Sources And Fates of Material Influencing Water Quality in the Agricultural Midwest

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Sources and Fates of Material Influencing Water Quality in the Agricultural Midwest

MANAGEMENT PRACTICES TO REDUCE FARM CHEMICAL LOSSES WITH AGRICULTURAL DRAINAGE

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ABSTRACT

Field losses of nutrients and pesticides with surface runoff represent concerns for the quality of receiving water with respect to aquatic life and human consumption. Movement of nitrate-nitrogen and possibly pesticides through the soil profile with excess precipitation can affect the quality of ground water through deep percolation or surface water resources through shallow subsurface drainage. The fate of a soil-applied chemical relative to these modes of loss is highly dependent on the chemical's persistence and interaction or absorption to the soil. In this paper chemicals will be classified as to major mode of loss (sediment, surface runoff water, or subsurface drainage) and reasonable management practices will be considered to control losses for each class. Included in the discussion will be conservation tillage; timing and methods of chemical application; chemical formulation, and additive choices; and monitoring to reduce chemical application.

INTRODUCTION

Efficient application of nutrients and pesticides to the soil in row-crop agriculture is important not only in economic and energy conservation terms, but also from the environmental viewpoint. Losses to surface and ground water resources represent a double concern. An understanding of the mechanism and magnitude of chemical losses is needed before methods of reducing losses are considered. The three most important factors affecting the fate of a soil-applied agricultural chemical are its persistence, its soil adsorption, and the hydrology of the soil to which it is applied.

Relative to persistence, transformations in the soil and plant uptake probably account for the major part of a pesticide or nutrient's disappearance or removal. For nutrients, it is estimated that each bushel of corn grain removes 0.8, 0.13, and 0.20 kg/ha of nitrogen, phosphorus, and potassium, respectively; corresponding numbers for a bushel of soybeans are 3.3, 0.34, and 1.0 kg/ha (Fertilizer Inst. 1976). A significant amount of inorganic nitrogen and phosphorus taken up by plants is also converted to organic forms left in the field as crop residues. Additional inorganic nitrogen and phosphorus can be tied up or immobilized by microorganisms that decompose crop residue. Of course, organic nitrogen and phosphorus previously formed is continuously mineralized or converted back to inorganic forms, with the overall net effect depending on reaction rates in the soil.

For nitrogen, losses to the atmosphere can result from the transformation of nitrate-nitrogen to nitrogen gas or nitrogen oxides, usually in wet soils under anaerobic conditions. The Fertilizer Institute (1976) estimates 10 to 40 percent of the nitrogen applied may be immobilized and 5 to 35 percent lost by denitrification. For phosphorus, they estimate immobilization to be 50 to 90 percent, but phosphorus immobilization includes formation of nearly insoluble compounds as well as tie-up by microorganisms. Keeney (1983) and Nelson and Logan (1983) have written detailed reviews on the fates, respectively, of the nitrogen and phosphorus in the soil.

For pesticides, the major pathway of dissipation is degradation in the soil, where the original compound is converted to other compounds through chemical or biological reactions. With only a few exceptions (for example, atrazine), currently used pesticides dissipate from the soil to the point that no obvious pesticidal properties remain within a year. Pesticides are often assumed to degrade at a rate proportional to the amount present (first-order reaction), and the term half-life is used to express the time it would take for half of the pesticide to degrade. For example, alachlor with a half-life of 2 weeks would be 75 percent dissipated in 4 weeks, 87.5 percent in 6 weeks, and
so on. Whether a first-order reaction represents pesticide degradation or not, very low levels (ppb) of pesticides could remain in the soil for some time. If so, little is known of the fate or environmental impact of these small amounts.

The degradation products of a pesticide are generally considered to be less toxic than the parent pesticide with some notable exceptions, such as aldrin (which is converted in the environment to dieldrin; both of these chlorinated insecticides are now banned). However, considering the large number of degradation products possible for each different pesticide, not enough is known about the fate and toxicity of all these possible compounds. In the past, emphasis has been placed in studying the parent pesticide. Plant uptake of pesticides is usually small compared to other dissipation processes. In a study on the fate of soil-applied dieldrin, Caro and Taylor (1971) estimated that uptake by corn plants was about 5 percent of that applied.

When a chemical is applied to the soil, which is a mixture of organic and inorganic materials along with water and air, the chemical attempts to distribute itself among these compartments. Depending somewhat on soil properties, but to a much greater degree on its own properties, a chemical will interact with or be adsorbed by the soil. One measure of the degree of adsorption of a chemical is the ratio of its concentration in soil to that in water when the chemical, soil, and water are mixed together and are at equilibrium.

For the neutral organic pesticides adsorbed by soil organic matter, this ratio, often called K, usually is in the range of 1 to 1000. For example, for a 2-percent organic matter soil, atrazine would have a value for K of about 10, which would result in runoff water from surface soil with 3 ppm, initially having about 0.3 mg/L of dissolved atrazine (or, K = \(10 = \frac{3}{0.3} \text{ mg/L}\)). Of course, concentrations in the surface soil and runoff water would decrease as atrazine was removed from the surface soil by both runoff water and water infiltrating into the soil.

Because of the fixed, negatively charged sites on clay surfaces associated with the cation exchange capacity of soils, cations (positive ions) such as ammonium are also adsorbed by soil (the organic portion may also have adsorption sites). For ammonium, K has been measured in the vicinity of 50 (Baker and Laffen, 1983a). For this high a value of K and for a soil without surface nitrogen fertilization, ammonium can actually be extracted from rainwater by the soil. For several storms, Johnson and Baker (1982, 1984) measured about 1 mg/L ammonium-nitrogen in rainwater, with concentrations in runoff from cropland being reduced to 0.1 to 0.2 mg/L.

To illustrate the importance of adsorption as opposed to solubility in determining runoff losses of pesticides, paraquat, a highly soluble but strongly adsorbed herbicide because of its cationic nature, was never observed in runoff water although it existed in the soil and sediment at ppm levels (Johnson and Baker, 1982).

For anions (negative ions) few fixed, positively charged sites exist, compared to negatively charged sites, and thus the soil capacity for adsorption is rather limited. In addition, the soil exhibits an order of preference for anions, with nitrate among the least preferred; therefore a K value of zero can usually be assumed for nitrate. The phosphate also forms slightly soluble compounds with aluminum and iron, and therefore, K values for phosphate (relative to available phosphorus in the soil) can exceed 1000.

Relative to hydrology, one of the key factors determining how much chemical is lost in surface runoff, particularly for the less strongly absorbed chemicals, is the amount of infiltration that occurs at the beginning of a storm before runoff begins. It is generally believed that only a thin layer of surface soil, perhaps 1 cm, interacts with and releases chemicals to rainfall and runoff. If a large amount of infiltration occurs, therefore, before surface runoff begins, not only is the volume of runoff reduced, but the concentrations as well—if infiltration moves some of the chemical below the mixing zone. The overall hydrology of the soil system will then determine how much excess water there is and the route, surface or subsurface, it will take to a water resource. Hydrology will also influence the potential for erosion and sediment loss. Table 1 illustrates the impact that initial infiltration, in conjunction with adsorption, can have on chemical losses in surface runoff.

Depending on the volumes of sediment, runoff water, and subsurface drainage, and on K or the degree of soil adsorption, therefore, the chemicals can be classified into three groups: those lost mainly with sediment (strongly adsorbed), those lost mainly with surface runoff water (moderately adsorbed), and those lost mainly with leaching or subsurface drainage water (weakly or nonadsorbed). Because chemical loss is a product of the volume of carrier (sediment or water) and the chemical concentration in that carrier, a reduction in concentration or carrier will reduce loss. Furthermore, knowledge of which group a chemical belongs to is necessary if loss reduction is to be achieved by reduction in the volume of carrier, be it sediment, surface runoff, or subsurface drainage.

**MANAGEMENT PRACTICES**

Because of present economic restraints, the farmer has a rather limited number of management alternatives he can pick from for reducing sediment and chemical losses. This is particularly true if he is locked into row-crop agriculture and cannot use or make a profit with less chemical-intensive crops such as forages and possibly small grains. Considering only row-crops, conservation tillage, soil incorporation of chemicals, improved timing and methods of chemical application, use of special formulations or additives, and reduction in application rates seem to be highest on the list of possibilities.

Conservation tillage, which leaves some or all of the previous year’s crop residues on the soil surface, has been very effective in controlling erosion and sediment-transported chemical losses. The degree of erosion control is highly correlated with the percentage of the soil surface covered with residue. For example, in one study (Barisas et al. 1978), a no-till system with 99 percent residue coverage reduced erosion by 98 percent relative to a moldboard-plowed area with only 2 percent of the surface residue covered.

Total nitrogen losses associated mainly with sediment were reduced by 81 percent. The nitrogen loss reduction was less than the soil loss reduction because the finer sediments lost with no-till were enriched with nitrogen, and solution nitrogen losses with no-till, although only a small part of the total, were up because of decreased incorporation of surface-applied fertilizer and possible leaching from residue. This is one example of the possibly counterproductive effects of conservation tillage.

Another concern is that the use of herbicides might increase as tillage weed control is reduced. Also, leaching might increase if infiltration is increased with conservation tillage, although water quality benefits would result from the reduced runoff. However in at least one study (Kanwar et al. 1985), the amount of nitrate-nitrogen leaching in a no-till plot was less than that in a moldboard-plowed plot. The fate of herbicides broadcast-sprayed on crop residue and susceptible to washoff and volatilization is another concern (Baker and Laffen, 1983a).

For the chemicals lost mainly with surface runoff water,
soil incorporation to decrease the concentration in the rainfall-runoff mixing zone would be an effective practice. However, tillage used for incorporation is a counterproductive practice if conservation tillage is needed, because tillage destroys soil-protecting surface residue. It has been shown that disk-incorporation of herbicides reduces surface runoff losses by about a factor of three compared to surface application without incorporation (Baker and Laflen, 1979) and that plowing down fertilizer results in runoff losses no greater than if no fertilizer were applied (Timmons et al. 1973). For chemicals lost primarily with subsurface drainage, particularly nitrate, better timing of applications to match crop needs should reduce the potential for leaching. In an irrigation experiment, Timmons and Dylla (1981) found that four nitrogen applications during the corn growing season reduced nitrate leaching losses about 20 percent compared to a single application of the same total nitrogen amount. Using mathematical modeling, Kanwar et al. (1984) also predicted reduced nitrate leaching with delayed nitrogen applications. A reduced application rate should also reduce leaching losses as a Minnesota study showed (Gast et al. 1979) where nitrate-nitrogen leaching losses increased nearly in proportion to the amount of initial infiltration that occurs before runoff begins. Had no infiltration occurred, runoff losses for the Initial Depth of mixing zone = 0.63 em Depth of mixing zone = 2.54 em

### RESEARCH NEEDS

To improve efficiency of uptake and to reduce nitrate leaching losses, a reliable soil test for nitrogen would be very valuable, particularly for humid regions. Such a test, now available, would have to take into account not only the inorganic nitrogen present, but that which would result from mineralization during the growing season. In addition, some accounting for transformation, uptake, and movement as affected by the variables of weather would have to be part of the overall scheme. This would probably require the use of mathematical models.

Logic suggests placing chemicals out of zones of higher water movement to reduce chemical movement. For example, there may be an advantage (currently unquantified) to placement of nitrogen in the ridge of a ridge-tillage system where a larger portion of the water would infiltrate in the valleys between the ridges. Herbicide banding just over the row would not only reduce the amount of chemical applied, but would avoid application to the traffic-compacted interrow areas where surface runoff losses are known to be higher because more runoff occurs sooner. On the other hand, chemical placement beneath a compacted zone may reduce the leaching potential because of less infiltration.

Baker et al. (1983) are working on a point-injector fertilizer applicator that can incorporate liquid fertilizer without incorporating crop residue. It also requires little power to pull and has potential for multiple nitrogen applications to improve nitrogen-use efficiency. Others are working on use of high pressure equipment to “blast” liquid fertilizer into the soil. Work on equipment for application of herbicides beneath crop residue in conservation tillage systems is also ongoing. This would be particularly useful for some of the herbicides on the market that are strongly adsorbed (hence not leached and with runoff losses controlled by erosion control) but require incorporation because of volatilization or photodegradation problems.

Finally, work on new equipment or concepts to control runoff and erosion is still needed. For example, a new tillage tool called the paraplow has some potential to increase infiltration without increasing erosion by fracturing the soil without inverting it and covering residue, and the concept of slot-mulch tillage, used in the west to reduce water movement to reduce chemical movement. For example, there may be an advantage (currently unquantified) to placement of nitrogen in the ridge of ridge-tillage system where a larger portion of the water would infiltrate in the valleys between the ridges. Herbicide banding just over the row would not only reduce the amount of chemical applied, but would avoid application to the traffic-compacted interrow areas where surface runoff losses are known to be higher because more runoff occurs sooner. On the other hand, chemical placement beneath a compacted zone may reduce the leaching potential because of less infiltration.

### Table 1.—Percent losses in runoff water as affected by depth of mixing zone, adsorption, and initial infiltration.

<table>
<thead>
<tr>
<th>Initial infiltration cm</th>
<th>Depth of mixing zone = 0.63 cm</th>
<th>K = 0</th>
<th>K = 2</th>
<th>K = 50</th>
<th>Depth of mixing zone = 2.54 cm</th>
<th>K = 0</th>
<th>K = 2</th>
<th>K = 10</th>
<th>K = 50</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.0</td>
<td>50.0</td>
<td>45.4</td>
<td>21.7</td>
<td>5.6</td>
<td>47.5</td>
<td>22.5</td>
<td>6.7</td>
<td>1.5</td>
<td></td>
</tr>
<tr>
<td>0.69</td>
<td>6.8</td>
<td>29.0</td>
<td>17.2</td>
<td>4.6</td>
<td>27.8</td>
<td>17.8</td>
<td>5.5</td>
<td>1.2</td>
<td></td>
</tr>
<tr>
<td>1.27</td>
<td>0.9</td>
<td>17.9</td>
<td>13.1</td>
<td>3.7</td>
<td>15.9</td>
<td>13.5</td>
<td>4.3</td>
<td>1.0</td>
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<tr>
<td>1.90</td>
<td>0.1</td>
<td>10.5</td>
<td>9.3</td>
<td>2.7</td>
<td>8.7</td>
<td>9.6</td>
<td>3.2</td>
<td>0.7</td>
<td></td>
</tr>
</tbody>
</table>

1From mixing zone assuming a 1-h rainfall at 3.8 cm/h and using a simple mixing model.
2Infiltration that occurs before runoff begins; after the initial infiltration period (0, 10, 20, or 30 min), it was assumed runoff was half of rainfall.
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THE FATE OF MATERIALS EXPORTED BY THE BIG BLUE AND THE BLACK VERMILLION RIVERS INTO TUTTLE CREEK RESERVOIR, KANSAS

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ABSTRACT

Suspended sediments and dissolved nutrients in midwestern reservoirs originate in agricultural watersheds. Rivers carry the sediments and nutrients to reservoirs formed by impounded rivers. Highest loadings occur during storm flows, thus the hydrologic regime represents a significant control variable. Most of the precipitation in the Great Plains occurs in late spring and early summer, usually in the form of thunderstorms, but little similarity exists from year to year. Once in the reservoir, wind generated currents keep materials in suspension, thus establishing turbidity gradients along the long axis of the reservoir. These gradients of suspended materials control many biological processes, e.g., photosynthesis, decomposition, particulate feeding by zooplankton and predation by zooplanktivorous fishes. These biological processes influence water quality; thus, while the original quality of the water entering the reservoir is directly related to watershed conditions, watershed properties more subtly and indirectly influence water quality.

INTRODUCTION

The number of reservoirs in North America has increased most in the last three decades. In regions where no natural lakes existed a short time ago, there are now significant numbers of large bodies of standing water. Reservoirs and lakes differ significantly in their morphometric and hydrologic characteristics (Neel, 1963; Baxter, 1977; Thornton, 1984; Marzolf, 1984), so that biological processes and their controlling variables also differ, being more dependent upon the river and watershed events.

Since larger reservoirs are formed by impounding major rivers, a reservoir is, in some ways, a lacustrine feature of its river; therefore, a river flowing into a reservoir has an important influence on biological processes in the reservoir. Thornton (1981) and Marzolf (1984) established the significance of river inflow on water quality parameters and reservoir productivity.

Silts and clays often dominate the particulate materials in the inflow to reservoirs in agricultural drainages. This report relates a study of the effects of a pulsed inflow resulting from storms in the summer of 1984 to the topic of nonpoint source pollution. The report compares the storm flow effects to water quality characteristics and biological processes during stable inflows.

COMPARISON BETWEEN RESERVOIRS AND LAKES

Lakes historically were the main focus of limnological investigations. Our conceptions of reservoir characteristics have therefore been biased towards those of lakes. The following comparison of lakes and reservoirs addresses that misconception.

Watershed Area. Lakes and reservoirs differ in watershed area. Lakes rarely have a ratio of watershed area to water surface area exceeding 10; the ratio for reservoirs in the Kansas River drainage basin averages 507 (Marzolf, 1984). These large ratios reflect the potential for greater nutrient and sediment loading to reservoirs. The watershed is the geomorphic unit drained by the river; the sizes of rivers are proportional to the sizes of watersheds. Consequently, the large ratio for reservoirs results from damming large rivers.

Tributaries. Reservoirs receive water primarily from the rivers impounded to form them. Lakes typically receive water from several small inflowing streams. Thus, a single dominant inflow point provides the materials in suspension and solution for reservoirs.

Hydraulics. The turnover time, or hydraulic retention time, is greater for lakes than reservoirs. Marzolf (1984) found average retention times for lakes to be 1,622 and for reservoirs, 427 days. Materials entering a reservoir, in most cases, are transported more rapidly through the reservoir than in lakes (Groeger and Kimmel, 1984). Marzolf (1984) referred to reservoirs as "continuous flow processors" and to lakes as "batch processors."

Depth. Reservoirs are shallower than lakes, and are shallower in relation to water surface area exposed to the wind. Many reservoirs in the Great Plains do not stratify because of wind mixing; the shallow depth and long fetch allow complete circulation. Particulate materials entering the reservoir from the impounded river remain suspended.

Shape. Elongate reservoir morphology (maximum length several orders of magnitude greater than width), large watersheds, and river-influences act together to create "lentic rivers" in that the materials in suspension and solution at any point in space are dependent upon those materials entering from points upstream.

TUTTLE CREEK RESERVOIR: A CASE STUDY

Tuttle Creek Reservoir, 8 km north of Manhattan, Kansas, is convenient to the Kansas State University campus, and it exemplifies the reservoir characteristics just described (Fig. 1). The reservoir was formed by damming the Big Blue River. The Black Vermillion River joins the Big Blue
Figure 1.—Map of Tuttle Creek Reservoir showing the sampling sites and associated reservoir morphometric parameters.

TUTTLE CREEK RESERVOIR INFLOW PATTERNS IN 1984

A large inflow occurred in June 1984. Inflows began to increase on June 7 (Fig. 2), peaked on June 16 at 161.8 million m³/day, and returned to normal or base flow conditions (10.6 to 1.4 million m³/day) on June 27. From this latter date to the end of the sampling season (November), inflows remained near base flow levels.

Inflow and outflow rates influence reservoir volume. The Army Corps of Engineers determined outflow rates based upon conditions in higher order rivers in the Kansas River and Missouri River drainages. The Corps of Engineers held water in Tuttle Creek Reservoir and reduced outflow during this storm to baseline levels (2.4 million m³/day, Fig. 2). Consequently, reservoir volume increased immediately after inflow increased (June 8), peaked at 149,672 ha-m on June 26 (1 day before inflow returned to baseline conditions), and gradually decreased to conservation pool levels (42,000 ha-m) by July 23. Outflow was increased when reservoir volume peaked, and returned to baseline flows on July 28. Reservoir volume was above normal for a period of 6 weeks; having increased 365 percent at peak volume. Retention times, by definition, follow volume and outflow patterns (Fig. 2).

Water samples were collected at eight stations along the longitudinal axis of the reservoir (Fig. 2) and analyzed for parameters believed to influence zooplankton nutrition (the investigation underway at the time). Standard sample collection and analytical methods for these parameters were employed; Shuman (in prep.) discusses the details.

The first sampling date, June 8, was the day after the initial increase in inflow (Fig. 2). Two additional sampling dates (June 13 and 22) fell during the period of high inflows and high reservoir volume (outflow was still at baseline levels). The next dates (July 5, July 13, and July 18) fell during high outflows, baseline inflows, and high-volume. The final two sampling dates included in this analysis (July 27 and August 2) were during stable reservoir conditions (baseline inflows, outflows, and volume). Of the eight sampling dates, the first six occurred during various stages of the pulsed inflow and high reservoir volume (the pulse period), and the final two dates occurred during stable hydrologic conditions (the stable period).

Figure 2.—Reservoir volume, inflow, outflow, and retention time during the pulse period and subsequent stable period in the summer of 1984. Arrows indicate sampling dates.

RESERVOIR RESPONSES TO THE STORM INFLOW AND SUBSEQUENT BASE INFLOWS

Suspected Particulate Matter

Concentrations were extremely high at the river input end of the reservoir (Fig. 3) on the first and second sampling dates. Concentrations of suspended solids exceeded 1,300 mg/L. These high loads moved down the reservoir as input continued to be high, decreasing upstream and increasing downstream concentrations. From July 5 to July 18, when outflow was high, the pulsed load continued to move down the reservoir as the retention time averaged about 19 days at peak outflow.

During the stable period, July 27 and August 2, gradients in suspended particulate matter were stabilized. These patterns typify data sets previously collected from Tuttle Creek Reservoir during stable reservoir periods (Dufford, 1970; Taylor, 1971; Osborne, 1972).

PATTERNS IN TUTTLE CREEK RESERVOIR INFLOW
Dissolved Solids

Patterns of dissolved solids (Fig. 3) are less obvious during the pulse period. When inflow was high, dissolved solids were low, particularly at the upstream end of the reservoir (mean concentration of 120.8 mg/L). They remained low throughout the pulse period, gradually increasing to levels between 225 and 421 mg/L during the stable period (mean concentration of 292 mg/L). Gradients of dissolved solids were evident during the stable period, with highest concentrations at the upstream end of the reservoir. Low concentrations during storm inflows may be caused by dilution; surface runoff is low in dissolved solids. Ground water sources of flow during stable periods are normally high in dissolved solids in this drainage; limestone formations contribute calcium carbonate during infiltration.

Alkalinity and pH

Patterns of alkalinity (Fig. 4) follow the trend for dissolved solids. Since the pH is always below 8.0, the inorganic carbon is primarily in the form of bicarbonate ions. Concentrations were low during high inflows (75.5 mg/L as CaCO₃ at the upstream end), remained low during the outflow period, and returned to normal levels (120–240 mg/L as CaCO₃) during the stable period. Gradients in alkalinity were present during this stable period, with highest alkalinitities occurring at the river input end of the reservoir.

pH follows a similar pattern (Fig. 4), with levels near or below neutrality during the pulse, and higher during the stable period (7.0 to 7.4). Alkalinity and pH are lower in surface runoff from agricultural watersheds during storm inflows, but higher in ground water sources during baseline inflows. Limestone, common in this region, accounts for the buffering capacity of ground water sources.

Dissolved Nitrogen and Phosphorus

Nitrogen and phosphorus are important plant nutrients, with complicated biogeochemical cycles in nature. This report describes temporal and spatial patterns for a short time during a single year. Nitrogen and phosphorus cycles in rivers may be different than in standing water; readers should therefore interpret the following with caution. For example, during stable base flow, nutrient kinetics may be heavily influenced by biological uptake and release. Storm flow nitrogen and phosphorus concentrations, on the other hand, may be most heavily influenced by hydrologic and chemical exchanges.

Nitrogen. Spatial patterns of dissolved ammonia (Fig. 5) were distinct during stable periods, but varied during the pulse period. When outflow rates were high, ammonia was low throughout the reservoir. Although ammonia concentrations were highest at the river input end at the onset of the storm inflow (June 8), they did not remain high with successive inflows. Since ammonia is rapidly assimilated by biota or oxidized to nitrate, high concentrations were expected only at the river input end of the reservoir. Except for the mid reservoir peak on June 13, this pattern characterized both pulse and stable periods.

Figure 3.—Suspended particulate matter and dissolved solids concentrations measured along the long axis of the reservoir during the pulse period (June 8 to July 18) and subsequent stable period (July 28 to August 2) in the summer of 1984.
Figure 4.—Alkalinity and pH measured along the long axis of the reservoir during the pulse and stable period.

Figure 5.—Dissolved ammonia, dissolved phosphate (SRP), and dissolved nitrate concentrations along the long axis of the reservoir during the pulse and stable period.
No obvious patterns in dissolved nitrate concentrations (Fig. 5) occurred during pulse or stable inflow periods. Concentrations were highest during the storm inflow, but since nitrate concentration was uniformly high (2-3 mg/L), changes through time represented only small percentages of the total. Thus, changes, while they may be large in an absolute sense, are not always detectable.

**Phosphorus.** During pulsed inflows, dissolved phosphate (Fig. 5) was highest at the river input end of the reservoir (mean concentration of 181.8 µg/L). Concentrations were higher during the pulse period than during the stable period. Concentrations during the stable period (July 27 to August 2) were lower at the river input end; measurements during August to November were similarly low. Mean dissolved phosphate concentrations on these two dates were 70.6 µg/L.

**Dissolved and Particulate Organic Carbon**

Dissolved organic carbon (DOC) concentrations (Fig. 6) at the river input end of the reservoir during storm inflows were similar to other sites in the reservoir (mean concentration of 6.8 mg/L). Similar results were found for the storm inflows in 1983. Adsorption of the organics to the more concentrated clay particles during floods possibly decreases the DOC concentrations in the incoming water. During stable hydrologic periods, with lower concentrations of inorganic particulates, higher DOC concentrations were observed at the river input end of the reservoir. This pattern persisted through November. The mean DOC concentration (averaged over all sites) on these two dates was 6.8 mg/L.

Particulate organic carbon (POC) was measured on five of the eight sampling dates (Fig. 6). Levels of POC during the periods of high inflow (June 8 and 22) were notably higher than levels during base inflow (July 13–August 2). The peak POC concentration detected during storm inflows was 42 mg/L at the river input end of the reservoir at the onset of the pulse. During the stable period, a gradient in POC is established, with highest concentrations at the river input end, suggesting adsorption of dissolved organics onto clay and silt particles entering at the river input end of the reservoir during pulse and stable periods.

Although the potential for high levels of DOC entering with pulsed inflows seems likely, adsorption onto the abundant particulates would remove these from solution and increase the levels of POC. Further, the high suspended sedimentation loads at this time are dominated by silts and clays, normally inorganic particles. Without adsorption of organics unto these particles, POC concentrations should be quite low at this time. Since the reverse is true, and the ratio of DOC to POC is less than 1, adsorption seems a significant process during inflows.

**Chlorophyll**

Chlorophyll levels (an indication of algal abundance) in Tuttle Creek Reservoir (Fig. 6) are normally low (<10 µg/L). Occasionally, concentrations (such as at the river input end on August 2) compare to reservoirs and lakes with low abiotic turbidity. However, concentrations below 2 µg/L are common, with the highest concentrations occurring at the river input end. Since light is most limiting there, the algae present are probably allochthonous in origin (Taylor, 1976; Swanson and Bachmann, 1976). Osborne (1972) showed primary productivity was lowest at the river input end, although chlorophyll was highest there. Since chlorophyll levels are low throughout the reservoir (often below detection), algae are unlikely to be a significant nutritive source for secondary producers in Tuttle Creek Reservoir.

Figure 6.—Particulate and dissolved organic carbon concentrations along the long axis of the reservoir during the pulse and stable period.
PERSPECTIVES ON NONPOINT SOURCE POLLUTION

WATERSHEDS: IMPLICATIONS FOR MANAGEMENT

Impoundment of rivers creates bodies of standing water in areas with few or no lakes. These reservoirs are different in morphometry and hydrologic patterns from lakes and from the river that was impounded. Silts and clays dominate the input of materials from rivers draining agricultural watersheds. Concentrations are highest during storm flows. Wind-generated currents maintain these materials in suspension, creating gradients along the long axis of the reservoir.

Since the input of these materials to reservoirs is a single point in space, agricultural runoff to reservoirs is actually a point source, that is, the impounded river. Runoff is only a nonpoint source in terms of the river itself. The magnitude of silt and clay loading depends on watershed size, land use practices, and hydrologic patterns.

Since reservoir water quality and biological productivity are strongly influenced by the inflow of these materials and establishment of gradients, the reservoir is functionally linked to its watershed. For reservoir management objectives, this implies that hydrologic patterns and land use in watersheds must be understood to adequately predict water quality and biological productivity. Developers of reservoir management strategies should recognize these linkages.

The categorization of agricultural runoff as a nonpoint source of pollution to reservoirs is ambiguous. Further, classifying runoff as a reservoir pollutant connotes detrimental effects. Whether biological and water quality responses to these inflows are negative depends on their importance in management strategies and goals.

REFERENCES


THE INTERACTION OF BIOLOGICAL AND HYDROLOGICAL PHENOMENA THAT MEDIATE THE QUALITIES OF WATER DRAINING NATIVE TALLGRASS PRAIRIE ON THE KONZA PRAIRIE RESEARCH NATURAL AREA

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ABSTRACT
The quality of water from native landscapes is the baseline against which the impact of pollutants on surface water resources must be evaluated. The King’s Creek watershed has been a U.S. Geological Survey benchmark watershed since 1979. It represents the landscape type that gave rise to much of the agricultural midwestern United States. Hydrologic, chemical, and biological measurements have been made by ecologists performing long-term ecological research (LTER) in tallgrass prairie. Streamwater chemistry varies seasonally with the amount and movement of ground water and soil water, with changes in prairie vegetation, and in response to changes in surface water discharge. Concentrations of organic carbon, organic and inorganic nitrogen, and phosphorus increase during storm flows. The particulate fractions in transport and in storage in the stream bed and on the flood plain vary seasonally with recent hydrologic history and changes in vegetation.

INTRODUCTION
The term water quality implies some standard against which waters can be compared. Since a continuum of physico-chemical conditions exists in nature, so does a similar range of standards. Water quality is a function of the natural watershed, and the condition of the watershed is a function of the geologic history of the basin. Therefore, each ecological region must have a characteristic set of standards. To establish these standards, measurements must be made on pristine systems within an ecological region. Baseline measurements taken over long time frames would provide an estimate of the true range of chemical and physical parameters for these ecological regions.

Recent research on stream ecosystem properties has shown the linkage between terrestrial and aquatic ecosystems (Likens and Bormann, 1974; Hynes 1975). In any lotic system there exists a continuum (Vannote et al. 1980) of physical, biological, and chemical conditions. These conditions represent the most probable states of stream reaches dependent on hydrologic and geologic constraints.

The import, transport and storage of organic and inorganic material in a given stream reach depends on both biotic and abiotic factors. Included in the biotic component is the nature of the microbial assemblages. The microbially form the tightest link between terrestrial and aquatic ecosystems (McArthur et al. 1985). Organisms that feed directly on microorganisms or upon organic material imported into the stream further alter the quality of the water through feeding (Wallace et al. 1982). Seasonal changes in the terrestrial ecosystem would further alter the amounts and qualities of material imported into a stream reach. Primary production both in the stream and within the watershed would alter the nutrient loading to the system.

While biological processes play an important role in the chemical quality of the water, physical (e.g., hydrologic) events control directly the import, transport and amount of storage. Extreme hydrologic events may act as reset mechanisms in flowing waters (Gurtz et al. 1982).

Tallgrass prairies once covered greater than 6 percent of the coterminous United States. This ecological system was exceeded in total area only by eastern deciduous forests. The landscape once represented by tallgrass prairie has been converted into much of the agricultural landscape of the midwestern United States. The Konza Prairie Research Natural Area (KPRNA) is the largest representative tract of tallgrass prairie. Less than 2 percent of the Konza Prairie has ever been plowed, and the KPRNA is managed to provide a range of conditions encompassing those of tallgrass prairie prior to settlement.

The 1,060 ha Kings Creek watershed is located entirely within the Konza boundaries. This watershed has been a U.S. Geological Survey benchmark watershed since 1979.

We present here preliminary data resulting from ecological research in tallgrass prairie. Streamwater chemistry has been monitored in the Kings Creek watershed for NO₃-N, dissolved and particulate organic matter, specific conductivity and phosphorus. These parameters have been measured over seasons and during storm events. Our results will provide useful values to help determine a meaningful measure of water quality in the midwestern United States.

METHODS
Baseflow water samples were collected weekly from the Kings Creek drainage system in 500 ml bottles. Storm events were sampled using automatic water samplers (ISCO Model 2100). The samplers collected every 30 minutes after being actuated by increased stage height. Water samples were analyzed for nitrate-N and soluble reac-
tive phosphorus on a Technicon AutoAnalyzer II and specific conductivity was measured. Total persulfate nitrogen (TPN) was determined by the nitrate method after an alkaline persulfate digestion using a modified method of D'Elia et al. (1977). Dissolved organic carbon analyses were done on a Beckman 915-B Total Carbon Analyzer after the samples had been filtered (Gelman Type A/E), acidified and sparged to remove inorganic carbon. Particulate organic carbon was separated into size fractions gravimetrically and combusted at 450°C for 4 hours. The results are expressed as ash free dry mass (AFDM). Discharge measurements were made from data collected by the U.S. Geological Survey and see Gurtz et al. (1982). Diurnal samples were collected by grab samples on hourly intervals in June 1983. The samples were analyzed as described above. Water collected during storm events was used as inoculum (10 μl) and spread on a minimal salts media with glucose added as the sole carbon source.

RESULTS AND DISCUSSION.

Hydrological measurements of water years 1980–84 on Kings Creek are shown in Figure 1. The annual hydrograph varies considerably from year to year in terms of duration of the flow period and the magnitude, and timing of major storm flows. This temporally variable hydrological regime imposes limits on biotic processes while influencing physical channel processes that further mediate biological phenomena. Life cycles, food requirements, and behavior of aquatic species reflect adaptations to the relatively unpredictable harsh extremes of drought and flood in prairie streams. Organic matter storage provides both structure and a potential food resource to aquatic biota and is strongly influenced by recent hydrologic history. Floods can cause high rates of export as well as movement of large organic matter from the channel to the adjacent banks of the floodplain (Gurtz et al. 1982).

The import, transport, deposition and entrainment of nutrients in stream ecosystems is strongly influenced by the hydrology. The frequency and/or intensity of hydrologic events are important in determining whether material is retained or transported through the system. Dynamics of dissolved organic carbon concentration differed during two storm events in 1981 (Fig. 2).

The early storm (Fig. 2a) shows dissolved organic carbon levels tracking changes in discharge. This storm (May 1981) occurred early in the water year. The pattern of transport changed after repeated storms (July 1981; Fig. 2b). Dissolved organic carbon concentrations peaked prior to the peak in discharge and subsequently decreased. This may be due to at least one of two factors: (1) dilution of the dissolved organic carbon by increased rain or (2) an actual decrease in available soluble carbon as a result of previous leaching.

Bacterial population densities in the water column follow patterns similar to the dissolved organic carbon (Fig. 2). McArthur (1984) has shown experimentally that bacteria respond immediately to inputs of dissolved organic carbon but these studies were not done during storm events. The increases in DOC and bacteria during storms may be from the same sources (floodplain). Dissolved organic carbon leached from the floodplain and bacteria associated with forest floor litter may be washed into the stream during storm events.

Particulate organic carbon enters the stream from direct litter inputs from riparian vegetation as well as lateral movement of matter at the soil surface. Smith (1982) showed that most of the decomposition of coarse particulate organic matter (CPOM) was due to microbial and physical processing. Macroinvertebrate shredders (Meritt and Cummins, 1978), organisms that consume CPOM, were an insignificant component in Kings Creek. In some streams shredding organisms can contribute significant amounts of fine particulate organic matter (FPOM), through their feeding activity and fecal material. Most of the particulate matter in transport in Kings Creek is smaller than 53 μm; since invertebrate activity is mini-

![Figure 1](image-url)
m al and input is predominantly CPOM in the lower reaches we hypothesize that this FPOM is originating in the prairie or from bank erosion.

The senescent growth of prairie grasses remains as standing dead material which is physically broken up by wind and precipitation and subsequently transported into the stream channel. Furthermore, when water is present in the grassland reaches of the stream and nutrient availability is high, extensive algal mats are formed. The sloughing of senescent algae further contributes to organic matter transport.

Nitrogen and phosphorus concentrations in surface waters draining the Midwestern States are among the highest in the country ranging from 0.3 to 5 mg N/L for nitrogen and 0.05 to 0.20 mg P/L for phosphorus and are related to the amount of watershed used for agriculture (Omernik, 1977). Despite this, nitrogen and phosphorus concentrations draining Kings Creek, a tallgrass prairie stream, are low enough to limit algal growth during the summer months (Tate, 1985).

Phosphorus concentrations in stream water are generally less than 0.01 mg P/L in Kings Creek (Tate, 1985) during base flow. Phosphate concentrations increase up to approximately 200 to 300 mg P/L during storm events (Fig. 3). The major export of phosphorus occurs during storm flows.

Inorganic nitrogen (NO₂⁻N) concentrations in Kings Creek fluctuate over seasons (Tate, 1985) and diurnal and storm events. Figure 4 depicts changes in nitrate concentrations and specific conductivity over time during one storm event (May 1983). Specific conductivity decreases in response to increased discharge and then increases as the stream returns to baseflow conditions. Nitrate patterns are the reciprocal of the specific conductivity measurements: there is an initial increase and then subsequent decrease. The major export of nitrogen occurs during storm events.

Less nitrogen is exported during the summer when terrestrial plants are actively growing. Significant amounts of nutrients are available for export in the fall after the growing season or in the spring prior to the growing season. The primary mechanism to maintain these nutrients within a stream reach is uptake by instream producers. Tate (1985) has shown lower nitrate concentrations in late spring and summer, a period corresponding to the growing season of terrestrial vegetation, and higher nitrate concentrations in fall, winter, and early spring corresponding to the dormant season of terrestrial vegetation.

Diurnal variation in nitrate concentrations depends on location within the drainage. Figure 5 shows nitrate concentrations at two sites over a 24-hour sampling period. Lowest nitrate concentrations at a site 2 km from channel origin occurred during the daylight hours, corresponding to the period when photosynthesis occurs. Night-time levels of nitrate at this site were more than double the day-time values. Farther downstream, 4 km from channel origin, in the gallery forest, the diurnal pattern in nitrate was much different. The stream channel at this lower location had a dense canopy inhibiting instream primary production and subsequently altering the transport of nitrogen.

Total persulfate nitrogen is composed of nitrate, nitrite, ammonia, and organic nitrogen. Tate (1985) reported nitrite and ammonia to be a negligible component of total nitrogen in the Kings Creek system. The TPN values in Table 1, therefore, represent the higher concentrations of organic nitrogen found in Kings Creek, a tallgrass prairie stream. Total, persulfate nitrogen concentrations are an

![Figure 2](image)

**Figure 2.** Changes in discharge (L/s), mean bacterial counts (Colonies/10 mL), and dissolved organic carbon (mg/L) during two storm events in Kings Creek. (A = May 1981, B = July 1981) Dash line = discharge. Histogram bars = plate counts. Numbers in parentheses are dissolved organic carbon concentrations.

![Figure 3](image)

**Figure 3.** Changes in nitrate-N concentration (µg/L) and specific conductivity (µmhos/cm) during one storm event in watershed 1D on KPRNA (May 1983).

![Figure 4](image)

**Figure 4.** Changes in nitrate-N concentration (µg/L) and specific conductivity (µmhos/cm) during one storm event in watershed 1D on KPRNA (May 1983).

![Figure 5](image)

**Figure 5.** Changes in nitrate-N concentration (µg/L) and specific conductivity (µmhos/cm) during one storm event in watershed 1D on KPRNA (May 1983).

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**Table 1.** Baseflow mean concentrations (µg/L) of nitrate-N (n = 17) and TPN in four watersheds on the Konza Prairie.

<table>
<thead>
<tr>
<th>Watershed</th>
<th>µg NO₃-N/L</th>
<th>µg TPN/L</th>
</tr>
</thead>
<tbody>
<tr>
<td>1D</td>
<td>1.6</td>
<td>1.58</td>
</tr>
<tr>
<td>2D</td>
<td>3.2</td>
<td>4.05</td>
</tr>
<tr>
<td>N1</td>
<td>24.2</td>
<td>13.14</td>
</tr>
<tr>
<td>N4</td>
<td>3.5</td>
<td>2.87</td>
</tr>
</tbody>
</table>
Figure 4.—Diurnal changes in nitrate–N (µg/L) in two reaches of Kings Creek, KPRNA (June 15-16 and 23-24, 1983).

order of magnitude higher than nitrate–N concentrations (Table 1). Organic nitrogen would enter the stream through the same vectors as organic carbon. Organic nitrogen is the major form of nitrogen exported from Kings Creek. Organic nitrogen was reported to be the major form of nitrogen exported in headwater streams draining the Cascade Mountains (Triska et al., 1984). Preliminary data suggest that concentrations of organic–N and ni-

Figure 5.—Changes in phosphate–P concentration (µg/L) during one storm event in watershed 2D on KPRNA (May 1982).
trate-N in soil water are similar to those in stream water, suggesting the contribution of soil water chemistry to stream water chemistry. In contrast, export from streams draining in the Hubbard Brook system (Likens et al. 1977) have higher levels of inorganic nitrogen export relative to organic nitrogen. The differences between these two ecosystems support the contention that water quality standards must be based on local ecosystem processes.

The dynamics of nutrient import, export and storage in aquatic ecosystems are controlled by many factors, including precipitation and runoff patterns, microbial response, and invertebrate feeding. Additionally, the seasonality of terrestrial plant uptake affects the movement of inorganic and organic nutrients through the soil water.

Processes that occur in the headwaters of a watershed will determine the nature of material imported into lower reaches, reservoirs and lakes. To adequately address whether particular land use is affecting the aquatic ecosystems draining the landscape type, we must know the extremes of the pristine system.

Concerns about the effects of artificial organic matter loading can best be approached after understanding how the system handles natural loading. Nutrient import from agricultural sources may affect in-stream processes but these effects cannot be definitely ascertained without suitable baseline information.

Baseline studies must be sufficiently extensive in spatial and temporal scale to allow water quality perspective to develop for an unmanipulated ecosystem. Only through acquisition of a large database can we develop an understanding of how natural processes influence water quality. Research must focus both on long-term changes which occur gradually over years or decades, as well as short-term phenomena such as storm events, which can have long-lasting consequences on stream water quality.

ACKNOWLEDGEMENTS: This work was supported by Contract E-AC09-76SR00819 between the U.S. Department of Energy and the University of Georgia's Institute of Ecology. We would like to acknowledge B. Brock and D. Loring for field collections of diurnal samples. D. Whittemore provided analyses of phosphorus data.

REFERENCES


IMPLICATIONS OF AIRSHED PROCESSES AND ATMOSPHERIC DEPOSITION OF NONPOINT POLLUTANTS

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ABSTRACT
Lake managers now routinely use estimates of chemical loadings from adjacent land surfaces and from groundwater in evaluating prospective responses to a treatment. However, large nonpoint sources of pollutants through atmospheric transport and deposition also need to be considered. The capacity for watersheds and lakes to assimilate the acidic component of these inputs varies widely on a site-by-site basis. Results are presented from studies of atmospheric chemical inputs and associated lake chemistries in northern Minnesota and Wisconsin. Precipitation sulfate input, lake depth, lake-water renewal time, and bedrock characteristics influence the variability in lake sulfate and alkalinity.

INTRODUCTION
Evaluating the potential response of lakes to management now routinely considers nonpoint chemical loadings from adjacent land and groundwater. However, nonpoint source loadings also result from atmospheric transport and airshed washout through scavenging by rain, fog, and snow. This paper examines the factors influencing the capacity of watersheds and lakes to assimilate these nonpoint inputs and the circumstances under which lake managers must take them into consideration.

To begin, we should consider data on the magnitude of essentially nonpoint source pollutants transported over the eastern United States in the lower atmosphere. Numerous studies (summarized by OTA, 1984) address one consequence of these transported pollutants: the alteration of precipitation chemistry over large areas. As with surface water nonpoint pollutants, the largest sources can be identified at some distances upwind. The 1980 nitrogen oxides and sulfur dioxide emissions of three States illustrate the burden released to the atmosphere:

- Missouri, $0.6 \times 10^8$ tons of nitrogen oxides and $1.3 \times 10^6$ tons sulfur dioxide,
- Illinois, $1.0 \times 10^8$ tons nitrogen oxides and $1.5 \times 10^6$ tons sulfur dioxide,
- Indiana, $0.8 \times 10^8$ tons nitrogen oxides and $2.0 \times 10^6$ tons sulfur dioxide (OTA, 1984).

For Indiana, these sulfur dioxide emissions amount to over 50 tons/square mile/year or 150 lb/acre, statewide. A substantial part of these emissions is deposited in the States of origin. Relatively near the larger sources, the annual wet plus dry deposition of oxidized sulfur and nitrogen compounds (oxides and acids) can exceed 500 lb/acre (McFee et al., 1984), scaling down to deposition of 10 to 30 lb/acre in the northern portions of the Midwest.

In 1978–79, researchers at the University of Wisconsin and at the Environmental Protection Agency in Duluth began a series of studies designed to evaluate these atmospheric chemical inputs in the softwater regions of the northern Midwest (Glass and Loucks, 1980). Plans for sampling up to 1,000 lake sites in the poorly buffered (few natural means to offset incoming acidity) regions of northeastern Minnesota, northern Wisconsin, and northern Michigan were developed. By 1983, slightly more than 1,000 lakes and streams had been sampled. Up to 70 watersheds properties also had been quantified in over 600 watersheds in the three States (Ellers et al., 1983; Rapp et al., in press). The goal of this paper is to review the results obtained concerning concentrations of nonpoint atmospheric chemical inputs across the region, the differences in watershed characteristics that lead to pH and alkalinity changes in lake and stream water quality, and the implications of acidic inputs and watershed sensitivity for lake management programs.

BACKGROUND
Watersheds (defined as land and water systems where inputs, internal processing, and outputs can be quantified, manipulated, and studied) have been a framework for ecosystem research for nearly 20 years (Likens and Bormann, 1972; Loucks, 1975; Schindler et al., 1976; Likens et al., 1977; Loucks and Odum, 1978). Most of this research involved intensive studies at a few specific sites to increase understanding of the flows of materials and the processes that link land and water ecosystems.

The understanding that various watershed processes and the wide variability in the physical properties of watersheds produce a specific land/lake response system is now being extended to a larger number (representative population) of watersheds and lakes within biogeographic regions (Rapp et al., in press). Studies of a population of lakes were not feasible until the general principles and first models of watershed functioning were developed in the 1970s. Now, with data on variations in chemical inputs across these regions, and data on the watershed characteristics and chemistry of a large population of lakes, recommendations can be extended to lake management.

ATMOSPHERIC CHEMICAL INPUTS ACROSS THE NORTHERN GREAT LAKES
Atmospheric deposition of chemicals has been monitored at eight sites across the three-State area since 1979 (Glass and Loucks, in review). Table 1 summarizes the results at the eight sites from December 1980 through November 1982. Deposition of acidic substances in northern Michigan (Douglas Lake) is less than at Hubbard Brook, New Hampshire (Glass and Loucks, in review), but much greater than in the interior of Norway, where moderate effects on lake fisheries are reported (Overrein et al., 1980). The difference in measured acidity in precipitation from the Fernberg, Minnesota, site is more than twice that of the Douglas Lake, Michigan, site. The differences between southwestern Minnesota and south-central Michigan are even larger, but few lakes in those regions are sensitive to acidic inputs.

Comparative analysis of the modern and historical precipitation chemistry across the northern part of this three-
State area (Glass and Loucks, in review) shows that the input loadings and the gradient (amounts of a substance carried in solution) have changed considerably since the first rain-chemistry measurements were made in the area during the 1950's. The changes are primarily increases in total nitrate and sulfate in the precipitation. Table 1 shows large increases in sulfate and nitrate anions (SO$_4^{2-}$ and NO$_3^{-}$) from west to east across the region, with almost a tripling in the deposition of acidity [H$^+$] across the three most northerly sites, Fernberg, Trout Lake, and Douglas Lake.

RESULTS FROM THE MONITORING OF LAKE RESPONSE

The 6-year UM-D/ERL-D studies emphasized monitoring of lake chemistry during the spring and fall turnover period. A moderately large number of lakes was measured as a single sampling in each of several years, but a small number of lakes (12 to 15 each in Minnesota and Wisconsin) were measured during the spring and fall each year. As of the end of 1983, 5 years of data were available in two of the States, and 2 years in Michigan.

The average pH, alkalinity, and conductivity at 12 Wisconsin lakes over 3 years show a slight decline in the first two measures (from 6.09 to 5.87 for pH and from 31.9 to 23.9 μeq/l for alkalinity) and the expected rise in conductivity (from 18.5 to 19.5 μSiemens/cm for conductivity). Most of the sites show this pattern in Wisconsin, but few do in Minnesota, where acidic inputs are lower and background alkalinity are higher. Research is underway now to evaluate more fully the statistical significance of these trends and their implications for lake management, taking into consideration the possible influence of weather cycles and the problem of averaging across different types of lakes.

WATERSHED CHARACTERIZATIONS ACROSS THE THREE STATES

To evaluate the role of watershed properties in mediating and neutralizing atmospheric inputs, quantitative data were obtained for 167 watersheds in Minnesota, 316 in Wisconsin, and 53 in Michigan. The characterization of Minnesota and Michigan watersheds was carried out primarily by a group at University of Minnesota-Duluth (UM-D) (Rapp et al. in press); the Wisconsin watershed data were developed originally by a group in the Wisconsin Department of Resources (Eilers et al. 1983) and expanded during 1984 by the UM-D group; and a combined University of Iowa and UM-D group has developed the watershed characterization for the upper peninsula of Michigan.

Various studies (Loucks and Glass, 1984; Loucks et al. 1984; Schnoor et al. 1984) have incorporated a limited set of the factors and processes considered important in watershed studies: atmospheric inputs of water and chemicals, evaporation, forest canopy alteration of chemical concentrations in rainfall, watershed relief and runoff processes, surface water detention/terrestrial neutralization processes, and various permeability and below-ground hydrologic flow parameters. In addition to these factors, important properties of chemical adsorption and acid neutralization in watersheds are associated with bedrock mineralogy, soil chemistry, and flows among various hydrologic compartments. In a mathematical model for predicting aqueous alkalinity in the receiving systems (lakes) of the study region, Schnoor et al. (1984) proposed using water volume and alkalinity in each compartment, along with flow rates, cumulative base cation export, and a rate constant for acid neutralization by soil or minerals.

When all of these factors are considered as processes within a watershed, the nine major classes of measurable properties shown in Table 2, all well known to lake managers; can be recognized as having the potential to influence the water chemistry of the responding system. Overall, data on 3 to 15 measures in each of these 9 categories were obtained for the watersheds studied in each of the 3 States.

ANALYSIS OF THE WATERSHED/LAKE RELATIONSHIPS

For lake management, we would like to know which lakes could show effects from acidic deposition in these northern Great Lakes States and over what time span (expressed in chemical flushing times of from 0.5 to over 30 years). Ideally, one would like to have data on the responses of a large population of lakes subjected to acidic deposition monitored throughout 3 to 5 decades. Because such data are not available for a wide range of lake types; variation in nature (the differences in watershed acid neutralizing capacity) to quantify the responses of watershed systems to the nonpoint atmospheric inputs can benefit research.

The relationships between transported pollutant deposition, watershed properties, and lake chemistry have been evaluated in two ways: first, through analysis of complete correlation matrices for inputs, watershed factors, and watershed processes identified in the lakes. Table 2.—Watershed factor groups and associated watershed processes recognized as important in mediating atmospheric inputs of pollutants (after Rapp et al. in press).

<table>
<thead>
<tr>
<th>Watershed factor group</th>
<th>Watershed processes</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Atmospheric deposition and meteorology</td>
<td>Input rates</td>
</tr>
<tr>
<td>2. Catchment size</td>
<td>Averaging responses</td>
</tr>
<tr>
<td>3. Topographic relief</td>
<td>Hydrologic flow rates</td>
</tr>
<tr>
<td>4. Hydrologic inputs</td>
<td>Flushing rates</td>
</tr>
<tr>
<td>5. Lake depth and volume</td>
<td>Resuspension</td>
</tr>
<tr>
<td>6. Bedrock type</td>
<td>Weathering</td>
</tr>
<tr>
<td>7. Surficial deposit depth and type</td>
<td>Neutralization</td>
</tr>
<tr>
<td>8. Land cover type</td>
<td>Remineralization</td>
</tr>
<tr>
<td>9. Roads and human development</td>
<td>Sedimentation</td>
</tr>
</tbody>
</table>

Table 1.—Annual ion deposition (in μeq/ha) based on the 2-year period from December 1980 through November 1982, using weekly composite rain samples from eight NADP precipitation sites (after Glass and Loucks, in review).

<table>
<thead>
<tr>
<th>Precip. cm.</th>
<th>H</th>
<th>SO$_4^{2-}$</th>
<th>NO$_3^{-}$</th>
<th>Ca</th>
<th>Mg</th>
<th>NH$_4$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lamberton, MN</td>
<td>62</td>
<td>23</td>
<td>210</td>
<td>136</td>
<td>130</td>
<td>44</td>
</tr>
<tr>
<td>Marcell, MN</td>
<td>78</td>
<td>72</td>
<td>220</td>
<td>128</td>
<td>98</td>
<td>35</td>
</tr>
<tr>
<td>Fernberg, MN</td>
<td>70</td>
<td>85</td>
<td>218</td>
<td>115</td>
<td>74</td>
<td>32</td>
</tr>
<tr>
<td>Spooner, WI</td>
<td>68</td>
<td>99</td>
<td>301</td>
<td>158</td>
<td>101</td>
<td>31</td>
</tr>
<tr>
<td>Trout Lake, WI</td>
<td>83</td>
<td>165</td>
<td>313</td>
<td>165</td>
<td>87</td>
<td>26</td>
</tr>
<tr>
<td>Douglas Lake, MI</td>
<td>72</td>
<td>284</td>
<td>367</td>
<td>195</td>
<td>100</td>
<td>38</td>
</tr>
<tr>
<td>Wellston, MI</td>
<td>88</td>
<td>368</td>
<td>494</td>
<td>279</td>
<td>124</td>
<td>40</td>
</tr>
<tr>
<td>Kellogg, MI</td>
<td>83</td>
<td>383</td>
<td>529</td>
<td>248</td>
<td>116</td>
<td>45</td>
</tr>
</tbody>
</table>
tershed output measures (color, in-lake $\text{SO}_4^{2-}$ and $\text{HCO}_3^{-}$); and second, through multiple regression modeling of the one or two variables in each of the nine classes that mechanistic models suggest would have the greatest influence on resultant lake chemistries.

Using a cluster analysis technique on 275 lake-watershed systems, the Wisconsin study (Eilerts et al. 1983) identified lake hydrologic type (seepage, drained, drainage, spring-lake, etc.) as by far the dominant watershed property controlling the prospective response of lakes to acidic inputs in Wisconsin. The dominance of hydrologic turnover time over other factors may have been influenced by the comparative homogeneity of the northern Wisconsin landscape in the study area; soil and bedrock vary little compared to the high diversity of the Northeastern States. Differences in precipitation inputs across the population of lakes were not considered at that time, but will be evaluated in the future.

The influence of chemicals in precipitation, and the ameliorating influence of other watershed properties on color, sulfate, and alkalinity were evaluated in Minnesota using stepwise multiple regression by Rapp et al. (in press). Here, the widespread occurrence of shallow bedrock was expected to yield quite different relationships from those in Wisconsin. The results in Table 3 show the watershed components accounting for the greatest variability in measured color are the surrounding vegetative characteristics, water renewal time, and evaporative concentration. Sulfate concentration in the lake water, however, is determined primarily by the within-State differences in atmospheric sulfate deposition (derived from the gradient in deposition across the lake cluster in Minnesota, interpolated from three sites in the surrounding area and the central Fernberg site). Bedrock type (scale 6 has some sulfide) and total water renewal time are also important in determining sulfate in the headwater lakes. The adjusted $r^2$ is largest for the headwater lakes (at 0.44) and is about half of that (0.24) when all watersheds of all sizes and characteristics are averaged.

Alkalinity is the most difficult of the water chemistry components to explain; the within-State gradient in atmospheric wet sulfate input is again significant. Water renewal time and maximum depth of the lake also contribute to the variability in alkalinity of these lakes. These results do not provide causal explanation, but they do identify variables likely to be important in managing a population of lakes or streams within a sensitive region.

**WATERSHED CHARACTERISTICS AND THEIR IMPLICATIONS FOR WATER QUALITY**

As is usually the case, some uncertainty remains regarding causality of the results presented, and continuing studies are planned. Alterations of lake and stream ecosystems are known to become expressed through the introduction of elevated quantities of chemicals through atmospheric deposition, and/or through changes in

### Table 3. Factors identified through stepwise multiple regression T-statistics as belonging to the best subset models for headwater lakes, and all lakes (after Rapp et al. in press).

<table>
<thead>
<tr>
<th>Headwater</th>
<th>All</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Best subsets for color</strong></td>
<td></td>
</tr>
<tr>
<td>Sulfate concentration</td>
<td>X</td>
</tr>
<tr>
<td>Annual runoff</td>
<td>X</td>
</tr>
<tr>
<td>Total renewal time</td>
<td>X</td>
</tr>
<tr>
<td>Maximum depth</td>
<td>X</td>
</tr>
<tr>
<td>% peat</td>
<td>X</td>
</tr>
<tr>
<td>% forest</td>
<td>X</td>
</tr>
<tr>
<td>% marsh</td>
<td>X</td>
</tr>
<tr>
<td>Adjusted $r^2$</td>
<td>.48</td>
</tr>
<tr>
<td>N</td>
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</tr>
<tr>
<td></td>
<td>244</td>
</tr>
<tr>
<td><strong>Best subsets for sulfate</strong></td>
<td></td>
</tr>
<tr>
<td>Chlorophyll-a</td>
<td>X</td>
</tr>
<tr>
<td>Wet-deposition sulfate</td>
<td>X</td>
</tr>
<tr>
<td>Total watershed area</td>
<td>X</td>
</tr>
<tr>
<td>Total watershed relief</td>
<td>X</td>
</tr>
<tr>
<td>Maximum depth</td>
<td>X</td>
</tr>
<tr>
<td>% scale-6 bedrock</td>
<td>X</td>
</tr>
<tr>
<td>% coniferous</td>
<td>X</td>
</tr>
<tr>
<td>% cultural development</td>
<td>X</td>
</tr>
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<td>Adjusted $r^2$</td>
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<tr>
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<tr>
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<tr>
<td>Chlorophyll-a</td>
<td>X</td>
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<tr>
<td>Wet-deposition sulfate</td>
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<tr>
<td>Total watershed area</td>
<td>X</td>
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<td>Total watershed relief</td>
<td>X</td>
</tr>
<tr>
<td>Maximum depth</td>
<td>X</td>
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<tr>
<td>Average bedrock scale</td>
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<tr>
<td>Average soil pH</td>
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<tr>
<td>% peat</td>
<td>X</td>
</tr>
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</table>

1 Numerical values for T-statistics deleted because they are intended as regression coefficients; magnitudes do not correspond with significance.
bioavailability and chemical forms induced through changes in pH. The work of Sollins et al. (1980) and the studies by the Electric Power Research Institute (see Chen et. al. 1983, and reports that follow) illustrate our ability to follow acidic substances through the diverse phase changes of a watershed or stream to the lake. Airborne acidic-forming compounds deposited with precipitation become part of the hydrocycle and can alter concentrations in the watershed, at least at critical times (Loucks et al. 1984).

To evaluate the potential pH depression of a lake or stream at a critical time—the period during and following snowmelt—the total deposition, through both wet and dry mechanisms in the snow pack, must be summarized. The relationships to terms in the general atmospheric deposition model have been evaluated by Ragland and Wilkening (1982), and Samson and Small (1984). The wet-only measurements at the three National Atmospheric Deposition Program monitoring sites associated with the three State clusters (Table 1) were used to compute the annual wet loadings. A calculated total deposition accumulated over winter in each watershed can be used as the net input to the watershed, stream, and lake from snowmelt.

The complex stoichiometry of lakewater or streamwater for each watershed must be examined carefully. The ionic composition and changes over time should be studied using "Stumm" ion balance diagrams (Schnoor and Stumm, 1984) in conjunction with the relationships given in the previous sections. In each region, a certain proportion of lakes shows a negative response to the snowmelt input, with the largest depressions in the Upper Peninsula of Michigan where SO$_4^{2-}$ deposition is largest (Glass and Loucks, in review).

**DISCUSSION AND CONCLUSIONS**

Evidence from the studies reported here, and from many others, is gradually documenting the lake-watershed properties that determine whether nonpoint pollutants deposited by atmospheric processes are likely to result in a pH or alkalinity depression in lakes or streams. When combined with computations on hydrology and lakewater renewal times, information also is being developed to determine whether the responses to SO$_4^{2-}$ inputs will involve a serious lag time before effects are expressed. The magnitude of SO$_4^{2-}$ deposition in the northern Great Lakes States is emerging as the most important variable determining lake sulfate and alkalinity, and aspects of watershed hydrology, lakewater renewal times, and surrounding vegetation types are important secondary considerations.

How can lake managers apply these results? Most probably, in situ management for lake or stream acidification effects will be extended soon from direct application of lime in the lake (which does not control aluminum mobilized in the watershed), to lime application on the adjacent watershed so as to control the transport of toxic metals. Over a longer period, we are likely to have risk assessment methods for those systems where the expression of acidification has a long latency caused by long water renewal times. Gradual reduction in loadings combined with lime treatments may be needed for these systems.

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**REFERENCES**


