C
ORGP(P_1)	$K_{P_1} + \frac{\sigma_P}{D}$	$R_{P_1} + \alpha_p \rho A$
ORTHO-P(P_2)	$\emptyset$	$R_{P_2} + K_{P_1} P_1 - \alpha_p \mu A + \frac{S_{P2}}{D}$
ORGN (N_1)	$K_{N_1} + \frac{\sigma_{N1}}{D}$	$R_{N_1} + \alpha_n \rho A$
AMMONIA-N (N_2)	$f_2 K_{N_2} + \frac{\sigma_{N2}}{D}$	$R_{N_2} + K_{N_1} N_1 - \Delta \alpha_N \mu A + \frac{S_{N2}}{D}$
NITRATE-N (N_3)	$\emptyset$	$R_{N_3} + f_2 K_{N_2} N_2 - (1-\Delta)\alpha_N \mu A + \frac{S_{N3}}{D}$
DISS. O_2 (O_x)	K_O	$K_O O_x + (\alpha_{OG}\mu - \alpha_{OR}\rho)A - f_1 K_C C - f_2 \alpha_{ON} K_{N_2} N_2 - \frac{S_B}{D}$

Basic Equation:

$$\frac{\partial CD}{\partial t} + \frac{\partial CU}{\partial x} + \frac{\partial CV}{\partial y} = \frac{\partial}{\partial x}\left(E_x \frac{\partial CD}{\partial x}\right) + \frac{\partial}{\partial y}\left(E_y \frac{\partial CD}{\partial y}\right) + RD - KCD$$

Terms:

ρ_G = grazing rate
ρ = respiration rate = $\emptyset.1\ \mu'$
σ_A = algae settling rate
μ = local specific growth rate (see below)
K_D = deoxygenation rate
f_1 = $O_x/(O_x+\emptyset.\emptyset1)$
σ_C = settling rate of CBOD
K_{P_1} = reaction rate of Organic P
σ_{P_1} = settling rate of Organic P
α_p = ratio of phosphorus to biomass
S_{P_2} = benthic source rate of Ortho-P
K_{N_1} = hydrolysis rate of Organic N
σ_{N_1} = settling rate of Organic N
α_N = ratio of nitrogen to biomass
K_{N_2} = nitrification rate
f_2 = $O_x/(O_x+\emptyset.5)$
σ_{N_2} = settling rate of ammonia
Δ = $\emptyset.9N_2/(\emptyset.9N_2+\emptyset.1N_3)$
S_{N_2} = benthic source rate of ammonia
S_{N_3} = benthic source rate of nitrate
K_O = reaeration rate
O_S = DO saturation
α_{OG} = photosynthic oxygen production rate
α_{OR} = oxygen consumption rate by respiration
α_{ON} = oxygen uptake per unit nitrification = 4.57

Supplementary Equations:

$$\mu = \mu'\left(\frac{N_2+N_3}{N_2+N_3+K_N}\right)\left(\frac{P_2}{P_2+K_P}\right)L$$

where:

μ' = maximum specific growth rate
K_N = half saturation value for inorganic N
K_P = half-saturation value for inorganic P
L = $\frac{ef}{K_e D}\left(\exp(-ab) - \exp(-b)\right)$
e = 2.718...
f = photoperiod (fraction of day)
K_e = extinction coefficient = $\frac{1.9}{H_s'} + \emptyset.\emptyset52A$
H_s' = algae-free secchi depth
a = $\exp(-K_e D)$
b = I_A/I_S
I_A = avg light intensity over daylight hours
I_s = saturation light intensity
K_O = $11.61\emptyset1\ U^{0.969}D^{-1.673} + K_{ow}\frac{W}{D}$
K_{ow} = wind coefficient for reaeration
W = wind speed
C_L = chlorophyll-a concentration = $\alpha_{C_L} A$
α_{C_L} = ratio of chlorophyll-a to biomass

Table 2 Calibration Constants

Photoperiod (f)	0.5714 fraction
Light Intensity - average (I_a/f)	320.5259 ly/day
Light Intensity - saturation (I_s)	300.0 ly/day
Chl. a to algal biomass (a_{CL})	66.45 µg chl/mg biom
CBOD to BOD_5 (r_L)	2.74
Deoxygenation rate (K_d)	0.097 day^{-1}
Organic Phosphorus decay (K_{op})	0.005 day^{-1}
Organic Nitrogen hydrolysis rate (K_n)	0.063 day^{-1}
Ammonia nitrification rate (K_{an})	0.3 day^{-1}
CBOD settling rate (σ_L)	0.07 ft/day
Org. Phosphorus settling rate ($\sigma_{p,1}$)	0.011 ft/day
Org. Nitrogen settling rate (σ_{n1})	0.014 ft/day
Algae settling rate (σ_{a2})	0.03 ft/day
Carbon to algal biomass (a_c)	0.4 mg c/mg biomass
Nitrogen to algal biomass (a_N)	0.083 mg N/mg biomass
Phosphorus to algal biomass (a_p)	0.012 mg P/mg biomass
O_2 production for algal growth	1.6 mg O_2/mg biomass
O_2 consumed for algal respiration	2.0 mg O_2/mg biomass

$$\frac{\partial H}{\partial t} = -\frac{\partial U}{\partial x} - \frac{\partial V}{\partial y} + R$$

These equations expressed in finite difference form are solved explicitly on a grid representing Tampa Bay. The resulting model was calibrated once in 1970 by the use of sodium fluoroscene dye which was released on target and photographed repeatedly from the air. The model has been verified in seven major studies including the study of this report, Ross (1984). Verifications in clude tide heights and velocities. A graphic comparison of net transports in Tampa Bay is given in Figures 3 and 4. Figure 3 is the results from the USGS model of Tampa Bay and Figure 4 is the results from the USF model with the same input data ,Goodwin and Ross(1984). Comparison of the results from these two very different models indicates that major circulations do exist in Tampa Bay. Net velocities in Tampa bay are shown in Figure 5 ,Ross(1973). It has been shown that suspended material tends to congregate in velocity gyres, Ross(1985).

For this study the driving functions for the model were obtained by fitting the observed tide data with an astronomical harmonic series, this allowed extensions beyond the observed period. Hydraulic calculations were performed for each one minute period for a real time of 14.4 days (a repeating tidal period). Transports, velocities, depths,and accumulated transports were recorded for each 10 minute interval. These data are the input for many other auxilliary models including a new Ecological model developed for this study.

THE ECOLOGICAL MODEL USED IN THIS STUDY

The ecology of Tampa Bay is shown in Figure 6. Instead of the open system suggested by Lewis (1980), a semi-closed system including the recycling from the carnivores is the basis for the new model.

The model is based upon the vertically integrated equations of conservation of mass including sources and rate processes. The basic equation and the applicable rates and parameters are shown in Tables 1 and 2.
An example of the output of the program is given in Figure 7. The form of the output shown is of invaluable aid in the calibration and verification process. The entire cycle is visible on one page. The rates used in the model were set at default rates as given by Zison(1978). A box model created to summarize the ecologic system (part of the main program) is useful in the calibration process. Average values of the concentrations were used to adjust approximate rate coefficients. This can be done in minutes. A long run shows that other changes may be necessary.

During calibration and verification , time varying hydraulic conditions were utilized in the hydraulic model as discussed. The ecologic model was initialized close to fact then time varying loads are introduced and the model run for a 40 day period . For discharges from the 39 sources, independent tests discovered the dynamic equilibrium time of 40 days for Tampa Bay.

PARTIAL RESULTS OF THE STUDY

Some of the important results of study are summarized in six figures. Figure 8 and Figure 9 show year 2000 with nonpoint source pollution only - present bottom conditions. Figures 10 and 11 show nonpoint sources with best practical treatment- present bottom conditions. Figures 12 and 13 show the results for nonpoint sources with BMP -and benthic uptake reduced by 50%.

CONCLUSIONS :

The six figures from 8 through 13 show that reducing nonpoint sources by 30% by the use of the Best Management Practice only slightly reduces the spatial extent of poor water quality, however, slightly reducing the benthic uptake of nutrients greatly improves water quality.

The computer simulations also verified that the phytoplankton in Hillsborough Bay are nitrogen limited and that they grow to a density at which they become light limited . Self shading increases the oxygen demand and detritus settling. The uptake of nutrients from the bottom contributes to the algae growth tending to fix the location of the high density. The process spreads slowly with increased deposition. This is the classic pattern of eutrophication. While the model does not calculate the vertical distribution of dissolved oxygen , it is safe to assume that in areas of high benthic demand , anoxic conditions exist in the lower water column much of the time even though the average concentrations throughout the water column may meet minimum standards.

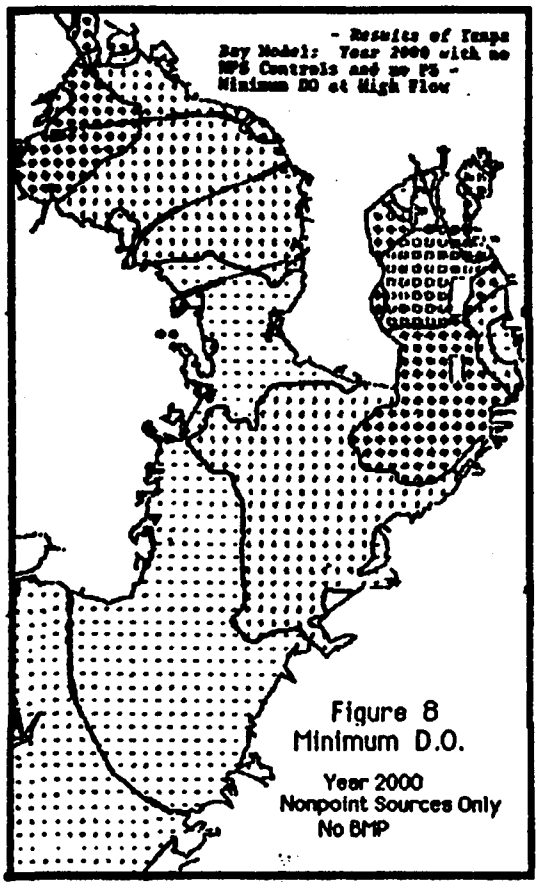


Figure 8
Minimum D.O.
Year 2000
Nonpoint Sources Only
No BMP

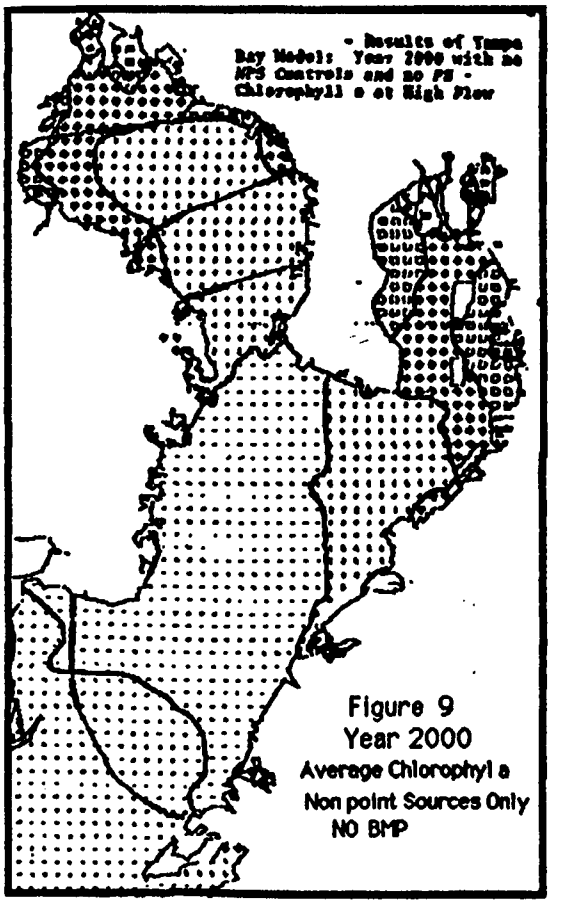


Figure 9
Year 2000
Average Chlorophyl a
Non point Sources Only
NO BMP

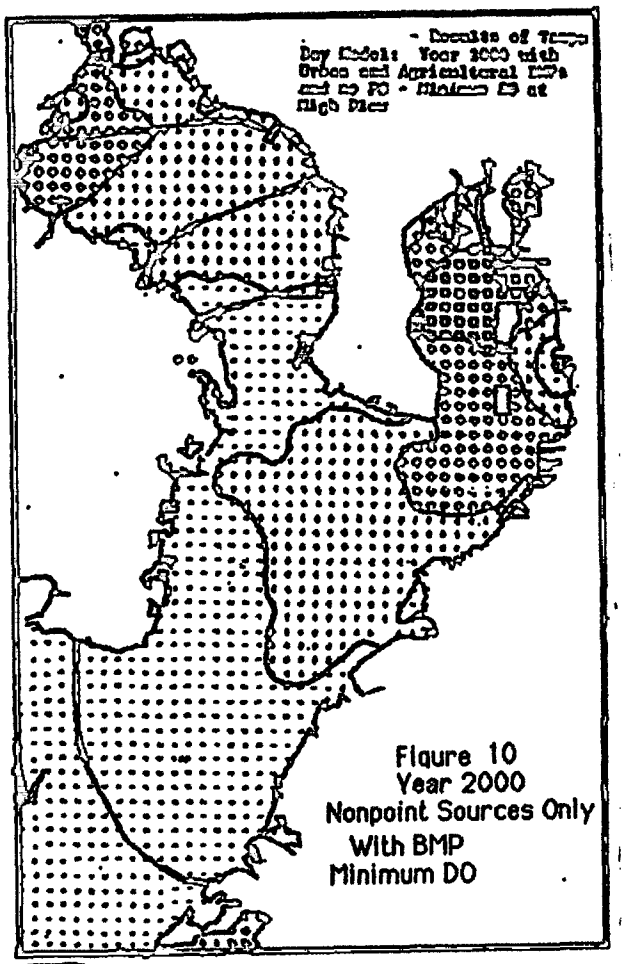


Figure 10
Year 2000
Nonpoint Sources Only
With BMP
Minimum DO

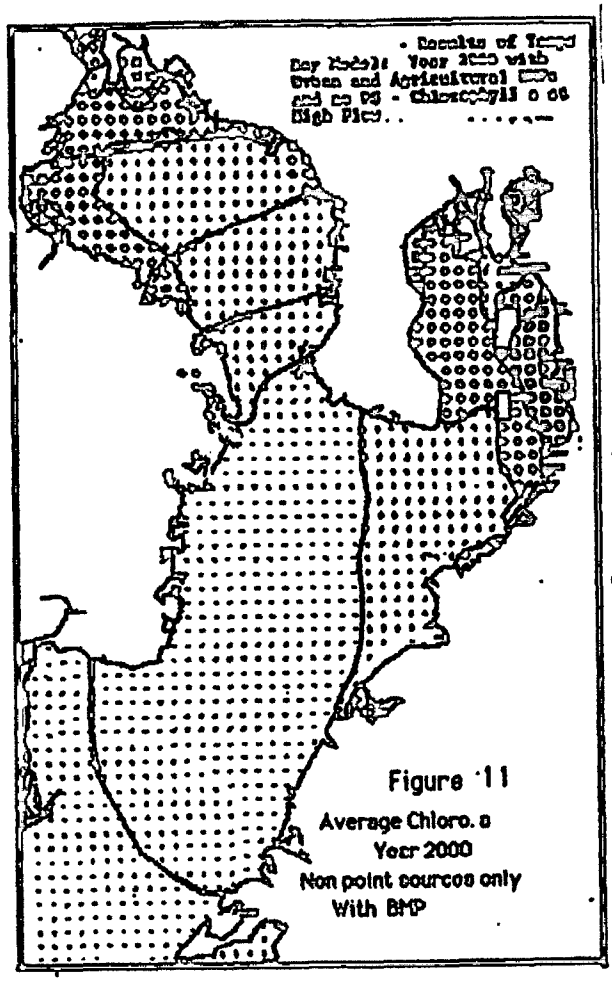


Figure 11
Average Chloro. a
Year 2000
Non point sources only
With BMP

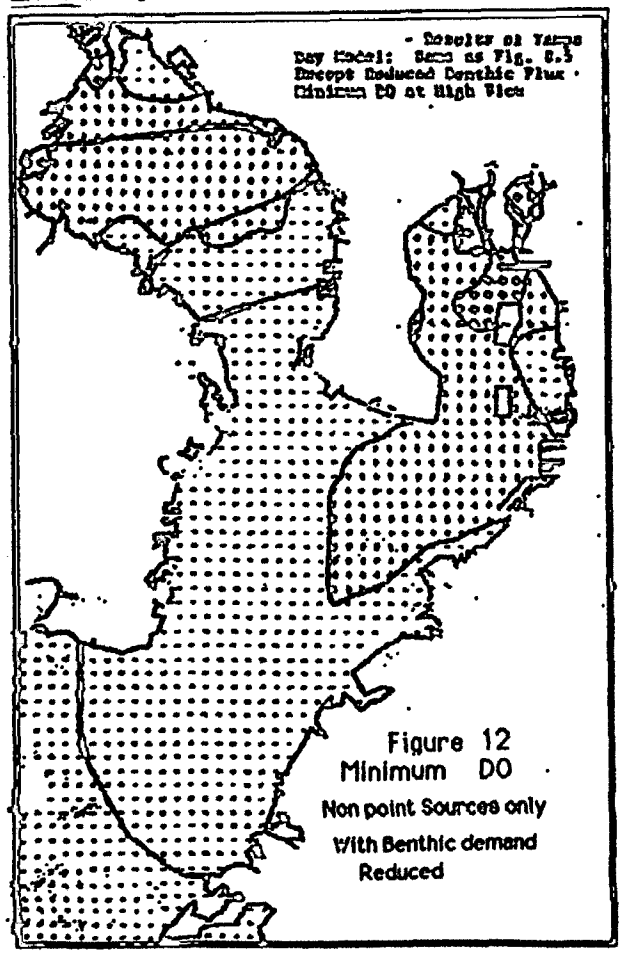


Figure 12
Minimum DO
Non point Sources only
With Benthic demand
Reduced

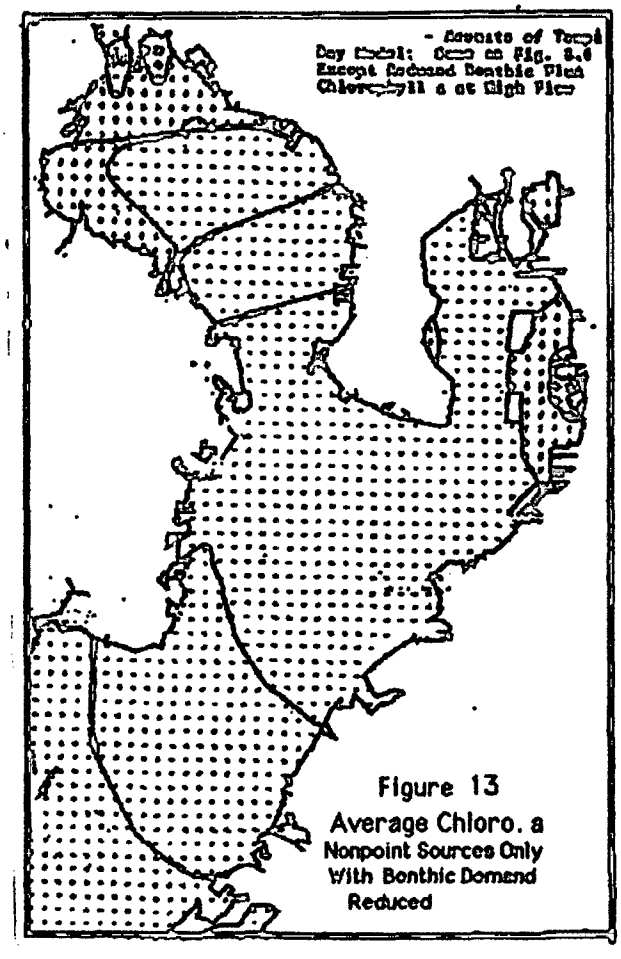


Figure 13
Average Chloro. a
Nonpoint Sources Only
With Benthic Demand
Reduced

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POLLUTION ABATEMENT IN THE RUHR RIVER BASIN, WEST GERMANY

D.R. Albrecht and K.R. Imhoff

1. The Ruhr System

Nonpoint pollution abatement has been an integrated part of the comprehensive water management in the Ruhr River basin since 72 years. Innovative approaches to water management in this area developed over seven decades ago as increasing pressure on a very limited supply of water by the massive urban-industrial complex in the Ruhr Industrial Area required decisive action to resolve severe water related problems. Special legislation created two organizations which were empowered, respectively, to care for water quantity and water quality. A joint directorship of the two was formed 47 years ago, thus institutionalizing the fact that management of water quantity and water quality are inextricably intertwined. The Ruhr River water management associations are characterized by compulsory membership for all major water users and suppliers, i.e. communities, industries and water works. A comprehensive water management in the Ruhr catchment is all the more necessary because approximately one-third of the average seasonal low flow is exported as drinking water to other river basins in the industrialized area (Fig. 1).

Northrhine-Westphalia is one of the most industrialized states in West Germany. In the industrial district - known under the name "Ruhr District" - water consumption and the volume of wastewater are seven times as high per km^2 as in the average of West Germany. The principal source of water supply in the district is the Ruhr River which at present has to meet 70 % of the whole demand. The Ruhr River has a drainage area of 4,488 km^2. It flows at the southern border in parallel to the large urban-industrialized district from east to west. The long term average precipitation varies within the river basin from 800 to 1,400 mm per year. Due to seasonal factors, the river flow varies between a minimum of 4 m^3/s and a maximum of 2,200 m^3/s. The present permanent export rate of drinking water abstracted from the Ruhr basin is 12 m^3/s. Thus, in dry periods a

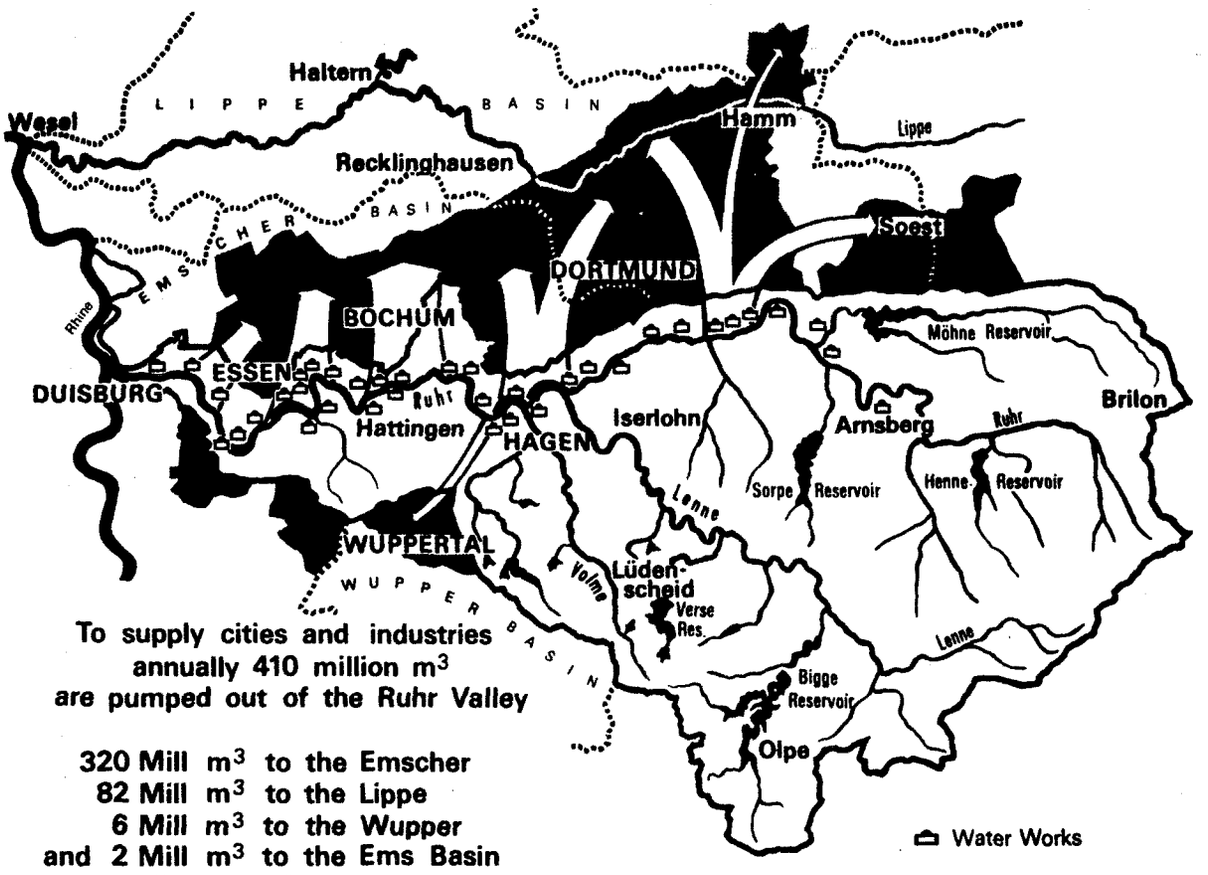


Fig. 1: Water export from the Ruhr to adjacent river basins

considerable water deficit must be compensated for by the operation of a system of reservoirs. In total a volume of 471 million m^3 of storage capacity exists to augment low flow.

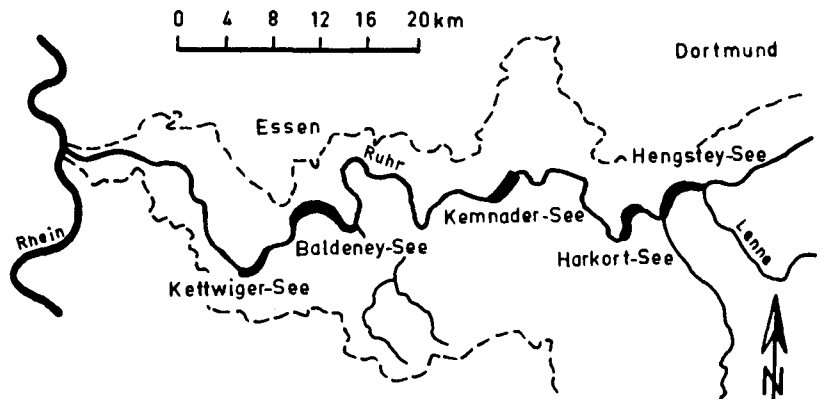
Besides providing sufficient water for water supply the Ruhr River also has the task of carrying the wastewater of about 2 million people living in its drainage area and of various industries located in the basin. Thus, effective wastewater treatment is necessary. The Ruhr River Association operates a system of 118 treatment works, 40 of which are vested with tertiary treatment by polishing lagoons.

One of the first tasks of the association was to assist the municipalities to design and construct sewer systems, in order to improve their hygenic conditions. At this stage the former nonpoint pollution of communities was changed into point pollution for the major rivers. In many cases smaller tributaries of the Ruhr were integrated into the sewerage system. The following period was characterized by the increasing control of point

pollution and increasing degrees of wastewater treatment. At present, the major load of point pollution is controlled by treatment efficiencies of more than 90 % in respect to BOD. In this situation the remaining nonpoint pollution has become the largest residual fraction of organic load to rivers. Sewage and unpolluted water of creeks are presently separated and many tributaries in the cities are now "renaturized".

2. Purification Lakes

According to the innovative design of the first director of the Ruhr River Association, Dr. Karl Imhoff, a system of five river impoundments has been constructed in addition to the system of treatment works to compensate for residual pollution from treatment plants and nonpoint pollution above major water works intakes [1], (Fig. 2). Both concepts, polishing lagoons and river impoundments provide facilities for sedimentation and extended biological activities. River water polishing by impounded lakes has proven to be a feasible measure to improve river water quality and to control nonpoint pollution. The always available large amount of fresh water has compensated for the considerable oxygen demand of storm water runoff at any time. In contrast to other German rivers, a fish kill was never observed in the impounded Ruhr after heavy rainfalls.



Impounded Lake	year of impounding	average depth (m)	surface (km^2)	capacity in the year of impounding (mio. m^3)
Hengstey	1929	1,90	1,70	3,3
Harkort	1930	2,30	1,38	3,2
Kemnade	1979	2,40	1,25	3,0
Baldeney	1932	3,25	2,60	8,7
Kettwig	1949	2,60	0,55	1,4

Fig. 2: River impoundments in the Ruhr valley

The selfpurification effect in river impoundments is mainly provided by sedimentation. In Lake Hengstey 400,000 m^3 of sediment have accumulated and in Lake Harkort some 350,000 m^3. During 50 years the volume of Lake Baldeney has diminished by 1.5×10^6 m^3. Subsidence caused by coal mining until 1970 is estimated to be of the same order. Consequently, some 3×10^6 m^3 have been accumulated, resulting in a sedimentation rate of 60,000 m^3 per year [2]. This rate depends on the flow characteristic of the river and on size and morphology of the impoundment. For example, in Lake Baldeney maximum sediment accumulation in terms of m^3 per day is observed at flows of the order of three times long term average flow, despite the then considerable turbulence. At lower flows circumstances for sedimentation are more favourable, but the input of settleable solids is then limited. At flows equal to ten times long term average, deposits start to be mobilized, and only flows exceeding 15 times average flow flush deposits partly into downstream reaches. Table 1 indicates analytical data of the sediments in Ruhr lakes.

Table 1: Average composition of sediments in the Ruhr lakes

Parameter	Dimension	Purification Lake		
		Hengstey	Harkort	Baldeney
Dry solids	%	39.4	45.8	45.6
volatile solids	% of dry sol.	12.3	11.0	14.0
Fe	mg/kg dry sol.	69,200	59,900	50,100
Cu	mg/kg dry sol.	1,240	830	730
Zn	mg/kg dry sol.	4,030	3,120	3,540
Ni	mg/kg dry sol.	261	250	312
Cr	mg/kg dry sol.	440	280	400
Pb	mg/kg dry sol.	450	480	525
Cd	mg/kg dry sol.	18	29	36
PO_4	mg/kg dry sol.	14,400	11,110	17,700
N total	mg/kg dry sol.	3,750	2,840	6,820

A detailed research program on the influence of the largest Ruhr River impoundment, Lake Baldeney, on the river water quality was performed from 1979 throughout 1982 [3]. Some results will be described below:

- On average, 30,000 tons of suspended solids are retained annually in Lake Baldeney, which is 20 % to 50 % of the input.

- The average difference between BOD_5-input and output is 2,200 tons per year, being equivalent to a purification capacity of 100,000 population equivalents. 20 % to 30 % of the BOD_5 input is retained. Fig. 3 represents the decrease of BOD_5-load in dependency of flow, distinguished for winter and summer half years.

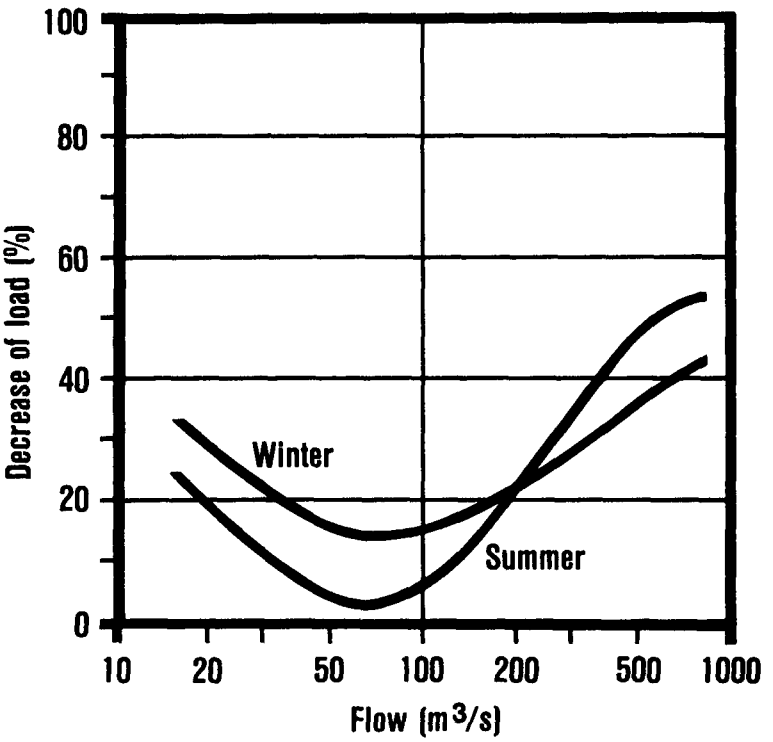


Fig. 3: Impounded Lake Baldeney: percentual decrease of BOD_5-load in dependency of flow

By evaluating 284 influent and effluent samples a detailed phosphorus balance was performed. Fig. 4 indicates the retention of the P-load in dependency of flow. At summer low flows 22 % or 70 tons of dissolved phosphorus are retained by algal development and subsequent sedimentation of phytoplankton or grazing zooplankton. During the winter half year 16 % or 40 tons of dissolved phosphorus are eliminated. In addition, 170 tons of phosphorus is settled per year with solid particles.

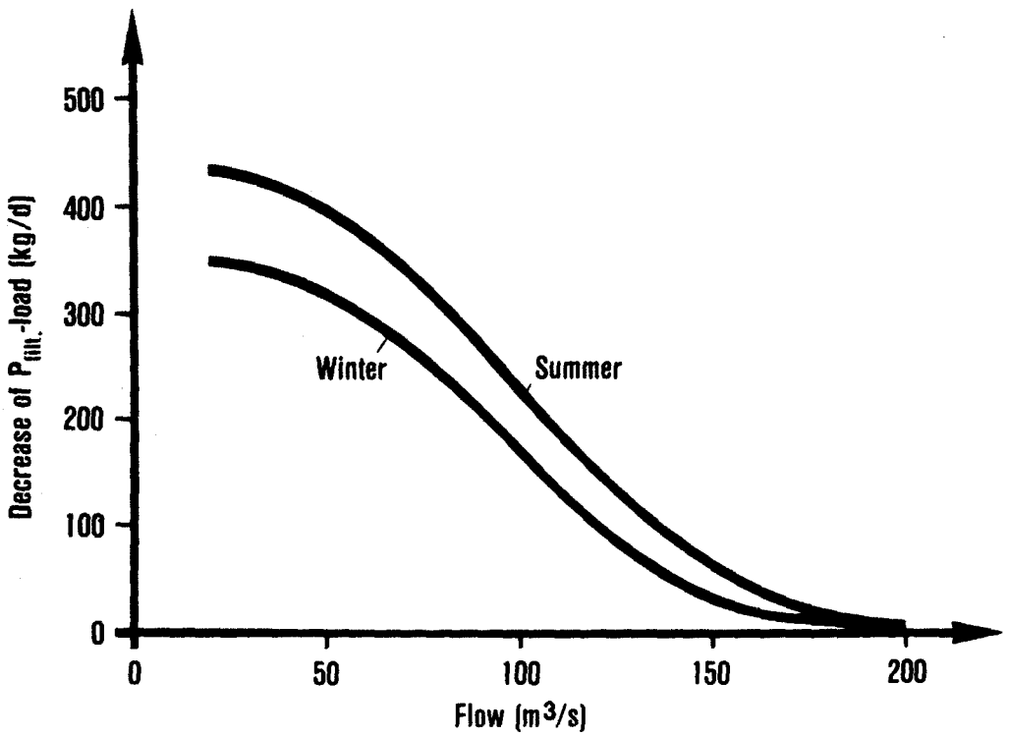


Fig. 4: Decrease of dissolved phosphorus in the impounded Lake Baldeney

- Due to geochemical reasons and because of a considerable number of metal finishing factories in the Ruhr basin, the heavy metal content of the sediments is ten times as high as usual. The average elimination rates for heavy metals in Lake Baldeney are as follows:

Element	Fe	Pb	Cr	Cu	Cd	Zn	Ni
Elimination rate %	50	46	38	33	33	31	10

A separate investigation on origin and fate of heavy metals in the Ruhr basin has indicated that a fraction of 41 % of the total heavy metals in Ruhr water is of geochemical origin, i.e. from nonpoint sources (Fig. 5) [4].

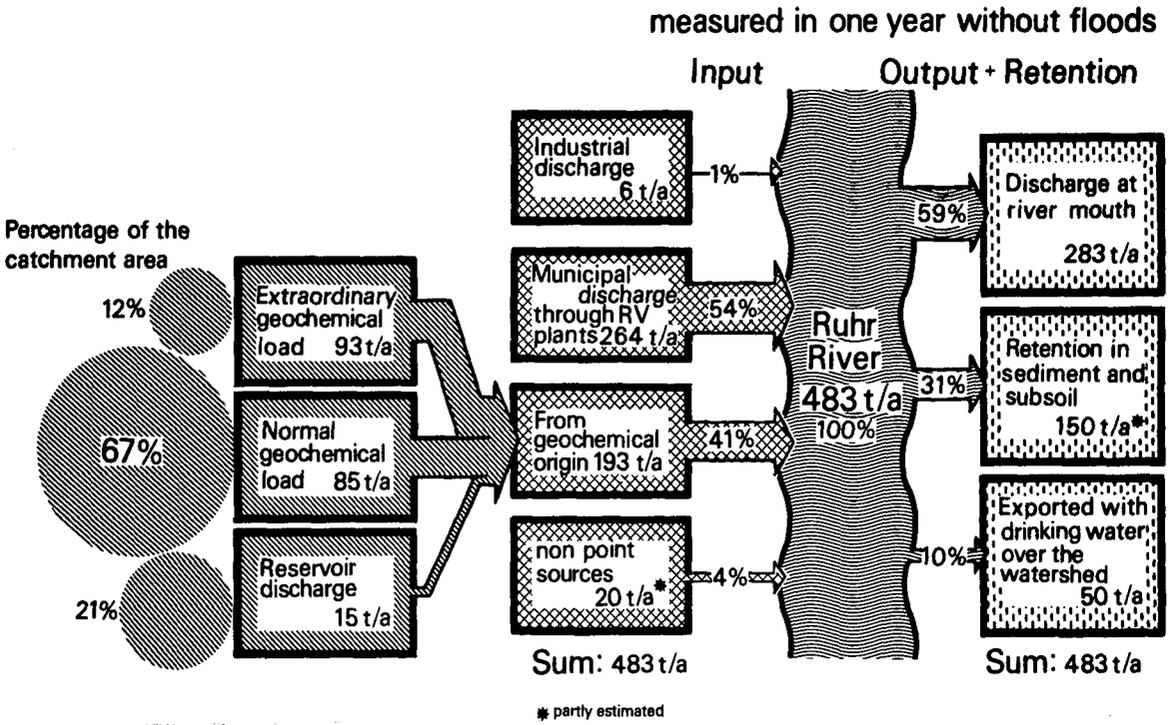


Fig. 5: Balance of heavy metal loads in the Ruhr River basin

- Bacterial counts are reduced in the lake by agglomeration at particles and sedimentation to the order of 90 %.

- Nitrification occurs with considerable lower rates in the lake as in free flowing reaches of the river. Oxygen consumption by nitrification processes is therefore not important in the impounded reaches of the Ruhr [5].

- A potential secondary pollution is created in the impoundments by mass algal development in summer. Consequently, oxygen depletions may result and have to be compensated for by instream aeration [6].

In order to keep sufficient conditions for boating and other recreational uses a minimum depth between 2 and 2.5 m is required. Sediment deposits in the Ruhr impoundments are observed annually by echo sounding monitoring. By hydraulic dredging 1.1 million m^3 of sediments were removed from Lake Baldeney in 1983/84. The total cost including sedimentation lagoons and land use was 11 million DM. As supernatant water from the lagoons flows back into the river (Fig. 6) its quality has been analyzed.

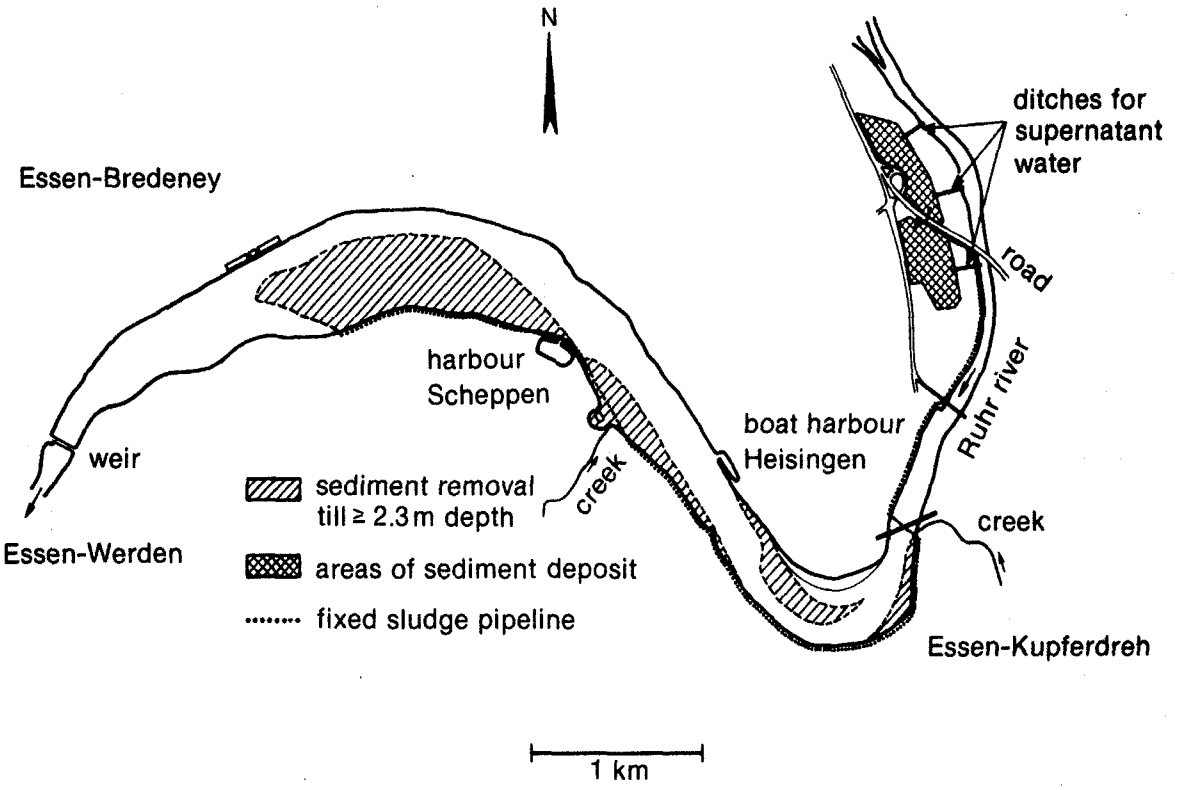


Fig. 6: Sediment removal from Lake Baldeney

The following results were obtained:

BOD_5	8.4	mg/l
COD	23	mg/l
NH_4	8	mg/l
Fe, Mn	<0.1	mg/l
Cu, Zn, Ni, Cr, Pb, Cd	<0.01	mg/l

Half a year after finishing sludge dredging wild herbs, bushes, and trees have grown already without any human aid to a height of half a meter on the surface of the sediment lagoons. Some rare species are among them. According to German legislation a detailed plan for landscaping of the lagoons had to be designed before starting the measure, but it is doubtful if this will be performed in future, as a new probably also valuable "substitute" habitate develops.

3. Stormwater Treatment

After completion of the system of biological wastewater treatment plants and purification lakes stormwater of the mostly combined sewerage systems

remained as the now major pollution to rivers. According to a basic study for the German Federal Government, the load of organic pollution from storm overflows without any treatment represents on average 12 % of the per capita organic load of domestic wastewater, which is shown as 0.12 pollution units (pu) per capita in Fig. 7.

Although the accurate figure is strongly dependent on the particular circumstances of a drainage area, i.e. topography, size, urban structure, and others, 0.12 pu/capita give a general idea of the relative order of magnitude of the waste load resulting from storm water. Fig. 7 indicates schematically the waste load to rivers which can be reduced by 50 % if in accordance to the German guideline A 128 [7] storm water flow is retained by storage and later is discharged to the municipal treatment plants, where up to twice the dry weather flow is treated biologically. The dimensioning rules of this guideline consider especially the storage of the highly polluted first flushing flood and ensure the transport of up to 90 % of the storm water flow into the treatment plant at critical conditions in the receiving water by calculating a critical storm water flow rate for any particular case. In dependency of the ratio between average low flow of the receiving water and the total wastewater flow including the storm water the critical rainfall rate is between 7 and 15 l/(s · ha). It is assumed that herewith 75 % of the annual organic load from storm water can be removed.

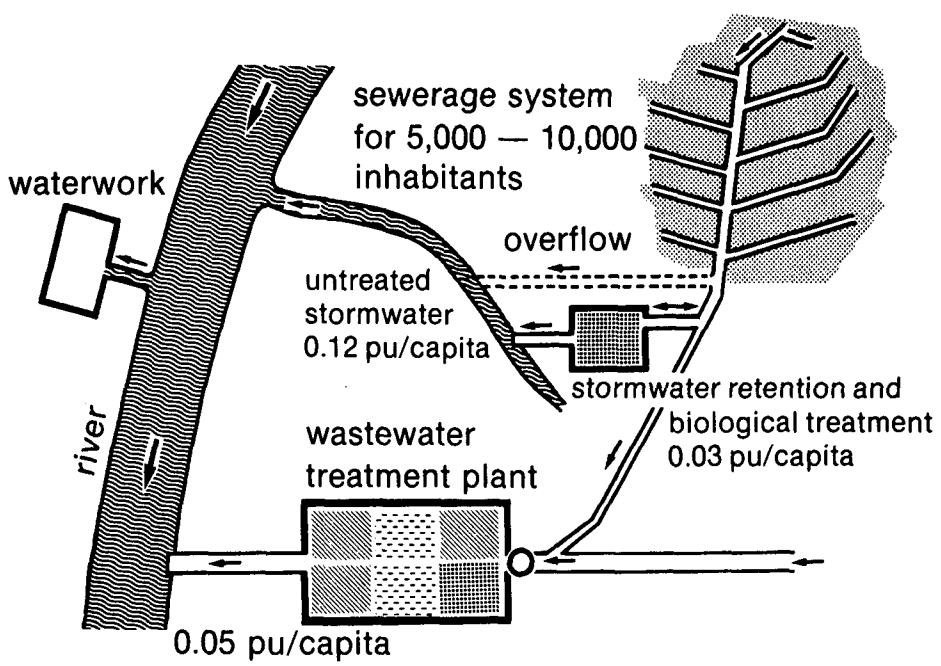


Fig. 7: The new storm water treatment concept results in more than 50 % load reduction

The German Federal Law on Wastewater Charges includes a charge for storm water discharges in addition to the charge for dry weather effluent of treatment works. According to State Water Laws the storm water charge can be reduced or even cancelled if storm water retention tanks of appropriate size are existent. Thus, a double incentive is given to construct storm water retention facilities from the financial and the water pollution control point of view. A new 100 million DM program to fight this part of pollution has therefore been approved for the Ruhr basin.

Some of the stringent State requirements were incentive to the development of sophisticated models to determine the pollution load from storm overflows. It seems, however, that this ambitious approach has failed because of the most complex interrelationships. As already the accurate determination of the discharged quantities from the particular systems is a very difficult approach, the Ruhrverband has decided to develop a practical hydraulic model. Recent discussions indicate the tendency of changing sophisticated State regulations on storm water charges to more simple calculations.

4. Particular Strategies

To avoid detrimental effects for the Ruhr water works from accidences on highways a detailed field study has been performed in the early sixties [8]. It resulted in design recommendations for retention tanks to be constructed at highways in the Ruhr valley. The effluent from highways thus passes in any case a tank where oil and other hazardous substances can be retained and abstracted before reaching the Ruhr River. The structures are simple and not equipped with automatical devices. They are designed for a storm water flow rate of 100 l/(s · ha) and a retention time of 10 minutes. The hydraulic surface loading is 12 m/h. The tanks are operated by highway maintenance services and observed by personnel of the Ruhr River Association.

A new French investigation has indicated that from double laned roads 230 to 400 kg COD/(km · a) result. For lead 0.9 to 1.3 kg/(km · a) were observed. The pollution load from roads is comparable to that of domestic wastewater related to the surface area. Only few heavy rainfalls may flush 30 % of the annual load into the environment [9].

Other strategies have to be applied in the case of major agricultural districts.

It is an aim to educate farmers to save artificial fertilizers, to apply manure properly, and to avoid erosion for example by appropriate ploughing in hilly areas, especially in the catchments of reservoirs, which are even more sensitive to eutrophication as river impoundments.

5. Summary and Conclusions

In the past, nonpoint pollution has been transferred into point pollution by the construction of sewerage systems. Consequently, this point pollution was more and more decreased over the decades. The greater the progress in this area of water pollution control the larger the percentage of residual nonpoint pollution as a part of the total river load. For the Ruhr basin a system of river impoundments has proven to be a feasible measure for river water polishing and nonpoint pollution reduction. As impoundments on the other hand may create disadvantages by forming secondary pollution from eutrophication processes, the task remains to find an optimum between impounded and free flowing reaches in a river system.

A new storm water treatment concept will lead to further reduction of residual pollution to rivers.

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USING LINEAR OPTIMIZATION FOR URBAN NONPOINT SOURCE POLLUTION WATER QUALITY MANAGEMENT

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ABSTRACT

Urban nonpoint source pollution is a principal cause of water quality problems in the United States. This pollution can be reduced by implementing management practices. One factor influencing management practice selection is the spending level at which diminishing marginal returns occur.

This study demonstrates diminishing marginal returns for spending to reduce nonpoint source pollution in Hampton, Virginia. A linear optimization program selected the best combination of management practices to minimize the cost of nonpoint source pollution control. Management practice options were grass swale roadways, porous pavers, detention basins, ponds and fertilizer management. Pollutants evaluated were total nitrogen, total phosphorus, suspended solids, fecal coliforms and five-day biochemical oxygen demand.

Additionally, a sensitivity analysis was conducted for total phosphorus to determine how sensitive overall nonpoint source pollution control costs were to key management practice data used in the linear optimization problem. Part of the sensitivity analysis examined the impact of using more site-specific cost data.

Significant diminishing marginal returns occurred above 40.0 percent removal for nitrogen and 50.0 percent removal for phosphorus, suspended solids and fecal coliforms. Overall pollution reduction cost was most sensitive to the pollution removal data for grass swale roadways and ponds. Using more site specific cost data in the sensitivity analysis lowered the spending level of diminishing marginal returns.

Keywords: Urban runoff pollution, nonpoint source pollution, linear optimization, water quality models

INTRODUCTION

In a recent United States Environmental Protection Agency report to Congress on nonpoint source (NPS) pollution, six of ten Agency regions specified NPS pollution as the principal cause of water quality problems (USEPA,1984). This report also listed urban NPS pollution as one of the most commonly cited problems. Urban NPS pollution presents a special challenge to water quality managers, because the many available pollution removal and treatment options (management practices) have cost and pollution removal characteristics which vary with land-use. This complicates selecting optimum treatment practices and spending levels. The focus of this study is on a factor which influences decisions on spending levels -- diminishing marginal returns. Diminishing marginal returns occur at the point where less pollution removal occurs per dollar spent.

This study's objectives were to find out if diminishing marginal returns would occur for spending on management practices in Hampton, Virginia and to evaluate how sensitive the point of diminishing marginal returns was to the management practice and land-use data in the problem. The sensitivity of the results to the data was evaluated to determine the importance of data accuracy. This information is essential to evaluate the reliability of the results and the effect of data assumptions.

METHODOLOGY

To achieve the diminishing marginal return objective, a linear optimization problem was created for each pollutant. These problems included the cost and pollution removal characteristics of every management practice in every land-use area. The data for the allowable pollution was set at the desired level and the problem was solved. The problem solution contained the overall cost of achieving the pollution level and a cost breakout, i.e., a list of the management practice area recommended in each land-use that contributed to the overall cost. The allowable pollution was made more stringent in 10.0% increments from 10.0% to 60.0% pollution reduction. This systematic adjustment provided cost and management practice data for a range of pollution removal which was used to analyze diminishing marginal returns. These results are reported in a graphic presentation of percent pollution removal and cost. The results are also presented in the form of cost per percentage pollution removal.

The sensitivity of the problem to the data was evaluated using a single pollutant. The approach was to change one data element and solve the linear optimization problem again at each of the pollution removal increments. The difference between the new (modified data) cost and the previous cost was the result of the data element change. These differences are reported for 50.0% pollution removal in the results. Another form of sensitivity evaluated was the sensitivity of the point of diminishing marginal returns to the level of detail of the land-use data. In this analysis a comparison was made between two linear optimization problems for the same city subbasin. A consolidated problem divided the subbasin into 8 areas and a parcel problem divided the subbasin into 33 smaller areas. The affect on the level of spending at which diminishing marginal returns occurs is shown.

Linear optimization has been used as a water quality management tool in the past. Anderson and Day (1968) minimized the regional operating costs

of conventional point source waste treatment with linear optimization. Revelle et al. (1968) compared linear programming results to dynamic programming to show both techniques yielded essentially the same result. Smith and Morris (1969) optimized water-use goals and management options using point sources as the modified management practice and dissolved oxygen as the water quality constraint. Watershed Handbook (USEPA,1981) illustrated how to use linear programming to minimize the cost of reducing point and NPS pollution flowing into a water body.

Linear optimization model

The linear optimization approach was to minimize the cost of NPS pollution control subject to land-use and water quality constraints. The land-use constraints required the sum of all the management practices and the land where there was no recommended management practice to equal the total area in each land-use category. Water quality constraints set limits on the NPS pollution permitted in each land-use area. Without water quality limitations, no management practices were necessary and all the urban NPS pollution reached the receiving water. Making the water quality constraints more stringent, i.e., requiring larger reductions in pollutants, caused the program to select a land-use area and management practice for implementation to reduce pollution.

The management practice options were grass swale roadways (GSR), porous pavers (PP) -- a composite management practice including porous pavement and modular pavement, detention basins (DB), ponds (PO) and fertilizer management education programs (FM). The land-use categories were commercial strip, heavy industry, light industry, high density residential, multi-family residential, central business district, low density residential and open land.

The objective function (Equation 1) minimized the total cost (C) of implementing area in each of the management practice options (Xij) with individual costs (Cij). The management practice options were represented by subscript j, and the land-use categories were represented by subscript i.

Equation 1. $$\text{Minimize } C = \sum_{i=1}^{m} \sum_{j=1}^{n} C_{ij} X_{ij}$$

The land category constraints (Equation 2) required the sum of the management practices in each land-use area i to equal the area in i (Ai).

Equation 2. $$\sum_{j=0}^{n} X_{ij} = A_i \qquad \text{for } i = 1,2,\ldots,m$$

Each hectare (ha) of management practice Xij was assigned a fixed amount of NPS pollution that would be generated yearly (Lij). The sum of LijXij for each land-use area was a water quality constraint equation (Equation 3). The right side of the water quality constraint (w.q.) was set at fixed increments. Non-negativity constraints required Xij be greater than zero ($X_{ij} \geq 0$) because negative area was not feasible. A detailed formulation of the problem is described by Garland (1985).

Equation 3. $$\sum_{j=1}^{n} L_{ij}X_{ij} \leq w.q. \quad \text{for } i = 1,2,\ldots,m$$

The result of these equations was a linear programming problem with 40 variables and 16 constraints for each pollutant.

Management practice costs

The assigned cost of each management practice ($/ha) was based on the management practice cost and the different characteristics of each land-use area (Table 1). Land-use influenced cost because most management practices could only be applied to a fraction of the total land-use area. Other land-use factors which affected cost were length of roadway, percentage of impervious area and dwelling unit density. These factors influenced GSR, PP and FM costs respectively.

GSR for example, can only be applied to roadways in certain land-use areas because they require wider right-of-ways and are not feasible on main highways with a high number of access roads. This led to the assumption that GSR were not applicable in commercial strip and central business districts, and they were only feasible on 20% of industrial land. GSR were not considered a management practice option for open land, because limited roadways would only provide very limited overall pollution reduction. The cost ($/meter) for GSR was obtained from the Hampton Roads Water Quality Management Agency (1978). The length of roadway in each hectare (m/ha) for each land-use was multiplied by the cost ($/m) and feasible area (e.g., 0.20 for industrial land) to calculate the GSR cost per hectare.

Management practice effectiveness

Both the rate of pollution generation and the ability of the management practice to remove the pollutant for each land-use area were needed to create the water quality constraint. Pollution loading data was obtained from a 1978 STORM (Storage Treatment Overflow Runoff Model) analysis of the Hampton area conducted by the Hampton Roads Water Quality Agency (1978).

Management practice effectiveness for this linear programming format was expressed in units of kg pollution loading/ha. Literature values provided figures for how effective each management practice was, but the literature did not detail how extensively the management practice could be applied in each land-use hectare. For example, porous pavers (PP) are 100% effective at removing surface runoff in a parking lot by infiltration if properly designed. However, PP are not 100% effective at removing NPS pollution on a hectare of any land-use category because they can not be applied to the entire area. The PP treatable percentage of each land-use category had to be estimated based on the impervious area of the land-use categories. The effectiveness data is displayed in Table 2. For example, in Table 2 the pond (PO) management practice in commercial strip land-use will remove 50% of the total P, SS and FC; 40% of the total N and 20% of the BOD_5.

Both PO and DB were assumed to be applicable to all of the area in every land category. Pollution removal effectiveness was based on the removal rate of three PO systems in the local area examined by Anderson and

TABLE 1

MANAGEMENT PRACTICE COST SUMMARY IN 1984 DOLLARS PER HECTARE

Land-Use	Area[a] (ha)	Management Practice ($)				
		Grass Swales	Porous Pavers	Pond[b]	Basin[b]	Fertilizer Management
Commercial Strip	811	...	44,702	11,577	36,964	...
Heavy Industry	333	911	38,935	4,787	15,284	...
Light Industry	249	911	27,399	4,532	14,472	...
High Density Residential	700	911	38,935	22,519	71,894	141
Multi-Family Residential	109	319	12,256	22,373	71,427	469
Low density Residential	5,760	190	...	15,875	50,679	32
Central Bus. District	67	...	47,587	75,582	241,312	...
Open Land	2,663	...	...	1,820	5,812	...

SOURCES: Hampton Roads Water Quality Management Plan (4) and Tourbier and Westmacott (10).

NOTE: Costs include equipment, material, installation, capital and maintenance factors. For a complete discussion of cost assumptions see Garland (3). All cost data converted to January 1984 dollars using the Department of Commerce Composite Cost Index. Infeasible combinations "...".

[a]The city-wide land-use data was from Hampton, Virginia. Hampton is a 10,753 ha city located on a peninsula.

[b]Volume determined by inflow to area ratio of 53 m^3/inflow ha for ponds and 170 m^3/inflow for detention basins. Price increased by a real estate factor in each land-use category.

TABLE 2

MANAGEMENT PRACTICE POLLUTION REMOVAL EFFECTIVENESS
(Percent)

Land-Use	Poll.	Mgt. Practice[a] GS	PP	PO	DB	FM	Land-Use	Poll.	Mgt. Practice GS	PP	PO	DB	FM
Comm. Strip	Total P	..	62	50	80	..	Multi-Family Resid.	Total P	50	17	50	80	10
	Total N	..	62	40	57	..		Total N	65	17	40	57	10
	BOD_5	..	62	20	63	..		BOD_5	73	17	20	63	..
	SS	..	62	50	70	..		SS	77	17	50	70	..
	FC	..	62	50	70	..		FC	77	17	50	70	..
Heavy Ind.	Total P	10	54	50	80	..	Low Density Resid.	Total P	50	..	50	80	10
	Total N	13	54	40	57	..		Total N	65	..	40	57	10
	BOD_5	15	54	20	63	..		BOD_5	73	..	20	63	..
	SS	15	54	50	70	..		SS	77	..	50	70	..
	FC	15	54	50	70	..		FC	77	..	50	70	..
Light Ind.	Total P	10	38	50	80	..	Cent. Bus. Dist.	Total P	..	66	50	80	..
	Total N	13	38	40	57	..		Total N	..	66	40	57	..
	BOD_5	15	38	20	63	..		BOD_5	..	66	20	63	..
	SS	15	38	50	70	..		SS	..	66	50	70	..
	FC	15	38	50	70	..		FC	..	66	50	70	..
High Density Resid.	Total P	50	54	50	80	10	Open Land	Total P	..	..	50	80	..
	Total N	65	54	40	57	10		Total N	..	..	40	57	..
	BOD_5	73	54	20	63	..		BOD_5	..	..	20	63	..
	SS	77	54	50	70	..		SS	..	..	50	70	..
	FC	77	54	50	70	..		FC	..	..	50	70	..

SOURCE: Anderson et al. (1982).

NOTE: For a complete discussion of effectiveness assumptions see Garland (1985). Infeasible combinations "..".

[a]GS -- grass roadway swales, PP -- porous pavers, PO -- ponds, DB -- detention basins and FM -- fertilizer management.

coworkers (1982). These researchers showed PO needed a 53 m^3/inflow hectare ratio in order to achieve a 50% SS reduction. A 50% removal rate for SS for DB corresponded to a storage volume to inflow ratio of 170 m^3/inflow hectare. FC were assumed to behave as particles, because most bacteria are attached to suspended solids.

FM programs were assumed to be applicable only in the residential areas since this management practice is directed toward the reduction of P and N pollution caused by excessive fertilizer application practices. The effectiveness assigned to this management practice was a 10.0% reduction in the total N and total P runoff.

RESULTS

The plots of the city-wide data for cost and percentage of pollution removal are shown in Figures 1, 2 and 3. A visual inspection of the plotted data revealed linear segments in the relationships between cost and percentage pollution removal. For BOD_5, the linear range extended from 10.0% to 60.0%. Total P, SS and FC appeared linear between 10.0% and 50.0%, and linear at a new slope between 50.0% and 60.0%. Total N appeared linear from 10.0% to 40.0% and then linear at a new slope above 40.0%.

A best fit line was developed with linear regression and the slope used to calculate the cost per percentage pollution removed ($/1%). This data is shown in Table 3. There was a perfect correlation of the data (one) when the points being measured represented costs from the same set of recommended management practices. When the correlation dropped slightly, e.g., to 0.9999, this indicated a region where a new management practice was added to the treatment set or one of the management practices was no longer recommended.

No feasible solution was possible above 57.0% removal for total N because none of the management practice options would remove more than 57.0% of the pollution load for industrial and open land.

BOD_5 was the most expensive pollutant to remove in the 10.0% to 50.0% removal range, costing about 1.5 times the removal cost of the other pollutants. BOD_5 cost more to remove due to the low effectiveness of PO. Because PO were only capable of removing 20.0% of the initial BOD_5, the DB management practice had to be used in some land categories for removal rates starting at 20.0%. DB cost between three and four times as much per hectare as PO creating the high cost for BOD_5 pollution removal.

Above 40.0%, total N removal costs also increased due to the switch in recommended management practices to DB instead of PO and GSR in industrial areas. For SS, FC and total P, DB were not selected until above 50.0% pollution removal.

The extremely high cost of total P removal above 50.0% was caused by the requirement to implement DB in residential areas while all the other pollutants were still removed by GSR management practice. DB cost from 70 to 270 times the cost of GSR per hectare in residential areas.

As Figure 4 shows, linear segments also occurred in the subbasin analysis by parcels and by consolidated land-use areas. The first segment was from 10.0% to 50.0% pollution removal. In this range, the parcel cost per percentage pollution removed was $47,825/1% removed and the consolidated cost was $48,679/1% removed. The second segment was 50.0% to 60.0% pollution removal, where parcel analysis cost was $2,097,985/1% removed and the consolidated cost was $1,231,235/1% removed.

The subbasin analysis showed that more detailed examination lowered

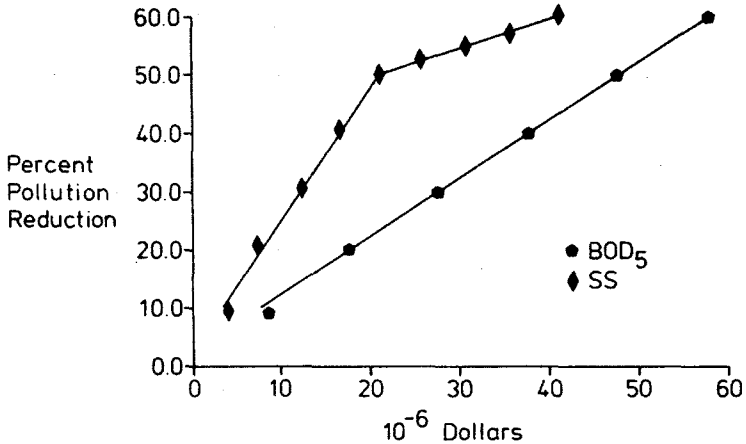

FIGURE 1. CITY-WIDE POLLUTION REDUCTION COSTS FOR BOD_5 AND SS IN 1984 DOLLARS.

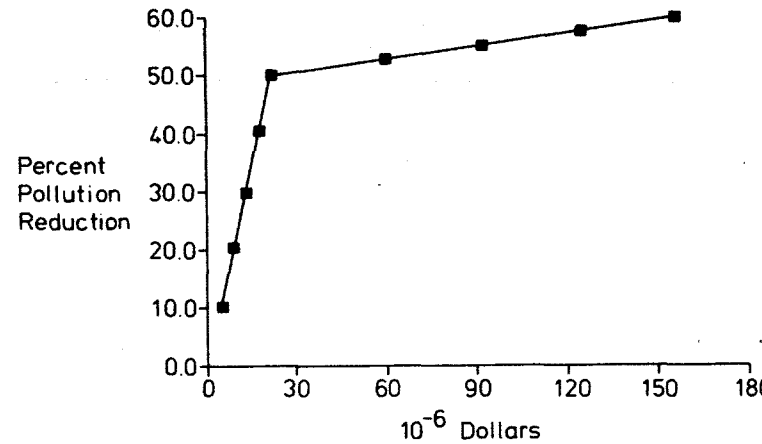

FIGURE 2. CITY-WIDE POLLUTION REDUCTION COSTS FOR TOTAL P IN 1984 DOLLARS.

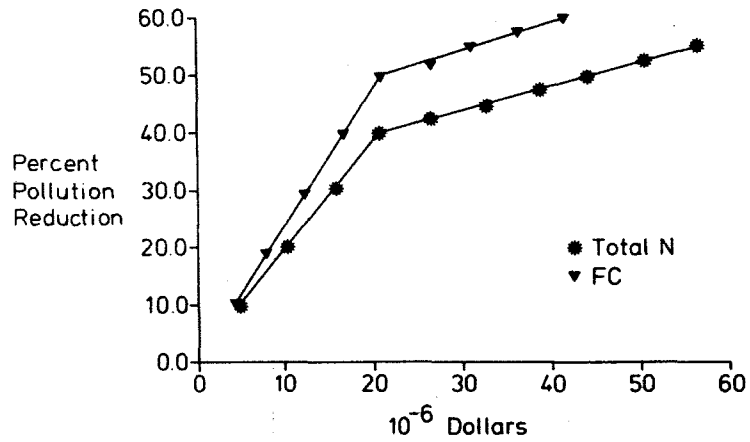

FIGURE 3. CITY-WIDE POLLUTION REDUCTION COSTS FOR FC AND TOTAL N IN 1984 DOLLARS.

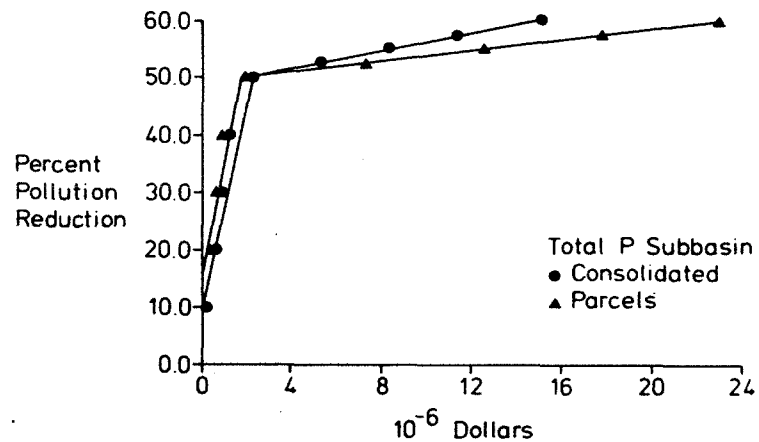

FIGURE 4. SUBBASIN AND PARCEL ANALYSIS POLLUTION REDUCTION COSTS FOR TOTAL P IN 1984 DOLLARS

TABLE 3

CITY-WIDE DATA ANALYSIS

Pollutant	Pollution Removal Range (%)	Cost/Percent Pollution Removed (1984 Dollars)
BOD_5	10-60	992,603
SS	10-50	427,460
	50-60	1,996,428
FC	10-50	429,938
	50-60	2,006,944
Total N	10-40	526,510
	40-60	2,364,271
Total P	10-50	592,273
	50-60	12,735,077

TABLE 4

SENSITIVITY ANALYSIS SUMMARY FOR 50% TOTAL P REMOVAL AND CITY-WIDE LAND-USE DATA

Mgt. Practice/ Variable	Change to all Land-use Areas	Total Cost % Increase
Grass Swale Roadways	Lower effectiveness 10%	154.38
Ponds	Lower effectiveness 10%	19.56
Ponds	Increase cost 10%	13.57
Open Land	Raise area 20% and decrease other areas fractionally	1.45
Porous Pavers	Increase cost 10%	0.86
Grass Swale Roadways	Increase cost 10%	0.62
Open Land	Raise area 10% and decrease other areas fractionally	0.53
Fertilizer Management	Raise cost 10.0%	0.00
	Lower effectiveness 10.0%	0.00

costs at low pollution removal levels and raised costs above 50% pollution removal. This effect was caused by using 33 different prices (one for each parcel for DO and DB) rather than 8 different prices (one average figure for PO and DB for each land-use category). Below 50.0% pollution removal in the parcel analysis, some PO and DB parcel prices were inexpensive enough to cause their selection while they were not selected in the consolidated problem which used the average figures. Above 50.0% pollution removal, all of the inexpensive PO and DB parcel area had been used. Additionally, there were no other management practice options which could achieve the desired pollution removal. The result was large overall cost increases in the parcel problem because more expensive PO and DB area was recommended.

The parcel approach had the additional advantage of providing more information on where to implement the recommended management practices. For example, the consolidated problem recommended 9.9 ha of GSR in low density residential area, while the parcel problem specified GSR in low density residential area in the following parcels: 0.0 ha in parcel A, 3.2 ha in parcel B, 0.8 ha in parcel C, 5.6 ha in parcel D, 0.3 ha in parcel E. The disadvantage of the parcel problem was the problem size -- 165 possible management practice and land-use combinations and 66 constraints.

The cost sensitivity analysis results are shown in Table 4. At the total P removal levels examined, the greatest overall cost change was caused by changing PO cost. FM was not selected at 50.0% pollution removal, so the change in FM cost had no impact on overall cost. Below 50.0% pollution removal, a 10.0% increase in FM cost created a change in overall cost of approximately 1%, e.g., at 10.0% pollution removal a 0.65% overall cost change occurred as the result of a 10.0% increase in FM cost. Increasing PO costs had a greater effect than other cost sensitivity changes because the PO fraction of the overall cost was greater. For example, the area recommended for PO in all the land-use categories (4,106 ha) accounts for 62.74% of total costs at 50.0% pollution removal compared to 6.20% of the overall cost for the GSR area (6,444 ha).

Decreasing management practice effectiveness caused an increase in pollution removal costs due to the increase in management practice area which had to be implemented to achieve the desired pollution reduction. In some cases the decrease in management practice effectiveness made it necessary to switch practices. The increases in overall cost caused by the drop in the management practice effectiveness of PO and GSR (19.56% and 154.38% respectively) at the 50.0% pollution removal level was caused by a switch in management practices. For example, at the 50.0% pollution removal level PO capable of removing 50.0% of the pollution could be implemented on all the land-use area and achieve the desired pollution reduction. When PO effectiveness is reduced to 45.0%, another more effective and more expensive management practice (DB) must be selected in order to achieve 50.0% pollution removal. When DB was selected to augment PO or GSR at the 50.0% pollution level the cost increased greatly for each percentage of pollution removal.

An additional sensitivity analysis raised the effectiveness of GSR in residential areas to 80.0% for total P. The primary reason for this analysis was to simulate the lack of diminishing marginal returns demonstrated by BOD_5 with another pollutant. One of the main differences between BOD_5 and the other pollutants was GSR removal effectiveness. GSR are 73.0% effective for removing BOD_5 in residential areas. Improving the GSR effectiveness to 80.0% for total P eliminated the need to recommend any other management practices in the residential land categories. This switch to one management practice

for the residential areas did cause a dampening of diminishing marginal returns between 40.0% and 50.0% pollution removal. The results were similar to those shown for BOD_5.

The fraction of the open land land-use area was examined to see how sensitive the results were to errors in land-use data collection caused by using zoning data. An open land area data error would occur if a 10 ha area zoned commercial strip was in fact 8 ha of commercial strip and 2 ha of open land.

Increasing open land area increased removal cost in the 10.0% to 50.0% pollution removal range and decreased removal cost above 50.0%. This occurred because the average cost of treatment of open land below 50.0% pollution removal exeeded the average cost of treatment in other land-use areas. However, above 50.0% removal DB and PO, the recommended practices on open land, were less expensive than the management practices recommended on other land-use areas.

CONCLUSIONS

Significant diminishing marginal returns did occur for all pollutants except BOD_5 at spending levels between $20 and $28 million. This spending level is too high to effect Hampton's expected NPS pollution control program, because it represents approximately 25% of the city's total expenditures.

Pollution removal costs were the sum of the costs of the management practices in each land-use category. As additional acres of a set of management practices were added to meet the more rigid water quality criteria, a linear relationship was formed between cost and pollution reduction. Diminishing marginal returns occurred when the cost of removing an additional percentage of pollution increased.

A new set of management practices was selected when the old set would no longer achieve the desired amount of pollution removal. This occurred at 40.0% pollution removal for total nitrogen and 50.0% removal for total phosphorus, suspended solids and fecal coliforms. New management practices were introduced causing a change in the cost and pollution removal relationship. Each new set of management practices was capable of removing less pollution per dollar spent than the previous management practice set thereby creating diminishing marginal returns.

The five-day biochemical oxygen demand cost per percentage removal remained approximately the same from 10.0% to 60.0% pollution removal levels. A sensitivity analysis showed this situation was caused primarily by the assumptions making grass swale roadways inexpensive and highly efficient at removing biochemical oxygen demand through 73.0% removal levels.

In general, pollution removal effectiveness was the most sensitive input variable, then management practice cost and then the percentage of open land. The point of diminishing marginal returns was also affected by the level of detail of the analysis. The parcel analysis showed more site specific management practice costs reduced the spending level of diminishing marginal returns. By using more detailed cost data for ponds and detention basins, this analysis showed the spending level needed to encounter diminishing marginal returns for total phosphorus in the subbasin decreased by $201,436.

While this study focused on spending levels for treating urban nonpoint source pollution, linear programming is a potentially valuable tool for selecting the optimum combination of acreage for application in each

management practice option. The optimum set of management practices under the assumptions of the study were similar for all pollutants. Ponds and detention basins were recommended for commercial strip. Porous pavers were recommended for central business districts. For industrial areas, grass swale roadways were recommended below 10.0% pollution removal. Between 10.0% and 50.0% pollution removal, a combination of ponds and grass swale roadways was recommended. Where five-day biochemical oxygen demand was the pollutant, detention basins were recommended instead of ponds in the 10.0% to 50.0% pollution removal range to compensate for poor biochemical oxygen demand removal by ponds. A combination of ponds and detention basins was recommended for all pollutants above 50.0% pollution reduction. The management practice recommendation below 50.0% removal for residential areas was grass swale roadways. Above 50.0% pollution reduction, a combination of detention basins and grass swale roadways was recommended. Open land treatment was by ponds below 50.0% pollution removal and a combination of ponds and detention basins above 50.0% pollution removal.

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DESIGN OF URBAN DETENTION BASINS FOR WATER QUALITY CONTROL

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ABSTRACT

Engineering effort has been traditionally placed on improvements in the hydraulic design and management of urban storm drainage detention facilities with quality control considerations of stormflow performance receiving relatively little attention. Nevertheless, the pollutional impact of such routed flood discharges on receiving streams in urban catchments can be significant. The paper examines pollutant settleability and removal efficiencies of flood detention basins and considers the effects of particle size, pollutant form and timing on trap efficiency.

It is shown that timing and partitioning of pollutants during stormflow events are not amenable to the quality control function of detention basins as best management engineering practice. The significance of particle size and settling velocities on pollutant removal within storage basins are demonstrated through sedimentation column tests and application of trap efficiency formulae. Efficiencies of between 18 to 62% were found for various pollutants although these values were considerably reduced if adjustments are made to allow for turbulent flow routing through the basin.

Keywords: Urban runoff pollution, pollution retention, trap efficiency, particle size and contaminant controls, detention basins.

NOMENCLATURE

V_s = particle setting velocity

L_b = basin length

$\bar{v}$ = mean flow velocity in basin

y = basin depth

n = manning roughness coefficient for basin floor

g = gravitational constant

v_* = shear velocity

η = efficiency function

INTRODUCTION

It is well recognised that urbanisation causes permanent changes in the storm runoff regime and leads to increases in both the magnitude and frequency of peak flows discharged to receiving streams. It is equally clear that urban developments lead to a marked deterioration in the water quality of these discharged stormflows despite the enhanced volume available for dilution. (Ellis, 1982).

A common low-cost engineering means of limiting the effects of development on the hydrological response of a catchment and of maintaining the predevelopment downstream river flow regime, is to provide temporary storm storage facilities within the urban catchment. In effect, a detention pond represents an attempt to replace the natural storage capacity lost through urban development, although since the latter was distributed throughout the catchment and the former is located at a single site, the comparison is not totally appropriate.

The design of such detention basins has been conventionally based on hydrological and hydraulic criteria with their size and capacity predetermined from selected peak flow rates for a specified recurrence probability. The design methods provided by Davis (1963) or Hall and Hockin (1985) allow the routed peak storm inflows to be delayed and attentuated through a restricted outlet or overflow capacity. Unfortunately, the design of what Whipple (1979) has referred to as dual-purpose detention basins has to resolve the conflicting requirements of water quantity and quality. There is a limited amount of post-project appraisal work available to evaluate the hydraulic design performance of these control structures (ASCE, 1982) but little consideration has been given to their trap efficiency in water quality terms. It is not known if the designated storm for any particular basin serves to retain the pollutant inflow loadings through sedimentation or if the release time is sufficient to regulate and minimise receiving stream quality impacts. The concept of a "settleable design storm" expressed in terms of specific pollutant parameters for flood detention basins remains to be formulated and tested.

POLLUTANT RETENTION IN FLOOD DETENTION BASINS

Whilst short term retardation and attenuation of urban flood flows by

storm detention basins may represent a best management practice in hydraulic terms, it does not necessarily provide the most effective means of retaining or reducing associated pollutant flows. Urban detention basins are normally designed to balance short-duration, high intensity storm events since these are the cause of the most frequent downstream flood activity in the receiving stream. These basins release the detained peak inflows over a period of a few hours at most, although gated and throttled outlets can retain water for rather longer periods. However, it can be argued that it is the aggregate effect of storms of a return period less than that of the design storm for flood control that is of most importance in terms of water quality. Although the major flood damage may result from the larger more infrequent storms, the bulk of the contaminant loading is carried by the small and medium sized storm events.

In addition, the proportion of pollutants incorporated within the "first flush" of the design flow that will be detained by the flood storage basin can be extremely variable. Fig. 1 demonstrates the runoff-pollutant timing curves for a flood flow of 10 year return period within a 350 hectare separately sewered urban catchment in Hendon, NW London.

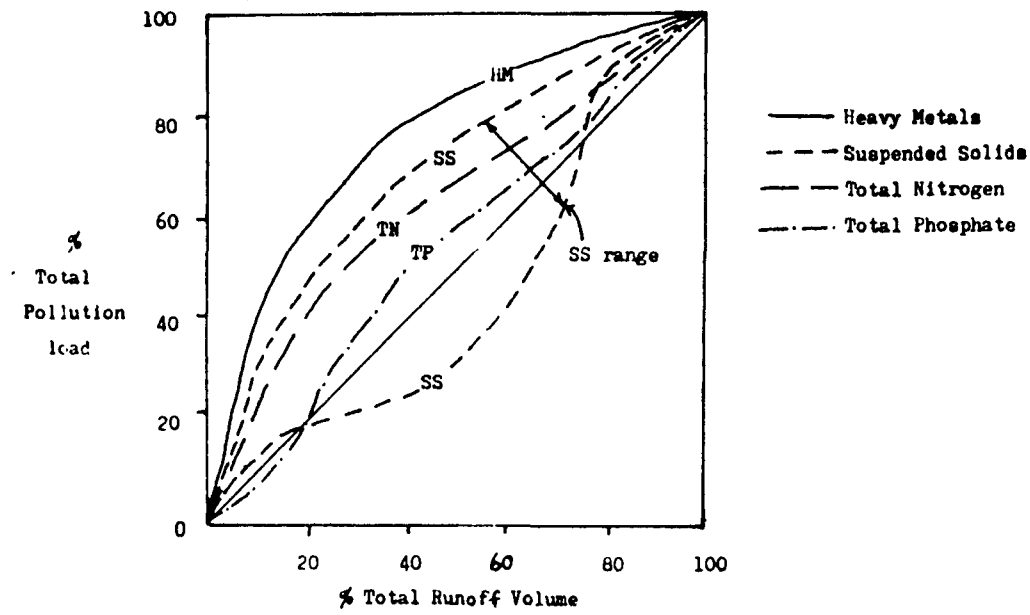

Figure 1. Runoff - Pollutant Timing Curves

Metals exhibit the greatest propensity towards a first flush effect and this might be expected given their known affinity to solids. The greatest deviations from peak flow are shown for dissolved constitutuents although delayed 'last flushes' for solids, cadmium, zinc and hydrocarbons are not uncommon (Revitt et al, 1982; Morrison et al, 1984) and the random variability of pollutant parameters in urban runoff is well documented. (US EPA, 1983.) Therefore, the initial two hours of the first flush design inflows to the urban detention basin can contain

between 60 to 80% of the total storm pollutant mass. (Ellis, 1979). However, the partitioning of pollutants between solid and dissolved states implies that those which remain in the latter phase will not be amenable to control through any solids retention and this may substantially reduce the notional retention percentages quoted. Sizeable fractions of nitrogen (60 to 75%) and phosphorous (30 to 55%) have been shown by Grizzard and Randall (1978) to exist in the dissolved phase of urban stormflows. Runoff studies by Morrison et al (1984) in NW London and Gothenburg, Sweden have shown that as much as 63%, 77%, 60% and 32% of total zinc, cadmium, lead and copper respectively are held in the dissolved or easily exchangeable form. Therefore, the form or speciation of the pollutant in storm flows needs to be ascertained to adequately evaluate the retention efficiency and design performance of the detention basin as a quality control structure.

A further and fundamental complication to the quality control performance of urban flood detention basins is in the particle size association of the storm runoff pollutants. Ahtough only some 8 to 25% of the total solids loading to the basin will be of particle size less than 150 microns, it is this fine size fraction which accounts for much of the pollution potential. This size range can typically include 30 to 50% of algal nutrients, 30% of heavy metals, 50 to 60% of hydrocarbons, 25% of the oxygen demand and 20 to 40% of total coliforms. (Sartor et al, 1974; Ellis, 1976). Given the low settling velocities of such fine contaminated particles and the strongly turbulent nature of both inlet and outlet flows, it is highly probable that such material will not be retained within the basin but will be routed in suspension directly through the structural facility. The high flow rates maintained during flood flows also stimulate disaggregation of organically bound flocs in the stormwater suspension (Ellis et al, 1982) and this further exacerbates the quality control problem.

Despite the anomalies presented by partitioning and size association, analysis and appreciation of the pollutant retention efficiency of flood detention basins must proceed through considerations of sedimentation rates and sediment related quality parameters.

TRAP EFFICIENCY OF FLOOD DETENTION BASINS

The average concentration of total suspended solids (TSS) in urban storm runoff varies between 100 and 600 mg l^{-1} (Ellis, 1982; US EPA, 1983) although individual low return storm flows can average as much as 1700 mg l^{-1}. Settleable solids greater than 150 microns, as determined by settling through a 1.83m column of 0.3m diameter for 24 hours, comprise between 60 and 90% of TSS and possess specific gravities ranging between 0.8 and 2.6. The settling velocities of stormwater runoff solids as determined under these static conditions varied from 0.001 cm sec^{-1} for fine silts/clays up to 8.5 cm sec^{-1} for coarse sand/grit particulate of 1/2 mm. The modal stormwater suspended sediment diameters of 15 to 20 microns (Ellis et al, 1982) and the 150 micron cut-off defining settleable solids possess settling velocities of 0.01 cm sec^{-1} respectively. These theoretically determined values do require some adjustment to allow for the relatively turbulent flow regime conditions obtained in the detention basin. It has been suggested that effective settling velocities, for the computation of net pollutant removal, may be as low as one-ninth of those determined under quiescent laboratory conditions. (Northern Virginia Planning District Commission, 1980.)

A number of methods are available to predict the pollutant trap efficiency of detention basins of which the oldest and simplest is that of Camp (1945). This method, like the more recent but complex modelling approach of Ferrara and Hildick-Smith (1982), utilises the geometry and hydraulics of the basin together with computed settling velocities. The three parameter sediment removal function of Camp is

$$\eta = \text{function}\left[\frac{V_s L_b}{\bar{v} y} \quad ; \quad \frac{V_s y^{1/6}}{\bar{v} n \sqrt{g}}\right]$$

and expresses removal as a function of surface loading with a turbulence function equivalent to V_*/V_s. The removal efficiencies can be read from the graph (Fig.2) following calculation and insertion of the appropriate values.

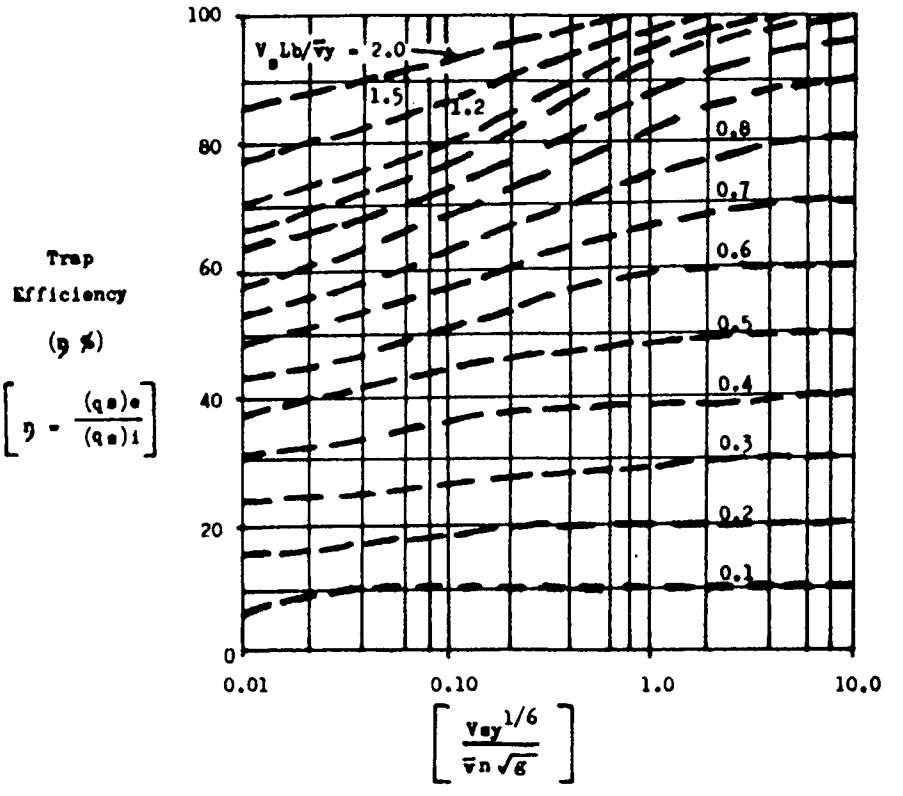

Figure 2. Detention Basin Trap Efficiency

Table 1 shows pollutant settleability and trap efficiencies for a number of test runs and storm events; the latter determined by the Camp method using data for a small urban detention pond draining 14.9 hectares of the larger separately sewered, residential catchment in NW London mentioned earlier.

The tabulated results illustrate the potential utility of the Camp methodology, which in conjunction with adequate hydraulic criteria can provide a dual purpose approach to detention basin design. The design also emphasise the relatively low efficiencies of existing structural designs in providing a quality control, particularly when subject to turbulent flow regime conditions. This conclusion is confirmed by the field observations and from independent studies (Ferrara and Witkowski,

TABLE 1

SETTEABILITY AND TRAP EFFICIENCIES OF DETENTION BASIN POLLUTANTS

Pollutant	(1) Imhoff Settleability (Average % Removal)	(2) 2 Hour Removal as % of 24 hour Removal	(3) % Trap Efficiency ($Vs_{150\mu m}$)	(4) % Trap Efficiency ($Vs_{150}/9$)	(5) Observed % Trap Efficiency
TSS	73	34	62	42	6 - 84 (249) - 86[abcd]
BOD_5	32	13	48	28	29 - 48
Hydrocarbons	67	18	60	35	20 - 60[cde]
Lead	62	30	59	42	10 - 40 (16) - 70[cd]
Cadmium	26	15	18	2	(48) - 21[c]
Total Phosphate	46	20	41	10	10 - 78[abf] 20 - 70
Total Coliforms	85	60	-	-	7 - 73

NOTE: Numbers in parentheses indicate negative trap efficiencies; data from following references:-

[a] Ferrara and Witkowski, 1983
[b] Grizzard et al, 1982
[c] Dally et al, 1983
[d] Whipple and Hunter, 1981
[e] Ellis et al, 1985
[f] Ferrara and Hildick-Smith, 1982

1983; Grizzard et al, 1982; Dally et al, 1983), although some recently derived modelling approaches would indicate very much higher trap efficiencies in respect of TSS. (Amandes and Bedient, 1980; McCuen, 1980.) However, application of the Camp method with settling velocities for particle sizes within the range 20 to 60 microns produce extremely low or event negative trap efficiencies and the validity of this is borne out from the extreme ranges observed in the field. Therefore, the particle size association of pollutants is of importance to the removal efficiency, despite assertions occasionally found in the literature that pollutant removal shows little correlation with sediment removal (Whipple and Hunter, 1981; Kuo and Ni, 1984). Nevertheless, the low computed and observed trap efficiencies would imply that factors other than the immediate storm hydralics and pollutant settleability are operating within the detention basin to determine the resultant outflow quality. The most obvious of these would be the exchange of pollutants between the deposited and reducing or anaerobic basal sediments to the overlying water phase of the detention basin. It is well documented that trace metals nutrients and hydrocarbons accumulate in detention and flood storage basin sediments, (Wignington et al, 1983; Hvitved-Jacovbsen et al, 1984; Ellis et al, 1985) and during storm plug inflows these pollutants can be released and mobilised from zones of maximum accumulation such as occur near outlets which experience the longest periods of standing water. This resuspension and mobilisation phase will also transfer some of the pollutant load to the dissolved phase, particularly in the case of pollutants such as cadmium, zinc or nitrate which have high exchangeable potential. It may be that the short retention time of materials in the basin during storm events would imply that substantial transformation of particulate to soluble form is not likely. However, the high degree of turbulence would facilitate exchange and in any case this argument could not be applied to 'wet' basins where flushing of soluble material generated during the preceeding dry period

would inevitable take place.

CONCLUSIONS

Considerations of the potential pollutant removal efficiency of urban storm detention basins require a coupling of their particle size and settling velocity characteristics. If this relationship can be successfully and reliably determined it can be used in conjunction with hydraulic and geometrical criteria to provide a dual purpose procedure for detention basin design. The operational quality performance of any basin, however, will also be influenced by post-design criteria which depend on complex biogeochemical exchange mechanisms between pollutants in the accumulated basal sediments and the overlying water phase. Proper and regular maintenance, including sediment removal, is therefore essential if the projected quality design limits are to be achieved.

Whilst it may be arguable to conclude that urban flood detention basins may have only a limited pollutant control function, it must be admitted that with proper maintenance they can only help improve receiving stream quality. In addition, they can and do provide substantial intangible aesthetic, as well as occasional direct recreational community benefits despite and irrespective of any adverse water quality conditions.

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USING THE UNDILUTED EVENT MEAN CONCENTRATION TO DETERMINE RUNOFF LOADS

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ABSTRACT

A three year urban runoff study was conducted on Rapid Creek near Rapid City, SD. A new parameter, the undiluted event mean concentration (UEMC), was introduced in order to directly relate water quality changes to the quality of runoff water. Thirty rainfall runoff events were monitored with about 20 data sets being complete enough to allow the calculation of UEMCs.

Six water quality sampling stations were installed to monitor water quality changes in Rapid Creek resulting from urban stormwater runoff in the Rapid City area. Five of the stations were in-stream locations on the creek while the sixth station was located on the drainage channel from a 2,000 acre (809 ha) watershed. All sampling sites were gaged for flow. Precipitation for the sampled events ranged from 0.08" (0.20 cm) to 2.99" (7.59 cm) and the urban discharge ranged from near zero to about 470 acre-feet (579×10^3 m^3). Concentrations of instream constituents varied widely throughout the study, with the majority of the constituents increasing in concentration at the downstream stations.

The UEMC is calculated for a particular event by subtracting out the upstream load. To obtain valid UEMCs for various parameters, careful determination of baseline flow and quality is needed as well as accurate determinations of flow and water quality during the runoff event. A statistical analysis of UEMCs will be presented for ammonia, chloride, COD, total lead, total phosphorus, and suspended solids. Results show that data are positively skewed with an error in the UEMC of about 5 percent.

THE STUDY AREA

General

The study area lies within the physical boundaries of Rapid City, South Dakota. Figure 1 indicates the locations of the six sampling stations. Rapid City is located at the foothills to the Black Hills in western South Dakota and is the county seat for Pennington County. Rapid City is the largest community in western South Dakota and is the business center for much of the surrounding area.

Between 1940 and 1960, Rapid City experienced a dramatic growth rate resulting in a 1960 population of 42,399, up from 13,844 in 1940. A more stable, much slower growth occurred in the next two decades, giving a 1980 census figure of 46,492.

Sampling Locations

The six sampling stations are indicated on Figure 1. Ideally, a sampling station should have been located just upstream of all urban development, and just downstream from the city. Station 1 essentially satisfies the upstream station, but some urban influence will manifest itself because of an increase in density of dwellings and upstream highway traffic. It was not hydraulically possible to establish a sampling station downstream from the study area before other non-point sources of runoff would enter Rapid Creek. Consequently, a theoretical station, No. 56, was developed which mathematically establishes downstream values based on mixing the loads present at Sta. 5 (the last station on Rapid Creek) and Sta. 6 (the last major urban drainage into Rapid Creek).

Drainage areas ranged in size from 33,730 acres (13,650 ha) at Sta. No. 1 to 1,610 acres (650 ha) at Sta. No. 5. Land usage varied from 96 percent non-urban at Sta. No. 1 to 19 percent non-urban at Sta. No. 5. Complete land use information can be obtained from a document by Harms et al. (1983).

TECHNICS

Field Methods

The majority of the water samples were collected using automated equipment but some samples were collected manually, primarily at the upstream stations. Baseline samples were taken at normal flow conditions periodically. Manual methods were used to obtain these samples. Even with the automated equipment in-place, every effort was made by the field crews to be on site during the runoff events.

Manually collected water samples were obtained by wading the streams and collecting depth integrated aliquots using between 10 to 15 verticals across the channel. In the first year of the study, manually collected baseline and precipitation runoff samples were collected directly into new, one-gallon (3.8 L) plastic milk containers. During 1981 and 1982, samples were collected by use of DH-77 depth-integrating samplers. After collection the samples were transferred to new, one-gallon (3.8 L) plastic milk containers. All sampling equipment and

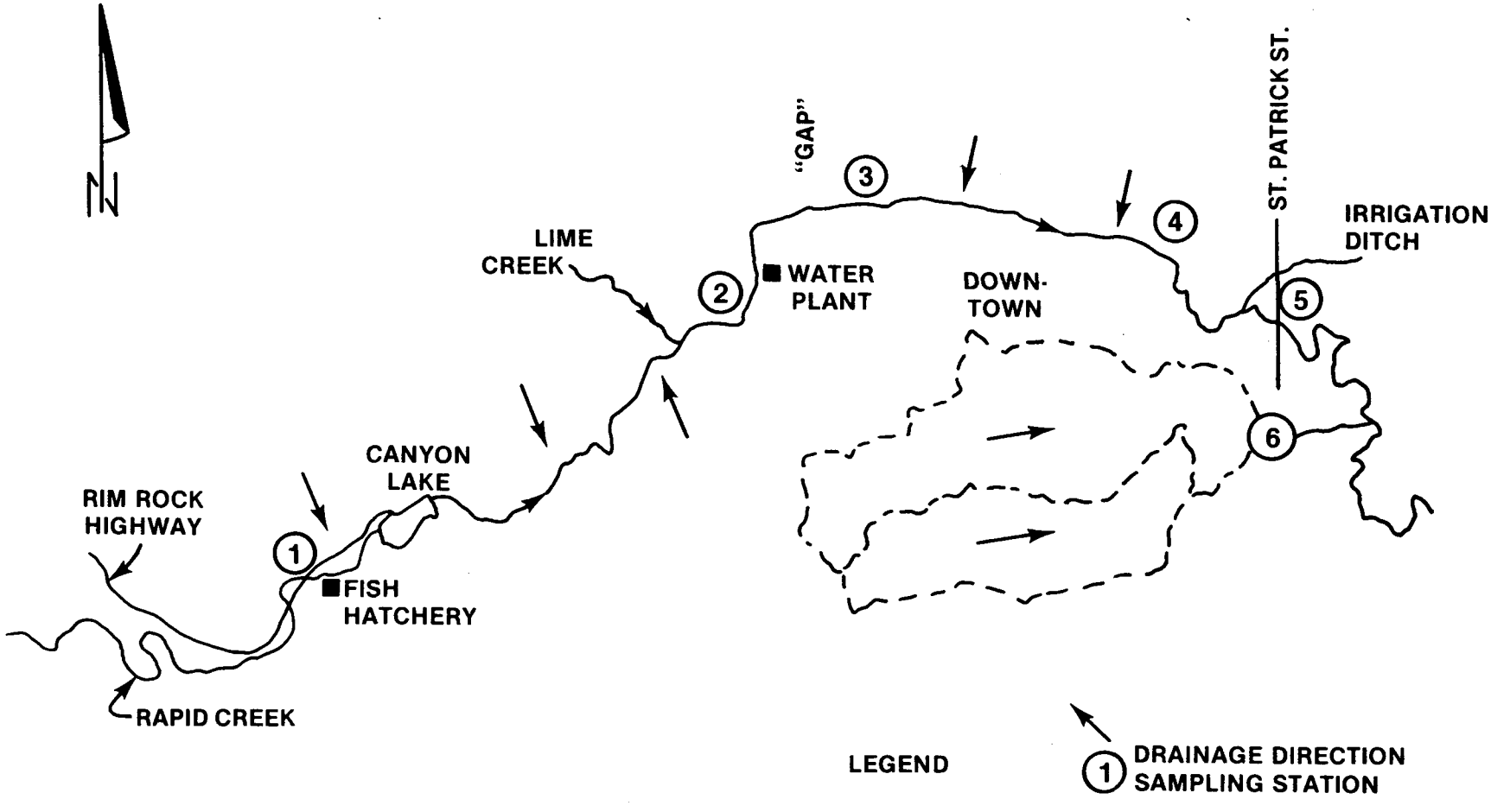

Figure 1. -- Map of Rapid Creek and Major Drainage Basins within Rapid City Proper; Sampling Site Locations.

containers were rinsed three times with native water prior to taking the first aliquot. The sample was immediately cooled to 4°C. Date, time, and stage were recorded immediately before and after sample collection.

The automated sampling equipment, referred to as an urban hydrology monitoring system by USGS, incorporates a microprocessor based system control unit to receive a record on site and to control the automated water sampling device. The system would switch on to storm mode when a select stage, corresponding to a definite discharge, was reached in-stream during a storm event. The system would make continuous recording data (time, date, stage, accumulated rainfall, and sequential sample number) at a pre-determined time interval ranging from 30 seconds to 1 hour. If a rapid rise in stage occurred, the timing sequence was over ridden and additional samples were collected. Samples were pumped into new, one-gallon (3.8 L) plastic containers which were stored in a refrigerator unit, at 4°C, and later, as soon as possible, transported to the South Dakota School of Mines and Technology (SDSM&T) environmental engineering laboratory. Under most conditions, samples were delivered to the lab within three hours of collection. A sketch of this equipment is shown in Figure 2.

Laboratory Methods

All sample preservation and preparation were done in the laboratory. Flow weighted composites were normally made for each station using the appropriate aliquot from each discrete sample. Volumes for each aliquot were determined by calculating the volume under the hydrograph that each sample represented, baseflow included. A computer program was developed to compute the correct aliquots, Harms and Smith (1983).

Composites were produced using a USGS cone splitter, USGS (1980). The splitter splits any sample into 10 equal aliquots to the nearest one percent. The cone splitter was evaluated for accuracy prior to use. After some practice the error averaged about 3 percent which was the sum of the cone splitter and analytical error. After the composite was properly consituted, the cone splitter was used to split the composite sample into individual containers for storage and/or preservation.

A quality assurance (QA) plan was developed and approved by EPA. Standards (both known and unknown), duplicates, blanks, and spikes were routinely analyzed. Values which did not satisfy the QA requirements were not entered into the data base.

UNDILUTED EVENT MEAN CONCENTRATIONS

The undiluted event mean concentration, UEMC, was selected to indicate the quality of runoff which entered the stream. The UEMC was calculated for each parameter from laboratory data for a particular event by subtracting out the portion associated with the baseline flow, as follows:

$$Q_{BL}\ C_{BL} + Q_{RO}\ UEMC = \overline{Q}\ EMC$$

and,

$$UEMC = \frac{\overline{Q}\ EMC - Q_{BL}\ C_{BL}}{Q_{RO}}$$

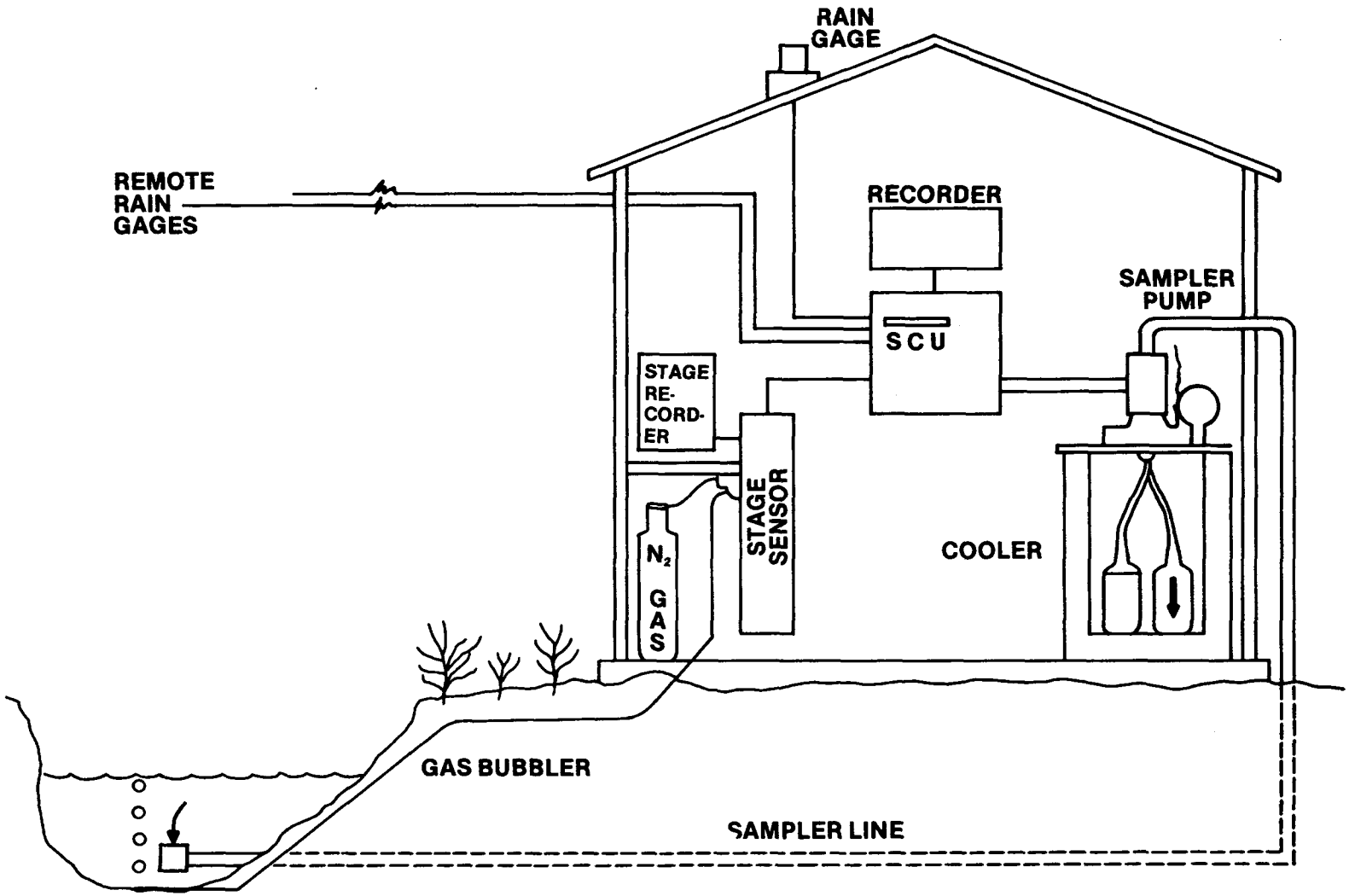

Figure 2. -- Automated sampling station.

where,

Q_{BL} = Base flow preceeding runoff events, cfs

C_{BL} = Assumed base flow concentration for the parameter under consideration. Value is assigned from baseline quality obtained from samples collected when runoff was not affecting the quality, mg/L.

Q_{RO} = Mean runoff flow, equal to in-stream mean flow recorded during a runoff event - base flow ($\overline{Q}$ - Q_{BL}), cfs.

UEMC = Undiluted event mean concentration, a calculated approximation of the mean concentration of a parameter in the runoff, mg/L.

$\overline{Q}$ = Mean in-stream flow during a runoff event, flow weighted; i.e., ($\overline{Q}$)(Runoff duration) = Volume of water passing a station during event, cfs.

EMC = Event mean concentration. Obtained by laboratory testing of a flow weighted composite sample, mg/L.

(Note: cfs x 2.8317×10^{-2} = m^3/s)

A simple example may best illustrate the UEMC:

Given: Station No. 3
Runoff Event No. 5
Base flow = Q_{BL} = 86 cfs (2.44 m^3/s)

Mean in-stream flow = $\overline{Q}$ = 110 cfs (3.15 m^3/s)
Instream suspended solids = EMC = 60 mg/L
Base flow suspended solids = C_{BL} = 8 mg/L

To Find: The concentration of suspended solids in the runoff above Sta. No. 3, UEMC.

$$UEMC = \frac{\overline{Q}\,EMC - Q_{BL}\,C_{BL}}{Q_{RO}} = \frac{(110)60-86(8)}{110-86}$$

= 246 mg/L ans.

The above procedure is not applicable at Sta. 56 (the theoretical downstream location) because the event duration was not exactly the same at Station 5 and Station 6. The UEMCs for Sta. 56 were calculated using the duration recorded at each station, as shown on the following page:

$$UEMC_{56} = \frac{\left|(UEMC)(Vol\ RO)\right|_5 + \left|(UEMC)(Vol\ RO)\right|_6}{Vol\ RO_5 + Vol\ RO_6}$$

where:

UEMC = as described above, mg/L
Vol RO = Volume of runoff (base flow subtracted) at each station, acre-feet (1.2335×10^3 m^3).

Because of the inherent error in sampling and measuring procedures. Some of the calculated UEMC values for Stations 1 through 5 yield a negative value. Obviously this is impossible as a negative 3.5 mg/L of chloride could not be present in the runoff, for example. Statistically, however, these negative values are just as valid as positive values and they were not discarded during subsequent calculations. Generally negative UEMC values were viewed as an indication that that particular constituent was very low in the runoff of the event under consideration.

Frequently, only a minor change in the measured concentration or flow would have produced a positive value. For example, the UEMC for Station 1, Storm 1 shows a -21.5 mg/L for a chloride concentration. If the baseflow used in the UEMC calculation was in error by only 3% (a very acceptable field measuring error), the UEMC would calculate to be a positive 14.2 mg/L. Thus, even though some of the UEMC values compute as negative quantities; they appear to give reasonable concentrations. Approximately 3.7% of the UEMC values for Sta. 1 through 5 yield negative concentrations.

TABLE 1 -- Log Normal mean values for UEMC, mg/L

Sta. No.	NH_3-N	COD	Cl^-	TKN	NO_3	Total P	Diss. P	SS	Tot Res.	VSS
1	0.26	12	5.7	0.56	0.35	0.04	0.01	22	240	3
2	0.12	52	4.8	1.30	0.38	0.25	0.03	289	533	33
3	0.09	91	4.3	1.14	0.35	0.29	0.03	350	552	35
4	0.12	113	5.5	2.42	0.60	0.62	0.06	693	771	81
5	0.11	95	5.8	2.19	0.46	0.55	0.04	595	621	67
6	0.18	190	5.5	3.87	0.52	1.48	0.08	2,443	2,786	228
56	0.15	119	5.9	2.56	0.51	0.80	0.05	1,038	1,161	104

A summary of the log normal mean values for the UEMCs is presented in Table 1. Although mean values can be misleading statistics, the data clearly show an increase above normal runoff in the stream (Sta. No. 1) as urban runoff enters Rapid Creek. When values from Sta. No. 5 are compared with the theoretical concentrations downstream at Sta. No. 56, the impact of the pollutants from the Robbinsdale watershed (Sta. No. 6) are very apparent.

A statistical analysis was done for the UEMC data sets for each parameter. Summaries of the statistics for the computed UEMC values for ammonia and chemical oxygen demand (COD) are presented in Tables No. 2 and 3. Ammonia is repre-

sentative of a dissolved species and shows some negative values while the COD is representative of a parameter associated with solids in suspension and does not show any negative values.

TABLE 2 -- Summary of UEMC for Ammonia-Nitrogen

Parameter	Station 1	2	3	4	5	6	7
Mean, mg/L	0.60	0.36	0.18	0.23	0.22	0.26	0.22
Log mean, mg/L	0.26	0.12	0.09	0.12	0.11	0.18	0.15
Median, mg/L	0.26	0.19	0.16	0.18	0.13	0.16	0.13
Min., mg/L	-0.11	0.02	0.02	0.02	0.02	-0.02	0.26
Max., mg/L	3.80	1.96	0.78	1.32	0.62	1.00	0.61
No. samples	15	18	19	19	21	20	19
No. negative values	2	0	0	0	0	2	0

TABLE 3 -- Summary of UEMC for COD

Parameter	Station 1	2	3	4	5	6	7
Mean, mg/L	53	69	102	137	112	214	135
Log mean, mg/L	12	52	91	113	95	190	119
Median, mg/L	2.5	50	78	106	85	171	100
Min., mg/L	2.5	11	47	33	21	87	51
Max., mg/L	359	276	251	377	339	599	367
No. samples	15	17	19	18	21	21	20
No. negative values	0	0	0	0	0	0	0

Runoff Loads

Rainfall runoff loads for each event were calculated by using the UEMC for each parameter. The UEMC was multiplied by the volume of runoff to give the total load to the stream in pounds (0.454 kg).

Obviously, there is a dramatic increase in material entering Rapid Creek as the stream moves through Rapid City. Table 4 shows these increases as a percentage of the load entering Rapid City (at Sta. No. 1). The higher percentages are associated with those constituents which fluctuate with suspended solids, the most predominate being total lead. Lead was almost entirely tied to particulates, and very little dissolved lead was detected.

TABLE 4 - Increase in Runoff Loads

Parameter	Percent Increase Sta. 1 to Sta. 56	Parameter	Percent Increase Sta. 1 to Sta. 56
Ammonia-Nitrogen	633	Total P	14,000
COD	24,400	Dissolved P	800
Chloride	12,900	Sodium	5,980
Total Kjeldahl N	2,530	Suspended Solids	63,500
Total Lead	104,000	Volatile SS	61,400
Nitrate + Nitrite	2,300	Total Residue	4,730

An important consideration is that these loads are deposited in the stream within a relatively short time-frame. Downstream from the Rapid City urban area is the municipal wastewater treatment facility. Assuming that the facility treats its sewage to normal levels, the runoff loads show that it would take three months for the treatment plant to discharge the same amount of solids that a moderate storm event would cause to be washed into the creek. It would take almost three years for the effluent to contribute solids equal to those deposited in Rapid Creek in one day by a 2.5 inch (6.35 cm) rain (Event No. 21).

CONCLUSION

If used in conjunction with accurate flow measurements and representative water samples, the undiluted event mean concentration is an effective parameter for computing runoff loads.

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WET WEATHER IN-STREAM WATER QUALITY EVALUATION

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ABSTRACT

The magnitude and duration of non-point source discharges are stochastic in nature. For this reason, in-stream water quality standards should be based upon an acceptable frequency of violation rather than an absolute level which should never be exceeded. The effect of various pollution abatement methodologies could then be evaluated based on their ability to meet the prescribed water quality criteria and associated acceptable frequency of violation. This type of approach requires characterization of receiving water responses to a large number of storm events.

A project was conducted for the city of Altoona, Pennsylvania, to quantify the degradation in water quality of the Little Juniata River as a result of combined sewer overflows. Dissolved oxygen was chosen as the principal indicator of in-stream water quality. Runoff flows and associated biochemical oxygen demand (BOD) loadings were calculated for the urban and surrounding rural areas by the Army Corps of Engineers' Storage, Treatment, Overflow, Runoff Model (STORM). A three year period from 1975 through 1977, containing 209 individual storms events, was chosen to characterize the statistical behavior of in-stream dissolved oxygen concentrations.

A one-dimensional, steady state, dissolved oxygen model was utilized to calculate spacial variation of dissolved oxygen concentration. Model input included STORM output (first flush flow and BOD loadings) for each storm event. First flush values were chosen to simulate worst possible conditions. The dissolved oxygen model calculated the critical or minimum dissolved oxygen value occurring within the river study area for each storm. A cumulative frequency distribution of minimum in-stream dissolved oxygen values was then calculated via a simple ranking procedure.

Results indicate that the seasonal operation of CSO facilities may be a viable option, with real economic savings. The impact of supplying various levels of detention storage for the urbanized area of Altoona was also investigated. Diminishing returns (smaller decreases in the frequency of violation) can be expected with increasing levels of detention storage.

Keywords: Urban runoff pollution, nonpoint pollution, nonpoint pollution abatement, stormwater quality, stormwater models, water quality models

INTRODUCTION

Combined sewer overflow (CSO) discharges pose a serious threat to receiving quality in many urban drainage basins. The combined discharge of untreated sanitary "dry weather" flow and urban runoff can introduce significant amounts of pollutants, resulting in severe degradation of receiving water quality.

In 1981, the Pennsylvania Department of Environmental Resources (DER) directed the City Authority of Altoona to analyze the effects of combined sewer overflows in the wastewater management plan being developed for the city. Altoona, a city of approximately 60,000 people, serves as the headwater region for two rivers: the Little Juniata River and the Beaverdam-Frankstown Branch of the Juniata River. The respective watershed areas are 343 and 395 square miles. Water quality in the two rivers is very sensitive to the impact of CSO discharges since the discharges constitute a significant portion of the total flow in upstream reaches of the rivers during periods of heavy rainfall or snowmelt.

This report will focus on the study performed in the Little Juniata River. The study was undertaken to satisfy two principal objectives: 1) to estimate the frequency of violation of dissolved oxygen standards as a result of existing combined sewer overflows; and 2) to evaluate the potential effectiveness of alternative CSO abatement measures. Mathematical models were used to simulate the generation of CSO discharges in Altoona and their effect on in-stream dissolved oxygen concentration. The models were used to evaluate abatement measures such as expanded sewage treatment plant capacity and the utilization of stormwater detention basins. The effectiveness of an abatement measure was expressed in two interrelated ways: 1) by its ability to reduce the biochemical oxygen demand (BOD) loadings attributed to combined sewer overflows; and 2) by its ability to reduce the frequency of violation of established dissolved oxygen standards during wet-weather events.

MODEL FORMULATION

The mathematical modeling effort was divided into two phases: a land phase and an in-stream phase. In the land phase, the Army Corps of Engineers' Storage Treatment, Overflow, Runoff Model (STORM) was used (6). STORM performs a continuous simulation of the rainfall-runoff and snowmelt-runoff processes on the land surface. It calculates the quantity of runoff generated in a drainage basin and the quality of the runoff in terms of 5-day BOD concentrations. The STORM model was modified in two minor areas: (1) to better represent street cleaning operations, changes in the sweeping efficiency function were made; and (2) the closure action of shear-gates at each diversion structure was incorporated in the model. The STORM model is well-documented for its runoff and pollutant loading calculations and no further discussion is provided.

The second model used was the Distributed Oxygen Analysis Program (DOAP), a model developed specifically for this project. In this model, the flow generated by STORM is mixed with the background flow in the river, and the total flow is tracked downstream. The model calculates in-stream dissolved oxygen concentrations along the length of the river using steady-state equations that perform an oxygen balance. The oxygen balance calculations

provide a simulation of the consumption of in-stream oxygen by the bacterial breakdown of organic material present in the combined sewer overflow and other waste inputs.

The oxygen mass balance equation used in the stream model is the well-known steady-state form:

$$D = \frac{L_0 k_1}{k_2 - k_1} [\exp^{jt} - \frac{S_1}{S_2} \exp^{gt}] + D_0 \exp^{gt} \quad (1)$$

where:

$$j = \frac{U^2}{2E} [1 - S_1] \quad (2)$$

$$g = \frac{U^2}{2E} [1 - S_2] \quad (3)$$

$$S_1 = 1 + \frac{4 k_1 E}{U^2} \quad (4)$$

and $$S_2 = 1 + \frac{4 k_2 E}{U^2} \quad (5)$$

The variables present in Eq. (1) to (5) are defined as: D = resultant DO deficit (mg/l); D_0 = initial DO deficit (mg/l); E = longitudinal dispersion coefficient (miles^2/hour); k_1 = deoxygenation coefficient (1/hour); k_2 = reaeration coefficient (1/hour); L_0 = ultimate BOD (mg/l); t = elapsed flow time (hours); and U = flow velocity (miles/hour).

The effects of photosynthesis, benthal oxygen demand, and plant respiration are not included in Eq. (1). It is expected that little photosynthetic activity occurs during runoff events due to cloudy weather and turbid water conditions. Similarly, plant respiration may have an insignificant effect because of the increased volume of flow in the river. The potential effect of benthal oxygen demand on DO concentrations can be quite significant, however. During wet weather, high flow velocities can act to scour sediements that can include large quantities of organic material. This phenomenon and its corresponding sediment oxygen demand are not represented in the present study. There is no evidence that benthal oxygen demand plays a significant role in depleting the DO in the Little Juniata River.

Controlling parameters in the DOAP model are: (1) k_1, the in-stream deoxygenation coefficient; and, (2) k_2, the reaeration coefficient. Because all DO depletion was assumed to take place through biological oxidation, the deoxygenation coefficient, k_1, was a sensitive parameter. It was estimated as a function of flow rate using a formulation by Wright and McDonnell (7):

$$k_1 = \gamma_1 \, Q^{\gamma_2} \tag{6}$$

where k_1 = in-stream deoxygenation coefficient at 20° C (1/day); Q = steady state flow rate (cfs); γ_1 and γ_2 are empirical coefficients to be evaluated by calibration against observed DO data from in-stream measurements during a storm. These were primary parameters in fitting the model to the Juniata River system.

The reaeration coefficient, k_2, was estimated by use of an equation developed by Owens et al. (5) for shallow streams:

$$k_2 = \frac{21.7 \, U^{0.67}}{H^{1.85}} \tag{7}$$

where k_2 = reaeration coefficient at 20°C (1/day); U = mean stream velocity (ft/sec); and H = mean stream depth (ft). Both k_1 and k_2 were adjusted for temperature using standard correction formulas with θ values of 1.068 and 1.024, respectively.

Since many inflows occur along the rivers, the in-stream flow rate, velocity and depth will vary greatly with distance. In the case of a headwater receiving stream, this variation can be several times greater than the base discharge during dry weather periods. Consequently, it is important to represent the spatial distribution of flows and pollutant discharges to the headwater receiving stream. In the present study, this has been accomplished by developing depth discharge and velocity-discharge relations for each reach. With changes in flow rate at each inflow point, the estimates of mean flow depth and velocity are updated. These revised stream values in turn are used to re-calculate k_1, k_2 and the flow time for each sub-reach (distance between inflows). A sub-area definition sketch is shown in Figure 1 for the 343 square mile Little Juniata River portion of the total Juniata River watershed.

FREQUENCY ANALYSIS

The strategy of analyzing the frequency of occurrence of low DO levels to evaluate CSO abatement alternatives has been introduced previously by Medina (3). The DO frequency analysis performed in DOAP is possible due to the model's ability to generate DO profiles for a large number of separate runoff events. One DO value from each event is used in the frequency analysis. This is the minimum in-stream DO concentration present, irrespective of its location in the river. Thus, the magnitude of the DO sag is considered, but its location is not. The minimum DO value from each event is then stored for subsequent frequency analysis. This set of DO minimums is then ordered and

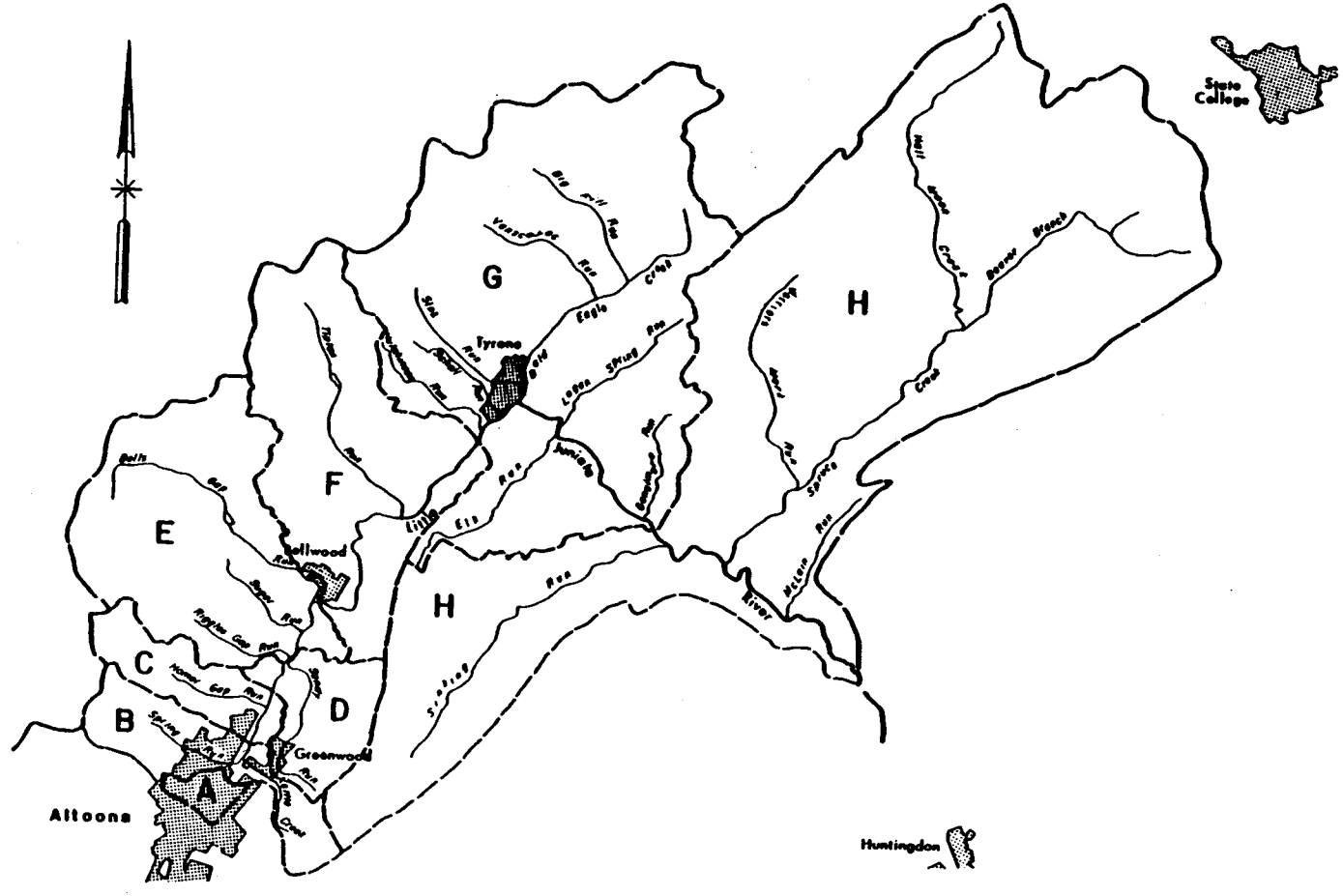


Figure 1: Little Juniata sub-area definition sketch.

assigned a plotting position using the formula: P = 100m/N, where P is the percentage of events exceeding a specified DO level, m is the rank in descending order, and N is the total number of events.

MODEL APPLICATION

The period of analysis was 1975 through 1977 -- a three-year period experiencing above-normal rainfall distribution with the watershed in a condition close to that projected over the life of any CSO abatement project. The definition of an individual "event" is based on the return to zero storage as employed by STORM. The timing of individual "events" is thus dependent on watershed storage and sewer system capacities, as well as on the hourly precipitation record. To reduce the dependence between events, a minimum inter-event time of six hours was used for separation purposes. Using this event definition criteria, STORM showed that 209 CSO events occurred in Subarea A (urban) during the three-year simulation period. The number of events in Subareas B-H (non-urbon) varied from 120 to 183, depending on watershed characteristics. Obviously, larger amounts of precipitation are required to produce runoff in the more highly pervious rural watersheds. Thus, many small storms that cause runoff in Altoona (Subarea A) produce no runoff in other subareas. The first flush or first hour flow rates in the 209 events ranged from 11 cfs to 389 cfs. BOD_5 concentrations in the CSO ranged from 9 mg/l to 302 mg/l. The wet-weather runoff from Subareas B-H varied between zero and several thousand cfs during the simulation period, with typical BOD_5 concentrations of 5 mg/l to 10 mg/l.

Model parameters in STORM were adjusted for runoff volume and pollutant (BOD_5) load by calibration against nine individual storms on three urban subareas where gaging instrumentation was installed. The details of STORM calibration and extension to the Little Juniata system in Figure 1 are presented elsewhere (1,2). The in-stream DO model, DOAP, was then structured to simulate point source loading from each subarea. Base flows were estimated from long-term quarterly averages. These were assumed to be at the same temperature as the wet-weather inflows and also were assumed to be DO saturated. Sewage treatment plant discharges were obtained and introduced in DOAP at their respective locations.

In-stream DO samples collected during wet-weather events were then used to calibrate DOAP. Samples were collected at different times during an event at several accessible locations. During calibration, efforts were made to fit the calculated DO profile to the observed DO data. Calibration was accomplished by adjusting the values of γ_1 and γ_2 in the deoxygenation coefficient equation (6). Since an attempt is made to simulate the worst in-stream condition during each storm event, the DO profile was fitted to the minimum DO concentration measured at each sampling station.

The wet-weather event that occurred on July 27, 1982 was selected for model calibration since the DO data collected on that day clearly illustrated the impact of combined sewer overflows. The best possible fit of the DO sag point was obtained for values of γ_1 and γ_2 set at 14.5 and -0.37, respectively. Several other wet-weather events sampled in 1982 were also used for model verification purposes. Calibration and verification of the DOAP model was successful in terms of matching the magnitude of the dissolved oxygen sag. The DOAP model was not very successful in matching the entire

measured dissolved oxygen profile. Discrepancies of up to 2.0 mg/l were observed at downstream locations.

The discrepancy between the calculated profile and the measured data is caused by several factors. First, due to problems in tracking CSO discharges downstream, the samples collected during an event may not reflect the true minimum DO concentration that occurred at a specific location. Secondly, the results from DOAP depend upon accurate wet-weather flow volumes and BOD loadings as calculated by the STORM model. During calibration of STORM it was observed that the predictions from STORM may not be reliable for any individual event, although the model matched observed values reasonably well over a series of events. Variation in the velocity-discharge relations is another possible source of error. Thus, matching the magnitude of the DO sag is believed to be compatible with the planning-level application of the DOAP model in this study.

RESULTS

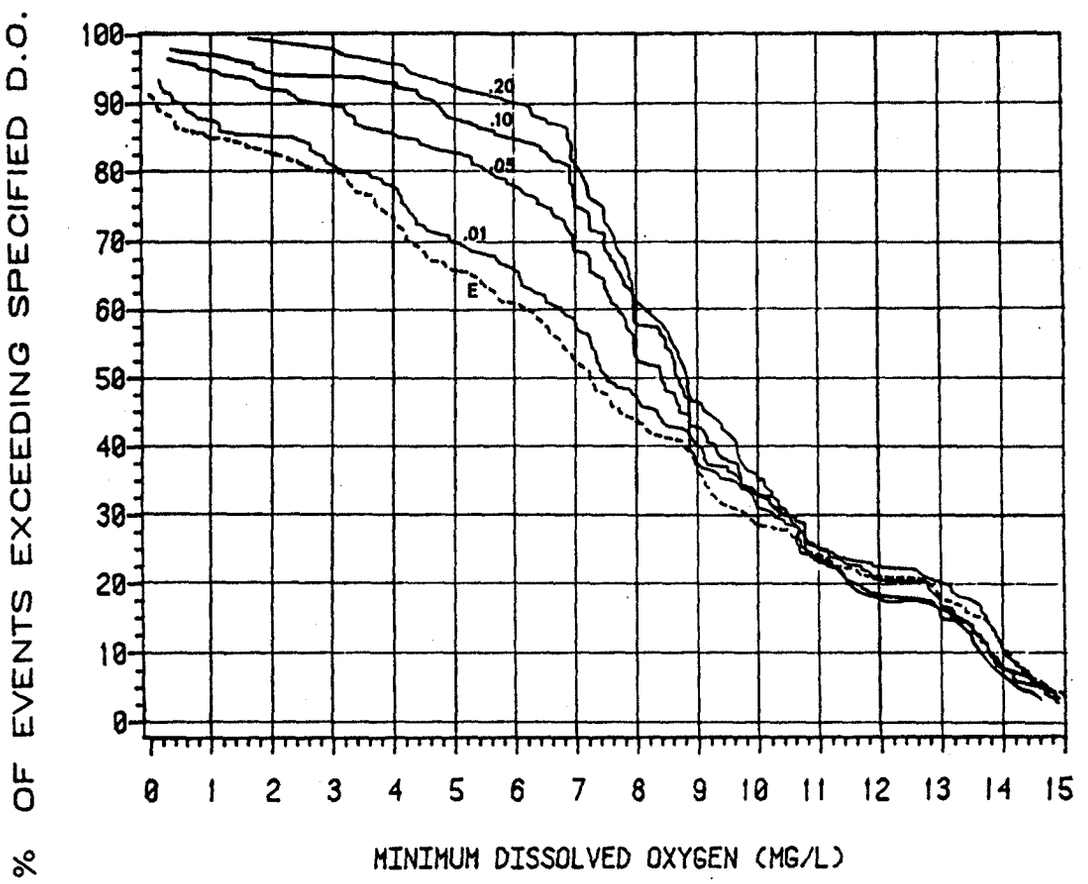
The mathematical models, STORM and DOAP, were used in tandem to simulate the effectiveness of various measures in reducing the water quality impacts of combined sewer overflows on the Little Juniata River. The results are presented in this section for: (1) existing conditions; (2) selected CSO storage treatment alternatives; and, (3) seasonal analysis of DO levels. It should be noted that the results described below are for first-hour flow rates and BOD loadings. This initial hour of each event was adopted as the most severe loading period in an effort to represent first flush impacts on the receiving stream.

Existing Conditions

As a means of establishing a reference or bench-mark for comparison of alternative CSO abatement alternations, existing conditions were analyzed. Results show that under existing conditions (existing diversion, no storage, and a 6.77 MGD wet-weather treatment capacity) combined sewer overflows frequently cause water quality conditions in violoation of the DER dissolved oxygen standard. During the simulation period, in-stream conditions during 65 of the 209 events violated the standard established for the Little Juniata River. The cumulative frequency curve for the 209 events under existing conditions is shown as a dashed line in Figure 2.

Storage Treatment Alternatives

A number of storage/treatment/diversion combinations were selected for analysis with the DOAP model to determine their effectiveness in protecting water quality in the Little Juniata River. Figure 2 shows the effectiveness of different storage capacities with existing flow diversion and existing treatment capacity. Diminishing returns (smaller decreases in the frequency of violation) is shown with increasing levels of detention storage. Similar analyses were completed for modified flow diversion and a doubled treatment capacity. The results from all of the analyses are summarized in Table 1. Results for each storage/treatment/diversion combination are expressed as an annual frequency of violation of the DER dissolved oxygen standard. The



E = Existing Condition

.01 = Storage Capacity (inches)

Figure 2: Minimum dissolved oxygen cumulative frequency curve.

Table 1: Frequency of Violation of DO Standard Under Various Storage/Treatment Conditions, Little Juniata River

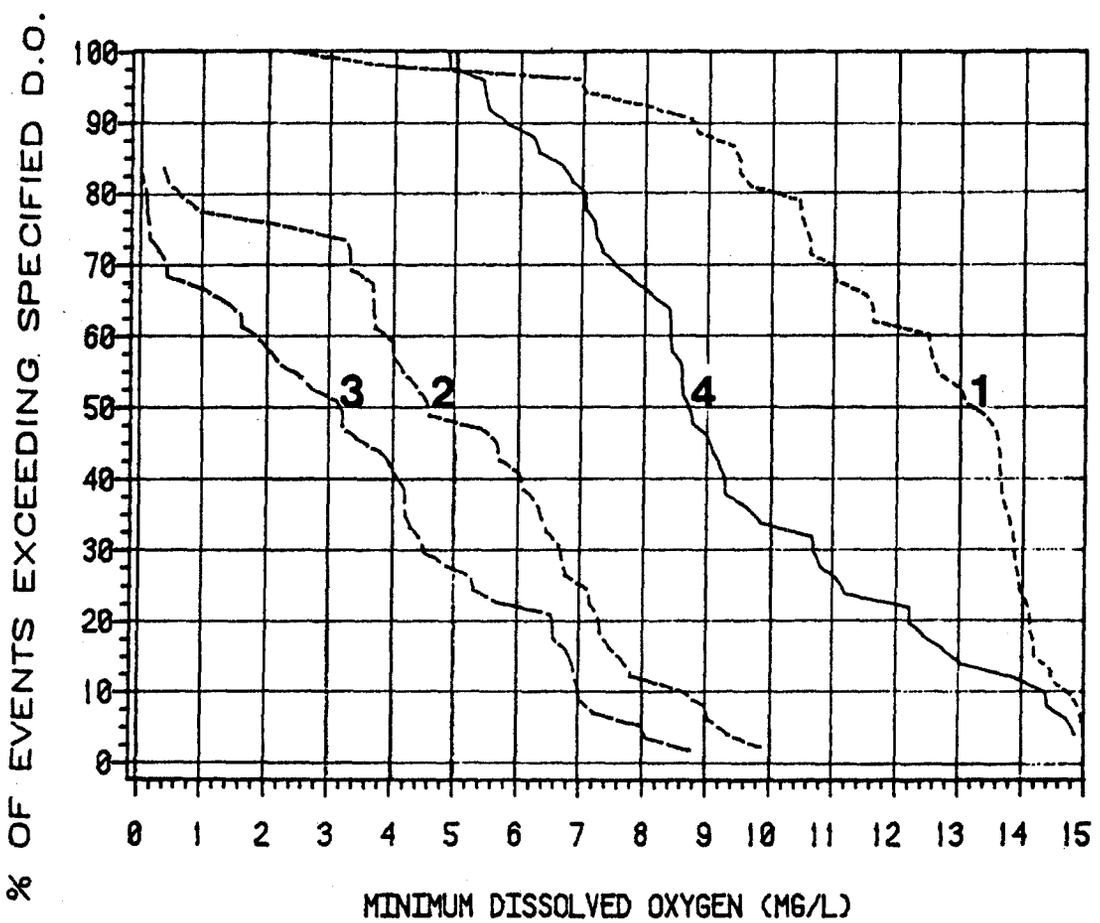
Type of Diversion	Treatment Capacity (MGD)[a]	Storage Treatment (in)[b]	Storage Treatment (MG)[c]	No. of Violations Per Year[d]
Existing	6.77	0.00	0.00	21.7
"	"	0.01	0.29	19.7
"	"	0.05	1.47	11.3
"	"	0.10	2.94	7.0
"	"	0.20	5.87	4.3
Modified	6.77	0.00	0.00	19.0
"	"	0.05	1.47	11.7
"	"	0.10	2.94	7.7
Modified	13.54	0.00	0.00	17.7
"	"	0.05	1.47	9.7
"	"	0.10	2.94	7.3

[a]MGD = million gallons per day; 1.0 gal. = 3.785 liter
[b]1.0 in. = 25.4 mm
[c]MG = million gallons; 1.0 gal. = 3.785 liter
[d]Any DO value below 5.0 mg/l from February 15 to July 31; any DO value below 4.0 mg/l during the remainder of the year.

present frequency of violation, 21.7 events per year, is significantly reduced by the larger CSO storage capacities. Modified flow diversion and expanded treatment capacity are effective in combination with small storage facilities, but do not significantly affect the frequency of violation when storage capacities are large.

Seasonal Effect

In an effort to observe seasonal trends in the minimum DO concentrations, each of the three years in the simulation period was divided into quarters. Results from the seasonal analysis are presented in Figure 3. It is evident that wet-weather water quality conditions are worst during the third quarter



1 = Quarter 1: January 1 - March 31

2 = Quarter 2: April 1 - June 30

3 = Quarter 3: July 1 - September 30

4 = Quarter 4: October 1 - December 31

Figure 3: Seasonal analysis of minimum dissolved oxygen cumulative frequency curve.

(July 1 to September 30). In general, CSO discharges have little effect on DO levels during the first and fourth quarters, although a few violations of the DO standards are observed in the first quarter. The seasonal operation of CSO facilities may provide adequate water quality protection while significantly reducing operating expenditures.

CONCLUSIONS

Several conclusions can be drawn from the results obtained in the study. Combined sewer overflows from Altoona presently introduce substantial amounts of biochemical oxygen demand in the Little Juniata River. During warm weather, the BOD loading is sufficient to cause frequent violations of the established dissolved oxygen standards. Due to the predominantly rural nature of the study area downstream of Altoona, wet-weather inflows dilute the impacts from combined sewer overflows. Combined sewer overflow abatement measures that include storage facilities, expanded wastewater treatment capacity, and modified CSO diversion have been shown by the modeling results to be effective to varying degrees. In general, storage facilities show the most promise in protecting receiving water quality.

The results from this study suggest that the Pennsylvania DER address the issue of modifying its water quality standards to include provisions for wet-weather conditions. Appropriate wet-weather provisions must be based on the significance of the frequency, magnitude, and duration of DO depletion in regard to the health of aquatic life. Meaningful cost-benefit abatement analyses will depend on a better understanding of the significance of low in-stream DO levels during wet-weather periods. This is of critical importance to the selection of alternative CSO abatement facilities.

ACKNOWLEDGMENT

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Potential and actual loading in a small mixed-use belgian basin.

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This paper emphasizes the result of a study of the load of the Dyle River in the center of Belgium. This little basin presents a high population density, and the ground is used for farming, industry and urban purposes. The population is mainly concentrated along the riversides. The first step of the study was a potential load approach, on the basis of two-yearly population, culture and crop, cattle, sheep, pig and poultry census and non point source loading values in the literature. The second one was the work on four automatic sampling stations, on time-mixed sample mode, for one and half year, in the upper basin of the Dyle river. So it has to be constated a sharp contrast between the relatively constant potential loading and the quite variable weekly mixed samples parameters values, emphasizing the preponderant role of the potential loading for the gross average load value and this of the short time meteorological conditions for the variations.

Introduction.

The study of the relationship between pollution potential and actual loads seems to be a greatly difficult question, because of a lot of either less controlled or even unknown parameters implicated in their estimation.The socioeconomic and demographic changes in the given area complicated the approach. A great discrepancy exists in the estimation of the unit potential loads as computed by Cmernik (1976) and Volleweider (1968), who attempted to find a common denominator between the many and diverse values of particular potenial load of grass, cattle, corn and beet field, poultry, industry, depending on the slope and the pedology of the ground, a.s.o., ... A review of the literature over more than 10 years, suggested me that, despite the revealed works of Cmernik and Vollenweider, the particular potential loads census is far from terminated, because the descriptive parameters of the many possible situations (some of them are numbered above) existing for a sole parameter potential effect, as nitrate, are not well systematically defined yet. A second discussion element, is the applicability of the literature data, which are most from an american source, to the european coastal continental theatre, of which Belgium is a part. For this latter scene, Vollenweider showed typical values of potential loads (1968). The american ground use seems to be of relatively the same intensive effect on the soil as the european continent one is. This is despite the particular history of these soils, the difference of climatic intensities, and other extrinsic factors differentially happening on the both sides of the Atlantic Ocean.

We had the opportunity to study a little drainage basin for three semesters in the years 1978-1979, with a small prototype sampling network system, using standard industrial refrigerated samplers, but with a modification, for remote working control by phone lines. So, from week to week, a complete historical data serie has been recorded for approximately one and a half year, at four locations (a, b, c and d on the map figure 1). The data were 2-hour, day and week means (Glazemakers and Dele, 1980, Dele 1984). Independent quantitative and rain data completed the data set. A full work on the pedological, socio-economical and agricultural factors of this basin has been performed, on the basis of statistical data,with the aim to compare the measured loads with the pictures resulting from the census.

The Upper Dyle basin. (figure 1)

The Dyle Basin is a little drainage basin at the east ofBrussels, with three main affluents: the Senne, where Brussels stands, the Demer with Hasselt as main town and the Dyle itself, with Louvain and the quasiconurbation Wavre-Ottignies-

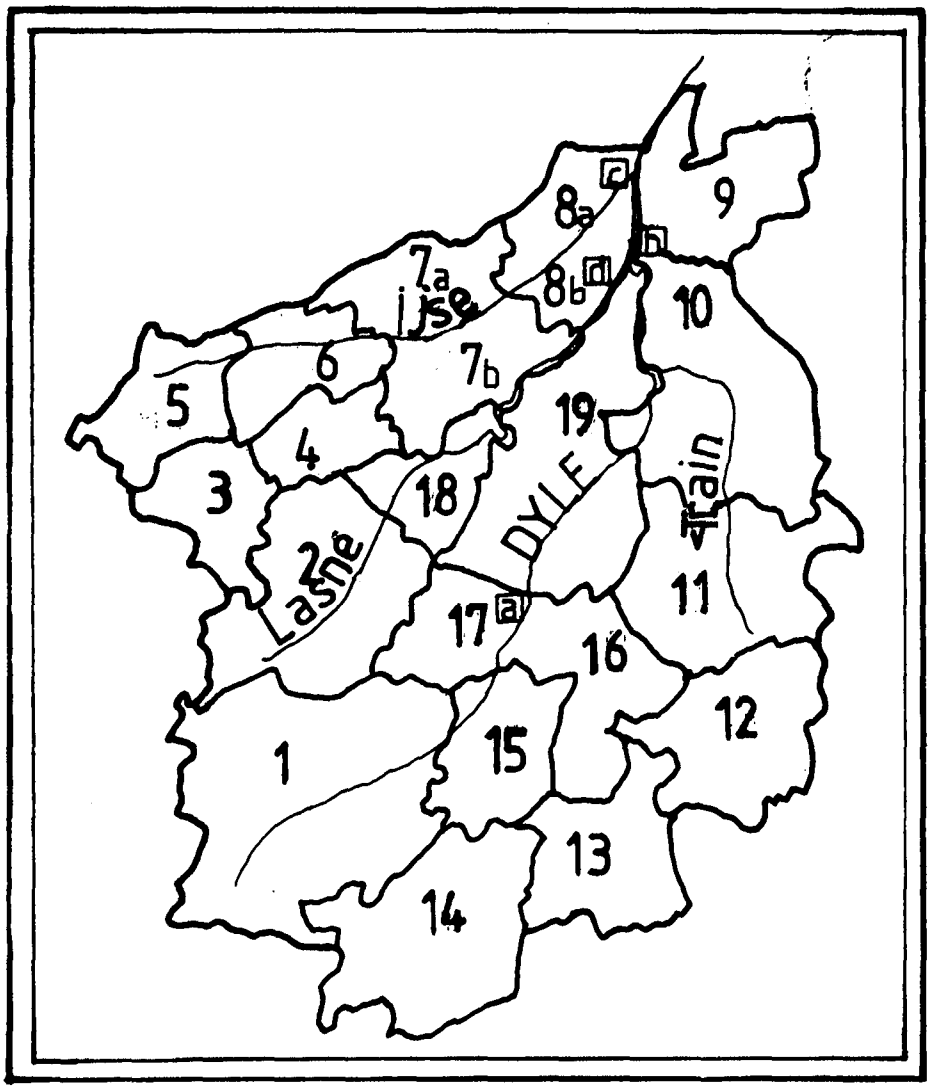

Fig. 1: The Higher Dyle Basin.

1. GENAPPE
2. LASNES
3. WATERLOO
4. LA HULPE
5. SINT-GENESIUS-RODE
6. HOEILAART
7. OVERIJSE
8. BULDENBERG
9. SINT-JORIS-WEERT
10. GREZ DOICEAU
11. CHAUMONT-GISTOUX
12. WALHAIN
13. CHASTRES
14. VILLERS-LA-VILLE
15. COURT-SAINT-ETIENNE
16. MONT SAINT GUIBERT
17. OTTIGNIES-LOUVAIN-LA-NEUVE
18. RIXENSART
19. WAVRE

Louvain-la-Neuve-Court-Saint-Etienne, and then a lot of subsidiaries and municipalities of diverse importances. Its surface is some 2700km2, with which this study concerns 599km2, or 22 %, called Upper Dyle basin, from the spring to the village of Sint-Joris-Weert. The pedology of the upper basin shows the predominance of calcarous sands (table 1), covered with loamy sand, then supporting the culture horizon. The landscape is various, with hills and valleys, and not truly plain surface. Some part of this country is called " Ardennes Braban\onnes ". An estimated 38% of the above mentionned surface is covered with forestry. The basin is inhabited by some 165000 inhabitants, with activities varying along the basin, from predominently farming at the South to equally agricultural and residential at the North of the partial basin. So,the dwelling density differs from 100 in the predominently agricultural zones to 1500 in the mostly residential ones: Rixensart and Waterloo (## 3 and 18 on the map figure 1), for example. The potential population is estimated as 180000 to 200000 inhabitant-equivalents (IE) (Dufour 1975, De Schepper 1976). Some industrial activity exists, but this momentarily figure of some 50000 IE's (De Schepper 1976) is now different from the present one. Some industries are typically bound to farming and then seasonal as sugar industry at the end of each year; some others have now disapppeared (as paper and metal) where they were traditional and new ones are appearing in quite different zones. From this point of view, this area is in constant evolution. The presented pictures are thus valuable for the years 1978 and 1979. The communication network is relatively dense, with any sorts of ways from the truck path to the great runways, as this one from Brussels to Namur (E41) and a dense railway. Thus, in a word, this area is typically a mixed-use basin, with high dwelling and use density, but on a few dozen of kilometers (20 to 30 mi.) along the river axis. As another singularity of this area as other ones in the neighbourhood of great cities, a great number of people travel daily for work to Brussels, Namur, Charleroi or Gembloux so that certain municipalities can be seen as "sleeping zones" of these main attractive towns.The potential load computations don't take into account these daily transhumances.

Material and methods.

The sampling program was started at four points in the area (a,b,c and d on the map figure 1), and performed with standard industrial Buehler PRP/12T type samplers, with standard refrigeration at 4[C and a 12 2.5 liter (2/3 USGal) high density PE bottles distributor.

table 1 Pedogeology on the Upper Dyle Basin underground.

Age	Nature of the formation	Tickness
Pleistocene	Loamy sand (agr.soil)	20m (60ft)
Brusselian	Coarse quartz sand	25m (80ft)
Ypresian	Clayed sand	40m(130ft)
Landenian	Fine glauconiferous sand	70m(250ft)
Masstrichtian	tuff	15m (50ft)
Senonian	marnas	15m (50ft)
Silurian	schists and quartzophyllades	30m(100ft)

Loy W (1978).

For particular technical reasons (Deleu 1981), it was preferred to sample in function of time, in place of in function of flow. For the same reasons, no conservation reagent has been added; only cold at 4[C is used, though it was recommanded by Standard methods (1978) and Belgian standards (1981), among many others.The analysis delay has been of one week after the last sampling. The samples were analyzed according to the methods of the Institute of Hygiene and Epidemiology (IHE 1976) for a total of 16 parameters. From this set, 6 are discussed here, because they seems to be the mostly referred in the li--terature. They are NO3-N, total N, Total P, o-PO4-P, SS and COD. Flow data came from the LGR telemetred hydrological network jointly operated by the former Noyau Administratif de l'Eau and by the Rural Engineering Department of the Faculty of Agronomy,Catholic University of Louvain at Louvain-la-Neuve. (Persoons 1979, Bazier, 1980). The Meteorological data came from the Royal Institute of Meteorology (IRM) (Van Diepenbeek 1980). The data are treated on a Hewlett -Packard HP97 computer (Hewlett-Packard 1977) and on a Tandy - Radio Schack TRS80 model 100 portable computer (Radio-Shack 1983).

For the estimation of the potential loading, it was seeking to different sources, either already written studies over the subject (De Schepper 1976, Laurent 1980, Oovaert 1983), informations graciously given by the Ministry ofEconomicAffairs (1979,1980) and the National Institute of Statistics (1979, 1980) or still, the known composition and quantitites of the main spreaded fertilizers and manures during the asolement period . Thus, there exist numerous approaches of the potential pollution loading estimation and comparison. But because the statistical informations are far more complete and giving 6 months pictures, and so permitting to see the possible variations in the potential loads along 2 years, then the results and discussion about potential pollution mainly will concern these statistical informations.

Results.

Figure 1 is a contour map with the administrative limits of the municipalities in the upper Dyle Basin.The pictures refer to the name listing of these municipalities. The small I.S.O. letters indicate the four sampling locations.

Figure 2 is an analysis of the different 6 month censuses for the population and human activities in the basin (figure 2 a) and for the culture (~~figure 2b~~), and the information is treated according to Bertin (1977). The activities are recensed following the river axis and following the values of

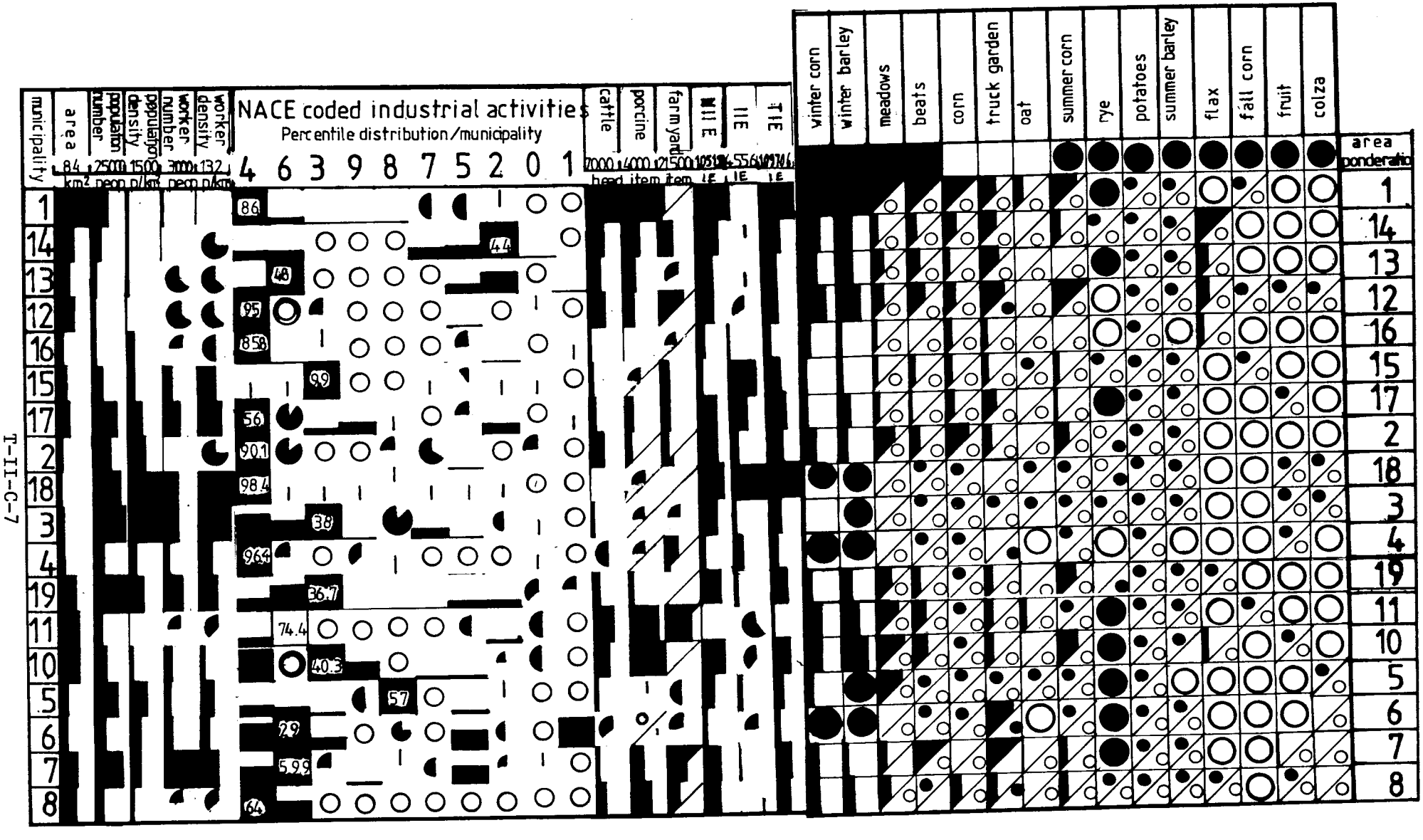

FIGURE 2 : human activities in the Upper Dyle Basin

the pollution load unit conversion coefficient for industrial used waters (Moniteur Belge 1974).

The lisibility of such pictures is somewhat particular. Each subfigure owns its own scale. The full black square expresses 100% value (in which the true value may be either inserted or drawn on the upper side), the other values are proportional to the width of the band. Under 5%, which is pictured with either a sharp line or a blank, the values are expressed in the closest round fraction of a disk, as 7/8, 3/4, 4/8, 1/2, 3/8, 1/4, 1/8,... A fraction of a thenth or less is indicated with a little sharp vertical upper line.A fraction of a twentieth less is drawn with a little sharp vertical lower line. When it is zero, the figure is O. The occasionnally seasonal activity (which is expressed with a 25 % variation between two consecutive 6-month actvities) is shown by partitioning the square figure in two triangles (upper: summer, june, lower: winter, december) with a diagonal,in wich both triangles the above mentionned code is also valuable. The industrial and service activities are distributed according to the NACE code,into categories numbered from 0 (farm and natural activities) to 9 (services) (Gewestelijke Ontwikkelingsmaatschappij voor Vlaamse Brabant 1979). The relative quantities (in percent) for a given NACE category is written in front of the concerned municipality. Contrary to the other comparison items, showing a vertical comparison line between municipalities, along the river axis, the distribution is here horizontal for the same municipality. By this way,the repartition of the human activities and the residential zones becomes quite well seen, with the dynamics of the changes along the river axis and the year.

The industrial activities are also taken into account as potential pollution load on a twin basis according to the NACE distribution on one side, and according to the standardized industrial equivalent-inhabitants (IIE), which is deduced on the legal definition of the equivalent-inhabitant. (Moniteur Belge 1974). The definition of the IE has been discussed elsewhere (Deleu 1983, Vanderborght 1981). Unfortunately, the lawmaker didn't use the NACE industrial activity distribution code and considered a little amount of the classical industry, with an arbitrary insistance over unattended details.The distribution of the recensed activities is then dependent for a given part to the subjectivity of the user.But the numerous chosen NACE categories are sufficient enough to minimize the possible estimation errors. The resulting IE are converted to potential pollution loads according to the I E legal definition. It becomes then apparent that such detailed approach of the potential polluting EH, even if misestimated for the abovementioned reasons, is much higher than the preceeding estimations (Dufour 1975, De Schepper 1976, Oovaere 1981).

Thus, owing to the loss of data due to non recensed ground u-

ses (ways, aso.,...) and the uncertainty of some of the industrial activities, the potential pollution estimates must be considered as partial ones, even if this part represents the great majority of the true estimation.

Figure 3 shows the variations of potential (a) computed from the literature minimum values, b) computed from the literature maximum values, c)computed from the non industrial equivalent living population, including the recensed cattle, d)computed from the industrial equivalent population, as recensed following the NACE code) and actual 6 months loadings (e) at the four sampling sites and for the six analyzed parameters. The values are expressed in metric ton/6 months. The lecture code is the same as for figure 2. Here, the seasonal activity is visible by itself, because the shown values are 6-monthly ones.

Figure 3 presents data corrected with respect to the length of a semester in weeks, e.g.26 weeks/half-year, and converted from kg/s to metric tons/6 months, with a time conversion coefficient from 86400 second a day to week of 7 day and a semester of 26 weeks. For each partial basin,the picture is the expression of a specific income load in the river for the concerned parameter over 6 months.

Figure 4 presents a distribution of the actual loads between the runoff (white area), the hypodermic (delayed) discharge (hatched area) and the ground discharge (black area).
The picture is drawn on the same scheme as the preceding one. The symbols are the same as above excepted one more : a squared one indicates insufficient data.When a particular discharge is estimated equal to zero, the figure shows a O. If the discharge value is under 5%, a as above fractionned disk is drawn over the zone,with the same drawing code of the distribution.The square hatched zone of the disk is without meaning.
The values are expressed with respect to the percentile distribution; 100 % is the value of each squared subfigure.

Discussion.

It is first worth saying that every measured values considered here are mean values, not point values. The total grab sampling number is within the 36000/year.station. The results and discussion are dramatically affected with such data collection because the majority of the past discussions are re-

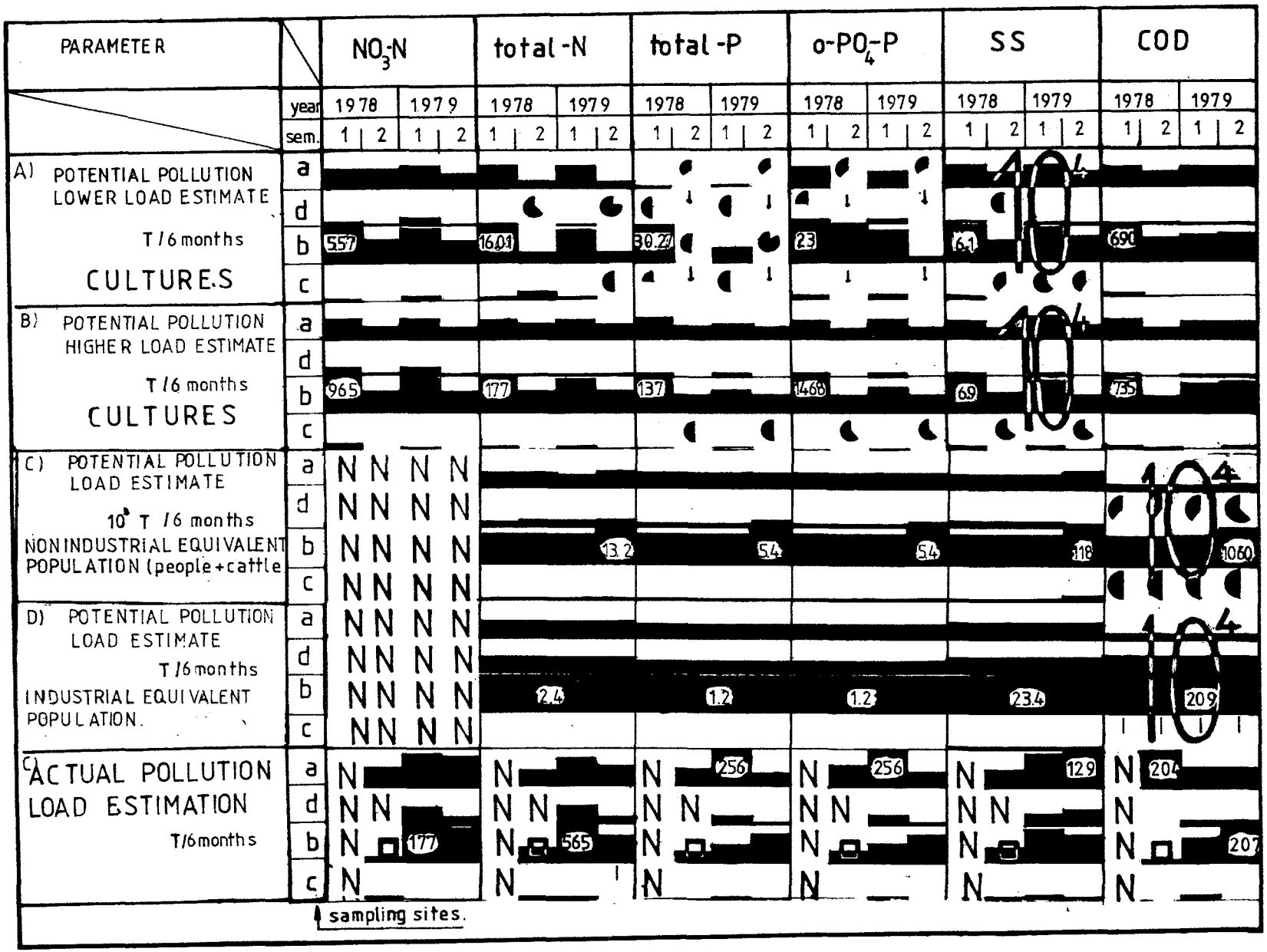

FIGURE 3: potential and actual loads in the Dyle Basin

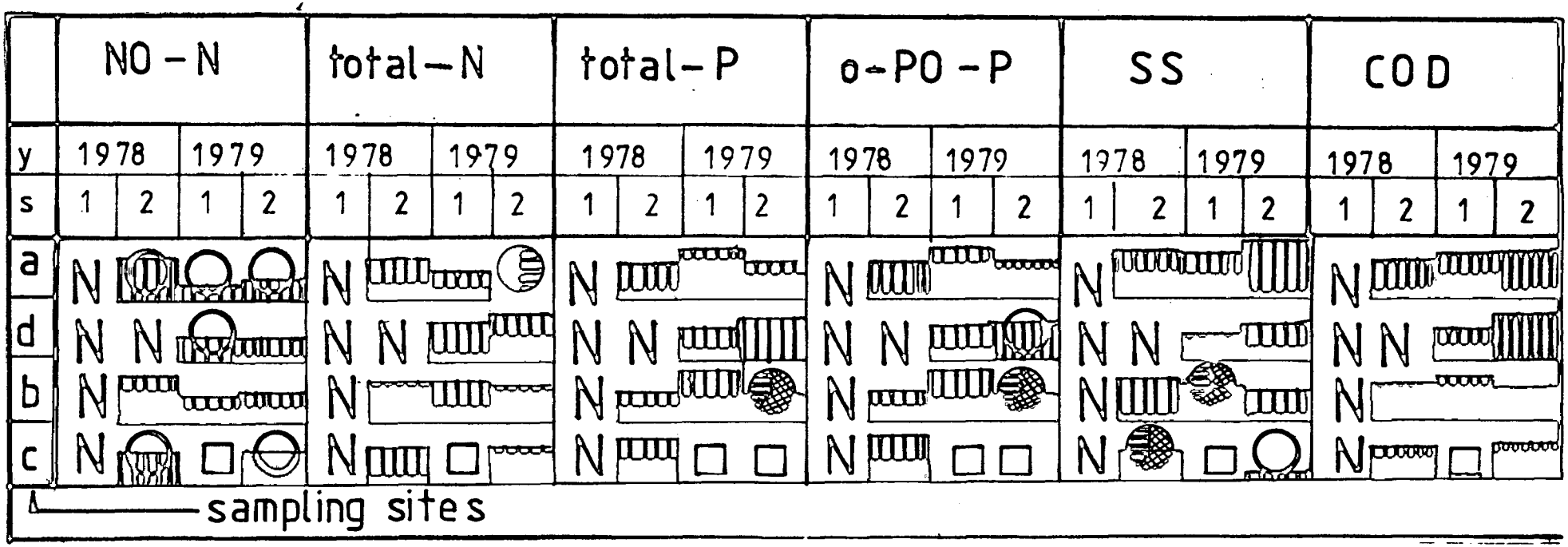

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FIGURE 4: percentile distribution of actual loads between ground discharge, hypodermical discharge and runoff

ferring to grab samples, which are taken at different time frequencies, or with a more or less complex relation with the flow.

On the other hand, by the census recensed surfaces are lower than the above cited picture of 599 km2. It remains about 85 km2 (14 %) out of the census, which are taken back as not recensed, for example some routes and forestry.
The pathways and the forestry are taken into account by themselves for the actual load computation, but in the potential load estimation not, excepted some forestry, as rural ones (National Institute for Statistics 1979, 1980).

On the figure 3, the effect of the variations from a semester to the other is remarkable, for the six concerned parameter, even in the case of the actual pollution loads. The alternance in the loads is quite apparent, giving an impression of predominance of the seasonal activities on the otherones.This same variation character of both potential and actual loads suggests that the potential pollution pressure also ownsfractal properties, as the actual ones have (Pearsons coefficient values up to more than 150 %). By the way, it was shown else where (Dele 1984), that the quality variations follow the laws of fractal curves, or the Mandelbrot theory (Mandelbrot 1975) from which the most remarkable property is the conservation of the standard deviation variations at every scale these variations are examined. The load pressure was assumed constant because the normal way of life of the populations and the indefatiguable industrial activity. Actually, the potential loading is quite more dynamic and variable than the a priori computations can show. Thus, it seems to be worth working the calculations at smaller scales than the abovementioned 6 months, for greater informational precision.

The second remarkable element is, shown in the figure 3, the great difference between potential (figure 3 a minimum, 3 b aximum, 3c non industrial EH, and 3d industrial EH) and actual (figure 3e) loads estimations, in the order of 1000 to 1.
The import from population and livestock appears much more important than the culture import. The actual load appears closer of the higher culture load estimations and a few tenths of percents of the population and livestock E H and its legal conversion.
Furthermore, the potential estimation is but partial,because not including the non recensed sources,though the actual estimation concerns everything passing through the Dyle. Logically, both are estimated with the same amount of population and ground use, minus what wasn't recensed in the potential estimate.
The most frequent (more than 95 % in frequency) polluting load import is concommittant with occurrence of rain,superficially appearing to have the central role in load transport and then which could explain the alternance from a semester to the another. But the few percents (5 to 30 %) of the

actually transported loads by net rains in the case of some parameters,forbit to consider its role as of the most important.In other words, the role of the rain seems less being the cause of transportation than its throttle. In non raining conditions, the ground discharge withdraws the greatest part (from 80 up to 100 %) of the total charge coming up to the river, despite of the lower flow, with stagnation and sedimentation in the waste pipes. When raining,the runoff withdraws a great part (from 30 up to 70 %) of the total charge and the resulting sedimented loads are pushed down with the runoff and then become active in the river water, giving the well known pollution peaks. But, an analysis of the base line of these events shows a great contribution of the hypodermic discharge to the polluting load (up to 80 %), with a time delay of about one week.The remaining of the charge (up to 75%) is taken in charge with the ground discharge and constitutes the basis of the loading. Sometimes, for nitrate-N,the ground discharge does not exist.

The effect of a seasonal activity on the water quality as the yearly sugar campaign (Muller 1966), may be quite expressive and dramatically change the parameter values even for arelatively long time length: two months,e.g.one third of semester (Glazemakers and Deleu, 1980). But as it could be seen on the figure 3e, this local effect remains invisible in the whole load mass, despite the time rate.

The dramatic difference between the measured charges and the potential load computation results will not lead to conclude that the earlier work must be repeated. It evently could be computed according to the unit values of Vollenweider (1968), but the parametrization of these unit values is not known yet for the Upper Dyle Basin. This difference overall underlines a great difference in the approaches in measuring and in computing.

On the other side,it could also be the sign of a great retention potency of the polluting load in the basin. This retention could be either physical (long time sedimentation, atmosheric elimination of NH3 (Loehr 1974) for example,) or depending on a long delay discharge, or degradative (natural purification), or still the three phenomena together as Benoit (1972) showed in the case of natural waters. The form of the purification processes could be more general and complex than assumed by Benoit, it is to say that they occur not only in the watercourses. Then, at the next step, the measured load is only a small fraction of complex origin of the initial load computed as potential load. Further, it appears that this purification process is of quite high efficiency, on a time delay on only a few days in the river for the runoff (Bazier 1981), one week for the hypodermic discharge. On quantitative basis, the known low values of runoff coefficient (.1 to .32 following De Pelsmaker (1980), .01 to .15 following our own less rigourous estimations) are assumed to

fill a part of the process conditions. In river water,the intensity of these degradation for NH4 is high (80 to 95 % oxidation)(Curtis, Durrant and Harman 1975). Joining these two values, 0.01 for the runoff coefficient and .05 for the degradation, one can estimate the remaining NH4-N load to the .05 % of the initial potential load.This is within the actual load of the NH4-N relative parameter NO3-N.The final explanation lies thus in a combination of degradation and quantity. It could maybe remain elsewhere,but with greater uncertainty.

Conclusions.

The above presented results indicate an unexpected great difference between potential and actual loads.This great difference between room calculations and field measurements indicates a retention potency of the polluting charge along the time and resulting in a quite efficient loss of load. A begin of complete explanation can be presented in a combination of degradation chemistry and hydrology.

The alternance of the results from 6 to 6 months, indicates a seasonal effect on both potential and actual pollution loads showing the possible short term variation of the potential load like the actual load variations.

During this study, the use of high density data, as automatic permanent samplers can collect, appear quite potent in the approach of long term complex water quality phenomena.

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MODELING SEDIMENT DELIVERY FROM FIELDS TO CHANNELS: A CASE FOR FIELD SCALE DATA AND HIERARCHAL DATA STRUCTURES

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ABSTRACT

There is a broadscale awareness of and concern regarding the water quality problems caused by nonpoint source pollution. There is also a growing committment to abating this type of pollution. In order to direct nonpoint pollution control activities in the most cost-effective manner, it is necessary to understand the erosion, sedimentation and sediment transport processes in a watershed. This understanding is best developed through the use of a mathematical model.

There are several watershed models available today that can be used to evaluate erosion, sedimentation and sediment transport. They vary considerably in their input requirements and output abilities. These models generally require that a watershed be subdivided into many small, square, cells. The unit of data is a grid cell. Data for the model is then collected, entered into the computer, and analyzed on a cell by cell basis. From the modeling experiences associated with the Wisconsin Nonpoint Source Program, it appears that this grid cell technique is not the best way to divide, collect and analyze watershed data.

An alternative modeling approach was subsequently defined and is presented. This approach incorporates field scale data and hierarchal data structures. Field scale data means that the unit of data is the farm field. Data is collected on a field by field basis. The use of field scale data allows for more homogeneous cell attributes and more applicable field analysis and management plans.

The use of field scale data also necessitates the use of hierarchal data structures. Hierarchal data structure refers to the manner with which data is accessed and processed in a computer program. Hierarchal data structure indicates a data storage technique by which all cells are accessed only by means of their relationship to other cells. The use of hierarchal data structures parallels the downslope movement and aggregation of runoff, and is the most appropriate data structure for the process being modeled. Most significantly, the use of field scale data and hierarchal data structures enables a model to be used to estimate the delivery of eroded sediment from a field to a channel.

Key Words: sediment delivery, sediment yield, delivery ratios, sediment trapping, nonpoint pollution.

INTRODUCTION

The Wisconsin Fund Nonpoint Source Program was initiated in 1978 to identify and rectify nonpoint pollution problems in the state. This program, with a current annual budget of $5,600,000, provides state cost-share dollars for nonpoint controls in targeted areas of designated Priority Watersheds. The Nonpoint Source Section of the Department of Natural Resources ("the Department") has the responsibility for developing a watershed plan for each Priority Watershed. Each plan includes an identification of the nonpoint pollution problems in the watershed, the water quality objectives, the critical sources (eligible areas) of nonpoint pollution, and the applicable best management practices. Each year there are four to six new watersheds added to the program, each of which is usually 250 to 750 square kilometers (100 to 300 square miles) in area. The analytical tools used in the identification of critical areas, the selection of remedial measures, and the estimation of resultant water qualtiy impacts, must be appicable to large areas in a timely and cost-effective manner.

Extensive land use iventories are conducted on each watershed. Currently, all fields within a quarter-mile of a channel are inventoried for location, landowner, soils, topography, land cover, rotation and other management practices. These inventories are conducted on a field by field basis by local county agencies under contract with the Department. An average annual rate of erosion is derived for each field using the Universal Soil Loss Equation (USLE). In the most recent round of watershed plans, one of the watersheds had over 8500 fields for which the USLE was calculated. Following these soil loss calculations, the watershed planner, in conjunction with the local agencies, selects a target soil loss rate and a prioritized list of best management practices applicable to the watershed. The computer then "applies" these practices one at a time to each field with a soil loss rate above the target level, recalulating the soil loss rate after application of each practice, until the rate is either below the target level or, infrequently, all practices have been applied. The fields are then aggregated by landowner. Landowners are in turn prioritized for eligibility in the program based upon the anticipated total reduction in soil loss.

This method has several advantages: The data base is similar to that used by several county and state resource management agencies. This sometimes allows for shared data and reduced data gathering time. It enables broadscale data collection to be carried out over many large areas simultaneously and requires no sophisticated data gathering tools or precise geolocator techniques. It also gives the local agencies a significant role in the planning process, which gives them a large measure of "ownership" in the plan. This method has two strong disadvantages however. It does not distinguish between differences in sediment delivery between fields, nor is it capable of estimating watershed sediment yield. Rather it treats all inventoried fields identically, regardless of the ultimate delivery of eroded sediment from each field to receiving waters.

Consequently, it does not distribute limited money for nonpoint source controls in the most cost-effective manner. In lieu of a better tool, it relegates this phase of the program to a function of soil erosion control rather than water quality control.

Recognizing this, in 1983 the Department procured a Federal grant to incorporate the ANSWERS model into it's planning and evaluation activities. ANSWERS is a distributed parameter hydrologic and sediment detachment and routing model, intended for primarily agricultural watersheds (Beasley and Huggins, 1982). A distributed parameter model, unlike a lumped parameter model, divides a watershed into many discrete cells. ANSWERS, like several other distributed models (MODANSW [Park and Mitchell, 1983] and AGNPS [BOSCH et al,1983]) is a grid based model. A square grid is placed over the watershed, and a row and column locator is assigned to each cell. For the ANSWERS model, the location (row and column), slope, direction of slope, soil type, land cover and channel class (if any) of each cell must be designated. Several variables are required for each soil type, including total porosity, field capacity, infiltration descriptors, antecedent soil moisture, and USLE "K" value. Several more variables are required for each land cover, including rainfall interception capacity, surface cover, surface shape descriptors, Manning's N, and USLE "C" and "P" values. Several of these variables fluctuate seasonally, requiring several data bases for different crop stages.

ANSWERS, like other distributed models, is event specific, i.e., it models runoff and sediment concentrations for single events. It also operates in a stepped time mode, i.e., under event simulation, the conditions of each cell are calculated and updated every few minutes for the duration of the event. Because of its distibuted nature and stepped time mode, ANSWERS requires large amounts of data and is computationally intensive for large watersheds.

The primary advantages of a distributed model like ANSWERS is that it can estimate net sediment erosion/deposition on each cell, total watershed sediment loadings, and the overall impact of changing land management on various cells on sediment loadings. However, there are two drawbacks to a distributed model like ANSWERS that, indirectly, are the subject of the rest of this paper. The first is the difficulty and inappropriateness of collecting and processing land based data, especially for large areas, in a grid based format. The second is the inability of a computer internal grid based data structure to estimate the eventual deposition or delivery of sediment from individual fields to receiving waters (as opposed to the net sediment movement on each cell). In response to these drawbacks and to the needs of the Wisconsin Nonpoint Source Program, an outline of a proposed sediment delivery model, incorporating field scale data and hierarchal data structures, is presented.

DISCUSSION

Field scale data

The data format that has most frequently been used in watershed modeling of sediment erosion and delivery has been the grid cell approach. Using this method, a grid is placed over a map of the area being modeled, which is then divided into numerous identically sized cells. The cells are thereafter identified by row and column numbers and they become the unit by which data is collected, entered into the computer, and processed. The utility of the grid cell approach can be examined in light of its attributes relative to those of a field scale data approach.

To its credit, the grid cell method enables an easy locator to be associated with each cell, i.e., any given row/column designation can be readily located on a map. Therefore, the input or output of the model can be easily graphed and visually displayed. Further, in the case of ANSWERS and MODANSW, it allows for a more accurate determination of runoff direction from each cell, i.e., the direction of flow is highly variable and can be proportioned between two adjoining cells. In the case of AGNPS, the direction of flow must be designated to one of four directly adjacent cells or four diagonally adjacent cells. Lastly, because of the rigid and simple relational locators, it makes programming and modifying grid based models a relatively simple task.

There are of course a few major drawbacks in using the grid cell approach. One relates to the size and homogeneity of the cells. The size of the cells should be a function of the diversity and the size of the area being modeled. The smaller the cells, the more homogeneous their contents, the greater their number.

Tradeoffs will often have to be made, sacrificing the homogeneity of smaller cells in favor of the practicality of larger ones. In the 727 hectare watershed that the Department calibrated and evaluated the ANSWERS model on, a cell size of one hectare was employed. Using this same cell size on the Priority Watersheds would require generally 25,000 to 75,000 cells for each watershed, obviously a prohibitive number on any account. Using larger cells, perhaps ten hectares, would alleviate that problem but would result in another one. In much of Wisconsin's diverse topography, ten hectare cells in a grid system would yield very high diversity in the many cells that encompased two or more very diverse land covers, slopes, soils, etc.. The authors of the AGNPS model (Bosch et al, 1983) suggest using a four hectare (ten acre) cell size for watersheds less than 800 hectares (three square miles), and 16 hectare (40 acre) cell size for larger watersheds. Given the diversity found in most of Wisconsin's landscape, cells of the latter size would prove far too heterogeneous for accurate representation of their contents.

In a recent related study by the author, the soil diversity of a 1024 hectare area of Dane County, Wisconsin was examined. The soils of the area had previously been digitized, and this analysis was performed by computer. A grid matrix composed of one hectare cells was superimposed over the area, and

the number of different soil classifications in each cell was counted. Only 153 cells had just one soil classification within them. Of the other 871 cells, 447, 313, 79, 28, and 4 cells contained, respectively, 2, 3, 4, 5, and 6 different soil classifications.

A second problem related to using grid cell data involves making field management decisions based on grid cell model outputs. The entire purpose of running a model is to make field management decisions related to water quality objectives. Yet how are management decisions made when there are cells straddling more than one field, or multiple cells in a field indicating divergent net soil movement? Field management is not typically based on or applied to a small portion of a field, but rather it is based on and applied to the entire field. Grid cells are less than ideal for making field scale management decisions.

A third formidable problem of the grid cell method relates to referencing the landscape to the grid cell pattern that was established. While it is a simple though tedious task to overlay a grid on a topographic map and determine elevations, slopes and flow directions of cells, it is not a simple task to accurately locate grid cell boundaries on unrectified air photos, soil maps, or worse yet, actual landscapes. Though the problem is not insurmountable, it presents significant difficulty in application.

There are alternatives to using a grid cell pattern. One very attractive alternative is using actual field boundaries as the basis for or unit of data collection and processing. The unit of data becomes the farm field. Data collected at this scale is, not surprisingly, termed field scale data. For these purposes, a field may be defined as an area of homogeneous land cover, managed as a unit by the same individual(s), without significant topographic or soil differences within it. There are several attributes which make field scale data collection and processing attractive.

The primary advantage of using field scale data is that it involves readily identifiable, relatively homogeneous, management oriented data units. There is little ambiguity about the boundaries of a field. They can easily be identified on air photos, soil surveys, and "in the field". Most significantly, land cover and land management are homogenous by definition. Further, field boundaries are often laid out following the contours and channels of topography and drainage and, as such, they tend to have more homogeneous slopes and soils then similar size grid cells placed on the landscape irrespective of naturally occuring features. Where fields do have significant topographic or soil differences within them, it may be in both the modeler's as well as the land owner's interest to subdivide them. Field size also tends to follow the diversity of the landscape. Where slopes are uniform, fields tend to be larger, and vice-versa. This relation coincides with the inherent preferences of the modeler, i.e., smaller scale data where needed and larger scale data where warranted. And of course, by definition, field scale data is most appropriate for making field management decisions. The land operator is already operating at this scale. Lastly, this is a common unit of data gathering, and several other agencies and organizations

collect this type of data in a similar format. This common data requirement and existance of common data files provides for the opportunity of shared use of common resources.

The use of field scale data has many appealing attributes but, as we shall see, it necessitates the use of hierarchal data structures.

Hierarchal data structures

The term data structures refers to the method by which data is organized and processed within a computer model. There are two types of data structures to be considered here: array and hierarchal.

An array data structure is (in this application) a two dimensional representation of data with X-Y or row-column coordinates. The array parallels the grid cell data format, (although a grid cell format, within which runoff is passed to one adjoining cell rather than two, may as well be represented in the computer by an hierarchal data structure). The array is processed sequentially, proceeding row by row, column by column, through each iteration. Runoff and sediment is routed out of each cell to adjoining cells.

There are two problems with array processing of runoff and sediment routing data. First and foremost, array processing makes tracking of sediment eroded out of one cell impractical to separate and track independently as it moves downslope and merges with sediment from other cells. Instead, the net sediment movement on each cell is tracked (deposition plus incoming sediment minus detachment minus outgoing sediment). A simple tally of net increase or decrease in sediment on each cell is kept, not the eventual deposition or delivery of sediment from each cell.

The second problem is one of efficiency or waste of computer resources. The size of arrays must be declared (allocated) before the computer program can run. Either the arrays must be declared large enough to accomodate the greatest anticipated cell numbers (how big is the watershed and how small are the cells?), or the program must be modified repeatedly to fit the individual watersheds, or the watershed must be subdivided to fit the program. Recall also that there are several parameters for each cell (slope, land cover, soil, flow direction, etc.), and that this problem is compounded by the number of parameters. Wasted allocation can rapidly limit the size of a watershed which a particular computer can accomodate. ANSWERS cleverly transforms its data into several one dimensional arrays, which alleviates the wasted allocation which results from watersheds not being square, i.e., not completely filling the corners of two dimensional arrays.

An alternative to the array data structure is the hierarchal data structure. Hierarchal data structure is a method of data storage and access by which each cell is accessed by its location relative to other cells. It does not have such absolute designations such as "row and column" or defined locations in an array. Instead, each cell is accessed only through access to a previous cell. The location in computer memory of each record is kept in and a part of another computer record.

The use of field scale data necessitates the use of hierarchal data structures, because the fields obviously are not uniformly rectalinear. Fields have a variety of shapes and sizes. They do not conform to a grid pattern either on the landscape or in the computer. Rather than being accessed by row and column in the computer, they are accessed by their position relative to other fields.

There are a couple of drawbacks to using hierarchal data structures however. Probably the most significant disadvantage is that it requires that all runoff from a field flow to a single adjoining downslope field or channel. Without some very complicated programing, which would in execution severly hinder model performance, it is impossible to split the flow of runoff from one field to two or more adjoining fields or channels. In designating a flow route, this requires making a selection based on dominant flow, worst case conditions, or some other criteria. A second less critical drawback is that field attributes (the input and output of the model) can not readily be mapped by the computer.

While the use of hierarchal data structures has some drawbacks, there are several advantages of this type of data structure that makes it the prefered choice in watershed modeling. To understand why this is so requires a more in-depth look at hierarchal data structures.

Hierarchal data structure is perhaps more commonly refered to as a tree structure, because the internal representation of the data approximates the structure of a tree (Figure 1.). A cell which has other cells flowing into it is called a root. Each root has one or more cells flowing into it called branches, each of which in turn may have one or more branches flowing into it. Each cell is a branch of the root to which it flows, and the root of the branches which flow into it. All of the cells that flow either directly or indirectly into a root cell are called daughter cells of that root. In Figure 1., cells B, G and H are branches of cell A, and cell A is the root of cells B, G and H. All cells other than A are daughter cells of cell A.

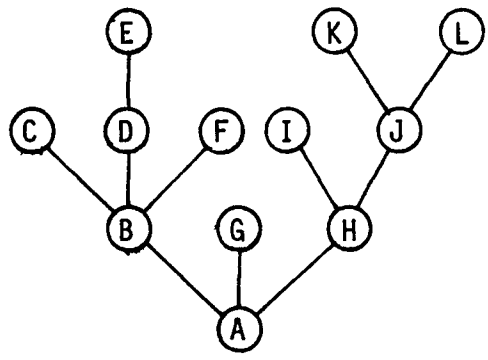

Figure 1. Pictoral Representation of a Hierarchal Data Structure

This tree structure closely mimics the natural flow pattern of surface runoff, aggregating from field to field into small intermittent channels, and those in turn into larger and larger perennial channels.

In a hierarchal tree structure, in order to proceed down the tree, each cell must have one data record associated with it indicating the cell to which it flows. This record must be determined beforehand and be included in the basic data file. In addition, each cell may have one or more cells flowing into it. To proceed up the tree then, each cell must have a list of which cells, if any, flow into it. This list does not have to be developed beforehand but can by built in the computer based on the root record of each of the cell's branches.

In addition to paralleling natural flow patterns, a significant advantage using hierarchal data structures may be obtained if the model is programed in a computer language called Pascal. This advantage results from a feature of Pascal, known as dynamic data allocation, which allows computer space to be allocated as needed, eliminates wasted allocation and maximizes computer storage potential. Incidentally, another outstanding feature of Pascal is its ability to do something called recursion, which allows a subroutine (subsection) of a program to reference (call) itself. This feature is a tremendous asset when using hierarchal data structures.

There is one other strong advantage of using hierarchal data structures, which is the ability to track sediment from a field to a channel. This feature may best be described in an overall discussion of modeling sediment and runoff, using the combined strengths of field scale data and hierarchal data structures.

Modeling using field scale data and hierarchal data structures

Natural flow patterns may be subdivided into and modeled as two distinct but integrated subsystems. There can be one hierarchal subsystem representing the channel segments, and another one representing fields flowing to the channel segments. In the channel subsystem, a channel segment must flow to another channel segment, but it may have either fields, channels or both flowing into it. On the other hand, a field can flow to another field or channel, but it can only have fields flowing into it, not channels. Clearly the dynamics of sediment routing operate differently in both subsystems. Also the distinction serves the purpose of identifying the extent of eroded sediment from an upland field that is delivered to a channel (as opposed to the mouth of the watershed). The need for and practicality of this limit will be made clear shortly.

When modeling a watershed using hierarchal data structures, one begins at the root of the entire watershed, i.e., the mouth of the main channel. For each iteration or time segment, the model proceeds up the tree, stopping at each cell to calculate sediment detachment, transport and deposition, routing the excess back down to its root, then proceeding up each of its branches, in turn repeating this process. The cells of Figure 1. for example would be processed

in alphabetical order. Therefore, for a main channel outlet that has two branches coming into it, one branch, and all of the watershed draining into it, will be processed during each interation before the analysis proceedes up the other branch. If the branch in question is a channel, then a group of paramaters describing that channel, and equations governing channel transport, can be called in. If it is a field, then the parameters must describe the field, and the equations governing overland detachment, deposition and transport must be used. In either case, the excess runoff and sediment is routed to the cell's root, be it a field or channel, and the cell's branches (if any) are in turn processed, be they fields, channels, or a combination of both. By looping through several iterations over time, it is possible to track the net sediment dynamics in each of the field and channel segments.

If the analysis ended at this point, the major differences between this and current models would be the units of the input data, the manner of cell processing, and the units of the output data. The output of the model could, in a parallel fashion to others, indicate net sediment gain or loss on each field and channel (not cell), as well as total watershed sediment loading. However, using the components of this model, it is possible to improve on this output significantly. With field scale data and hierarchal data structures it is possible to track the amount of sediment eroded off of a field that is delivered to the channel network. This is possible because the length of overland flow prior to channelization is generally relatively short. The SCS (1972) indicates that it is seldom more than 1000 feet. In the small watershed that ANSWERS was calibrated on, overland flow was seldom across more than two or three fields before channelizing. In that watershed, the greatest number of fields of consequtive overland flow was five. These five fields were narrow, contour cropped, and on the side of a long gently sloping hill.

This rather rapid channelization of overland flow means that the tree structures representing fields in the computer are typically quite short, i.e., there are a lot of them but they do not branch very far. As a consequence, it is practical to do two things in the computer that would allow a program to track sediment downslope. The first is to, for every source field, keep a linked list (a one dimensional tree) representing the sediment in suspension on each field over which overland flow from the source field must traverse before reaching a channel. Fields adjoining a channel would have only one record in their linked list. Fields with two intervening fields between themselves and receiving channels would have three records in their linked list. Sibling cells (branches with the same root) would have the same number of records in their linked list. Each of their records would, in parallel, refer to the sediment originating from their respective cells in suspension on coinciding downslope cells. The sediment in suspension on each field at any one time would be equal to the sum of the sediment in the coinciding cell of the linked list of each of its daughter cells. Again, using Pascal and dynamic data allocation, the required storage capacity would be minimized.

The second component necessary to track sediment downslope is a process, not an item. Whenever sediment dynamics on a given field are calculated, a

traversal through each of the cell's daughter cells is required. The purpose of this traversal is to modify the record on each of the daughter cell's linked list corresponding to the field in question. On the record of the linked list corresponding to the field, it would proportionally decrement the sediment load for deposition (if any) and downslope transport. Then it would increment the adjoining downslope record for inflow from the upslope record. When the root cell flows to a channel (as opposed to another field) the downslope transport represents delivery to the channel. By this means sediment delivery from a field to a channel can be tabulated. The reason that this is possible but limited to overland flow is that the computer storage and execution effort required with each cell to perform these calculations increases linearly with the number of daughter cell it has. A field seldom has more than a few daughter cells, but a channel can have hundreds to thousands.

The above description outlines a proposed flow of control for a suggested sediment delivery model. References to specific methods for calculating overland and channel flow and sediment transport are noticeably lacking, but they can be selected from several in existence.

The data that would be necessary to run the proposed model includes two detailed types of inventory (field and channel), as well as the selection of appropriate design events. The field inventory would require field by field descriptions, including all of the data presently collected in the Piority Watersheds to estimate erosion using the USLE, plus two additional data records. One would be the overland flow distance of the field. This, in conjunction with the field acreage, would be needed to estimate field dimensions, as flows are usually calculated in unit volume per unit time per unit width. The second additional data record would be a designation of the field or stream to which each field flows. Anticipating that at some time in the not too distant future the Department may procure such a model, the Nonpoint Source Section is evaluating a method by which these additional pieces of information may be obtained.

The second part of the inventory consists of a channel inventory. The channel inventory would require a channel segment identifier, a physical description of the channel segment (width, profile, gradient and vegetation if any), length of the segment, and a designation of the channel segment that this segment flows to. This would not be difficult for perennial streams, but could become tricky when applied to small intermittent streams, grassed waterways, roadside ditches, etc.. The present channel inventory that is undertaken in Priority Watersheds, mainly to evaluate habitat and biotic potential, would have to be upgraded significantly to accomodate this data.

CONCLUSIONS

The Wisconsin Nonpoint Source Program requires a model capable of estimating the sediment load from fields to streams, and sediment transport within streams, under current and projected field management conditions. This model

must accept data that can be collected for large areas in a reasonable amount of time and with a high degree of reliability. Lastly, the output of the model must be appropriate for making field scale management decisions based on water quality objectives. It is argued that the proposed model could demonstrate all of those attributes. Finally, it is suggested that appropriate agencies consider funding the implementation of such a model.

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EUTROPHICATION PROBLEMS IN NORTH CAROLINA AND MANAGEMENT APPROACH FOR NONPOINT SOURCES

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The State of North Carolina has been experiencing eutrophication problems in several coastal rivers for the past decade. These problems are characterized by extensive surface blooms of blue-green algae during June through September. Two recently filled multi-purpose reservoirs in the lower piedmont have also been exhibiting signs of eutrophication although only minor blooms of nuisance algae have as occurred yet. Extensive monitoring and research efforts have been conducted in the watersheds of these eutrophic systems to characterize the magnitude of nonpoint sources of nutrients and sediment from urban, agricultural and forested watersheds. The effectiveness of "best management practices" (BMP's) has also been evaluated.

In an effort to mitigate the occurrence of nuisance blooms one coastal river and prevent the occurrence of these problems on the piedmont reservoirs, the State has classified the watersheds of these systems as "Nutrient Sensitive Waters" (NSW). This classification provides the regulatory authority to limit the input of nutrients from point sources. However, agricultural and silvicultural activities are explicitly exempted from this regulatory authority. Although urban areas are not exempted from this regulatory authority, there are limited approaches for controlling nonpoint nutrient inputs from existing urban areas.

Despite the limited regulatory authority, the State has used the NSW classification to educate the public, local government officials, and the General Assembly about nonpoint problems. The threat of point source nutrient controls has been used to force local governments to plan and control runoff from new urban expansion. It was also a major factor in motivating the General Assembly in 1984 to provide $2.5 million for the voluntary implementation of cropland and animal operation BMP's in NSW watersheds. These efforts in NSW watersheds are the first test case for the effectiveness of nonpoint source planning and control measures on a large scale in North Carolina.

Keywords: Eutrophication, nuisance algal blooms, nutrient sensitive waters (NSW), nonpoint source pollution control, agricultural runoff, urban runoff, future development.

INTRODUCTION

Freshwater sections of several coastal rivers in North Carolina have been experiencing severe eutrophication problems since the early 1970's. The Chowan River (Fig. 1) was the first of these rivers to bring public and regulatory attention to eutrophication problems with the development of extensive surface mats of blue-green algae. These surface blooms have occurred nearly every year since 1970 and in 1972, 1978 and 1983 surface mats covered nearly all of the 30 km long lower river during part of the summer and fall (N.C. DNRCD, 1982a).

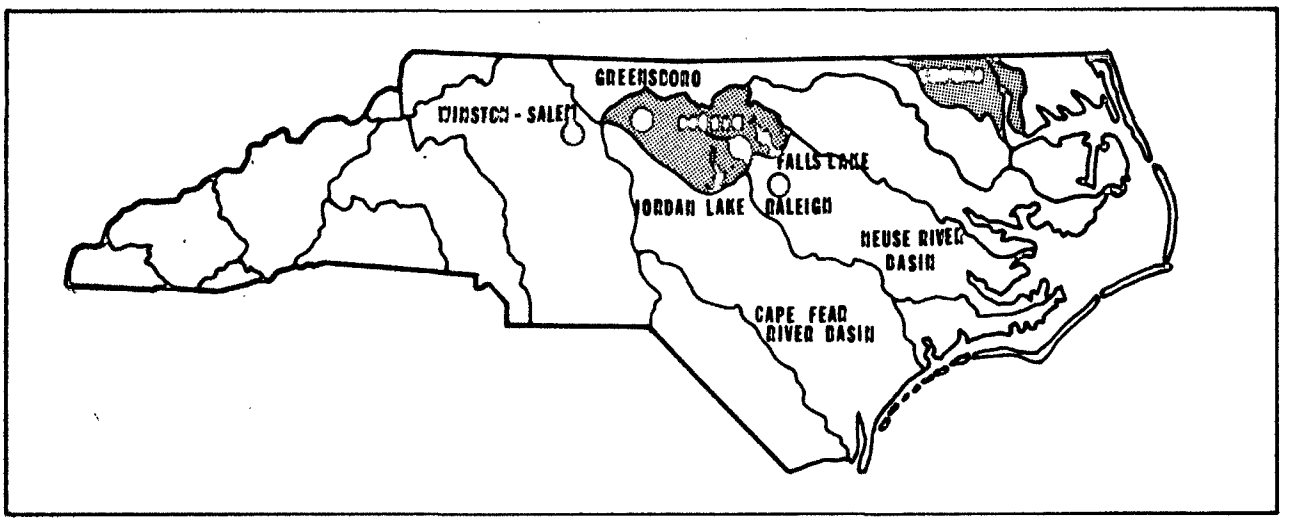

Fig. 1 Nutrient Sensitive Watersheds in N.C.

N. C. inland lakes and reservoirs have a wide range of trophic conditions. Trophic state classification surveys of nearly 80 bodies of waters conducted in the mid 1970s and early 1980s indicate that 30 to 40 percent of these lake/reservoirs are eutrophic or hypereutrophic (Weiss and Kunzler, 1976; N.C. DNRCD, 1982b). Extensive surface blooms of blue-green algae have not occurred in lake/reservoir systems. However, many water treatment plants using these systems as raw water supplies periodically have to modify treatment to minimize taste and odor problems. Two recently impounded reservoirs, B. Everett Jordan Lake in the Upper Cape Fear River Basin and Falls Lake in the Neuse River Basin (Fig.1), are already considered the most eutrophic reservoirs in N. C. These reservoirs are planned as water supplies and recreational areas for the rapidly growing Raleigh/Durham/Chapel Hill area and there is considerable concern over the impact of severe eutrophication on uses of the lakes (N.C. DNRCD, 1983a).

The N. C. Division of Environmental Management (DEM) has developed a watershed classification entitled "Nutrient Sensitive Waters" (NSW) as a regulatory means to deal with eutrophication. The NSW classification provides the regulatory authority to limit inputs of nutrients to background levels, if necessary. The authority is effective in controlling nutrient inputs from point source discharges, however the North Carolina General Assembly has passed legislation which has exempted agricultural activities (other than concentrated

feed-lots) and silvicultural activities from this of the authority. Urban runoff is not exempted from the DEM authority provided by the NSW classification but there are limited approaches for effectively reducing nutrient inputs from existing urban areas.

The Chowan River watershed has been classified as NSW since 1979 while the watersheds to Falls and Jordan Lakes were classified NSW in 1983. Point source nutrient controls (nitrogen and phosphorus in the Chowan River watershed and phosphorus controls in Falls/Jordan watersheds) are currently being implemented. Nonpoint source control implementation has initially been slow, however a special funding program provided by the N. C. General Assembly is making progress on agricultural Best Management Practices (BMP) implementation in all three NSW watersheds. Also, a unique local/state government cooperative approach seems to be making progress in minimizing runoff problems from urban expansion as well as showing some potential to control some existing urban runoff problems in the Falls/Jordan watersheds.

CHARACTERIZATION OF EUTROPHICATION PROBLEMS

Chowan River

The Chowan River is located in Northeastern North Carolina (Fig. 1). It is formed by the confluence of the Blackwater and Nottway rivers near the Virginia/N.C.state line and stretches about 80 km south to Albemarle Sound. The watershed is approximately 12,700 km^2 with 73 percent in the Piedmont and inner Coastal Plain of southeastern Va. The watershed in North Carolina is entirely in the coastal plain and the Chowan is tidally influenced over its entire N. C. length. Despite the tidal influence, the Chowan River is essentially a freshwater system except during periods of extreme drought.

The Chowan River can be divided into two district regions. The upper portion of the river, from the Virginia state line to Holiday Island (48 km) is a relatively narrow (90 to 1200 m in width), sluggish river with depths up to 12 m. The lower river from Holiday Island to Albemarle Sound (32 km) ranges from 1200 to 3000 m. wide and behaves like a shallow (mean depth 4.0 m), rapidly flushing lake. Surface mats of blue-green algae occur primarily in the lower river due to the large surface area, shallow depth, and lake-like tendencies (N.C. DNRCD, 1982a).

Detailed monitoring by DEM has been used to estimate nutrient inputs to the lower Chowan River from 1977 to 1981 (N.C. DNRCD, 1982a);

Annual Flow (M^3/sec)		Total Nitrogen (kg x 10^3/yr)		Total Phosphorus (kg x 10^3/yr)	
Ave	Range	Ave	Range	Ave	Range
133.7	34-242	40.3	6.8-75.1	4.3	0.8-7.5

These estimates indicate that the N:P ratio is in a range where either nitrogen or phosphorus could be limiting. Detailed research studies on phosphorus and nitrogen uptake, nitrogen fixation, and sediment nutrient recycling confirmed that nitrogen and phosphorus limitations varied both seasonally and as a function of the phytoplankton community. Phosphorus was identified as the primary nutrient requiring control since the major sympton of the eutrophication problem was surface blooms of nitrogen fixing blue-green algae in the lower river (Kuenzler, et.al., 1980). A 30 to 40 percent reduction in phosphorus inputs was recommended based on modelling analyses by DEM (N.C. DNRCD, 1982a) and confirmed by research results (Paerl, 1982). A 15 to 20 percent reduction in total nitrogen inputs was also recommended since nitrogen was considered a major factor in occasional blooms of Microcystis spp., a non-nitrogen fixing blue-green, in the upper portion of the Chowan (Witherspoon and Pearce, 1982).

The magnitude of blue-green blooms seems to be related to hydrologic conditions (Fig. 2). The most severe blooms seem to occur after above average spring tributary flows and moderate summer flows. If high river flows are sustained through the summer, the flushing action minimizes blue-green growth. Likewise, if stream flow is extremely low, insufficient nutrients are delivered to the lower river to sustain algal growth. For example, during 1980-81, a 15 month drought limited blue-green growth in 1980 from a lack of nutrients. By the summer of 1981, saltwater had moved up the Chowan River above (Holiday Island) completely eliminating the growth of nuisance species of blue-green algae (N.C. DNRCD, 1982a).

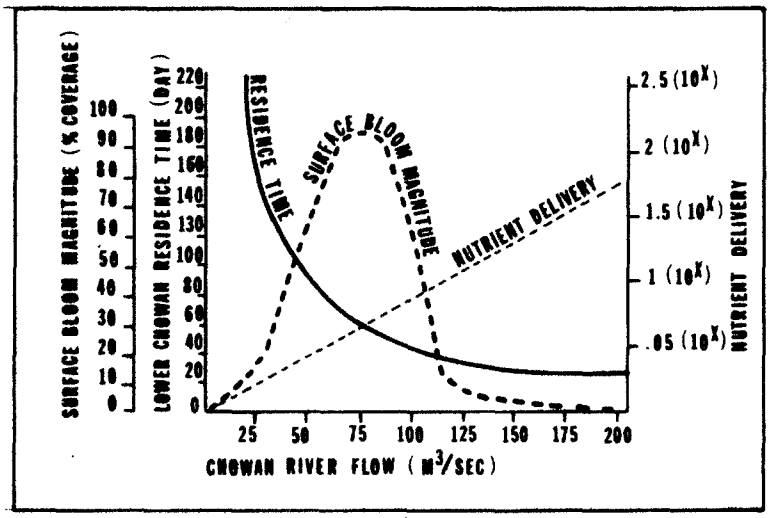

Fig. 2. Relationship between flow, residence time, nutrient delivery, and algal bloom magnitude in the lower Chowan River(N.C. DNRCD, 1982a).

Falls and Jordan Lakes

Falls and Jordan Lakes are located near the edge of the lower Piedmont and lower Coastal Plain in N. C. (Figure 3). Falls Lake

is a 5000 ha impoundment of the Neuse River which was filled in February 1983 and has a watershed area of 2000 km^2. The reservoir is 39 km long with a mean depth of about 4 m. Jordan Lake is an impoundment of two major tributaries of the Cape Fear River - Haw and New Hope Rivers and was filled in September 1981. The watershed to Jordan Lake is 4464 km^2, the lake area is about 5500 ha, and the mean depth is approximately 5 m.

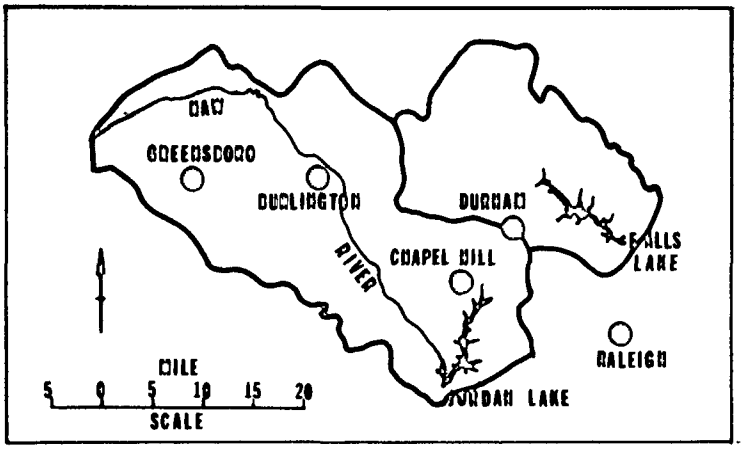

Fig. 3. Falls and Jordan Lake Watersheds.

There has been considerable controversy over both impoundments. Both reservoirs are U. S. Army Corps of Engineers projects for purposes of flood control, recreation, water supply, wildlife protection and downstream water quality control. However, due to extensive urban, industrial and agricultural development in both watersheds the trophic state of the lakes was a major issue. Also, the suitability of Jordan Lake as a water supply is still being examined due to the presence of 170 permitted wastewater dischargers as well as extensive urban and agricultural runoff in the watershed (N.C. DNRCD, 1985).

Water quality monitoring conducted during the first few years after filling the reservoirs has confirmed some of the predictions of lake quality. Jordan and Falls Lakes are the most eutrophic reservoirs in N.C. although the trophic status has not yet impaired uses of the lakes (N. C. DNRCD, 1984b). Chlorophyll *a* levels in both lakes have at times exceeded 200 to 300 mg/m^3. However, this has been the result of blooms of phytoplankton which are not nuisance species (e.g. *Chrysochromulina spp.*) (N.C. DNRCD, 1984a). Chlorophyll *a* levels have been averaging between 40 to 80 mg/m^3 during the summers of 1983 and 1984. There seems to be a trend towards more dominance of some blue-green species in Jordan Lake, but more time is needed to confirm this trend. These reservoirs have relatively short average retention times of 7 and 270 days in the Haw and New Hope arms of Jordan Lake, respectively, and 120 days in Falls Lake. Already there have been several occasions where extreme flows have flushed substantial phytoplankton population from the lake (N.C. DNRCD, 1983a; Weiss et.al., 1984; Francisco et.al., 1984).

Table 1 summarizes inflow information and estimated nutrient inputs to the lakes for mean annual flow conditions. This information indicates N:P ratios in the range of 5:1 which

indicates more nitrogen limitation than in the Chowan River. The general abundance of nutrients and nitrogen limited conditions are major reasons for the concern over future problems from nuisance nitrogen fixing blue-green algal species.

Table 1. Estimated Nutrient Inputs to Falls and Jordan Lakes From Instream Monitoring Results (N.C. DNRCD, 1983).

Lake Watershed	Mean Annual Flow (m^3/sec	Total Nitrogen (kg x 10^4/yr)	Total Phosphorus (kg x 10^4/yr)
Falls	22.5	77.9	13.5
Jordan - total	48.6	227.0	55.6
Haw River Arm	39.3	181.8	45.7
New Hope River Arm	9.3	45.2	9.4

NUTRIENT SOURCES

Chowan River

The Chowan River watershed is predominantly rural in both N.C. and Va. It's watershed is 80 percent forested and wetland areas, 19 percent agricultural and 1 percent urban. The total population in the watershed was a little more than 200,000 in 1980. Municipalities are generally quite small. The largest dischargers in Va. are Emporia (0.6 MGD), Blackstone (0.5 MGD), and Franklin (0.8 MGD), while in N.C. Ahoskie (0.8 MGD) and Edenton (1.0 MGD) are the largest municipalities. There is one large industrial discharger in the Va. watershed. Union Camp is a pulp and paper facility located near the state line which discharges in excess of 380 x 10^6 l/day (100 MGD) from holding ponds only during the months of December through March when the Chowan River can assimilate the waste. In N.C., there was a large fertilizer manufacturer (C.F. Industries) in operation until 1983 which was initially blamed for the occurence of nuisance blooms in the early 1970s. C.F. Industries did discharge significant quantities of nutrients in the late 1960s and early 1970s. This discharge ceased by 1975 and thereafter, this company contributed nutrients to the river primarily through air fallout, site runoff and contamination of groundwater. (N.C. DNRCD, 1982a).

Figure 4 depicts nutrient budgets for the Chowan River watershed in N.C. (N.C. DNRCD, 1982a). Agricultural runoff (50%), forest and wetlands (22%), direct precipitation (5%) and urban runoff (0.5%) account for about 78% of the phosphorus inputs from the N.C. watershed. Nonpoint source contributions of nitrogen are similar, 80 percent.

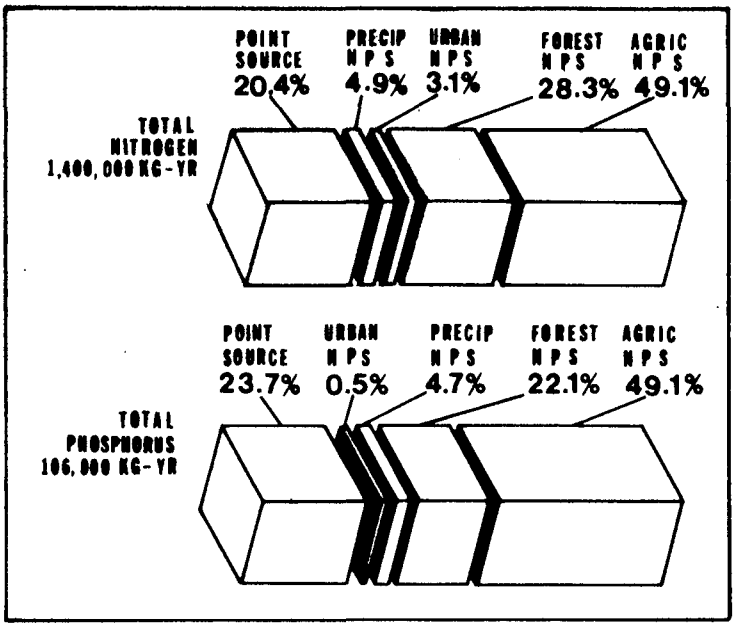

Fig. 4. Land use/point source nutrient budget for the Chowan River Watershed in N.C.(N.C. DNRCD, 1982a).

Falls and Jordan Lakes

In contrast to the Chowan River watersheds, portions of the Neuse and Cape Fear River basins draining to Falls and Jordan contain some of the most rapidly developing areas in N.C. The Jordan Lake watershed includes the urban centers of Greensboro, Burlington, Chapel Hill and the southern part of Durham with a total population in 1980 of about 475,000 people. There are 10 major municipal dischargers in the watershed releasing nearly 300 x 10^6 l/day (80 MGD). In addition, there are several major industrial dischargers and a total of 170 permitted dischargers in the watershed. Urbanization is continuing at a rapid pace in this watershed with the projected population for the year 2000 to exceed 600,000 people.

Development in the Falls Lake watershed is not currently as extensive as in the Jordan but the area is also growing rapidly. Major urban areas include Hillsborough and northern Durham while the northern portion of the Raleigh metropolitan area is also pushing into the watershed. The 1980 population was about 150,000 people and is estimated to exceed 210,000 people by the year 2000. There are two major municipal dischargers and a total of approximately 60 permitted dischargers in the watershed.

Based on analyses of land use and point source data nonpoint sources account for approximately 45 percent of the nitrogen and 40 percent of the phosphorus into Jordan Lake and 70 percent of the nitrogen and 57 percent of the phosphorus into Falls Lake (Figure's 5 and 6). Nutrient inputs seem to be readily controllable through point source actions. However, the lakes currently have such high level of nutrients that the systems may remain eutrophic even if all point source inputs of phosphorus were eliminated (N.C. DNRCD, 1983a; Francisco et.al., 1984). Future development in the watershed could also dramatically

increase nutrient contributions from urban runoff which are already substantial.

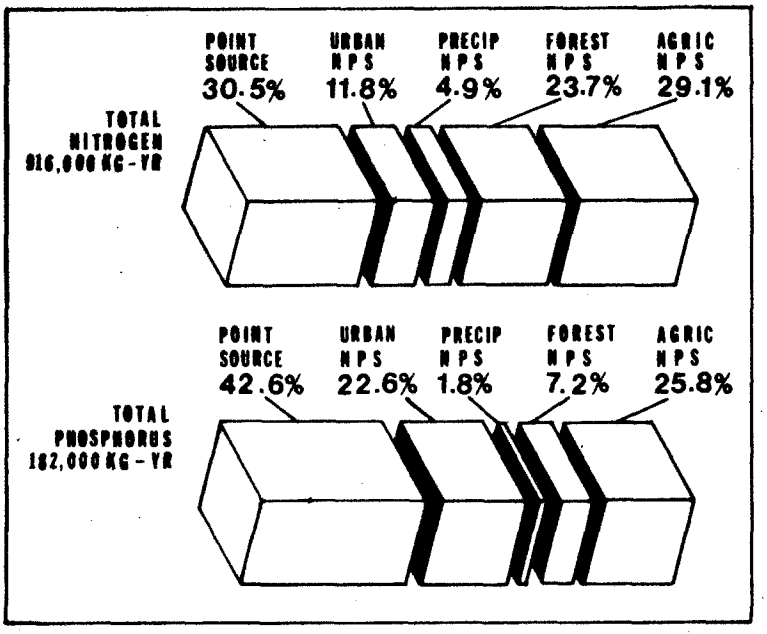

Fig. 5. Land use/point source nutrient budget for the Falls Lake Watershed (N.C. DNRCD, 1983a).

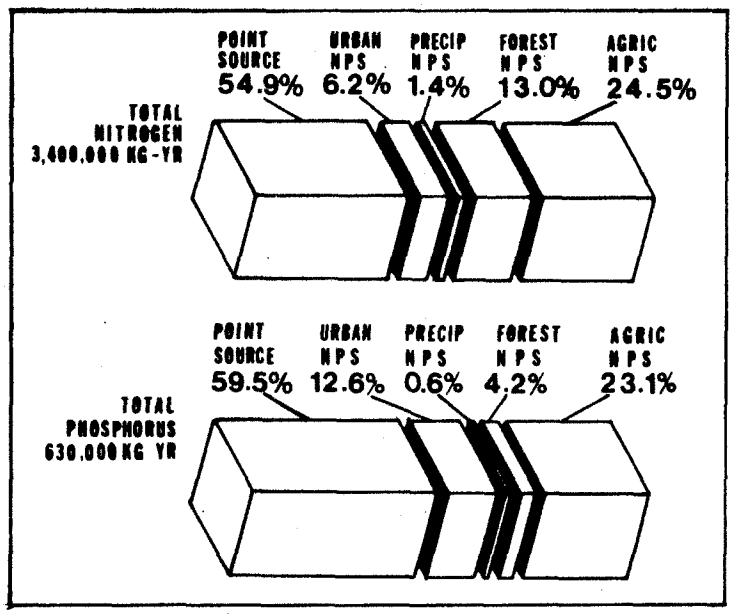

Fig. 6. Land use/point source nutrient budget for the Jordan Lake Watershed (N.C. DNRCD, 1983a).

NONPOINT AND POINT SOURCE MANAGEMENT

Chowan River

Nonpoint source control initiatives began slowly. When the Chowan was classified as NSW in 1979, agricultural agencies were unwilling to admit to any contribution to eutrophication problems of the Chowan. This unwillingness along with legislation limiting the ability of the DEM to require nutrient reductions

from agricultural and silvicultural operations severely limited options to reduce nutrients in N.C.

With these restrictions, the State took two major initiatives; 1) Actions were started to eliminate or minimize all point source contributions of nitrogen and phosphorus and 2) Research/education efforts were begun to verify agricultural contributions to the nutrient enrichment problems, identify the effectiveness of various best management practices, and to offer technical assistance for BMP implementation.

Point source efforts were relatively straight-forward in that all the municipal facilities were small and required upgrading. Facilities were promoted on the priority list for construction grants and all facilities began to conduct non-discharging to land application, wastewater disposal. These systems will all be operational by 1987. Small industrial discharges were required to meet total nitrogen and phosphorus limits of 3 and 1 mg/l, respectively, and several enforcement steps were taken to limit nutrients in runoff and seepage from the abandoned C.F. Industries fertilizer manufacturing site. These actions have eliminated point source nutrient contributions as an issue in the North Carolina watershed, and have focused attention on agricultural contributions (N.C. DNRCD, 1982a).

A three-year research study from five watersheds from 200 to 2800 ha in the Chowan basin in N.C. was conducted from 1979 to 1982 (to examine agricultural runoff) (Humenik et.al., 1983). The study was not able to quantify before and after effects from BMP implementation due to the short research time. However, valuable survey information on existing management practices and producer attitudes towards nonpoint source controls were obtained. The information was utilized in developing supportive publicity, educational and technical assistance programs for the entire Chowan watershed. Estimated annual yields from the forested (control) watershed and the four agricultural watersheds (averaging 50 percent cropland) are shown on Table 2.

Despite the failure of the research to demonstrate BMP effectiveness, these studies in combination with other State action helped to involve agricultural agencies in the development of a nonpoint control program. Also, the study indicated that estimates of nutrient contributions from agriculture shown in Fig. 4 may have been somewhat conservative. In addition, survey results helped to target cropland requiring control measures and animal operations requiring waste management systems. BMP effectiveness was demonstrated as part of research in other parts of N.C. and demonstration farmswere set up to demonstrate structural BMPs , provide education on fertilizer application, serve as advocates for soil testing, and demonstrate animal waste management systems.

Table 2. Sediment and Nutrient Yields from Five Watersheds in the Chowan River Watershed (Humenik et.al., 1983).

Parameter	Annual Yield (kg/km^2) Forested Watershed	Agricultural Watersheds Ave.	Range
Suspended Sediment	13,700	71,940	4972-400,400
Total Nitrogen	135	627	110-1331
Ammonium-Nitrogen	8.8	28	1.1-73
Nitrate Nitrogen	17	275	6.6-672
Organic Nitrogen	101	303	84-1071
Total Phosphorus	13	114	19-351

At the end of 1982 and early 1983, this put the agricultural control program in a position to begin implementation. Based on the overall nutrient reduction goals of 30 to 40 percent for phosphorus and 15 to 20 percent for nitrogen from watershed areas in N.C. and Va. as well as the reductions being obtained from point sources, goals of 30 percent effectiveness for phosphorus and 20 percent effectiveness were set for the agricultural control program (N.C. DNRCD 1982a). However, there was little or no special funding to encourage implementation of BMPs so the program was entirely voluntary. The implementation program had to rely on existing staff from the N.C. Agricultural Research Service and N.C. Soil and Water Conservation Districts to provide technical assistance. The capability of the N.C. Department of Agriculture to greatly expand soil testing in the Chowan watershed was also insufficient. Cost sharing funds from the ASCS also had to be relied upon for structural BMP implementation.

Falls and Jordan Lakes

When Falls and Jordan Lakes were classified as NSW in October 1983, this clearly indicated the intent of DEM to regulate point sources of nutrients beginning with phosphorus. Initial policies for the lake were to limit all new facilities to a total phosphorus limit of 1 mg/l and encourage land application wastewater treatment systems whenever possible. This applied primarily to rural subdivision developments on both watersheds but also is applicable to any expansion of existing wastewater facilities. Existing facilities in the New Hope arm of Jordan Lake were issued compliance schedules to meet a 1 mg/l phosphorus limit in 3 years. All other facilities were issued letters indicating that the watershed was NSW and nutrient limits might be established in the future.

Clearly substantial phosphorus reductions to both lakes could be accomplished based on budgets in Figure 5 and 6 by requiring phosphorus removal at point sources. However, contractors studying the lake for the Corps of Engineers have expressed doubts over the estimated nutrient contributions from point

sources in the Jordan watershed, the effectiveness of phosphorus reductions for controlling eutrophication, and the need to control eutrophication since no uses are currently being impaired. (Francisco et. al., 1984). This opinion was considered in developing the initial control strategy for point sources. There has been little disagreement that actions are necessary to encourage proper planning to minimize future nonpoint source contributions of sediment, nutrients and toxicants. All three pollutant categories are a major concern relative to water supply usage of the lakes.

Public concern over the future of these lakes put state agencies in a favorable position to get cooperation from local governments to protect the lakes. The threat of point source controls on phosphorus has also assisted in getting cooperation from government groups in the upper part of the watershed which do not get recreational and water supply benefits from these lakes. Committees were formed with state and local officials and, with assistance from DEM staff and local/regional planning groups, guidance was developed for local actions to protect the lakes from future nonpoint source inputs. Essential elements of these action plans are as follows;

- Local sedimentation and erosion control plans,
- County soil and water plans for high erosion areas (cropland),
- Requirement of storage of first 1/2" runoff from new development,
- 50 foot stream buffers,
- Designation of water quality critical areas near the lakes
    - 6% impervious limit
    - Restricted commercial/industrial development
    - No sewer extentions,
- 12% impervious limit for unsewered, non-critical areas,
- 30% impervious limit for sewered, non-critical areas,
- Designation of limited industry areas.

In addition to these efforts to minimize nonpoint source impacts from new development, DEM is continuing to investigate means to reduce runoff pollutants from existing urban areas. Using loading rate information from the Nationwide Urban Runoff Program study in Winston-Salem, N. C. and nationwide, portions of the urban areas have been targeted for controls(N.C. DNRCD, 1983b) . Stormwater detention basins for the 30 percent of urban areas with highest pollutant yields could reduce metals inputs from urban runoff by as much as 50 percent and nutrient inputs by 20 to 30 percent (N. C. DNRCD, 1985). This information is being provided to local planning agencies and funding is being considered for urban detention basin demonstration projects.

NSW LEGISLATION

In June 1984, the N. C. General Assembly approved special funding for the implementation of nonpoint source controls in NSW watersheds. Approximately $2.5 million/year was provided to the county soil and water conservation districts through the N. C. Division of Soil and Water. The bulk of this funding will be

used for 75 percent cost-sharing of animal waste management systems and cropland BMP's. A portion of the funding was also used to hire an additional staff member for each of the counties in NSW watersheds to provide technical assistance.

During the first six months of its existance this program concentrated on publicity and getting agricultural producers interested in the program. A good portion of the cost-sharing funds were used for animal waste management. Specific targeting of BMPs for high erosion rate cropland has not occurred during the first year since it takes some time to get producers involved in this type of voluntary/cost incentive program and it is important for continued funding to utilize all funds provided. Targeting of highly erodable cropland for BMPs is planned for the second year of implementation.

In addition to agricultural cost-sharing funds, additional funds were provided for enforcement of state sediment and erosion control laws. Also, special research funding is available to local governments to investigate means to control nutrients from municipalities. Research will likely center on biological means to reduce phosphorus at existing wastewater facilities. However, these funds may be used in the future to investigate the effectiveness of detention ponds for urban nonpoint source pollution control.

FUTURE DIRECTIONS

The NSW watersheds are the first test case for the effectiveness of nonpoint source planning and control measures on a large scale in North Carolina. At present, initial nutrient control needs have been established and a program has been developed for agricultural land use. A program is being further developed to control urban runoff. For the Chowan River Basin, the State of Virginia has just developed a plan for point and nonpoint source controls for protection of their watershed and is beginning to be implemented.

The future success of the nonpoint control programs in NSW watersheds depends on continued cooperation between state and local agencies, continuing legislative funding, and research/study efforts.

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REDUCING BACTERIAL NON-POINT POLLUTION IN TILLAMOOK BAY, TILLAMOOK, OREGON

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ABSTRACT

High coliform counts in the waters of Tillamook Bay closed oyster harvest. A review of possible sources of pollution revealed that sewage treatment plants, failing on-site domestic septic systems and the county's 18,000 dairy cows were the primary candidates.

Extensive water sampling confirmed all three contributed. Major improvements were made in the sewage treatment plants and the on-site systems but the nonpoint contribution from the 100 plus dairy operations proved to be challenging.

A computer program was developed to evaluate management practices and predict the movement of organisms from the cow to the Bay. This assisted dairy operators in selecting the specific component to improve or add to their unique operation to make the greatest positive impact on water quality. State and federal agency personnel also used the model to identify those operations with the greatest need and distribute federal Rural Clean Water Program cost share monies to assist in upgrading facilities.

Several local citizens' groups, clean water committees, state and federal agencies worked with the oyster growers and the dairymen toward the common goal of improved water quality. The program is still underway but recent sampling suggests an improvement in Bay water quality.

INTRODUCTION

There are many documented water quality problems in the United States. Over the past 15 years nutrients, primarily nitrogen, have received the greatest attention as pollutants. Currently several cases of bacterial pollution from non-point sources have drawn attention to these problems and to the general lack of research effort in this area.

Organisms have been responsible for closures of oyster harvesting areas in a number of bays and estuaries. One such closure occurred in Tillamook Bay in 1977 (1). Tillamook Bay is on the Pacific Coast in the northwest corner of Oregon (Figure 1).

This closure brought attention to the problem and focused the efforts of several state and federal agencies and local groups. Water quality sampling conducted by the federal Food and Drug Administration (FDA) (2) and the state Health Division (HD) and Department of Environmental Quality (DEQ) during the seventies showed that bay water quality did not meet standards of the National Shellfish Sanitation Program (NSSP) (6) (3). In general these studies showed that a potential hazard did exist as high levels of total and fecal coliform bacteria were present in oyster beds regardless of weather and tidal conditions.

From reviewing the studies and input from involved agency personnel, three major sources of bacterial pollution have been identified. These are the five sewage treatment plants in the watershed, failing on site domestic sewage systems and livestock wastes. A two step plan was proposed to address the pollution from these sources. First to reduce the bacterial input by implementing technical and institutional options such as improved sewage treatment, land zoning, and installing best management practices on the dairy farms (5). The second part was to develop a bay closure scheme when failing equipment or heavy rainfall would provide conditions that would excess the sanitation standards set by the NSSP.

The remainder of this paper will address the efforts to reduce those organisms originating from the dairy operations. There are some 18,000 dairy cows on 118 farms in the watershed. These dairies support the Tillamook cheese plant which converts 25% of all the milk produced in Oregon into cheese. The coastal marine climate brings 150 inches of rainfall to the upper part of the watershed and 90 inches of precipitation in Tillamook. Most of the rain falls during the winter while the summers are dry. Within the project area there are 23,500 acres of agricultural lands. The dairies which occupy almost all of the agricultural land are located on the tidal flood plains and river terraces. These flat uplands are dedicated to pastures that are utilized by grazing during the dry 6 months of the year. Cows' access to the streams for watering purposes provide opportunity for fecal organisms contamination during the summer pasturing season. Manure storage and spreading are the waste management practices of concern during the winter season, when cows are confined in and around the buildings.

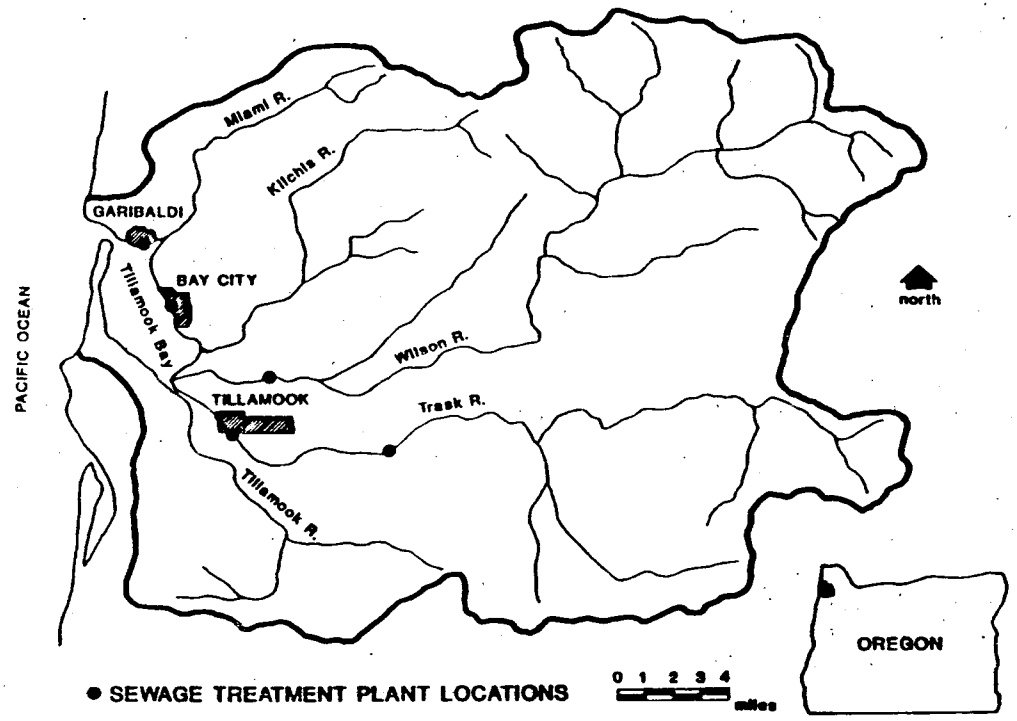

Figure 1. Sewage treatment plant locations on the five rivers in Tillamook Bay, Tillamook, Oregon.

Personnel from the Soil Conservation Service (SCS) and Oregon State University (OSU) provided leadership in a group which developed a list of best management practices. Many of the practices on the list have been proven effective in reducing nutrient pollution but were only speculated to be effective in reducing bacterial pollution.

A research project was funded to assist in evaluating manure management practices and help operators select effective practices to reduce organism movement and escape. This research developed a computer model to evaluate manure management practices. The evaluator was developed using coefficients in the literature to estimate die-off, losses and movement in runoff. This model follows organisms on a daily basis, from the cows through manure handling to land spreading and possible escape in runoff.

A flow chart of the model is shown in Figure 2. The dairyman inputs the cow numbers, manure collection techniques and other farm specific information. Organims die-off begins in storage and varies slightly between dry and liquid storage systems. If the operator elects to spread manure on his pasture land the application method and rate influences the number of organisms available to runoff. Seasonal environmental conditions influence the die-off rate of bacteria both in the storage system and on the land surface.

Twenty-nine years of weather records were evaluated to generate a wet, dry and average year in order to evaluate different management practices. Utilizing the daily rainfall and irrigation as hydraulic input and interial drainage as outflow, a moisture balance is kept for lands receiving manure. Once a precipitation event occurs the antecedent moisture conditions and field soil characteristics are utilized to calculate if and when soil saturation will occur and runoff will begin. A percentage of available bacteria are moved into the soil with infiltration and are considered removed from the system in terms of impact on surface water quality. When saturated soil conditions occur a first order kinetic relationship is utilized to calculate the bacteria that are moved overland in the runoff water. If the operator is using a buffer strip a reduction of the transported bacteria occurs. The reader is directed to a paper entitled "Modeling Dairy Waste Management Systems' Influence on Coliform Concentration in Runoff" for details of the model (4).

Using the model on a large number of farms generated the following guidelines for reducing the bacterial pollution potential of dairy operations and land applied wastes:

(1) Storage of wastes will increase management options and reduce pollution of surface waters.

(2) Installing tile will improve soil drainage, reducing runoff events and the movement of bacteria.

(3) Save well drained soils for waste receiver sites during high rainfall periods.

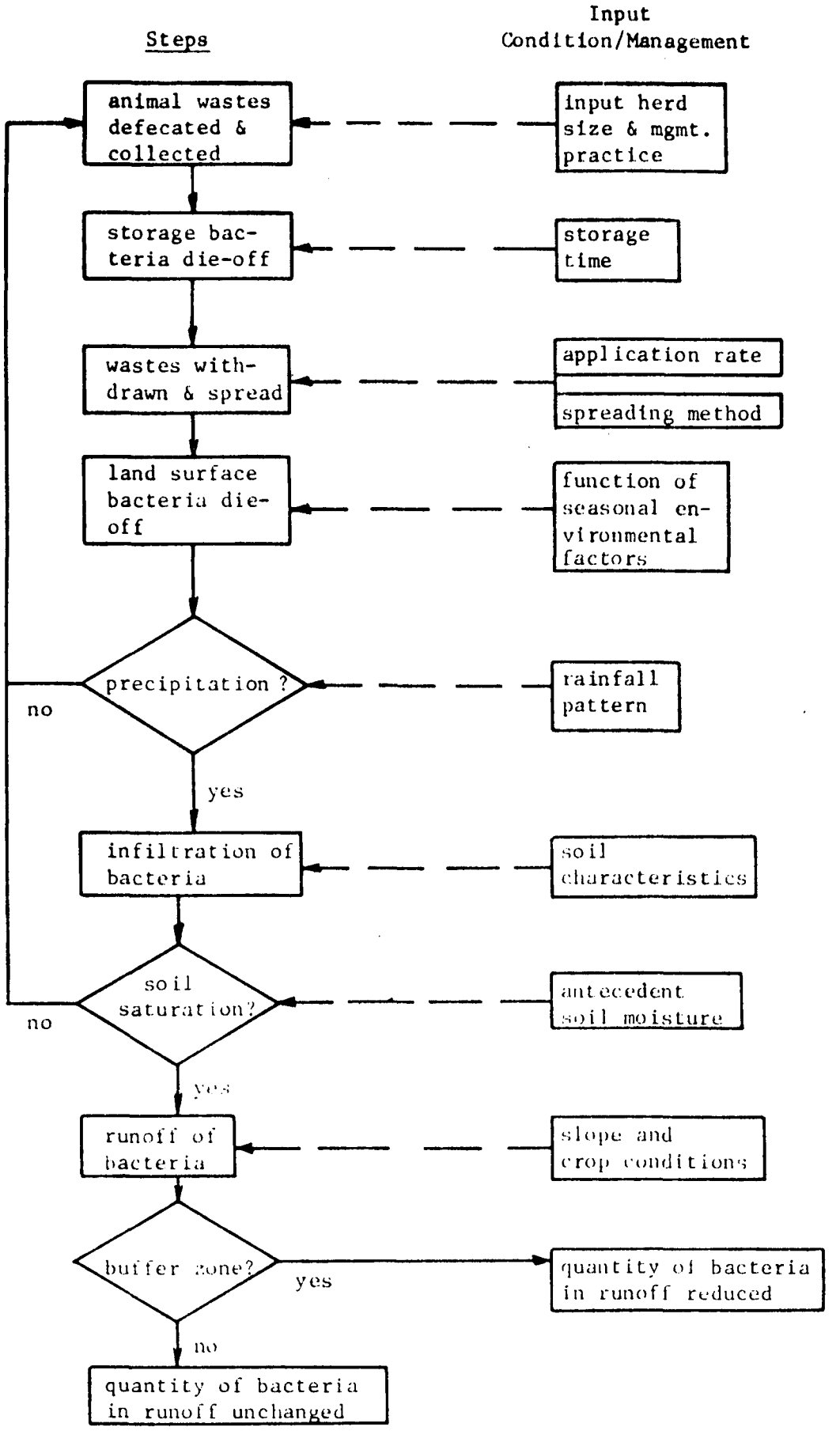

Figure 2. Flowchart of daily bacterial transport model.

(4) Utilize buffer strips between land spreading areas and streams.

(5) Use low spreading application rates over large areas when possible.

(6) Reduce barnyard runoff by minimizing lot area, guttering buildings and diking. This source is the greatest pollution potential. This means keep the clean water clean by rerouting and collect and hold dirty water.

Obviously management is a critical ingredient of any successful water pollution reduction program. Utilizing windows or dry weather periods when they occur to spread manure is perhaps the most important management choice made by dairymen.

The many positive actions of those involved in the project; agency personnel, local clean water committees, oystermen and dairymen were instrumental in presenting a successful grant application to the federal Rural Clean Water Program. This brought over 2 million dollars of cost share funds into the program to assist dairymen in constructing and installing best management practices.

Some 69 operations have signed contracts to implement best management practices as spelled out in their farm plan. By the end of November 1984 some 60 dairy operations had spent over $3,690,000 in construction costs to install new pollution control facilities. Approximately two-thirds of these funds were cost share funds provided by the federal government.

The model and a broad based evaluation system was used to evaluate every dairy which signed up for assistance and cost share funds in order to establish where the funds should be spent to make the greatest positive impact. The model was also utilized to select which practice(s) or component(s) should be installed to have the greatest contribution in reducing bacterial movement to surface waters. Table 1 shows how each component can be ranked when evaluated using the computer model. As suspected, the precipitation level influences the effectiveness of management components. That is, drain tile will be more effective during a high rainfall year, etc.

On site farm visits and educational programs were conducted by members of the DEQ, SCS and Extension Service personnel. Publications have been written specifically for dairymen in this project area. Once the project was underway tours of newly installed practices and facilities were conducted. Local, state, and national press were utilized to tell the story of the efforts and results of the project.

The program is a success in terms of moving the dairymen to improve their facilities. The program began in 1980 and while much construction has been accomplished, the entire program is not expected to be complete until 1990.

Table 1. Model comparison of different management procedures (each examined using a 100 acre field of pasture during the winter with the specified characteristics).

Example	Storage (days)	Management System D=dry; S=semi; L=liquid	Application Rate (tons/ac)	Buffer Strip (yes,no)	Drainage Tile (yes,no)	Soil Type (loam,clay)	Avg. ppt. Net Runoff FC ($x10^9$ org)	Avg. ppt. Rank	Dry ppt. Net Runoff FC ($x10^9$ org)	Dry ppt. Rank	Heavy ppt. Net Runoff FC ($x10^9$ org)	Heavy ppt. Rank
1	0	D	10	Y	N	L	1,848.5	15	1,186.8	3	2,685.6	3
2	0	S	10	Y	N	L	1,388.2	10	891.1	1	2,016.8	1
3	0	L	10	Y	N	L	1,829.9	14	1,188.1	2	2,647.0	2
4	20	D	10	Y	N	L	440.0	4				
5	20	S	10	Y	N	L	330.3	2				
6	20	L	10	Y	N	L	453.2	6				
7	40	D	10	Y	N	L	154.3	1				
8	0	D	40	Y	N	L	7,393.8	19				
9	0	D	10	Y	N	C	2,128.8	16				
10	0	S	10	Y	N	C	1,598.6	11				
11	0	L	10	Y	N	C	2,157.8	17				
12	0	D	10	N	N	L	4,261.2	18				
13	0	D	10	Y	Y	L	838.3	9				
14	0	S	10	Y	Y	L	616.8	7				
15	0	L	10	Y	Y	L	696.3	8				
16	0	D	10	Y	Y	C	1,697.9	13				
17	0	L	10	Y	Y	C	1,624.9	12				
18	20	D	10	Y	Y	C	440.5	5				
19	20	L	10	Y	Y	C	439.7	3				
20	0	D	400	N	N	C	212,873.0	20				

The ultimate success of this program will be measured by sampling the water quality in the bay. A comprehensive sampling program has been planned to begin in the fall of 1985. Routine samples to date indicate an improved quality in the streams and bay.

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Abatement of Nonpoint Pollution of Semi-arid Streams Using Livestock Grazing Management, Vegetation, Instream Flow Structures, and Beaver

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ABSTRACT

An important but often ignored contribution to nonpoint pollution in the semi-arid western United States is sediment. Accelerated erosion and sediment in stream flow is often thought caused by livestock grazing on rangeland in less than excellent condition. Erosion of watersheds is often recognized by stream channel condition. Poor condition streams are often downcut and show active bank rilling, bank sloughing, and piping. Lateral movements of gullies away from downcut streams through upland ecosystems are, in addition, contributing sources for increased sediment pollution. Equally important, when downcut streams occur, associated riparian vegetation is often reduced in areal distribution, disappears, or is replaced by pseudoriparian plant species. Loss of riparian vegetation and the hydrologic support of these limited but valuable ecosystems may further reduce stream channel stability and sediment trapping efficiency of nonpoint source pollution transported to the stream by overland flow and ground water.

This paper addresses theory and methodology being applied in two research programs on perennial and ephemeral streams designed to: 1) promote water storage and vegetation yield, 2) control nonpoint source pollution, 3) reverse desertification of riparian zones and, 4) advance the state of knowledge for management of stream side ecosystems subjected to user pressure.

Livestock grazing treatments, instream flow structures, willow management, and beaver activities are being used to manipulate stream flow velocity. Reduced velocity causes sediment deposition on channel banks where it is then stabilized by vegetation. Encroachment of stream banks should cause aggradation of stream channels, promote water spreading over or under dry flood plains, and increase area and health of riparian zones.

Key words: Nonpoint Pollution, Sediment, Riparian, Desert, Streams, Best Management Practices

INTRODUCTION

Riparian Zones

Riparian zones are wetlands supported by a high water table because of proximity to surface or subsurface water. They are characterized by distinct soils and plant communities with high productivity as well as species diversity. They normally occur as an ecotone between more xeric lowland communities and aquatic ecosystems (Brown 1978). Strictly speaking, riparian zones are dominated by plant species that depend on sub-irrigation throughout their growing season. When sub-irrigation does not occur riparian communities are replaced by more xerophytic communties.

Importance

The riparian zone has a high level of use by a wide variety of interests as noted by Busby (1978), Johnson (1978), Tubbs (1980), Thomas et al. (1978), Haugen et al (1980), and numerous others. Apparent causes for concentration of multiple uses in riparian zones are the vegetation species diversity, productivity, and proximity to open water. High species diversity in the riparian zone is reported by Campbell and Green (1968), Brown et al. (1978), Ewel (1978), Haugen (1980), Speck (1981), and Kauffman et al. (1983a). Diversity is associated with zones of variation in soil moisture (Miller 1979). Higher plant biomass production due to increased soil moisture is described by Thomas (1978), from research by Minore (1970), Minore and Smith (1971), and Miller (1979). Flowing water wetlands have higher rates of gross primary productivity and net biomass production than still water wetlands (Brown et al. 1978).

Riparian areas provide a buffering ecotone between the aquatic and dryer lowland and upland ecosystems. The beneficial effect of this ectone to stream condition, water quality, and fish populations has been reported by Odum (1978), Jahn (1978), and Haugen et al. (1980). Water quantity and quality as well as physical features of streams also depend on characteristics of the watersheds drained by each stream (Jahn 1978). Characterization of watersheds and riparian zones is therefore of importance in evaluating user impacts.

User Impacts

A complex interrelationship exists between soils, water, and riparian vegetation (Miller 1979). Impacts compacting the soils, such as livestock grazing and recreation, may reduce infiltration, lower water tables, and increase erosion (Peterson 1950, Schmidly and Ditton 1978, Meehan and Platts 1978, and Thomas 1978). Methods to increase water harvest on channelized streams will also affect the soil moisture regime and vegetation (Peterson 1950, Jahn 1978, Campbell 1970, McCall and Knox 1978) through dewatering the riparian zone. Such impacts as loss of plant vigor and changes in species composition in plant communities result from livestock grazing and trampling, roads, mining, and recreation (Peterson 1950, Gunderson 1968, Martin 1978, Thomas 1978, Schmidly and Ditton 1978, and Speck 1981). The loss of capacity to bind

soil from a decrease in vegetation increases the potential for erosion, loss of stream channel stability, and results in sedimentation within channels.

Peterson (1950), Busby (1978), Meehan and Platts (1978), Thomas (1978), Roath (1982), Kauffman et al. (1983b), and Kauffman and Krueger (1984) indicate an increase in erosion has resulted from livestock grazing, water development, and recreational use of riparian zones. Channel instability indicated by sloughed stream banks results in wider channels, more shallow water depths and altered flow characteristics (Platts 1981). Gunderson (1968), Duff (1978), Platts (1981) and Kauffman et al (1984b) reported channel widening due to impacts of livestock grazing. Haugen et al. (1980) included mining, water development, and timber harvest as user impacts that altered stream channels. Sediment from erosion results in a loss of habitat for aquatic organisms such as invertebrates, reduces fish production, and alters water quality (Thomas 1978, Duff 1978, Gunderson 1968, and Platts 1981). Thus impacts to streams caused by users of riparian zones have been well documented.

This paper addresses two research programs designed to improve riparian zones of an ephemeral and a perennial desert steppe stream by 1) promoting water storage and vegetation yield season long, 2) controlling nonpoint source pollution and, 3) advancing the state of knowledge for management of stream side ecosystems subjected to user pressure.

Theory for Reclamation of Cold Desert Steppe Streams

Reclamation of degraded streams should be initiated on hydraulic units where treatments will most likely reverse impacted channel conditions. Criteria for selecting hydraulic units should allow experimental replication on similar stream reaches or between streams. Our study sites are selected based on seven geomorphologic and hydraulic criteria. The criteria used are as follows: 1) the stream reach gradient must be low, 2) the stream reach should have a mature meander pattern, 3) the channel morphology of the stream should have a developed or developing flood plain, 4) stream flow should be losing to surrounding alluvium during high flow, 5) channel damming within the selected stream reach should cause maximum response upstream, 6) a high potential exists for water spreading to adjoining flood plains as reclamation progresses and, 7) the stream reach selected should be typical of those found and similarly managed for grazing in the immediate area.

Stream Gradient

The success or failure of reclaimation of degraded stream channels and adjoining riparian zones depends on deposition of sediment on stream banks or bottoms if dams are constructed. Channels are usually wide compared to mature natural conditions. Streams with high sediment load usually cross areas with steep gradients but lose sediment in areas of low gradient because of a drop in flow velocity. If one wishes to achieve maximum deposition of sediment for reclamation of stream

channels, reclamation should start on reaches with the lowest gradient along the stream course.

Stream Maturity

Mature stream reaches are those which meander from valley wall to valley wall through a developed flood plain of deposited alluvium. Maturity is associated with low stream gradient. Alluvium is present because of a decrease in flow velocity and meandering is a natural physical flow response across low gradient areas. Because of meandering, valley flood plain width is maximum. Meandering also increases the length of stream per lenght of valley floor. This increases the contact of water with banks and the length of time of exposure of water to a valley reach. Reclamation of degraded streams on reaches where meandering occurs should subject more land area to flooding and promote riparian habitat.

Flood Plain

Flood plains along a stream course reflect response of flow regime to channel width. Degraded desert streams that have been downcut tend to widen because of wet-dry cycles that cause bank slumping. Piping and bank erosion also add sediment to the channel bottom. Slumped banks and eroded sediment deposited on the channel bottom are flushed downstream during periods of high flow. This flushed sediment is often more than is contributed to stream flow by scouring of channel banks (Leopold et al. 1964) As downcut channels widen, streams during low flow begin to meander within the confinement of high banks and deposition of sediment occurs on low banks.

Stream dynamics and channel morphology related to sediment transport and delivery are generally reviewed in Leopold et al. (1964), Morisowa (1968), Graf (1971), ASCE Task Committee (1975) and Simons and Senturk (1977). Hydraulics of sediment transport in stream flow is further reviewed by Parker (1978a,b). Andrews (1983) studied bedload transport and deposition within a stream in western Wyoming and clarified the relationship between flow regime and bank deposition of sediment. Shumm (1963) was able to show trapping of fine sediments by vegetation on banks in eastern Colorado. Andrews (1983) work related to Shumm's by showing that build up of bank sediment occurs before stream bottoms rise. During periods of low flow sediment is first deposited on banks. Built up banks cause the channel to become narrower. Bed load is then moved dowstream by increased flow velocity because of the new narrow channel. If aboveground and belowground biomass does not solidify the new bank sediment before the next high flow event, bank accumulation will be transported downstream as suspended sediment or bed load. This response to high flow maintains the channel at a given width and depth corresponding to a yearly average flow regime. The width and depth of a stable channel however corresponds to the mean average bank full flow (Andrews 1983). Stable banks are not eroded because high flows overtopping stable banks and dissipated over a well vegetated flood plain reduces flow velocity. This hydraulic process is explained in part by Parker (1978a,b). Andrews (1983) points out that although a stream maintains an average

width, it will move laterally from year to year thus fitting Leopold and Lanbein's (1966) description of meandering streams. The presence of undercut banks along stable stream systems are evidence of lateral movement of meanders.

Bank first sediment deposition, vegetation encroachment into the new bank sediments, and reduced flow velocity by dewatering or ponding are important points to consider when reclaiming degraded streams. Vegetation which is allowed to become well established within newly deposited sediment on banks creates channel bank stability and a roughness factor to decrease flow velocity during flood stages. Reduced flow velocity will cause flood waters laden with sediment to deposit some of its load. Sediment dropped on vegetated banks will be further stabilized by regrowth of plants. This process will continue to narrow a channel until it adjusts to a mean width sufficient to meet the annual flow regime and is often observed in desert streams within the confinement of high banks. When this happens the channel is mature with a developed and stable flood plain.

Reclamation of downcut streams in areas where flood plains are developed and well vegetated can most effectively be accomplished in low gradient stream reaches. Meander patterns can further be broken down into straight and meander reaches. Manipulation of stream flow by daming to decrease flow velocity should be implemented on straight stream reaches. This reduces the chance of losing dams because of channel lateral movement along the meander's concave curve. In addition, belowground and aboveground biomass offer resistance to erosion and flow velocity when instream flow structures are utilized to create ponding and entrapment of bed load.

Losing Streams

Stream channel transmission loss of water downstream during high periods of flow to surrounding alluvium should cause a decrease in flow velocity and consequently a loss in sediment. Sediment deposition should be maximized in mature stream reaches in areas of low gradient where meandering increases time in travel of flow downstream. Transmission loss of stream flow in desert streams has been shown by Lane (1970). Loss of stream flow should be maximum near the mouth of a drainage basin or in larger basins of losing desert streams because of increased travel time of water during any one runoff event (Glymph and Holtan 1969). Location of manipulative practices to reclaim degraded streams based on loss of flow and sediment deposition should therefore be placed to maximize water travel time after considering the previously discussed criteria.

Damming

Damming by instream flow structures, like check dams or trash collectors, and biological damming by beaver or constrictive channel dams caused by encroaching banks and riparian zones may cause: 1) reduced flow velocity, 2) stable bed load and 3) storage of water in banks proximal to the dam. Heede (1978 and 1982) discusses reclamation of gullies by raising a local base level of an ephemeral stream reach to decrease gradient slopes upstream using a variety of check dams. The

lower gradient reduces sediment transport. Deposition occurs upstream in a wedge shape until reaching an elevation change upstream equal to the elevation of the dam. Following Heede's 1978 and 1982 research, dams should be placed downstream just above a tributary junction. To achieve restoration of riparian habitat however, the dam should also be located on a stream reach having a low gradient where meandering occurs and a stable floodplain exists. The dam will then cause bank deposition of sediment and maximum channel filling within the upstream drainage network.

Water Spreading

Water spreading across flood plains formerly left dry because of stream downcutting and occupied by pseudoriparian vegetation decreases flow velocity. Shrubby type vegetation may cause sediment to be deposited in rows downstream from woody plants parallel to the main stream channel. These rows of deposited sediment may form many smaller channels (braiding) which may carry water during high flow events. Braiding may also occur in wide channels when vegetation establishes on the channel bottom and sediment deposits downstream. Once braided channels become stable because of vegetation, multiple check dams can be established to maximize water spreading over larger areas.

Water spreading increases riparian zone area by promoting return flow as runoff and ground water to the main channel. Where degraded streams have downcut, widened, and formed a mature flood plain within high banks, water spreading is limited. However, constrictive damming by encroached riparian zones into a main channel, instream flow structures, and beaver dams often cause peak flows to flood over high banks upstream thereby creating desired riparian habitat. Wider alluvial valley floors are desirable to maximize water spreading.

Grazing Response

Knowing how to maintain riparian zones at desired conditions when used by livestock and wildlife necessitates accumulating verifiable information on: 1) how behavior of animals using uplands and riparian zones under any one grazing management strategy vary, 2) which season of use best maintains riparian habitat, and 3) how different stocking rates affect vegetation and stable banks. Information gathered about these three questions provide input for developing sound strategies to manage livestock in cold desert regions where the area of riparian zones is small in comparison to surrounding uplands.

Grazing by livestock on degraded downcut streams that have a mature flood plain confined within high banks may be beneficial. High banks channelize stream high flow causing an increase in velocity and prevent water spreading. Hoof action by livestock may round or flatten high banks by trampling. Hoof imprinting may cause depression storage and increase infiltration.

Approach Taken for Reclaiming Cold Desert Steppe Streams

Two field research facilities were selected to study reclamation of degraded desert streams and associated riparian zones. One is located

on an ephemeral stream (15 Mile Creek) in North Central Wyoming near the town of Worland. The second, a perennial stream (Muddy Creek), is located in south central Wyoming between the towns of Rawlins and Baggs. Both study areas were selected by following the basic criteria previously discussed. High concentrations of sediment are present in flow of both streams.

Ephemeral Channels-Facilities

The 15 Mile Creek facility is divided into 5 pastures. Pasture 1, downstream, is going to be used for the study of watershed improvement practices designed to deposit sediment and improve riparian habitat. Pastures 2, 3, and 4, located upstream are being used to study the effect of spring, summer, and fall grazing on stream channel morphology and vegetation.

Pasture 5 serves as a control and is located upstream from the other four pastures. Livestock behavior studies are being conducted inside grazed pastures as well as outside to document seasonally selective use of upland and riparian habitats. Livestock use studies in a mature riparian zone and utilization of associated riparian plants is being conducted in two separate pastures placed on a second stream near to the 15 Mile Creek facility. Utilization levels on plant species in relation to season of use and stocking rate are being determined.

Ephemeral Channel Measurements

Change in stream channel morphology is being monitored using cross section techniques. Permanent cross sections placed on meander and straight stream reaches are located downstream, within, and above the pasture facility. Changes in channel depth and in cross-sectional area are being determined.

Comparative vegetation analysis are being conducted upstream and downstream from the pasture system. Within the pastures, permanent vegetation transects located at channel cross section locations are established to monitor change in production and shrub density. Vegetation transects extend from uplands to within 50' of the low channel bank, from 50' to the low bank, from the top of the low bank to the interim channel, and along the edge of the interim channel. Degree of encroachment of vegetation (above and belowground) across the interim channel is also being measured. Core samples on low banks are also being taken to determine belowground biomass.

Monthly monitoring of soil moisture using neutron scattering techniques follows vegetation transects from upland to low bank edge in all pastures. In addition, soil moisture data is being collected across meanders and straight sections. Water table depth is recorded and precipitation monitored continuously during spring, summer, and fall. Substrates on cross-sections have been core-sampled and analyzed for particle size distribution and textural classification. We intend to use these data to characterize sediment deposition on banks and with belowground biomass examine bank resistivity to erosion.

Willow propogation and planting trials have been initiated within the interim channel of the downstream pasture (1). Survival relative to planting depth and irrigated-nonirrigated treatments are being evaluated. The purpose of the planting trial is to study the feasibility of forming a channel plug of willows to filter sediment from flow and raise bed load level.

Perennial Stream-Facilities

The Muddy Creek facility is divided into six hydraulic stream reach units including areas above and below designated treatment areas. The first hydraulic unit upstream demonstrates stream headcut problems encroaching into riparian zones formed by beaver dams and good riparian zone habitat management. Because of waterspreading and constrictive damming, this riparian zone has caused sediment deposition and maturation of the stream side zone upstream. Existing differences in livestock grazing management within this unit illustrate negative impacts of inappropriate cattle grazing on willow communities and beaver damming activity as well as the subsequent deterioration in bank stability.

Downstream, (Unit 2) below active headcutting, channelization has occurred but meandering flood plains are confined within high banks. Willows are present but limited and need rest from grazing to stabilize flood plains. In addition, recent sediment deposited should be planted with willows to encourage beaver damming as well as provide potential sites for installing instream flow structures to raise water tables, cause deposition of sediment on flood plains, and stabilize bed load. This channelized stream reach, hydraulic unit number 2, is planned as a research area for documenting the hydrologic response affiliated with reclamation of cold desert perennial streams.

Hydrologic data will be collected for three years to develop a water balance of this degraded stream and riparian zone. Three more years are also planned to stabilize flood plains using willows. Four additional years will be devoted to using instream flow structures and beaver to raise the water table and bed load profile over a distance of approximately three miles and six feet change in elevation. Pertinent questions to be answered are:

1) How much water is stored underground by natural damming and riparian zone improvement practices?
2) What change in downstream flow regime occurs in the form of prolonged release of water season long?
3) What water is lost to downstream users during periods of need and right?
4) What water balance and bank stability response occurs because of watershed improvement practices like brush control (burning, spraying) and range fertilization?
5) What is the significance of good riparian zone management to abatement of nonpoint and point source pollution?
6) What water balance, bank stability, and vegetation response occurs because of livestock and wildlife grazing on improved riparian zones because of watershed improvement practices?

7) What are the up and downstream effects of riparian zone and watershed improvement practices?
8) What are the economic costs and benefits of improving degraded riparian zones?

Research methodology initiated after three years of hydrologic monitoring of this three mile stream reach will be based on pilot research being conducted downstream. Water leaving the designated research facility (Unit 2) flows into the third hydraulic unit. The downstream portion of hydraulic unit three, approximately three miles of a 6 mile stream reach, is characterized by filling with sediment caused by waterspreading and beaver damming downstream in the fourth hydraulic unit.

Hydraulic unit number three is in various stages of plant succession and dissected by braided stream channels caused by flooded shrub vegetation and deposited sediment. Mud flats are evident. Willows are virtually absent, perhaps caused by historic early spring livestock grazing. This reach has been fenced to manipulate spring grazing and promote willow establishment. This unit will serve as an area for initiating riparian watershed improvement practices on a response area caused by good riparian zone management downstream. Grass and willow planting trials will be initiated on exposed sediments.

Hydraulic unit number four is a stream reach approximately ten miles long and has been managed to maintain riparian habitat. The main stream has one diversion to spread water and return flow enters the braided main channel. Channels are dammed by beaver. Grass has been established to stablize sediment and provide livestock forage. Willows are prominent and grazing by livestock occurs during late fall and winter.

Next downstream, hydraulic unit number five, approximately three miles long, is downcut and has an immature flood plain constrained between high banks. This unit may reflect response to flow that has been stripped of sediment by the improved riparian zone upstream. Flow in this unit, because of less sediment, should have more power to erode the existing channel. No work is currently planned in this area.

Hydraulic unit number six changes from a stream channel with an immature flood plain to a channel with mature floodplains abruptly below the confluence with an ephemeral stream. Perhaps this change in flood plain development is caused by an influx of sediment to Muddy Creek during flow events from the ephemeral channel. Willow growth on the floodplains of hydraulic unit number six grades from light just below the confluence of the ephemeral stream channel to heavy downstream. This entire unit (three miles in length) has been fenced to demonstrate the use trash collectors as a tool for reclaiming riparian zones. Fencing will allow control of livestock grazing trials on streamside vegetation during different grazing seasons and a wildlife exclosure will provide information on effects of livestock and wildlife vegetation on riparian zone vegetation and channel morphology. Grazing trials have begun coincident with installation of trash catchers and fences.

Trash collectors have been installed on straight stream reaches within three pastures to be used for livestock grazing during spring, summer, and fall followed by winter wildlife use. They have also been installed in the wildlife exclosure and in a pasture downstream adjoining the wildlife exclosure where wildlife grazing but no livestock grazing will occur. A control reach is downstream from all fenced areas and has no trash collectors. This entire three mile stream reach is now in a ponded condition during low flow due to trash collectors. Beaver have now used three trash collectors for constructing dams and winter food storage.

Hydraulic unit number six was selected based on the criteria discussed previously for testing trash collectors as a tool to reclaim degraded streams and riparian zones. It is hypothesized these structures will: 1) promote bank storage of water, 2) cause deposition of sediment on channel flood plains thereby causing channel narrowing, 3) stablize and raise bed load levels and 4) cause upstream flooding because of constrictive damming of high flow events and thereby sediment deposition on upstream banks. Willow establishment caused by increased height of watertables and water spreading should promote beaver damming. This would eliminate the need for further installation of trash collectors in hydraulic unit number 6. Planting of willows on new floodplains in hydraulic unit number five would promote channel stabilization and sustain trash collector improvement practices upstream if the hypothesized effects in unit six do occur.

Perennial Stream Measurements

Measurements recorded follow those discussed for the 15 Mile Creek study. Changes in stream channel morphology will be evaluated using the cross section technique. Species composition, production, and utilization of the channel, flood plains and riparian zone vegetaion are being monitored. Bank-stream interflow and flood plain soil moisture in hydraulic unit number 6 is ready for monitoring to evaluate change caused by trash collectors. Willow propogation in planting trials will be evaluated in hydraulic units numbers 1, 3, and 6. Trash collector stability and new installations will be evaluated for collecting bedload, causing bank flooding upstream, and any rise in water tables. Installation of equipment to construct a water balance is planned for hydraulic units number 2 and 2 through 6 inclusive.

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MANAGEMENT ALTERNATIVES FOR URBAN STORMWATER

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A. Introduction

This paper is a brief example of how suitable stormwater management alternatives can be designed using available monitoring information. Milwaukee area urban runoff has been extensively studied by the Wisconsin Department of Natural Resources, the U.S. Geological Survey and the Southeastern Wisconsin Regional Planning Commission (SEWRPC), among others, for about the past ten years. Two major research projects were conducted for the International Joint Commission (IJC) on Great Lakes Research (Bannerman, et al, 1979) and for the Nationwide Urban Runoff Program (NURP) for the U.S. EPA (Bannerman, et al, 1983). The IJC project involved detailed urban runoff monitoring of many single land use and mixed land use sites, while the NURP project examined urban runoff at eight single land use sites for two years. This information, in conjunction with other regional and nationwide urban runoff characterization and control data, was recently used by the Nonpoint Source and Land Management Section of the DNR to prepare a Construction Site and Stormwater Management Plan and Model Ordinance (Pitt 1985). This paper uses this summarized information in several related examples to show how stormwater management controls can be designed to meet both flood control and urban runoff quality objectives.

B. Required Inventory Information

Before an evaluation of suitable control measures can be made, certain information about the study area must be obtained. This inventory information pertains to the importance of the various source areas in contributing runoff flow or pollutants to the outfall. Land cover information is needed to describe the percentage of the study area covered by various surfaces, such as streets, roofs, landscaping and parking areas. This information is usually obtained by direct measurement from aerial photographs. Additional information is needed to describe the aspects of these areas that modify their pollutant or flow generating capabilities. Roof drain connections, pavement texture and condition and the type of drainage system are examples. This information requires field investigations of representative areas for each land use subcategory. Table 1 is an example of land cover information

for typical medium density residential and industrial areas recently studied (Pitt and McLean 1985). Landscaping comprises the most common surface cover in the single family residential area, while rooftops and paved parking and storage areas are the most common surfaces in the industrial area. These major differences can result in significantly different runoff characteristics.

Figures 1 through 4 are based on extensive runoff monitoring of these two study areas and show how the different areas contributed to the outfall total solids and lead yields for different rain volumes (Pitt and McLean 1985). Rain has a much greater effect on the relative contributions for the residential area. The larger amount of impervious surfaces in the industrial area reduces the variations observed. Table 2 summarizes this information for a selection of pollutants for large rains (fairly stable source contributions occur with large rains because of the decreased importance of pervious area rain infiltration). Roof and paved parking/storage area runoff contribute most of the runoff volume and most of many of the pollutants in industrial areas. Flow and pollutant contributions are much more diverse in residential areas. This information is used in the following example to estimate the effectiveness of the different control measures in reducing runoff flows and pollutant contributions.

C. Infiltration and Detention Basin Combined Effects in Areas Having Poor Soil Percolation

1. Hypothetical Site Characteristics.

A series of calculations were made to investigate possible alternative flow and pollutant reduction options available for a hypothetical study area. Options investigated included dry and wet detention basins, roof and parking/storage area infiltration, and grass waterway drainages. The benefits examined included flow volume and rate reductions along with pollutant concentration and yield reductions.

These calculations are intended to generally describe the benefits of alternative urban runoff controls available. The calculation procedures, along with suitable design guidelines, are described in the recently prepared Stormwater Management Plan (Manual of Practice) (Pitt 1985). As stated above, these calculations are based on many field monitoring projects, including several conducted during the past ten years in the Milwaukee and Menomonee River watersheds (Bannerman, et al, 1979 and 1983).

2. Flow Estimates.

Table 3 shows the land use areas in the example watershed, divided by hypothetical soil percolation characteristics. Table 4 shows the calculations used to

estimate the average urban runoff flow rates for a variety of large storm conditions. These flow rate estimates are based on the monitoring results of several hundred Milwaukee area urban runoff events studied by Bannerman, et al (1979 and 1983). Statistical analyses of these runoff data also indicated that peak runoff flow rates for large urban runoff events are about 6.5 times the average flow rates. The peak flow rates generally only occur for several minutes during the event.

3. Dry Detention Basin Design.

Dry detention basin storage requirements were estimated for the different land uses. The basins should drain within a few days to enable control of storms likely to follow in a short time. For the critical 100 year storm, basin outflow rates of between 100 and 200 cfs would result in reasonable drainage times. Table 5 shows required basin sizes for various outflow rates. Total basin areas of about three percent of the watershed area may be required to control the critical storm. Alternative basin criteria can also be used to determine the required basin sizes.

Alternative detention basin design objectives and associated required basin sizes are summarized on Table 6. The minimum basin size would reduce the expected peak flow rate to the assumed channel carrying capacity and would require about 0.3 percent of the watershed area. The maximum basin size is designed to allow basin drainage in three days to obtain a maximum flow rate reduction and would require about three percent of the watershed area. A reasonable basin objective is intermediate to these two sizes and would reduce the peak flow rates to the average flow rates. A basin size of about 0.5 percent of the watershed area would be required. Even though these dry detention basins would substantially reduce the peak flow rates, they do not change the total volume of water or the amounts of pollutants entering the creek and may not benefit flood control or water quality in downstream receiving waters.

4. Wet Detention Basin Design.

Many monitoring projects have demonstrated substantial water quality benefits associated with well designed wet detention basins (EPA 1983). Wet basins can be used in conjunction with dry basins (and/or other urban runoff control devices) to help meet both flood control and water quality objectives. Wet basins also do not reduce the total volume of water discharged to the receiving water, but can remove much of the settleable solids.

High control levels of pollutants in wet basins are only reasonably possible for the more frequent rains. Water quality is not usually of much concern during very large infrequent events, such as the 100-year storm. The most cost-effective benefits can be relealized during common, smaller events. A wet basin about 0.3 percent of the watershed area, having a wet depth of about four feet and a

freeboard height of about 12 feet would produce about 75 percent solids control. This freeboard requirement is needed to reduce the wet pond outflow velocity to desired values and may be larger than desired. It would be possible to reduce this freeboard requirement with the use of a surge dry basin or upland controls.

5. Upland Infiltration Design.

Upland infiltration can be used to reduce the flow rates to downstream detention basins, but more importantly, they reduce the total volume of water and amounts of pollutants discharged. Upland infiltration devices recharge the local groundwater and increase critical low flows during summer dry months, further benefiting the receiving water. Care must be taken to protect significant groundwaters, of course. Upland infiltration can also be used only in areas with appropriate soil percolation conditions. The soils in the hypothetical watershed are of two percolation classes: poorly drained and moderatly well to well drained. The following calculations only consider the application of upland infiltration in the better drained soils.

Table 7 shows the maximum effects that roof and parking/storage area infiltration may have on flow rate and volume reductions for several large rains in residential and industrial areas. These reductions can be substantial, especially for industrial areas where about 80 percent of the water originates from roofs and parking/storage areas. These improvements must be reduced because of the poor soils in the assumed study area. Roof runoff infiltration performance in the residential area would be reduced to about ten percent, roof runoff infiltration performance in the industrial area would be reduced to about 15 percent, parking/storage area infiltration performance in the industrial area would be reduced to about five percent and roof plus parking/storage area runoff infiltration performance in the industrial area would be reduced to about 20 percent.

6. Grass Waterway Design.

Grass waterways may also significantly reduce flow volumes through infiltration. Table 8 shows the effects of grass drainages, instead of underground sewerage. Flow rates and volumes during the large 100 year storm may be reduced by as much as 20 percent by these devices. Table 9 shows the total expected watershed flow rates using these controls.

7. Revised Dry Detention Basin Design Considering Upland Infiltration.

Table 10 shows the revised dry basin storage requirements with these upland controls. The area requirements have been slightly reduced, to about 0.4 percent for the 100 year storm and for peak shaving, but the outflow rates have been reduced substantially for many conditions. The revised wet basin area and freeboard

requirements are slightly reduced (down to about 10 feet over about 0.3 percent of the watershed area).

D. Summary.

Table 10 presents a summary of the expected water quality and quantity improvements for the various alternatives investigated. No water quality or water volume improvements will occur for the dry basins alone, but substantial reductions in flow rates, water volume, and pollutant yields and concentrations will likely occur for a combination of control devices.

In conclusion, a reasonable amount of the watershed land area can be used to effectively control urban runoff quantity and quality. Dry basins comprising about 0.4 percent of the watershed area (about six feet deep) can be used with wet basins requiring about 0.3 percent of the area (having a wet pond depth of about four feet and a freeboard elevation of about ten feet) and upland infiltration devices (in areas having suitable soils) to achieve significant flow and pollutant reductions. Similar control devices have been monitored and proven effective during many urban runoff research projects. Special restricting conditions (such as poor percolating soils) can be considered in the design of these control devices.

E. References

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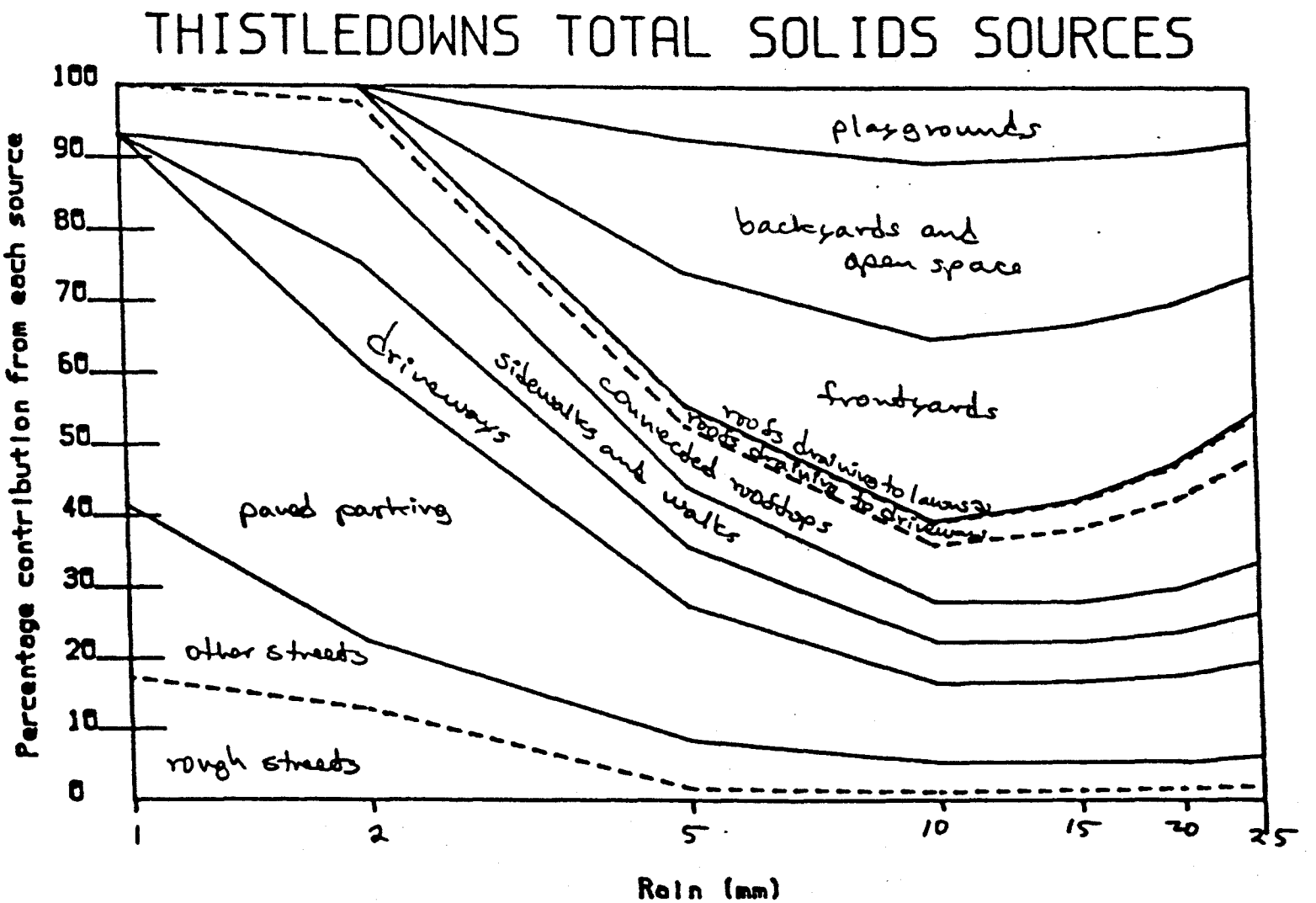
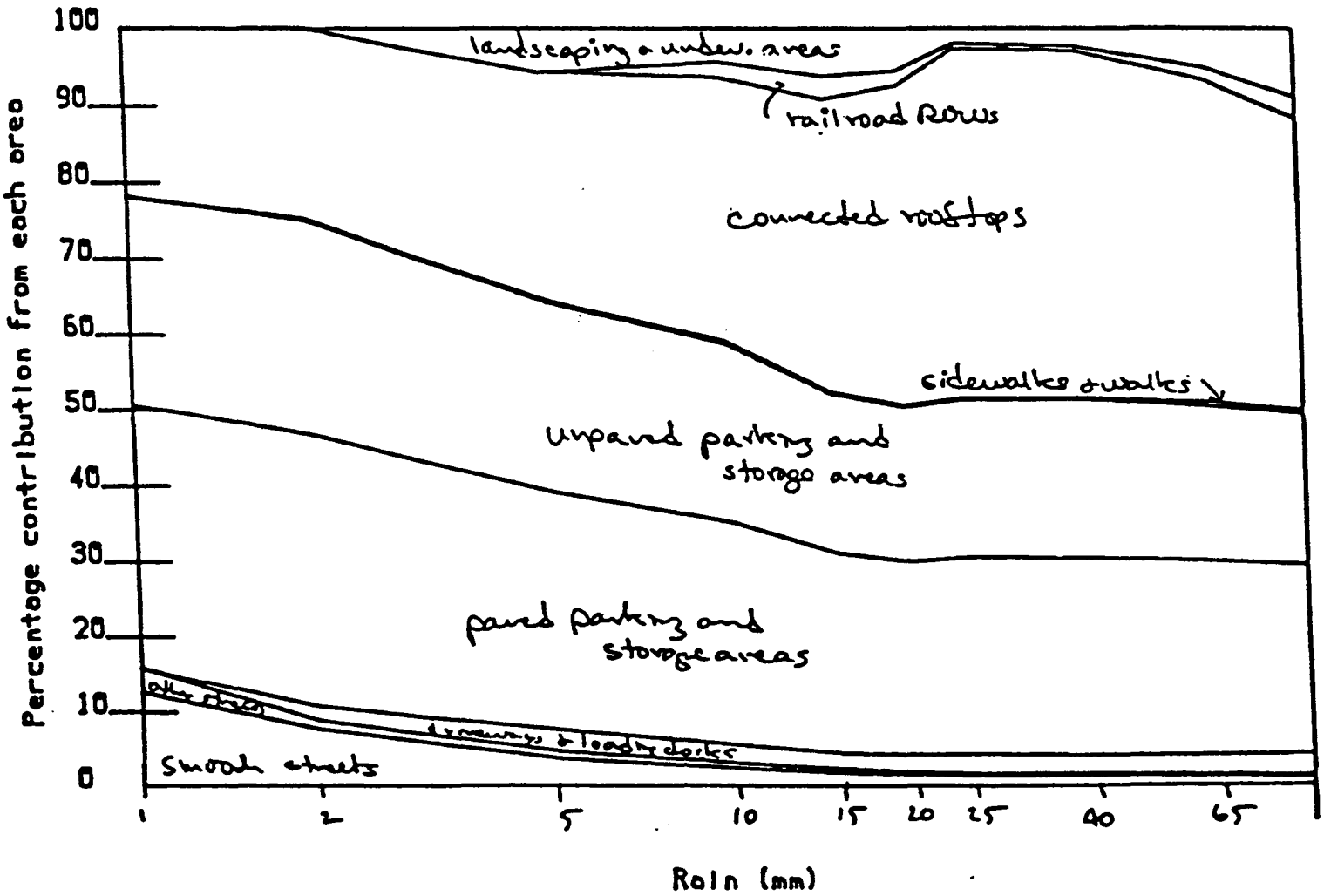
Figure 1



Source: Pitt and McLean 1985

Figure 2

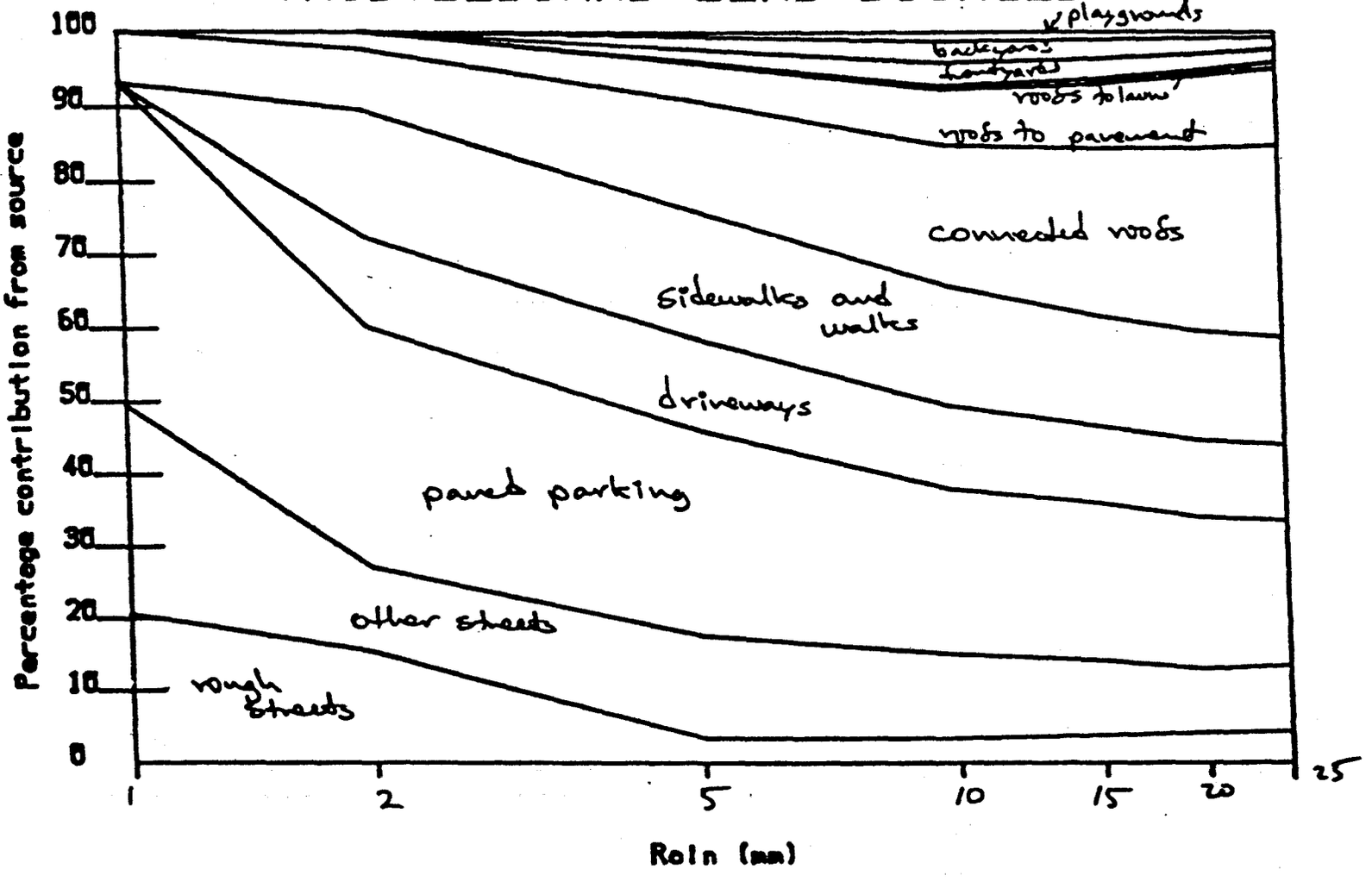
EMERY TOTAL SOLIDS SOURCES



Source: Pitt and McLean 1985

Figure 3

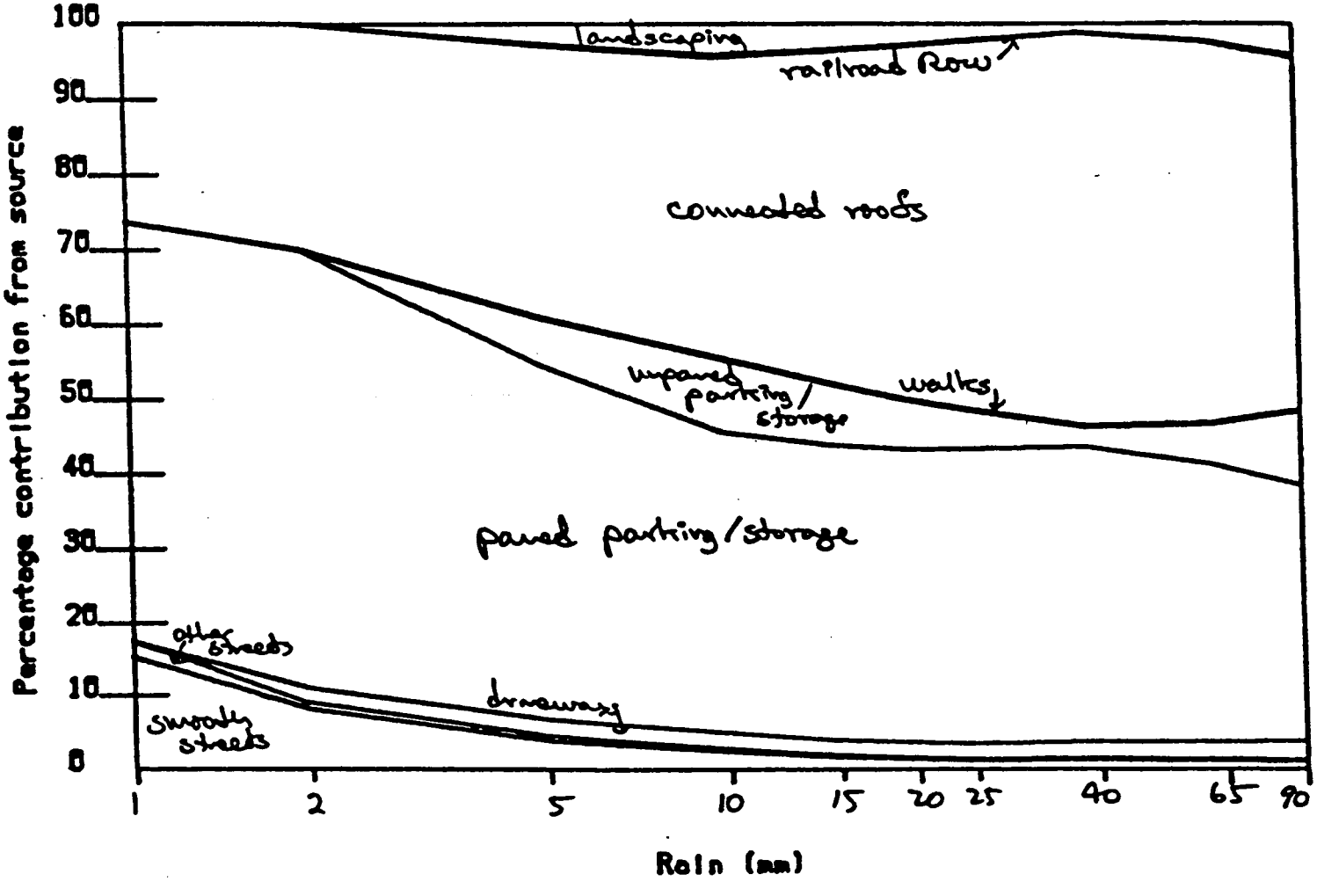
THISTLEDOWNS LEAD SOURCES



Source: Pitt and McLean [redacted]5

Figure 4

EMERY LEAD SOURCES



Source: Pitt and McLean 1985

Table 1. Industrial and Medium Density Residential Land Covers (%)

Land Cover	Industrial	Residential
Streets	5.6%	11.4%
Swales	0	3.1
Grass strips	0	9.2
Concrete walks	0.4	3.7
Front landscaping	10.2	16.3
Paved driveways	2.2	6.4
Paved parking/storage	19.9	0.1
Unpaved parking/storage	15.4	0
Sidewalks	0.1	0.7
Back landscaping	0	24.5
Undeveloped	12.4	0
Railroad	2.6	0
Connected roofs	31.1	4.9
Roofs draining to driveways	0	8.7
Roofs draining to lawns	0	10.9
Total	99.9	99.7
Total Impervious	47.4	36.4
Total Pervious	30.4	47.6
Total Disconnected Impervious	22.2	16.0

Source: Pitt and McLean 1985

Table 2. Sources of Pollutants During Large Events

Industrial Areas:

Constituent	Streets	Walks	Front Lands.	Drive-ways	Paved Park.	Unpaved Park.	Open Space	Connected Roofs
Total solids	2%	1%	2%	3%	26&	20%	10%	36%
Suspended solids	3	1	10	4	42	20	14	6
Phosphorus	1	1	10	2	20	8	10	48
Kjeldahl N	1	1	6	2	30	3	5	52
Phenolics	2	1	2	5	58	8	1	23
COD	1	1	2	3	37	8	3	45
Fecal colif.	2	3	6	2	21	25	5	36
Copper	3	1	6	4	50	12	4	20
Lead	2	1	3	3	35	12	2	42
Zinc	1	1-	2	4	51	8	2	32
Flow	9	1-	4	2	30	3	3	49
Area	6	1-	10	2	21	15	15	31

Mixed Residential Area:	Streets	Walks	Front Lands.	Drive-ways	Paved Park.	Open/ Rear Lands.	Connect. Roofs	Roofs to Drive.	Roofs to Lawns
Total solids	6	6	20	8	14	26	14	6	1-
Suspended solids	8	4	25	8	18	32	3	2	1-
Phosphorus	3	6	24	3	8	29	20	6	1
Kjeldahl N	8	24	3	2	6	6	36	13	2
Phenolics	3	17	1-	2	4	1-	14	56	4
COD	8	27	4	2	3	7	34	13	2
Fecal Colif.	8	59	7	1	2	14	6	3	1-
Copper	14	4	4	14	26	6	21	10	1
Lead	12	12	2	10	22	3	26	12	1
Zinc	3	5	1-	2	5	1-	59	24	2
Flow	23	3	8	6	8	15	19	9	2
Area	10	4	23	5	5	31	8	7	7

Source: Pitt and McLean 1985

Table 3. Example Land Use by Soil Type (acres)

	Poorly Drained Soil	Well to Moderately Well Drained	Total
Residential	1125	1445	2570
Industrial	435	235	670
Commercial and Institutional	110	110	220
Open Space	125	85	210
Total	1795	1875	3670

Table 4. Average Flow by Land Use (cubic feet per second)

Return Freq. (yrs)	Rain Dur. (hrs)	Total Rain (in.)	Rain Int. (in/hr)	Residential	Institutional, Commercial, or Industrial	Open Space
1	0.5	0.9	1.80	0.83	1.1	0.06
1	1	1.1	1.10	0.46	0.59	0.04
1	2	1.3	0.65	0.27	0.36	0.03
1	3	1.5	0.50	0.23	0.30	0.03
1	6	1.7	0.28	0.14	0.18	0.01
1	12	2.0	0.17	0.081	0.10	0.01
1	24	2.3	0.10	0.045	0.056	0.01
10	24	3.9	0.16	0.083	0.10	0.02
100	24	5.5	0.23	0.13	0.15	0.05

Table 5. Alternative Dry Basin Designs for Hypothetical Example

1) Minimum design to reduce peak runoff to assumed channel capacity (1500 cfs):

Storm Freq. (24 hr)	Peak Inflow (cfs) (1)	Reqd. Storage (acre-ft)	Reqd. Surface (acres) (2)	Reqd. % of 3700 acre basin	Pond Drain (days)
1 yr	1100	0	0	0%	0
10	2000	21	3.5	0.1	1.3
100	3100	66	11	0.3	2.1

2) Maximum design to allow pond drainage in three days:

Storm Freq. (24hr)	Ave. Inflow (cfs)	Reqd. Outflow (cfs)	Reqd. 24 hr Storage (acre-ft)	Reqd. Surface (acres) (2)	Reqd. % of 3700 acre basin
1yr	170	57	220	37	1.0%
10	300	100	400	67	1.8
100	480	160	630	105	2.8

3) Reasonable design to reduce short-term peak to ave. flow (peak shaving):

Storm Freq. (24hr)	Peak Inflow (cfs) (1)	Ave. Inflow (cfs)	Reqd. 30-min Storage (acre-ft)	Reqd. Surface (acres) (2)	Reqd. % of 3700 acre basin
1 yr	1100	170	38	6.3	0.2%
10	2000	300	70	12	0.3
100	3100	480	108	18	0.5

Footnotes:

(1) Peak flow duration is assumed to be 30 minutes
(2) Dry pond depth is six feet

Table 6. Effects of Upland Infiltration on Runoff Rates (cfs per acre)

Residential Area

Storm Freq. (yrs)	Storm Dur. (hrs)	Ave. Flow With No Infilt.	Ave. Flow With Roof Disconnect.	Percentage Ave. Flow Reduction
1	0.5	0.83	0.67	19%
1	1	0.46	0.38	17
1	2	0.27	0.23	15
1	3	0.23	0.19	17
1	6	0.14	0.12	14
1	12	0.081	0.065	20
1	24	0.045	0.037	18
10	24	0.083	0.071	15
100	24	0.13	0.11	15

Industrial Area:

			Roof Dis.:		Parking Dis.:		Roof and Parking	
Storm Freq. (yrs)	Storm Dur. (hrs)	Ave. Flow With No Infilt.	New Ave. Flow	% Reduc.	New Ave. Flow	% Reduc.	New Ave. Flow	% Reduc.
1	0.5	1.1	0.54	51%	0.90	18%	0.37	66
1	1	0.59	0.30	49	0.50	15	0.21	64
1	2	0.36	0.19	47	0.31	14	0.13	64
1	3	0.30	0.15	50	0.25	17	0.11	63
1	6	0.18	0.094	48	0.15	17	0.064	64
1	12	0.10	0.053	47	0.088	12	0.038	62
1	24	0.056	0.031	45	0.048	14	0.022	61
10	24	0.10	0.060	40	0.090	10	0.048	52
100	24	0.15	0.098	35	0.13	13	0.080	47

Table 7. Effects of Swale Drainages on Runoff Rates (Percent)

Storm Freq. (yrs)	Storm Dur. (hrs.)	Percentage Flow Reducrion for: Residential (3 ft. swales)	Industrial (7 ft. swales)
1	0.5	15-	14-
1	1	15-	14-
1	2	15-	14-
1	3	15-	14-
1	6	19	14
1	12	27	23
1	24	50	35
10	24	33	18
100	24	23	14-

Table 8. New Runoff Volumes with Upland Infiltration (cfs)

Storm Freq. (yrs)	Storm Dur. (hrs.)	Ave. Flow With No Infilt.	Ave. Flow With Infilt.	Peak Flow With No Infilt.	Peak Flow With Infilt.	Percentage Flow Reduction
1	0.5	3100	2760	NA	NA	11%
1	1	1700	1520	NA	NA	11
1	2	1000	900	NA	NA	11
1	3	860	760	NA	NA	12
1	6	520	425	NA	NA	22
1	12	300	230	NA	NA	22
1	24	170	120	1100	750	32
10	24	300	240	2000	1600	21
100	24	480	380	3100	2500	21

Table 9. Alternative Dry Basin Designs for Hypothetical Example With Upland Infiltration

1) Minimum design to reduce peak runoff to assumed channel capacity (1500 cfs):

Storm Freq. (24 hr)	Peak Inflow (cfs) (1)	Reqd. Storage (acre-ft)	Reqd. Surface (acres) (2)	Reqd. % of 3700 acre basin	Pond Drain (days)	Pond Acres Saved
1 yr	750	0	0	0%	0	0
10	1600	4	0.7	0.02	1.1	2.8
100	2500	41	6.8	0.2	1.7	4.2

2) Maximum design to allow pond drainage in three days:

Storm Freq. (24hr)	Ave. Inflow (cfs)	Reqd. Outflow (cfs)	Reqd. 24 hr Storage (acre-ft)	Reqd. Surface (acres) (2)	Reqd. % of 3700 acre basin	Pond Acres Saved
1yr	115	38	150	25	0.7%	12
10	240	80	320	53	1.4	14
100	380	130	500	83	2.2	22

3) Reasonable design to reduce short-term peak to ave. flow (peak shaving):

Storm Freq. (24hr)	Peak Inflow (cfs) (1)	Ave. Inflow (cfs)	Reqd. 30-min Storage (acre-ft)	Reqd. Surface (acres) (2)	Reqd. % of 3700 acre basin	Pond Acres Saved
1 yr	750	115	26	4.4	0.1%	1.9
10	1600	240	56	9.4	0.25	2.6
100	2500	380	88	15	0.4	3.0

Footnotes:

(1) Peak flow duration is assumed to be 30 minutes
(2) Dry pond depth is six feet

Table 10. Pollutant Control Alternatives (Approximate Percentage Control at Outfall)

Control Option:	Total solids	Sus. solids	Phos.	TKN	Phen.	COD	Fecal colif.	Cu	Pb	Zn	Flow volume	Ave. flow rate	Peak flow rate	Pond area (%)
1 1yr	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1 10yr	0	0	0	0	0	0	0	0	0	0	0	0	25	0.1
1 100yr	0	0	0	0	0	0	0	0	0	0	0	0	52	0.3
2 1yr	0	0	0	0	0	0	0	0	0	0	0	67	95	1.0
2 10yr	0	0	0	0	0	0	0	0	0	0	0	67	95	1.8
2 100yr	0	0	0	0	0	0	0	0	0	0	0	67	95	2.8
3 1yr	0	0	0	0	0	0	0	0	0	0	0	0	85	0.2
3 10yr	0	0	0	0	0	0	0	0	0	0	0	0	85	0.3
3 100yr	0	0	0	0	0	0	0	0	0	0	0	0	85	0.5
4 1yr	90	60	55	35	50	55	70	70	70	55	0	12	86	0.7
5 1yr	75	50	45	30	40	45	60	60	60	45	0	0	74	0.3
6 1yr	34	28	34	44	36	44	30	37	40	54	32	32	32	0
6 10yr	25	19	25	35	27	35	21	28	31	45	20	20	20	0
6 100yr	20	14	20	30	22	30	16	23	26	40	19	19	19	0
7 1yr	34	28	34	44	36	44	30	37	40	54	32	32	0	0
7 10yr	25	19	25	35	27	35	21	28	31	45	21	21	6	0.02
7 100yr	20	14	20	30	22	30	16	23	26	40	21	21	40	0.2
8 1yr	34	28	34	44	36	44	30	37	40	54	32	32	97	0.7
8 10yr	25	19	25	35	27	35	21	28	31	45	21	21	96	1.4
8 100yr	20	14	20	30	22	30	16	23	26	40	21	21	96	2.2
9 1yr	34	28	34	44	36	44	30	37	40	54	32	32	90	0.1
9 10yr	25	19	25	35	27	35	21	28	31	45	21	21	88	0.25
9 100yr	20	14	20	30	22	30	16	23	26	40	21	21	88	0.4
10 1yr	84	64	64	61	62	69	72	75	76	75	32	32	90	0.7

Codes:

1. Minimum dry detention to meet channel carrying capacity
2. Maximum dry detention to drain in three days
3. Peak shaving dry basin
4. Wet basin for 90% solids control for 1 yr, 24 hr storm only
5. Wet basin for 75% solids control for 1 yr, 24 hr storm only
6. Upland infiltration and swales
7. Minimum dry basin plus upland infiltration and swales
8. Maximum dry basin plus upland infiltration and swales
9. Peak shaving dry basin plus upland infiltration and swales
10. Peak shaving dry basin plus small (75%) wet basin plus upland infiltration and swales

NONPOINT SOURCE POLLUTION OF THE VENICE LAGOON: PERSPECTIVES OF LONG-TERM ABATEMENT

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ABSTRACT

The paper adresses the major research topics and achievements related to nonpoint source pollution (NPSP) abatement of the Lagoon of Venice (Italy) aiming at source evaluation for the 2000 km^2 wide mainland and water quality forecasting of the receiving Lagoon.

The relevance of the work lies in the forthcoming need of setting control policies for the planned man-made flood barrages regulating the exchange between the sea and the Lagoon. In fact, control strategies should account for general health of the city of Venice, of the natural environment and of the macro- and micro-economics. Hydrological, chemical, hydraulic and physical aspects are worked out in a tentatively synoptic framework.

Detailed aspects are considered for the control of urban and agricultural NPS, for nutrient transports through the Lagoon system and for possibility of controlling algal blooms via Catastrophe Theory.

KEYWORDS

Non Point Source Pollution, Non Point Source Abatement, Water Quality Modeling, Tailoring Runoff Models to Receiving Water Models, Catastrophe Theory, Algal Blooms, Lagoon of Venice.

INTRODUCTION

The paper is centered on a discussion of the research progress on nutrient Non Point Source Pollution (NPSP) abatement of the Lagoon of Venice (Italy).

The first part of the paper summarizes the physical setting of the system and the main results of the hydrological, hydraulic and chemical research, aimed at selection of:
- source distribution and magnitude;
- coding of overall mass transports and balances;
- modeling aspects.

The second part puts forth some new ideas on forecasting of the eutrophication of the Lagoon, with special emphasis on modeling of algal blooms via application of combined mass transport models and Catastrophe Theory.

The third part briefly outlines some likely possibilities of long term abatement, given the social and environmental texture of the problem.

THE LAGOON SYSTEM

The Venice Lagoon (Figure 1) is a complex tidal embayment which consists of interconnected sloughs, marshes, often meandering natural and man-made channels. Unlike other similar tidal water bodies, the Lagoon system can be represented by one basic tidal type, that of well-mixed shallow waters. The salinity, in the whole reach, ranges from the salinity value of sea water near the three mouths, to that of nearly fresh water only along a confined belt at the tributaries outlet. Nevertheless salinity concentrations are mantained at values close to that of sea water for almost the entire length span. At the boundary, minor salinity stratification may occur in winter as a result of runoff during periods of heavy rainfall; otherwise the Lagoon water is isohaline due to low fresh water inflows and strong wind- and tide-induced mixing (Ghetti, 1979).

The Lagoon receives drainage waters from a large basin at its north-western reach, whereas the ratio between drainage area and the Lagoon surface adds up to about four (2000 Km^2 / 500 km^2). The system receives about 40% of its input of fresh water from pumped outlets draining a complex network of lowland areas, mainly devoted to modern agriculture, whereby large and steadily growing urban runoff pours into the system (Rinaldo, 1982; Cossu et al., 1984; Bendoricchio et al., 1984; Bendoricchio et al.,1985).

No major tributary flows into the Lagoon: in fact this unique

natural environment has been preserved through the centuries by diverting the estuaries of two rivers, Sile and Brenta (the works lasted from 1630 and 1850 ca.) once flowing within the reach of the system. The actual tributaries (Zero, Dese, Marzenego, Naviglio Brenta) peak at about 20÷60 m3/s without major solid transport occurences. Input of fresh water, besides the mentioned urban and agricultural components, is completed by other small tributaries (up to 27 overall delivery locations), sewage treatment plants and industrial discharges surroundings the Lagoon proper (Figure 1) (Rinaldo andRinaldo, 1983; Cossuet al., 1984).

Tidal propagation throughout the Lagoon is ruled by the hydraulics of deep channels (in some instances 10÷15 m deep) connected to broad shallow shoals alongside. The phase delay of tidal elevations to the forcing tide at the three marine openings (substantially altered with respect to natural conditions because of a man-made navigation channel lined through the mid Lagoon) ranges from about 1 hour at the city of Venice to 2,5 hours at the northern reach (Goldmann et al., 1975). The tides in the Lagoon do not show properties of standing waves, underlining the importance of both inertial and dissipative effects (Di Silvio and D'Alpaos, 1972).

Water circulation in the Venice Lagoon system is a combination of tide-driven and wind-driven currents, whereas baroclinic density-driven currents seem negligible (Ghetti,1979). Total water volumes stored within the system range from 160 to 510 millions of cubic meters (Mcum), where the dominating tidal exchange (for a gauged M2 tide) over half period may amount to 330 Mcum (through the Lido (130 Mcum), Malamocco (136 Mcum) and Chioggia (64 Mcum) mouths) (Ministero dei Lavori Pubblici, 1979).

The further lack of coastal currents in the outer Adriatic sea induces further uncertainty to environmental studies: it has been inferred, in fact, from some experimental evidence that some 15÷30% of the water volumes ebb-released by the system are not dispersed and reenter the mass balance of the Lagoon for a typical M2 tide (Cossu et al., 1984).

NPSP SOURCES

The task of predicting overall non-point sources (mainly nutrient for the actual purposes) into the receiving Lagoon is demanding due to the geomorphologic and hydraulic complexity of the drainage watershed both in its agricultural and urban components.

Urban runoff tied to about 1,000,000 inhabitants within the whole mainland, is noteworthy and mainly non-point in nature. Only 250,000 units are collected through sewage

systems and sewage treatment plants. The remnants load directly the drainage network and flow into the Lagoon. Furthermore, the inhabitants of the city of Venice and of the islands plus seasonal presences due to intense tourism amount to 100,000 further equivalent inhabitants, untreated at present stage. Hence, total urban NPS Nutrient load has been estimated in about 4,000 t/year of total Nitrogen and 1,500 t/year of Phosphorus (Bendoricchio et al.,1985).

As far as other NPSP components are concerned, the nature of the watersheds and the range of the land uses have called for a thorough model selection phase, in view also of the difficulties of calibration. The possibility of use of zero-order approximation (like empirical runoff-concentration curves) has been discarded, in view of their conceptual inadequacy (Zingales et al.,1984). In fact, significant differences between rising and falling stage discharge/concentration relations can be fully explained by quite simple mass transfer relationships referring to interactions with channel bottom materials and soil adsorption mechanisms. In this pattern, pollutant load/discharged total water volume relations seem fit to linear correlation scheems, as proved also by the results of smoothing of experimental data gathered in a testing watershed in environment of the Lagoon, for which two annual daily data collections had been carried out (Zingales et al., 1982).

An important result of such conceptual mass transfer and balance relations consists of the validation of the first-order approaches introducing reference weighted concentrations in the runoff, or loading functions for long-term, or event-based, time scale (Haith and Dougherty, 1976; Haith, 1982; Dickeroff Delwiche and Haith, 1983).

In view of these facts, screening model have been deemed to the point for the preliminary phase of the case study at hand, the flow-averaged concentrations being realistically linear with respect to total water quantities at large time scales. Incidentally, this fact holds true also for other indices, like BOD pollution loadings (Whipple and Hunter, 1977) and, even more notably, for solid transport (Haith, 1982).

A second-order class of modeling approaches, suitable to simulation of runoff and dissolved nutrient losses, consists of the development of unit-mass response functions (UMRF) for the source area. Such approach is either of experimental nature (based, for istance, on parameter indentification via constrained linear search) (Jolankai,1983) or of conceptual nature (Bendoricchio and Rinaldo, 1981; Zingales et al., 1984), or of statistical derivation (Marani and Bendoricchio, 1984). Such approaches, which portray in depth most features of time development of pollution loadings,

rely on hydrological and chemical parameters which require calibration and hence prove suitable to large scale prediction upon field-size calibrations. In fact, although predictive errors of screening or UMRF models may seem excessive if compared to observed values, they should be placed in perspectives: errors in model predictions are frequently comparable to sampling and analysis errors, given the difficulty to accurately measure NPSP and, e.g., the rapid transformations (changes in crop practices; urbanizations) of the drainage areas.

At this stage, there seem no point in trying to implement distributed parameter modeling approaches (among the several outstanding contributions in the field) to the actual case study.

The model adopted, as preliminary screening phase of source areas, is a modification of loading function approach suited to the available data. The waste loading, either of solid-phase or dissolved (or total) Nitrogen and Phosphorus, is defined as a nutrient export from unit source area due to runoff event on i-th day, eventually integrated over a time span. Combined effects of crop type, soil nature and hydrology are accounted for in the scheme (Haith, 1982). Computational sequences for dissolved and suspended chemicals are separated at small time scales, reflecting the different mechanisms of leaching from croplands (runoff-erosion); nevertheless both phases can be treated jointly for long-term total nutrient losses evaluations. Examples of typical average concentration ranges in the runoff, as from the U.S. and European literature and of the wide variability of these parameters have been discussed in the literature (Jolankai,1983).

Table 1 - Yearly urban (URB) and agricultural (AGR) loads of N and P split into Lagoon subsystems (after Bendoricchio et al.,1985).

NUTRIENT	SOURCE	LOADS ON LAGOON SUB-BASINS (tons/year)			LAGOON LOADS (tons/year)
		Northern	Central	Southern	
N	URB	2,200	1,300	750	4,250
	AGR	1,900	1,750	1,500	5,200
P	URB	850	500	300	1,650
	AGR	350	300	300	950

The overall results of source magnitude evaluation are

reported in Table 1 for comparison among the single contributions. A detailed description of the results is shown in Figure 2. The discussion of modeling and physical implications are reported elsewhere (Zingales et al.,1984; Bendoricchio et al., 1985).

As a sole remark, the discretization of input load delivery locations is relevant to the receiving water model. In fact, the limited number of NPS yields a lowered priority for spatial transport modeling, at least in the scale needed for evaluation of residence time within the Lagoon. On the other hand, integration of runoff and receiving water models needs a time discretization which forces the user into considering tidal effects on dispersion of the water body (Rinaldo and Marani 1984). Therefore, once long-term nutrient discharge sites are pointed out via screening models, the further step is related to the need of shrinking the time scale of event-based responses (which, for the mainland of the Lagoon, range from one day to dozen of days) to, at least, inertidal scales. Such a time scale is deemed necessary for water quality modeling of the receiving water body (Rinaldo and Marani, 1984). Experimental results have allowed calibration of conceptual UMRF approach which fits the level of detail required for prediction of lagoonal conditions (on the order of the hour) (Zingales et al., 1984).

In this pattern, the concept of reference weighted concentration of pollutants in the runoff has been linked to the determination of parameters of conceptual UMRF (Zingales et al., 1984) portraying mass transfer kinetics and flow rates generation. Even though any assumptions concerning extrapolations of calibrated parameters seem, in principle, arbitrary, it is certainly possible to tailor UMRFs to total quantities predicted via screening models.

NPS POLLUTANT TRANSPORT

One of the foremost issues of long term NPSP forecasting and control is related to feasible integrations of runoff and receiving water models. The need of tailoring the tools suitable to reliable mass balances for large contributing areas and complex receiving water bodies is manifest: in fact, the conceptual chain of events to be simulated is constitued by the array of sources, the mechanisms of transport, plus the net exchange with a larger water body (in the actual case, the Adriatic Sea) which is, somehow questionably, considered as a vessel of infinite capacity. Hence, NPSP model selection is affected by the characteristics and the quality standard of the receiving water body. In fact, the output of NPSP model simulations serves as input for the water quality model, whose level of sophistication needs be tailored to foreseable predicting

ability and scaling-timing of NPSP models. Quality and extent of the data base, either for the source areas or for the hydrodynamic regimes of the water body, also play a major role in model selection.

Remarkable efforts have been put, in the recent past, in establishing the theoretical grounds for simulation of the hydrodynamic and environmental regimes of the Venice Lagoon. Hydraulic approaces (summarized in thorough state-of-the-art contributions: Ghetti, 1979; Di Silvio and Fiorillo, 1980) range from zero order lumped mathematical models to complex formulations of the water quality model. For instance (Ghetti, 1979; Ghetti et al., 1971; Rinaldo and Marani, 1984), the simpler models of tidal behavior of the Lagoon work satisfactorily with respect to prediction of water levels and global water balances, while detailed prediction of local features (velocities and concentrations) needs further specializations (Ghetti, 1979; Di Silvio and Fiorillo, 1980; Leendertse, 1967; Cheng and Walters, 1982). Detailed modeling of movements and transformations of intensive parameters (Dejak et al., 1981; Jorgensen, 1983; Leendertse and Gritton, 1971) like temperature, salinity or nutrient concentrations, induced by the decay or reaction properties of given tracers, casts further uncertainty to the prediction ability of the theoretical tools.

All of the dispersion models considered are, in some way, based on mass, momentum and energy balances, ranging from microscopic transport to macroscopic intertidal phenomena. Albeit perhaps, the dispersion type of description is locally good for the Lagoon, it need not be true at the scale of the entire Lagoon. In other words, the traditional equations of turbulent convection-diffusion mass balance might not be useful for describing fluid motions that are on a scale comparable with the system itself (Himmelblau and Bishoff, 1968). In view also of these fact, the representation of net long term mass transport in a model is inseparably related to the spatial structure of the physical system (Shanahan and Harleman, 1984) and needs be resolved via spatial lumping of appropriate level (Rinaldo and Marani, 1984).

Furthermore the net transport of mass by time-varying currents like those tide- or wind-driven within the Lagoon of Venice, must consider properly velocity fields other than the mean velocity in fixed vertical sections. Stokes' drifts (dependent on the time and the distance scales of the fluctuating currents) must also be accounted for in tidal period-averaged net transports, according to the Eulerian framework (Longuet-Higgins, 1969). Quite recently direct calculations of Lagrangian residual circulations have also been shown to be feasible for large tidal water bodies similar to the Lagoon of Venice (Cheng and Casulli, 1982). Lagrangian random walk-type simulations have also been proposed for far-field dispersion (Chin and Roberts, 1985).

Furthermore:
a) - while the mechanism of leakage of urban and industrial point sources is not affected by the hydrodynamic conditions and is characterized by relatively high local concentrations, NPS tracers are characteristically discharge dependent as far as either concentration or total quantities are concerned. The higher dissolved or solid-phase loads are carried, in fact, by the floods, which at the same time, yield important dilution of the loading. Low concentrations, difficult simulations of movements and transformations of its components, time variability (dependent upon hydrologic and hydraulic parameters) and large total quantities involved in mass balances are therefore the main features of NPSP;
b) - to simulate long-term concentration variations, a lagoon water quality model must consider the influence of net mass transport. Long-term lagoon water quality modeling should therefore implement fast and efficient algorithms, and subtidal time discretization. Certain discontinuities at the sea-lagoon interface need be accounted for, because generrally the outflowing water holds higher nutrient concentrations than the inflowing ones. It seems that dispersion models, as the one already implemented for environmental studies on the Lagoon (Di Silvio and Fiorillo, 1980; Dejak et al., 1981), are tailored to simulations of current and tracer patterns in a specified tidal or wind condition only for relatively short simulation pattern.

At the present stage, it is therefore clear the difficult in meeting all requirements of prediction/simulation characterized by short time step and long time span.

In view of the fulfilling of such twofold objective, the rules for the modeler seem to follow some speculations:
1) - detailed studies of simplified schemes of two-dimensional tidal propagations have been carried out (Rinaldo and Marani, 1984). The most competitive tool for lumping hydraulic fluxes into any level of aggregation is related to filtering of velocity and elevation fields calculated by advanced finite element techniques (Putti et al., 1985);
2) - multiple-vessel spectral models serve at best the overall purposes of long-term forecasting of dispersion at spatial scales like that of the Lagoon and time scales like those forced by NPSP inputs and by eutrophication processes (Rinaldo and Marani, 1984);
3) - the spatial discretization of the system of cells (vessels) simulating the water quality variations over the horizontal landscape can be linked to the finite element grid of hydrodynamic simulation (Putti et al., 1985);
4) - theory of signal processing and finds of numerical spectral analysis induce important simplifications in the

mathematics and in the computational loads.

Due to limited field data available for the Venice Lagoon system, to date hybrid modeling techniques (mainly referring to model calibration via more sophisticated models runs) have been implemented.

EUTROPHICATION OF THE LAGOON

Eutrophication phenomena imply the most ostensibile and undesirable facets of phytoplanktonic growth and often represent an ultimate concern for the environmentalist. Albeit the phenomenon has long been known within the Lagoon (as some toponyms referring to algae might suggest) alarming frequencies and spatial spreading of algal blooms have been occurring only in recent years. Among the noteworthy spinoffs of the modified trophic conditions, fish death-blows (either in fisheries or within the tidal system), odors and often limitation to recreational activities are perhaps to be mentioned.

While the "cause" in plankton-based systems (certainly due in most part to the increase in nutrient release from the drainage area) seems amenable to numerical simulation, model selection and application for eutrophication effects is demanding, due to the lack of continuous data on a complex system which is further characterized by tidal oscillations of the parameters and marked hydraulic advection (Rinaldo and Marani, 1984).

In the actual case of Venice, for which the scheme of homogeneous well-mixed interconnected vessels proved satisfactory, a feasible approach for simulation has turned out to refer to Catastrophe Theory (Zeeman, 1978; Poston and Stewart, 1978). The foremost advantages of such an approach are:

- step by step refinements of the qualitative description;
- effective representation of the processes and overall reliable forecasting of blooming phenomena.

Catastrophe, within environmental systems, means a jump from an initial equilibrium state to another one, usually far from the former, induced by slow changes of the control variables. Analytical manipulations of the discontinuity allow application of the theory if certain requirements are met. Such constraints seemingly fit eutrophication phenomena. The recent literature has, in fact, proved noteworthy successful in application to the field of algal population growth (Dubois, 1979; Duckstein et al., 1979; Van Nguyen and Wood, 1979; Kempf et al., 1984).

Catastrophe models may be applied to bimodal (Voinov and Svirezhev, 1984), piecewise continuous, hysteretical or

divergent processes provided that control variables can be clearly pointed out: algal levels in coastal or estuarine waters are amenable to control via Chlorophill-a (a-Chl) concentration while, if the bloom only is to be modeled, the control parameters may be cast into a growth factor, specialized by an index of nutrient content (Liebig's limiting factor concentration in most applications to date), plus an index combining the effects of temperature and insulation. During the bloom, in fact, bacterial reactions, net zooplankton assumptions and other biological events do not affect substantially the overall processes.

As far as phytoplanktonic mass growth is concerned, a typical bimodal trend is shown to exist. This behavior is characterized by a state of survival at low concentrations (10÷30 ppb of a-Chl) and a further one at high concentrations (100÷300 ppb of a-Chl) which is proper to algal blooming. The back-jump from high to low conditions is patterned on paths other than that labelling the blooms. Furthermore, evolution patterns of the phenomena show typical hysteretical trends of catastrophes.

The only catastrophic surface portrayed by two control is Whitney's cusp, i.e. a cubic equation dependent upon the catastrophic variable. In this context, the catastrophic dependent variable is the phytoplankton mass measured as concentration of a-Chl; the splitting variable is concentration of limiting nutrient; and the normal variable is a lumped parameter of light intensity and water temperature.

Albeit Whitney's model does not seem very sophisticated indeed if compared to other schemes (Swartzman and Bentley, 1979), in simulating all phenomena connected to the algal blooms, nevertheless its structure allows a certain degree of accuracy in portraying most critical events and a simple graphical layout.

As experimental evidence supporting these ideas, a clear bimodal trend and an overall catastrophic occurence, with change of trend on the 24-th day, is shown by the results of Figures 3 and 4. A field experiments campaign of chemical and biological nature (Zingales et al.,1983) has, in fact, been carried out aiming at pointing out the parameters controlling the eutrophication of the Lagoon of Venice. In particular, attention has been focusing on the water bodies adjacent the industrial sites of larger size and marked by limited tidal circulation. Existing temperature gauging have also been checked. The campaign, performed during spring time due to its potential embedding of algal blooms, lasted up to 25 days and considered as potential indices: pH; turbidity; dissolved oxigen; salinity; water temperature; and sylicon, ortophosphate, nitrogen (in various forms) and Chlorophyll-a concentrations.

Algal blooms have been detected only at the station quite apart from the industrial sites, due to the presence of Ammonium-ion levels toxic to algal life. The foremost data (temperature, total Nitrogen and Chlorophyll-a) gaged at the two stations involved in blooming phenomena are reported in Figure 3. Figure 4 further illustrates, for the station near Tessera, the plot of sequences of equilibrium paths on a cusp surface.

The normal control variable (water temperature) undergoes a five-degree excursion connected with the deeper insulation and wind-calm. Nonetheless, the high degree of mixing experienced by lagoonal canals, river outlets or the openings to the sea, where tidal convection is strong, yields a generally critical determination of the parameters unless for shallow low-current sites.

The second control variable turned out to be total Nitrogen. Concentration changes have, in fact, experienced a noteworthy increase on the 10-th day (right at a 30 mm rainfall event in the area) released most likely by croplands just fertilized. A decrease in salinity and an increase in turbidity have also been observed.

Algae populations, the ~~in~~dependent variable of the model are formed by 80% diatoms (the most common species) plus chlorophiceae and cyanophyceae in much smaller proportions. Certain correlation exists between maxima of Chlorophyll-a and limiting nutrient concentrations. In fact, a-Chl concentrations of 250 ppb against maximum Nitrogen concentrations of 3 ppm are shown at blooming stations (Tessera) whereas otherwhise (S. Giuliano) the maxima reach 120 ppb (a-Chl) and 2 ppm (Nitrogen). The amplitudes of Chlorophyll-a concentration oscillations may be further related to tidal convection and, partially, to the analytical method.

PERSPECTIVES OF LONG-TERM ABATEMENT AND CONCLUSIONS

Nutrient NPSP pollution abatement of the Lagoon of Venice is feasible (may be at low cost practices) provided that major sources be pointed out and controlled. The logical step forward is hence related to a detailed knowledge of: nutrient release mechanisms; nutrient movement and trasformations within the drainage network and within the Lagoon proper; autodepurative properties of the Lagoon system. The tool is primarily a continuous data acquisition system focused on the environmental aspects of the phenomenon which is, to date, still discussed. Top-priority is therefore to be granted to creation and maintenance of such data base and on the related verification of the theoretical progresses, namely generation, propagation, near-

and far-field mixing and dispersion models.

Domestic pollution abatement has also been considered (Bendoricchio et al., 1985): upon completion of the planned sewage systems (and via first and second stages of wastewater treatment), the total domestic NPS load turns out to be 3,400 t/year of Nitrogen and 1,500 t/year of Phosphorus. In practice, the overall abatement will range at about 20% lower levels. Sea outfall for industrial and urban point sources are being planned, with the further aid of a realised by-law reduction in the content of polyphosphates within commercial domestic detergents from 8% to 5%.

Urban runoff pollution abatement, albeit not yet considered in the array of studies in progress, may benefit from the parallel research progress in the field.

Agricultural releases may be controlled via a set of techniques of increasing diffusion, mainly referring to slow release fertilizers and controlled drainage (Skaggs and Gilliam, 1984). In fact, 50% abatements in Nitrate losses have been measured without losses of production through Nitrate reduction via submerged drains, an increasingly popular practice in the zone. These techniques added to the results of the oncoming studies of the Best Management Practices commissioned by the Regione del Veneto (the local Government) should guarantee consistent improvements.

Aids to reduction of eutrophication phenomena may also yielded by modifications of tidal circulation within suffering shoals via revitalization of dead water zones.

The basic idea underlined by the actual work is to single out the relative importance of each elementary mechanism affecting the eutrophication of the Lagoon and to establish the theoretical grounds for an assessment of control policies for the foremost causes. Hence, the mentioned abatement approaches, efficient in a cost/benefit perspective, have been outlined.

The study is focusing on the fate of nutrients in surface waters as a major contributor to the actual decay, since toxic substances of industrial, agricultural and urban source may just require a thorough control even in commercial routes.

As a final comment, a noteworthy environmental aspect, as far as either nutrients or toxic substances are concerned, is related to lagoonal sediments. They have not been mentioned insofar because they might require ad-hoc interventions on low-current zones and mechanical remotion in some instances.

Through the complex set of planned interventions, among which

only a part already operational, the complexity of the problems can be inferred and hence the fascination to researchers, professionals and, may be, politicians.

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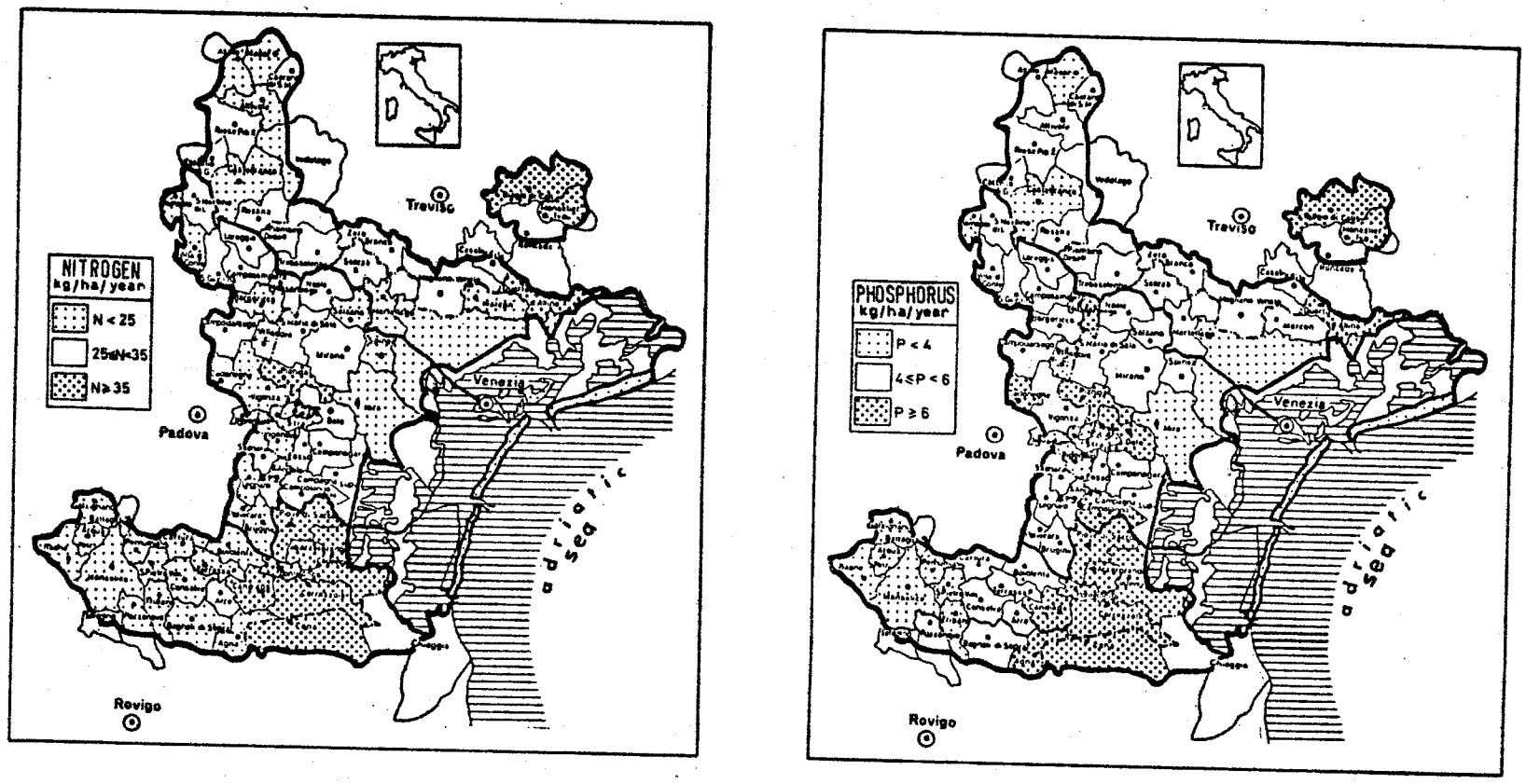


Figure 2 - Mainland draining into the Lagoon of Venice: spatial distribution of nutrient loads.

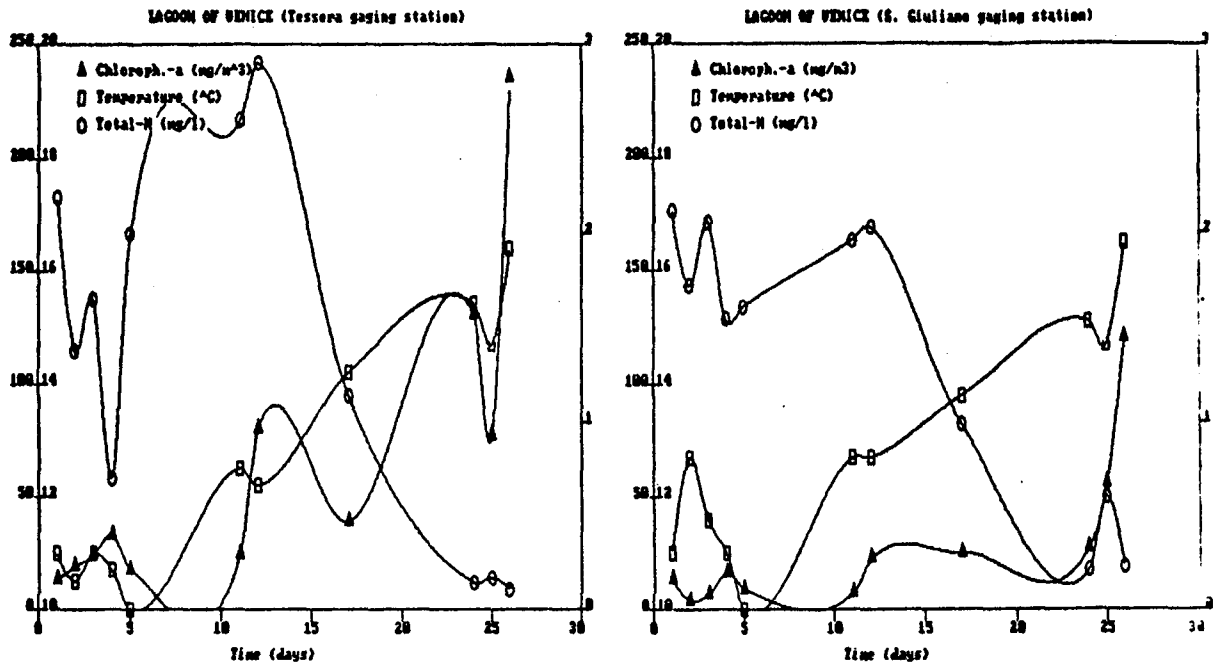


Figure 3 - Values of Chlorophyll-a, temperature and total Nitrogen gaged at two stations (Tessera and S. Giuliano), as during the field campaign of spring, 1981.

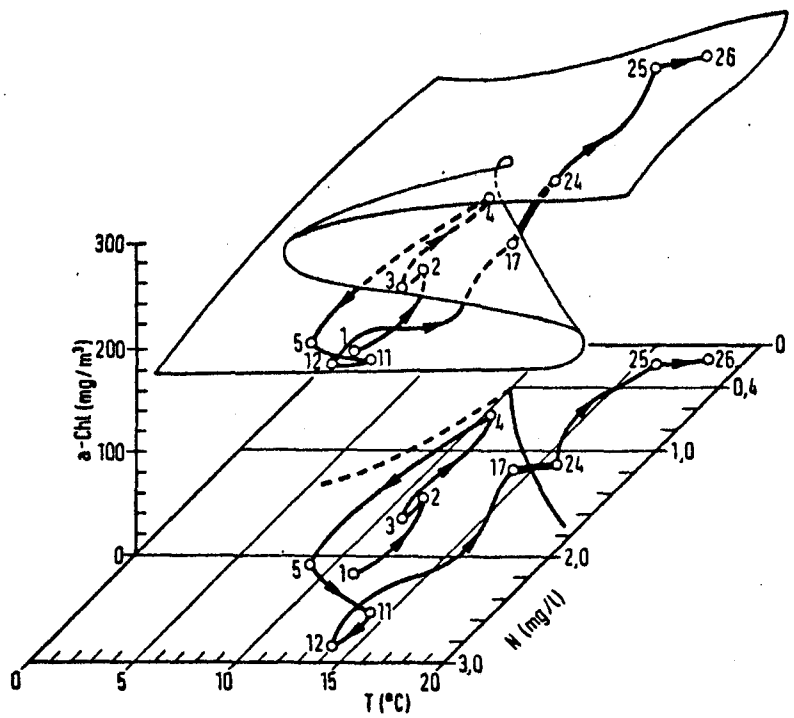
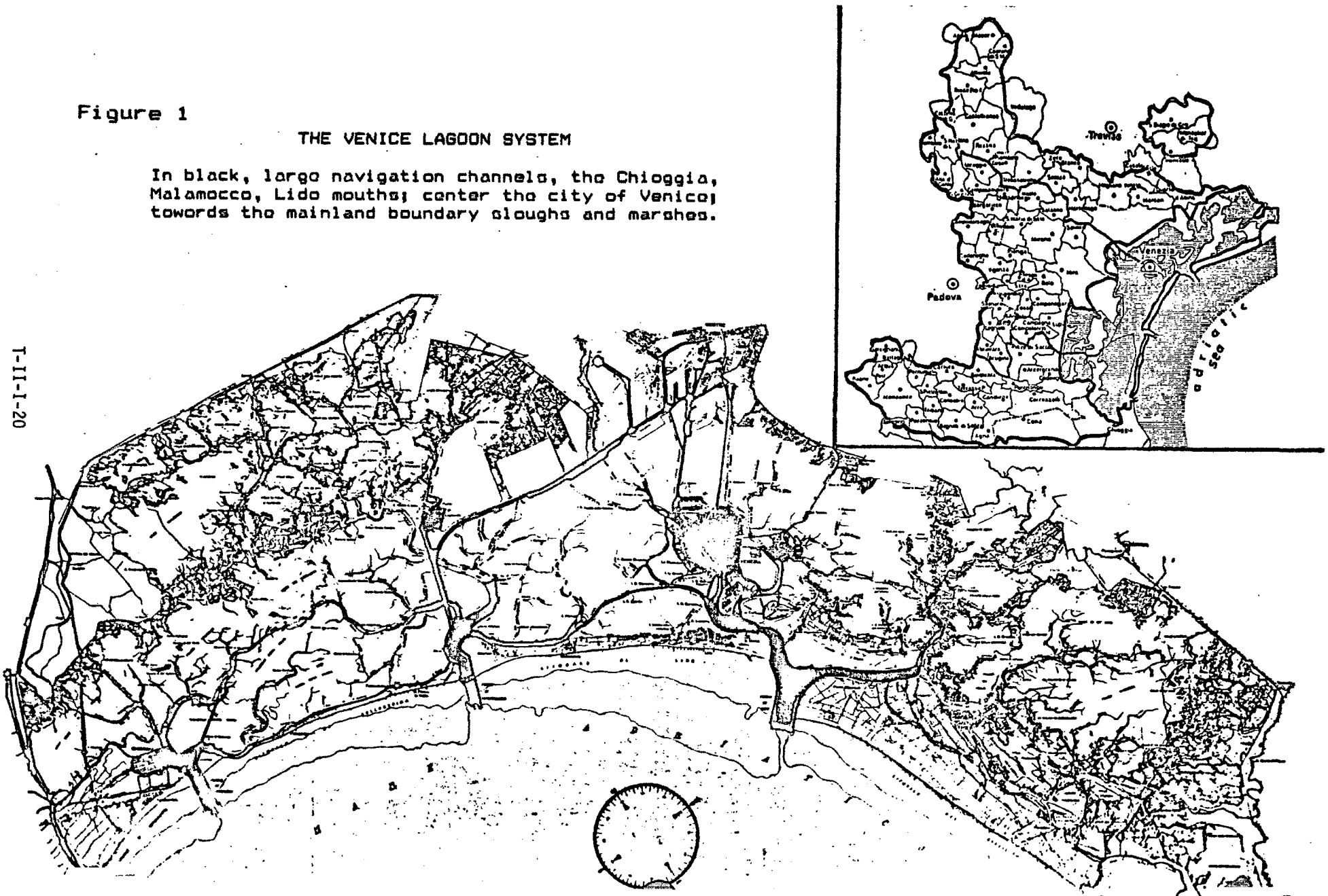


Figure 4 - Tessera gaiging station: sequence of Phytoplankton states. The catastrophic jump is underlined by the thick solid line. The numbers point out the days in which the sample has been collected.

Figure 1

THE VENICE LAGOON SYSTEM

In black, large navigation channels, the Chioggia, Malamocco, Lido mouths; center the city of Venice; towards the mainland boundary sloughs and marshes.