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Estimating Lake Susceptibility to Acidification Due to Acid Deposition

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National Forest System lands contain a wide variety of surface waters ranging from small, sensitive, remote wilderness lakes to large, heavily used recreational reservoirs. The USDA Forest Service is responsible for the management and protection of these diverse water resources. Over the years, much time and effort has been spent on determining how the various activities carried out on national forest lands—timber harvest, road construction, mining, grazing, recreation, etc.—affect water resources, and how adverse effects can be reduced, mitigated, or eliminated. Considerations of impacts on water quantity and quality are an integral part of most land-use decisions made by Forest Service land managers.

Acid deposition and the acidification of lakes and streams present the land manager with a different kind of problem: How to best manage water resources and water-related resources that may be affected by fossil fuel combustion, ore smelting, and other emissions-producing activities taking place outside the national forest boundary, perhaps hundreds of miles away. The decision to build or maintain a campground or trail system, or to develop or manage certain kinds of fisheries, may hinge on whether particular lakes or streams are acidified or whether they are likely to become acidified if the rate of acid deposition increases. In the case of Class I wilderness areas, Public Law 95-95 (the Clean Air Act Amendments of 1977) charges Federal land managers with the responsibility to protect "air quality related values" in these areas, values which may include sensitive lakes and streams and other aquatic ecosystems. Under PL 95-95, if emissions from a proposed new emissions source may impact a Class I area, the appropriate federal land manager has an obligation to review the construction permit to determine whether any air quality related values will be adversely affected.

This paper describes a graphical method of using lake chemistry data and rates of sulfur (and nitrogen) deposition to estimate the likelihood of lakes becoming acidified if subjected to increased levels of deposition. Becoming acidified, in this context, means that a lake's pH decreases to the point that aquatic flora and fauna are adversely affected.

The purpose of this paper is to help the land manager answer the following questions:

1. Below what rate of sulfur (and nitrogen) deposition can the land manager have a high degree of confidence that the lake(s) of concern will not become acidified?

2. Above what rate of sulfur (and nitrogen) deposition can the land manager have a high degree of confidence that some sensitive lake(s) will become acidified?

Between these two deposition levels the fate of lakes is more difficult to assess. Additional data may have to be collected or additional studies conducted. Factors such as background sulfate (SO_4) levels, mineral weathering rates, lake and watershed hydrology, SO_4 adsorption capacity of watershed soils, and acid-neutralizing capacity (ANC) generation due to SO_4 and nitrate (NO_3) reduction in wetlands and lake sediments can all influence the degree to which individual lakes are affected by acid deposition.

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The concept for this paper originated at a May 2-5, 1988, "Workshop on Air Pollution Effects on Wilderness" held at the Mary Flagler Cary Arboretum in Milbrook, NY. The workshop was a joint effort of the Institute of Ecosystem Studies, New York Botanical Garden, and the USDA Forest Service. It was attended by 38 participants, including scientists working in the field of air pollution and acid deposition effects, and USDA Forest Service managers with air resource management responsibilities. The purpose of the workshop was to initiate development of a screening procedure to help Forest Service land managers evaluate "PSD" permit applications to determine whether proposed emissions sources will adversely affect Class I areas. The results of that workshop are reported in a USDA Forest Service General Technical Report (Fox *et al.* 1989). This paper describes, in considerably more detail, the methods outlined by Fox *et al.* (1989) for estimating lake susceptibility to acidification.

THE LAKE ACIDIFICATION PROCESS

Lakes vary widely in their sensitivity to acidic deposition. In some lakes, a relatively small acid loading will result in a sharp drop in pH and deleterious effects on the biota. In others, the same loading, or even a much larger acid loading, will be quickly buffered and have little or no effect on aquatic organisms.

The ability of a lake to buffer incoming acids—i.e., its ANC or alkalinity—depends primarily on the amount of bicarbonate (HCO_3) in the water and the rate at which it is supplied from the watershed. Bicarbonate supply depends on watershed geology and hydrology. In the watershed, HCO_3 is produced by the reactions of water and carbon dioxide with carbonate minerals, such as calcite and dolomite, and cation aluminum silicate minerals like feldspars (orthoclase, albite, anorthite) and micas (muscovite, biotite). These weathering reactions release HCO_3 into solution, accompanied by approximately equal amounts (on an equivalent weight or μeq/l basis) of base cations—calcium (Ca), magnesium (Mg), sodium (Na), and potassium (K) (Stumm and Morgan 1981). How much HCO_3 and other mineral-weathering products actually reach a lake depends on the flow paths of water through the watershed. Water that moves rapidly over the land surface or along shallow subsurface pathways usually contains much less dissolved material than water that takes a slow, deep route to a lake. Ground water is typically high in dissolved ions. Lakes with strong ground water inputs usually have high buffering capacities.

In addition to that supplied by the watershed, some alkalinity is generated in lakes by the processes of SO_4 and NO_3 reduction in the bottom sediment. This can be a significant source of ANC in lakes that receive only minimal amounts of HCO_3 from their watersheds (Rudd *et al.* 1986a, 1986b; Kelly *et al.* 1987; Baker and Brezonik 1988; Baker *et al.* 1988).

Because approximately equal amounts of base cations and HCO_3 are released by weathering of carbonate and aluminum silicate minerals, the ratio between ANC and base cations is fairly close to 1:1 in lakes that are unaffected by acid deposition. When a sensitive lake/watershed system is subjected to acidic deposition, incoming acids are neutralized by HCO_3, which is consumed in the process and replaced by SO_4. If, in a particular lake, HCO_3 is consumed faster than it is supplied by mineral weathering in the watershed and by in-lake processes, then ANC eventually falls to zero and the lake becomes acidic. While lake HCO_3 decreases in response to acid loading, the concentrations of the base cations may remain unchanged, or may increase to some extent due to acid-induced increases in rates of mineral weathering and cation leaching in the watershed.

Figure 1 shows how ANC varies with base cations in three groups of sensitive lakes subjected to different amounts of deposition (lake data from EPA's Eastern Lake Survey, Kanciruk *et al.* 1986; and Western Lake Survey, Eilers *et al.* 1987). The high-elevation lakes of the Pacific Northwest receive low amounts of deposition, an average of about 0.75 to 2.0 kg total nonmarine SO_4-S/ha/yr, close to natural background levels; the Midwestern lakes shown in figure 1 are exposed to moderate deposition rates of about 5 to 7 kg/ha/yr; and, the Northeastern lakes shown are located in areas that receive a high rate of deposition, about 11 to 14 kg/ha/yr. The slopes of the ANC/base cation regression lines for the three lake groups are similar. ANC increases by about 0.75 to 0.85 μeq/l for every 1.0 μeq/l increase in base cations, due to the ratio of HCO_3 and cations produced by mineral weathering. But, the point at which the regression lines intercept the x-axis in figure 1—i.e., the average concentration of base cations in lakes with zero ANC—varies with deposition rate.

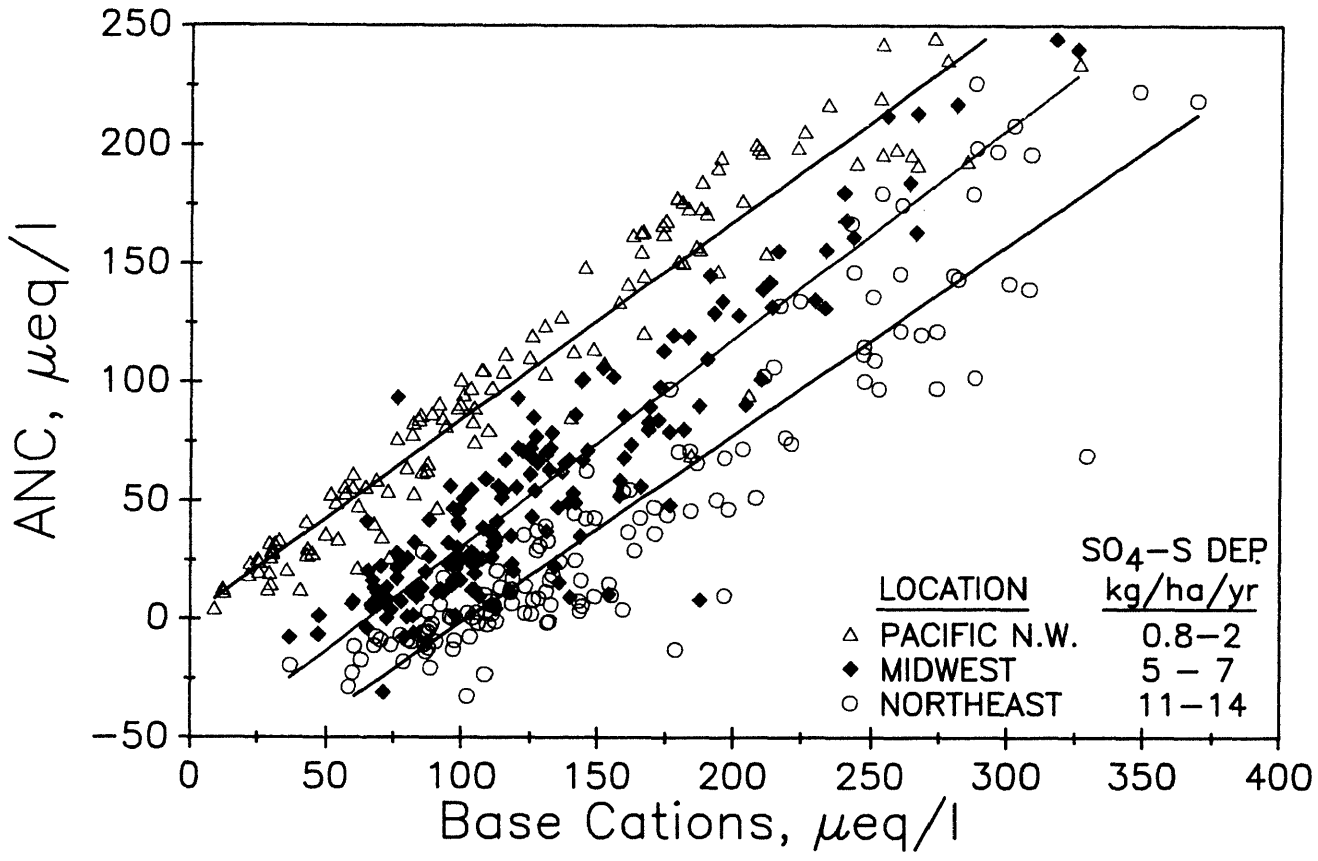

Figure 1.—*Relationship between base cations and ANC in three groups of sensitive lakes subject to different deposition rates. (Lake chemistry data from Kanciruk* et al. *1986 and Eilers* et al. *1987.)*

For the Pacific Northwest lakes, the x-intercept of the ANC/base cation regression line is close to zero. Lakes that have ANC's near zero also have base cation concentrations that are close to zero. The Western Lake Survey found no lakes in the Pacific Northwest with zero ANC. Deposition has not noticeably altered the natural ANC/base cation relationships in these lakes. As deposition increases, more HCO_3 is consumed by acidity (while additional amounts of base cations may be leached from the watersheds), and a deficit develops in lake ANC compared with base cations. For the Midwestern lakes shown in figure 1, the regression line is shifted about 65 μeq/l to the right. Lakes in this group that have base cation concentrations of about 65 μeq/l have ANC's near zero. Also, as figure 1 shows, a number of Midwestern lakes have ANC's of zero or less and are acidic. These are not dark-colored lakes that are naturally acidic because of high concentrations of organic acids from peatlands; only clear or lightly colored lakes (color less than 50 Pt-Co units) are included in figure 1. For the Northeastern lakes shown—lakes subjected to high rates of deposition—the x-intercept of the regression line increases to about 100 μeq/l; and, the incidence of acidic lakes increases.

In pristine lakes, ANC is a good indicator of sensitivity. But ANC changes in response to acid deposition. Because it tends to be more stable, the concentration of base cations has been widely employed as a measure of inherent lake sensitivity and pre-acid deposition ANC. Increases in lake SO_4 and decreases in lake ANC compared with base cations have often been used as indicators of the lake acidification process. The relations among cations, SO_4, HCO_3, and acid loading are the basis of several simple empirical models for predicting whether lake acidification will occur at given acid deposition levels (e.g., Almer *et al.* 1978, Dickson 1980, Henriksen 1980, Wright 1983, Nichols and Verry 1985, Nichols

and McRoberts 1986, Dupont and Grimard 1988). These models are often referred to as "titration models," because they treat lake acidification like a large-scale titration of a bicarbonate solution with a strong acid (sulfuric acid).

These models have some shortcomings. Differences in natural background SO_4 concentrations of lakes and in-lake ANC generation are two sources of variability that are not adequately addressed by titration models. Also, the amount by which base cation concentrations change in response to acid deposition, though probably small in very sensitive lakes, is not easily measured, and can only be roughly estimated for most lakes. Many models ignore this factor. In highly colored lakes, the abundance of organic anions can skew the cation/HCO_3 relationships. In spite of these difficulties, titration models of lake acidification are useful tools. In most cases, they provide a reasonable estimate of lake susceptibility to acidification at various acid-loading levels, without requiring an excessive amount of data.

In this paper, basic titration model theory is applied to lake chemistry data from EPA's Eastern Lake and Western Lake Surveys and some simple nomographs are constructed to estimate lake chemistry response to acid deposition. These nomographs are intended to provide the land manager with guidelines for assessing whether a lake is likely to become acidic and its biota damaged by a particular level of deposition.

EFFECTS OF ACIDITY ON FISH AND OTHER AQUATIC ORGANISMS

Lake acidification appears to have little effect on fish or other aquatic organisms as long as the pH remains at 6.5 or higher; however, noticeable adverse effects begin to occur as lake pH declines to about 6.0. The most acid-sensitive fish species, such as the fathead minnow (*Pimephales promelas*), experience reproductive failure at pH's near 6.0. Some important food organisms such as the opossum shrimp (*Mysis relicta*) disappear at pH's slightly below 6.0. In general, fish eggs and fry are more sensitive to acidity than are adult fish, and fish populations succumb to reproductive failures at pH's above which adult fish can survive. In lakes where average pH is below 5.0, most fish species, almost all molluscks, and many other groups of bottom-dwelling organisms are eliminated. The food web is severely disrupted. Mats of algae frequently cover the bottom (Magnuson *et al.* 1984).

Rahel and Magnuson (1983) studied the relationship between pH and the distribution of fish species in 138 lakes in Wisconsin. They found that several species of minnows (Cyprinidae) and darters (*Etheostoma* spp.) were absent at pH's below 6.0. Black crappie (*Pomoxis nigromaculatus*), muskellunge (*Esox masquinongy*), smallmouth bass (*Micropterus dolomieui*), walleye (*Stizostedion vitreum vitreum*), and northern pike (*Esox lucius*) were absent at pH's below 5.5. A few species, such as the bluegill (*Lepomis macrochirus*) and yellow perch (*Perca flavescens*), survive and successfully reproduce at pH 4.5.

A study by Schindler *et al.* (1985) involving the experimental acidification of a 27-ha lake, Lake 223, in the Experimental Lakes Area in western Ontario documented many of the changes in biota that occur with decreasing pH. Sulfuric acid was added to the lake each spring for 8 years. Table 1 summarizes some of the major findings of that study.

NOMOGRAPH CONSTRUCTION

Lake Sensitivity

Figure 2 shows how lake pH is related to ANC in 2,117 lakes in the Eastern, Midwestern, and Western United States. The data are from Kanciruk *et al.* (1986) and Eilers *et al.* (1987), EPA's Eastern and Western Lake Surveys. Due mostly to the buffering action of HCO_3, lake pH is relatively stable over most of the range of ANC values. At the lower end of the ANC range, however, buffering is negligible and small ANC losses result in large pH declines.

Twenty-five μeq/l is about the lower limit of the range of lake ANC values over which the land manager can be reasonably certain that fish and other aquatic organisms will be unaffected by acid deposition. As figure 2 shows, essentially all clear lakes (color less than 25 Pt-Co units) with ANC's of 25 μeq/l or more have pH's of at least 6.5. These lakes should have a normal, healthy biota. Below 25 μeq/l of ANC, lake pH's drop sharply. Because of inputs of organic acids from peatlands or other wetlands, some moderately colored lakes (color between 25 and 75) with ANC's of from 25 to 50 μeq/l have pH's between 6.5 and 6.0. While marginal, these lakes should still have essentially normal fauna and flora populations. A few highly colored lakes (color greater than 75) in this ANC range have pH's as low as 5.5; some of the most acid-sensitive species would probably be absent from these lakes.

Table 1.–*Biological effects of the acidification of Lake 223*

Year	Lake pH	Effects
1975	6.8	Pre-acidification.
1976	6.8	ANC decline, little pH change, little biological change.
1977	6.13	Slight decline in Chrysophytes (golden-brown algae) and a slight increase in Chlorophytes (green algae).
		Increased emergence of dipterans and rotifers.
1978	5.93	Reproductive failure in fathead minnow (*Pimephales promelas*).
		Severe decline in opossum shrimp (*Mysis relicta*).
		Disappearance of the copepod *Diaptomus sicillis*.
		Increase in Chlorophytes and Cyanophytes (blue-green algae).
		Decline in Chrysophytes and diatoms.
1979	5.64	Fathead minnow nearly extinct.
		Decline in slimy sculpin (*Cottus cognatus*).
		Crayfish (*Orconectes virilis*) exoskeletons soft.
		Decline in other acid-sensitive species of zooplankton and increases in acid-tolerant species.
		Mats of filamentous algae (*Mougeotia* spp.) form in littoral areas, and cover some lake trout spawning areas.
		Continued change in planktonic algae populations as in 1978.
1980	5.59	Reproductive failure in lake trout (*Salvelinus namaycush*).
		Reproductive impairment or failure in crayfish.
		Continued changes in populations of zooplankton and phyto-plankton. Total biomass of plankton has actually increased.
1981	5.02	Reproductive failure in white sucker (*Catostomus commersoni*).
		Lake trout condition declining.
		Crayfish population dwindling.
		Phytoplankton and zooplankton populations continue to change to favor acid-tolerant species.
1982	5.09	No young-of-the-year of any fish species observed.
		Lake trout condition poor due to disruption of food web.
		Plankton populations about the same as previous year.
1983	5.13	Lake trout still numerous, but condition very poor.
		Crayfish, leeches, and the mayfly *Hexagenia*, once abundant, now absent.
		Plankton populations stable—species composition radically different from pre-acidification conditions.

In lakes with ANC's of about -5 μeq/l or less, the land manager can be reasonably certain that populations of aquatic organisms have been affected by lake acidity. Almost all lakes with ANC's of -5 μeq/l or less have pH's between about 5.3 and 4.3 (fig. 2), a pH range in which severe effects on aquatic ecosystems have been well documented.

In figures 3 and 4, total "nonmarine" sulfate sulfur (SO_4-S) deposition loadings are plotted against "Cl-adjusted" or "nonmarine" base cation concentrations for two groups of lakes sampled in the Eastern and Western Lake Surveys. One group of lakes, represented in the figures by open (nonfilled) symbols, consists of lakes that under present deposition conditions have ANC's between 10 and 25 μeq/l. The lakes are further divided into subgroups according to their geographic location and the percent of total precipitation that flows from their watersheds as runoff. These lakes are at or just below the "safe" ANC level for aquatic biota, the level of ANC (25 μeq/l) above which aquatic organisms are unlikely to be adversely affected by acidity.

The other group of lakes shown in figures 3 and 4, represented by filled symbols, consists of lakes that under present deposition conditions have ANC's of -5 to -20 μeq/l. These are lakes that are at or just below the "danger" ANC level, the level of ANC (-5 μeq/l) below which it is likely that aquatic organisms are

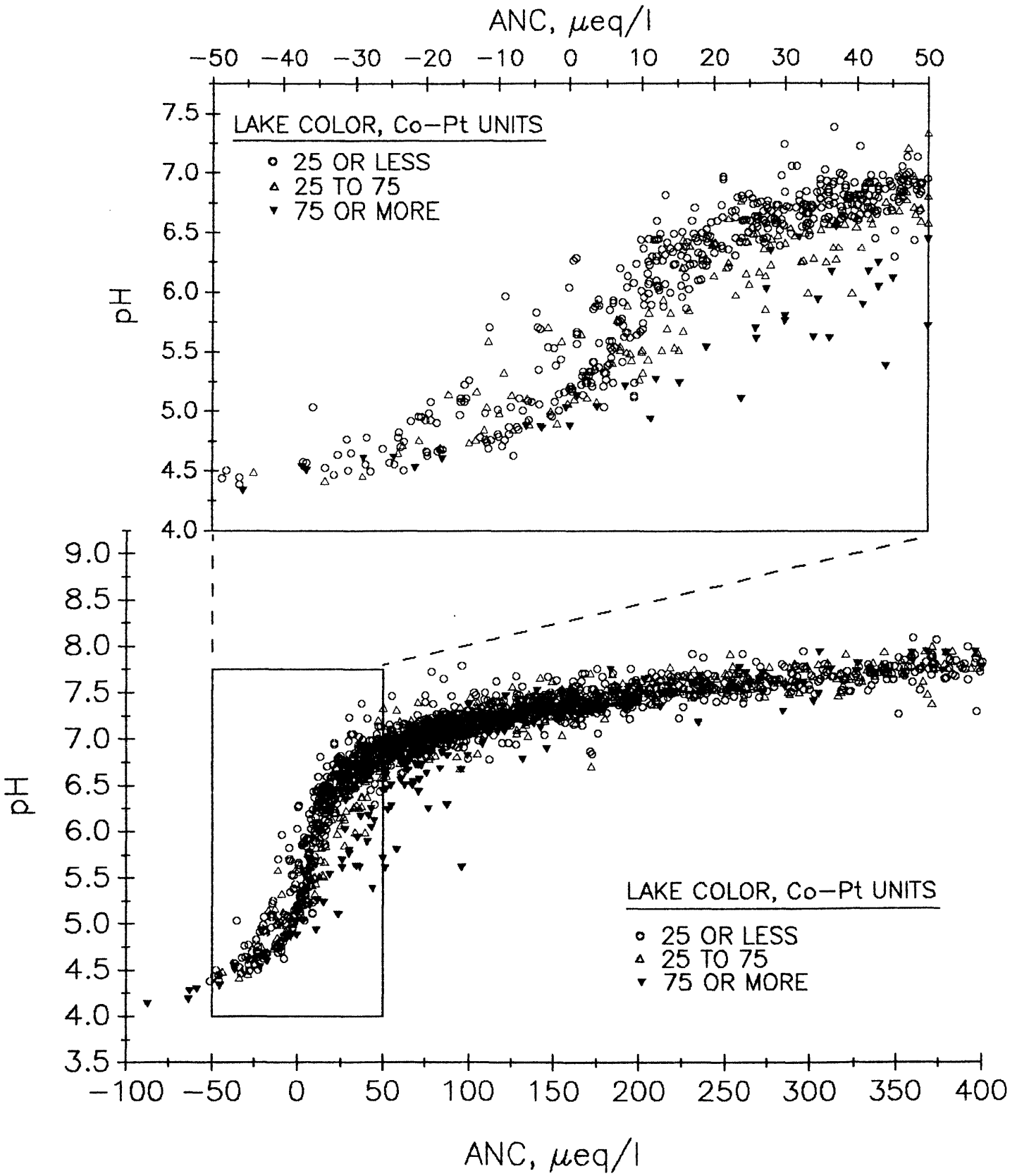

Figure 2.—*ANC and pH of 2,117 lakes in the Eastern and Western United States. (Lake chemistry data from Kanciruk* et al. *1986 and Eilers* et al. *1987.)*

adversely affected. This group contains lakes located only in the Northeastern and Midwestern United States. Except for one apparently naturally acidic lake in Yellowstone National Park, no lakes were found in the Western Lake Survey that had ANC's less than 10 μeq/l. For reasons that are explained later, no data for Florida lakes are included in figures 3 and 4.

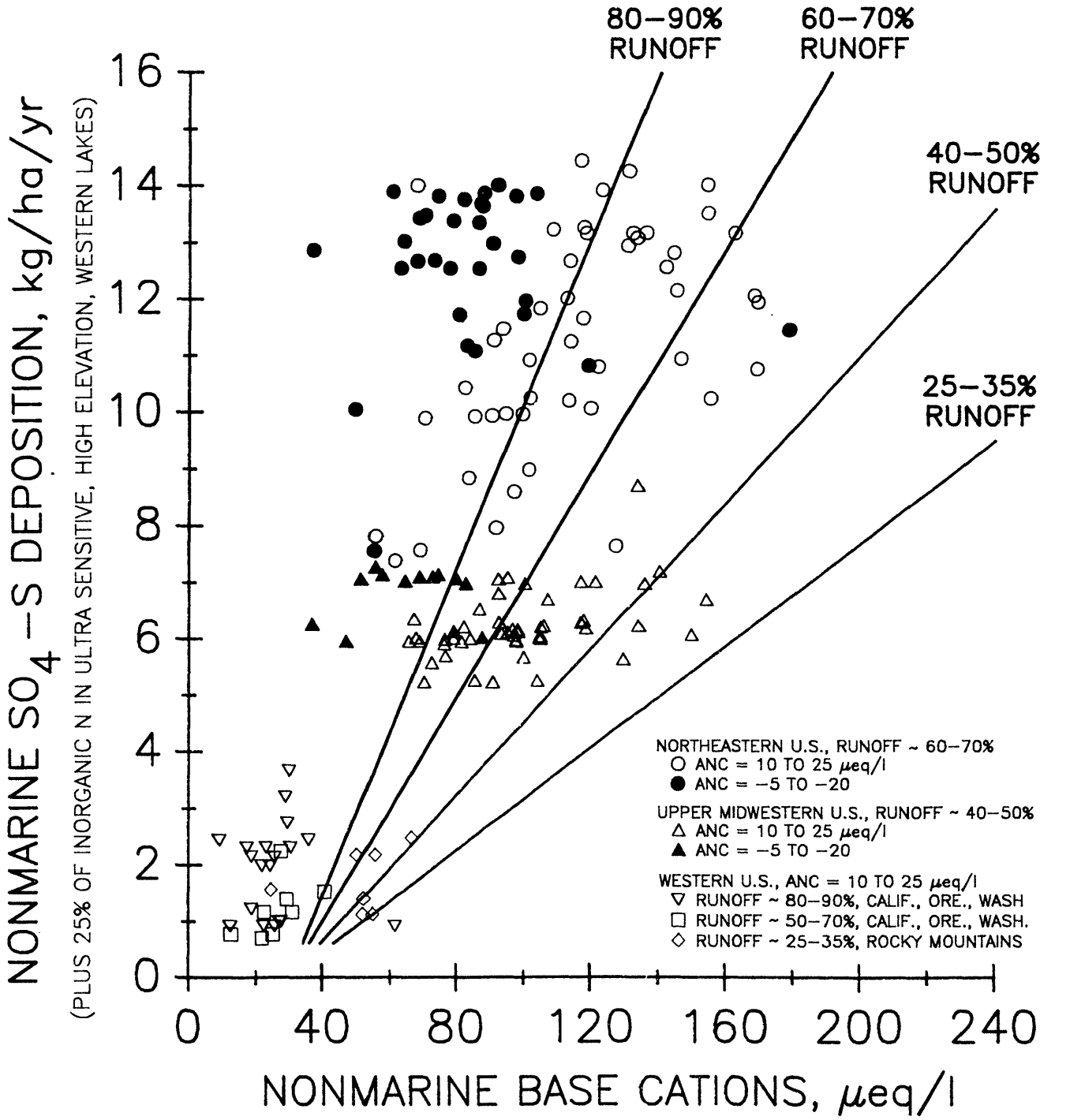

Figure 3.—*Base cations vs. deposition, "safe" lines. Lakes to the right of the appropriate runoff line are unlikely to be acidic. (Lake chemistry data from Kanciruk* et al. *1986 and Eilers* et al. *1987.)*

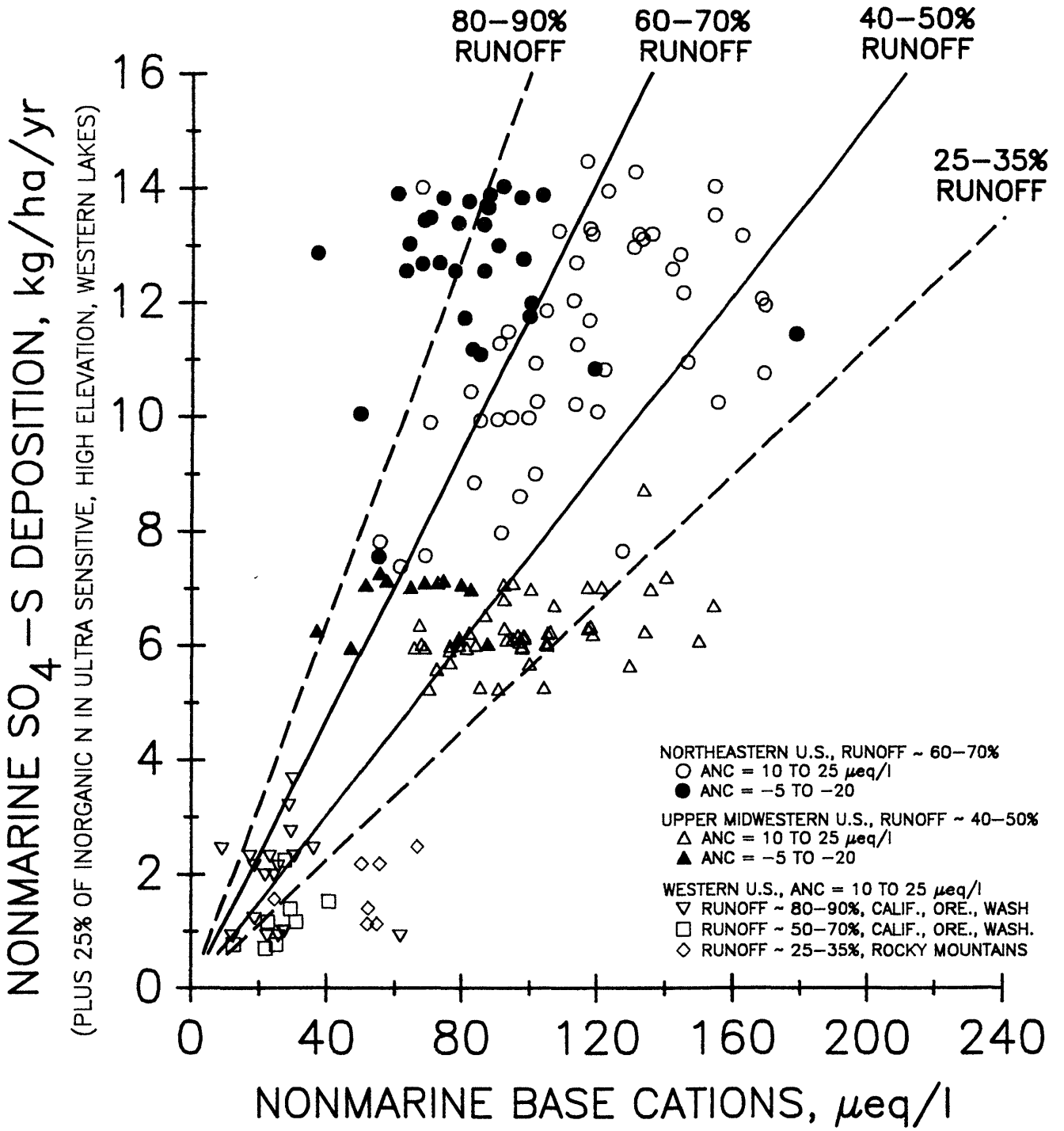

Figure 4.—*Base cations vs. deposition, "danger" lines. Lakes to the left of the appropriate runoff line are likely to be acidic. (Lake chemistry data from Kanciruk* et al. *1986 and Eilers* et al. *1987.)*

In figure 3, four lines estimate the limits of deposition to which lakes with a particular base cation concentration and runoff percentage can be subjected and still maintain a "safe" level of ANC. The lower end (zero deposition) of each line begins at a base cation concentration of 30 μeq/l. This value was selected empirically from the data in figure 1—among the lakes in the Pacific Northwest, where deposition is low, lakes with ANC's around 25 have base cation concentrations of about 30 μeq/l. The "safe" lines are drawn so that, except for a few outliers, all of the lakes in a particular runoff percentage category with ANC's of 10 to 25 μeq/l fall on or to the left of the line for that runoff category. Lakes in the 80-90 percent runoff category are only found at high elevations in the mountains along the Pacific Coast. Here deposition rates are low. For want of actual lake data, the position of the high deposition end of the "safe" lines for 80-90 percent runoff was estimated from the slopes of the 40-50 percent and 60-70 percent lines and the "evapoconcentration" ratios for the different runoff percentages. The same was done for the 35-45 percent runoff line for Rocky Mountain lakes. Runoff percentages were estimated from average annual precipitation and runoff maps compiled by Moody *et al.* (1985).

The amount of runoff relative to the amount of precipitation on a lake/watershed system affects how lake chemistry responds to acid loadings. As more of the precipitation is lost to evapotranspiration and less is yielded to runoff, acid deposition is in effect concentrated. This process is often referred to as evapoconcentration. The impact on the lake of a given acid loading is greater as the evapoconcentration ratio increases. For 50 percent runoff, the evapoconcentration ratio is 2:1; for 65 percent runoff, 1.54:1, etc. Also, larger runoff amounts are associated, on the average, with larger precipitation amounts. For a given rate of SO_4-S deposition, the greater the amount of precipitation, the lower the average SO_4-S concentration is in the precipitation, and the less the lake chemistry is affected. The effect of different runoff percentages or evapoconcentration ratios is illustrated in figure 3 through 6. The relationships among ANC, base cations, and deposition differ depending on the percentage of precipitation that is yielded as runoff.

Suppose a land manager is concerned about whether some lakes in a sensitive area—lakes that are presently not acidic and have ANC's of 25 μeq/l or greater—will become acidic if deposition were to increase to a particular level. If, given the projected deposition level and the lakes' base cation concentrations, the lakes fall to the right of the appropriate "safe" line in figure 3 for the lakes' runoff category, then the land manager can feel fairly certain that the lakes will maintain an ANC of at least 25 μeq/l and that the biota will not be adversely affected at that deposition level. A lake that falls to the left of the "safe" line at a particular deposition level will not necessarily be harmed by that deposition level. However, the farther to the left of its "safe" line a lake plots, the lower its ANC and pH are likely to be, on the average, as a result of deposition, and the more likely it is to be adversely affected.

In figure 4, four "danger" lines are shown, representing deposition amounts above which lakes with given cation concentrations and runoff percentages are likely to be adversely affected. The lower end of each line begins at a base cation concentration of zero. This represents a hypothetical "distilled water" lake with zero HCO_3 and zero cations, affected by zero deposition. Since it has no buffering capacity, essentially any acid deposition added to this hypothetical lake would acidify it. The "danger" lines for 40-50 percent and 60-70 percent runoff are drawn so that, except for a few outliers, all of the lakes in a particular runoff class with ANC's of -5 to -20 μeq/l fall on or to the left of the line for that runoff class. The slopes of the 25-35 percent and 80-90 percent runoff "danger" lines are estimated from the slopes of the 40-50 percent and 60-70 percent runoff "danger" lines and the relationship among the slopes of the four "safe" lines. If, given a lake's base cation concentration and the deposition level to which it is estimated the lake will be subjected, a lake plots to the left of the appropriate "danger" line for the lake's runoff class, then the land manager can be reasonably certain that the lake will be adversely affected. The farther to the left of the "danger" line a lake lies, the more likely it is, on the average, to be affected.

A land manager may need to estimate the susceptibility to acid deposition of lakes for which few or no base cation data are available. In this situation, lake water conductivity can be used as a "quick and dirty" substitute for base cation concentration. Conductivity is easily and cheaply measured in the field, and conductivity data are often available for more lakes than are cation data. Figures 5 and 6 are similar to figures 3 and 4, except that deposition is plotted against adjusted lake conductivity values instead of base cation concentrations. The "safe" and "danger" lines in figures 5 and 6 can be interpreted the same way as in figures 3 and 4. The data used in all four

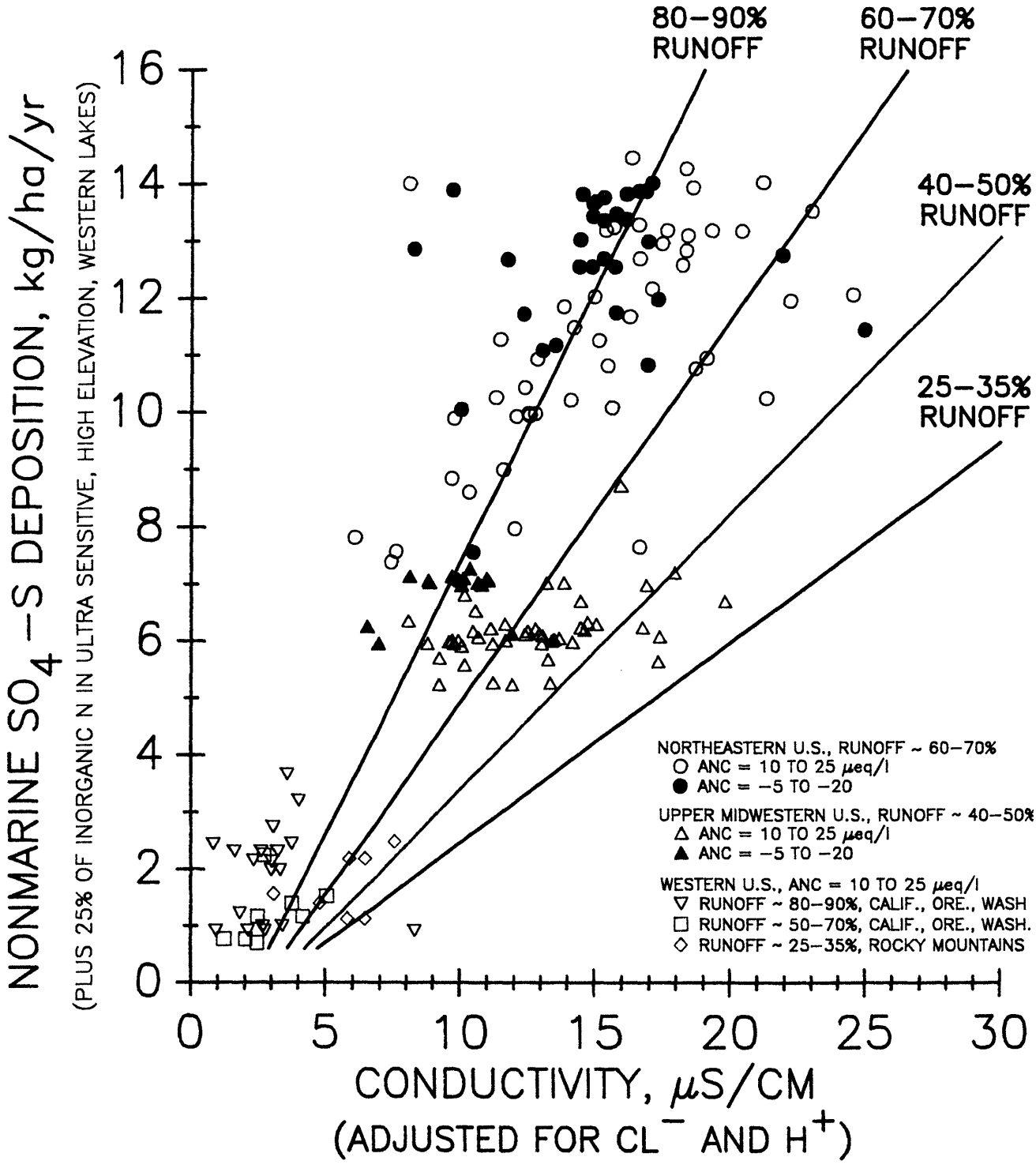

Figure 5.—*Conductivity vs. deposition, "safe" lines. Lakes to the right of the appropriate runoff line are unlikely to be acidic. (Lake chemistry data from Kanciruk* et al. *1986 and Eilers* et al. *1987.)*

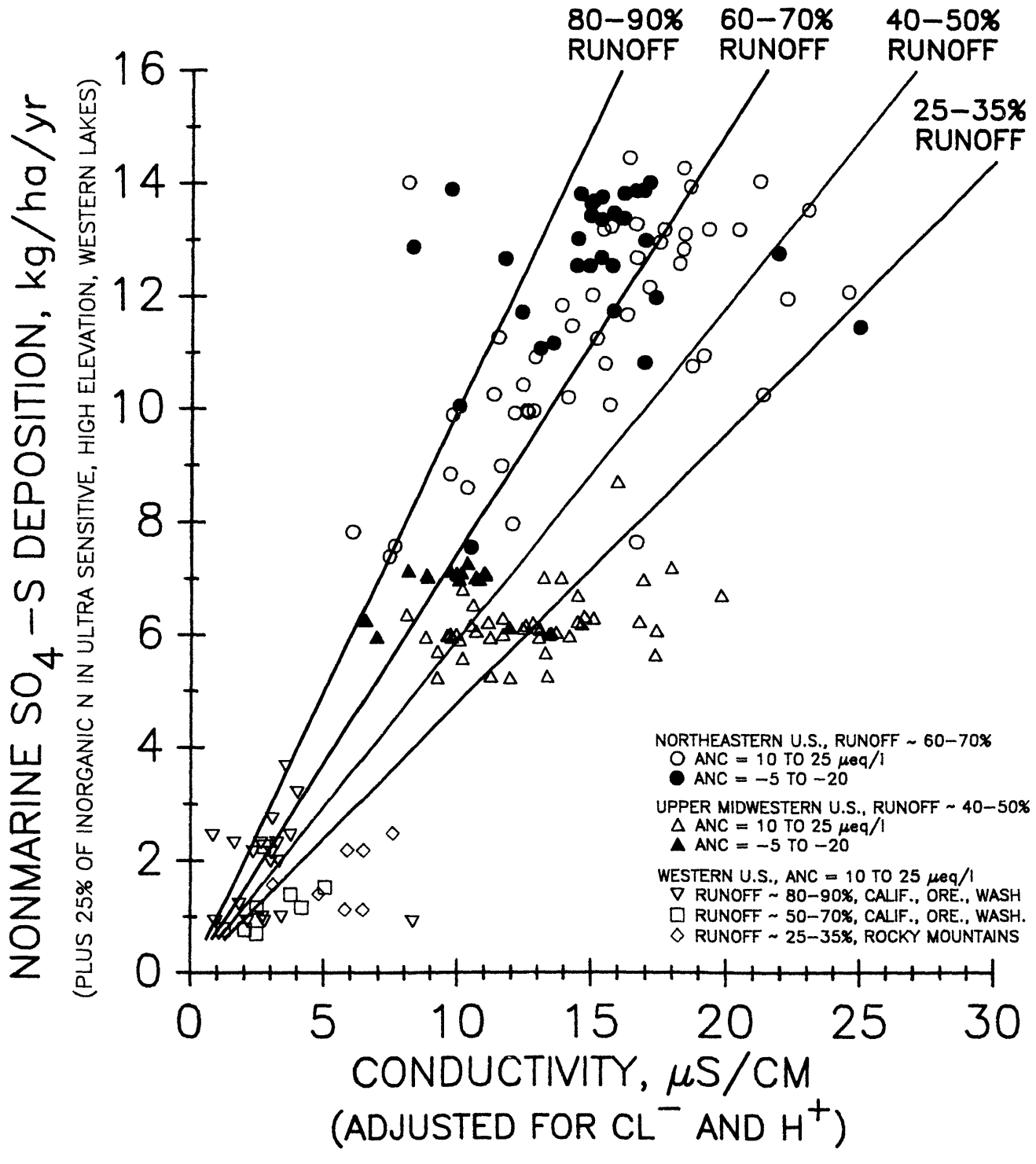

Figure 6.–*Conductivity vs. deposition, "danger" lines. Lakes to the left of the appropriate runoff line are likely to be acidic. (Lake chemistry data from Kanciruk* et al. *1986 and Eilers* et al. *1987).*

figures are from the same lakes. The conductivity of a water sample is determined by the concentrations of all ions dissolved in the water, not just base cations. Consequently, the relationship between ANC and conductivity is more variable than that between ANC and base cations. This can be seen in the considerable overlap between lakes in the two ANC categories in figures 5 and 6. Conductivity should only be used as an indicator of lake sensitivity where cation data are not available.

Sulfur Deposition

The total SO_4-S deposition rates shown in figures 3 through 7 were calculated as the product of the mean annual precipitation amount and mean volume-weighted SO_4-S concentration in the precipitation at each lake (wet deposition), plus an estimated amount of dry deposition. For the Northeastern and Upper Midwestern United States, mean volume-weighted concentrations of SO_4-S in precipitation were calculated from NADP/NTN (National Atmospheric Deposition Program / National Trends Network) data for all stations of record in the Northeast/Midwest area (70 sites) from July 1978 to May 1987 (NADP 1988). In addition, mean wet SO_4-S concentrations for 17 sites in southern Ontario were calculated from APIOS (Acid Precipitation in Ontario Study) data (APIOS 1987). From these data, contour maps of SO_4-S concentration in precipitation in the Northeast and Midwest were drawn, with contour intervals of 0.1 mg/l. Maps were hand-drawn separately by the author and by Dr. E. S. Verry, USDA, Forest Service, North Central Forest Experiment Station, Grand Rapids, MN. The maps were then compared and differences (minor) reconciled. On these maps were drawn 0.2° by 0.2° latitude/longitude grids, and the wet SO_4-S concentration at each node was estimated. Each lake of interest was then located by latitude and longitude and a mean volume-weighted SO_4-S concentration in precipitation at that location was computed from the concentrations at the nearest four nodes.

Precipitation SO_4-S concentrations for the lakes in the Western United States, located in the Sierra-Nevada and Cascade mountain ranges and the Rocky Mountains, were estimated from the relatively few higher elevation NADP collection sites and from high-elevation snow chemistry studies (Brown and Skau 1975, Melack *et al.* 1982, Reddy and Claassen 1985, Duncan *et al.* 1986, Laird *et al.* 1986, Loranger and Brakke 1988).

Wet SO_4-S deposition amounts for lakes in the southern Blue Ridge region and in Florida, shown in figure 9, are from concentrations calculated by Husar (1986) from NADP data for 1982 through 1984, and precipitation amounts from the same period.

Precipitation amounts for lakes in the Northeast were taken primarily from Knox and Nordenson (1955), who described and mapped annual precipitation patterns for all of New England and most of New York for the period 1930 to 1949. Knox and Nordenson developed equations for estimating precipitation amounts from elevation, exposure, distance from the Atlantic Coast, etc., and used streamflow records to verify and adjust these estimates. Precipitation for south-central New York and northern New Jersey were taken from annual precipitation maps for the period 1931 to 1955 (USCOMM 1958, 1970). For Pennsylvania, precipitation data were taken from maps for the period 1931 to 1960 (USCOMM 1968). In the Upper Midwest, precipitation data were taken from maps supplied by the respective state climatologists for the following periods: Wisconsin, 1951 to 1980; Michigan, 1940 to 1969; Minnesota, 1951 to 1980. Precipitation data from all base periods were adjusted to a 1951 to 1980 base through analysis of long-term precipitation records for appropriate HCN (Historical Climatology Network) sites (Quinlan *et al.* 1987). Annual precipitation estimates for the Western lakes were derived from a variety of published and unpublished maps, and from consultations with state climatologists and national park and national forest hydrologists.

For the Eastern, Southern, and Midwestern lakes, dry SO_4-S deposition was estimated to be equal to 30 percent of wet deposition, and this amount was added to the wet deposition for each lake to give an estimate of total SO_4-S deposition. This 30 percent value represents the combined judgment of the participants at the 1988 Cary Arboretum Workshop (Fox *et al.* 1989). The deposition estimates for the Western lakes were based to a large extent on snow chemistry studies that involved collection and analysis of bulk snow samples which, unlike NADP/NTN samples, contained both wet and dry deposition. Consequently, no additional correction for dry deposition was made for the Western lakes.

Nitrogen Deposition

The primary anions associated with acids in atmospheric deposition are SO_4 and NO_3. Ammonium

(NH_4) in deposition also has acidification potential when it is assimilated by the biota or oxidized to NO_3 in the soil or water. Lake acidification models typically concentrate on SO_4 and ignore nitrogen forms. Nitrate is taken up rapidly by aquatic and terrestrial biota. This process produces alkalinity that neutralizes the acidity associated with the NO_3; thus, NO_3 does not appear to promote lake acidification in most cases (Galloway *et al.* 1983, Driscoll and Likens 1982). There are some exceptions to this general observation.

Terrestrial ecosystems that have received high levels of N deposition for many years may become N saturated and export significant amounts of NO_3. This has been observed in Europe (Grennfelt and Hultberg 1986). Also, large NO_3 losses often occur following severe disruptions of the vegetation, such as after clear cuts and fires (Likens *et al.* 1970, Bormann and Likens 1979). Lake/watershed systems located in rocky basins with thin to nonexistent soils and scant vegetation, primarily at high elevations in the Western United States, may not be able to assimilate their entire NO_3 load, even if NO_3 deposition is fairly low. In Norwegian lake/watershed systems, which are chemically and biologically similar to high-elevation areas in the Western United States, the terrestrial portions of the systems have been found to retain from 75 to 100 percent of incoming NO_3 (Henriksen and Brakke 1988). The Western lakes shown in figures 3 through 6 are the most sensitive lakes among some 752 sampled in EPA's Western Lake Survey. For these lakes, with base cation concentrations generally less than 50 μeq/l, a nitrogen assimilation of 80 percent was assumed, with an acidifying effect on the lakes by the other 20 percent. This was the general guideline recommended by the 1988 Cary Arboretum Workshop participants for use in high-elevation Western lakes where actual NO_3 assimilation data are not available (Fox *et al.* 1989). The effective deposition rates for these lakes, shown in figures 3 through 6, were estimated to be equal to total SO_4-S deposition plus 25 percent of N deposition (25 percent rather than 20 because, on an equivalent weight basis, 1 g of N = 1.15 g of S).

Marine Influence Adjustment–Precipitation

Titration models of lake acidification assume that the SO_4 content of precipitation is proportional to precipitation acidity and its potential to acidify lakes. Near the seacoasts, however, precipitation and airborne particulates contain substantial amounts of chloride (Cl), Na, Mg, and SO_4, and lesser amounts of other ions derived from ocean spray. Sea salts increase SO_4 deposition in coastal areas, but this additional SO_4 is associated with base cations rather than acids and does not have the same acidification potential. Therefore, before precipitation chemistry data can be used in a lake acidification model, the SO_4 derived from sea salts must be subtracted.

Chloride is typically used as an indicator of the presence of sea salts in precipitation. The concentration of Cl in seawater is about 19,000 mg/l. Except near the coasts, rain and snow generally contain less than 0.2 mg/l. To correct for sea salt effects, it is assumed that all Cl in precipitation is from sea salts, and an amount is subtracted from the SO_4 concentration in proportion to its occurrence in seawater relative to Cl (table 2):

"Nonmarine" SO_4 in precipitation =
total SO_4 - (Cl x 0.105).

The nonmarine SO_4-S deposition rates for the lakes shown in figures 3 through 6 were calculated using this method. The deposition rates in the figures are in kg/ha/yr of SO_4-S rather than SO_4. To convert, SO_4-S equals SO_4 divided by 3. In a study of seawater and marine precipitation chemistry, Keene *et al.* (1986) suggested that not all Cl in precipitation is of marine origin, that some may be due to hydrochloric acid vapors scavenged from the air, and that individual precipitation chemistry data sets should be analyzed to determine whether Cl is an unbiased indicator of marine influence. Such an analysis, however, is beyond the scope of this paper. For the purposes of this paper, all Cl in precipitation is assumed to be of sea salt origin.

Table 2.–*The principal dissolved ions in sea water (from Hem 1970)*

	mg/l	μeq/l	Percent of Cl μeq/l basis
Chloride (Cl)	19,000	535,990	100.0
Sodium (Na)	10,500	456,750	85.2
Sulfate (SO_4)	2,700	56,210	10.5
Magnesium (Mg)	1,350	111,050	20.7
Calcium (Ca)	400	19,960	3.7
Potassium (K)	380	9,720	1.8
Bicarbonate (HCO_3)	142	2,840	.5

Marine Influence Adjustment—Lake Base Cations

A major premise of the titration model of lake acidification is that, in lakes unaffected by acid deposition, the concentration of base cations is proportional to ANC. Therefore, for sensitive lakes that have been affected by acid deposition, base cations can be used as an indicator of the lakes' original ANC and the amount of ANC that has been lost. In figures 3 and 4, the concentrations of the basic cations Ca, Mg, K, and Na are used as an indicator of original lake ANC's. Titration models for lake acidification are commonly based on only the divalent cations Ca and Mg. While Ca and Mg are usually the dominant cations associated with ANC, the weathering of minerals containing K and Na can also contribute significantly to ANC, and these cations are included here. The titration model is confounded by lakes containing significant amounts of base cations that are balanced by anions other than HCO_3 (the main component of ANC), mainly Cl and SO_4. The source of much of this Cl and SO_4 and associated base cations is directly or indirectly marine, including sea spray, salt applied to roads in the winter for deicing, and sedimentary rock formations of marine origin. When base cations are used as to estimate lake sensitivity to acidification, as in figures 3 and 4, elevated concentrations of cations not associated with ANC can lead to erroneous conclusions. Consequently, the base cation concentrations of the lakes shown in figures 3 and 4 have been adjusted as much as possible for these influences.

In the same way that Cl is used to estimate the sea salt component of various ions in precipitation, the Cl content of lakes can also be used to estimate that portion of the base cations derived from road salt, sea spray, sea salts in marine rocks, etc. In most nonmarine rock types, Cl is present in lower concentrations than any of the other common ions found in water (Hem 1970); and in lakes unaffected by sea salts, chloride is usually low (0.5 mg/l or less). Chloride-bearing minerals such as feldspathoid sodalite ($Na_8(Cl_2(AlSiO_4)_6)$) do occur in igneous rocks, but typically not in significant amounts. As with precipitation, the "non-sea-salt" portion of various ions in lake water is estimated by assuming that all the Cl in lake water is from marine sources, and subtracting from the concentration of each ion an amount proportional to the ratio of each ion to Cl in seawater (table 2). The base cation concentrations of all the lakes plotted in figures 3 and 4 have been "Cl-adjusted" following this procedure:

"Cl-adjusted" base cations =

$$(Ca - (Cl \times 0.037)) + (Mg - (Cl \times 0.207)) + (Na - (Cl \times 0.852)) + (K - (Cl \times 0.018)) = Ca+Mg+K+Na - (Cl \times 1.114)$$

(all concentrations expressed in μeq/l).

In using the Cl content of precipitation or lake water to estimate the portion of the base cations coming from sea salt or other marine sources, it is assumed that the marine-derived ions are supplied in the same proportion relative to each other in which they occur in seawater. Where salts from sea spray are the only marine source, this assumption is probably valid and the Cl correction procedure reasonably accurate. If lakes are affected by rock salt spread on roads for deicing, this procedure overestimates marine-derived cations by a few percent, because the amount of SO_4 and associated cations in rock salt is less than in seawater.

Where marine sedimentary rock formations influence lake chemistry, the potential error in the use of Cl to estimate "marine" cations is much greater. Two common sources of Cl and SO_4 and associated cations in marine formations are the minerals halite (sodium chloride) and gypsum or anhydrite (calcium sulfate). These minerals may occur separately in relatively pure deposits, or together in any proportion. In addition, Twenhofel (1950) lists 25 other chloride and sulfate minerals that commonly occur in marine sedimentary formations. Various sulfide minerals that produce SO_4 upon weathering are also found. Obviously, while Cl is a good general indicator of marine influence, the concentrations of various ions relative to Cl in marine formations, and in the surface and ground water draining from them, may or may not be similar to that of seawater. Because of this uncertainty, any lakes with Cl concentrations greater than 400 μeq/l (about 14 mg/l) have been eliminated from figures 3 through 6. In the areas included in EPA's Eastern Lake Survey, high Cl lakes are common in Rhode Island, Massachusetts, and southern New Hampshire; from northeastern Pennsylvania through northern New Jersey and southeastern New York to southwestern Connecticut; and in Florida. This reflects the marine geologic history of these regions.

Marine Influence Adjustment—Lake Conductivity

In figures 5 and 6, conductivity is used in place of base cations as an indicator of lake sensitivity. Because conductivity is a measure of the total concentration of ions dissolved in water, it is related to, and is used here as a substitute for, base cations. As with base cations, conductivity must be corrected for sea salt influences. This correction is critical because conductivity is affected not only by sea salt base cations, but also by the marine anions, mainly Cl and SO_4. The contribution to conductivity of the ions commonly found in water is shown in table 3.

Table 3.–*Contributions of various ions to conductivity (Vanysek 1984)*

Ion	Conversion factor
Hydrogen (H)	0.34965
Calcium (Ca)	.05947
Magnesium (Mg)	.05300
Potassium (K)	.07348
Sodium (Na)	.05008
Bicarbonate (HCO_3)	.04450
Sulfate (SO_4)	.08000
Chloride (Cl)	.07631

Concentration of individual ion (μeq/l) x Conversion Factor = Conductivity (μSiemens/cm) due to that ion

Again, all Cl in the lake water is assumed to be of marine origin. The contribution to conductivity of the sea salt portion of all the major ions is estimated from the concentrations of the ions in seawater relative to Cl (table 2) and the concentration-to-conductivity conversion factor for each ion (table 3).

Non-sea-salt conductivity = total measured conductivity - (Cl x 0.1422) (Cl concentration expressed in μeq/l and conductivity in μSiemens/cm)

To be used as a reasonable substitute for base cations, conductivity must also be adjusted for pH (hydrogen ion concentration). This is especially true in acidic waters. At equal concentrations (μeq/l), hydrogen ions contribute from four to seven times as much to conductivity as the other common ions (table 3).

Adjusted conductivity = unadjusted conductivity - (H x 0.34965), where H is the hydrogen ion concentration in μeq/l (H = 10 raised to the -pH power, then that quantity multiplied by 1,000,000). The conductivities for all lakes shown in figures 5 and 6 have been adjusted by subtraction of that portion derived from either marine influences or pH. The lakes shown in figures 5 and 6 are the same lakes shown in figures 3 and 4; lakes with Cl concentrations greater than 400 μeq/l have been eliminated.

EXAMPLE OF NOMOGRAPH USE

Figures 7 and 8 illustrate the use of the nomographs to assess the effect of a 3 kg/ha/yr increase in total nonmarine SO_4-S deposition on two groups of sensitive Midwestern lakes—50 lakes in northeastern Minnesota and 53 lakes in northwestern Wisconsin (lake chemistry data from the Eastern Lake Survey). The base cation concentration (fig. 7) and conductivity (fig. 8) of each lake are plotted against present SO_4-S deposition and present-plus-3 kg/ha/yr, and compared to the "safe" and the "danger" or "acidic" lines for 40-50 percent runoff.

Although they are sensitive, the Minnesota lakes shown in figures 7 and 8 appear to be sufficiently buffered to withstand the present deposition rate of about 3.2 to 3.6 kg/ha/yr with no damage; based on either conductivity or cation concentration, all fall in the "safe" area of the nomographs. They all have ANC's above 40 μeq/l. All pH's are above 6.5 except for three highly colored lakes strongly influenced by organic acids. Given a substantial 3 kg/ha/yr increase in SO_4-S deposition, one Minnesota lake falls a little to the left of the cation-based "safe" line and others move close to it (fig. 7). Several fall to the left of the conductivity-based "safe" line (fig. 8). This does not mean that the most sensitive of these lakes will necessarily become acidic, but it does increase the likelihood that ANC will go below 25 μeq/l and pH below 6.5, and that some aquatic organisms may begin to be affected.

About 200 to 300 miles to the southeast in northwestern Wisconsin, the situation is different. Here some very sensitive lakes are found, and deposition is greater. At present SO_4-S deposition rates of about 5.0 to 5.5 kg/ha/yr, almost 25 percent of the Wisconsin lakes shown fall in the critical area to the left of the cation-based "safe" line. According to Eastern Lake Survey data, 7 of the 53 lakes shown have

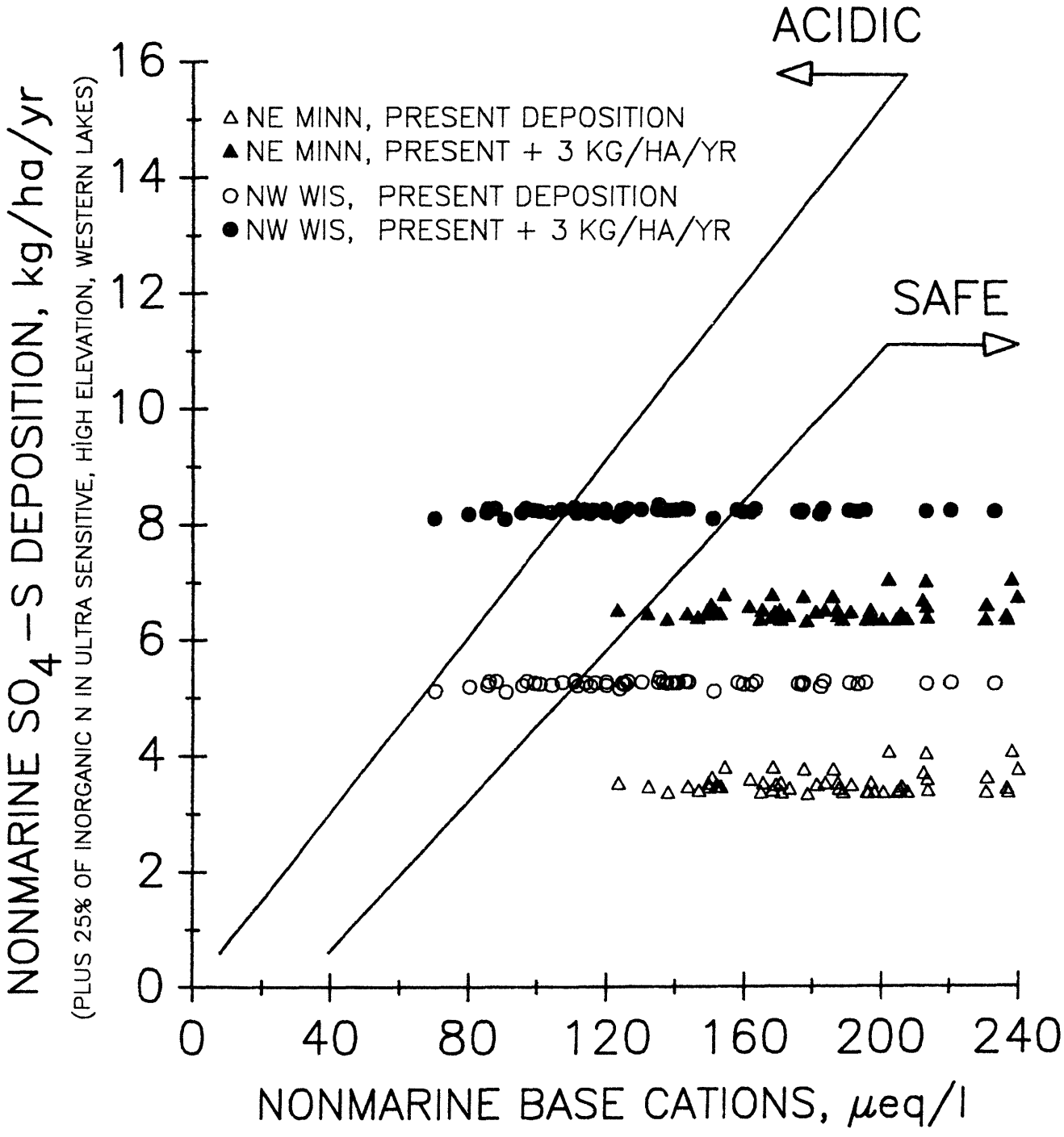

Figure 7.—*Base cation and deposition relationship of two groups of Midwestern lakes, at present deposition rate and at present deposition plus 3 kg/ha/yr. (Lake chemistry data from Kanciruk* et al. *1986 and Eilers* et al. *1987.)*

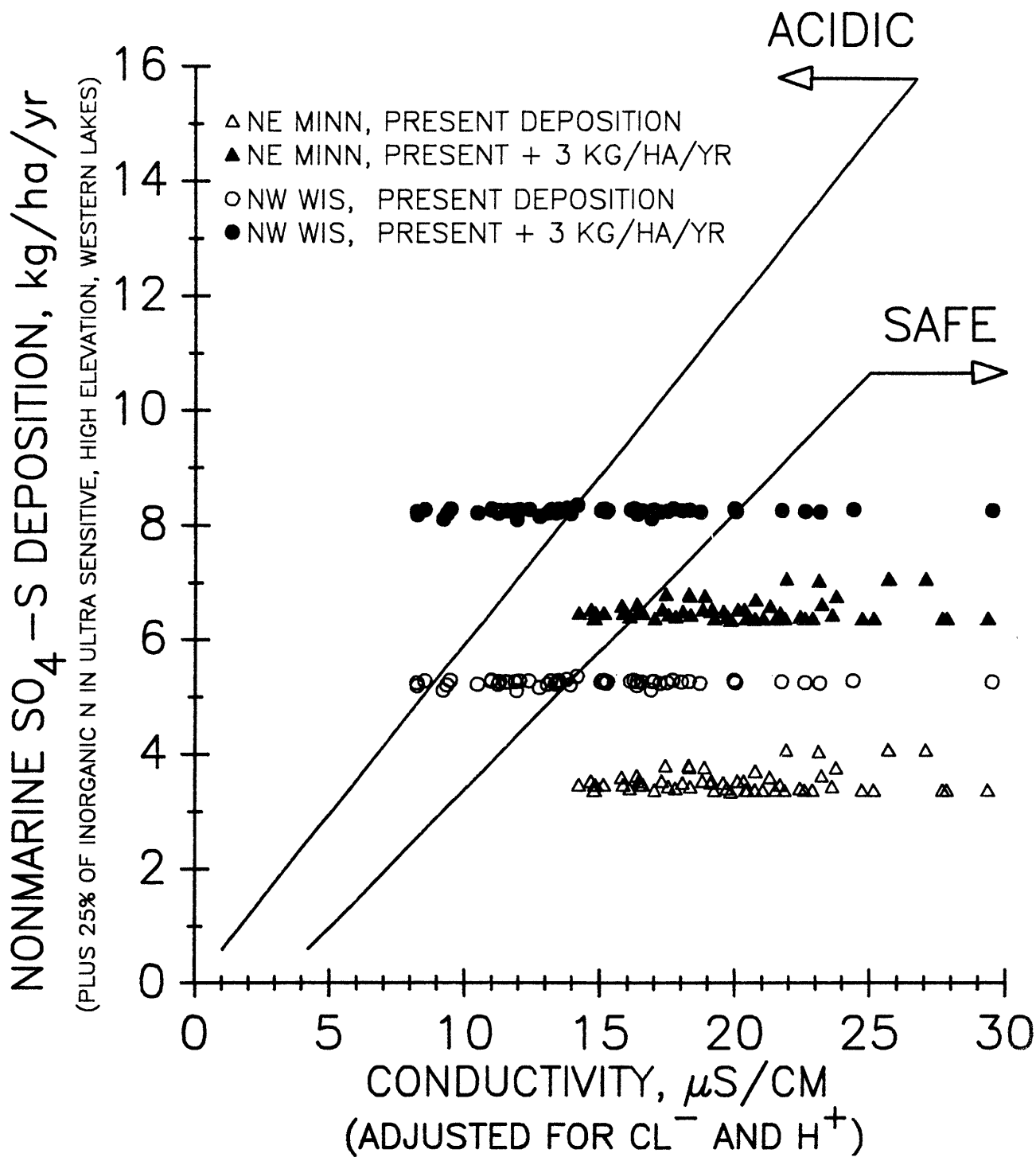

Figure 8.—*Conductivity and deposition relationship of two groups of Midwestern lakes, at present deposition rate and at present deposition plus 3 kg/ha/yr. (Lake chemistry data from Kanciruk* et al. *1986 and Eilers* et al. *1987.)*

ANC's of 25 μeq/l or less. Six have pH's below 6.0. The biota of these lakes probably reflects the low pH and ANC values. Any significant increase in deposition is likely to result in noticeable damage to some lake ecosystems. As figures 7 and 8 indicate, a 3 kg/ha/yr deposition increase would be likely to cause many of these Wisconsin lakes to become acidic, with pH's in the 5.0 to 5.5 range, and seriously affect populations of fish and other aquatic organisms.

DISCUSSION

In figure 9, the SO_4 concentrations of nine groups of sensitive lakes (ANC ≤ 200 μeq/l) from the Eastern and Western Lake Surveys are plotted against the median nonmarine SO_4-S deposition loads for each group. For each group of lakes, the minimum, 15th percentile, median, 85th percentile, and the maximum SO_4 concentrations are shown. While there is much variation, lake SO_4 concentrations are on the average very low where deposition is low, and increase linearly as deposition increases. Overall, concentrations of SO_4 in lakes increase with increasing deposition, from the Sierra Nevada and Cascade mountain ranges on the West Coast to the Adirondack, Catskill, and Pocono mountains in New York and Pennsylvania, at the rate of about 9 to 10 μeq/l of lake SO_4 for each kg/ha/yr of SO_4-S deposition.

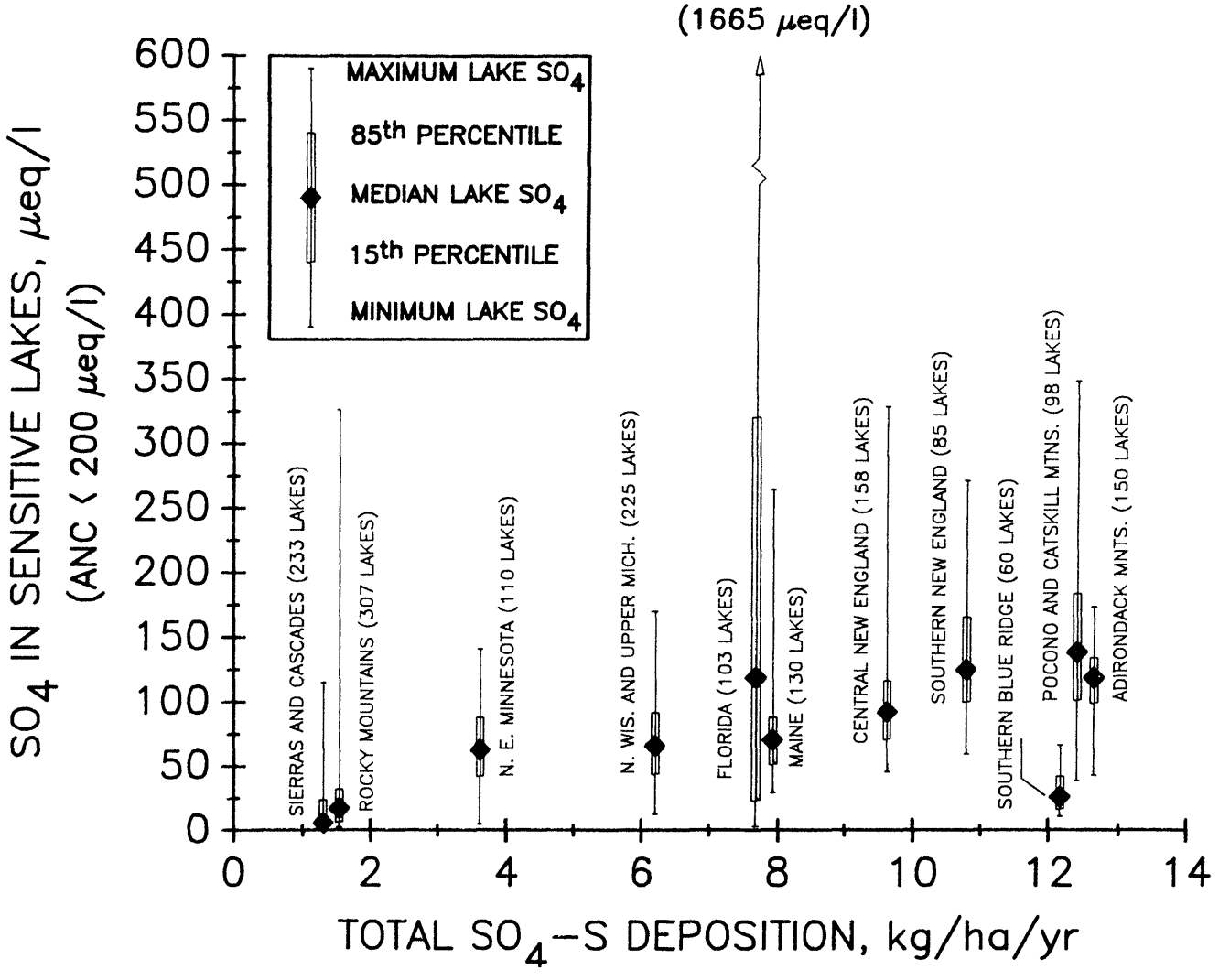

Figure 9.—*Total SO_4-S deposition and lake SO_4 concentration for sensitive lakes (ANC < 200 μeq/l) in the Eastern and Western United States. (Lake chemistry data from Kanciruk* et al. *1986 and Eilers* et al. *1987.)*

To maintain the balance between cations and anions in solution, an acid deposition-induced increase in lake SO_4 must be accompanied by an increase in cations, due to increased mineral weathering and cation leaching, and/or by a decrease in ANC, due to the addition of acids and the consumption of HCO_3. The amount by which base cations increase relative to the increase in lake SO_4 is often called the cation weathering enhancement factor (f):

$$f = \Delta \text{ cations} / \Delta SO_4.$$

The f value is an expression of how lake chemistry responds to acid deposition. If for a particular lake f = 1, base cations would increase with SO_4 in response to acid deposition and ANC would be unchanged. If f = 0, the lake's response to acid deposition would be that ANC would decrease as SO_4 increased while the concentration of base cations would stay the same. The f value of a lake depends on the mineralogy of the watershed and flow paths of water through it. In general, the more weatherable the carbonate and cation-aluminum-silicate minerals with which water comes in contact as it flows to a lake, the higher the f value of the lake will be. Using data from a variety of Scandinavian and North American lakes, Wright (1983) estimated that typical f values are between 0 and 0.4. Later, Wright *et al.* (1986) estimated that f values could go as high as 0.6. More recently, Henriksen *et al.* (1987, 1988), after analyzing data from 1,000 Norwegian lakes, suggested that f values probably range from 0 in lakes with very low concentrations of base cations to 1.0 in lakes with high base cation contents. Simulation models used by Labienec *et al.* (1988) indicate that f values can decline over time with continued acid deposition.

The f values of the lakes shown in figures 3 through 6 are not known. These figures were drawn assuming f values of zero. These very sensitive lakes, containing only small amounts of base cations and ANC, are sensitive because water flowing from their watersheds comes in contact with only small amounts of weatherable minerals that supply cations and ANC to the lakes. Under these conditions, acid deposition is not likely to increase mineral weathering and cation leaching. The response of these lakes to acid deposition is likely to be a loss of ANC while cation concentrations remain about the same. Figures 3 through 6 will overestimate the sensitivity of lakes that have f values significantly greater than zero.

While lake SO_4 generally increases as SO_4 deposition increases, there is considerable variation among lake SO_4 values at any particular deposition level (fig. 9). This variation is due to differences in the supply of SO_4 from natural sources, as well as differences in the amounts of SO_4 from both deposition and natural sources that are retained by the watersheds or that pass through the watersheds to the lakes. Sulfur is widely distributed in both igneous and sedimentary rocks, either in reduced forms as metallic sulfides such as pyrite (FeS_2), chalcopyrite ($CuFeS_2$), pyrrhotite (FeS), and galena (PbS), or as calcium, magnesium, potassium, and sodium sulfate minerals, the most common of which is gypsum ($CaSO_4 \cdot 2H_2O$) (Hem 1970, Twenhofel 1950). Base cations associated with geologic SO_4 add considerable variation to the relationship between ANC and base cations. In figures 3 through 6, much of the scattering of data points among lakes in any given deposition-runoff-ANC category can be attributed to differences in SO_4 content.

At a given base cation concentration or conductivity, a lake that contains a substantial amount of SO_4 will have proportionately less ANC and will be more susceptible to acid deposition than a lake with less SO_4. For example, in a study of lake chemistry and its relationship to acid deposition in the Upper Midwest, Nichols and McRoberts (1986) found that as deposition levels increased from north-central Minnesota to central Lower Michigan, lake ANC's decreased linearly with respect to base cation concentrations and lake SO_4 levels increased by similar amounts. However, the SO_4 content of a group of lakes sampled near the eastern end of the Superior National Forest in northeastern Minnesota averaged about 50 μeq/l higher than would be expected from deposition in that area, and the ANC's of these lakes were about 50 μeq/l less than expected compared with their base cation concentrations. Nichols and McRoberts, citing a study of Minnesota geology (Sims and Morey 1972), attributed these differences to the occurrence of extensive sulfide mineral deposits in the bedrock formation with which these lakes are associated.

Not all of the SO_4 that enters a lake/watershed system from atmospheric deposition or from mineral weathering shows up as SO_4 in the lake. Sulfate reduction in lake sediments was mentioned earlier as a mechanism for removing SO_4 from lake water and generating ANC. In addition, other processes occur

generating ANC. In addition, other processes occur in the watershed that can prevent much of the deposited or geologically generated SO_4 from reaching the lake.

Sulfate is adsorbed by attraction to positively charged sites in the soil, sites associated with hydrous oxides of iron (Fe) and aluminum (Al) and the edges of 1:1 clay mineral particles (Bohn *et al.* 1986). Highly weathered soils normally contain considerable quantities of Fe and Al hydrous oxides and 1:1 clays and have high capacities to adsorb SO_4. Adsorption capacity increases with decreasing soil pH. Figure 9 shows that, compared with other groups of lakes sampled in the Eastern Lake Survey, sensitive lakes located in the southern Blue Ridge region contain very low amounts of SO_4 relative to SO_4-S deposition. Most of the SO_4 that falls on these watersheds is apparently adsorbed by the old, highly weathered soils of the region and doesn't reach the lakes, at least at present. In the future, less SO_4 may be retained by the soil as SO_4 adsorption sites become saturated.

Much SO_4 is also retained in peatlands by plant uptake and subsequent accumulation in the peat, and by biochemical reduction in a process similar to that occurring in lake sediments (Hemond 1980, Rippon *et al.* 1980, Bayley *et al.* 1986, Verry 1987). While peatlands typically retain on an average annual basis significantly more SO_4 than terrestrial ecosystems, much season-to-season and year-to-year variation has been observed in peatland SO_4 retention related to climatic and hydrologic conditions. Nichols (1987) found that highly colored lakes in the Upper Midwest contained less SO_4 than clear lakes, presumably due to SO_4 reduction in peatlands in the watersheds of the colored lakes.

Among the lakes sampled in the Eastern Lake Survey, the Florida lakes are an extreme example of the variability of lake SO_4 content. The SO_4 content of sensitive (ANC $\leq$ 200 µeq/l) Florida lakes shown in figure 9 ranges from close to 0 to almost 1,700 µeq/l. Sulfate reduction has been found to be an important mechanism for removing SO_4 from many sensitive Florida lakes (Baker *et al.* 1986). The extremely high concentrations of SO_4 found in many Florida lakes is apparently due to natural geologic sources. Many lakes and streams in Florida are influenced by discharge from the Floridan aquifer system, a series of limestone and dolomite formations of Late Paleocene to Early Miocene age. Like many marine formations, the Floridan aquifer contains large deposits of gypsum that result in high SO_4 concentrations in the water in some areas (Johnston and Bush 1988). J. Eilers, Environmental Chemistry, Inc., Corvallis, OR (personal communication), suggests that the SO_4 in some Florida lakes may stem from irrigation of agricultural lands with Floridan aquifer water rather than from natural discharges from the aquifer.

Variation in natural sources and sinks of SO_4 is a potential source of error in the use of figures 3 through 6 to estimate the susceptibility of lakes to acidification. Lakes that, because of sources of SO_4 in their watersheds, have a much higher SO_4 concentration than would be expected for the rate of deposition to which they have been subjected (fig. 9), would be more likely to become acidified than would be indicated by their base cation concentrations or conductivities. For lakes with very low SO_4 levels relative to deposition due to SO_4 uptake by vegetation, SO_4 adsorption by upland soils, or SO_4 reduction and retention in wetlands or lake sediments, the opposite is true. Most of the outliers seen in figures 3 through 6—lakes with higher cation concentrations and conductivities than other lakes in their ANC and runoff groups—are high SO_4 lakes. No Florida lakes are included in figures 3 through 6. The extreme variation in SO_4 among Florida lakes makes these nomographs useless for estimating susceptibility to acidification. Many high SO_4 lakes in Florida with base cation and conductivities that would indicate high ANC's and pH's are actually highly acidic. Because it is easily measured, conductivity may be known or can be quickly determined for many lakes for which base cation concentrations are unknown. Figures 5 and 6 are intended for use in this situation; however, their limitations must be recognized. The relationship between ANC and conductivity in lakes is considerably more variable than that between ANC and base cations. Differences in the geologic SO_4 supply add much more variation to the ANC:conductivity relationship than to the ANC:base cation relationship, because both SO_4 and base cations contribute to conductivity. For lakes in any given deposition-runoff-ANC category, the scatter among the data points and the overlap between groups of acidic and nonacidic lakes is greater when conductivity is used as a measure of lake sensitivity (figs. 5 and 6) than when base cation concentration is used (figs. 3 and 4). For this reason, base cations rather than conductivity should be used whenever base cation data are available.

If, in addition to base cation concentrations, reliable ANC and SO_4 concentration values are available, these data can allow the land manager to make a more accurate estimate of lake susceptibility to acidification than can be made from cation concentrations alone. Consider, for example, a Midwestern lake with a nonmarine base cation concentration of 125 μeq/l, in an area subject to a SO_4 deposition rate of about 6 kg/ha/yr. This lake would fall just about on the "safe" line for Midwestern lakes in figure 3, indicating that the lake should have an ANC of 25 μeq/l or greater at its present deposition rate. At a SO_4 deposition rate of 7 kg/ha/yr, the lake would fall to the left of the "safe" line, into the questionable zone between the "safe" and "danger" lines. It would be difficult to estimate from cation data alone whether the lake would be adversely affected. If the natural SO_4 content of the lake was low and the actual ANC of the lake was known to be greater than 25 μeq/l, say 50-60 μeq/l, then an increase in SO_4 deposition from 6 to 7 kg/ha/yr would probably not cause a problem. If, conversely, much of the cation content of the lake was associated with a naturally high SO_4 content or, as in the case of highly colored lakes, associated with organic anions, and the lake's ANC was actually less than 25 μeq/l already, then the increased SO_4 deposition would likely have adverse effects.

The lake chemistry data upon which figures 3 through 6 are based were collected during the fall of the year. In late summer and fall, lake ANC is probably as high as at any time during the year. At higher latitudes and elevations, large amounts of SO_4 and NO_3 and associated acids, depending on deposition rates, may accumulate in the winter snowpack. In the spring, lake ANC's and pH's may be temporarily depressed by the acid load released during snowmelt. Some very sensitive lakes that have small, positive ANC's and pH's above 6.0 or 6.5 in the summer and fall may be partially acidic for a time during and just after the snowmelt period. This phenomenon is beyond the scope of this paper, but it may be an important consideration in evaluating the impacts of proposed emissions sources.

The nomographs presented in figures 3 through 6 are based on a simple empirical model. In general, simple models require less data, but yield results with more potential for error, than complex models. For any given acid deposition rate, figures 3 through 6 are intended to allow the land manager, with a minimum of available data, to place lakes into three groups: (1) those that are very unlikely to be adversely affected, (2) those that are very likely to be adversely affected, and (3) those for which more information is needed before the determination can be made. The nomographs were constructed so that the error involved in assigning a lake to group 1 or 2 is minimized. The obvious consequence is that more lakes are assigned to group 3. In some cases, with additional data such as ANC, SO_4, color, etc., some group 3 lakes can probably be reassigned to group 1 or 2. However, if the land manager requires a more accurate assessment of acid deposition effects, additional data will have to be collected and a more complex model used. Several more sophisticated models have been developed for predicting lake acidification, for example the MAGIC (modeling the acidification of ground water in catchments) model (Cosby *et al.* 1985), the ILWAS (integrated lake-watershed acidification study) model (Gherini *et al.* 1985), the ALaRM (acid lake reacidification model) model (DePinto *et al.* 1987), and the enhanced trickle-down model (Nikolaidis *et al.* 1988). After calibration for specific conditions unique to the area being studied, these models can simulate various aspects of lake response to acid deposition with better accuracy than can simple titration models. However, because of the time and data needed, they are not well suited to the problem addressed by this paper—a "first cut" evaluation of deposition impacts on lakes for which data are sparse, to determine whether or not a projected deposition rate is likely to acidify lakes and damage aquatic ecosystems.

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Presents a graphical procedure for evaluating the sensitivity of lakes to acidification due to acid deposition. The procedure is based on empirical relationships between sulfur (and in some cases nitrogen) deposition rates and lake pH, acid-neutralizing capacity, base cation concentrations, and the amount of runoff.

KEY WORDS: Acid-neutralizing capacity, base cations, conductivity, nitrate, sulfate.