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SECTION 11

NONPOINT SOURCE POLLUTION

ACTIONS TO TREAT SEDIMENT WATER QUALITY PROBLEMS

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ABSTRACT

Methodologies for estimating sediment sources and sediment yield for with and without project actions are presented in this paper. These form the basis for planning for improvement or protection of water quality based on sediment trapping and sediment yield reduction. Sediment and pollutant trapping effects of conservation measures must be considered as well as the erosion reduction or change in availability of the pollutants. Alternatives are considered on the basis of physical and economic feasibility as well as the anticipated effects on water resources. The concepts of measuring water quality parameters in monetary and non-monetary terms with regard to the planning processes is also presented.

INTRODUCTION

Water quality concerns have become of interest to most Americans. New legislation and health concerns have made the quality of our surface and ground water an issue that affects the families of our nation. In the last decade, institutional concerns for water quality have begun to change their focus from industrial discharges and municipal waste disposal to the water quality effects that are caused by nonpoint sources of pollution.

This paper describes the investigation, formulation, and evaluation of actions that deal with sediment, sediment-attached substances, and their impairment of surface water. It considers the effects of sediment on water use, the resultant impairment, and concepts of developing and evaluating activities that would reduce the introduction of sediment into water to improve or protect its quality.

DEFINING WATER QUALITY

Water quality is its state of being--those properties that make it suitable for use by humans or by other living things. The water resource is managed by each state and, where appropriate, has been designated suitable for specific uses. This designation is made through the use of state water quality standards which are an extension of state law. Where the quality of the water in the water body is not suitable to support the designated present or future use, it is said to be "impaired," either fully or partially.

Water use suitabilities are defined on the bases of physical, chemical, and biological parameters which are criteria by which the ability for plants and animals to use the water may be measured. Physical criteria include the temperature and oxygen content of water which is the habitat for a plant or animal. Sediment, when considered separately from the attached contaminants, is a physical parameter because it occurs in an undissolved state and affects the water use.

The natural characteristics of surface water may be modified by the use and management of the land. Water quality is a function of the <u>availability</u> of a given substance in or on the earth (soil is nearly always available); its <u>detachment</u> from its source through solution or mechanical forces of erosion; <u>transport</u> through runoff or percolation into soil and ultimately to a water course or aquifer; and the <u>integration</u> of the substance when it reaches the receiving water.

Water uses are impaired by nonpoint source substances that come from industrial activities, residential neighborhoods, mining, agricultural production, and other sources. Sediment that is produced by agricultural activities is a water quality concern because it reduces water uses by the action of filling or taking up space, causes turbidity, deteriorates aquatic habitat, and it serves as a medium to carry unwanted chemicals. Other sources of sediment produce similar effects.

DEFINING THE WATER QUALITY PROBLEM

A problem with water quality should be defined as a full or partial impairment of the designated use. This impairment might occur when a physical condition, substance, chemical, or biological entity is present in the water and renders it less suitable or unsuitable for the use. Most waters of every state have designated uses, but where such uses have not been designated, water quality criteria are generally based on aesthetic values such as color, odor, or appearance that are found in the state water quality standards.

Sediment and nutrients are substances that may impair water use. Some nutrient forms are attached to and are transported by sediment carried by water, while other nutrients and minerals are dissolved in the water. Examples of sediment and nutrient impairment of water uses include (1) sediment (or sediment-related substances) impairing a stream or lake fishery, (2) impairment of water-based recreation, or (3) loss of storage in a lake or reservoir.

Habitats of warm-water fish, such as largemouth bass or catfish, may be damaged by the deposition of sediment and nutrients on and within the substrate. Suspended sediment may also harm the fish themselves by restricting sight feeding, respiration, and propagation. The sediment related problems for a warm water fishery may also happen to colder water fisheries (trout, salmon, small mouth bass). While cold water fisheries tend to be associated with lower sediment yields, they also exhibit less tolerance to the effects of sediment, chemicals, and low oxygen levels.

Swimming and boating become hazardous when the amount of sediment or algae hides subsurface hazards or restricts depth. The appearance of water that has a high turbidity from sediment or vegetation reduces the quality of the recreational experience. The value of the water stored in a lake or reservoir used for municipal, industrial, irrigation, flood control, fish and wildlife, or recreation is based upon its ability to supply these purposes.

FORMULATING ACTIONS FOR WATER QUALITY PROBLEMS

The formulation of federal water quality projects under Principles and Guidelines (US Water Resources Council, 1983) is based on future conditions that are projected to occur if no action is taken to remedy a water resource problem. The forecast for the future without action condition is based on physical conditions that affect the quality of the water resource. Planning goals are set, based on this future condition, to provide a base for project formulation.

Forecasting future without action conditions includes the use of national, regional, or local resource forecasts. Future land use and management forecasts may be the most useful guide because of their direct relationship to the quality of surface and ground water. Population changes, increases in prosperity, new demands for water uses, and present and future government programs that would stimulate economic activity are all valuable indicators. Forecasts describe conditions at a discrete location which is the site of the water use impairment. The forecast reflects the impairing effects of the substance(s), sediment and nutrients, that cause the water quality problem. All sources, agricultural or nonagricultural, point or nonpoint, are inventoried and routed to the point of impairment. Different sources of impairing substances have different rates of delivery efficiency. The amount of the substance delivered is the forecasted measure of the water quality impairment.

SETTING PRIORITIES FOR WATER QUALITY ACTIONS

Planning goals for water quality are set at a level at which the desired water use would be protected, improved or restored. For example, a water quality goal might be to improve or restore a fishery but not necessarily to reduce sheet and rill erosion, which may be the sediment source.

Reduction or elimination of the flux of the water impairing substance alone may not provide the desired effect on the water use. Additional treatment at the point of impairment may be required to achieve the desired water use potential. Such treatment may include a change in the aquatic environment such as a modification of a stream habitat or weed harvest in a lake. For example, dredging of a portion of water body to provide an adequate environment (available storage) for the water use might be an alternative to address the impaired water use. Reduction in the flux of the polluting substance might not even be considered. Where a remedial action involves only a reduction in the flux of the impairing substance in the source area, the potential for the recovery of an impaired stream reach, lake, or aquifer without onsite action must be evaluated.

METHODOLOGIES FOR ESTIMATING EROSION RATES AND SEDIMENT YIELD

The magnitude, extent, and location of sediment sources must be identified and quantified in order to choose effective measures to address the water use impairment. Sediment yield or the quantity of sediment at the point of impairment is used as the basis of formulation for water quality problems. The delivery of sediment and associated substances may be measured or estimated for the present and future conditions, with or without remedial actions. Therefore, project actions should adapt a predictive methodology which identifies sediment sources, quantifies sediment transport and deposition, routes sediment to points where damages have been identified, considers the effect on the water use impairment, and accounts for the change in water quality from alternative remedial approaches.

In order for any remedial action to be effective, the sources of the elements that cause water quality problems must be identified in terms of:

The cause of the problem, such as excessive soil erosion (detachment) or application of nutrients in excess of the ability of a crop to use them (greater availability: soil is always available).

Location of the source of the impairing substances, as well as their dispersion or concentration in the watershed relative to the impaired water uses (availability: soil is always available).

Substance yield from each source (detachment and transport), defining the relative magnitudes of production and delivery of sediment or nutrients from each of the source areas.

Conditions under which a problem occurs (detachment, transport, and integration) such as the rainfall or runoff conditions contributing to the use impairment or the magnitude or frequency of storm events associated with the most problems.

Processes that impair the water use, such as sediment deposition or suspension.

Tools for estimating erosion (detachment)

Quantifying erosion will use separate predictive tools for each erosion process (type of detachment). The understanding of the relative contribution of sediment from each source directly affects the formulation of solutions to the water quality problems that are related to sediment. Sediment derived from sheet and rill erosion will have textures similar to the surface soil layers, whereas sediment derived from gully erosion may reflect textures from several soil horizons or rock units. Sediment derived from surface soil horizons will normally have higher concentrations of agrichemicals than from subsoil horizons or rock units.

Sheet and rill erosion may be predicted using the Universal Soil Loss Equation (USLE) (Wischmeier, 1978) and will soon be estimated by the Revised USLE (RUSLE). RUSLE will allow the user to determine seasonal factors for erosion prediction. The eventual replacement for the USLE, known as the Water Erosion Prediction Project (WEPP) (USDA and USDI, 1987; USDA, ARS, 1989), is currently under development. In addition to rill, interill, and concentrated flow detachment processes, WEPP will be able to estimate sediment yields (transport) on small drainage areas.

Ephemeral gully erosion is a concentrated flow erosion process that occurs on cultivated cropland. It is characterized by erosion channels that are partially or completely refilled by normal tillage operations; hence,

the ephemeral nature of this type of gully. Ephemeral gully erosion can be estimated using one of the following **methods**: Ephemeral Gully Erosion Model (USDA, SCS 1986 and 1988; Merkel et al, 1989); inventory plot data, direct measurement method (USDA, SCS, SNTC Bull. S210-7-4, 1986); and the Thorne Method (Thorne, 1985). Ephemeral gully erosion prediction capability will be included in the Watershed and Grid versions of WEPP.

Classic gully erosion is the formation of concentrated flow channels that may not be crossed or filled in by normal tillage equipment and cannot be used for production. Classic gully erosion may be estimated using simplified hydrologic techniques (USDA, 1966); air photo interpretation; and geomorphic principles (Schumm et al, 1984).

Stream erosion is the erosion of bank and bed materials by tractive stresses, bank shearing, and slipping or sliding. Forecasting channel erosion is based on geomorphic and stream mechanics analysis. If the bed is being eroded, the sediment produced by this process, as well as the sediment produced by bank erosion, must be included when assessing and forecasting stream channel erosion. Methods to predict stream channel erosion include air photo interpretation, onsite evaluation, geomorphic analyses, and bedload transport analyses.

Irrigation-induced erosion can be estimated by the direct measurement method, based on the changes in the physical dimensions of the furrow voids, by the CREAMS (USDA, Dec 1984) and GLEAMS (Knisel et al, 1989) computer models, and other models currently under development. The Watershed version of WEPP is being designed to estimate irrigation-induced erosion.

Mass wasting is erosion caused by gravity (e.g. landslides, soil creep) and can be a major sediment source on steeply sloping terrains. Prediction may be made using geomorphic techniques.

Wind erosion (detachment) can be estimated by using the Wind Erosion Equation (Woodruff, 1965), but the WEQ does not predict transport or deposition of sediment from wind.

Gross erosion is the summation of all types of erosion. The amount of gross erosion is not as important as the magnitude of the various types of erosion that may cause a water quality impairment. The summary should show (1) magnitude of erosion by type and land use; (2) quality of eroded soil, including texture and attached constituents; (3) spatial distribution; and (4) relationship to the water quality problem.

Determining Sediment Yields (Transport)

Not all eroded soil is moved from a watershed by the same storm event that caused the detachment, but it is continuously deposited, resuspended, and moved by dynamic processes. Some eroded soil remains as sediment within the field of origin while other sediment is deposited in the first drainageway downstream, and so on throughout the drainage network.

Sediment delivery ratios (SDR) can be used to estimate the percentage of eroded soil that is transported to a specific point in a watershed. This method may use geomorphic relationships, such as watershed size (USDA, 1983), relief/length ratio (USDA, 1979), Roehl's morphometric analysis (Roehl, 1962), the slope profile method for very small watersheds (Clarke and Waldo, 1984), the large watershed methods (Waldo, 1986), and the depositional area/total area relationship (Flaxman, 1974).

SDR's can be used with the greatest level of confidence where supported by actual measurements of sediment transport or deposition rates, as from reservoir surveys or from stream gage analyses. An SDR is normally valid for planning purposes only within a specific geomorphic or physiographic area.

Sediment yield predictive models may be used to estimate sediment yields. Examples of these models include the Modified Universal Soil Loss Equation (Williams, 1977), the Pacific Southwest Interagency Committee method (PSIAC, 1986), and Flaxman's Predictive Sediment Yield Equation (Flaxman, 1974). Computer models which are based on the physical processes of erosion and sediment transport include CREAMS (USDA, 1984), GLEAMS (Knisel et. al., 1989), AGNPS (Young et. al., 1987), ANSWERS (Beasley et. al., 1980), SWRRB

(Williams, 1985), and the Watershed version of WEPP. These models combine algorithms for the erosion processes (detachment) and hydraulic transport capability to route sediment from its source to an evaluation location. Users of sediment yield models must be properly trained in application limitations, data requirements, and in the interpretation of the output.

Sediment transport models may be used to estimate the movement and deposition of sediment within streams or channels. The suspended and bedload portions of the sediment transported by streams are considered both in terms of volume and particle size. The design of practices to trap sediment must consider the effective particle settling velocities if the practice is to be effective. Therefore, the degree to which the silt and clay portions of the load travel as aggregated bundles and not discrete particles is a critical element to the process of formulating measures to detain or trap them.

The effect of sediment yield on the water quality problem is also a function of the movement of the suspended sediment through the specific point of water use impairment and/or deposition of the sediment at the problem location (integration into receiving waters). Estimates of sediment trapping or deposition are used to characterize these effects. Such analyses may be made using transport models or trap efficiency models (USDA, NEH3)

Sediment yield summaries should be made for each point or reach where sediment related impairment of water quality has been identified. Each summary should combine the amounts of sediment yielded by source to the evaluation point, the quality of sediment (texture and attached constituents), and the spatial distribution of the sources.

FORMULATING ALTERNATIVE PLANS FOR WATER QUALITY

Sediment yield and sediment deposition, not erosion rate or volume, are the focus for formulating solutions to water quality problems caused by sediment. The formulated action must be directed towards a goal that is determined to be the level at which the water resource will be restored, improved, or protected for its designated water use. For example, reducing sheet and rill erosion on fields that are remote from the site of water quality impairment will not be as effective in reducing the effects of sediment as treatment of fields that are closer to the site. Similar variation in effectiveness will occur when comparing a field which flows into a channel that is interconnected with the stream network, to a field which flows into a channel that is not interconnected.

The development of actions to reduce sediment yields is based on the efficiency in reducing the effects of sediment at the problem site. The actions usually involve a combination of management or vegetative practices that reduce chemical availability or soil and chemical detachment, or structural practices that reduce sediment transport by hydraulic trapping, allowing attached chemicals to immobilize, decay, or to be utilized through plant growth. The remedial treatment may also include measures such as dredging or confinement to treat the substances at the impairment site.

Practices that achieve the desired result must be chosen to become alternative plans. For example, if it has been determined that ephemeral gully erosion is a major cause of a water quality problem, conservation tillage alone is not likely to be an adequate solution because it does not fully address the source. Practices such as grassed waterways, water and sediment control basins, and terraces would be needed to control a concentrated flow erosion problem. Where goals for reducing sediment yield cannot be reached by treatment to reduce erosion of watershed lands, trapping sediments with impoundments or treatment at the point of impairment may be viable alternatives.

Potential conflicts in formulating solutions to water quality impairments may arise, such as:

* Protecting ground water from infiltrating contaminants versus inducing infiltration to protect surface waters.

* Installing practices to promote production vs. those needed to protect water quality and quantity.

Forecasting the effects of remedial actions on sediment yield must consider the effects of ongoing conservation programs and the future impact of legislation in retiring and then possibly releasing highly erodible lands from permanent vegetation contracts. Sediment yields based on future conditions without remedial action should be displayed for each point of water quality concern by source, quality of sediment, and spatial distribution of sediment sources (maps).

Forecasting the effects of remedial actions on sediment yield involves the same methods described above. A summary and analysis are needed for each point or reach of water quality concern to describe the effects of the action. Each alternative can then be readily compared with sediment yields for future without action conditions to assess its merits.

PROJECT FORMULATION: WHAT IS THE SOLUTION?

Two methods may be used to develop alternative plans for actions to solve water quality problems related to sediment: the incremental analysis procedure (USDA Bull. 200-4-3, 1984) and the conservation options procedure (COP) (USDA Tech. Note 200-LI-2, 1988). The SCS develops alternatives to solve water resources problems through a systematic planning process which culminates in the development of at least two types of plans:

* The **National Economic Development (NED) plan** which addresses the federal objective is the plan which maximizes net economic benefits consistent with protecting the Nation's environment.

* The **Resource Protection (RP) plan(s)** may provide less than maximum net economic benefits in order to address other Federal, state, local and international concerns.

Water resources plans are formulated in a systematic manner to include all reasonable alternative plans and to allow the decisionmaker the opportunity to judge their relative merits.

Incremental Analysis (USDA, 1984) is a procedure that identifies the NED plan by using incremental benefit/cost ratios to evaluate practices and combinations of practices to solve the water quality problem. Practices or actions are added incrementally if their added benefits exceed added costs. The first step is to list all practices considered to be acceptable means to solve the problem and the costs and benefits of each increment. The practice which has the greatest benefit per dollar of costs is selected as the first increment. More practices are then added to the most efficient practice to form an alternative plan that addresses the water quality problem. The procedure continues until other practices that provide positive net incremental benefits cannot be added, defining the NED plan. An accurate analysis will result only if the practices are added in decreasing order of efficiency.

Incremental analysis essentially formulates and selects the practices in the NED alternative plan. The procedure could lead to the selection of an unacceptable plan; i.e., one that doesn't resolve the water resource problems being considered.

Conservation Options Procedure (COP)

COP (USDA, 1988) is an economic method which uses cost efficiency, non-monetary environmental and social effects, and net benefits to evaluate alternatives. The procedure identifies the NED plan and RP plans, from which the Recommended plan may be chosen. Input from an interdisciplinary team is a required feature of this procedure, especially where non-monetary values enter into plan selection decisions.

Stage I of COP is an analysis for cost-effectiveness of practices, and their combinations, that are selected by the interdisciplinary team and are technically feasible to address the water quality problem. The first step is to identify and list all of these technically feasible practices and then combine them into alternative plans. The next step is to select a common base for comparison which becomes a common denominator for the cost effectiveness analysis. If the water quality problem is caused by sedimentation, the common denominator might be the cost per ton of reduced sediment yield.

The alternative plans are then displayed in a tabular format. The tables should be as complete as possible to provide a record of the process and identify feasible options that were considered. The alternatives which are not technically feasible should be eliminated from further analysis. Likewise, those alternatives not deemed cost effective should also be eliminated.

Stage II of COP is the net monetary analysis of the alternatives surviving the initial cost effective analysis. The alternatives brought forward from Stage I are those which are the most cost effective and satisfy the state standard or remedial goal based on a parameter that solves the water quality problems. The time and cost savings of eliminating inefficient alternatives in Stage I become apparent in this stage.

The first step in Stage II is for the interdisciplinary team to quantify the physical, social, and environmental effects of each of the alternatives. These effects must be estimated before their monetary values can be determined. The next step in Stage II is to determine all monetary benefits, including net benefits for each alternative.

Stage III of COP is the evaluation of all alternatives, the formulation of alternative plans, and the subsequent identification of the NED plan and most desirable RP plans. In this stage the non-monetary, physical, social, and environmental effects are considered along with the monetary effects of Stage II. The alternative plan with the greatest net monetary benefits becomes the NED plan. The alternative plans that achieve acceptable levels of resource protection becomes the RP plan. Efficiency, social, environmental, and economic trade-offs define achievability of the RP plan. The actual criteria for the selection of the best RP plan is a joint effort of the interdisciplinary team and the decision makers. The resultant choice or recommendation to solve the water quality problem represents the best alternative considering all criteria.

SUMMARY

Formulating solutions to water quality problems related to sediment requires an intimate understanding of the relationship between the water quality problem, the specific watershed areas and erosion processes that are responsible for producing the damaging sediment, and the sediment transport characteristics of the watershed.

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REMEDIATION OF CONTAMINATED SEDIMENTS-GREAT LAKES

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ABSTRACT

In-place, contaminated sediments represent the most serious, unaddressed, nonpoint source pollution problem facing our nation's waters. Nowhere is the contaminated sediments problem as critical as in the Great Lakes basin, where both ecosystem and human health are at risk. The magnitude and extent of sediments contaminated by toxic substances are presented in general for the entire basin as well as in more detail for several hotspots. While remediation initiatives have been delayed for years, navigation dredging has proceeded in these areas and the associated resuspension of toxic substances poses environmental risks. About 5 million m^3 of material is dredged in the basin each year at a cost of about $24 million. Approximately 90 percent of the total represents U.S. projects and 60 percent of the total is in the Lake Erie basin. Progress is reviewed in meeting commitments made by the U.S. and Canada in the Great Lakes Water Quality Agreement to remediate contaminated sediments. Policy implications for the use of Superfund authorities and the special "Confined Disposal Facility" program created in Section 123 of the Rivers and Harbors Act are discussed. Not only initiatives under the Agreement but also new budget authorities will be needed to address this huge multi-ten billion dollar problem in the Great Lakes basin.

INTRODUCTION

Purpose and Scope

More than any large waterbody in North America, the Great Lakes system is suffering from almost a century of releases of toxic substances. The contaminants have permeated the ecosystem to the point that reproduction of some aquatic and terrestrial wildlife is impaired, fish commonly suffer tumors, and increased risks exist for human health impacts. This situation has significant implications for relations between the U.S. and Canada. Remedial actions to address the contamination are covering many different sources of toxic substances, including in-place contaminated sediments. Because point source discharges of these toxic substances have been greatly reduced, the in-place contaminated sediments are implicated as significant pollution sources. This paper describes this situation and presents an evaluation of progress in dealing with these in-place contaminants.

Risks Posed by Toxic Substances in the Great Lakes Basin

The last twenty years have seen dramatic improvements in water quality of the Great Lakes system. The Cuyahoga River no longer burns, Lake Erie is no longer dying. The ten billion dollars spent on sewage treatment by governments have made dramatic strides. Unfortunately, an even bigger challenge lies ahead -- that of addressing persistent toxic substances which have accumulated in the Great Lakes Basin Ecosystem. Residues of persistent toxic substances -- PCB's, dioxins, furans, heavy metals, chlorobenzenes, mirex, toxaphene, DDT, -- permeate the ecosystem. The contamination represents a "soup" of toxicants with 209 different PCB's, 135 furans, and 75 dioxins, and many compounds are not readily identifiable.

Sources are as varied as the contaminants themselves - point source discharges, atmospheric emissions, nonpoint sources, contaminated ground water, and contaminated sediments. The extent of contamination is described in detail by the International Joint Commission's (IJC) Water Quality Board (1987). Predators at the top of the food chain, including humans, are vulnerable to biomagnification of these chemicals. Information is now sufficient to link chemical contaminants to a variety of biological abnormalities. At least two dozen top predators -- ranging from fish to birds, reptiles, and mammals -- continue to experience reproductive problems, behavioral abnormalities, and population declines in the basin (Colborn, 1989). These persistent toxic substances are teratogenic in nature -- they affect the offspring of exposed adults by causing abnormal development, immune system suppression, birth defects, and behavioral abnormalities.

Why be concerned with these abnormalities in wildlife? The reason is because there is direct, statistically significant evidence that similar teratogenic responses are occurring in offspring of females who eat contaminated Great Lakes fish. As described by Colborn (1989), the reproductive outcomes of women who ate Lake Michigan fish 2-3 times per month, were significantly different from those who did not eat Lake Michigan fish. Adverse affects were demonstrated on birth weight, gestation period, skull circumference, and cognitive/motor/behavioral development of infants. Fish and wildlife may be serving as the "canary in the coal mine" in warning of environmental danger. For more information please refer to the IJC's Science Advisory Board (1989).

SIGNIFICANCE OF CONTAMINATED SEDIMENTS

Only recently has the importance of recycling and bioaccumulation of sediment -associated toxic substances been recognized. Before the 1970s, it was believed that once these compounds were buried in the sediments, they were lost to the system. Subsequent research has shown that natural processes, such as resuspension events from storms and uptake by biota, as well as activities like shipping and dredging, can significantly affect the cycling and impact of sediment-associated contaminants.

The IJC's Water Quality Board has expressed concern for years about unaddressed areas of toxic substance contamination, and in 1985 the Board identified 42 hot spots of toxic pollution known as Areas of Concern (AOCs) in the basin. Harbors and rivers such as Duluth, Green Bay, Milwaukee, Grand Calumet, Detroit, Maumee, Saginaw, Cleveland, Buffalo, Rochester, and the Niagara River were designated as AOCs, and all but one of them have a concern for contaminated sediments. In addition, there are many other areas around the basin with contaminated sediments.

Despite the significant reduction of point source discharges of these toxic substances, biological damage is still occurring in areas with contaminated sediments as the "old" pollutants are recycled into biological systems. In fact, work conducted in the Great Lakes AOCs has identified high frequencies of fish tumors associated with these contaminated sediments (International Joint Commission, 1989). Analyses conducted by the IJC's Water Quality Board (1989) demonstrate that these tumors or fish cancers are widespread in the 42

AOCs, and restrictions on dredging as well as degradation of aquatic communities also represent associated impairments.

Economic Implications

The sediments of harbors and channels throughout the Great Lakes are routinely dredged to maintain adequate depths for ships. Concerns about open lake disposal of dredged material surfaced in the 1960's and the U.S. Army Corps of Engineers (COE) began pilot projects to test alternatives. By the late 1960's, binational Boards of the International Joint Commission (1969) recommended that open water disposal of these sediments be stopped and that disposal in confined areas be used to protect the Great Lakes. Studies showed that 77 of the 129 harbors in the U.S. portion of the Great Lakes had contaminated sediments unsuitable for open lake disposal.

The U.S. responded with Congress passing PL 91-611, which authorized section 123 of the Rivers and Harbors Act for construction, operation, and maintenance of Confined Disposal Facilities (CDFs) for about 37 Great Lakes dredging projects at a cost of about one third billion dollars. Canada also constructed similar facilities for dredged spoil disposal. Figure 1, adapted from an IJC workshop on CDFs shows the location of CDFs in the Great Lakes basin, and readers are referred to the workshop proceedings for information (International Joint Commission, 1986). Some small CDFs in the U.S., such as Duluth, Minnesota cost about $1.0 million to construct. Others such as Buffalo, Cleveland, and Detroit (Monroe, Michigan) cost $15 million, $28 million, and $51 million, respectively. With many of these facilities filled or close to filling, serious Federal budget implications exist for new CDFs as well as for the cost of maintenance dredging. Under the 1972 Great Lakes Water Quality Agreement between the U.S. and Canada, experts from both countries staff a work group under the IJC to assemble a register of dredging

FIGURE 1. Confined Disposal Facilities (CDFs) in the Great Lakes Basin.

projects in the basin. The 1975 version found that about 10 million m^3 of sediment was dredged annually in the 1960's and early 1970's, with about 80 percent being conducted in the U.S. and 50 percent in the Lake Erie basin.

The 1982 version (International Joint Commission, 1982) found that only about 5 million m^3 of sediment was dredged annually by the late 1970's. This was attributed to the low water levels of the 1960's necessitating more dredging than the high levels of the 1970's. The U.S. proportion increased to 90 percent and the Lake Erie proportion grew to 60 percent. The amount placed in CDFs or upland disposal increased from virtually none in the 1960's to 61 percent of dredged materials in the period 1975-1979. Estimates of average annual pollutant loadings in dredged materials were also made in 1982. The annual estimates were 0.23 metric tones (t) of PCBs, 1.0 t of mercury, and about 1000 t of other heavy metals like As, Pb, Cu, Zn, Cr, Cd were disposed of in open water. About 2.5 t of PCBs, 1.2 t of mercury, and 2540 t of other heavy metals were estimated to have been placed in CDFs or in upland areas. Clearly, pollutants are being disturbed. The most recent dredging register (International Joint Commission, 1990a) shows little change in volume or proportion and confined or upland disposal increased to 70 percent of all material. Confined disposal increased costs about 100 percent, and about $21 million per year was spent in the U.S. on dredging.

Policy Implications

While remedial actions to address contaminated sediments outside navigational channels have been stalled, dredging in some of the most highly contaminated areas continues for navigational purposes. This continued activity could, in large part, be responsible for the ecosystem damage and human health risks being experienced around the lakes. Old style bucket and hopper dredges are predominantly used rather than innovative technology used in Japan or Europe (International Joint Commission, 1990b). Overflow dredging practices resuspend a great deal of sediment as well as contaminants and have been the subject of controversial Congressional inquiry (U.S. General Accounting Office, 1988). Guidelines for determining "polluted" sediments are weaker in the U.S. compared to Canada for critical pollutants such as mercury and PCBs, which results in more open water disposal of contaminants.

The CDFs have been designed to leak as part of the dewatering process (International Joint Commission, 1990b) and have resulted in contaminant bioaccumulation over ambient conditions (Dobos et al, 1990). The implication is that Federal dredging activities contribute -- perhaps significantly -- to ecosystem and human health risks by resuspending toxic pollutants and that existing CDFs (without the use of more expensive best management practices for confinement) create "hazardous waste sites" that will need to be remedied under Superfund authorities. For example, the Green Bay CDF is estimated to contain about 176,000 kg of PCBs (Clean Water Action Council of Northeast Wisconsin, 1990). Will liability for cleanup end up being transferred from potential responsible parties (which released the toxic substances) to U.S. taxpayers as a result of navigation dredging?

There are also policy implications if remediation activities remain stalled. These contaminated sediments do not stay buried in one place. Storms and vessel traffic can resuspend them. Good examples involve mirex in Lake Ontario and mercury in Lake St. Clair-Lake Erie. As noted in Appendix B of the 1987 report of IJC's Water Quality Board (1987), mirex from two distinct isolated sources in sediments was widely dispersed throughout Lake Ontario in only one decade; and the mirex is now bioaccumulating in endangered Beluga whales downstream in the St. Lawrence River. Mercury from a chlor-alkali plant in the St. Clair River accumulated in sediments. In the absence of remedial actions, the mercury then dispersed widely in one decade to cause widespread contamination in Lake Erie. The implications are that if governments continue to delay remediation, the toxic substances will be transported away from harbors or hot spots to entire lakes, with possibly long-term impairments to aquatic life or increased risks to humans. It would then become impossible to conduct remedial activities. There are also policy implications if navigation dredging is stopped or delayed. A good example is the Grand Calumet - Indiana Harbor area near the steel mills of Gary, Indiana. Maintenance dredging has been stalled since 1972 because of contaminants and up to 20 feet of highly toxic sediments have accumulated. The multi-billion dollar shipping industry as well as land-based industry would grind to a halt if navigation channels are not kept open.

REMEDIAL ACTIONS UNDER THE GREAT LAKES WATER QUALITY AGREEMENT

In 1972, the U.S. and Canada made commitments to clean up the Great Lakes in signing the Great Lakes Water Quality Agreement. As understanding of the pollution problems changed, the Agreement was revised in 1978 to focus on toxic substances and was amended in 1987 to establish new institutions and multimedia pollution abatement commitments to foster cleanup of toxic substances. The IJC's historical role has been as a facilitator in providing a binational forum for experts to come to consensus on actions needed to be taken and to evaluate progress on actions that were actually taken. The dredging register described earlier represented one means of tracking progress. Other efforts to facilitate action in addressing contaminated sediments included: workshops on the use of CDFs, on options for remediation of contaminated sediments, on biological assessment techniques and a technology transfer symposium (International Joint Commission, 1986, 1988, 1989, 1990b).

Now that specific implementation cleanup commitments have been made in the 1987 amended Agreement, the IJC does not need to facilitate movement in addressing these sediments. Rather, it can now concentrate on evaluating actual cleanup progress. Important commitments made in 1987 by both countries are contained in new Annexes 2 and 14. In Annex 2, both countries committed to preparing and implementing a Remedial Action Plan for each AOC to restore uses impaired by contaminants -- no matter the sources of contaminants. Among impairments to be addressed are degradation of biological populations which might be caused by contaminated sediments as well as restoration of sediments in areas where restrictions on dredging exist. In Annex 14, specific commitments were included for conducting studies and research as well as demonstration projects for remediation of contaminated sediments. In addition, common criteria for classification of and similar procedures for the long-term management of these toxic sediments would be adopted. These commitments represent an important part of an overall multi-media strategy for managing toxic substances. The following outlines progress made since 1987.

Remedial Action Plans (RAPs)

Concurrent with the IJC's Water Quality Board identifying 42 AOCs in 1985 in the Great Lakes basin, the Board recommended preparation of RAPs for restoring impaired uses in AOCs. Since that time, the eight Great Lakes states and Ontario have been preparing RAPs for each AOC, and the U.S. and Canada formalized their commitment to RAP implementation in the 1987 Agreement. In Annex 2, commitments were made for doing the following as part of each RAP: defining impaired uses, identifying all sources of pollution, identifying remedial actions, scheduling implementation actions, establishing responsible agencies/parties, and enacting a process for evaluating the effectiveness of actions. This process represents the first truly comprehensive effort to restore seriously degraded areas in the lakes by implementing multi-media remedial actions. Central to the development of RAPs is the participation of the public -- all stakeholders -- in the process. The public has been empowered by the IJC to participate in this decisionmaking to help overcome longstanding social and political barriers. The IJC also reviews the RAPs at 3 stages to determine their sufficiency, which results in closing the loop on accountability.

Progress on RAPs has been slow, with only 18 of the 42 having been submitted to the IJC for review by mid 1990. Multi-media sources of pollution have resulted in great complexity, public involvement has taken time, and financial implications are enormous. In some AOCs such as the Rouge River, Michigan, Clean Water Act regulatory authorities have been used to require remedial dredging activities for contaminated sediments. In four AOCs (Waukeegan, IL; Sheboygan, WS; Ashtabula, OH; and Massena, NY), Superfund authorities are being utilized, and dredging followed by thermal destruction of PCBs is scheduled -- with costs up to $135 million. More detail can be found in the evaluation by the IJC's Water Quality Board (1989). Unfortunately, none of these Superfund actions cover all the sediments that are contaminated in the 42 AOCs -- only localized hot spots. In addition, many AOCs have not been rated high on the National Priorities List because the Hazard Ranking System doesn't recognize exposure routes from eating contaminated fish. Overall, as noted by the Water Quality Board (1989), progress has been too slow and funding has been too low. Nonetheless, progress is being made, studies and planning have proceeded, and now the challenge exists to make the transition to full-scale implementation.

Annex 14-Contaminated Sediments

Various research, monitoring, and demonstration projects have been undertaken by the COE over the last twenty years relating to contaminated sediments. Recent efforts are described by the International Joint Commission (1986b, 1988b, 1990b). Section 118 of the Clean Water Act was passed to meet U.S. commitments under Annex 14. It includes a demonstration program for remediating contaminated sediments in 5 AOCs (Saginaw Bay, Sheboygan, Grand Calumet, Ashtabula and Buffalo) to be completed by 1992. Innovative technology such as incineration, oxidation, extraction, and solidification are being employed. With Superfund remedies utilizing incineration, navigation dredging utilizing CDFs, and the demonstration program, full-scale implementation need not be delayed any longer.

POLICY IMPLICATIONS OF NEEDED ACTIONS

Both the U.S. and Canada have incurred a massive environmental quality deficit quite analogous to the federal budget deficits. Just as the two nations keep postponing payments to reduce the national deficit, payments to reduce the water quality/environmental deficit are overdue and the bill will be borne by our children. The toxic substances problem in the Great Lakes is so complex and solutions so controversial that significant federal leadership is needed in both countries to hasten cleanup. In no area is this more true than with in-place, contaminated sediments -- the most significant source of persistent toxic substances to the Great Lakes Basin Ecosystem.

By 1992, the research will have been completed, many active pollution sources will have been corrected, and extensive plans for remedial action will have been prepared. Will our institutions be ready to respond with full scale implementation? The policy implications for such actions are enormous. Both nations may be facing a total $50 billion corrective action program for decontamination of toxic sediments in just the Great Lakes basin. As part of a uniform, binational strategy for remediating these sediments, uniform national and international criteria and standards for contaminated sediments are needed. Extensive biological testing criteria rather than just bulk chemistry should be included. Best available technology from Europe or Japan would need to replace old style dredging equipment to minimize disturbance of contaminated sediments. Best management practices would need to be employed in constructing CDFs to ensure minimal leakage. Treatment with new technologies such as thermal extraction and destruction techniques used in AOCs as part of Superfund remedial actions should become widespread. The existing CDFs which leak or pose risks will require expensive corrective actions to be implemented. Superfund remedial and Resource Conservation and Recovery Act corrective action programs need to be focused on the Great Lakes basin to ensure that the polluter pays the cost of cleanup rather than taxpayers.

New authority and billions of dollars in appropriations would be needed for the COE to dredge outside navigation channels to completely remediate contaminated sediments in AOCs as part of RAP implementation. The Great Lakes Environmental Action Program proposed by the COE could meet this need if authorization and appropriations were provided by Congress. Additional use of Clean Water Act permit authorities can assist in requiring industry to take remedial cleanup actions. Institutionally, deadlines and funding to complete RAPs should be a priority as would a process for including public participation in siting disposal areas for dredged material following treatment. National criteria to protect wildlife from bioaccumulation of toxicants and effective nonpoint pollution control programs (to greatly reduce tributary sediment loads which result in the need for dredging) would be enacted as part of the Clean Water Act reauthorization scheduled for 1991-1992. Even improvements in data collection and reporting to the IJC under the Agreement have been recommended by the IJC's Water Quality Board (1989) to help provide a basis for addressing in-place sediments.

Canada and the U.S. can no longer just hope that the contaminated sediments problem will go away. While areas such as the James River in Virginia have a great deal of sediment moving to cover old deposits of toxic substances like kepone, much less sediment is transported by most Great Lakes tributaries.

Further delays in cleanup of these toxic hotspots mean that the contaminants will become more widely dispersed in the basin and result in either skyrocketing offsite ecosystem rehabilitation costs or perhaps long-term, essentially irreversible impairment of ecosystem and human health. Billions of dollars in annual benefits from the Great Lakes fishery and from land-based industry dependent on shipping are at risk. Also at risk is the relationship between the U.S. and Canada. Work has begun to reverse this situation under the Great Lakes Water Quality Agreement. Commitments have been made to remediate in-place contaminated sediments and the IJC is tracking progress in implementation. As noted repeatedly by the IJC, the risk of not taking immediate action is great -- not only politically but also economically and socially. Toxic contaminants stored in sediments of the Great Lakes will have to be addressed ... and will have to be addressed soon.

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MUDFLOW HAZARD DELINEATION ON ALLUVIAL FANS

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INTRODUCTION

Mudflow hazard areas are assigned a special zone designation (M) for the preparation of a Flood Hazard Boundary Map (FHBM) by the Federal Emergency Management Agency (FEMA). At the present time, FEMA guidelines have not been developed for the delineation of mud hazards, but related fan flooding can be evaluated on the basis of alluvial fan flooding guidelines. The National Flood Insurance Program (NFIP) guidelines for flood hazard delineation on alluvial fans are based on the 100-year return period flood event and a probabilistic determination of the flood hydraulics. This methodology for flood insurance mapping, promulgated by the Federal Emergency Management Agency (FEMA), has numerous limitations and may underestimate the potential flood hazard on alluvial fans. More accurate definition of flow hydraulics and flood boundaries can be predicted for unconfined flow surfaces on alluvial fans and floodplains using a two-dimensional routing model **FLO-2D**. This paper discusses the limitations of the FEMA methodology and presents the results of applying **FLO-2D** for both water floods and mudflow on alluvial fans.

FEMA'S METHOD FOR FLOOD-HAZARD DELINEATION ON ALLUVIAL FANS

Accurate flood hazard delineation on alluvial fans is essential. Alluvial fans are estimated to cover 15 to 25 percent of the semiarid southwest and are now important sites for urban expansion (DMA, 1985). Not only are the fans considered aesthetically desirable areas for their view, but their development is becoming inevitable as space on the valley bottom is exhausted. Flood insurance which fails to reflect actual flood hazard levels may encourage unsound development in flood-prone areas. Further, poorly delineated flood hazard areas may require disproportionately high payments from those in less flood-prone areas.

Special flood hazard areas on alluvial fans are defined in the FEMA 1985 Guidelines and Specifications for Study Contractors as areas "...subject to a one percent or greater annual chance of flooding, to a depth of 0.5 foot or greater, which is characterized by variable flowpaths, high flow velocity, erosion, and debris deposition." The special flood hazard area is subdivided into zones in which the flow depths and velocities differ by less than an average of 1.0 foot and 1.0 foot per second (fps) respectively. Flood hydraulics used to define each separate zone are predicted by a single channel approach developed by Dawdy (1979) or a multi-channel method recommended by DMA (1985). Flood insurance rates are based on flood hydraulics with a risk assessment of the one percent return period flood event for any point on an alluvial fan that exceeds the shallow flooding criteria of 0.5 foot.

The limitations of the FEMA methodology are multifarious and raise concern over whether the method is sufficiently conservative to generate accurate hazard mapping. The most important limitations associated with determining flow hydraulics on alluvial fans with the FEMA method are:

* FEMA's method assumes an equal probability and random distribution of the channel across a given fan contour. The fan is assumed to have a uniform surface at locations equidistant from the fan apex.

The application of the FEMA method is limited to very idealized, undeveloped, uniform topography alluvial fans (a smooth cone surface). Local relief, upstream and downstream controls, and fan entrenchment are not considered. The flow is assumed to be confined to an identifiable channel which migrates over the fan during the flood event. The channel avulsion factor of 1.5 is arbitrary and without basis. Loss of the channel and shallow overland flow are not accounted for by the procedure.

* FEMA's method does not consider fan development and flood-control measures which effect the flow path and peak discharge.

The assessment of flood hazards are generally associated with developed alluvial fans where the flow paths have been altered by buildings, levees or debris basins. FEMA's method assumes the flow is unobstructed or unconstrained. "If the flow tends to follow previous flow paths or is guided by structures in the flood path, the probability of an event on the fan is different from that assumed in the standard method." (Dawdy et. al., 1989).

* Fan hydraulics predicted with the FEMA method are based on empirical channel geometry relationships for ephemeral streams.

The exponents of the channel geometry regime equations used in the single channel method were based on the hydraulic geometry of ephemeral channels in the semiarid United States (Dawdy, 1979). These equations for velocity V and depth d as a function of discharge Q are:

$$V = 1.5\ Q^{0.2} \qquad\qquad d = 0.07\ Q^{0.4}$$

The coefficients were derived by assuming that the channel stabilized at a point where the change in depth is related to the change in channel width W according to the equation:

$$dd/dW = -0.005$$

The validity of these empirical equations and their suitability for alluvial fan channels has not been verified. The regime relationships predict channel geometry for mildly sloped semiarid streams and do not reflect the geometry of incising or avulsing channels found on steep sloped alluvial fans. FEMA's single channel method also does not account for the variation of channel geometry in a down-fan direction.

* FEMA's single channel method assumes a critical depth flow regime.

The critical flow depth assumption has not been verified. Supercritical flows on steep alluvial fans are not uncommon (French, 1986). Considering the variable slope and roughness of alluvial fans and the transient nature of flood flows, an assumption of critical flow may result in underestimated flow velocities.

* FEMA's method assumes there is no attentuation or surging the peak discharge as the flow progresses over the fan.

Flood wave attenuation and surging cause large variation in the flow depth and velocity as it progresses over the fan. "On the lower part of the alluvial fan it is not reasonable to expect the model to be accurate, particularly if there is infiltration and storage of flood flows on the alluvial fan..." (Dawdy, et. al., 1989). Flow storage, variable roughness and topography, fan rainfall and obstructions will affect the peak discharge. On large alluvial fans the increase in discharge from fan rainfall alone can be significant.

* Fan hydraulics are assumed to be unaffected by sediment concentration.

The FEMA method predicts flow hydraulics for water only. A sediment concentration of 40 percent by volume would bulk the peak discharge by 1.67 times that for water. Besides discharge bulking, high sediment concentrations can result in a significant variation in the fluid properties of viscosity and yield stress.

* FEMA's methodology employs the log Pearson type III probability distribution to assess peak discharge return periods.

The application of the log Pearson type III probability distribution for alluvial fan or hyperconcentrated sediment flows is unproven. Mudflows and water floods can have substantially different peak discharges for the same magnitude rainfall.

In summary, three conclusions can be drawn: 1) the FEMA method for the prediction of flood velocity and depth on alluvial fans does not represent a realistic assessment of the physical processes active on alluvial fans; 2) there has been no verification of the method assumptions, empirical equations or the results; and 3) risk assessment based on the predicted velocity and depth boundaries is uncertain because the method assumes an equal probability for the velocity and depth for any point on a given elevation contour.

Most fan flood studies are conducted on fans that are developed or have development proposed. Considering the limitations of the FEMA method and the lack of an historical flow record on virtually all fans, a consultant may not have a clear direction of how to proceed. Flood mitigation design or a master drainage plan may be requested and perhaps street flow, flow through subdivisions, and culvert and channel discharge will have to be evaluated to define the extent of the flood inundation. Additionally, the consultant may propose a flood containment wall to effectively remove a subdivision from the delineated hazard area. To conduct these analyses, the consultant may be required to apply the FEMA method to predict the flood hydraulics and he/she will be faced with the following dilemmas: 1) The FEMA method is applicable to only uniform topography fans; 2) There are no published guidelines on the limits of its applicability; and 3) It absolutely should not be used for flood mitigation design. Alternative methods for the prediction of flood hydraulics should be considered for both flood inundation and design of mitigation measures.

ALTERNATIVE METHODS FOR HAZARD DELINEATION

FEMA suggests that alternative methods should be applied in the analysis of alluvial fan flooding where development or modified flow paths will affect flow hydraulics. According to FEMA Guidelines (p. A5-1, 1985),

> "In portions of alluvial fans in which natural alluvial fan processes may not occur, such as in areas of entrenched channels, areas protected by flood control works, and heavily developed areas, the study contractor should exercise good engineering judgement in determining the appropriate methodology or combination of methodologies."

This language enhances the prospects for developing innovative approaches and applying new technology, but the technical review process discourages this initiative by repeatedly requesting additional information and money for evaluation. Revisions to flood insurance maps in areas where alluvial fan flood predictions have been refined using an alternative methodology have to be supported by appropriate documentation. The information must include a comparison of the original FEMA mapping and hydraulics and the alternative results. Further, an explanation for the superiority of the alternative methodology must be presented.

TWO-DIMENSIONAL FLOOD SIMULATION ON ALLUVIAL FANS

How should the flood hazard be delineated for various return period events analyzing both water floods and viscous mudflows? In a variety of alluvial fan projects, a two-dimensional flood routing model **FLO-2D** has been applied. This multifaceted model can simulate viscous mud or water flow over surfaces with complex topography and roughness. Split flow, shallow overland flow and flow in multiple channels can be modeled. In developed areas, the effects of flow obstructions such as buildings or walls which limit storage or constrict flow paths can be simulated. The model also has a street flow component. A detailed description of **FLO-2D** and its verification is presented in "Flood Hazard Delineation on Alluvial Fans and Urban Floodplains," by O'Brien (1990).

FLO-2D has been utilized to: analyze split flow around a hillside from overbank flooding of a concrete-lined channel; define flooding in an area of coalescing alluvial fans; design a flood containment wall in a alluvial wash; predict the water flooding of an urban floodplain; investigate mudflow inundation of an urbanized fan.

FLO-2D uses a central difference, finite difference routing scheme and uniform grid elements in the application of the continuity and momentum equations. Its important attributes are:

* Overland flow on unconfined surfaces is modeled in two-dimensions.

* Channel flows are routed with a variable geometry cross section to represent river reaches.

* The flow regime can vary between subcritical and supercritical.

* Viscous mudflows can be simulated.

* Return flow to the channel from the floodplain is simulated.

* Buildings and obstructions are simulated by assigning area and width reduction factors to the grid elements.

* Streets are modeled as shallow rectangular channels.

* The data file can be created with AutoCAD-DCA software.

Linking **FLO-2D** with AutoCAD-DCA was a substantial improvement in the modeling technique. Topographic maps are digitized into a file, then AutoCAD-DCA creates an overlaid grid system and extracts the point elevation for each grid. A linkage program creates a data file in the proper format to run the model. Revisions to the data file to accommodate changes in the scale of the grid system or the area of study are simple to incorporate.

To apply the model the following approach is employed:

1. Establish the hydrology and various return period flood hydrographs for the study fan at the fan apex. If necessary use **FLO-2D** to develop the flood hydrograph from the upstream watershed and route it to the fan apex.

2. Create the grid system encompassing the study area with AutoCAD-DCA.

3. Use **FLO-2D** to route the flow in the channel through the study area or over the unconfined alluvial fan.

4. Evaluate potential flood scenarios which may include: failure of the channel along selected points of the water course; relocation of the inflow hydrograph; obstruction or redirection of the flow path.

5. Identify areas of particular interest that may require a more detailed simulation such as alternative locations for proposed levees, bridges or buildings.

For ungaged watersheds, runoff hydrographs from various return period rainfall events should be developed at the fan apex. This can be accomplished with various hydrologic models such as HEC-1 or alternatively, **FLO-2D** could be used. The flood hydrographs can then be routed over the alluvial fan using **FLO-2D**. Mudflows are simulated with a corresponding sediment concentration hydrograph. The sediment concentration can be varied to maximize the peak discharge. Each runoff event can be routed both as a water flood and a mudflow.

An example of hazard delineation with **FLO-2D** for Telluride, Colorado is shown in **Fig. 1**. These results were obtained by simulating channel overflow and street flow.

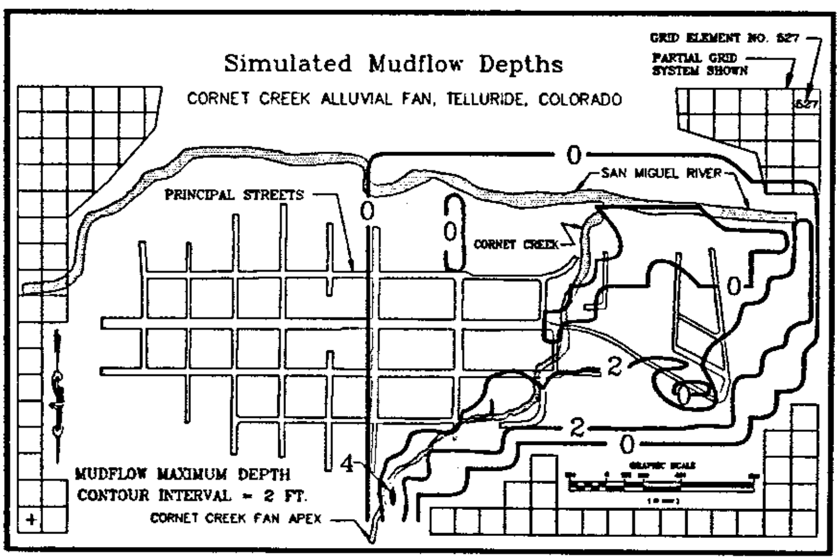

Figure 1. Simulated Mudflow Depths, Telluride Colorado.

A second example illustrates the results for water routing in a alluvial wash, Hiko Springs Wash, near Laughlin, Nevada (**Fig. 2**). Compare these results with **Fig. 3** computed with FEMA's method. Flow hydraulics were predicted near the downstream extent of the wash for the design of a flood containment wall. The wall design would have been based on a flow depth of 1.5 feet and velocity of

6 fps for the 100-year event using the FEMA results. **FLO-2D** predicted a flow depth of 2.7 feet and a velocity of 11.0 fps. The hydraulics predicted by the FEMA method would have resulted in a less conservative wall design.

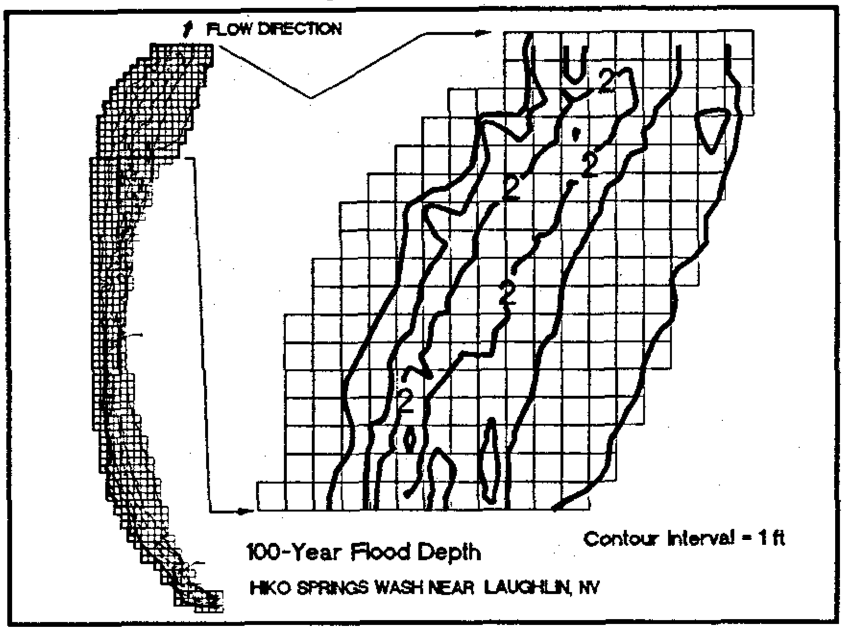

Figure 2. 100-Year Flood Depths, Hiko Springs Wash.

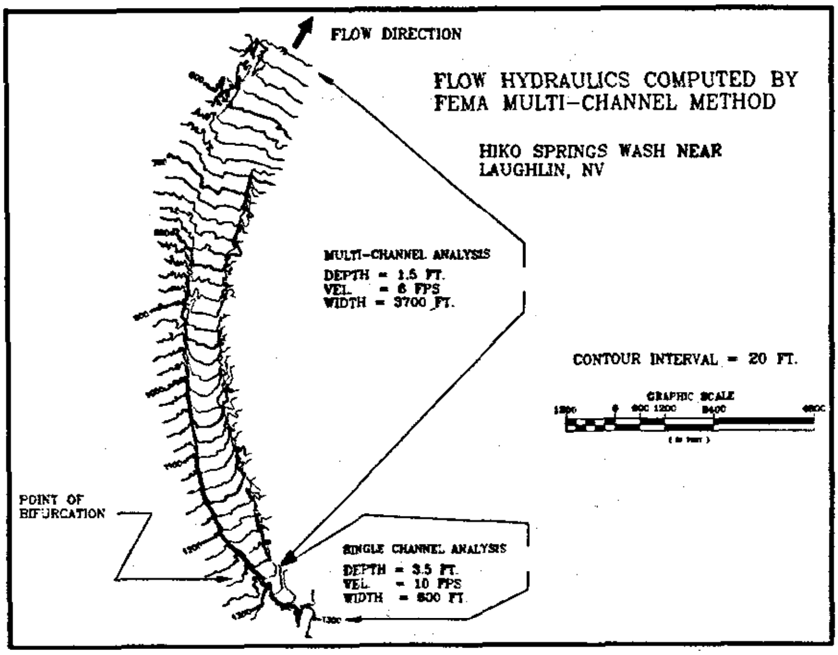

Figure 3. Flow Hydraulics Using FEMA Method

The ultimate goal of flood hazard delineation is the establishment of flood boundaries and insurance rates. The two elements required to achieve this goal are the prediction of hydraulics and the risk assessment for any point on the fan. The existing FEMA method embodies risk assessment with the prediction of hydraulics.

Is there a better method to establish risk? Risk assessment can be accomplished through the investigation of multiple flow scenarios. Is it appropriate to analyze various flow scenarios and what constitutes a viable flow scenario? Engineering judgement is required in choosing various scenarios. For example, where will the channel plug or be partially obstructed? What probabilities should be assigned to alternate channel migration patterns?

The prediction of alluvial fan hydraulics with the **FLO-2D** model is more accurate and definitive than the FEMA method. Channel or culvert blockage and channel migration can be simulated. Can more accurate insurance maps be prepared as a result? The capability to simulate various return period events is a decided advantage over the FEMA method. Mudflows can be analyzed for each event, another obvious advantage. Assigning probabilities to each scenario is a difficult endeavor, but as complex as this task might appear, would it not be more realistic than FEMA's assumption of equal probability for channel location across a contour?

CONCLUSIONS

Risk assessment and flood hazard delineation for alluvial fan flooding should be determined with a more sophisticated model than presented in the FEMA guidelines. The FEMA methodology is limited to undeveloped, unconfined fans with unaltered flowpaths. It is not applicable for the prediction of mudflow hydraulics, urbanized or channelized fans or the design of structural flood mitigation. A two-dimensional routing model **FLO-2D** provides an alternative to the FEMA method. This model is a flexible tool for the hydraulic engineer to accurately determine peak flow hydraulics, identify areas of inundation and evaluate options for flood containment.

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NUTRIENT ENRICHMENT IN RUNOFF AND LAND MANAGEMENT

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ABSTRACT

Due to easier identification and control of point sources, nonpoint inputs of nitrogen (N) and phosphorus (P) from agricultural runoff, now account for a larger share of all surface water inputs than a decade ago. Accurate simulations of sediment-associated (particulate) N (PN) and P (PP) loss are needed for more efficient evaluation of relative management effects on the eutrophic response of a water body, in agricultural areas characterized by high erosion rates. A regression equation relating soil loss and nutrient enrichment (ER) in runoff, was developed from simulated rainfall studies and used to predict PN and PP transport in runoff. Inaccurate predictions were attributed to use of simulated rainfall and disturbed soil, however, limited field information is available. Thus, ER of PN and PP in runoff from 28 Southern Plains watersheds, were calculated for periods of up to 12 years. The consistently greater ER of PN (3.80) than PP (2.60) observed, was attributed to the fact that the former is mainly of lighter organic origin compared to the inorganic orgin of PP forms. Slope and intercept values of the logarithmic soil loss-ER relationship, were greater for reduced compared to conventional tillage practices at each watershed location. Making these values a function of surface area and density of eroded material and particle size ER, rather than total soil loss, may improve PN and PP predictions.

INTRODUCTION

Nonpoint source pollution of lakes and impoundments by the transport in agricultural runoff of eroded soil and agricultural chemicals, such as N, P, and pesticides, is recognized as one of the major water quality problems in the U.S. (USEPA, 1984). The role of soluble and PN and PP in accelerating the biological productivity of surface waters is well documented (Schindler, 1977; Vollenweider and Kerekes, 1980). The term particulate includes organic matter and N and P sorbed by soil particles eroded during runoff and may account for up to 90% of total nutrient transported in runoff from cultivated land (Schuman et al., 1973a, b). While soluble N and P (SP) are immediately available for biological uptake, PN and PP are less readily available and may be a long-term source of these nutrients to aquatic biota.

Selective erosion of clay-sized particles in runoff has led to the concept of enrichment ratios (ER), defined as the ratio of nutrient content of sediment (eroded soil) to that of source soil, which is used to predict PN and PP loss (Menzel, 1980; Sharpley et al., 1985). Particulate N and PP concentrations of runoff are calculated from total N and P contents of surface soil (mg kg^{-1}), respectively, and sediment concentration (g L^{-1}).

$$PN = \text{Soil total N} * \text{Sediment concentration} * PNER \qquad [1]$$

$$PP = \text{Soil total P} * \text{Sediment concentration} * PPER \qquad [2]$$

Enrichments ratios for PN (NER) and PP (PER) are predicted from soil loss (kg ha^{-1}):

$$\text{Ln ER} = i - s \text{ Ln (Soil loss)} \quad [3]$$

From limited field data, Menzel (1980) obtained slope (s) and intercept (i) values of 0.20 and 2.00, respectively, while Sharpley (1985) obtained values of 0.16 and 1.21, respectively, from simulated rainfall studies. Use of these latter values by Sharpley et al. (1985, 1988), provided reliable estimates of PN and PP loss in runoff. Inaccuracies, however, were attributed to the use of simulated rainfall and disturbed soil and that preliminary data indicated relationships between soil loss and ER varied between watersheds (Sharpley et al., 1985). Limited information is available, however, on the relationship between soil loss and measured ER's in runoff from watersheds under natural rainfall conditions.

This paper presents information on the enrichment of PN and PP in runoff from 28 unfertilized and fertilized, grassed and cropped watersheds in the Southern Plains for periods of up to 12 years.

MATERIAL AND METHODS

Study Area

The location and management of the 28 watersheds used in this study are presented in Table 1 and are representative of agricultural land use in the Southern Plains region of Oklahoma and Texas. Tillage practice varied between watersheds. At Bushland, reduced tillage (*RT*) consisted of stubble mulch tillage, with sweeps to kill weeds and leave the crop residue on the surface for wind erosion control. Conventional tillage (*CT*) at El Reno consisted of ploughing (chisel, moldboard, and sweeps on FR5, FR6, and FR8, respectively), with *CT* wheat on FR7 converted to no till (*NT*) in 1984. At Ft. Cobb, *CT* consisted of ploughing (chisel and/or moldboard), followed by harrowing and disking prior to planting. Conventional tillage at Riesel consisted of chisel ploughing and harrowing and at Woodward, ploughing (sweeps and disking). For wheat, fertilizer N and P was applied during fall planting with additional N applications broadcast in late February or early March. For the other crops, fertilizer was broadcast during harrowing in March at Ft. Cobb and December or January at Riesel, at rates determined by soil test recommendations (Table 1).

Watershed runoff was measured using precalibrated flumes or weirs equipped with FW-1 stage recorders, with 5 to 15 samples collected during each runoff event. The samples were composited in proportion to flow, to provide a single representative sample, which was stored at 277 K until analysis. Aliquots of each runoff sample were centrifuged (266 km sec^{-1} for 5 min) and filtered prior to SP determination by the colorimetric method of Murphy and Riley (1962). The same method was used for TP following perchloric acid digestion of unfiltered samples (Olsen and Sommers, 1982). Particulate P was calculated as the difference between TP and SP. Particulate N was determined as total Kjeldhal N content of unfiltered runoff by automated procedures (USEPA, 1979). Suspended sediment concentration of runoff was determined in duplicate as the difference in weight of 250 mL aliquots of unfiltered and filtered runoff after evaporation (378 K) to dryness.

Table 1. Watershed management and mean annual flow-weighted concentration (mg L^{-1}) and amount (kg ha^{-1} y
of sediment, soluble P, and particulate P and N in runoff.

Watershed	Study period	Management[+]	Fertilizer applied N (kg ha^{-1} yr^{-1})	Fertilizer applied P (kg ha^{-1} yr^{-1})	Sediment	Runoff (cm)	Soluble P Conc.	Soluble P Amt.	Particulate P Conc.	Particulate P Amt.
Bushland, Texas										
G10A	1984–1988	NT 3-year	0	0	268	3.27	0.25	0.08	0.06	0.21
G11A	1984–1988	NT wheat	0	0	547	3.46	0.22	0.08	1.02	0.36
G12A	1984–1988	NT sorghum	0	0	1039	6.18	0.19	0.12	0.91	0.79
G10B	1984–1988	RT fallow	0	0	579	1.99	0.20	0.03	2.32	0.41
G11B	1984–1988	RT rotation	0	0	984	2.86	0.19	0.06	1.60	0.41
G12B	1984–1988	RT	0	0	2324	0.75	0.16	0.01	1.59	0.09
El Reno, Oklahoma										
FR1	1977–1988	Native Grass	0	0	47	10.01	0.11	0.11	0.09	0.09
FR2	1977–1988	Native Grass	29	2	34	9.13	0.13	0.13	0.12	0.09
FR3	1977–1988	Native Grass	0	0	39	8.54	0.13	0.10	0.14	0.09
FR4	1977–1988	Native Grass	29	2	49	7.67	0.30	0.23	0.11	0.07
FR5	1978–1988	CT wheat	75	16	2596	9.27	0.27	0.25	1.74	2.99
FR6	1978–1988	CT wheat	76	12	5003	8.42	0.37	0.18	1.83	2.62
FR7	1978–1988	NT wheat	56	13	271	7.59	0.42	0.43	0.60	0.35
FR8	1978–1988	CT wheat	56	13	2675	6.29	0.27	0.19	1.45	1.66
Ft. Cobb, Oklahoma										
FC1	1982–1988	CT peanuts/grain	46	19	19842	13.40	0.19	0.23	4.36	5.84
FC2	1982–1988	CT sorghum rotation	43	18	20803	9.51	0.22	0.20	7.19	6.84
Riesel, Texas										
Y	1976–1982	Mixed‡	19	4	599	11.45	0.17	0.16	0.44	0.51
Y2	1976–1982	Mixed	10	1	901	12.65	0.15	0.16	0.58	0.70
Y6	1976–1982	CT cotton/oats/	54	14	2083	8.25	0.06	0.05	1.37	1.05
Y8	1976–1981	CT sorghum	47	12	1363	9.60	0.07	0.06	1.05	0.82
Y10	1976–1982	CT rotation	71	15	4015	14.37	0.07	0.08	1.70	2.38
Y14	1976–1982	Introduced grass	42	0	603	15.17	0.11	0.19	0.26	0.35
W10	1976–1981	"	21	0	74	11.18	0.10	0.11	0.09	0.11
SW11	1976–1982	"	40	8	870	13.10	0.36	0.17	0.62	0.75
Woodward, Oklahoma										
W1	1977–1988	Native Grass	0	0	40	0.57	0.19	0.01	0.37	0.02
W2	1977–1988	Native Grass	0	2	565	1.46	0.24	0.02	1.26	0.22
W3	1978–1988	CT wheat	95	23	8781	4.85	0.42	0.23	4.49	0.38
W4	1978–1988	NT wheat	85	23	640	4.63	0.36	0.29	1.00	0.49

[+]NT, RT, and CT represent no till, reduced till and conventional till practices.

‡60% bermuda grass and 40% a 3-year rotation of cotton, oats, and sorghum.

The major soil types at Bushland, El Reno, Ft. Cobb, Riesel, and Woodward are Pullman clay loam (fine, mixed, thermic, Torrertic Paleustolls), Kirkland silt loam (fine, mixed, thermic, Uertic Paleustolls), Cobb fine sandy loam (fine-loamy, mixed, thermic, Udic Haplustalfs), Houston Black clay (fine, montmorillonitic, thermic, Udic Pellusterts), and Woodward loam (coarse, silty, mixed, thermic, Typic Ustochrepts), respectively. Surface soil samples (0-50 mm depth) were collected annually in March at four sites near the flume of each watershed, composited, air dried, and sieved (2 mm). Total N content of each soil sample was determined by a semi-micro Kjeldhal procedure (Bremner and Mulvaney, 1982), total P by perchloric acid digestion (Olsen and Sommers, 1982), and Bray P by the Bray I procedure (Bray and Kurtz, 1945).

RESULTS

Concentrations and Amounts

At each location, a reduction in tillage decreased erosion (Table 1). At El Reno and Woodward, soil loss from ***NT*** wheat (FR7 and W4) was 8 and 7%, respectively, that from ***CT*** practices (FR5, FR6, FR8, and W3). As expected, soil loss was least from native grass compared to cultivated watersheds at each location employing these practices. Similarly, PN and PP loss was greater from ***CT*** than ***RT*** and ***NT*** watersheds, where similar amounts of fertilizer were applied (Table 1). In contrast to PN and PP, SP concentrations were greater from ***NT*** (FR7 and W4) compared to ***CT*** watersheds (FR5, FR6, FR8, and W3), even though similar amounts of fertilizer P were applied to both practices. (Table 1). This difference in SP concentration between ***NT*** and ***CT***, results from leaching of crop residue P and a build up of P in the surface soil with reduced mechanical mixing of ***NT***. Nevertheless, PN and PP accounted for the major proportion of total N and P, respectively, transported from grassed (81 and 57%), ***NT*** (93 and 70%), ***RT*** (95 and 90%), and ***CT*** (98 and 89%) watersheds.

The loss of sediment and associated N and P varied with agricultural management and may be attributed to physical and chemical effects of crop cover on erosion and nutrient content of surface soil, respectively. These effects can be evaluated by determination of the enrichment of TN and PP in runoff for each watershed and location.

Relationship between sediment and nutrient transport

The enrichment of PN and PP decreased with an increase in soil loss as exampled by data for ***NG*** (FR2) and ***CT*** (FR6) watersheds at El Reno (Fig. 1). A similar situation existed for the other watersheds, covering a range in ER (1 - 40 and 1 - 16 for PN and PP, respectively) and soil loss (0.05 - 19800 kg ha^{-1}). However, the enrichment of PN (average value of 3.80) was greater (significant at the 1% level) than PP (average value of 2.68) for all events, watersheds, and management practices.

Particulate N and PP enrichment was significantly related (at the 0.1% level) to soil loss for all watersheds and management practices (Fig. 1 and Table 2). Slope and intercept values of this logarithmic relationship for PN, however, were consistently greater than for PP (Table 2). Further, both PN and PP slope values for the ***NG***, ***NT***, and ***RT*** watersheds were in general, greater (significant at the 5% level) than slope values for ***CT*** watersheds at each location (Table 2).

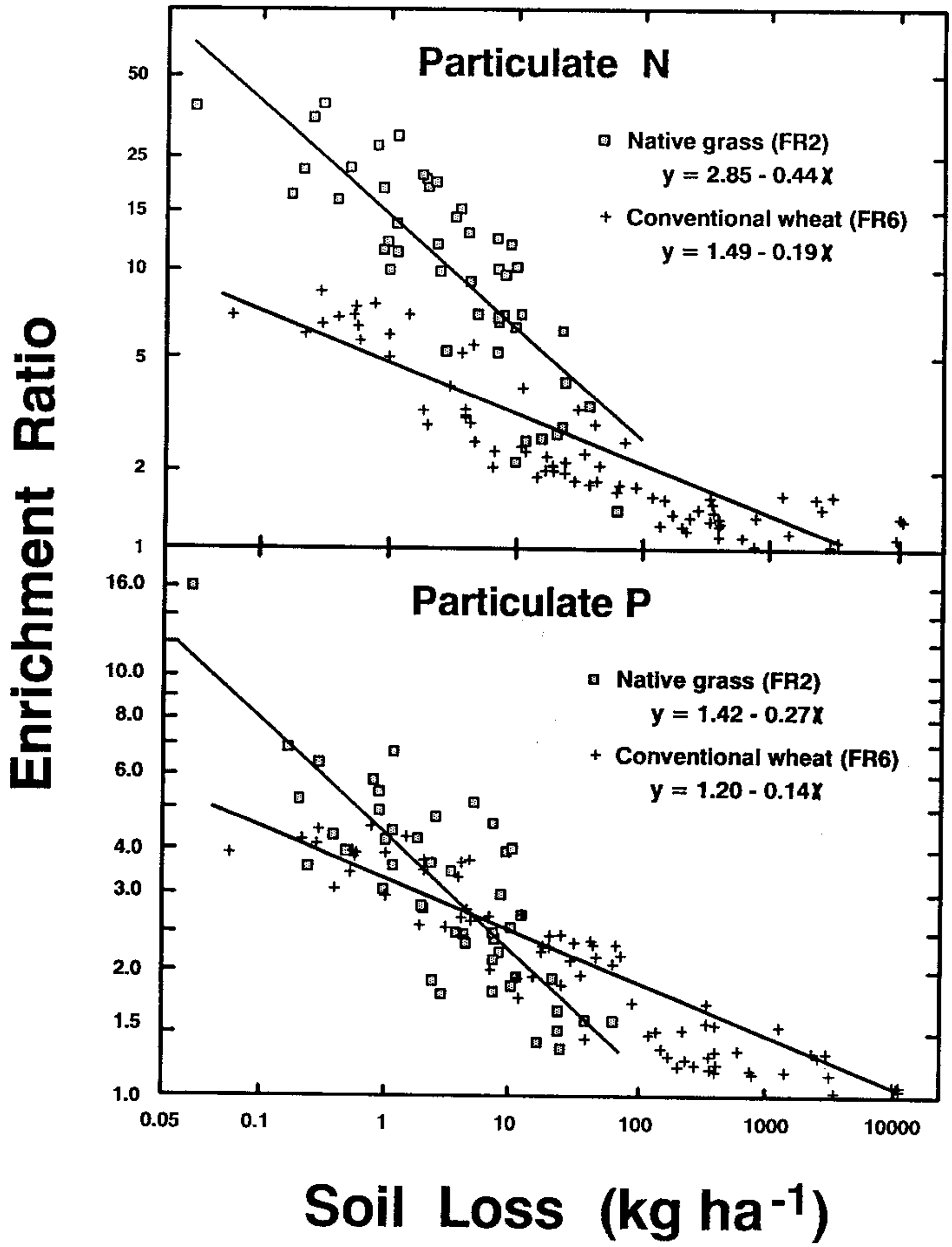

Figure 1. Relationship between soil loss and enrichment of particulate N and P in runoff from native grass (FR2) and conventional tilled wheat (FR6) watersheds.

Table 2. Regression analysis of the logarithmic relationship between soil loss and enrichment ratio†

Watershed	Mgt.	Particulate P			Particulate N		
		Slope‡	Intercept	r^2	Slope	Intercept	r^2
Bushland, Texas							
G10A	NT	-0.24a	1.62	0.94	-0.33a	2.63	0.86
G11A	NT	-0.24a	1.59	0.95	-0.32a	2.59	0.84
G12A	NT	-0.24a	1.73	0.86	-0.32a	2.54	0.88
G10B	RT	-0.21a	1.72	0.92	-0.33a	2.85	0.84
G11B	RT	-0.20a	1.45	0.95	-0.33a	2.59	0.94
G12B	RT	-0.20a	1.35	0.81	-0.29a	2.18	0.86
All		-0.22	1.55	0.81	-0.30	2.50	0.81
El Reno, Oklahoma							
FR1	NG	-0.22ab	1.55	0.61	-0.35ab	2.29	0.76
FR2	NG	-0.27a	1.42	0.67	-0.44b	2.85	0.69
FR3	NG	-0.17b	1.33	0.68	-0.33a	2.47	0.65
FR4	NG	-0.23ab	1.37	0.76	-0.34a	2.55	0.65
FR5	CT	-0.16b	1.21	0.86	-0.18c	1.35	0.85
FR6	CT	-0.14b	1.20	0.87	-0.19c	1.49	0.80
FR7	NT	-0.24a	1.56	0.73	-0.26ac	1.78	0.77
FR8	CT	-0.16b	1.27	0.77	-0.21c	1.54	0.74
All		-0.18	1.35	0.76	-0.29	2.04	0.67
Ft. Cobb, Oklahoma							
FC1	CT	-0.20a	1.95	0.71	-0.27a	2.86	0.72
FC2	CT	-0.18a	1.94	0.78	-0.28a	3.14	0.92
All		-0.19	1.91	0.72	-0.27	2.91	0.74
Riesel, Texas							
Y	M	-0.22a	1.43	0.75	-0.36a	2.11	0.78
Y2	M	-0.20a	1.43	0.73	-0.24b	1.62	0.81
Y6	CT	-0.18a	1.43	0.82	-0.24b	1.78	0.74
Y8	CT	-0.18a	1.37	0.93	-0.34a	2.44	0.93
Y10	CT	-0.19a	1.47	0.84	-0.25b	1.88	0.79
Y14	IG	-0.28b	1.81	0.87	-0.37a	2.62	0.78
W10	IG	-0.30b	1.79	0.65	-0.44c	2.34	0.79
SW11	IG	-0.32b	2.13	0.85	-0.38ac	2.43	0.83
All		-0.20	1.49	0.79	-0.30	2.09	0.76
Woodward, Oklahoma							
W1	NG	-0.31a	1.90	0.93	-0.34a	2.37	0.91
W2	NG	-0.27a	1.89	0.81	-0.33a	2.04	0.86
W3	CT	-0.18b	1.94	0.78	-0.23b	2.41	0.68
W4	NT	-0.25ab	2.23	0.80	-0.28ab	2.58	0.80
All		-0.23	2.05	0.76	-0.27	2.39	0.74

†All relationships are significant at 0.1% level.

‡Slopes for a given watershed, followed by the same letter are not significantly different at 1% level.

Nevertheless, when the data for each watershed at a given location were combined, significant (0.1% level) relationships between soil loss and enrichment ratio were obtained (Table 2). Similarly, combining the data for all watersheds gave the following regression equations:

$$\mathrm{Ln}\ (\mathrm{PNER}) = 2.14 - 0.27\ \mathrm{Ln}\ (\mathrm{Soil\ loss}) \quad r^2 = 0.67 \qquad [4]$$

$$\mathrm{Ln}\ (\mathrm{PPER}) = 1.58 - 0.20\ \mathrm{Ln}\ (\mathrm{Soil\ loss}) \quad r^2 = 0.68 \qquad [5]$$

DISCUSSION

It is apparent that the enrichment of PN and PP in runoff sediment from the Southern Plains watersheds, varied as a function of nutrient form and management practice. The consistently greater enrichment of PN than PP in runoff from all watersheds may result from the difference in origin of the parent material transported. In the case of N, particulate forms transported are mainly of organic origin, where as PP is comprised mainly of inorganic material. Consequently, preferential transport of lighter organic than inorganic material during runoff will result in a greater loss and enrichment of PN than PP. Knoblauch et al. (1942) and Neal (1944) also observed higher organic matter (4.31) and TN (4.12) than TP (1.84) ER's in runoff from Collington sandy loam in New Jersey. Similarly, Sharpley (1985) found ER's for organic particulates (2.00 and 1.61 for organic carbon and TN, respectively) to be greater than for inorganic particulates (1.56 and 1.52 for clay amd PP, respectively) for 6 soils varying in physical and chemical properties under simulated rainfall.

Slope and intercept values of the logarithmic soil loss - ER relationship for all data (Eqs. [4] and [5]) varied from those obtained in earlier studies by Menzel (1980) (-0.20 and 2.00, respectively) and Sharpley (1985) (-0.16 and 1.21, respectively) and used in predicting particulate nutrient losses (Sharpley et al., 1985). It may, thus, be inappropriate to use constant slope and intercept values for different management practices, where the runoff - surface soil interaction may differ greatly. For example, in grassed and reduced tillage practices, organic matter may contribute a greater proportion of particulate material transported. This is substantiated by the significantly greater regression slope and intercept values obtained for these practices, compared to conventional tillage (Table 2). Further, at low runoff and consequently low soil loss, TN and PP enrichment will be more a function of the transport of lighter organic matter than heavier soil particulates.

As the relationship between ER and soil loss is logarithmic, predicted values of ER will be affected more by a unit quantity of soil loss at low values of loss (<50 kg $ha^{-1}yr^{-1}$) than higher values (>500 kg $ha^{-1}yr^{-1}$). Consequently, making slope and intercept of Eqs. [4] and [5] a function of factors affecting soil loss or runoff, such as rainfall intensity, vegetative cover, and managment practice, should improve predictions. This may involve use of specific surface area and density of eroded material, and enrichment of particle size fractions, rather than total sediment loss. Although conservation tillage practices were effective in reducing erosion, they may not be as effective in controlling nutrient loss. As a greater amount of P and N associated with organic matter and crop residues may be transported in runoff from conservation compared to conventional tillage practices, control of organic as well as soil material, must be considered in the former practice.

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SEDIMENT CHARACTERISTICS OF HIGHWAY STORMWATER RUNOFF

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ABSTRACT

The paper summarizes and discusses the results of an analysis of the sediment and associated pollutant characteristics of highway runoff, and the observed influence of several factors on pollutant levels in the runoff from highway sites. The presentation emphasizes total suspended solids (TSS) concentrations and toxic heavy metals, which are present in highway stormwater at significant levels. The analysis was performed on a data base comprising nearly 1000 separate storm events at a variety of sites in the United States.

Significant differences in runoff quality were determined to exist between highway sites in urban versus rural settings, and between normal storm runoff and that resulting from the melting and washoff of snow accumulations. These differences are characterized and discussed. No significant direct quantitative relationship between pollutant levels and average traffic density at a site was indicated by the data analyzed.

Introduction

In a study supported by the Federal Highway Administration (FHWA), we assembled and analyzed data drawn from a number of different investigations which conducted field monitoring of highway stormwater runoff quality (1). The source studies were completed over the past 10 years and were either directly or indirectly supported by FHWA. Monitoring data from 993 separate storm events at 31 highway runoff sites distributed among 11 States were examined. Data sets for most sites consistently included about 10 pollutants, including heavy metals, nutrients, oxygen demanding substances and suspended solids. The source characteristics and mechanisms involved in the generation of highway runoff pollutant loadings were considered, and the characteristics of different highway sites were compared.

This presentation emphasizes the analysis procedures used, and the results obtained for total suspended solids (TSS) and heavy metals.

Data Analysis Procedures

Pollutant concentrations in the studies examined were reported either as event mean concentrations (EMCs) or as the concentrations in discrete sequential samples collected at intervals during a single storm event. The EMC is defined as the average pollutant concentration present in the total volume of runoff from a storm event. It is equal to the total pollutant mass discharged divided by the total volume of the runoff. In most cases, the data reported by the individual study results were based on a flow-weighted composite sample collected over the entire storm event, which provides the EMC directly. For studies that collected a set of sequential discrete samples during storm events, the reported data were analyzed to estimate the EMC, by integrating the hydrograph (plot of flow rate vs time) and pollutograph (plot of concentration vs time).Total mass divided by the total runoff volume provided the estimate of the EMC.

We investigated the hypothesis that probability distributions of all EMCs could be represented by the lognormal distribution. This initial assumption was made because it is consistent with similar findings for a variety of other storm-generated pollutant discharges (2,3). The assessment of the lognormality of EMCs in highway runoff was made by examination of probability plots of individual pollutant EMCs in the storm events monitored at each of the sites.

The median and coefficient of variation of the EMCs for all of the storms monitored at each of the sites, were used to completely describe the magnitude and variability of pollutant discharges

Table 1 Runoff EMCs at Milwaukee I-794 Site

EVENT	DATE (MDY)	RUNOFF (in.)	TSS (mg/l)	Pb (mg/l)	CL (mg/l)
1	6/18/76	0.72	176		
2	7/28/76	0.28	111	1.10	59
3	7/30/76	1.39			
4	8/5/76	0.04	92	1.10	110
5	8/13/76	0.58	146	3.10	20
6	8/25/76	0.12	179	1.70	45
7	8/28/76	1.17	173	1.80	13
8	9/9/76	0.85	87	0.90	10
9	9/19/76	0.28	61	0.80	21
10	10/30/76	0.15	193	2.60	422
16	3/27/77	0.20	201	1.80	220
17	3/28/77	0.85	416	2.50	62
18	5/31/77	0.18	86	0.80	24
19	6/5/77	0.66	119	1.10	16
20	6/5/77	0.46	475		
21	6/8/77	0.24	185	1.70	45
22	6/10/77	0.04	170	1.50	118
23	6/11/77	1.15	127	1.20	13
24	6/17/77	0.61	169	2.10	30
25	6/30/77	0.76	109	1.10	16
26	7/3/77	0.33	63		
27	7/17/77	1.66	26		
28	8/4/77	0.17			
29	8/5/77	0.14			
30	8/5/77	0.15			
31	8/13/77	0.93			
32	8/28/77	0.79	128		
33	9/23/77	0.01	433		
34	9/23/77	0.11	228		
35	9/24/77	0.77	90		

	N	30	25	17	17
	Mean	0.68	172	1.59	69
	Median	0.32	140	1.46	39
	COV	1.88	0.70	0.44	1.47

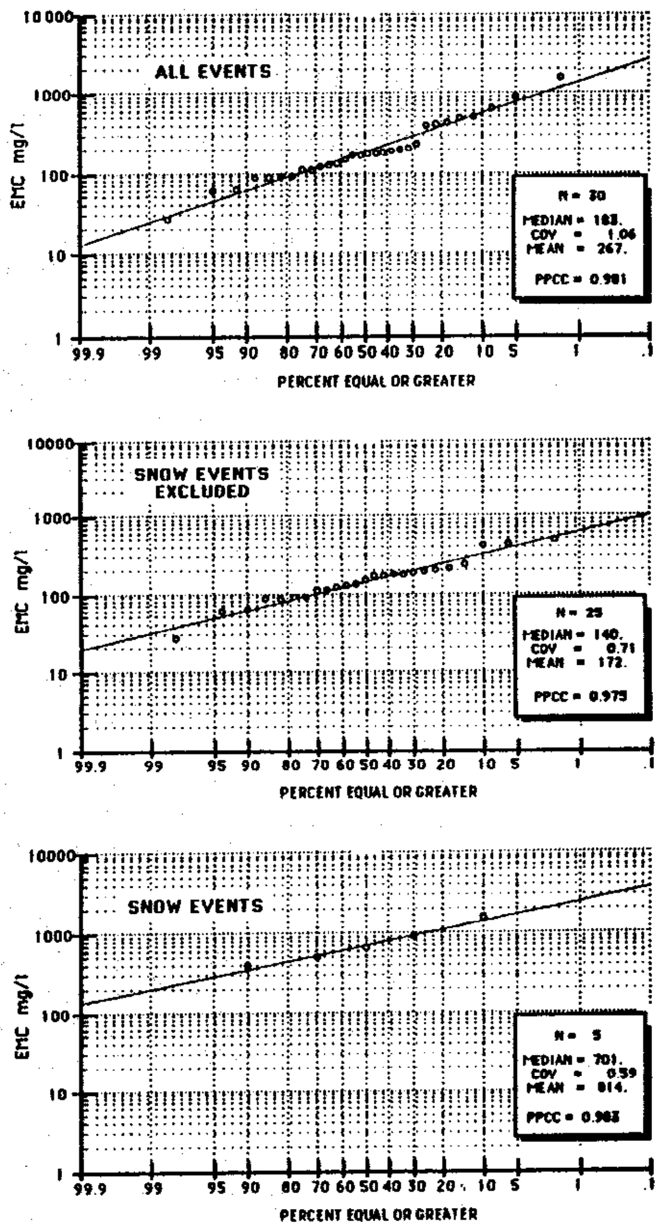
SNOWMELT / SNOW WASHOFF EVENTS

11	2/23/77	0.10	1576	13.10	1,030
12	3/3/77	0.11	632	5.00	13,300
13	3/3/77	0.43	496	5.00	425
14	3/12/77	0.21	886	7.40	299
15	3/17/77	0.18	387	3.90	1,063

	N	5	5	5	5
	Mean	0.21	814	6.98	3,404
	Median	0.18'	701	6.24	1,131
	COV	0.64	0.59	0.50	2.84

TSS = total suspended solids
COV = coefficient of variation

Figure 1
Probability Distribution of TSS EMCs at Milwaukee I-794 Site



characteristic of that site. Since the coefficients of variation were found to fall within a consistent, relatively narrow range, the median for each pollutant at a site (designated the site median concentration (SMC)) was used for comparing different sites and examining the influence of physical and other site factors of interest.

Table 1 indicates the type of initial data summary that was prepared, using the Milwaukee I-794 site as an illustration. The tabulations were developed in spreadsheet format to provide a convenient record of all the data. This format permits visual inspection to provide a qualitative appreciation of the magnitude, range and variability of EMCs of pollutants in the runoff, and organizes the data so that the desired analyses can be conveniently performed. The statistics listed at the bottom of each column (mean, median and coefficient of variation) are computed from the EMCs using a procedure that assumes they follow a lognormal probability distribution. A test of the validity of this assumption is illustrated by Figure 1, which shows probability plots of the EMC data for TSS and Lead. The match between the plotted points and the line that represents a lognormal distribution of the data set are typical of the vast bulk of the data sets in the full data base. The variability of pollutant EMCs in stormwater runoff is quantified by the value for coefficient of variation listed in Table 1 for each pollutant, and the slope of the probability plot.

Assigning a distribution allows one to define the central tendency of the EMCs (the median value), and the spread in the values which is characterized by the coefficient of variation (COV). Estimates of other useful items of information can also be made. For example, for the illustrated site data, one can determine that for TSS (median = 140 mg/l, COV = 0.70), 95 percent of all storm events at the site are estimated to have EMCs equal to or less than 400 mg/l.

The probability distributions in Figure 1 show the expected probability of each observed EMC as a plotted point, for comparison with the theoretical lognormal cumulative distribution shown by the line. Lognormality is judged by how well the plotted points correspond with the theoretical straight line. Also tabulated on each plot is the Plotting Position Correlation Coefficient (PPCC), which provides a statistical measure of the goodness of fit between data and lognormal distribution (4,5).

We concluded that the assumption of a lognormal distribution for runoff pollutant EMCs is a satisfactory practical approximation for all pollutants at all of the study sites.

Highway Runoff Quality - Site Median Concentrations (SMCs)

The SMCs for total suspended solids (TSS) and heavy metals for all of the highway sites are summarized by Table 2. The data summaries represented by this table are based on data that exclude snow melt / washoff events, and reflect runoff conditions common to all areas of the country. Snow related events were analyzed separately and results are discussed later.

The data in Table 2 are sorted into two groups. Sites with an average traffic density (ADT) in excess of 30,000 vehicles per day (VPD) were assigned to the Urban highway classification. Otherwise the site was assigned to the Rural group. The eight sites in the latter grouping include all five of the sites described as"rural" in the land use descriptions, and three sites that were classified as suburban.

Probability plots of the highway site SMCs are presented in Figure 2. In the figure, the upper left hand plot shows the distribution of SMCs for all sites. The upper and lower plots on the right provide separate probability distribution plots for urban and rural highways, respectively. An inspection of the plots indicates that the lognormal distribution assumption holds, and that the median of the SMCs for each group is significantly different. The latter observation is tested by computing the 90 percent confidence intervals for the median. Results are summarized by the box plots presented in the lower left quadrant of Figure 2. In the box plot, the upper and lower tics represent the 90th and 10th percentiles of the site median concentrations. The upper and lower bounds of the rectangle conform to the 75th and 25th percentiles. The pinched-in section indicates the median and the 90 percent confidence range for its value.

Similar results were obtained for all the other pollutants and demonstrate that rural highways with lower traffic densities have substantially lower pollutant discharge levels than highways in urban

Table 2. Site Median Concentrations for Highway Sites

SITE NAME	SITE CODE	ADT 1000 vpd	AREA (acres)	TSS (mg/l)	Fe (mg/l)	Cu (µg/l)	Pb (µg/l)
URBAN HIGHWAYS							
LOS ANGELES I-405	CA-1	200	3.2	172			987
DENVER I-25	CO-1	149	35.3	406	14.3	104	705
MIAMI I-95	FL-2	140	1.43	67		43	623
MILWAUKEE I-94	WI-3	116	7.6	143	6.0	155	817
SEATTLE I-5	WA-1	106	1.22	93		37	451
SACRAMENTO HWY 50	CA-2	100	2.45	90	3.2	68	278
NASHVILLE I-40	TN-1	88	55.6	190	4.9	56	411
MILWAUKEE HWY 45	WI-1	85	106	334	11.7	75	738
SEATTLE SR-520	WA-2	84	0.099	244		72	1065
MINNEAPOLIS I-94	MN-1	80	21	51		20	116
WALNUT CREEK I-680	CA-3	70	2.1	218			900
ST PAUL I-94	MN-2	65	16.3	85		30	407
HARRISBURG I-81	PA-2	56	2.81	184	7.6	87	26
MILWAUKEE I-794	WI-2	53	2.1	140	6.2	88	1457
LITTLE ROCK I-30	AR-1	42	1.5	112	2.9	19	108
SPOKANE I-90	WA-7	35	0.22	119		41	173
URBAN HIGHWAYS - MEDIAN OF SMCs				142	6.2	54	400
RURAL HIGHWAYS							
EFLAND I-85	NC-1	26	2.49	20	2.4	38	11
HARRISBURG I-81	PA-1	24	18.5	25	1.0	29	91
BROWARD Co HWY 834	FL-1	20	58.3	9	0.2	5	236
VANCOUVER I-205	WA-3	17	0.28	34		17	46
SNOQ. PASS I-90	WA-4	15	0.18	43		25	65
MONTESANO SR-12	WA-5	7.3	0.28	126		36	175
PULLMAN SR-270E	WA-9	5.0	0.25	104		26	130
PASCO SR-12	WA-6	4.0	1.25	101		25	101
RURAL HIGHWAYS - MEDIAN OF SMCs				42	0.7	22	80
ALL HIGHWAY SITES - MEDIAN OF SMCs				93	3.5	39	234

ADT = Average Daily Traffic
vpd = vehicles per day

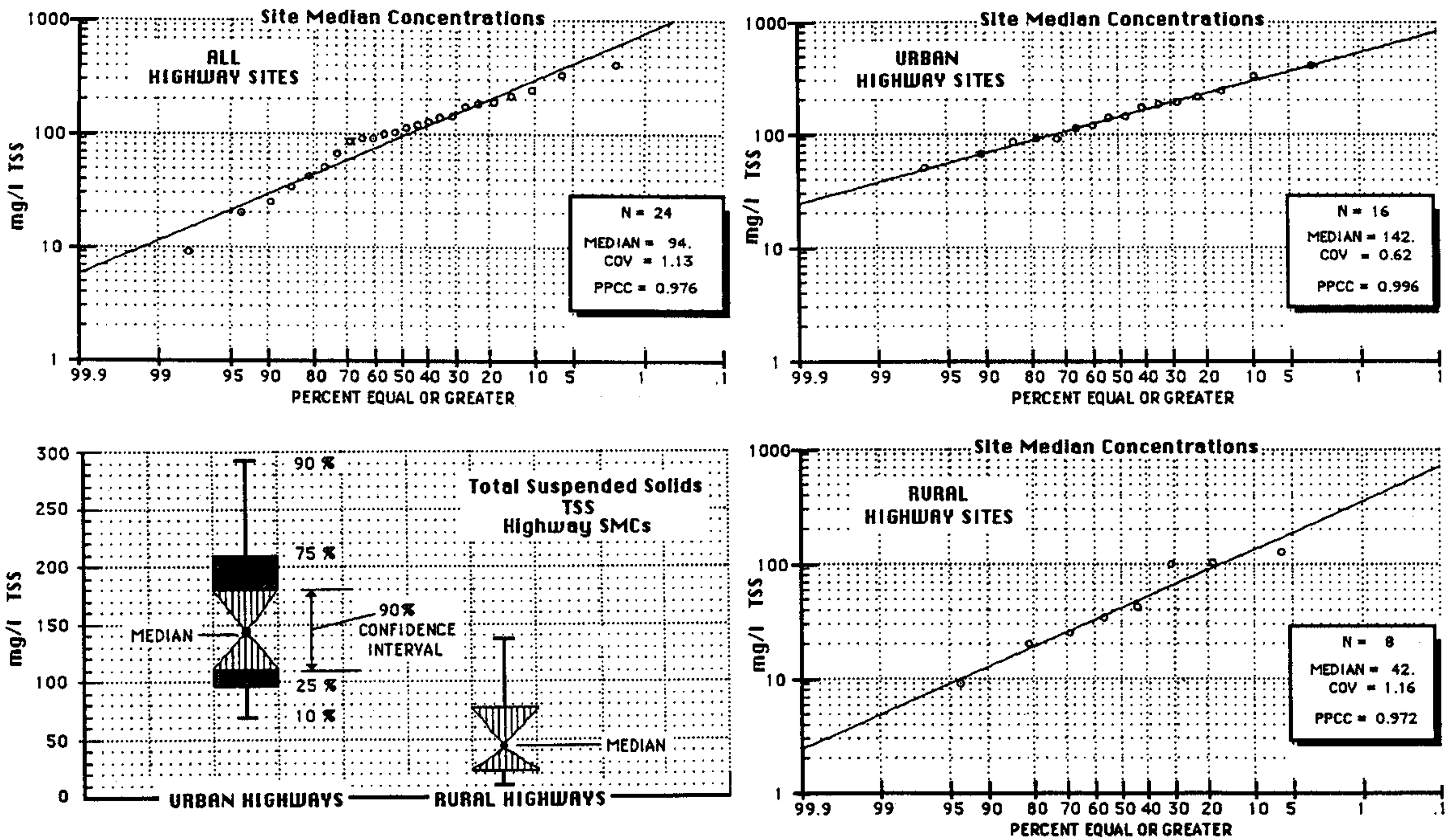


Figure 2 Highway Site Median Concentrations (SMCs) for Total Suspended Solids (TSS)

settings with traffic densities in excess of 30,000 vehicles per day. The "median" urban site is typically about 2 to 4 times higher than the median rural site, in runoff concentrations of the pollutants analyzed. The lack of overlap in the confidence bands indicates that there is a statistically significant difference in the data sets.

Effect of Traffic Density

Previous studies have suggested that the traffic density is a factor of importance (6). The effect of average daily traffic (ADT) on runoff quality was examined by testing the correlation the SMC and ADT (1000 vehicles per day). Total ADT levels used follow the conventional practice which reports ADT as the total number of vehicles which pass by the highway segment in question in both directions. The number used is a daily average value, and averages out differences based on hour of the day, day of the week, or season of the year.

It is not clear whether the distinctly different pollutant levels for rural and urban highways result primarily from the traffic levels, or from other conditions related to the character of the surrounding area (e.g., ambient air quality). Since the two factors are so strongly correlated themselves, it is impossible to distinguish them. The analysis examined the influence of ADT on runoff quality for sites in the urban highway group. Figure 3 shows the correlations between ADT and site median concentrations for total suspended solids (TSS), and a few heavy metals.. The results provide no basis for attributing a quantitative influence on pollutant level to ADT, since the "r-squared" values which quantify the variation due to ADT, are so low. For TSS, it is possible that other sources of TSS from surrounding areas could be significant contributors to accumulations on highway surfaces and levels in runoff. However, the heavy metals are more likely to relate to vehicular traffic and the correlation of most metals with ADT is also extremely poor.

Effect of Surrounding Land Use

The differences in the site groupings that was summarized by Table 2 are considered to be primarily attributable to general differences in atmospheric quality and deposition between urban and rural locales. Urban highway pollutant levels are about 3 times higher (5 times for some metals) than their rural counterparts. This general finding is supported by an independent study of runoff from highways in West Germany (7), which concluded that "the amount of pollutants discharged is not dependent on the traffic frequency, but much more on the characteristics of the area". The available data also indicate that unusual local factors can result in abnormal levels for specific pollutants. For example, non-typical zinc concentrations were found at one of the Washington State sites which was located near a zinc smelter.

Snow Washoff Events

The runoff quality characteristics presented above excluded data from snow washoff events. The data indicated these events were fundamentally different in nature than rainfall/runoff events. Some of the monitored runoff events at sites where snowfall occurs were the result of rainfall on snowpack or snowpack melt. These events were identified and segregated based upon an inspection of event chloride and total solids concentrations and were confirmed by notations in original field records, project reports, event dates, or the data sets themselves. These are not snowmelt events in the strict sense of the word, i.e., they typically did not result solely from melting snow. A snowpack traps pollutants by preventing normal dissipation by natural or traffic-induced winds, and concentrates them due to evaporation and melting. In many instances the snow pack had been on the ground for some time, thereby tending to concentrate any pollutants present, especially those in particulate form. The rainfall that gave rise to a monitored snowmelt runoff event caused additional melting and carried much of the accumulated pollutant load with it.

TSS and metal concentrations in snowmelt events may be 2 to 4 times higher than concentrations generated by most rain events. Table 3 lists the SMCs for snow-washoff events, and also the ratio of the SMC from snow events to the SMC produced by rainfall events. These entries represent multipliers that convert median rain event data to median snow event data. Since these events

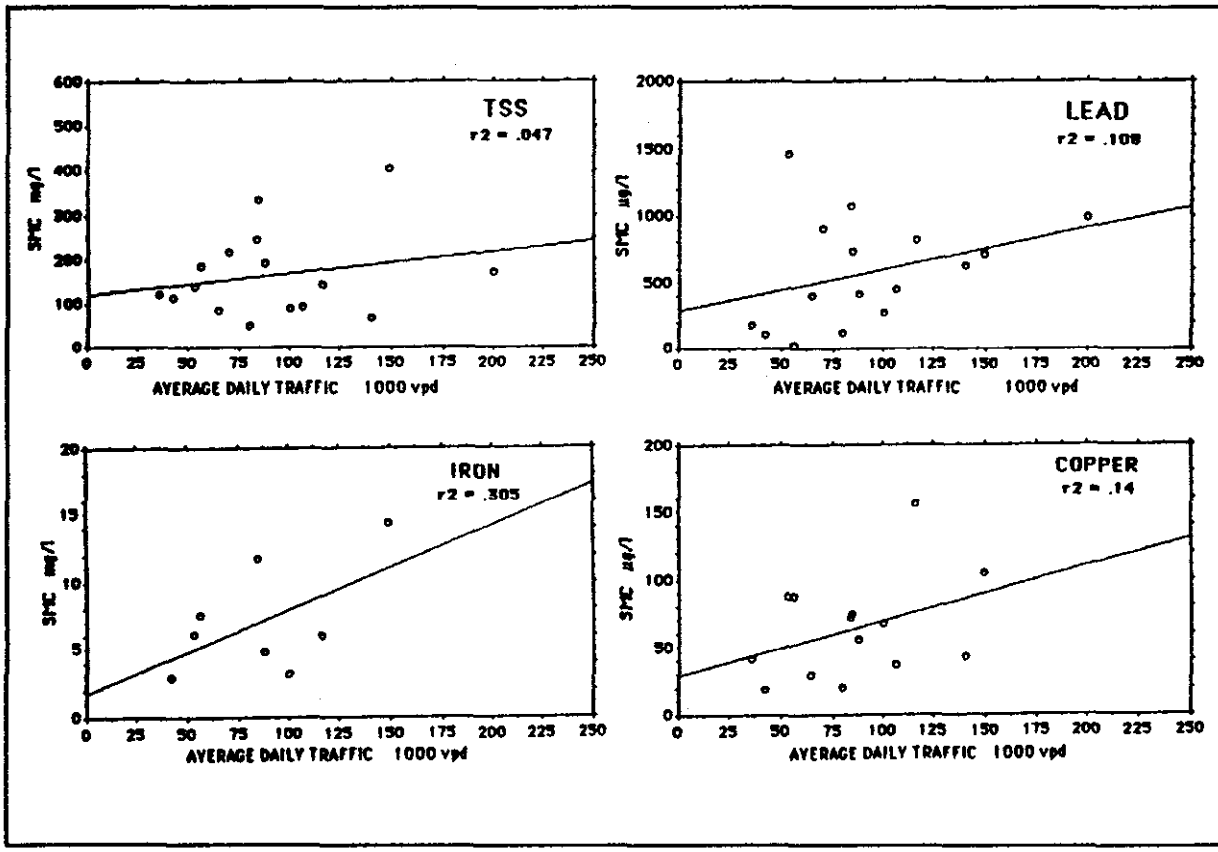


Figure 3 Relationship Between ADT and SMCs

Table 3 Pollutant Characteristics of Snow-Washoff Events

SITE			SITE MEDIAN CONCENTRATIONS (mg/l) TSS	(µg/l) Cu	(µg/l) Pb	(µg/l) Zn	RATIO snow events / rain events TSS	Cu	Pb	Zn
AR-1	LITTLE ROCK	I-30	61	36	202	303	0.5	2.0	1.9	1.8
MN-1	MINNEAPOLIS	I-94	175	78	416		3.4	3.8	3.6	
MN-2	ST PAUL	I-94	200	122	1579		2.4	4.0	3.9	
NC-1	EFLAND	I-85	11	23	20	127	0.7	0.6	1.8	2.5
PA-1	HARRISBURG	I-81	102	62	165	165	4.1	2.1	1.8	3.2
WA-8	PASCO	SR-12	155	1000	300	259	1.5		3.0	0.8
WA-9	PULLMAN	SR-270	465	40	378	310	4.5	1.5	2.9	3.1
WA-1	SEATTLE	I-5	150	63	524	402	1.6	1.7	1.2	1.1
WA-2	SEATTLE	SR-520	435	94	2004	577	1.8	1.3	1.9	2.1
WA-4	SNOQ. PASS	I-90	124	23	168	113	2.9	0.9	2.6	1.6
WA-7	SPOKANE	I-90	752	136	1213	6786	6.3	3.4	7.0	2.3
WA-3	VANCOUVER	I-205	114	50		86	3.4	2.9		2.2
WI-1	MILWAUKEE	HWY-45	375	196	1510	750	1.1	2.6	2.0	2.0
WI-2	MILWAUKEE	I-794	701	285	6239	1415	5.0	3.2	4.3	4.2
WI-3	MILWAUKEE	I-94	240	233	2285	1537	1.7	1.5	2.8	3.3
Median			**204**	**91**	**549**	**420**	**2.4**	**1.4**	**2.4**	**2.0**

occur relatively infrequently, the sample sizes are much smaller than those that were used in computing the rain event results. The data for nearly half of the sites were computed from five or fewer monitored events and, as a result, have much wider confidence intervals associated with them.

Influence of Other Factors

A variety of factors have been postulated to influence the pollutant loadings that will result from highway stormwater runoff. The most commonly identified factors include (a) surface wind speed and direction; (b) configuration (elevated, ground level, depressed); (c) pavement composition, quantity, condition; (d) design, geometry, cross-sections; (e) vegetation types on right-of-way; (f) drainage features (curbs and pipes, vegetated swales); (g) traffic characteristics (density, speed, braking); (h) vehicle characteristics (type, emission,age,maintenance); (i) vehicular transported, generated and deposited inputs; (j) maintenance practices (sweeping, mowing, weed control, repair); (k) institutional characteristics (litter laws, speed limit enforcement, car emission regulations).

It is reasonable to expect that each of these factors can have some effect on pollutant levels at a particular site, and that all other things being equal, a change in a particular factor would affect the pollutant discharge. Individual studies have identified the influence of a number of these factors at particular locations. However, even the large data base that was examined, proved to be much too small to confirm, much less quantify, effects and possible interactions across all of these possible explanatory variables. Among all the competing influences that contribute to variability and the median EMC concentration at highway sites, the overall effect of any particular factor is lost in the "noise" resulting from all other influences. Only the two conditions identified (urban versus rural surrounding area and snow washoff events) exhibited a sufficiently large and consistent influence to emerge as significant, demonstrable general influences on highway runoff pollutant levels.

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SEDIMENT, A NONPOINT POLLUTANT

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ABSTRACT

Section 319 of the Water Quality Act of 1987 established a new National Nonpoint Source (NPS) action program. This statute required States to assess their waters, and to develop NPS management programs to control and reduce specific nonpoint sources of pollution. This paper will summarize the extent of NPS sediment pollution as reported by the States, the management programs developed by the States to control it and provide an overview of the relationship between the problem and proposed solution.

INTRODUCTION

In the 1987 amendments to the Clean Water Act, Section 319 was added to further address NPS pollution. Section 319 requires States to submit NPS Assessment Reports identifying State waters which, without additional control of NPS pollution cannot be expected to attain or maintain designated uses. In addition, Assessment Reports were to identify those categories and subcategories of NPSs causing the impacts or threatening use impairment; describe the process by which the State selects best management practices (BMPs); and identify and describe State and local programs for controlling NPS pollution. Section 319 also encourages States to submit State NPS Management Programs to address the NPS problems identified in the Assessment Reports. These Management Programs are expected to include implementation schedules and milestones for the first four years and to include the State Attorney General's certification that the state has adequate authority to implement the proposed activities. These activities should include BMP identification; regulatory and non-regulatory efforts to implement the program; and creation of a framework for implementing NPS abatement procedures.

The States had eighteen months from the time the statute was enacted February 7, 1987, to complete their Assessment Reports and Management Programs. Due to the short timeframe most States utilized existing information and identified areas needing further study and analysis to fulfill the requirements of the Assessment Report. The Management Programs to a large degree focused on utilizing existing programs to satisfy the Section 319 requirements.

Status Section 319

As of June 1, 1990, 49 States and 2 territories' EPA had approved the Section 319 Assessment Reports. EPA also approved, by the same date the Management Programs of 54 States and territories, 42 fully approved and 12 partially.

SEDIMENT, AS A POLLUTANT

Sediment is defined as a solid material which has been transported from its place of origin by erosion (Davenport 1983). Erosion is defined as the detachment and movement of soil or rock fragment by water, wind, ice or gravity. There are five principal types of water erosion; 1) gully; 2) natural; 3) rill; 4) sheet; and 5) splash erosion. Soil erosion is not synonymous with soil loss, nor is soil loss equal to sediment yield (Dunne, 1978) since the sediment-producing process involves soil detachment, transport and deposition.

Sediment in water is a pollutant when it adversely impacts water quality by itself or acts as a transport mechanism for chemicals, toxic or hazardous, to man or aquatic life. For example, clean sediment can harm fisheries as fine particles fill the egg pockets in spawning areas and effectively suffocating the embryos (USEPA, 1987a). Contaminated sediment can be a major source of pollutants to overlying waters as these pollutants reach equilibrium between the soluble and adsorbed phases (USEPA, 1987b). Sediment is a significant pollutant because of its physical properties, its many potential chemical interactions, and the sheer magnitude of sediment transport (load) to the Nation's waterways. Sediment can be either suspended or deposited.

METHODOLOGY

Assessment data were taken from the approved assessment reports of 41 States, Puerto Rico, and the Virgin Islands for inclusion in this national summary. Quantitative information was not found in 10 nonpoint source assessments (Alaska, the District of Columbia, Florida, Michigan, Mississippi, Nebraska, New Hampshire, New Jersey, Oregon, and South Carolina).

All States provided a list of waters affected by nonpoint source pollution, but the informational content and quality of these lists varied considerably across the nation. For example, North Dakota provided a list that was extremely informative regarding the geographic nature of nonpoint source problems, quantification of nonpoint source problems, quantification of nonpoint source impacts to each waterbody, the range of pollutants causing nonpoint source problems, and the pollutant sources themselves. Alaska, at the other extreme, provided only a very brief summary of its nonpoint source problem, with no quantification of nonpoint source impacts.

EPA extracted and aggregated assessment data in two ways because of the nature of the data provided by the States. The first, more general approach uses statewide information provided by the States. Quantitative statewide assessment data were provided by 7 States. Thirty-four States, Puerto Rico, and the Virgin Islands provided lists of NPS impacted waters from which EPA estimated quantitative (size) information regarding waterbody type, assessment methodology used, extent of use impairment, causes of use impairment, and pollutant sources. The waterbody specific data at the State level was aggregated for the purpose of this report.

To facilitate and standardize the process of extracting this waterbody-specific information from State Assessments Reports, EPA developed a data form and a set of rules. The major rules utilized in this process are:

- State assessments that did not include measures of the size of impacted waters were not included in the analysis. Sixteen States and district of Columbia failed to meet this criterion, and only seven of these States (DE,ID,KY,MD,NM,VT,& WV) provided quantitative statewide data.

- If multiple uses were reported for any given waterbody, then the full size of impairment was counted against each use (multiple counting).

- If multiple pollutant sources were reported for any given waterbody, then the size of impairment was divided equally among each of the major sources, and subdivided equally among each of the minor sources (subcategories); that is, no multiple counting.

- If multiple pollutants were reported for any given waterbody, then the size of impairment was divided equally among each of the pollutants; that is, no multiple counting.

- No distinction was made among high, moderate, low, or slight impacts, but threatened waters were counted separately from impaired (non-support or partial support) waters.

- If a State reported impaired waters but no partially-supporting waters, then non-support was assumed.

- If pollutants were not reported but impacted size and pollutant sources were reported, then "pollutant not reported" was assumed.

RECOMMENDATIONS FOR DATA INTERPRETATION

Both absolute values and percentages are presented in the 1990 Draft Report. The reader was cautioned to utilize both types of summary data in interpreting this information. It is all too often the case that percentages are lifted from national reports and utilized inappropriately. Also, since the assessment data were provided in two forms, (i.e., statewide and waterbody-specific) and since EPA applied a consistent analytic approach to only about four-fifths of the data (34 States, PR & VI), the reader was advised to apply a ten percent, or more, uncertainty to the results presented.

RESULTS

The results are based on data presented in EPA's DRAFT Section 319 Report to Congress for fiscal year 1989 (USEPA, 1990). Siltation and in-place contaminants are the two sediment related impacts that States reported in their Section 319 Assessment Reports. Siltation is classified as a pollutant/cause and in-place contaminants are classified as a pollutant source.

Siltation

Table 1 shows the extent of siltation as reported by the States. Siltation includes suspended solids, turbidity, and sedimentation. The relative ranking associated with siltation for each waterbody type indicates it is a major pollutant of concern for rivers, lakes, and wetlands. Based upon our summary of State-reported data, siltation from NPS activities should not be considered a nationally significant pollutant for coastal waters or estuaries. At the State level, these ranking may vary considerably, however, as shown by the fact that Hawaii reported all the coastal impacts due to siltation as accounting for 27% of its impacted coastal area.

Similarly the extent to which siltation affects river water quality varies from state to state. For example, siltation impacts river uses in 86 percent of Missouri's NPS-impacted river miles but in only two percent of NPS-impacted miles in Massachusetts. Figure 1 shows the proportion of NPS-impacted river mileage affected by siltation in each State.

TABLE 1

Extent of Siltation by waterbody type (USEPA, 1990)

Waterbody	Extent	Percent*	Rank
Rivers (1) (miles)	36,273	38	1
Lakes (2) (acres)	774,763	21	2
Wetlands (3) (acres)	14,986	30	1
Coastal Water (4) (acres)	10,774	2	7
Estuary (5) (miles ^{2})	75	1	8

(1) 33 States
(2) 25 States
(3) 2 States
(4) 5 States
(5) 11 States
(*) percentage of impacted miles (non-support, partial support, plus threatened) for which pollutants are reported.

The wetlands data are misleading since filled wetlands are not included and the sample size is limited to two states: Iowa and California. The greatest share of reported lake siltation problems were reported by California (27 percent) and Oklahoma (25 percent). Figure 2 shows the share of NPS impacts to lakes that is attributed to siltation across the Nation.

Management programs were analyzed to see if states had identified existing or proposed state regulatory programs to control nonpoint sources of pollution. Two-thirds of the regulations described included information on the types of pollutants covered by the regulation, and the pollutants most commonly covered were sediment, nutrients, pesticides, priority organics, and oil and grease. Table 2 shows the existing and proposed regulation by source category for those sources generally associated with sediment production (46 State reporting).

TABLE 2

Existing and proposed NPS regulations (USEPA, 1990)

Source Category	Existing	Proposed	Total
Agriculture	148	51	199
Silviculture	47	10	57
Construction	68	19	87
Urban	63	35	98
Mining	126	22	148
Hydromodification	110	25	135

This information indicates that the major sources of sediment should be addressed by most States' regulatory programs, however, the extent to which these regulations control sediment-producing activities is unclear. The data in Table 2 seem to show a continued focus on mining and hydromodification and an increased emphasis on agriculture, urban and construction activities. For example, the proposed regulations for hydromodification represent 18 percent of all hydromodification regulations, but 35 proposed urban regulations account for 36 percent of urban regulations. The source category with the largest number of proposed regulations (51) is agriculture. According to Susan Offutt, Senior Examiner, National Resource Division, Office of Management and Budget, agriculture is the remaining, major unregulated source of environmental, primarily water, pollutants (JSWC, 1990). This analysis of existing and proposed regulations indicates the States are attempting to address agricultural NPS pollution, however, to what degree is unclear.

In-place Contaminants

In-place pollutants (contaminated sediments) are classified as a NPS in accordance the 1987 guidance on developing State NPS Assessment Reports (USEPA, 1987c). Existing NPS assessment and control efforts have not been targeted toward contaminated sediments. In addition, our understanding of the nature and extent of the problem of contaminated sediments is hampered by the fact that the method to collect and interpret sediment quality data are not fully developed.

The overall magnitude of the problem in terms of areal extent and severity has not been assessed. The potential, however, is staggering given the historic use of our waterways as a disposal area and the fact that the U.S. has over 39, 400,000 acres of inland lakes, at least 3.5 million miles of river, 32,000 square miles of estuaries, and 57,000 miles of ocean coastline (ASIWPCA, 1985). Even if only a small percentage were affected with contaminated sediment, it would represent a very significant problem. Table 3 shows the extent to which the States reported in-place contaminants as a NPS problem.

TABLE 3

Extent of In-Place Contaminants (USEPA, 1990)

Water body	Extent	Percent*	Rank
River (1) (miles)	977	0.5	0
Lakes (2) (acres)	117,770	3	7
Wetlands (3) (acres)	0	0	0
Coasted Waters (4) (acres)	40,320	3	5
Estuary (5) (miles ^{2})	908	16	1

(1) 37 States and Puerto Rico
(2) 26 States and Puerto Rico
(3) 2 States
(4) 5 States and Puerto Rico and Virgin Island
(5) 12 States and Puerto Rico
*percent of NPS-impacted area for which sources were reported

In-place contaminants were reported as a significant source for estuaries and a minor source for coastal waters and lakes. It is the largest source of NPS pollutions in estuaries. No State received EPA approval of an in-place contaminant component in their Management Program, and only two States (CT & WI) described regulatory programs that addressed in-place contaminants. In general, State Section 319 NPS Assessment Reports and Management Program did not extensively address contaminated sediments. In response to the apparent lack of State action and the need to address the serious problem with contaminated sediments, EPA has initiated a process to develop a Sediment Management Strategy. The purpose of this strategy is to establish a framework for EPA and States to comprehensively address the contaminated sediment problem. The Strategy has three major components: Assessment, Prevention, and Remediation. Presently, NPS activities are included in the Prevention component and consist of mainly targeting State Section 319 activities toward watersheds with contaminated sediment problems. Within Region V, contaminated sediment activities receive a high priority for funding under the Section 319 Program.

CONCLUSION

Siltation is a major nonpoint source pollutant in rivers, lakes and wetlands. In-place contaminants are a significant nonpoint source of pollution to the Nation's estuaries. The States are generally either prepared or are preparing to address the siltation problem as part of their Section 319 NPS Management Programs, but the benefits to be gained from these activities are not yet clear. In-place contaminants, however, are not adequately addressed in most State NPS Management Programs despite the fact that estuaries, lakes, and coastal waters are impacted by this pollutant source. Additional action is needed by both the States and EPA to address this little-understood problem.

To fully assess the impact of sediment on water quality, States and EPA need to ensure future water quality assessment work which includes the impacts of clean sediment, in-place contaminants, and documenting the effectiveness of the implementation of the State Section 319 Management Programs.

DISCLAIMER; The contents of this paper do not necessarily reflect the views or policies of the United States Environmental Protection Agency.

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RIVER IMPACTS CAUSED BY SILTATION
(FRACTION OF KNOWN POLLUTANTS)

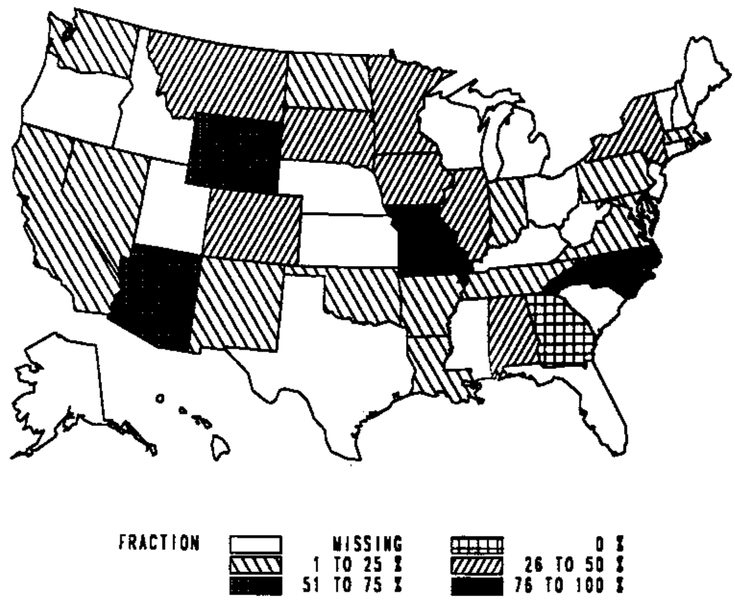

Figure - 1

LAKE IMPACTS CAUSED BY SILTATION
(FRACTION OF KNOWN POLLUTANTS)

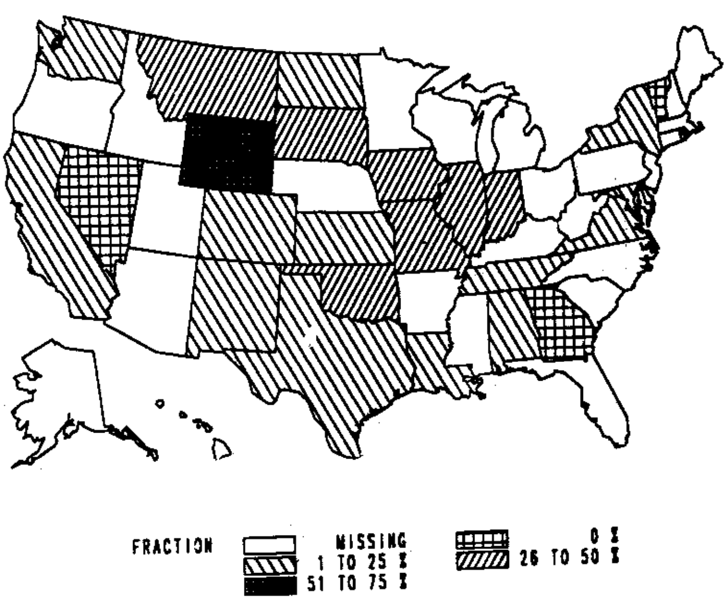

Figure - 2

EFFECTS OF FINE SEDIMENT ON FISH POPULATIONS

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ABSTRACT

To describe conditions in natural redds of steelhead trout (Oncorhynchus mykiss), we evaluated the particle size distribution of egg pockets, redd pits and tailspills, artificially constructed redds, and undisturbed substrate outside redds. Egg pockets were located in upper strata an average of 14.9 cm below the substrate surface. Egg pockets contained fewer fines (<6.35 mm) than other sites, except in artificial redds. As substrate depth increased, percent fines tended to increase at all sampling sites and variability in percent fines among sites was reduced. We observed no change in percent fines sampled from May to July, except in cleaned intrusion sites where free interstitial areas rapidly accumulated sediments. Dissolved oxygen levels declined in redds during incubation. Limitations of the data are discussed.

INTRODUCTION

Increased sediment production within watersheds can amplify the transport and deposition of sediment in stream channels. Fishery biologists have been concerned about the effects of fine sediment on salmonids for more than 60 years (Grost 1989). A large body of research suggests that deposition of fine sediments in streams is detrimental to the spawning and incubation of salmonids (Cordone and Kelly 1961, Everest et al. 1987). To determine the consequences of sediment deposition in spawning areas, researchers have linked models that predict sediment delivery with sediment-based predictions of survival to emergence (Young et al. 1989).

Much of the past research on sediment-based estimates of survival has consisted of laboratory studies that may not accurately reflect conditions in the natural environment (Everest et al. 1987, Chapman 1988). Few data are available to characterize conditions including particle size distributions, dissolved oxygen concentrations, water temperatures, and intragravel velocities in natural redds, especially at the site of egg deposition, the "egg pocket." As Chapman (1988) observed, quantitative predictors depend upon careful definition of egg pocket structure and on intensive study of egg pocket conditions. An understanding of conditions in natural redds can aid in determining the location and method of sampling to assess sediment effects (Grost 1989). Conditions outside egg pockets may not be related to conditions within egg pockets (Young et al. 1989). Knowledge of the structure and composition of the egg pocket can be applied to laboratory conditions, making better analogs of natural conditions (Chapman 1988). Knowledge of spatial differences can aid in determining the location and timing of sampling to assess sediment effects.

We incorporated an interdisciplinary approach involving biologists and hydrologists to describe conditions in natural redds of steelhead trout (Oncorhynchus mykiss). We also compared conditions in egg pockets to those in the surrounding substrate, both within the redd environment and exterior to it. We further describe temporal changes in egg pockets and the surrounding substrate as incubation progressed. Finally, we compared conditions in natural egg pockets and artificially constructed redds. The Idaho Department of Health and Welfare-

Division of Environmental Quality (DEQ) has proposed artificial redd monitoring as a technique to measure the effects of fine sediment on beneficial uses (Burton et al. 1990). The validity of this method depends on an understanding of conditions in natural redds.

METHODS

Study sites were selected in a 1.6 km reach of the mainstem Salmon River approximately 19 km upstream from Stanley, Idaho. The river flows north through a broad valley composed of alluvium and glacial deposits and drains over 500 km^2 of the White Cloud and Sawtooth mountains in the Sawtooth National Forest. The area is mostly underlain with granitic rock with a small area of Paleozoic sedimentary rock in the southeast portion of the basin. Elevations range from 3050 m at the divides to 1950 m in the study reach. Peak stream discharge typically occurs during 6 weeks in May and June during snowmelt. Base flows occur from September through January.

Steelhead trout are indigenous to and formerly spawned in the study reach. As a result of severe declines in wild fish escapements, hatchery-reared steelhead trout have been introduced in the drainage (IDFG 1984). We obtained 30 pairs of hatchery-reared steelhead trout, ranging from 60 to 80 cm in length, from the Idaho Department of Fish and Game's Pahsimeroi Hatchery. The fish were released in the mainstem Salmon River near the confluence of Hell Roaring Creek on April 20, 1990. Within 4 days we identified redds and randomly selected 26, 10 in a diversion channel adjacent to the river and 0.8 km downstream from the release site and 16 near the confluence of Hell Roaring Creek. Subsequently, these two locations will be referred to as the Hell Roaring (HR) and Diversion (DV) reach.

We applied terminology described by Grost (1989) as follows: redd, an area of the substrate modified by a salmonid during reproduction, usually elliptical and composed of a pit and tailspill. Pit, the deeper, concave, upstream portion of the redd. Tailspill, the shallower, convex, downstream portion of the redd. Egg pocket, a concentration of eggs within the redd.

On April 24, 1990, we inserted probes into redds from the downstream end of the pit and at the boundary of the undisturbed substrate and tailspill. We attempted to insert the probes adjacent to the egg pockets within the redds. Probes were designed to allow extraction of water from the interior of the redd for measuring dissolved oxygen (D.O.), temperature, and apparent velocity. Probes were constructed of continuous coil slot well-screen (30.5 cm by 3.2 cm diameter) capped on both ends with PVC fittings. An inner perforated rigid tube was attached to a 152 cm length of 6.3 mm (inside diameter) flexible tubing (Burton et al. 1990). The flexible tubing was left exposed on the substrate.

Adjacent to each natural redd, we established three additional sampling sites for: (1) construction of an artificial redd, (2) sampling natural substrate (undisturbed substrate outside of redds), and (3) measuring the intrusion rate of fine sediments into "cleaned" substrate. We selected the locations of the three sites by establishing four 3 x 3 m quadrants around each natural redd and randomly selected a quadrant for each site. Within the selected quadrant we randomly selected a location that met the following criteria: water depth of 0.37 to 0.6 m, velocity of 0.5 to 0.65 m/s, and surface substrate with diameters of 0.3 to 7 cm and 10 to 25% cobble (Cochnauer and Elms 1986). Occasionally, multiple sites were located in a single quadrant. Sampling sites were located to minimize disturbance of natural redds.

Artificial redds were constructed on April 24 with assistance from experienced DEQ personnel. An egg basket and monitoring probe were placed in each artificial redd. Burton et al. (1990) describe the specific techniques of artifical redd construction and the construction and installation of egg baskets. We obtained water-hardened hatchery steelhead trout eggs from the Idaho Department of Fish and Game's Sawtooth Hatchery and inserted 100 in each egg basket. Eggs were distributed throughout the lower portion of the baskets.

Natural substrate sites were marked with surveyors stakes. We excavated a 30 cm by 38 cm diameter depression at the sediment intrusion sites. The excavated substrate was sieved and material smaller than 6.35 mm diameter was discarded. We placed a 40 cm by 30.5 cm diameter nylon bucket (T. Lisle, USDA-Forest Service, personal communication) in a collapsed position in the depression and filled it with the material larger than 6.35 mm.

We collected substrate samples to describe the particle size distribution in the sampling sites. At monthly intervals, we randomly selected two natural redds (and the associated two artificial redds and two natural substrate sites) each in the HR and DV reaches for freeze coring. Additionally, four intrusion sites were chosen at the HR site for extraction of collapsable buckets. A tri-tube freeze coring procedure using liquid CO_2 (Everest et al. 1980) was used to extract a substrate sample approximately 30 cm long and 15 cm in diameter from the tailspill and pit in natural redds, the egg baskets in the artificial redds, and the natural substrate sites. Using definitions proposed by Young et al. (1989), we labeled cores as egg pocket samples if we observed eggs or alevins in the core.

Core samples from each site were stratified by 10 cm depths. The sediment core samples were air dried and sieved for 5 minutes on a mechanical shaker through the following mesh sizes: 128, 64, 32, 16, 9.5, 8, 6.35, 4, 2, 1, 0.85, 0.5, 0.25, 0.125, and 0.063 mm. We weighed the material retained on each sieve.

The bias associated with very large particles has been recognized and many workers have excluded them from analysis (Adams and Beschta 1980, Tappel and Bjornn 1983). We observed larger particles being lost as our cores were extracted. To avoid bias, we excluded particles larger than 64 mm from our analysis. As Kondolf (1988) observed, where the percentage of fine sediment is to be compared among sites or sampling times, the sample can be computed based on the truncated distribution. Stowell et al. (1983) defined fines as particles smaller than 6.35 mm and used the percent fines as an index of gravel quality in the Idaho batholith. We adopted their definition.

We sampled the D.O. concentration of water within the natural and artificial redds at 2 week intervals. We used a hand activated vacuum pump to extract water samples from the monitoring probes. Approximately 250 ml of water was initially extracted to flush the probe and measure water temperature. A second extraction was used to fill a 300 ml BOD bottle. Water samples were fixed and titrated in the field using a modified Winkler procedure to estimate D.O. (Standard Methods 1989). We estimated percent saturation by correcting for water temperature and elevation.

We compared the percentage of the substrate less than 6.35 mm for each location by strata using a two-way ANOVA. We performed multiple comparisons using the Tukey-Kramer procedure for unequal sample sizes (Sokal and Rohlf 1981). We examined differences between the vertical strata, differences between reaches, and temporal changes in sediment composition. We compared conditions in natural and artificial redds by examining differences in substrate composition and dissolved oxygen. We accepted P=0.05 as the level of significance.

RESULTS

Egg pockets in steelhead trout redds were located in upper strata. The mid point of egg pockets averaged 14.5 and 15.2 cm below the surface, respectively, in the HR and DV reaches. Egg pocket depth ranged from 7.6 to 21.2 cm. Egg pockets in artificially constructed redds were deeper, averaging 22.5 and 19.9 cm below the surface, respectively, in the HR and DV reaches and ranging from 15 to 27.5 cm.

We observed differences in the percent fines (<6.35 mm) between sites in the upper 10 cm strata. Within the HR reach, comparisons revealed significantly less fines in natural redd egg pockets than in pit and tailspill sites within the redd but outside the egg pocket (Figure 1). Within the DV reach, natural redd egg pockets also exhibited less fines than other sites within the redd, but differences were not significant. There was no significant difference in the percent fines in natural substrate outside the redds as compared to pit and tailspill sites in the upper strata samples from either reach. Artificial redds in both reaches exhibited significantly less fines than sites outside the redd or outside the egg pocket. There was no significant difference between the percent fines in egg pockets and artificial redds.

Percent fines were similar for the same types of sampling sites in the HR and DV reaches (Figure 1). This trend was similar for sites in natural redd tailspills and egg pockets, natural substrate, and artificial redds.

As the depth of strata increased, differences in the percent fines tended to be reduced. Within the HR reach, there were no significant differences in the percent fines between sites in the 10-20, 20-30, or 0-30 cm strata. Within the DV reach, artificial redds retained significantly fewer fines than other sites across all three strata.

Percent fines generally increased in deeper strata, particularly below egg pockets. The percent fines at egg pocket sites in the HR reach increased from 12.9% at 0-10 cm to 33.6% at 10-20 cm, and 33.1% at 20-30 cm. In the DV reach, the percent fines at egg pocket sites increased from 17.5% at 10-20 cm to 32.2% at 10-20 cm to 51% at 20-30 cm. Percent fines also increased at other sites from the 0-10 to 10-20 cm strata and remained similar at strata below 20 cm.

With the exception of the intrusion sites, there were no consistent temporal changes in percent fines. Within redd pits and tailspills, undisturbed natural substrate, and artificial redds, there was no significant change in the percent fines sampled in May as compared to July. Figure 2 illustrates the data by reach for pooled 0-30 cm strata. Similarly, no significant temporal change was detectable in the 0-10, 10-20, or 20-30 cm strata. We did not locate sufficient egg pockets to compare temporal changes in them. Within the intrusion sites, substrate cleaned to remove fines less than 6.35 mm rapidly accumulated fines between May and July. This suggests that sediment was moving through the reach and was deposited where free interstitial areas existed.

D.O. levels (expressed as percent saturation) declined in both natural and artificial redds during the sampling period. During the first two sampling dates, D.O. levels in natural and artificial redds were close to 100% saturation in both reaches (Figure 3). D.O. levels declined significantly after May 15 and averaged less than 80% saturation on July 11. With the exception of the July 11 sample

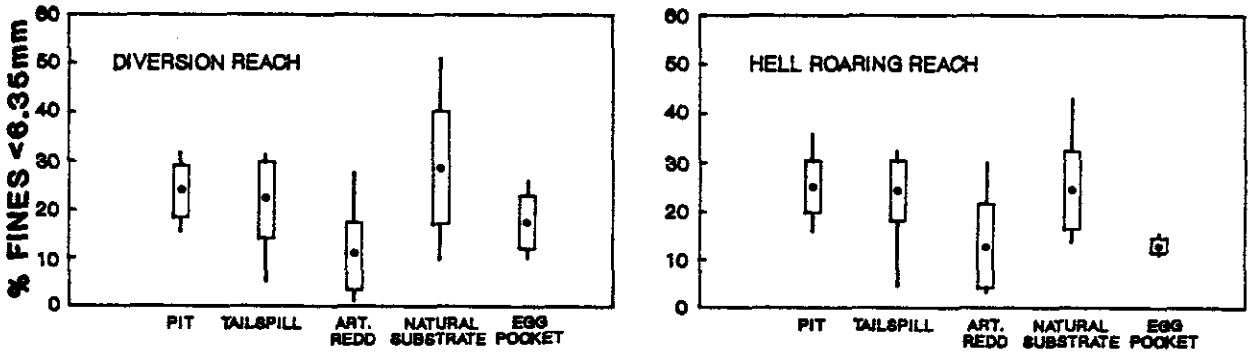

Figure 1. Percent of the substrate less than 6.35mm by sample site and reach, 0-10cm strata, Salmon River. Points, boxes, and vertical lines represent the mean, standard deviation, and range, respectively.

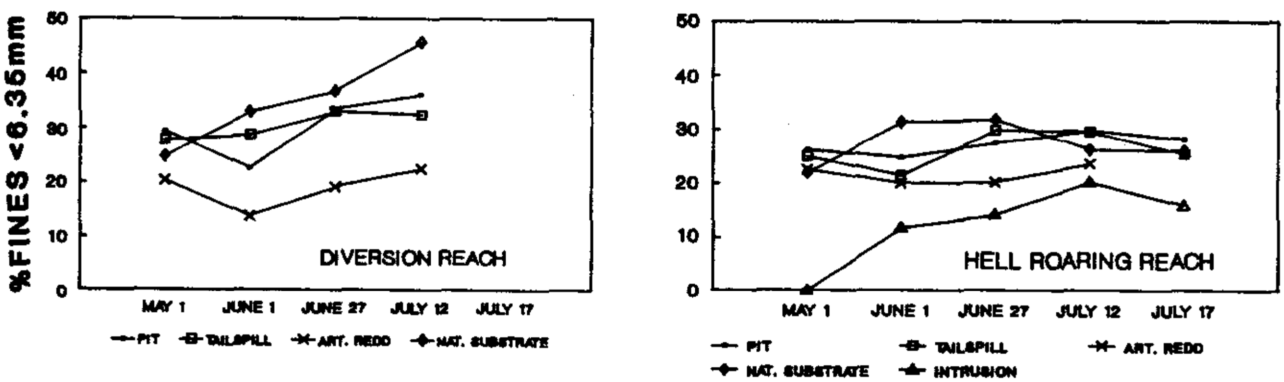

Figure 2. Comparison of the mean percent substrate less than 6.35mm by date, sample site, and reach, 0-30cm strata, Salmon River.

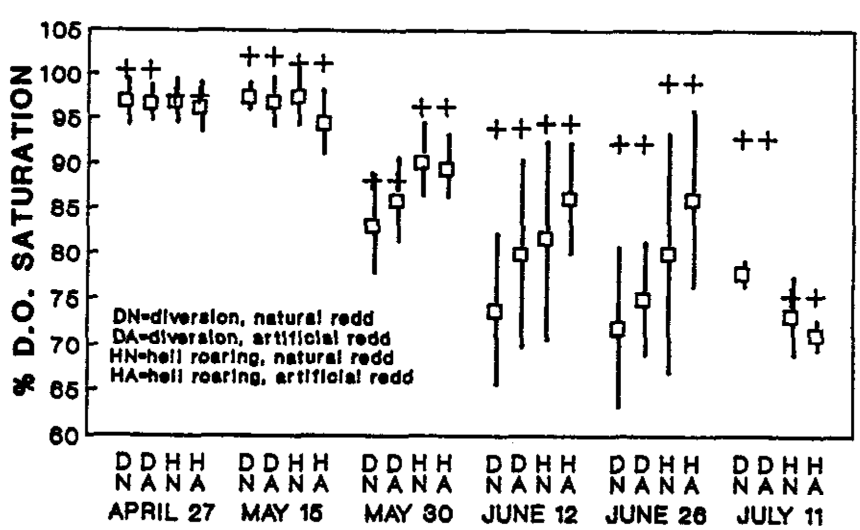

Figure 3. Means (boxes) and standard deviations (lines) of the dissolved oxygen percent saturation of water extracted from natural and artificial redds, by reach and date, Salmon River. Pluses represent the mean percent dissolved oxygen saturation of the river water column.

at the HR reach, D.O. levels in the river water column exceeded 90% saturation throughout the sampling period. The average D.O. saturation was higher at the Hell Roaring site on 5/30, 6/12, and 6/26. The difference was significant on 5/30.

The D.O. decline over the sampling period was most prevalent in natural egg pockets. We measured D.O. levels in three natural egg pockets from April 27 to June 26. D.O. levels averaged 94% saturation on 4/27, 97.3% on 5/15, 85% on 5/30, 76.9% on 6/12, and 65.1% on 6/26. We measured D.O. levels in six pits where we did not locate eggs or alevins. D.O. levels averaged 96.8% on 4/27, 96% on 5/15, 88.3% on 5/30, 80.1% on 6/12, and 81.7% on 6/26. Artificial redds maintained the largest average D.O. saturation during the sampling period, although not significantly larger than the natural redds. The D.O. levels in nine artificial redds averaged 95.5% saturation on 4/27, 94.7% on 5/15, 88.2% on 5/30, 82.6% on 6/12, and 82.7% on 6/26.

Temperatures within natural and artificial redds were generally within 1 °C of the temperature of the river water column. Water temperatures in the water column and in natural and artificial redds increased during the sampling period. Temperatures in natural and artificial redds in the HR reach averaged 3.9 °C and 3.9 °C, respectively on 4/27, and 12.2 °C and 12.3 °C, respectively on 7/11. Within the DV reach, temperatures in natural and artificial redds averaged 4.8 °C and 4.6 °C, respectively on 4/27, and 14 °C and 13.7 °C, respectively on 6/26. The HR reach maintained colder temperatures than the DV reach, probably as a result of tributary inflow.

DISCUSSION

Steelhead trout spawning in the Salmon River were apparently able to remove fine sediments less than 6.35 mm from the upper strata of the egg pocket. As Everest et al. (1987) observed, salmonids are not passive spawners, and redd building activities remove fine sediment. Our data suggest that the substrate in the egg pocket contains less fines than other sites, either inside or outside the redd. These results confirm the unique substrate composition of the egg pocket (Young et al. 1989) and tend to support the conclusions of Chapman (1988) who recommended sampling of natural egg pockets as being essential to quantitatively predicting effects of fines on salmonids.

Our inability to detect differences in the percent fines in lower strata suggests that it is critical to preserve the vertical stratification of substrate samples. Everest et.al. (1980) and Young et. al. (1989) reached similar conclusions.

The rapid increase in percent fines in the intrusion sites suggests that, where interstitial areas were free of fines, these areas retained available fines moving as bedload. Because the egg pockets also contain free substrate, sands and fine gravels could also be expected to intrude into the relatively clean gravel. We were unable to test this hypothesis as a result of an insufficient sample of egg pockets. It is also possible that a seal of fine particles, as Chapman (1988) describes, formed over the egg pocket or that hydraulic conditions above the redd in concert with a seal prevent deeper intrusion over the egg pocket. Our inability to detect differences in percent fines over time at sites outside the egg pocket suggests that interstices were filled.

Accelerated temporal depression of D.O. in egg pockets, compared to pits without eggs, suggests that developing embryos created an oxygen demand. Chapman (1988) reported that the rate of oxygen uptake increases steadily from fertilization to hatching. We observed the lowest D.O. levels in egg pockets where we observed decaying eggs and levels declined to an average of 5.2 mg/L at 65% saturation. Davis (1975) suggested a mean threshold of 8.1 mg/L and 76% saturation for successful hatching and incubation of salmonids. The presence of decaying eggs in artificial redds that did not display accelerated D.O. depression suggests that water samples from some artificial redds did not accurately reflect conditions in the redd. This may have been caused by placement of the probes, which are exterior to the egg baskets.

Several important uncertainties exist in our data. Sites in streams are not homogeneous, and conditions in individual redds vary. As a result, the wide natural variability in particle sizes within the same types of sites (i.e., across redd tailspills) makes comparison of conditions between sites difficult. A larger sample size would increase the reliability of comparisons. Also, our inability to core sample the same sites over time (as a result of destructive sampling) caused us to assume that temporal changes would be reflected in sites of the same type. This may not be a correct assumption.

We found it extremely difficult to precisely locate egg pockets in natural redds and sampled only nine. Further, placement of the probes inside the redd may have altered the structure of the egg pocket and precluded collection of accurate data describing particle size and D.O. (Young et al. 1989).

Our analysis of substrate composition applied a single index of gravel quality, percent fines less than 6.35 mm. Everest et al. (1987) and Kondolf (1988) have questioned using a single criterion to describe gravel quality. Our analysis is incomplete until we examine the distribution of other particle sizes. In addition, the geometric mean diameter (Platts et al. 1983) and Fredle index (Lotspeich and Everest 1981) have been proposed as more suitable indices of gravel quality. As Kondolf (1988) observed, geometric mean diameter and Fredle index are both heavily influenced by mean gravel size. To adequately evaluate differences in particle size distribution, several indices should be calculated for the sites by individual vertical substrata and for the composite (pooled) core. Our analysis is also incomplete until we examine the relationship between particle size, D.O., and apparent velocity.

An ultimate goal of this research is to contribute to the development of relationships between: stream sediment transport, sediment movement into redds, and the effects of sediment on egg and alevin survival. Additional research is in progress to describe the egg pocket environment of chinook salmon (Oncorhynchus tshawytscha) in the Salmon River. Our initial experiences with steelhead trout and chinook salmon illustrate that this research has many constraints and is labor intensive. As Chapman (1988) observed, the alternative to investigating natural redds is to use data of unknown accuracy and to extrapolate inappropriate data to natural redd environments.

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