

RESEARCH NEEDS FOR SECTION 208 PLANNING

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Continuous improvements are made in the control of point sources of pollution. However, even with the application of best available treatment to all sources of pollution, it is doubtful that the goals of the Federal Water Pollution Control Act Amendment (FWPCA, PL-92-500) will be attained by 1983. To a large degree this is due to pollution from non-point sources. In many regions of the nation non-point sources of pollution contribute significantly to the overall pollution problem. Unless efforts are made to control non-point sources of pollution, the 1983 interim national goals on water quality cannot be attained.

The National Commission on Water Quality reported in 1976 that non-point sources will contribute at least half of the following major pollutants to the nation's waters, after achievement of the 1977 practicable technology standard:

total phosphorus	80%
total nitrogen	50%
coliform bacteria	50%
biochemical oxygen demand	50%
suspended solids	50%

EPA estimates that 50% or more of the nation's current water quality problems are caused by non-point sources. Furthermore, non-point sources are also major carriers of toxic substances, whose effects may overwhelm pollutant reductions from point source control. For example, toxic material from urban runoff frequently exceeds that from typical industrial discharges (Westman, 1977).

The significant contribution of non-point sources to water pollution was recognized in the FWPCA of 1972. Yet, until 1975, little was done to

combat non-point sources of pollution in a systematic way. On the other hand, much money has been, and is currently being, spent on reducing pollution from point sources. Federal grants for sewage-treatment plant construction for FY 1977 alone are estimated at \$3.8 billion, close to what is being spent for the combined costs of hydroelectric dams, flood control projects, and public power plants (Westman, 1977).

If indeed 50% or more of the nation's water quality problems are caused by non-point sources, it may be expected that spending levels for non-point source pollution control will eventually be greatly increased.

Examples of Non-Point Source Categories:

The following categories are generally considered in conjunction with non-point sources of pollution.

Urban runoff: This is defined as storm and combined sewer discharges and overland or sheet urban flow. It is estimated that during intense storms 95% of the BOD load is directly contributed to runoff. Also a moderate size city may discharge between 100,000 to 250,000 pounds of lead and 6,000 to 30,000 pounds of mercury each year through storm water runoff (Pisano, 1975).

Construction activities: Pollution by runoff from sites of residential and commercial construction during the period of construction. It is estimated that about 1 million acres per year are disturbed for this purpose, generating much sediment.

Residuals disposal: This includes pollution resulting from the disposal of waste residuals (e.g. garbage, refuse, sludge from waste treatment operations, etc.) It consists primarily of runoff and leachate from improperly managed disposal sites. This category may become of significant importance in New Mexico in view of proposed storage of radioactive wastes.

Mining: Pollution generated through the extraction of minerals, ores, fossil fuels and other materials from the earth.

Forestry: Pollution caused by man-made disturbances associated with timber harvesting (i.e., timber harvesting, reforestation, fire fighting, fire prevention, etc.).

Agriculture: Activities associated with irrigated and non-irrigated crop production and animal production. Over 400 million acres are in cropland, and deliver 2 billion tons of sediment annually to streams and lakes, containing much of the 440 million pounds of pesticides used annually. Irrigated agriculture contributes to increased salt loads in rivers downstream.

Estimating Non-Point Source Pollution:

Non-point source pollution is very difficult to quantify, because it generally occurs over large land areas, and because its magnitude varies greatly with time. For example, the two billion tons of sediment contributed annually to streams and lakes comes from 400 million acres of cropland. Also a major portion of the pollutant contribution from land runoff may occur during a few intense rainstorms.

Loading of the runoff with pollutants or toxic metals is furthermore extremely variable in time and may depend on factors such as how long it has been since the previous storm, or whether or not chemicals were applied on the soil just prior to the storm.

A complicating factor in assessing non-point source pollution is further the fact that sediment and nutrients are discharged by natural processes and are part of natural aquatic ecosystems. In order then to assess non-point source pollution the natural background has to be determined, and subtracted

from man-made causes. A point in question is the quality degradation of river water along the Rio Grande in New Mexico (Wierenga and Patterson, 1972). Although the increase in total dissolved solids along the Rio Grande was more than twice as great in an irrigated portion of the river as compared to areas with no irrigation, the natural increase in total dissolved solids is significant. Studies in the San Joaquin Valley in California have shown that a large percentage of the nitrates found in drainage from the valley originates from sub-surface deposits and not from application of fertilizers. These two examples illustrate that background or natural pollution can be quite extensive, and that an assessment of non-point sources of pollution should take into account these background levels of pollution.

Efforts to manage non-point sources of pollution should be aimed at first determining the relative contributions of non-point sources to water quality degradations in a specific drainage area, for example in relation to existing point sources of pollution. Secondly, all significant contributing sources should be determined, as well as their levels of background pollution. And finally, an economic analysis should yield information as to what levels of non-point source control are achievable and at what costs.

Determination of the relative contributions of non-point sources to water quality degradation is no small task, however, either because the proper data are not available or because models used to estimate the contributions from non-point sources have not been adequately tested. To quote a recent report prepared for the EPA on loading functions for assessment of water pollution from non-point sources, "The estimation of non-point pollution is an approximate science, in its present stage of development. In some instances the term science is not appropriate." (McElroy et al. 1976). It appears therefore that research in pollution from non-point sources is very much needed. Research is necessary to determine the present contribution of non-point sources to water

pollution. Research is further needed to predict the effects of alternative management strategies on pollution from non-point sources. Thirdly, models which allow the prediction of the effects of alternative management techniques on water quality, should be combined with economic models to evaluate the costs of alternative management schemes, including cost estimates of maintaining present practices.

Research Needs for Non-Point Source Pollution:

In view of the large number of categories of non-point source pollution a complete review of past, present and future needs for 208 planning is clearly beyond the scope of this presentation. However, in reviewing the various categories of non-point sources of pollution, it becomes obvious that non-point source water pollution, whether caused by mining, agriculture, or construction consist essentially of two main mechanisms:

- A. Runoff of water with soil and toxic chemicals into open waters
- B. Transport of water and chemicals through soil into drainage systems, or into groundwater.

All categories listed in the introduction contribute to either surface runoff, or groundwater pollution, or both. Furthermore the processes involved in surface runoff or in movement to groundwater, whether from cropland, construction sites or mining operations, are very similar, and the same physical principles apply to all of them. Thus although parameters used in equations to predict sediment loss may be quite different for cropland as compared to parameters used to predict losses from a construction site, the general equations should be quite similar, and basic research performed in one non-point source category should have application in other non-point source categories. The same is true for chemical transport. The prediction of leaching from a waste disposal site is not essentially different from the prediction of nitrate losses to the groundwater from an irrigated farm. Because the physical and chemical

processes for the various 208 categories are very similar and center around runoff of water with soil and/or toxic chemicals, as well as around flow of water and chemicals in subsoil we will restrict our discussion of research needs to these two areas.

Water Pollution from Surface Runoff:

Water pollution by runoff from watersheds is a result of the sediment carried with the water and the various chemicals, associated with the runoff, either adsorbed on the sediment, or in solution. The control of non-point pollution therefore is, to a large extent, determined by adequate control of erosion. Estimating surface runoff is of primary importance, because surface runoff causes enormous losses of soil and subsequent pollution of streams and lakes with sediment laden water.

Substantial progress has been made in recent years in the development of basic mathematical models to predict runoff and soil loss. Unlike the Universal Soil Loss Equation which is an empirical equation and is discussed below, mathematical models combine fundamental principles, concepts and relationships of erosion mechanics, hydrology, hydraulics, soil science, and meteorology to simulate the erosion and sedimentation processes (Wischmeier, 1976). The advantage of such models is that they lead to a much better understanding of the actual erosion and sedimentation processes. The models have potential for predicting soil losses from specific events and for predicting spatial distribution of erosion and sedimentation. However, they are very complicated to use, and much additional research and information is necessary to allow their use on a routine basis.

A quite different approach is the use of the Universal Soil Loss Equation. This equation which combines the principal factors that influence surface soil erosion by water, has been successfully used to compute sediment losses from agricultural land.

The equation takes the form:

$$A = R.K.L.S.C.P. \quad (1)$$

where

A is the average soil loss, in tons per acre,

R is the rainfall factor, K is the soil erodibility factor,

L is the slope length factor, S is the slope-steepness factor,

C is the cropping and management factor, and P is the erosion control practice factor.

The equation has been used successfully by the Soil Conservation Service for predicting soil losses and conservation measures on agricultural land for more than a decade in the 37 states east of the Rocky Mountains, and to a limited extent in the western states. The Universal Soil Loss Equation is an empirical approach toward predicting soil loss. It is based on data from 10,000-plot-years of erosion studies at 42 research stations (Wischmeier, 1976), and as such is well founded. However, because of the need to predict soil losses other than cropland, e.g. construction sites, rangelands in the western United States, forested areas, etc., further studies are needed to improve the applicability of the Universal Soil Loss Equation. For example, a primary difficulty in using the University Soil Loss Equation for predicting soil erosion on construction areas is the evaluation of the erodibilities of subsoils, which are commonly heavier in texture than the surface soils for which existing relations have been derived. For urban construction sites an improved method of relating the soil erodibility to basic soil parameters must be developed (Roth, Nelson and Römken, 1974).

A major difficulty with the Universal Soil Loss Equation, apart from its applicability in a certain region, is that although the equation allows prediction of erosion rates in croplands it does not predict the percentage of this eroded soil that reaches a continuous stream. The equation was developed

for soil conservaton and not for sediment control, even though sediment control is presently of interest from the standpoint of non-point source pollution control. According to Wischmeier(1976), development of a better understanding of the basic sedimentation and erosion processes involved between the time when runoff leaves a field area and when it reaches a continuous stream system is one of the greatest erosion and sediment research needs. For New Mexico additional research with the Universal Soil Loss Equation is also needed for application to strip mine areas and spoil banks, and to burned areas in forests. Particular attention should be given to predict soil losses from specific high intensity rainstorms, since these cannot now be predicted accurately.

Although, as shown above, much additional research is necessary to make the Universal Soil Loss Equation a reliable tool for predicting sediment delivered to a stream, the need for quantiative estimates of non-point source pollution, the relative ease of its use, and the lack of alternative methods resulted in extensive use of the equation for prediction of sediment loading in streams and lakes. Because pesticides, nutrients and toxic chemicals are present in surface soils, stream pollution with these substances has also been predicted with the Universal Soil Loss Equation.

The principal method by which this is done is by first calculating the sediment yield and then multiplying sediment yields by factors which denote concentrations of these substances in the soil and enrichment in the erosion process. As an example let us consider phosphorus, since phosphorus is strongly adsorbed on soils and does not readily leach into the subsoil. Thus phosphorus is carried almost entirely on sediment. According to the equation (1) the average soil loss from an area is:

$$A = R.K.L.S.C.P. \quad (2)$$

The loading function, or the amount of sediment delivered to a stream is then:

$$Y = B (R.K.L.S.C.P.S_d) \quad (3)$$

where B denotes the surface of the source area and S_d the sediment delivery ratio. This is the portion of sediment delivered from the erosion source to the receptor water. The load of phosphorus coming into the stream is now obtained from:

$$YP = Y \cdot CON \cdot RF \quad (4)$$

where CON is the concentration of the phosphorus in the soil and RF the enrichment factor. The enrichment factor is a measure of the increase in the concentration of a pollutant associated with the sediment that actually reaches a stream compared to the concentration in the watershed soil. According to Stewart and Woolhiser (1976) the concentration usually increases because more nutrients and pesticides are adsorbed on fine-textured particles than on coarse particles, and more coarse particles are deposited as the sediment moves from the field area to the stream.

McElroy et al. (1976) used this procedure to predict sediment loading, and loading with nutrients, organic matter, pesticides and heavy metals.

The use of loading functions on the basis of the Universal Soil Loss Equation for estimating non-point source pollution by overland flow, has the same restriction as the Universal Soil Loss Equation. In addition, the approach suffers from lack of data on the enrichment factor, which is different for different surface soils and chemicals. The sediment delivery ratio also needs further study. For example, the delivery ratio takes into account both the deposition of sediment as it moves toward the stream and the gains from channel erosion. However, pesticides and nutrients are usually not associated with sediment from channel erosion (Stewart and Woolhiser, 1976). It should also be recognized that loading functions yield average values, usually on an annual basis. Because of the lack of data it is still difficult to make predictions of extremes over a period of years. This is of importance for New Mexico since much sediment is transported during extreme rainstorms.

Transfer of Chemicals Through Soil

When chemicals are applied on a crop or onto the surface of soil it is quite likely that a large portion of the chemical will enter into the soil with rainwater or irrigation water. In the soil a number of different processes can interact with the chemical, each of which greatly affects its ability to move to the groundwater table. Some of these processes are adsorption, ion exchange, microbial degradation, radioactive decay, plant uptake, and chemical precipitation.

The main mechanism for chemical transport in soil is mass movement with water. Diffusion is important for uptake by roots, but this process is too slow to play a significant role in the transport of chemicals to the groundwater. Pollution of groundwater is therefore largely dependent on the presence of water movement to the groundwater table. Without such water movement the risks of polluted groundwater from surface application of chemicals are virtually nil. This is one of the reasons why arid areas have frequently been proposed as storage repositories for radioactive wastes (Winograd, 1974). In order to predict whether surface applied chemicals will reach the groundwater table and in how much time, one has to know fairly precisely the rate at which water is moving through the unsaturated zone above the groundwater table. Determination of the rate of percolation of water through the unsaturated zone is extremely difficult, and remains, after many years of research, one of the most difficult tasks. The rate of percolation through subsoil depends on rainfall rates, evapotranspiration, vegetative cover, the soil water storage characteristics, the root zone, the soil and geologic conditions below the root zone, and the amount and timing of direct runoff. Stewart et al. (1975), estimated the annual percolation potential of land in the United States. According to their estimates the percolation potential varies from more than 7 inches/year in the east, to between 0 and 1 inches per year in the central United States.

Most of the western United States was omitted, because rainfall is too variable, and much of the area is irrigated. With low rainfall rates, prevalent in much of the western United States, the amount of water percolating to the groundwater becomes especially difficult to estimate. In New Mexico relatively little is known about the rates of deep percolation in the various areas of the state. Extensive research is needed, not only with regard to groundwater pollution, but also to determine if and at what rate aquifers are being recharged. This information is further needed to estimate the efficiency of irrigation systems and to predict consumptive use of crops.

As mentioned before, the rate of transfer of surface applied chemicals to the groundwater table is also dependent upon a number of physical and chemical processes in the unsaturated zone.

Adsorption is one of the main mechanisms delaying downward movement of surface applied chemicals. An example is adsorption of phosphorus on soil. Phosphorus is strongly adsorbed on most soils, and little if any pollution has been found as a result of phosphorus being leached into the groundwater. On the other hand, surface water pollution may occur due to surface runoff and erosion of surface soil containing adsorbed phosphorus. Many pesticides are also strongly adsorbed, especially on organic matter in surface soils. Unfortunately the adsorption characteristics of most pesticides and soils vary enormously, and determinations have to be made on each soil and pesticide to be able to predict leaching potential of a given pesticide. In view of the large number of pesticides continuously being introduced, it would be quite helpful if with the introduction of each new pesticide data were presented on their degree of adsorption on standard soils, and also on their rate of degradation under standard conditions. With this information soil scientists, knowing the physical and chemical characteristics of local

soils with respect to the standard soils, could at least predict the leachability and rate of degradation of the new pesticide under local conditions, and estimate its local environmental impact.

Degradation is another main mechanism which slows down the rate of movement of surface applied chemicals to the groundwater table. According to Stewart et al. (1975) the persistence of pesticides in soils is highly variable and under moderate climatic conditions may vary from less than eight months for phenoxy herbicides to more than two years for organochloride insecticides. For example, Scifres et al. (1977) found that 2,4,5-T applied to range land at three locations in east Texas (mean annual rainfall \pm 100 cm) was reduced to trace levels in soil within 7, 28 and 56 days, respectively. No residues were found below 15 cm in the soil. On the other hand the highly mobile picloram was reduced to trace levels within 56 to 112 days. Some picloram was found as deep as 60 cm, but most was restricted to the upper 15 cm. Monitoring of cumulative residues in water, soils, and vegetation showed that about 75% of the picloram was dissipated from the ecosystem within 28 days after application and over 90% was lost after 112 days. These data show the great importance of degradation on leaching to the subsoil and pollution of groundwater.

Ion exchange can also delay the downward movement of chemicals through soil. Many soils have a fairly large ion exchange capacity, particularly soils with high clay content and high organic matter content. Ion exchange reactions may be important for some pesticides, and are of great interest for predicting the composition of irrigation return flow. Because of ion exchange, precipitation and dissolution reactions accurate prediction of the composition of irrigation return flow is difficult. The chemistry of the various reactions taking place in the unsaturated surface soil during leaching with irrigation water is not completely understood and models developed to predict changes in irrigation return flow quality need more testing under field conditions.

The rate of decay of radioactive chemicals is well known, and incorporation of decay in predictive models offers no particular difficulty. A very beneficial effect of radioactive decay is that it delays the transport of radioactive chemicals through soil even further. In addition to radioactive decay, radioactive chemicals are also subject to adsorption and precipitation, which further delay the movement of these chemicals through soil. Accurate predictions of travel times of these chemicals to the groundwater cannot be made however, unless information is available on the chemical and physical interaction between the chemical and the soil or soils through which the chemical must pass.

Movement of nitrates to groundwater is also of interest, because high levels of nitrate make this water unsuitable for drinking water purposes. During the past ten to fifteen years much research has been done to determine the pollution potential of surface applied nitrates. Nitrates move quite readily with water, and several instances of high levels of nitrates in groundwater have been reported. Yet, even though much has been learned about the behavior of nitrates in soil, it is still difficult to predict for most field conditions the rate of movement of nitrates to groundwater. According to McElroy et al. (1976) methods to predict leaching of nitrates to groundwater are not available, and "local experience, data and expertise must be relied upon". The problem with nitrates is that they are subject to transformation. Thus NO_3^- may be reduced to NH_3^+ , which is adsorbed and does not move readily. Nitrogen may also be reduced to nitrogen gas and escape into the atmosphere. This reduction takes place mainly in saturated soils in the presence of a carbon source, but has also been reported for unsaturated soils in microsites. Present knowledge of the conditions in the soil, which determine the form of nitrogen in soil is far from complete. Unless more is known of the various reactions taking place in a given soil

profile or in geologic formations accurate predictions of nitrogen movement into groundwater cannot be made. Plant uptake of chemicals in the root zone of crops reduces the chance for such chemicals to be leached to the subsoil. Plant uptake is one of the main factors which cause a reduction in nutrient concentration in surface soil. Nutrients which are not taken up or otherwise interact with the soil, may be lost by leaching to the subsoil.

Precipitation of salts in soil added with irrigation water is of importance in irrigated areas. It has been shown that precipitation of salts in soil can be encouraged by minimized leaching, and that precipitation can result in a substantial reduction in the salt burden of drainage water (Rhoades et al. 1973b). However, results depend on the composition of the irrigation water, the type of soil and many other factors. Additional research is needed to determine the conditions in which minimized leaching can be practiced successfully without harmful effects on crop yields.

Modelling transport of water and chemicals

Because of the above mentioned interactions between the soil and chemicals and the frequently slow movement of water in the unsaturated zone, it may take many years before surface applied chemicals reach the groundwater table. In order to predict the effects of surface applied chemicals on water quality and the impact of management practices a large number of computer programs have been developed in recent years. An example is ACTMO, an agricultural chemical transport model which links together a hydrology model, an erosion model and a chemical model. It is used to predict the concentration and amount of pesticides and nutrients on a storm-by-storm basis for a farm-sized watershed (Frere, 1976). Other models have been developed for N and P loads in water (Hagin and Amberger, 1974), and for predicting the quality of irrigation

return flow (Shaffer and Ribbens, 1977). Although many models are presently available to predict long term trends in water quality, few of these models have been adequately tested under field conditions. Future research in water quality modelling should be directed toward field evaluation of these models. In addition, the description of the spatially varied aspects of soil, crop and aquifer systems needs further study.

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