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Areas Having Soil Characteristics That May Indicate Sensitivity to Acidic Deposition Under Alternative Forest Damage Hypotheses

R. S. Turner
R. J. Olson
C. C. Brandt

Environmental Sciences Division
Publication No. 2720

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ENVIRONMENTAL SCIENCES DIVISION

AREAS HAVING SOIL CHARACTERISTICS THAT MAY INDICATE SENSITIVITY TO ACIDIC DEPOSITION UNDER ALTERNATIVE FOREST DAMAGE HYPOTHESES

by

R. S. Turner
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Environmental Sciences Division
Publication No. 2720

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ABSTRACT

TURNER, R. S., R. J. OLSON, and C. C. BRANDT. 1986. Areas having soil characteristics that may indicate sensitivity to acidic deposition under alternative forest damage hypotheses. ORNL/TM-9917. Oak Ridge National Laboratory, Oak Ridge, Tennessee. 138 pp.

Recent evidence of forest decline and dieback has revived interest in mapping soil characteristics that may be useful in predicting areas where further forest damage could occur. Hypothesized causes of forest decline include direct effects of air pollutants on aboveground tree tissues and indirect effects on trees via belowground, soil-mediated processes. This report (1) reviews the potential for forest soil changes caused by acidic deposition and the role of these changes in damage to trees and (2) uses recently available soils data and computer display techniques to map the areal extent of soil characteristics that may indicate where such changes could occur.

Hypotheses on causes of forest decline and dieback mediated by soil changes caused by acidic deposition generally fall into two categories: (1) reduction in available base cation plant nutrients below a level required to support the standing biomass and (2) increased availability of substances toxic to plants or other soil organisms. Chemical and physical characteristics of soils were selected as mapping criteria based on a review of experimental results and conceptual and mathematical models of biogeochemical processes occurring in forest soils. For the nutrient-leaching hypothesis, soils having a cation exchange capacity of <15 cmol(+) kg^{-1}, base saturation of 20 to 60%, and a pH >4.5 were selected. For the

toxicity hypothesis, soils having a base saturation <20%, a pH <4.5, and organic carbon <5% were selected. Variations in these criteria were tested, and restrictions on weathering rates and rooting depth were also added. Soils data available in several national and regional data bases were linked with each other and with the selected criteria, to allow extrapolation from the site-specific and theoretical studies to maps of potentially sensitive soils at three different scales: (1) the eastern United States, (2) the northeastern United States, and (3) the Adirondack region of New York. Using the criteria selected and the data available for mapping, from 3 to 40% of the forested soils of the eastern United States may be sensitive to changes that could result in reduced forest productivity (within an undefined time frame). To predict the time until a certain magnitude of soil change will occur, a dynamic model and additional data are needed to relate deposition rates, weathering rates, forest cycling and accumulation rates, changes in anion mobility, and climate and hydrologic processes (leaching rates) to present soil characteristics.

The data base and computer analysis and display techniques developed here can facilitate future regional-scale ecosystem modeling and assessment of resources available or at risk. Additional process-level studies on the causes of forest decline and dieback and additional regional data on critical soil processes are needed before a more certain and more quantitative assessment of forest resources at risk to acidic deposition can be made.

1. INTRODUCTION

This reanalysis of a topic previously addressed by Norton (1980), McFee (1980), Klopatek et al. (1980), and Olson et al. (1982) was prompted by a variety of new developments. There is new evidence of forest decline in this country and central Europe. Public awareness of and concern for effects of acidic deposition on terrestrial ecosystems have increased. There is a need to evaluate new scientific understanding of effects of acidic deposition on forests and soils. Finally, additional data on regional forest and soil characteristics have become available, allowing new maps, with increased resolution and better data, to be constructed. This project was undertaken as part of the National Acid Precipitation Assessment Program (NAPAP) 1985 Assessment of Forest Effects and this document reports the results of the linking of regional soils data, conceptual and mathematical models of acidic deposition-soil-plant interactions, and a geographic information system (GIS) for display of spatial patterns. Further development of these techniques is ongoing to allow more linkages for use in future assessments.

An international workshop on soil and water acidification in March 1984 brought together experts and ideas from the United States and Europe (Johnson et al. 1985a) and reached a general consensus on a conceptual framework of soil- and vegetation-mediated mechanisms of surface water acidification. Workshop participants agreed that many of the same mechanisms may be involved in forest response to acidic

deposition but also acknowledged that many more factors are involved and could not, at present, be sorted out into a general framework of forest response mechanisms.

An exchange of North American and German scientists in April and June 1984 facilitated comparisons of sites, hypotheses, and research results from areas of observed forest decline and dieback in the United States and Federal Republic of Germany. The exchange participants were unable to define a generalizable hypothesis that explained all that they saw but agreed that multiple factors were responsible, were perhaps different at each site, and included natural and anthropogenic stresses. They developed a list of hypotheses that could be tested at each site (Schütt and Cowling 1985; Johnson et al., in preparation).

While air pollution, including acidic deposition, is considered by many scientists in the United States and Europe to be a prime factor in causing forest damage, canopy dieback is a widespread phenomenon in forests of the central and south Pacific where there is no industrial pollution (Mueller-Dombois 1983). It is difficult to argue that air pollution could not play a role in forest decline, but it is unclear how air pollution is to be linked to dieback in regions with clean air. We need always to keep our minds open to possible alternative causes.

The purposes of this report are (1) to review and evaluate, in light of recent findings, the potential for forest soil changes caused by acidic deposition and the roles of these soil changes in damage to trees; (2) to map the areal extent of soil characteristics that may indicate where such changes might occur; and (3) to evaluate the

validity, usefulness, and means of improving the maps. Sensitive soils are defined here as soils (or soil solutions) which may undergo changes (caused by acidic deposition) that lead to a reduction in forest productivity. Because the existing knowledge of mechanisms of forest decline is so limited and the availability of suitable soil data on a regional scale is also limited, the maps that are presented in this report should be interpreted carefully. The maps show areas having soils with characteristics that are hypothesized to be sensitive to change. They do not necessarily translate to areas of potential forest damage (decrease in productivity) or show bounds to areas of potential damage. The maps do indicate areas in the eastern United States where further process-level research should be concentrated to test the various hypotheses or chain-of-damage scenarios. The accuracy of these maps must also be tested. Results of that research, along with the experience gained in generating these maps, will determine the model and data needs and the appropriate scale of analysis required to produce results that will be more useful in quantifying forest resources at risk to acidic deposition.

The geographic area covered by this project has slowly expanded westward. Areas in the eastern United States have been of greatest concern because of greater amounts of more acidic deposition than is experienced in most of the West. Also, there has been more clamor about forest effects attributed to acidic deposition in the East. The techniques used here can readily be extended to the rest of the country.

2. HYPOTHESES ON FOREST EFFECTS THAT MAY BE MEDIATED BY SOIL CHANGES CAUSED BY ACIDIC DEPOSITION

Numerous chain-of-damage scenarios, several of which include hypothesized forest soil changes as components, have been suggested to explain observed forest damage symptoms. While some researchers believe that the symptoms can all be explained through direct foliar effects without considering soil conditions, most scientists agree that many factors, including soil conditions, affect forest response to chronic and acute stresses that may be imposed on them. Hypotheses on causes of forest decline/dieback mediated by soil changes caused by acidic deposition generally fall into two categories: (1) changes in soil-plant nutrient relationships and (2) changes in the availability of substances toxic to plants or other soil organisms. These hypotheses are discussed in the following subsections.

2.1 CHANGES IN SOIL-PLANT NUTRIENT RELATIONSHIPS

One of the most commonly cited "potential" effects of acidic deposition on forest soils is the leaching of nutrient cations from the rooting zone, that is, acidification of soils by leaching of base cations (K^+, Ca^{2+}, Mg^{2+}, Na^+, etc.) and replacement on exchange sites by H^+ and aluminum ions (Morrison 1984; Wiklander 1973/74, 1975, 1980; Malmer 1974; Frink and Voigt 1976, 1977; McFee et al. 1976; Bache 1980 and others). Soil acidification or reduction in the percentage base saturation (BS) is a natural process occurring in most soils in humid regions. To assess the importance of acidic deposition in reducing forest productivity in different areas, two sets of

questions must be answered: (1) to what extent does acidic deposition increase the rate of acidification in different soils (i.e., which soils are more sensitive to increased acidification), and (2) does an increased rate of soil acidification have any effect on forest productivity, is there a threshold BS below which forest productivity drops, or do new forest species move in to replace ones that decline in importance with little net change in forest productivity? The question of whether a forest species composition shift to (perhaps) less valuable or desirable species then also could become an issue.

Numerous analyses have attempted to answer the first question, using simple calculations, models, soil columns, and field investigations. It is relatively easy to calculate for a hypothetical forest soil how long it would take for rainfall that averages pH 4.0 to acidify the soil a certain amount compared with rainfall that averages pH 5.6, assuming no other inputs, for example, weathering. McFee et al. (1976, 1977) performed this exercise, but pointed out that of course there are other inputs. Reuss (Reuss and Johnson 1985) and Cosby et al. (1985a,b) have developed more sophisticated models that take other inputs, sinks, and controlling factors into account. Such models predict equilibrium solution concentrations and changes over time of soil characteristics, but no model has yet been validated for regional use, and most important, none explicitly incorporates soil weathering inputs or uptake and redistribution of elements by vegetation.

It is much more difficult to test whether such calculated or modeled changes actually take place. Laboratory column studies have

shown that higher simulated-rain acidity will leach more base cations as well as more H^+ and aluminum. In addition to changes in column soil solution, short-term decreases in BS of the column soils from the beginning to the end of the acidification experiments have been reported (Bjor and Teigen 1980; Schier 1984; Huete and McColl 1984). The columns represent unnatural systems, however, because moisture, temperature, soil structure, biological activity, plant cycling, and weathering are altered. These characteristics are critical determinants of the distribution of ions on sorption sites and in soil solution. Also, acid loading to the columns is often much greater than ambient field conditions so as to produce results more quickly. Thus, while column studies show that increased leaching can occur under certain conditions, they are unlikely to be representative of field processes. Columns have not been used to quantify relative differences in leaching rates between different soils, although using intact cores under controlled, relatively long-term conditions might be a useful tool for assessment purposes.

Investigations of soil leaching in the field have also shown base cation leaching, at least in surface-soil horizons. Abrahamsen (1980) irrigated field lysimeters filled with "undisturbed soil monoliths" (40-cm depth and 29-cm diam) and found increased leaching of K^+, Ca^{2+}, and Mg^{2+} with water of increasing acidity. Stuanes (1980) found a reduction in BS of O and E horizons (30-cm depth) of a Typic Udipsamment in response to 4 years of irrigation with artificial rain of varying acidity. The decrease was much greater for Ca^{2+} and Mg^{2+} than it was for K^+. Haman (1977) leached three Finnish soils

with a range of acid solutions and reported that a fine sandy loam lost more K^+, Ca^{2+}, and Mg^{2+} than did heavier soils. Lee and Weber (1982) irrigated model deciduous ecosystem plots (Typic Haplumbrept soils) with rain of varying acidity and observed increased leaching of Ca^{2+}, Mg^{2+}, and, to a lesser extent, K^+ at 20-cm depth for all treatments after 3 years. At the end of the 3 years, even the pH 3 treatment showed no effect at 1-m depth. While it can be argued that the most extreme treatments used in these experiments are unrealistic, even the more moderate treatments showed significant leaching effects over the relatively short periods of observation.

Despite the fact that under some experimental conditions increased base cation leaching has been shown to occur, efforts to measure long-term changes in soil nutrients or pH have not shown conclusively that acidic-deposition-induced changes actually occur under field conditions. Wiklander (1973/74) found no evidence of pH changes in a podzol soil in Sweden over a 39-year period from 1934 to 1973. Troedsson (1980) reported results from a survey in Sweden showing a strong correlation between increasing age of the forest and decreasing pH in the A_0 horizon (humus layer). Results also showed that during the period 1961-63 to 1971-73 exchangeable Ca^{2+}, Mg^{2+}, and K^+ decreased significantly, while H^+ and Al^{3+} increased. Troedsson concluded that the effect of the coniferous trees was more important than the effect of atmospheric deposition. Linzon and Temple (1980) found no significant change in pH of podzolized sandy Ontario soils sampled in 1960 and again in 1978. Alban (1982) found a depletion of Ca^{2+} and a lowering of pH in subsoils and enrichment of Ca^{2+} and an

increase in pH in surface soils under aspen and white spruce over a 40-year period in Minnesota, apparently attributable to high rates of uptake and cycling of Ca^{2+}. Johnson et al. (in preparation) found a redistribution of calcium from subsoils to surface soils in acidic soils of oak-hickory stands in Tennessee from 1971 to 1982 due to Ca^{2+} cycling. Faust et al. (1984) found no significant change in pH, exchangeable bases, or heavy metals in a variety of New Jersey soils that were sampled in 1947 and again in 1983 from two geomorphic provinces. In 1984 Andersen and Johnson (1985) resampled organic and mineral horizons at 68 sites that had been sampled in the Adirondacks of New York by Heimberger in the period 1930 through 1932. They found a decrease in pH and Ca^{2+} from organic horizons that had a pH > 4 in the period 1930 through 1932. They could not discern the relative importance of leaching, biomass accumulation, or the effects of fire and logging in these changes. Wiklander (1973/74) suggested that acidic deposition would cause little or no pH change in calcareous soils with a high buffering capacity or in already acidic soils with pH < 5 (which includes most forest soils). He proposed that sensitive soils would be noncalcareous sand soils with pH > 5.

The studies previously mentioned and many others have pointed out gaps in our knowledge of ecosystem processes, critical to assessing the sensitivity of soils to leaching by acidic deposition. Cation leaching must be considered within the context of accompanying anion mobility. Cations can only move if balanced by an equivalent charge of anions. Soils in which nitrate is removed biologically and sulfate is adsorbed physically or transformed biologically will not lose greater than

natural amounts of nutrient cations (balanced by bicarbonate and organic anions; sulfate and nitrate are the predominant anions added by acidic deposition). Most forest soils are thought to be nitrogen deficient, and many soils have been shown to retain sulfate, but soils that become nitrate- or sulfate-saturated will leach cations readily. Research has not yet quantified the net change in nutrient leaching caused by a switchover from a carbonic/weak acid anion leaching system to a predominantly sulfate/nitrate anion system. Susceptibility of soils to nutrient loss due to acidic deposition thus depends on atmospheric anion inputs and on the anion retention properties of the forest soils in question. We know a great deal about anion mobility in a few places but lack good knowledge of the regional variability of these properties for most forest soils.

Numerous additional controls on base cation distribution in soils (other than replacement by percolating H^+) are also not well quantified. Weathering of minerals within the soil profile continually resupplies nutrients to exchange sites. Trees are continually accumulating, cycling, and redistributing nutrients within soils. Atmospheric inputs of dry and wet deposition continually add base cations in addition to acidic cations and anions. The relationships between acidic deposition and these processes are not well known even for intensively studied sites, and regional differences in the importance of these processes have not been quantified.

The question raised previously, whether increased nutrient leaching affects net forest productivity, has not been answered by research to date. It is generally thought that humid-area forests are

nitrogen limited, and, therefore, some cation leaching may not initially affect tree growth. Morrison (1984) reviewed the results of numerous field and greenhouse growth experiments using irrigation with artificial acid rain. Some experiments showed slight increases in growth; others showed slight decreases. The consensus interpretation of these results seems to be that acid irrigation may cause a short-term increase in growth due to a nitrogen fertilization effect, but also that long-term irrigation would result in depletion of base cations such that growth would eventually decline. None of the experiments was carried out long enough to test this interpretation.

Some relatively restricted areas of the United States are known to be potassium or magnesium deficient (see Stone and Kszystnisk 1977; Heiberg and White 1951). These areas probably would experience reduced forest productivity if further leaching of those elements occurred. However, when the availability of an element becomes limited in the soil, it is strongly held by the exchange complex and tightly cycled by the vegetation, and other more common elements are leached. Soils that have been highly leached naturally (e.g., Ultisols) may support a large biomass, though the rate of regeneration may be low compared with nutrient-rich sites if the biomass is removed.

It also has not been established whether increased nutrient leaching from shallow soils can be compensated for by greater tree uptake through deeper roots. Stone and McKittrick (1984) have shown that Pinus resinosa trees can obtain at least 16% of their total foliar potassium from depths below 110 cm in Entic Haplorthod soils in New York. A relatively small increase in uptake by deep roots could

make up for nutrients lost from the shallow rooting zone. Once taken up they would probably be strongly retained by a highly leached system, and more common ions such as aluminum would be released.

Although regional surveys of forest damage have generally shown little relationship to soil-nutrient status, data from some sites and fertilizer treatment plots indicate that trees show less damage on some higher nutrient status (magnesium and calcium) sites and may respond positively to added magnesium and calcium. In the Black Forest (Federal Republic of Germany), Zöttl and Mies (1983) have correlated "healthy" trees with soils formed on slate and "sick" trees with soils formed on granite. The slate soils have 2 to 3 times higher exchangeable magnesium and calcium than the granite-derived soils, and the leaves of the sick trees have substantially lower magnesium and calcium than the healthy trees. Bauch (1983) examined the fine roots of healthy and declining Norway spruce and silver fir and found that (1) in the declining firs, the root cortex did not contain calcium and magnesium and (2) in the declining spruce, the cortex did not contain calcium. In other areas of Germany, Zech and Popp (1983), Hüttermann and Ulrich (1984), and Kenk et al. (1985) have found less tree damage on fertilized plots, as well as a positive response to magnesium and calcium fertilization of trees that have not progressed to an advanced state of decline. Kenk et al. (1985) present evidence of an interaction of emissions and climatic factors, along with nutrition difficulties, to explain forest damage at their site in the Buntsandstein region of the Odenwald. Although they acknowledge that nutrition difficulties alone have never caused widespread decline and

dieback in forests of central Europe, they propose fertilization of some sites as a mitigative measure until emissions are reduced.

Fertilization of forests by nitrogen and sulfur in acidic deposition certainly occurs, but evidence for increased forest growth due to acidic deposition has not been clearly documented. Sulfur deficiency in eastern U.S. forest soils is very rare (Leaf 1968). Nitrogen deficiency, however, is common. Abrahamsen (1984) reviews the results of fertilizer experiments in Europe and North America that show a strong response to nitrogen fertilization in northern soils of both continents but less response in soils of central Europe and North America because of shortages of other nutrients in addition to nitrogen. Such studies suggest that atmospheric nitrogen deposition is less likely to influence growth in southern areas than in northern forests. Another hypothesis suggests that some forests are nitrogen-saturated, and atmospherically deposited nitrogen could lead to changes in cell volume, hormone balance, and root-to-shoot ratio. This could result in problems for the trees in coping with frost and drought or in obtaining adequate nutrients (Nihlgard 1985). Whether this effect occurs and how it relates to soil characteristics are unknown. Fertilization of forests by base cations in acidic deposition is not commonly considered, and base cation deposition is not well quantified because dry deposition amounts are unknown. However, atmospheric inputs of base cations may equal or exceed base cations leached from some nutrient-poor soils (Heinrichs and Mayer 1977; Turner 1983; Johnson et al. 1985).

To summarize, Tamm (1979) has suggested that forest stands adapt their biomass to the amounts of nutrients available on the site. To the extent that growth-limiting nutrients are leached from soils by acidic deposition and not replaced by weathering or atmospheric inputs, we would expect forest stands to decline in biomass or basal area or at least to be more sensitive to other stresses. Experiments have shown that under some conditions, leaching of nutrients from soils can be increased by acidic deposition and that leaching rates can differ with soils of different characteristics. However, rates of leaching, given different ambient atmospheric inputs and undisturbed soil characteristics, are poorly quantified to date.

Also poorly understood is the extent to which mineral weathering and biological cycling/redistribution of nutrients can resupply base cations to exchange sites. Increased leaching may be offset by increased resupply at some locations because increased weathering is expected to result from deposition of acidic substances. On the other hand, soils with restricted root zones could be sensitive because of the inability of the vegetation to cycle and redistribute nutrients weathered at depth in the profile up to the surface. We can probably predict the relative mix of cations that will leach from a soil by knowing percentage BS, exchangeable cations, cation selectivity coefficients, aluminum solubility, pH, and CO_2 partial pressure in the soil. However, soil-solution concentrations and the extent of increased leaching over time depend on anion mobility, which is a function of biological activity (production of CO_2 ---> HCO_3^-), nitrate and sulfate inputs to the system, and nitrate and sulfate

retention in the soil profile. The physical conditions and biological transformations that control anion mobility are imperfectly understood.

2.2 CHANGES IN AVAILABILITY OF TOXIC SUBSTANCES

Acidic forest soils may experience pulses of extremely acid soil solution in response to the release of sulfate and nitrate anions from the litter and/or soil. These releases are mediated by seasonal changes in soil moisture, temperature, and biological activity, and short-term climatic changes or events such as warm-dry followed by cool-wet periods, or storms (Hüttermann and Ulrich 1984; Turner et al. 1985). During these acidic pulses, toxic substances such as Al^{n+} and other trace metals that may be damaging to roots, mycorrhizae, decomposers, and other soil microbiota may be mobilized into solution. The releases of Al are due to the salt effect, described by Seip (1980) and Reuss (in Johnson et al. 1985), in which H^+ and Al^{3+} concentrations in soil solution increase in response to an increase in total mineral acid anion concentration. Weathering, biological reactions, and redistribution of elements may take place between pulses so that further acidification of the bulk soil does not actually occur. In this scenario, damaging effects may occur in acidic forest soils with no need to further acidify the soils. Hypotheses concerning the potential toxic effects of trace metals on roots and soil microbiological processes have been reviewed by Morrison (1984) and Tabatabai (1985).

It is generally accepted that high concentrations of free aluminum can damage tree roots, although how high that concentration must be is

disputed and is probably species specific. Studies of aluminum toxicity reviewed by Abrahamsen and Tveite (1983) have suggested that concentrations of 80 to 160 mg L^{-1} of aluminum in solution are required for damage to roots of forest species. Ulrich et al. (1980) concluded that aluminum concentrations of 1 to 2 mg L^{-1} in equilibrium soil solution could damage root systems of trees. Matzner and Ulrich (1981) found aluminum concentrations of up to 6 mg L^{-1} in leachate under beech forest and up to 15 mg L^{-1} under spruce forest. Ulrich (1983) discussed the importance of the interaction between aluminum and calcium in the soil and roots and concluded that aluminum injury is likely when the molecular ratio of calcium to aluminum in the equilibrium soil solution or in the roots is <1. Abrahamsen (1984) reported results of an experiment with Norway spruce showing aluminum toxicity when aluminum concentration in solution reached 20 mg L^{-1} or above with a calcium concentration of 3.5 mg L^{-1}.

Aluminum concentration in roots and foliage has not been shown to be clearly associated with visible symptoms of decline. Johnson and Siccama (1984) measured the aluminum content of red spruce roots and foliage and found no consistent difference in aluminum content between healthy and declining trees. Bauch (1983) reported that the aluminum concentration in the outer cortex of fine roots did not differ substantially between healthy and declining firs.

Although the aluminum hypothesis has been supported by research in the Harz Mountains, there may be limitations to its applicability. Organic and inorganic ligands may effectively complex aluminum in the

soil (David and Driscoll 1984; Turner et al. 1985; Tabatabai 1985) and thus render the aluminum unavailable or nontoxic to roots. The high-elevation spruce forests in the United States root primarily in organic soils rather than in mineral soils as is the case for the forests in Solling, Federal Republic of Germany. Considering the present evidence, aluminum mobilization and toxicity is probably not a significant mechanism in the North American forests currently showing damage. However, aluminum toxicity cannot be ruled out in a consideration of forests growing on highly weathered soils low in organic matter.

Toxic effects of other trace elements on trees in the field have not been documented except around point sources such as smelters. Changes in microbial activities in soils resulting from direct or indirect effects of acidic deposition have been reviewed by Tabatabai (1985), but no generalizations are possible on the direction of the effect (i.e., both increases and decreases in activity have been observed). Results obtained have been found to be dependent on experimental conditions, types of soils tested, and degree of acidification. Organic forest floors are effective accumulators of some heavy metals, and additional research should investigate possible direct and indirect effects of these accumulations on forest productivity. For the present, however, there are no documented mechanisms of regional heavy metal effects on forests and no reliable data on geographic distribution of metals in forests.

3. CRITERIA FOR MAPPING SOILS

3.1 PREVIOUS WORK

Claims about the pervasive threat of acidic deposition to large areas of the United States and Canada have led to numerous attempts to map areas that are sensitive to acidic deposition (Hendrey et al. 1980; Norton 1980; McFee 1980; Klopatek et al. 1980; Wang and Coote 1981; Olson et al. 1982; Hollowaychuk and Lindsey 1982; Wada et al. 1983; USEPA 1983; Lucas and Cowell 1984). The basis of these maps has ranged from single-factor considerations of bedrock sensitivity to multifactor analyses including bedrock, surficial deposits, physiography, soil depth, soil drainage, soil chemistry, vegetation type, and vegetation cover. Most maps have shown areas of high, moderate, and low sensitivity based on educated, but arbitrary, selection and weighting of sensitivity values of each factor included. These maps all show relative sensitivity of one region compared with another, dependent on the environmental factors considered, the relative weighting of each factor in the final map, the scale of the maps, and the accuracy of the data used.

The maps have, however, been criticized for their lack of differentiation between criteria for aquatic and terrestrial effects; the use of unnatural mapping units; their inability to show variability within mapping units; the surrogate and inaccurate data used; the arbitrary nature of the selection and weighting of sensitivity values used; and for the factors not considered, such as dry deposition inputs, mineral weathering, hydrologic pathways through soils, and

elemental cycling by vegetation. The maps were produced at different scales on different projections in nondigital form and are difficult to overlay with maps of other factors, such as deposition inputs, forest types, or water quality. There was no intent to delineate dose-response or threshold (target) loading relationships with most of these maps, but to be more useful for assessment purposes, maps must show areas with characteristics that can relate deposition inputs to level of change or damage in terrestrial or aquatic ecosystems. The maps produced to date have not been tested for accuracy, and they have led to the generation of very little, if any, new data suitable for such regional assessments. It is doubtful whether they have been used to identify sensitive areas in which to locate new process-level research. The lack of interest in the existing maps probably stems from the numerous problems and unknown uncertainty embodied in them and also from the fact that they show very few U.S. areas to be sensitive to acidic deposition.

3.2 RATIONALE

Given past experience with soil sensitivity maps, why is there a renewed interest in such maps? An assessment of forest resources at risk to acidic deposition ideally would quantify the volume (and value) of timber threatened by acidic deposition within a given time period, and accompanying maps would show where the timber is located. When the 1985 NAPAP Assessment was being planned it appeared that research in this country and Europe would elucidate mechanisms or hypotheses of forest decline that could be related quantitatively to newly available

survey data (USDA SCS 1984a) and more detailed soil chemical data for regional analysis than had been available previously (Holzhey 1985). However, the current hypotheses of forest decline include soil characteristics as only one of numerous contributing factors. The research has not progressed to the point of showing threshold values of any soil characteristics affecting forest decline. Rather, it suggests that areas with nutrient-poor soils may show damage earlier or more severely and that under conditions of acidic pulses in some soils aluminum toxicity may result.

The maps produced in this section are not maps of sensitivity to forest damage but show areas with soil characteristics that may contribute to forest decline. They need to be overlaid with maps of other contributing factors to show areas potentially at risk to damage.

3.3 EMPIRICAL APPROACH

An empirical approach might be used for selecting sensitivity criteria by identifying the soil characteristics of areas where forest decline is observed. However, damage occurs in forests on many soil types in the Federal Republic of Germany, although some studies indicate that it may be more severe on sites poor in magnesium and calcium. It is possible that a closer link between damage severity and soil characteristics could be shown if more soil chemical data were available from analyses showing effective nutrient availability, rather than the standard agricultural soil analyses. In this country, clearly documented damage is currently confined to high-elevation forests, for which very few soils data are available. Thus, the empirical approach

to selecting sensitivity criteria is not very useful, given present data limitations.

3.4 THEORETICAL APPROACH

Because cause-effect mechanisms and dose-response relationships for forest effects are not available, we consider hypothesized mechanisms and, necessarily, theoretical, untested sensitivity criteria. As concluded in Sect. 2, the hypothesized mechanisms of soil-mediated forest decline include (1) nutrient leaching leading to nutrient deficiency and (2) toxic effects of dissolved aluminum in acidic soils.

Table 1 lists potential sensitivity criteria for each hypothesized mechanism. Bounds on some of the criteria are not set more specifically than high, medium, or low because quantitative estimates of what they should be are not known. For forest effects these criteria have previously been applied primarily to surface soil horizons, because most tree roots occur in approximately the top 25 cm of soil. However, to the extent that deeper rooting is important to incremental nutrient and water uptake by trees, subsoil horizons must also be considered. This at least doubles the data requirements and the complexity of analysis and mapping but probably is necessary.

The nutrient-leaching criteria are essentially those of Wiklander (1973/74) plus the mobile anion concept (Nye and Greenland 1960; Johnson and Cole 1980) and have been used in most other soil sensitivity schemes. Other sensitivity schemes have not explicitly

Table 1. Theoretical soil sensitivity criteria and data availability for two hypothesized forest-damage mechanisms

Hypothesis and criteria	Data and data source[a] (Surrogate data in parentheses)
Increased nutrient leaching leading to nutrient deficiency	
Moderate base saturation (BS), 20% < BS < 60%	BS for soil taxonomic subgroups - Pedon Data Base
Low-moderate cation exchange capacity, (CEC), CEC < 15 cmol(+)/kg	CEC for subgroups - Pedon Data Base
Moderate pH, pH > 4.5	pH for subgroups - Pedon Data Base
Low atmospheric replacement	NADP data for base cations - not available
Low weathering replacement in soil or from depth in soil profile by biological cycling	(Nonreactive bedrock - not available, taxonomic class mineralogy - Soils 5 File, forest type - NRI)
Anion mobility	
Low SO_4^{2-} retention	(Spodosols, others?)
High NO_3^- availability	(Forest type, stand age? - NRI)
Good soil drainage	Soil drainage class - Soils 5
Current nutrient deficiency	
Low exchangeable bases (Mg, Ca)	Exchangeable bases for subgroups - Pedon Data Base, (site index, forest type - NRI)
Aluminum toxicity	
Low BS, BS < 20%	BS for subgroups - Pedon Data Base
Low pH, pH < 4.5	pH for subgroups - Pedon Data Base
High Al solubility	(Not available)
Low cation selectivity coefficient	(Not available)
Very high anion concentrations	(Very high acidic deposition)
Seasonal/event pulses	(Extremely well-drained/droughty soils? - Soils 5 File, specific climatic/weather conditions? - NOAA)
Low Al complexing	
Low organic carbon, < 2%, 5%	Organic carbon for subgroups - Pedon Data Base
Low inorganic ligands	(Not available)

[a]See text, Sect. 4.

considered weathering replacement, redistribution of elements by vegetation, or atmospheric inputs of base cations. The criteria for current nutrient deficiency are simply low exchangeable bases and low replacement of bases. Magnesium and calcium are of primary concern, based on the field observations and column leaching experiments discussed in Sect. 2. Availability of nitrogen, phosphorus, or potassium has not been shown to be affected adversely by acidic deposition.

Combinations of soil chemical characteristics that may lead to aluminum toxicity in forests may be predicted using the equilibrium model of Reuss and Johnson (1985). Model inputs are BS, cation selectivity coefficient (KS), solubility of aluminum (KA), CO_2 partial pressure, and amount of mobile anions. Not all of these variables are available in existing soils data bases, but by using some worst-case estimates for the missing ones (i.e., KS = 0.1, KA = 9.0, CO_2 partial pressure = 3%, SO_4^{2-} = 500 meq L^{-1}, Cl^{-1} = 50 meq L^{-1}) the model results suggest that a soil must have a BS < 20% and pH < 4.6 before Al^{3+} in solution reaches 0.5 mg L^{-1}. The BS must drop below 15% before Al^{3+} reaches 1.0 mg L^{-1}, and below 10% before the calcium and aluminum molar ratio becomes < 1. Note that these are predicted worst-case results; model output for several New York and Maine soils sampled in the 1984 EPA Pilot Soil Survey (USEPA 1985) indicates that Al^{3+} concentration in solution would not exceed 0.26 mg L^{-1} and total monomeric aluminum in solution would not exceed 0.4 mg L^{-1} for those soils. Any predicted high Al^{3+} concentration would be toxic only if it were not complexed by organic or inorganic ligands.

4. DATA BASE AND ANALYSIS

4.1 AVAILABLE DATA

Table 1 shows the data and potential surrogate data available for each theoretical sensitivity criterion. Soils data for making regional assessments of sensitivity to acidic deposition are limited. Reliable regional data on atmospheric deposition, geology and weathering rates, and forest characteristics are not yet available in a uniform, validated, machine-readable format.

The 1982 SCS National Resources Inventory (NRI) (USDA SCS 1984a) contains limited data on soil physical characteristics for over 500,000 statistically selected points in the eastern United States. The data base also includes land use, cover type, forest mensurational data, an areal expansion factor, and a link to the SCS Soils-5 data base for each point. The NRI is potentially useful for a forest damage assessment but was originally designed for assessment of soil erosion and thus does not have many of the desired data. Its potential use is tied to the extent that it can be linked to a soil-chemistry data base. The NRI is very large and difficult to manipulate or to convert to aggregated map units other than the SCS Major Land Resource Areas (MLRAs). The individual points' locations are not identified.

The SCS Soils-5 data base (USDA SCS 1984b) contains taxonomic descriptions, data on physical characteristics and limited soil chemical properties (ranges of pH and organic matter), plus interpretive data on the suitability of over 22,000 soils for various uses. The lack of soil chemical data in the data base limits its

usefulness for acidic deposition assessment, although a large amount of general information on soil chemical characteristics can be interpreted from the soil taxonomic descriptions for each soil.

The SCS National Pedon Data Base (Holzhey 1985) contains chemical and physical data on over 12,000 pedons in the United States (Pedons are three-dimensional soil bodies usually about 1 m square on the ground surface.) This data base includes much of the type of chemical data needed [pH, exchangeable acidity, exchangeable aluminum, cation exchange capacity (CEC), exchangeable bases, BS, and others]. Unfortunately, many forest soils are not included. The analyses used are standard SCS procedures for agricultural soils with cation extractions at a buffered pH of 7, rather than at field pH, making the data difficult to interpret for acidic forest soils or to use in equilibrium models. The data base does not include information on sulfate adsorption capacity, although it may have other parameters that can serve as surrogates (e.g. Fe, Al oxides; these and other parameters need testing for their relationship to sulfate adsorption). Pedons in this data base were selected in a nonstatistical fashion for different specific purposes and were sampled at different seasons of the year from 1976 to 1984. It is uncertain what biases may be inherent in these data. The pedons with chemistry data are, however, reasonably well distributed geographically across major land resource areas (MLRAs), accounting for 25 to 50% of all soil series in MLRAs of the eastern United States.

Many states also have essentially independent soils data bases in diverse formats not easily machine readable and not yet tied to the

National Pedon Data Base. These include the SCS Soil Survey Investigations Reports (USDA SCS 1966-1981) and more recent SCS and state publications for numerous soils, but almost all suffer the same limitations as the National Pedon Data Base, in addition to not being readily accessible by computer. Data for the 600 pedons in the Pennsylvania Pedon Data Base (Cunningham 1985) and also for the 64 pedons in the 1984 EPA Pilot Soil Survey in New York and Maine (USEPA 1985) have been added to the SCS pedons in an integrated pedon data base being developed at ORNL.

The Geoecology Data Base (Olson et al. 1980) contains soil chemistry data (pH, CEC, BS, organic matter, and clay content) for typical pedons of most soil great groups of the eastern United States. These data were derived for the upper 20 to 25 cm of soil from the pedon descriptions in the SCS Soil Classification, Seventh Approximation (Soil Survey Staff 1960). These data also are based on pH 7 extractions, not field pH.

Acidic deposition data presented in Appendix A are from the Acid Deposition Data Network (ADDNET) data base maintained at ORNL for the NAPAP (Olson et al. in preparation).

4.2 ANALYTICAL APPROACH

The analysis involved (1) defining sensitivity criteria for each hypothesis in terms of available data as discussed previously, (2) applying the criteria to the data base to classify soil types and corresponding map units, (3) linking the classification results with the digital map files to display patterns of potentially sensitive

soils, and (4) generating statistical summaries of the extent of soils having the selected characteristics. This process was repeated at three levels of spatial resolution, as is discussed in the next section. The process required extensive efforts to develop the data resources and digital files for analysis and display, including (1) obtaining the various data files; (2) inputting and reformatting the data to allow merging of the files; (3) checking for data inconsistencies and errors using listings, exploratory statistical analyses, and thematic maps; (4) extracting data from thematic files, for example, wet deposition levels; and (5) compiling digital boundaries of soil map units. An attributes data base was created using SAS (1982) for storage, manipulation, and display, and digital map files were maintained, manipulated, and displayed using the ORNL GIS (Durfee et al., 1986). The sources of data, data base development procedures, and data base documentation are given in more detail in Brandt et al. (in preparation). That report describes both the attributes data base and the geographic files and processing. The attributes data will be made available through the ADDNET project (Olson et al., in preparation) as a set of SAS data sets.

5. MAPS AND INTERPRETATIONS

5.1 MAPPING UNITS

Soil characteristics are mapped at three different levels of detail on three different base maps in this section. Most of these maps are multifactor maps showing areas with different combinations of soil chemical and physical characteristics. Appendix A shows wet acidic deposition for the same areas, and Appendix B shows single-factor maps of numerous soil chemical and physical characteristics. Appendix C lists the soil subgroups and percentage of various forest types in different regions of the eastern United States that meet the selected soils criteria.

The maps of the eastern United States use MLRAs as the basic mapping units. MLRAs are natural regions defined by areas with similar climate, physiography, soils, water resources, potential natural vegetation, and land use (USDA 1981). Soil characteristics were aggregated for the MLRAs by calculating and mapping the percentage of total forested soil area that has soil characteristics within certain ranges (by linking NRI data points to the Soils-5 and Pedon data bases). For example, Fig. 2 shows 5 gradations, the lightest being MLRAs with < 5% of their area having soil characteristics that may lead to a reduction in nutrient status due to leaching by acid deposition, the darkest being MLRAs with > 50% of their area having such characteristics. This procedure allows us to identify MLRAs that have even relatively small areas of potentially sensitive soils.

The maps of the northeastern United States use the 1:2,500,000 scale General Soil Map of the Northeastern United States (Smith 1984) as a base. The mapping units show areas with similar soils, relief, and drainage. Each map unit consists of up to three soil great groups which are listed with their percentage of the area of each unit on the General Soil Map. Great groups are taxonomic subsets of soil orders and suborders, based on characteristics of upper pedogenic horizons. Soils within each great group have in common similar soil moisture and temperature regimes, base status, and presence or absence of diagnostic layers, such as fragipans. These maps also display percentages of each map unit that have certain forest soil characteristics, based on mean characteristics of the soil subgroups that make up the great groups.

The maps of the Adirondack Mountain region in New York use soil series or associations by slope phases as the mapping units. These maps were digitized by the Adirondack Park Agency from county maps produced by the SCS at a scale of 1:62,500 for each of the counties in the Adirondack Park (Ray Curran, personal communication). For map units with more than one series, the approximate percentage area of each series within the unit is known. We have mapped the percentage of area within each map unit having certain forest soil characteristics for these maps as well.

Figure 1 displays the percentage of forested soils in the MLRAs of the eastern United States. This map should be used in conjunction with the other maps that follow to better understand what percentage of the total area in each MLRA is actually forested. The potential for

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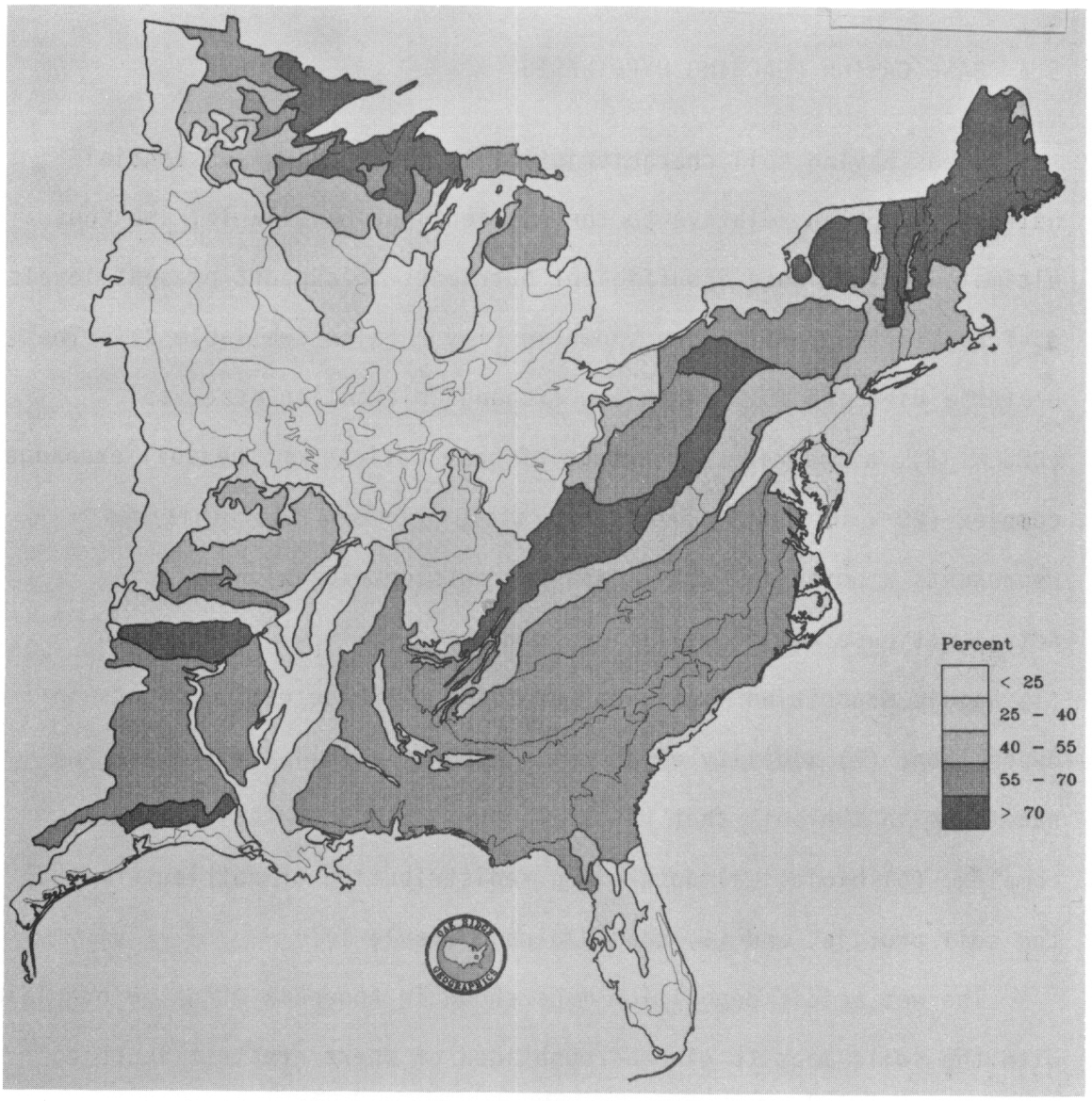

Fig. 1. Percentage of area in MLRAs of the eastern United States that is forested.

changes in forested soils is of most concern because forested soils generally are not limed, fertilized, or managed like agricultural soils and urban/suburban lands.

5.2 BASE CATION LEACHING HYPOTHESIS

Areas having soil characteristics that may allow substantial nutrient leaching relative to the nutrient pool available, and thus ultimately experience insufficient nutrients to support present levels of forest productivity, are shown in Fig. 2 (also see Table 2). The criteria used are soils with low exchangeable cation reserves (CEC < 15), a moderate percentage of base cations on the soil exchange complex (20 < BS < 60), and a moderate pH (pH > 4.5). This map represents a potential worst case for nutrient-leaching effects. Actual net base cation depletion from these soils will depend on (1) acidic deposition inputs to the soil, (2) base cation inputs in deposition, (3) mobility of anions through the soil, (4) weathering reactions in the soil that resupply base cations to the exchange complex, (5) biological uptake and redistribution of nutrients within the soil profile, and (6) soil drainage (Table 1).

The wet acidic deposition maps shown in Appendix A can be overlaid with the soils maps to give a rough idea of where greatest leaching could occur if sulfate and nitrate anions were mobile in the soil. A simple worst-case calculation (see Appendix D) suggests that for a soil with CEC = 10 cmol (+)/kg and 50% BS, 30 years of the highest deposition rates would be required to reduce BS by 20%, to 40% BS; 80 years of worst case leaching would reduce BS to 25%. It is unknown

what magnitude of decline in available nutrients would affect forest productivity. Spatial quantification of base cation deposition is not yet available for the eastern United States.

Anion mobility through the soil determines whether cations can be leached, because a charge balance must be maintained in the soil solution. Sulfate is the predominant anion in acidic deposition. Most nonglaciated soils adsorb sulfate until their adsorption capacity is reached; most glaciated soils have very low sulfate adsorption capacities. Therefore, most central and southern U.S. soils will have relatively lower cation leaching rates until the sulfate adsorption capacity is filled up, whereas sulfate leaches readily through many northeastern and upper midwestern soils. Data on sulfate adsorption capacity in soils are limited, but studies suggest that sulfate mobility in southern soils is increasing (Smith and Alexander 1983; Waide and Swank 1985; Johnson et al. 1984; B. J. Cosby, personal communication). The nitrate anion is also present in acidic deposition and may be generated within forest soils as well. Nitrate is highly mobile through soils. However, most forests are nitrate limited, and nitrate is readily removed from precipitation and soil solution by foliage and roots. There is some evidence that high-elevation, mature forests may leak nitrate; several field studies are investigating under what conditions this may occur. At present, it is not possible to reliably relate sulfate or nitrate mobility to other available data. Thus, the most we can say is that sulfate mobility is generally lower in nonglaciated soils than in glaciated soils, and base cation leaching

would be lower in central and southern U.S. soils, all other factors being the same, until the sulfate retention capacity was used up.

Weathering of primary and secondary minerals in the soil releases base cations to the soil exchange complex. The rate of weathering and the chemical content of the minerals present determine whether or over what period of time a net reduction in base cations or of a particular nutrient cation occurs on the soil exchangers. Weathering reactions may resupply base cations to the soil exchange complex as rapidly as acidic deposition leaches them. We currently have very few measures of weathering rates in soils, especially over the large areas of the eastern United States. Norton (1980) has mapped potentially sensitive areas, based on bedrock weatherability interpreted from state geology maps. These maps, however, are not digitized, nor are they uniform on a regional scale; thus, they are not very useful in this context. Soil taxonomic family names include a mineralogy class, telling the dominant mineral in the diagnostic B horizon. Figure 3 shows soils that have the same chemical characteristics as those in Fig. 2, and that also have family mineralogy classes indicating that a great deal of weathering has already occurred in those soils. These are soils in which substantial leaching could occur without rapid replacement by weathering within the soil profile, resulting in large net decreases in BS. Based on these criteria, 18% of eastern U.S. soils, mostly in the southeastern and mid-Atlantic piedmont and coastal plain, would be sensitive to net nutrient reduction. Research is ongoing to determine what factors can be used to predict weathering and leaching rates on a

regional scale. This would facilitate assessment of where weathering can, in fact, keep pace with nutrient leaching.

Biological uptake of nutrients from the soil can also have a significant acidifying effect on soils. These processes occur naturally over the life of a forest stand as it matures, because trees preferentially remove more basic cations than anions, maintaining charge balance by releasing H^+ ions to the soil from the root. Over a long period of time the stand would come into a state of equilibrium, with base cations that were removed being replaced by the decomposition of dead leaves and wood. However, many eastern forests today are growing on soils that were once farmed (often plowed and limed) or disturbed and eroded to the subsoil, and they are reacidifying the soil to its equilibrium state. Also, many forests are harvested, so that the nutrient cycle is never completed; nutrients are removed from the system before they can decompose and resupply the exchange complex. Soil acidification is thus exacerbated. We do not at present have a way of measuring soil acidification due to forest uptake on a regional scale. Numerous site-specific studies have been carried out, and more are under way. In the future, results of these studies may be linked to data on forest type, stand age, management practices, and land use history, to estimate the magnitude and relative importance of regional rates of acidification caused by forest uptake.

As mentioned previously, forests cycle nutrients from the soil to forest biomass and back to the soil, withdrawing nutrients from some depth in the soil profile and redepositing them on the surface. In some forest types this cycle is very tight, involving only the upper

25 cm or so of the soil, with very little leakage of nutrients below that depth. In other forest types, roots extend to greater depths in the soil, where they are able to take up nutrients that may have leached out of the upper soil horizon or to take up nutrients weathered from the relatively greater abundance of weatherable minerals at depth. Of the soils that are subject to nutrient leaching by acidic deposition, the shallower ones may ultimately be more sensitive, because plants are unable to extend their roots to greater depths to retrieve or replace the nutrients that are leached. Figure 4 shows soils that have the same chemical characteristics as those in Fig. 2, and that have a restrictive root layer [as defined by USDA SCS (1983) based on soil texture and bulk density] or depth to bedrock < 50 cm. These are soils in which substantial leaching could occur without replacement through recycling from depth by vegetation. Approximately 13% of the forested area of the eastern United States meets these criteria.

If water does not move through the soil, either downward through the profile or laterally downslope, net removal of nutrients from the soil probably will not occur, even with the most acidic deposition. Likewise, soils in low-lying areas may actually experience a net nutrient gain if water flows to those soils from upslope positions. When we delete soils with aquic moisture regimes from areas having the soil chemical criteria for nutrient leaching, the percentage of potentially sensitive forest soils drops from 41 to 35% (Table 2). This reduction occurs in all areas of the eastern United States but is

Table 2. Percentage of forested area in the eastern United States that meets selected sensitivity criteria[a]

Hypothesis and selection criteria	Percentage of Forested areas	Figure
Base cation leaching[b]		
A/E horizon	41	2,6,7
A/E & B horizon mean	55	
Base cation leaching except pH > 5 (A/E horizon)[c]	36	5
Base cation leaching except cation exchange capacity (CEC) < 10 (A/E horizon)[d]	30	
Base cation leaching and low weathering (A/E horizon)[e]	18	3
Base cation leaching and restrictive root layer or shallow bedrock (A/E horizon)[f]	13	4,9
Base cation leaching and good soil drainage[g]	35	6
Low exchangeable bases [< 3 cmol(+)/kg]		
A/E horizon	24	10,11,12
A/E & B horizon mean	20	
Low exchangeable bases [< 1 cmol(+)/kg]		
A/E horizon	6	B-9, B-10
A/E & B horizon mean	5	
Aluminum toxicity[h]		
A/E horizon	4	13,14,15
A/E & B horizon mean	1	
Aluminum toxicity except pH < 5 (A/E horizon)[i]	3	
Aluminum toxicity except organic carbon < 2% (A/E horizon)[j]	3	16
Aluminum toxicity except organic carbon < 10% (A/E horizon)[k]	5	

[a]See Appendix C for percentage of area by forest type, region of the United States, and soil subgroups meeting the selected criteria.

[b]CEC < 15 cmol (+)/kg, 20% < BS < 60%, pH > 4.5.

[c]CEC < 15 cmol (+)/kg, 20% < BS < 60%, pH > 5.

[d]CEC < 10 cmol (+)/kg, 20% < BS < 60%, pH > 4.5.

[e]CEC < 15 cmol (+)/kg, 20% < BS < 60%, pH > 4.5, mineralogy classes ferritic, gibbsitic, oxidic, siliceous, halloysitic, kaolinitic, or quartzipsamments.

[f]CEC < 15 cmol (+)/kg, 20% < BS < 60%, pH > 4.5, restrictive root layer or shallow bedrock < 50 cm.

[g]CEC < 15 cmol (+)/kg, 20% < BS < 60%, pH > 4.5, udic or dryer moisture regime.

[h]BS < 20%, pH < 4.5, organic carbon < 5%.

[i]BS < 20%, pH < 5, organic carbon < 2%.

[j]BS < 20%, pH < 4.5, organic carbon < 2%.

[k]BS < 20%, pH < 4.5, organic carbon < 10%.

particularly noticeable in soils of the southeastern coastal plain and Florida (Fig. 5).

Figures 2-6 and Table 2 indicate that there are significant areas that may be sensitive to acidification, reduction of base status or reduction in the pool of some nutrients available to forests. It is not known how long it will take for these changes to occur; how the changes might affect forest productivity; how the areas on the maps would differ if we varied some of the sensitivity criteria (because we have only hypothesized criteria, not field-tested thresholds); or what areas, if any, might have changed already.

Without knowing regional deposition rates of acids and bases, anion mobility in different soils, weathering rates, and vegetation cycling and removal rates, it is impossible to predict time to significant soil change with any certainty. We are currently gathering some of these data and searching for reasonable surrogates or estimates for the rest. We are also developing a simple predictive model that can be used with the data and that can be linked to a GIS, for spatial analysis and display.

The best soil-change model or display is of limited use to forest sensitivity/productivity assessments if we do not know forest damage thresholds or relationships between nutrient supply rate and forest productivity. These differ in a complex manner for different species growing under different conditions, and we may never be able to deterministically model forest response to acid deposition or other air pollutant stresses. Reviewers of this project have suggested that mapping sensitive soils is premature until we know what the appropriate

sensitivity or damage criteria really are. However, mapping hypothesized criteria is useful for exploratory analysis, estimating possible worst-case scenarios based on explicit hypothesized criteria, suggesting areas for field surveys and process research, and developing more sophisticated means of spatial and temporal analysis for future forest sensitivity/productivity assessment work. We must emphasize that the maps presented here are the results of ongoing analyses and do not necessarily represent forested areas sensitive to acid deposition. The maps represent the current state-of-the-art and state of science of regional data availability, data analysis based on hypothesized understanding of soil changes and soil-plant relationships, and data display.

Table 2 shows the results of changing some of the selection criteria. For example, increasing the pH threshold for the base cation leaching criteria to 5 decreases the potentially sensitive area by about 5%. This difference is significant in the southern and upper midwestern soils (compare Fig. 6, with Fig. 2), which are generally higher in pH than northeastern soils. Reducing the CEC from 15 to 10 cmol (+)/kg decreases the potentially sensitive area by about 10%, to 30% of the total forested area.

Figures 7 and 8 display the areas that meet the chemical criteria for base cation leaching for the northeastern states and for the Adirondack Mountain region of New York. Note the increasingly heterogeneous distribution of potentially sensitive mapping units (darker colors) as the level of detail in the maps gets finer. The northeastern United States map is based on soil chemical data

aggregated to the great group taxonomic level. It is not possible to relate soil series-specific mineralogy classes or soil depths to great groups, so we cannot reliably indicate soils in which nutrient leaching may be high and weathering rates low or soil depths shallow on the present map. Figure 9 shows areas in the Adirondacks where net leaching of base cations could occur without recycling by plants from below 50 cm in the soil profile. There are no soils in the Adirondacks with mineralogy classes that indicate that they are already highly weathered. Space limits the number of "what if" games that can be played here, but the data base and GIS are available to generate maps and summary statistics for new criteria as they are hypothesized or verified in the field.

Whether there are soils in the eastern United States that may already have acidified in response to acid deposition is difficult to determine using the available data. We can, however, display areas with soils that currently have low amounts of exchangeable bases (Figs. 10-12). These areas may currently have base cation deficiencies that are growth limiting to some species or that predispose forests to damage from other stresses. To date, forest decline that can be attributed to base cation deficiency has not been documented in the eastern United States, but the forest decline observed in some areas of the Adirondacks, the New Jersey Pine Barrens, and in some southeastern pines may be linked to low nutrient levels.

Table 2 shows the large difference in area between a threshold of 3 and a threshold of 1 cmol(+)/kg of exchangeable bases. The critical threshold will be species specific; as exchangeable bases are leached,

ORNL-DWG 85-10286

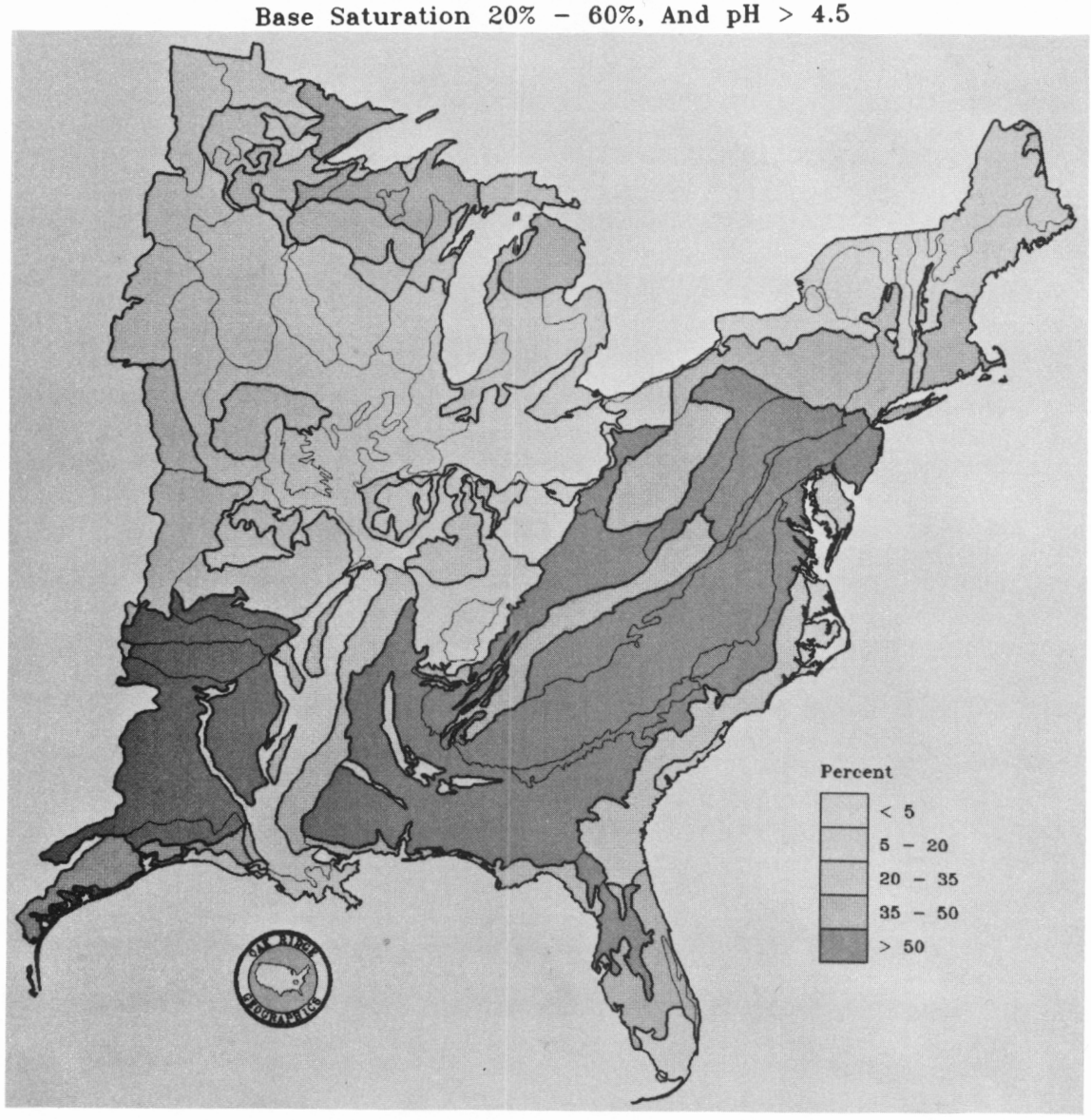

Fig. 2. Percentage of area in MLRAs of the eastern United States having forested surface soils with chemical characteristics that may lead to a reduction in base nutrient status because of leaching by acid deposition.

ORNL-DWG 86-6939

Percent Forested Soils with Cation Exchange Capacity < 15 cmol(+)/Kg, Base Saturation 20%–60%, pH > 4.5, and Low Weathering Potential

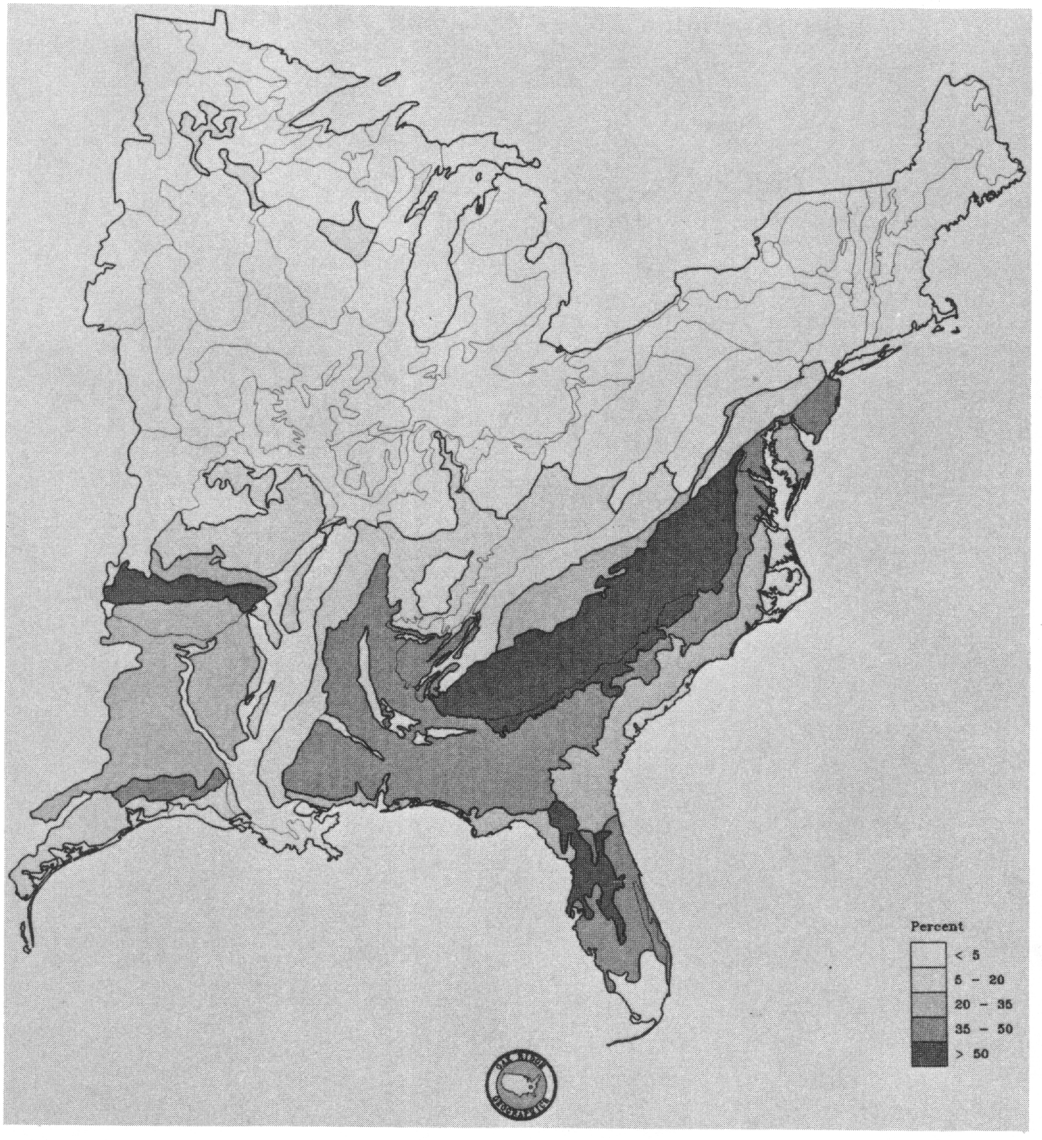

Fig. 3. Percentage of area in MLRAs of the eastern United States having (1) forested surface soils with chemical characteristics that may lead to a reduction in base nutrient status because of leaching by acid deposition and (2) low base cation replacement by weathering.

ORNL-DWG 86-6940

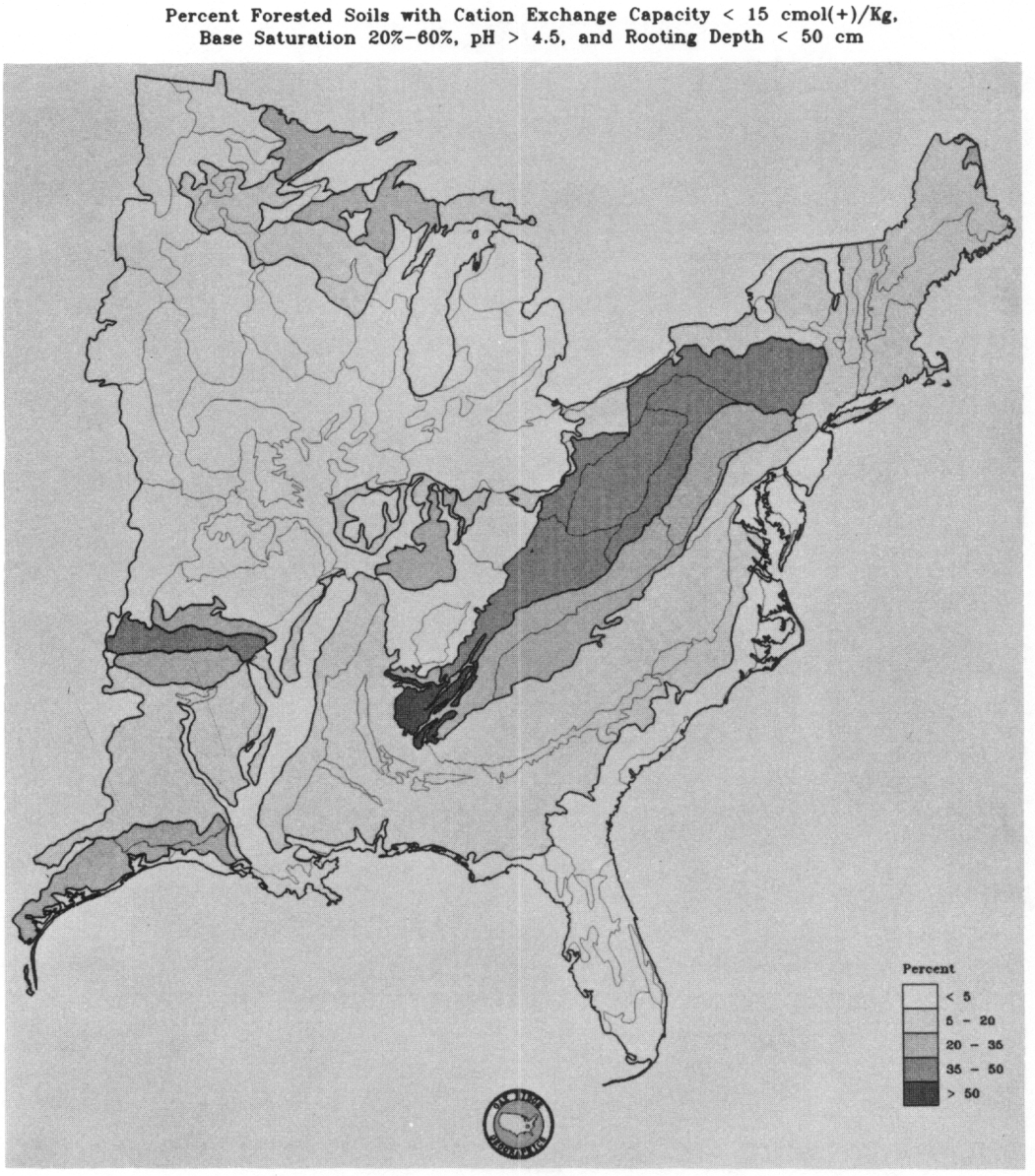

Fig. 4. Percentage of area in MLRAs of the eastern United States having (1) forested surface soils with chemical characteristics that may lead to a reduction in base nutrient status because of leaching by acid deposition and (2) shallow (< 50 cm) depth to a root-restrictive horizon or depth to bedrock resulting in low replacement of base cations by vegetation cycling from lower soil horizons to the surface.

ORNL-DWG 86-6948

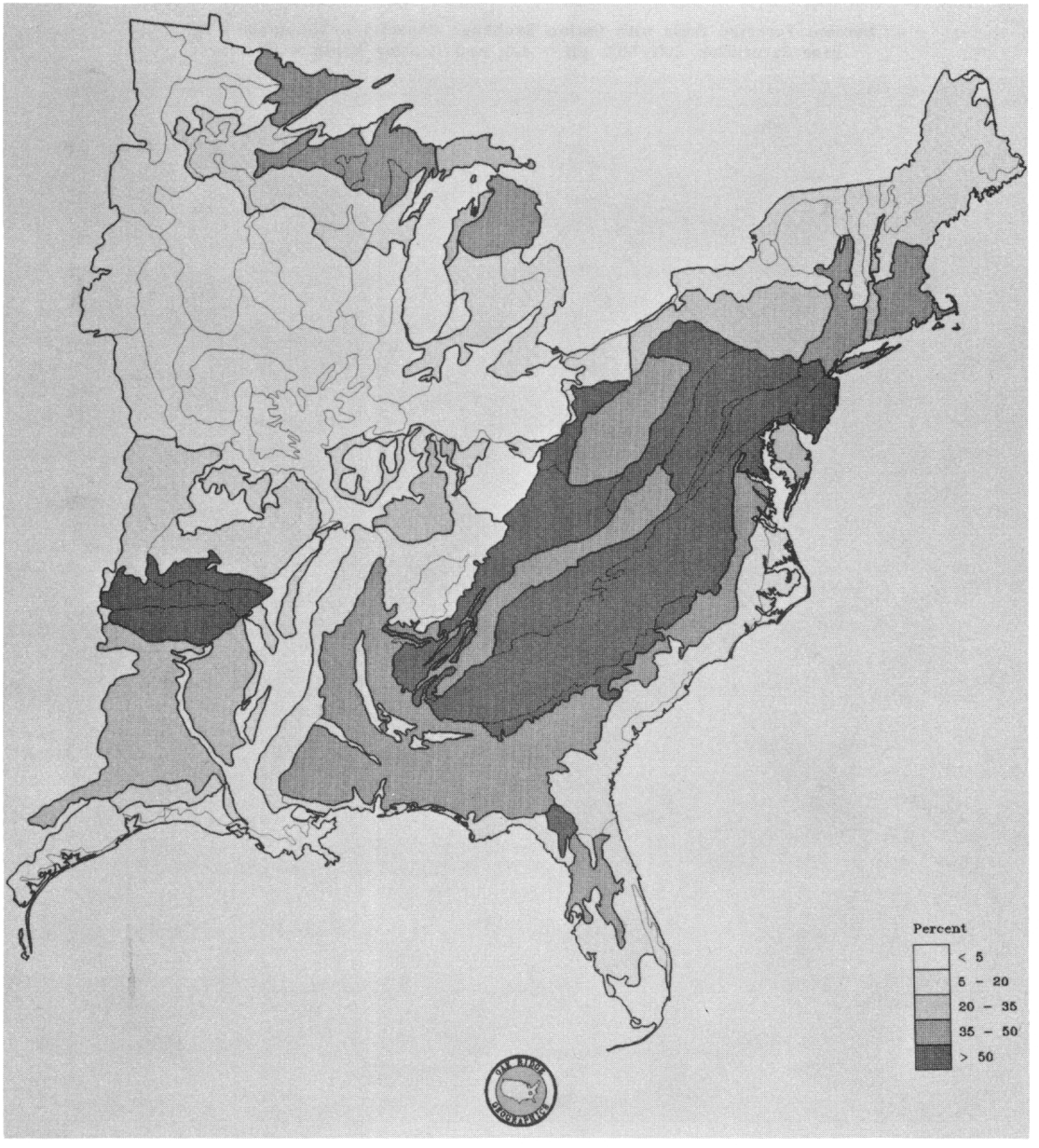

Fig. 5. Percentage of area in MLRAs of the eastern United States having (1) forested surface soils with chemical characteristics that may lead to a reduction in base nutrient status because of leaching by acid deposition, and (2) udic or dryer soil moisture regime.

ORNL-DWG 86-6941

Percent Forested Soils with Cation Exchange Capacity < 15 cmol(+)/Kg, Base Saturation 20%–60%, and pH > 5

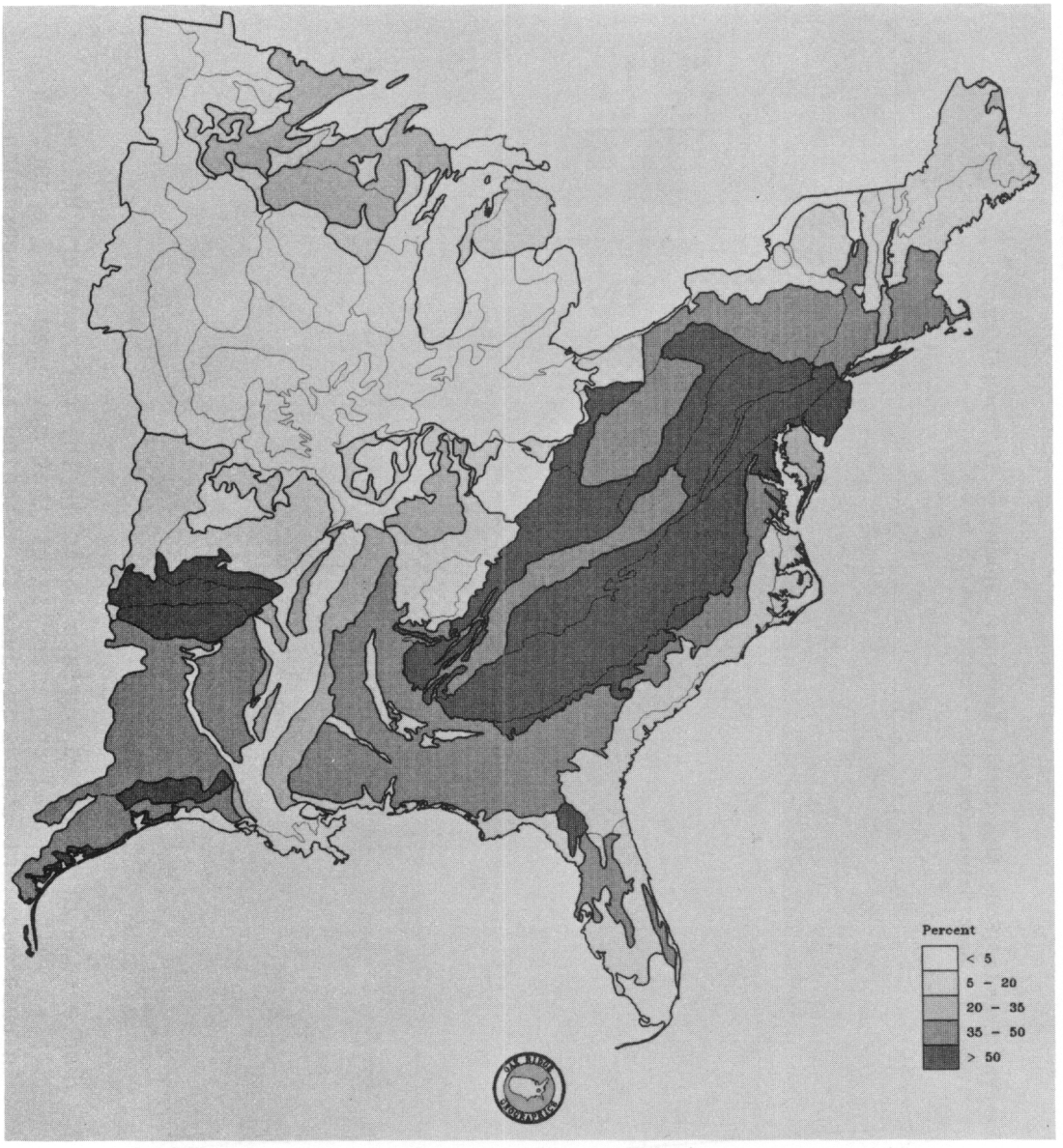

Fig. 6. Percentage of area in MLRAs of the eastern United States having forested surface soils with chemical characteristics that may lead to a reduction in base nutrient status because of leaching by acid deposition (pH threshold set at 5).

ORNL-DWG 85-10287

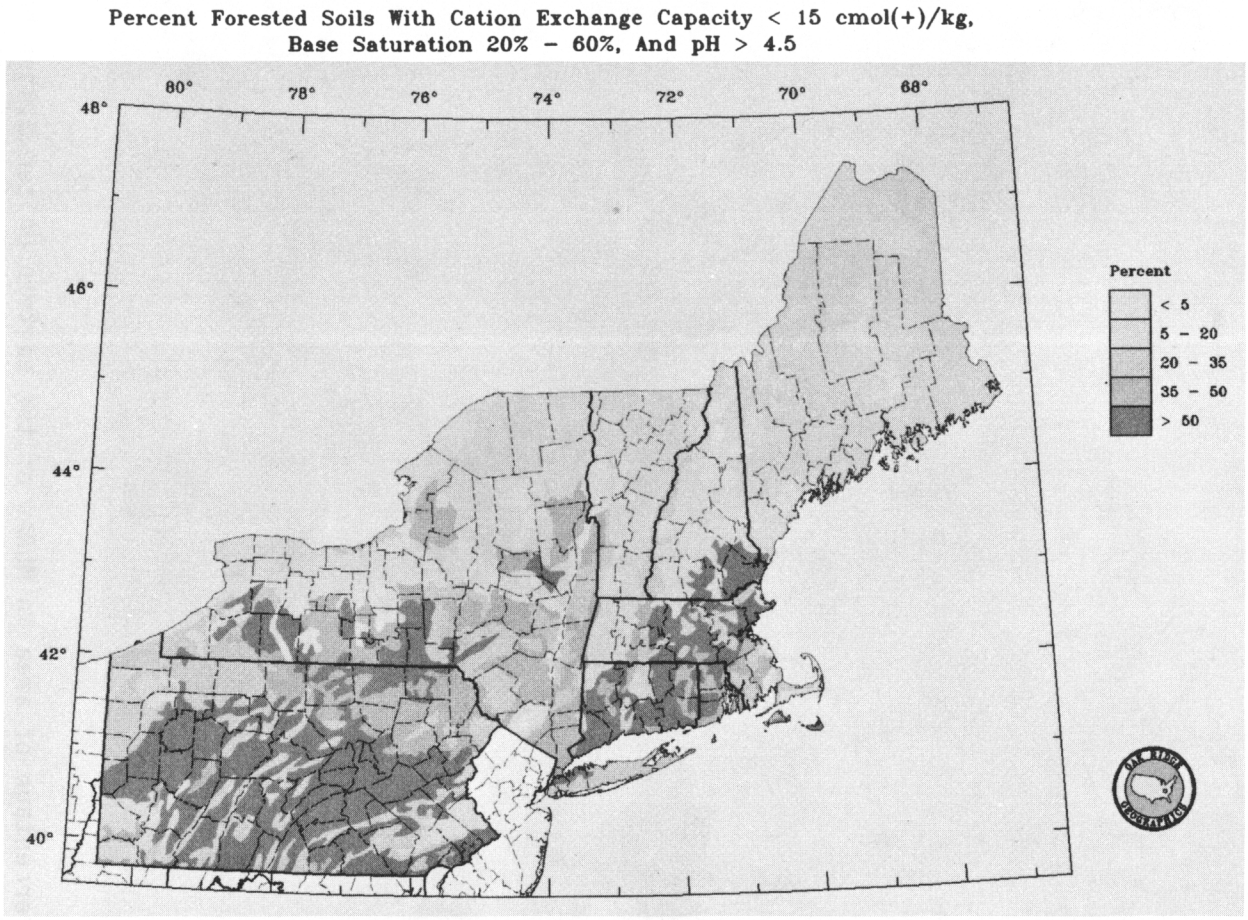

Fig. 7. Percentage of forested surface soils in the northeastern United States with chemical characteristics that may lead to a reduction in base nutrient status because of leaching by acid deposition.

ORNL-DWG 85-10288

Percent Forested Soils With Cation Exchange Capacity < 15 cmol(+)/kg, Base Saturation 20%–60%, and pH > 4.5

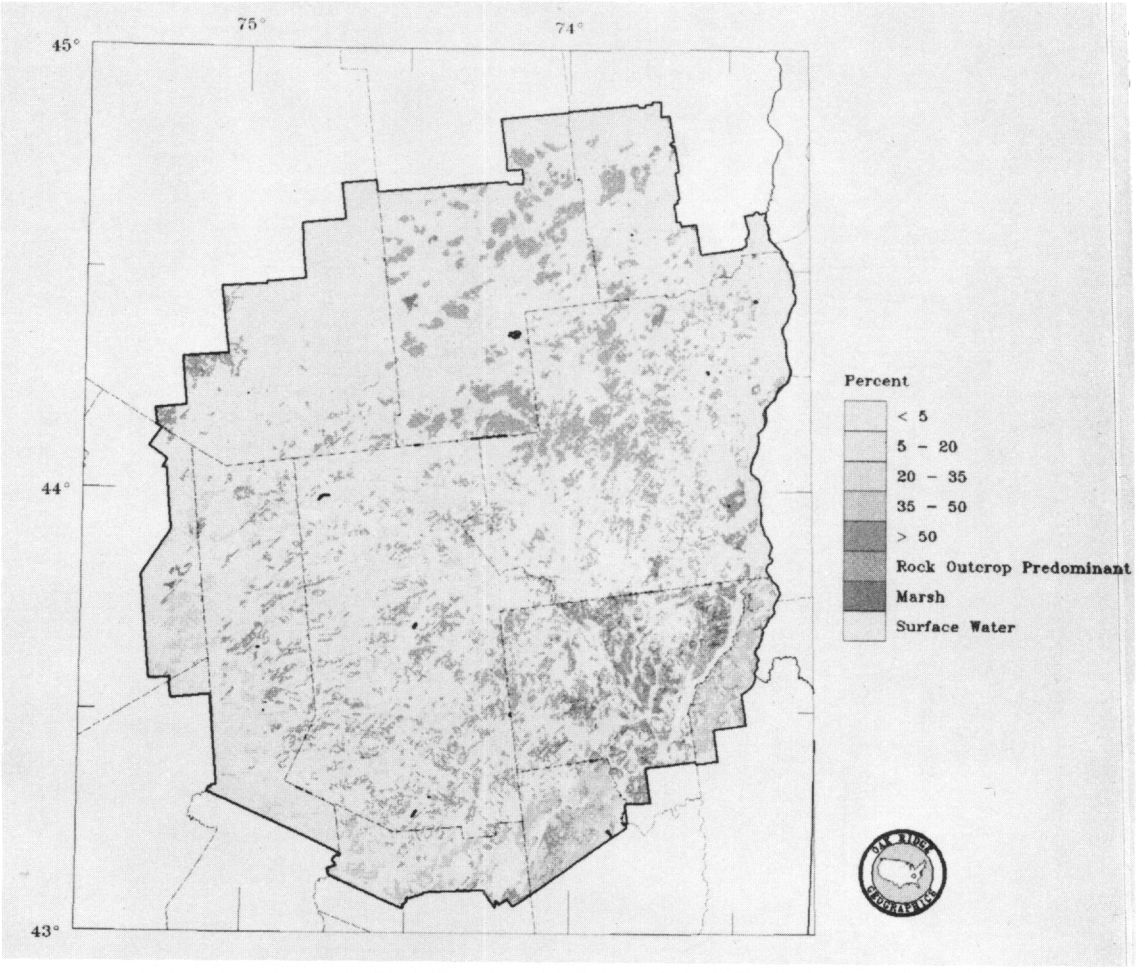

Fig. 8. Percentage of forested surface soils in the Adirondack region of New York having surface soils with chemical characteristics that may lead to a reduction in base nutrient status because of leaching by acid deposition.

ORNL-DWG 86-6942

Percent Forested Soils with Cation Exchange Capacity < 15 cmol(+)/Kg, Base Saturation 20%-60%, pH > 4.5, and Rooting Depth < 50 cm

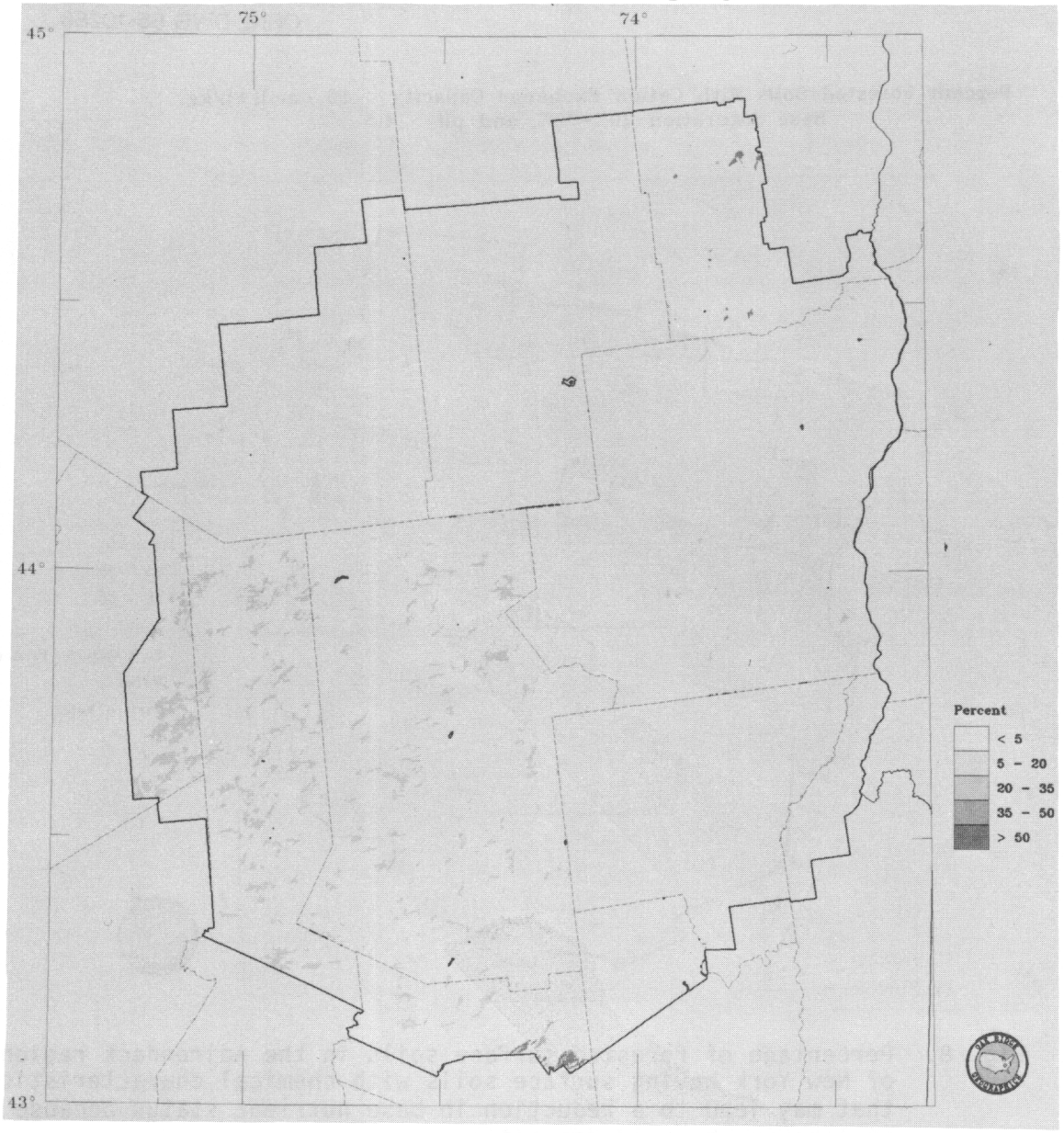

Fig. 9. Percentage of forested surface soils in the Adirondack region of New York with (1) chemical characteristics that may lead to a reduction in base nutrient status because of leaching by acid deposition and (2) shallow (< 50-cm) depth to a root restrictive horizon or depth to bedrock resulting in low replacement of base cations by vegetation cycling from lower soil horizons to the surface.

ORNL DWG 85-15735

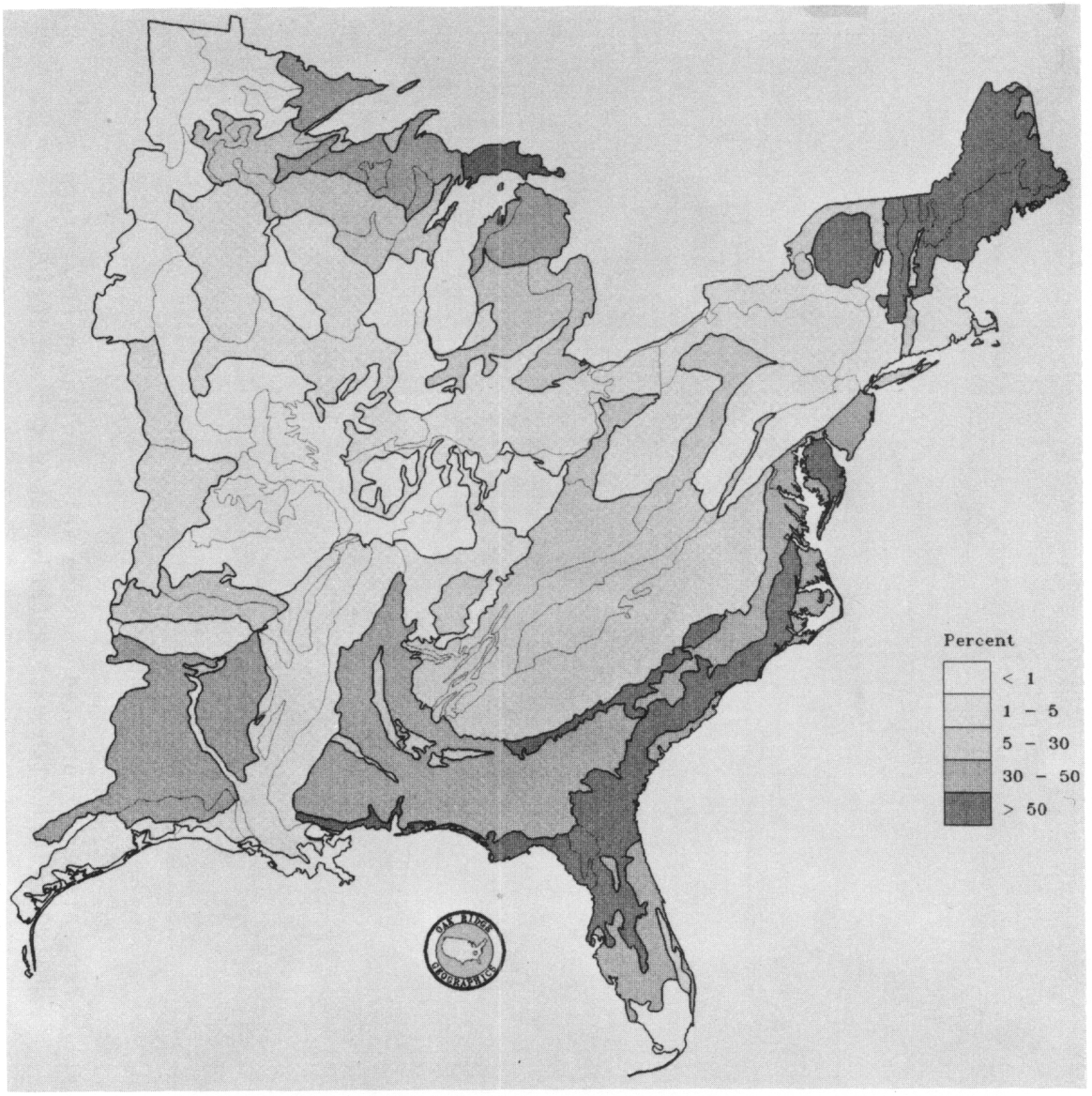

Fig. 10. Percentage of area in MLRAs of the eastern United States having forested surface soils with exchangeable bases < 3 cmol (+)/kg.

ORNL-DWG 86-6960

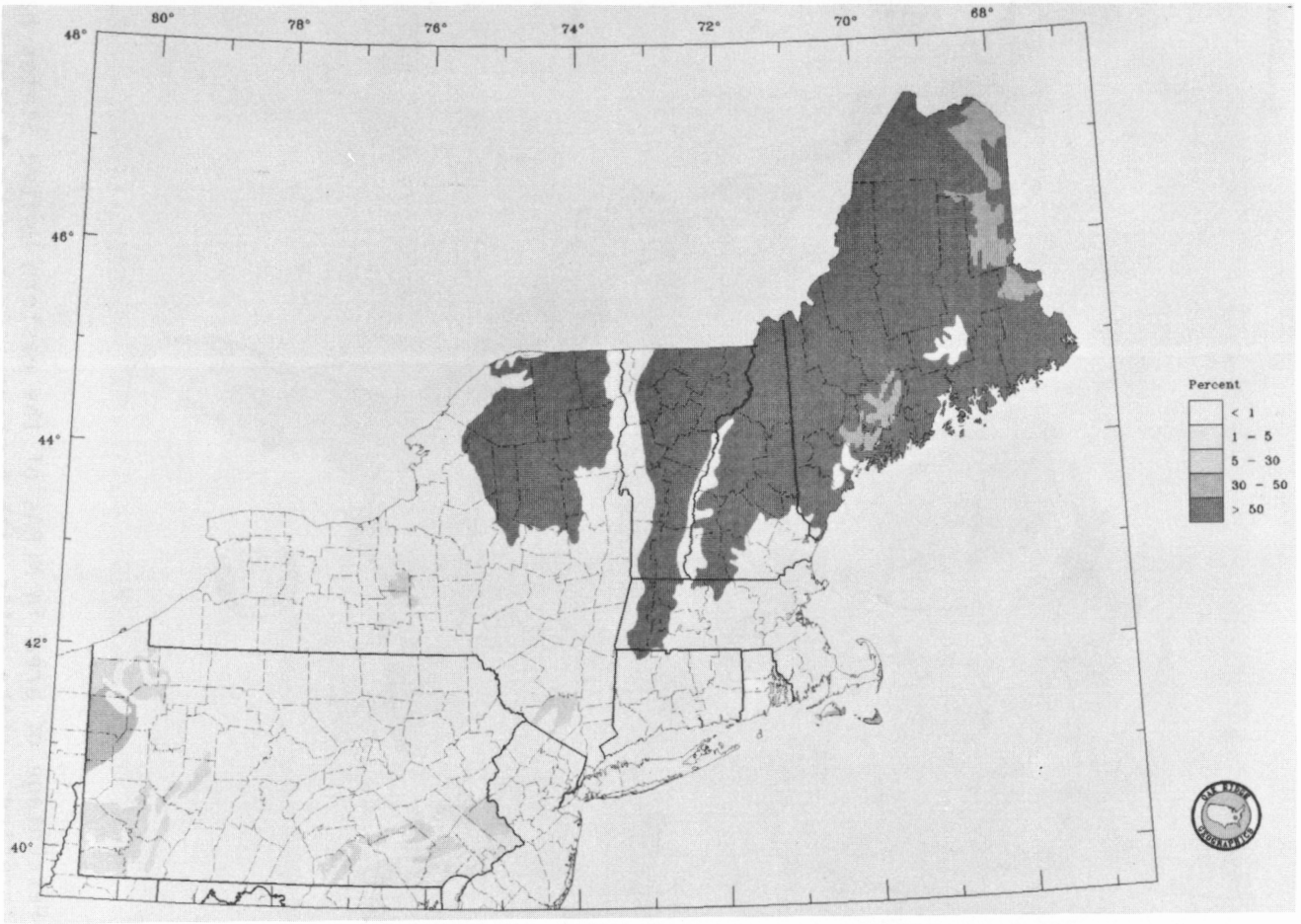

Fig. 11. Percentage of forested surface soils in the northeastern United States with exchangeable bases < 3 cmol(+)/kg.

ORNL-DWG 86-6943

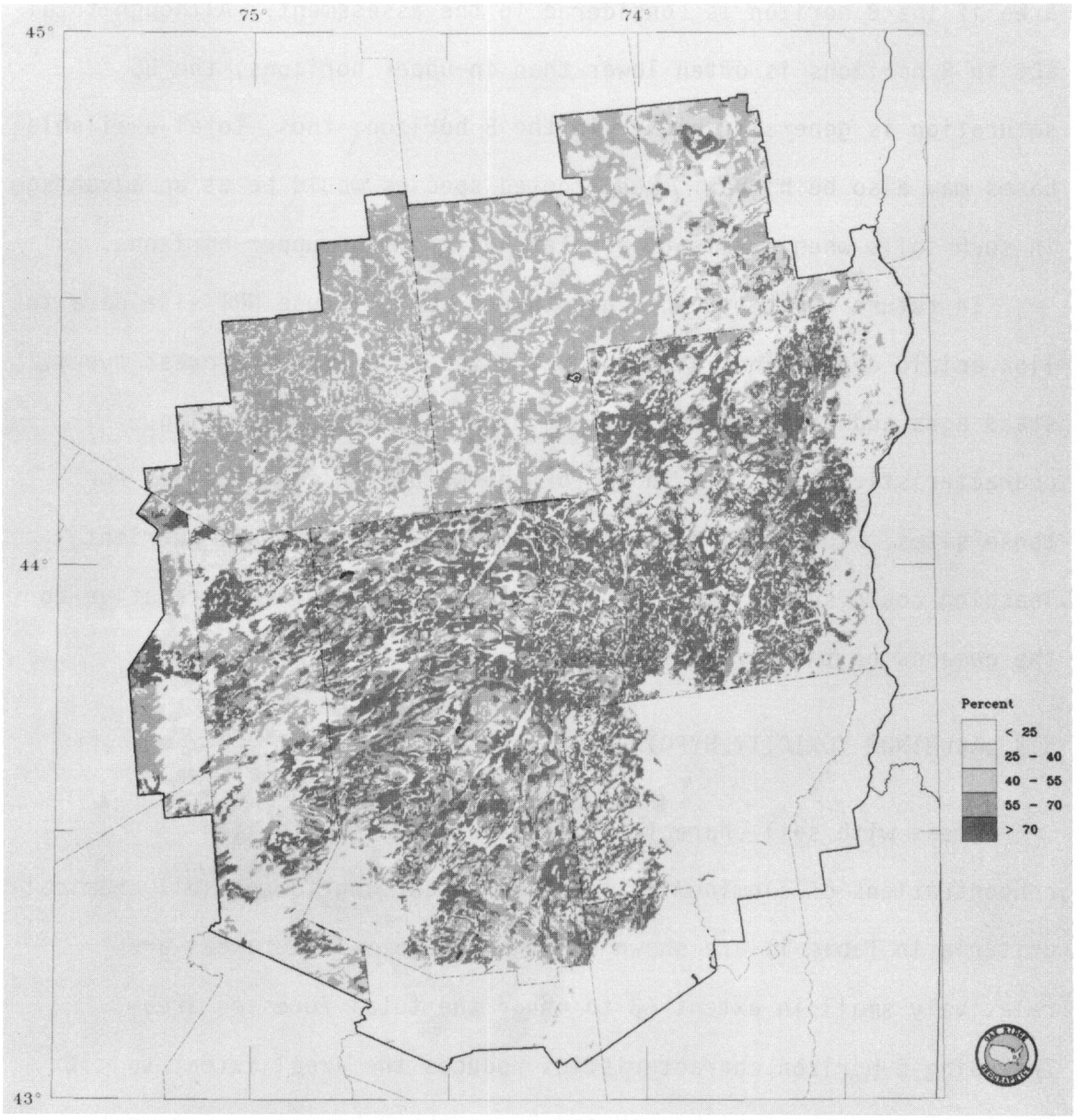

Fig. 12. Percentage of forested surface soils in the Adirondack region of New York with exchangeable bases < 3 cmol(+)/kg.

species requiring large soil-nutrient reserves will decline, and more tolerant species will invade. Table 2 also shows a lower percentage of area if the B horizon is considered in the assessment. Although total CEC in B horizons is often lower than in upper horizons, the BS saturation is generally higher in the B horizon; thus, total available bases may also be higher. Deep-rooted species would be at an advantage in such soils when nutrients are leached from the upper horizons.

In future work with this hypothesis, we will use NRI site data to link acidic deposition rates for the sites with present forest types, stand ages and basal areas, soil nutrient status and leaching characteristics, and site index for the Kuchler climax species for those sites. This will enable assessment of whether soil-nutrient leaching could substantially reduce base cation nutrients relative to the demands of the forest type of each site.

5.3 ALUMINUM TOXICITY HYPOTHESIS

Areas with soil characteristics that may lead to high concentrations of aluminum in solution (according to the soil chemical criteria in Table 1) are shown in Figs. 13-15. These areas are relatively small in extent (3 to 4% of the total forested area). Including B-horizon characteristics reduces the areal extent to <1% (Table 2) because of higher pH and BS in the B horizon. Somewhat larger areas would result if soil organic carbon were not so high in these forest soils, especially in the Northeast. Organic matter effectively complexes aluminum in soils and in soil solution, although it is uncertain how the percentage of organic carbon in the bulk soil

relates to dissolved organic carbon in solution. Figure 16 shows areas with potential aluminum toxicity problems if the soil organic carbon threshold is reduced to 2%. Such a reduction eliminates any potentially sensitive areas in the Northeast.

5.4 UNCERTAINTY IN THE ANALYSIS

Uncertainty in this analysis stems from many sources, including the validity of the hypothesized sensitivity criteria, accuracy of the soil chemistry data, and appropriateness of the map units selected. There are always inclusions of different soils within map units no matter what scale of mapping is selected. It is important to be able to quantify the variability of soil characteristics within map units, to compare that with the variability among map units, and to quantify the effect of that variability on the assessment being made.

Further process-level research is needed to validate and improve the sensitivity criteria. Threshold values of nutrient and toxin levels must be determined for species of concern before an analysis of soil characteristics can lead to a quantitative assessment of potential forest damage.

Soil chemistry can generally be determined to ± 10% precision. Systematic variability between soils laboratories in different states or regions is thought to be minimal but may occur (USEPA, 1985). Such systematic variability could bias regional comparisons of data but is at present unquantifiable.

Of much more concern is the uncertainty of the variability within soil types. Analysis of the variability within taxonomic subgroups and

of the variability of soil series within the subgroups is continuing in an effort to determine whether the subgroup is a useful level of aggregation for soil chemical characteristics or whether the series, or some other grouping of taxonomic classes, should be used. This analysis will aid in determining the most useful and economic level of aggregation, additional data needed, and the scale of analysis and display required in future assessments to be more useful in quantifying forest resources at risk to acidic deposition.

ORNL-DWG 85-10290

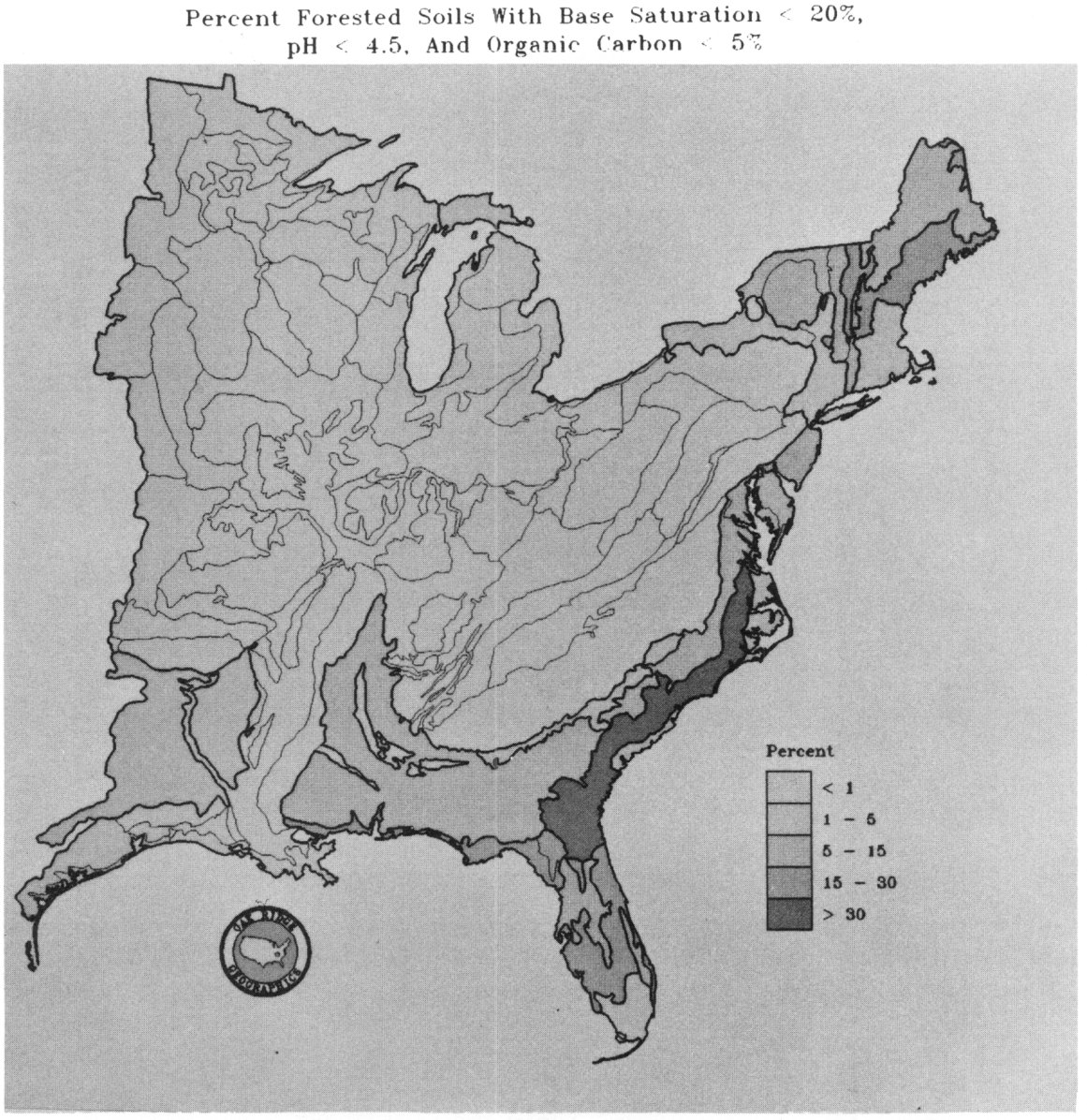

Fig. 13. Percentage of area in MLRAs of the eastern United States having forested surface soils with chemical characteristics that could lead to toxic levels of aluminum in soil solution for some forest species.

ORNL-DWG 85-10291

Percent Forested Soils with Base Saturation < 20%,
pH < 4.5 And Organic Carbon < 5%

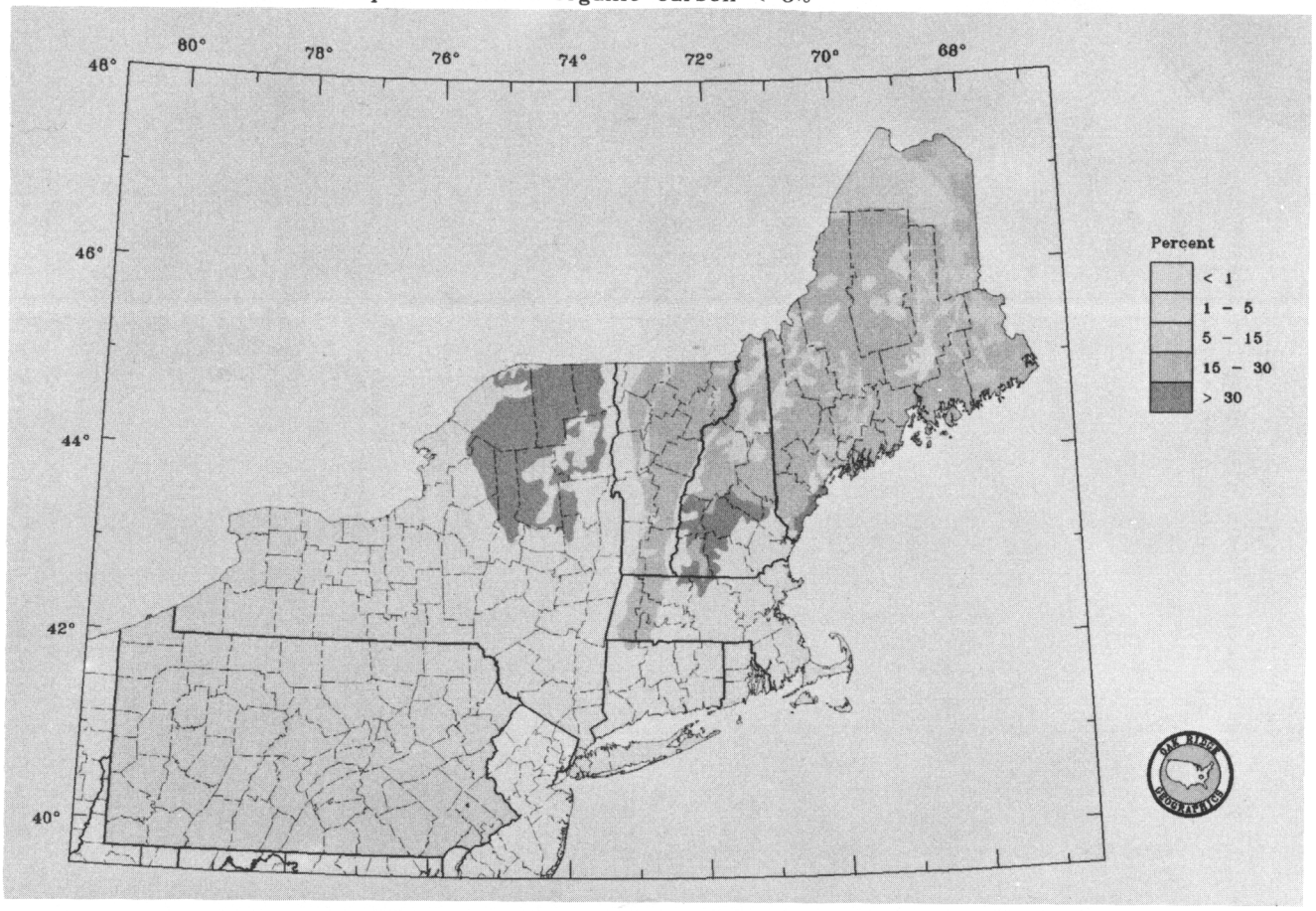

Fig. 14. Percentage of forested surface soils in the northeastern United States with chemical characteristics that could lead to toxic levels of aluminum in soil solution for some forest species.

ORNL-DWG 85-10292

Percent Forested Soils With Base Saturation < 20%,
pH > 4.5, And Organic Carbon < 5%

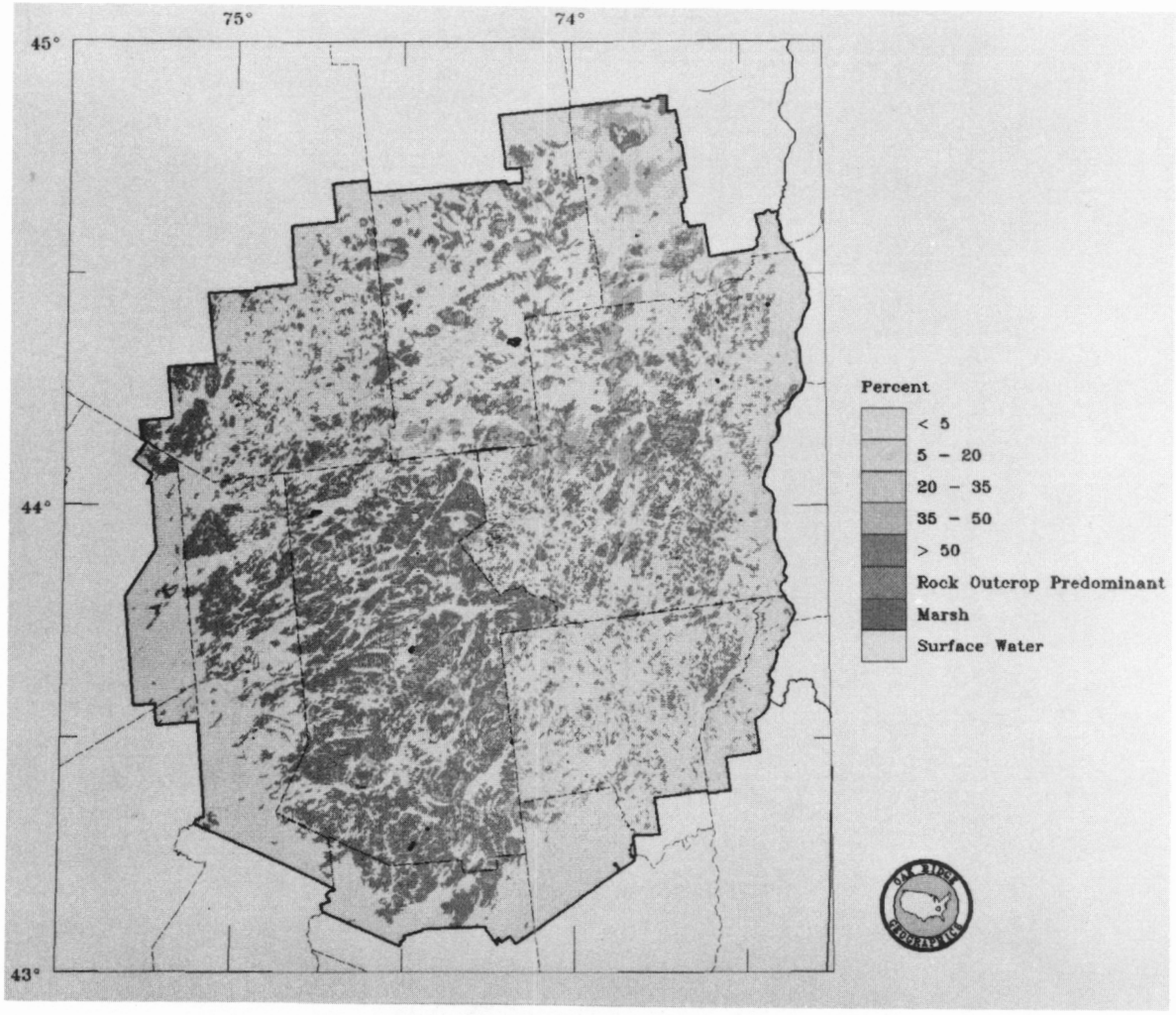

Fig. 15. Percentage of forested surface soils in the Adirondack region of New York with chemical characteristics that could lead to toxic levels of aluminum in soil solution for some forest species.

ORNL DWG 85-15742

Percent Forested Soils with Base Saturation < 20%, pH < 4.5, and Organic Carbon < 2%

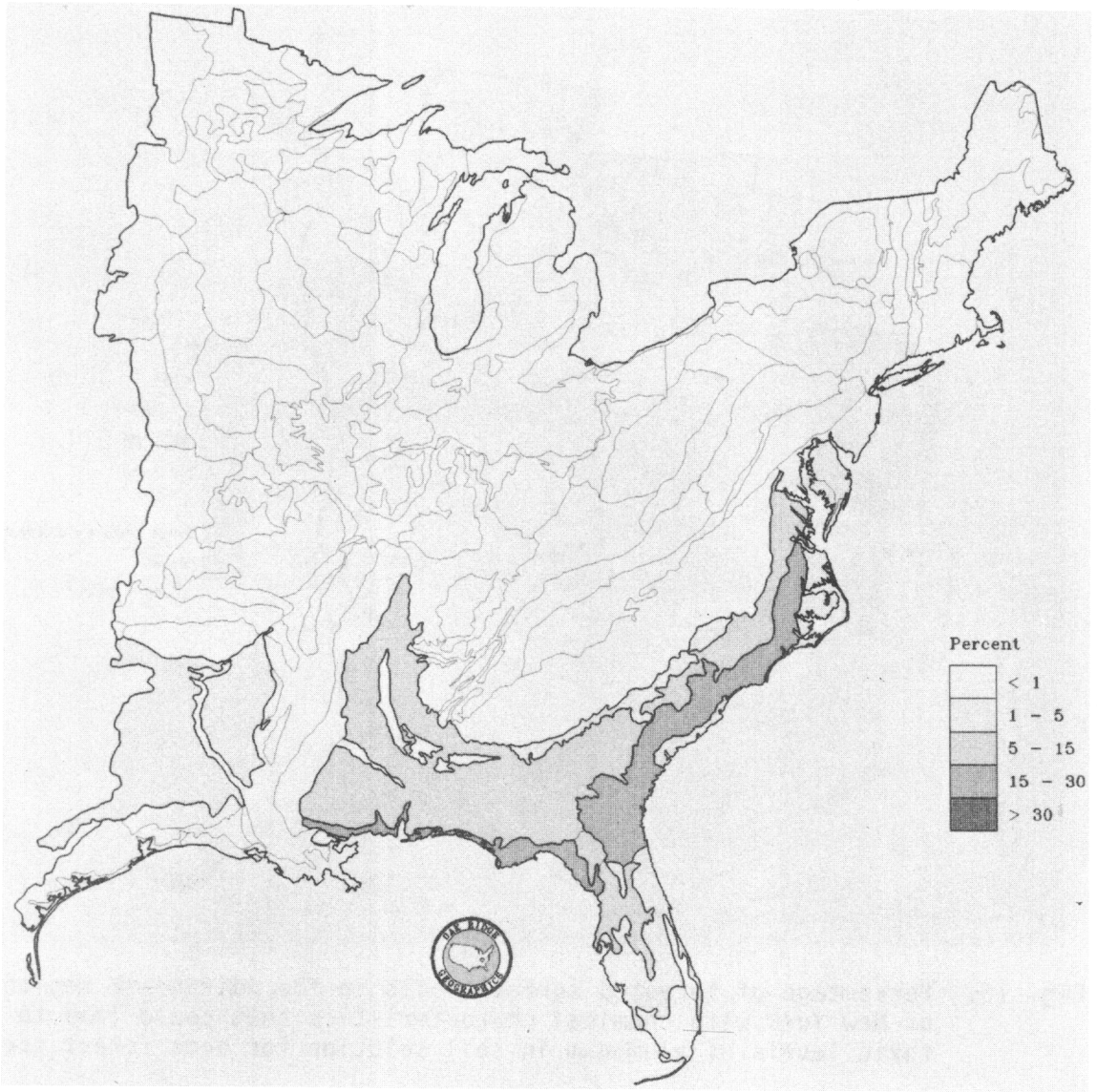

Fig. 16. Percentage of area in MLRAs of the eastern United States having forested surface soils with chemical characteristics that could lead to toxic levels of aluminum in soil solution for some forest species (organic carbon threshold set to 2%).

6. SUMMARY AND CONCLUSIONS

6.1 COMPARISON WITH SENSITIVITY FOR LAKES AND WATERSHEDS

Sensitivity characteristics of soils for forest effects are similar to those for acidification of aquatic systems. In both cases soils with moderate-to-high CEC and BS are considered less sensitive or insensitive because of a high capacity for base cation exchange buffering. In both cases acidic soils with low BS are considered sensitive because of the salt-effect-mediated release of H^+ and Al^{3+} to soil solution. This release may have toxic effects on plants and aquatic organisms but also may reduce solution alkalinity below zero, causing the water to retain the low soil-solution pH when it emerges into surface waters and degasses CO_2. Without this salt-effect reduction in alkalinity the low-pH soil solution would rise substantially in pH when it emerges in surface waters and degasses CO_2 (Reuss and Johnson 1985).

In the past, soil sensitivity criteria for forest effects have considered only surface-soil characteristics. It is now thought that for deep soils and deep-rooting forest species, the more benign characteristics of the deeper horizons may allow trees to accommodate nutrient deficiency or toxic substances in surface soils. Soil sensitivity for aquatic effects is critically dependent on the depth of the soil and unconsolidated materials, the chemical characteristics of the deeper materials, spatial variability of soils with topographic location within watersheds, and hydrologic pathways through these materials. Aquatic chemistry is controlled by the relative amounts of

water flowing through materials with different chemical characteristics in different parts of the watershed (Goldstein and Gherini 1984; Turner et al. 1986). Thus, soils for forest effects assessment are likely to be the upper horizons of the spatially more important upland soils, whereas the soils critical to aquatic effects assessment may be the deeper horizons of upland soils and surface and subsoil horizons of side-slope and lowland soils where runoff is generated in watersheds.

Figure 17 shows areas in the eastern United States having extremely acidic surface soils with rapid, shallow runoff. Under these conditions, the salt-effect-mediated release of H^+ and Al^{3+} to soil solution could result in acidic surface waters. Areas having these soil conditions, plus high runoff rates (Fig. 18), and high acidic deposition rates (appendix A) would be expected to have a greater proportion of acidic lakes and streams than other areas.

6.2 VALIDATION AND TESTING OF MAPS

These maps and ongoing analyses of pedon data and map-unit variability have been used in planning the EPA soil survey to be carried out as part of the USEPA Direct/Delayed Response Project assessing regional sensitivity of aquatic systems to acidic deposition. Results of that soil survey will be used to validate and improve these maps. These maps should also be used in planning the terrestrial effects survey and in locating process-level research on forest effects. Results of the terrestrial survey and effects research can also be used to validate and improve the mapping criteria, map units, and data base.

ORNL DWG 85-15744

Percent Forested Soils with pH < 4.5, Base Saturation < 20%, and Hydrologic Group C or D

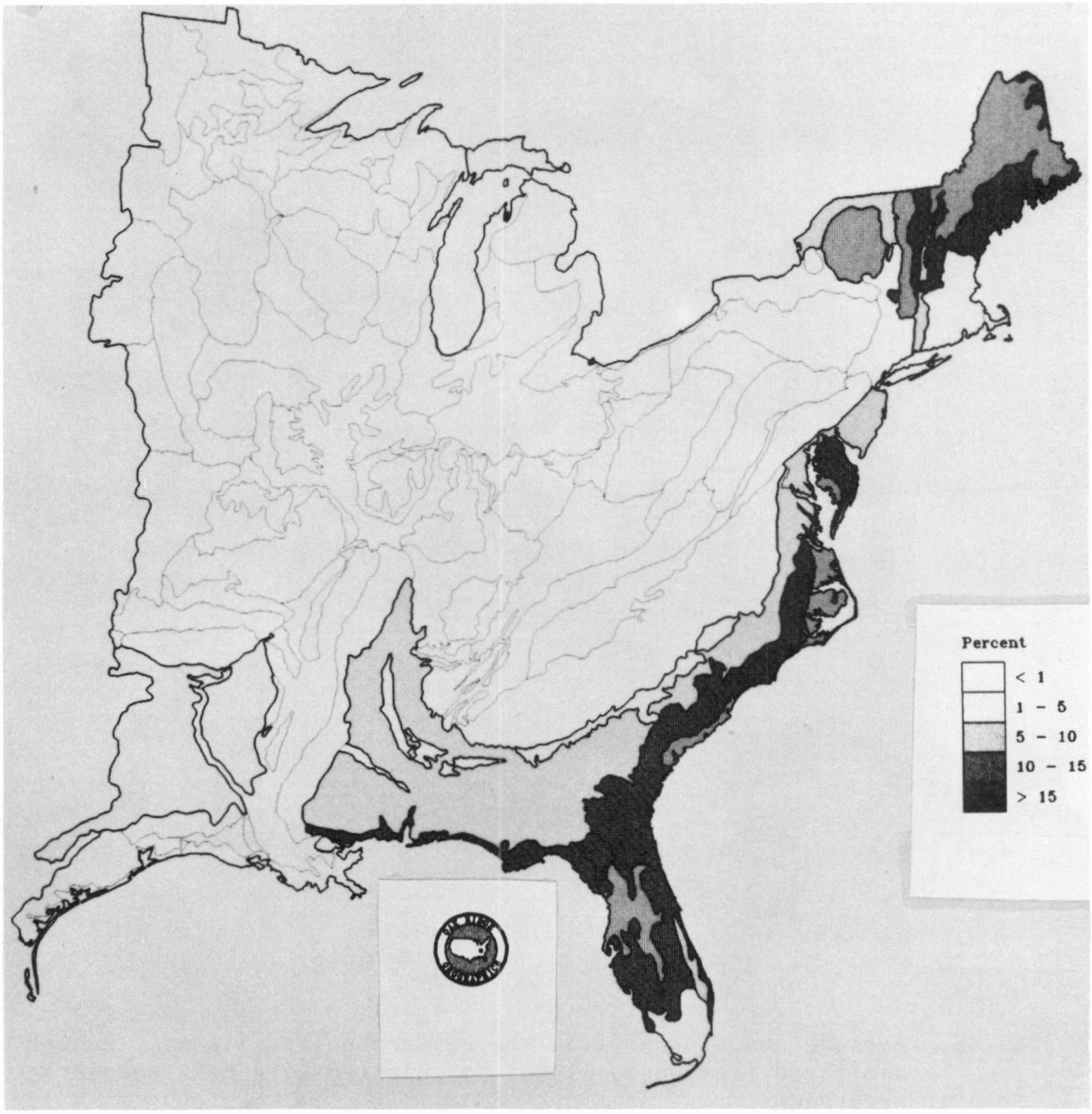

Fig. 17. Percentage of area in MLRAs of the eastern United States having extremely acidic forested surface soils with rapid, shallow runoff.

ORNL-DWG 85-10300

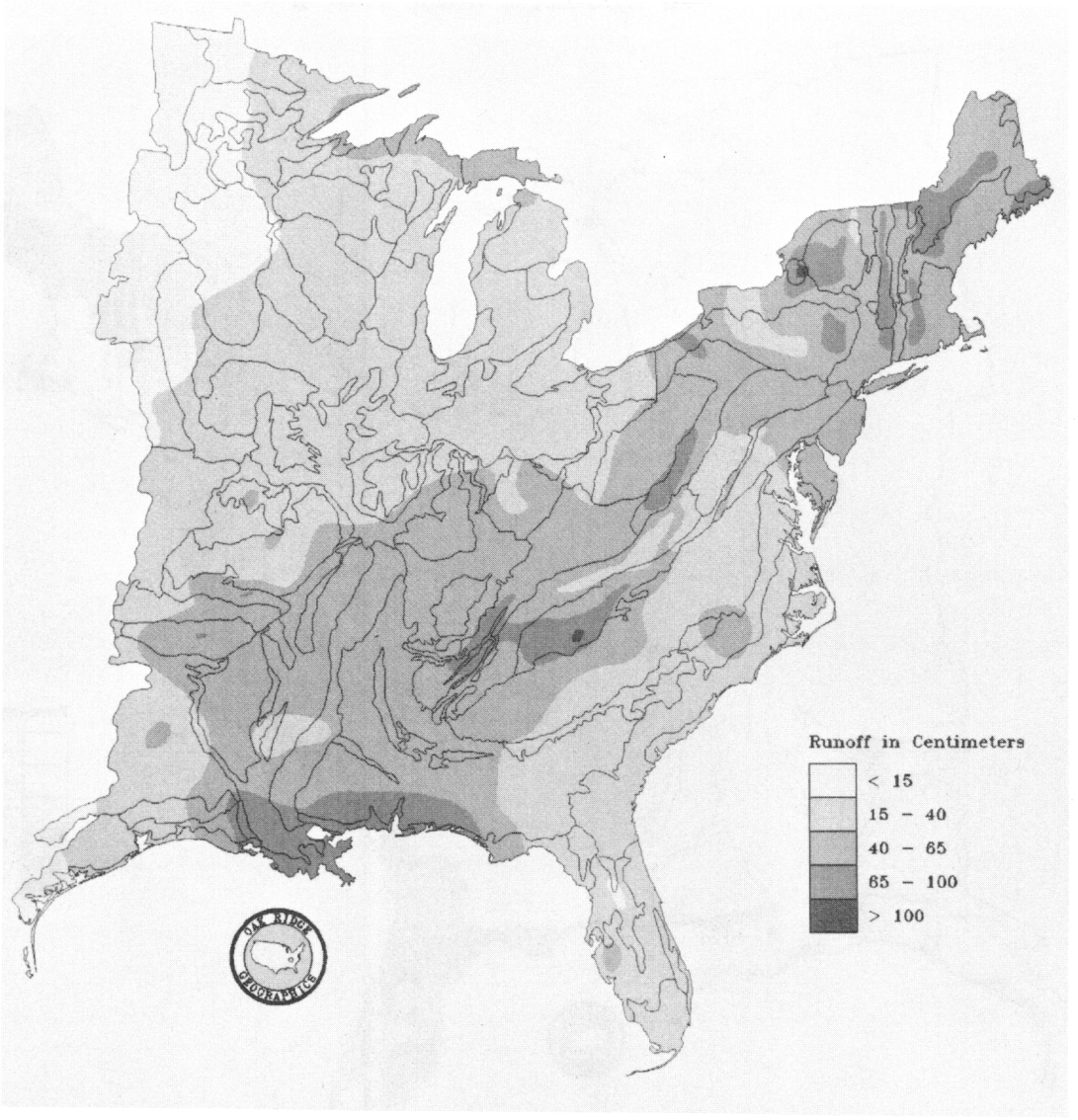

Fig. 18. Average annual runoff in the eastern United States. Runoff was digitized from Busby (1966) and plotted with MLRA boundaries for reference.

6.3 CONCLUSIONS

Given the current understanding of acidic deposition effects on soils and of potential soil-mediated effects on forest productivity, and given the available soil chemical data, there appears to be a significant percentage of forested soils (up to 40%) that may be subject to a decrease in nutrient status because of leaching by acidic deposition. A few soils, occupying small areas of the eastern United States (up to 4% of forested area), may have characteristics that could lead to aluminum toxicity for some forest species. The number of soils and the area could be greater if substantial leaching of base cations occurs in soils with pH < 4.5 without replacement by weathering or atmospheric inputs. However, changes in many soils in the central and southern United States may be delayed by the large sulfate adsorption capacity of many of those soils. There may already be substantial areas of soils with low base-cation nutrient status, especially in the Southeast and in northern New England. Forests in these areas may be predisposed to damage by other stresses.

To better assess potential soil-mediated effects of acidic deposition on forest productivity, we need a better understanding of the mechanisms of forest decline. Dose-response relationships and threshold values for different species need to be determined from forest growth studies and field manipulations in different types of soils. Other factors, such as pollutant loading and climate, can then be added to the assessment in an ecosystem-level analysis for different forest types in different locations.

Also needed is a better understanding of spatial patterns of soil chemical characteristics, especially anion mobility and rates of weathering replacement of base cations. We need to determine the most useful and economic level of aggregation of soils and to further develop techniques of spatial analysis at a scale that will allow us to quantify and locate areas with different forest types at risk to acidic deposition or other pollutant inputs. Until we can better identify forest decline mechanisms, dose-response relationships, and threshold values of associated soil chemical characteristics, simple conceptual models are probably most useful for assessment purposes. The models can be used to estimate areas potentially at risk as a function of a range of criteria for specific ecosystem characteristics, such as soil and vegetation types. As our understanding of the damage mechanisms and threshold values evolves, we can build that knowledge into more complex models.

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APPENDIX A. WET ACIDIC DEPOSITION MAPS.

ORNL-DWG 85-10294

1980–1982 Average Annual Hydrogen Ion Wet Deposition
With MLRA Boundaries

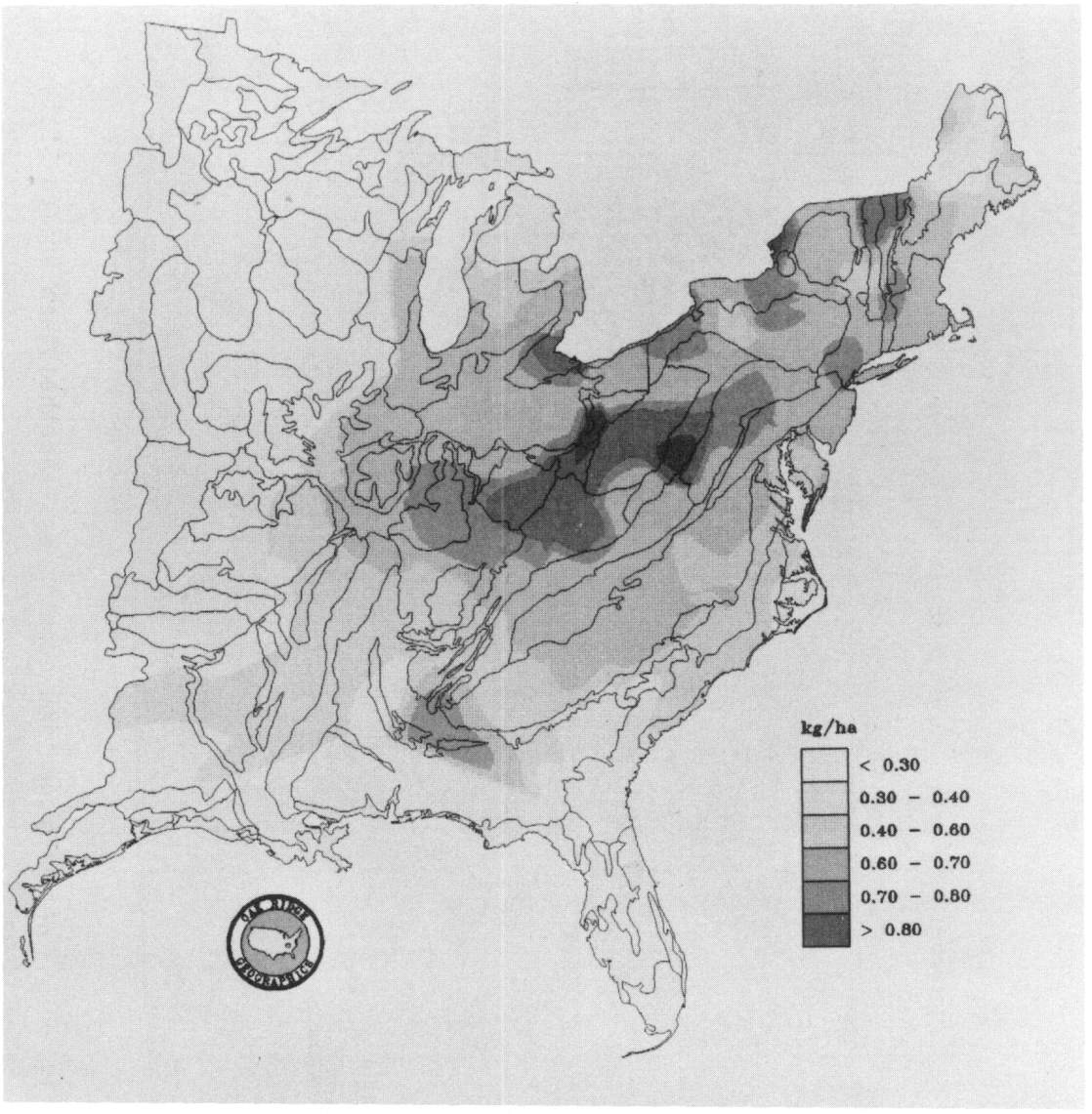

Fig. A-1 Wet hydrogen ion deposition – eastern United States.

ORNL-DWG 85-10295

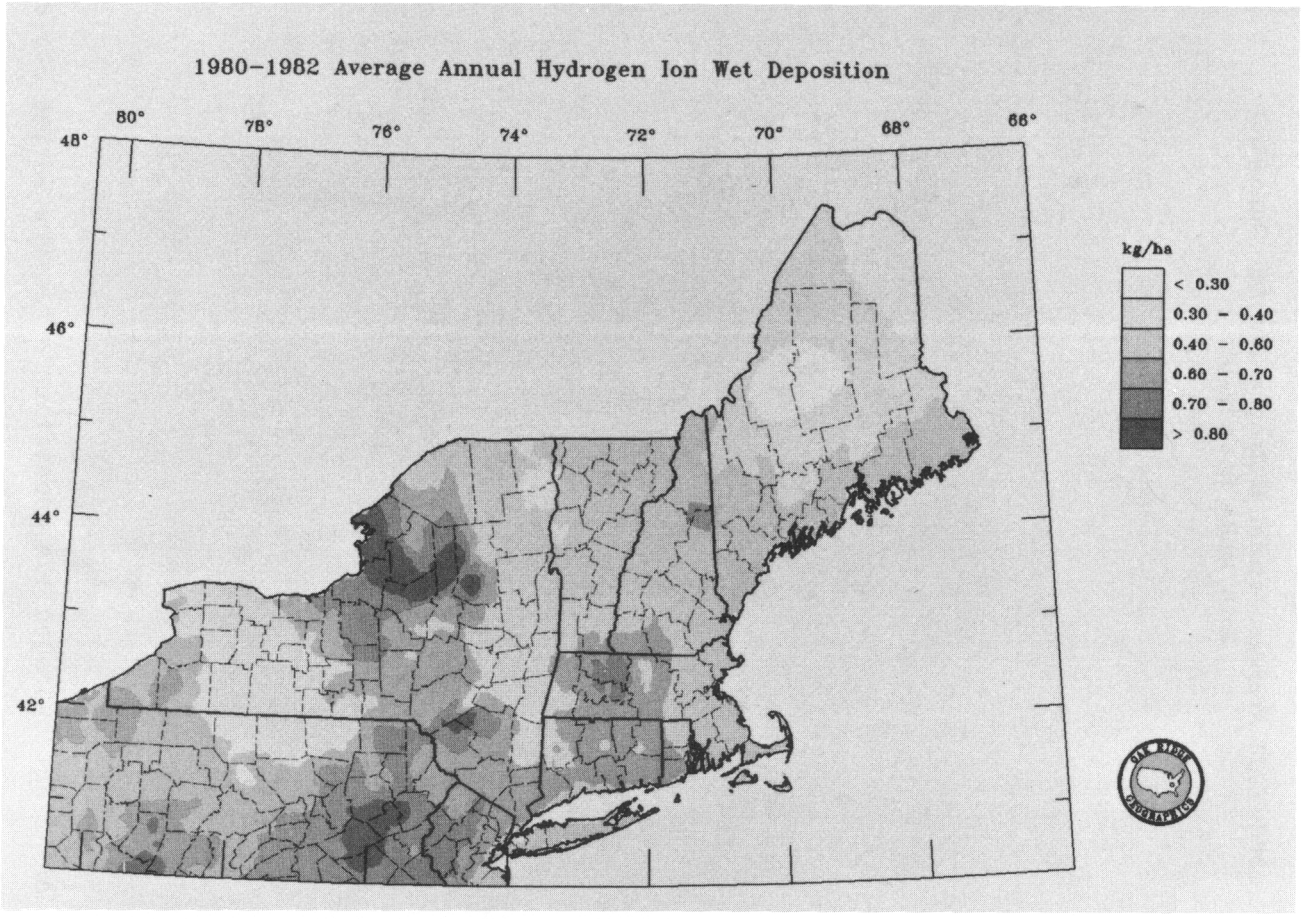

Fig. A-2 Wet hydrogen ion deposition - northeastern United States.

ORNL-DWG 85-10296

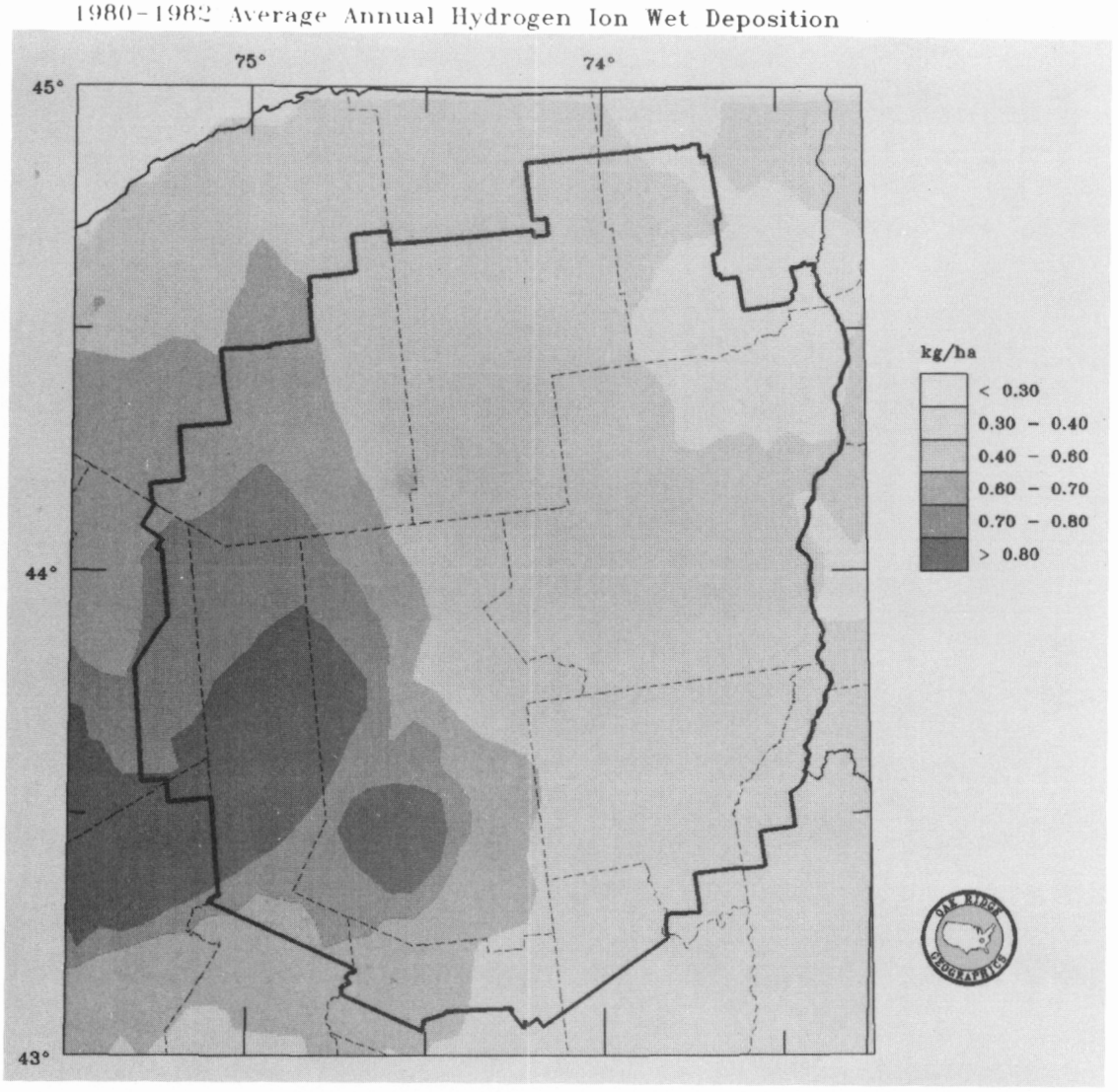

Fig. A-3 Wet hydrogen ion deposition - Adirondack region of New York.

ORNL-DWG 86-6946

1980–1982 Average Annual Sulfate Ion Wet Deposition With MLRA Boundaries

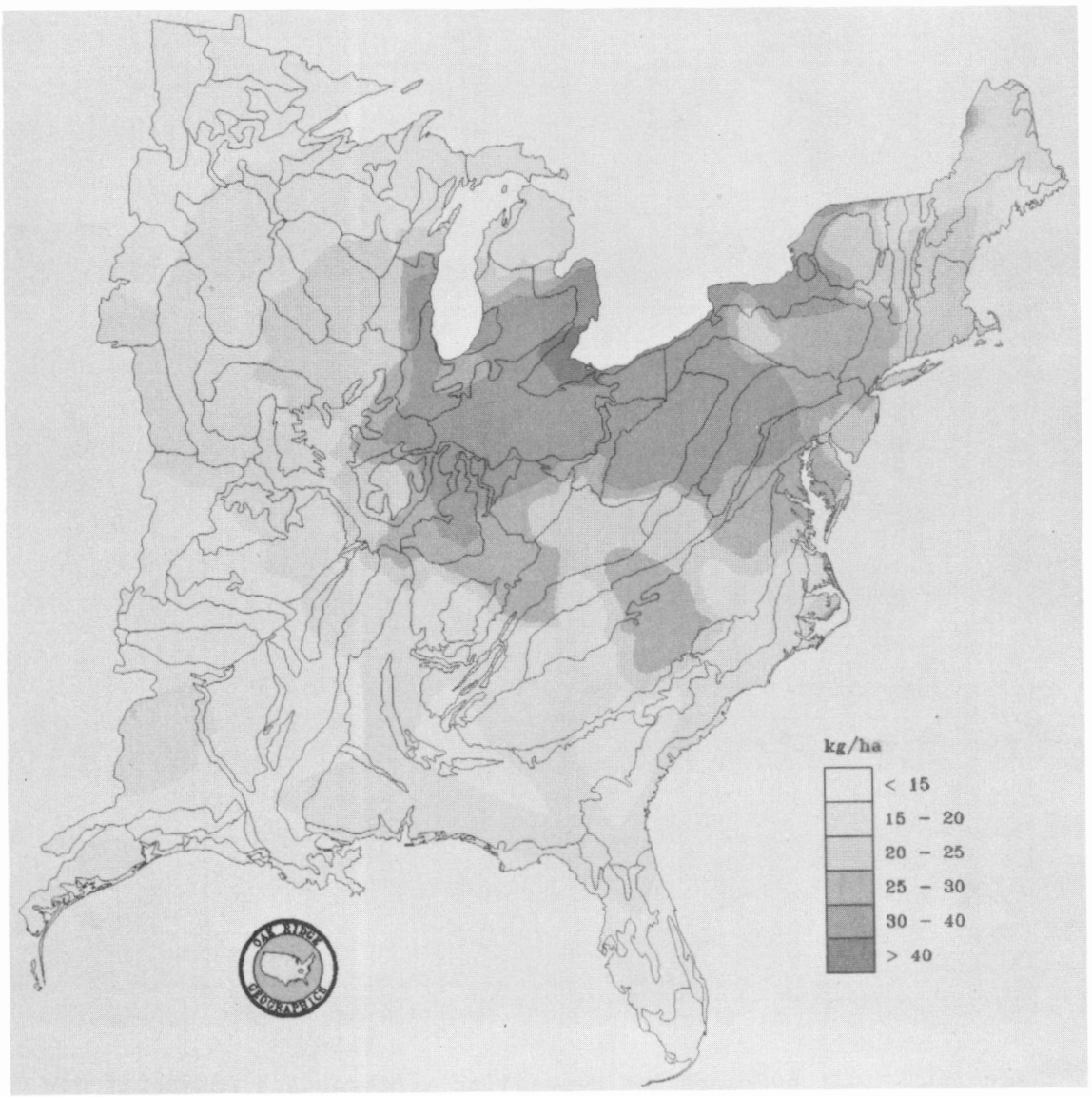

Fig. A-4 Wet sulfate deposition – eastern United States.

ORNL–DWG 85-11271

1980–1982 Average Annual Sulfate Wet Deposition

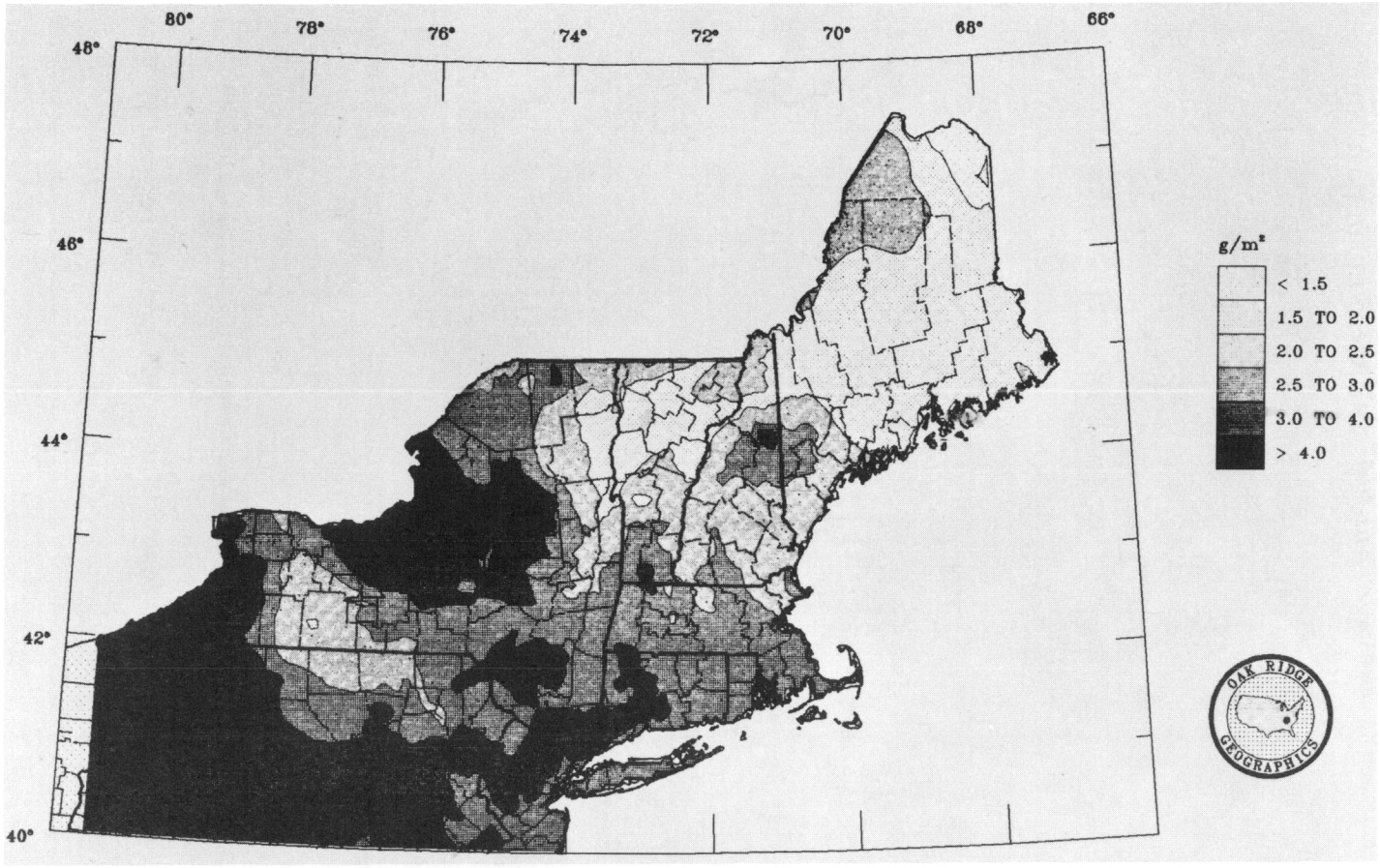

Fig. A-5 Wet sulfate deposition - northeastern United States.

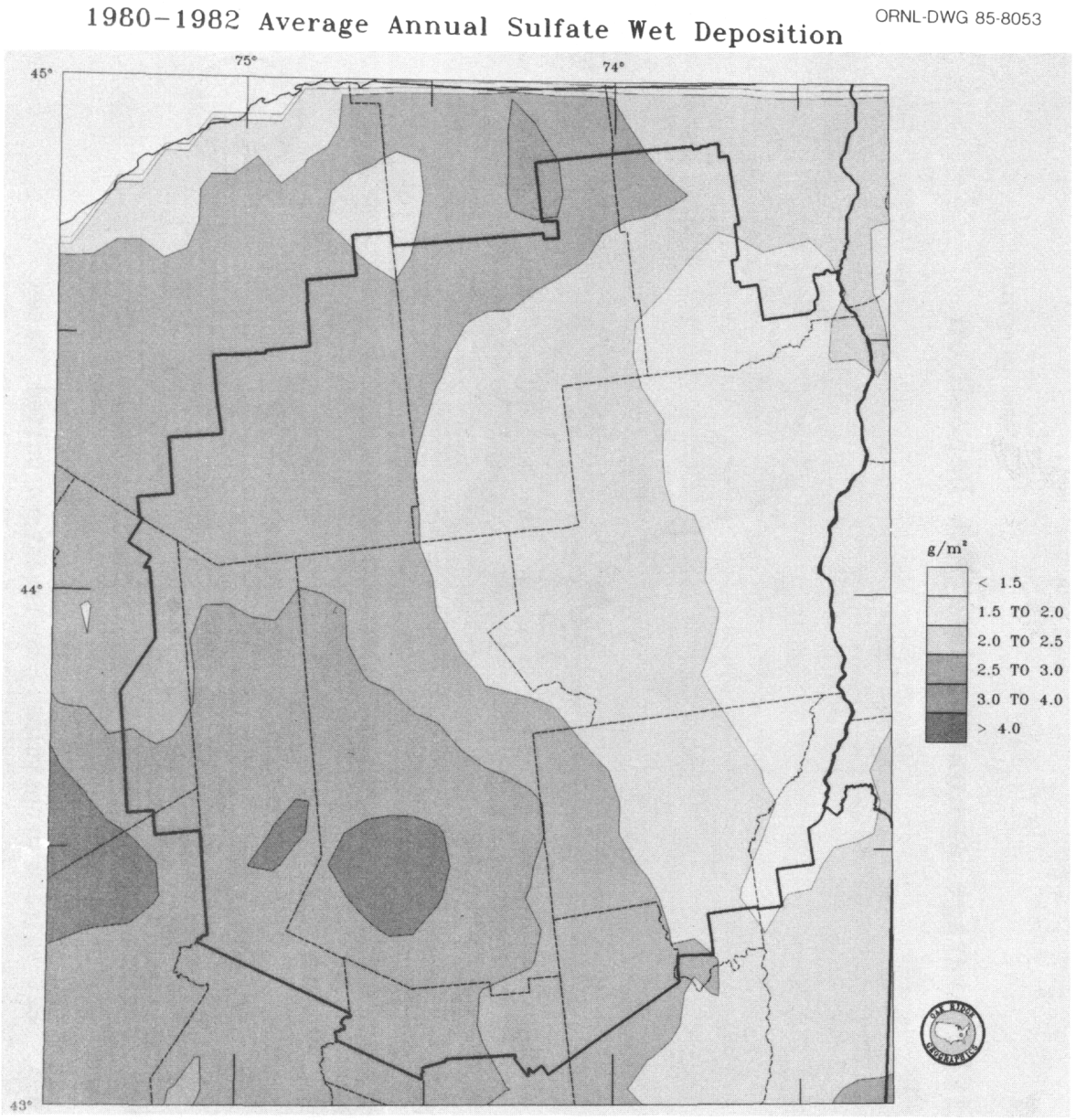

Fig. A-6 Wet sulfate deposition - Adirondack region of New York.

ORNL-DWG 86-6945

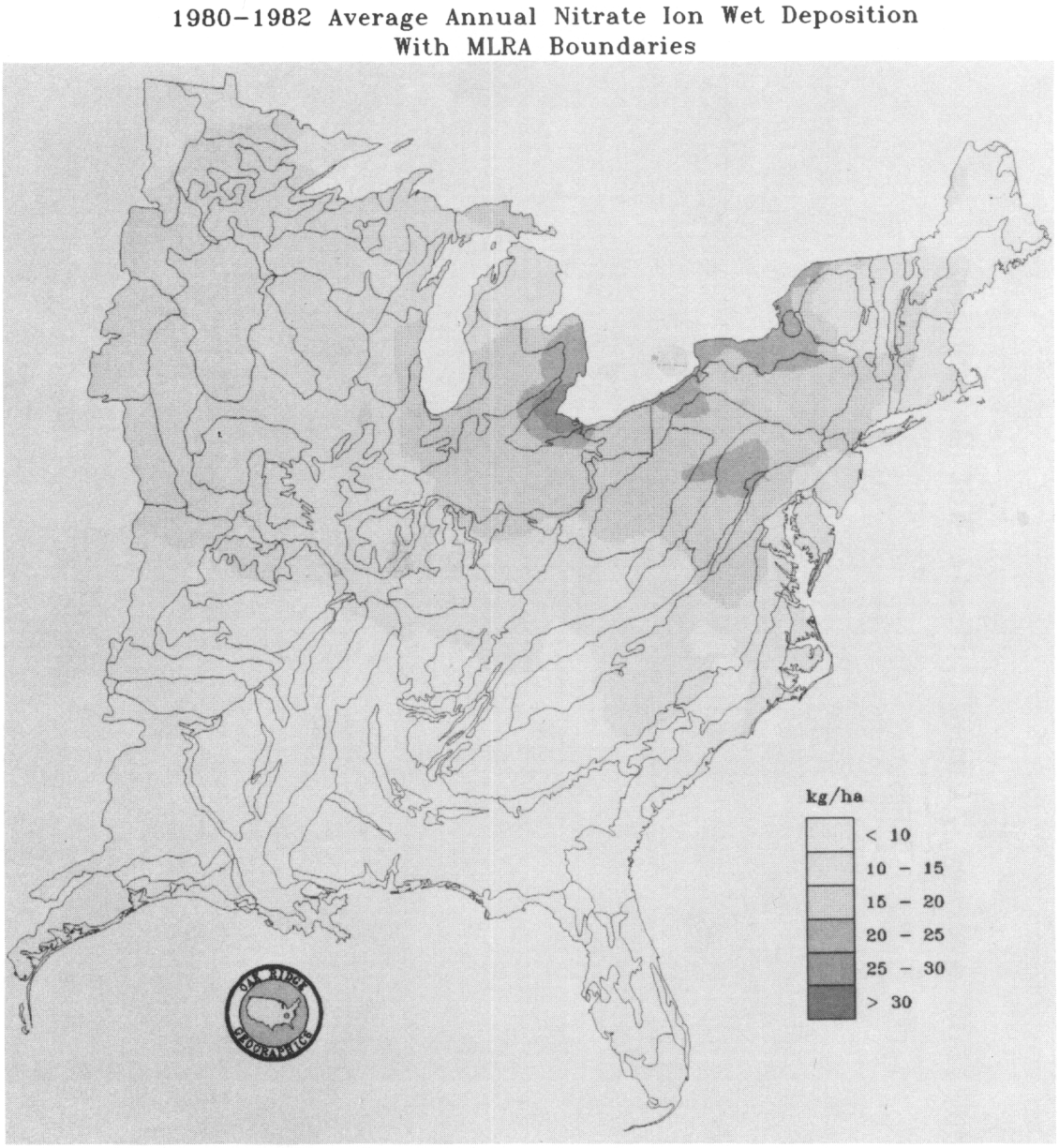

Fig. A-7 Wet nitrate deposition – eastern United States.

ORNL–DWG 85-11272

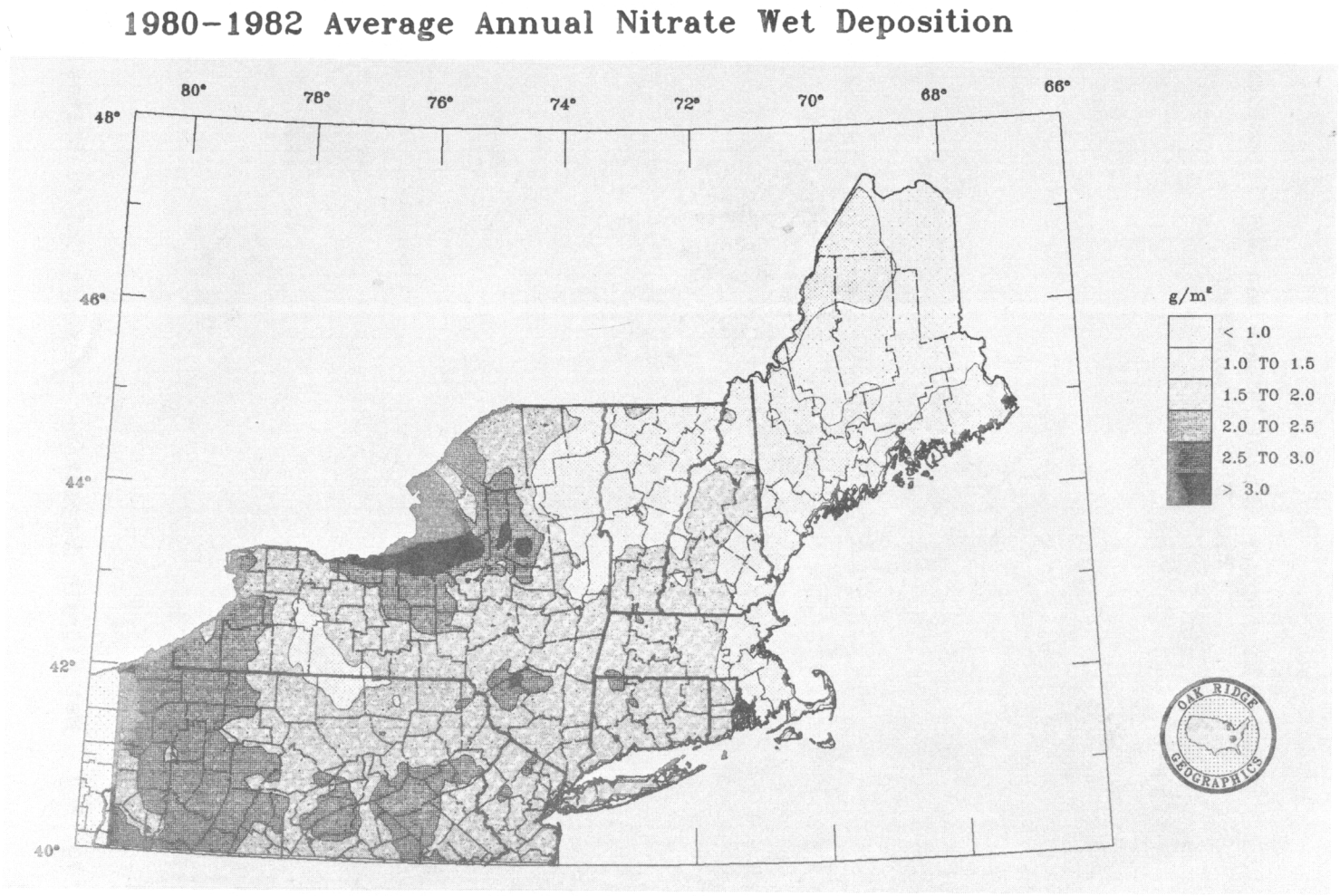

Fig. A-8 Wet nitrate deposition - northeastern United States.

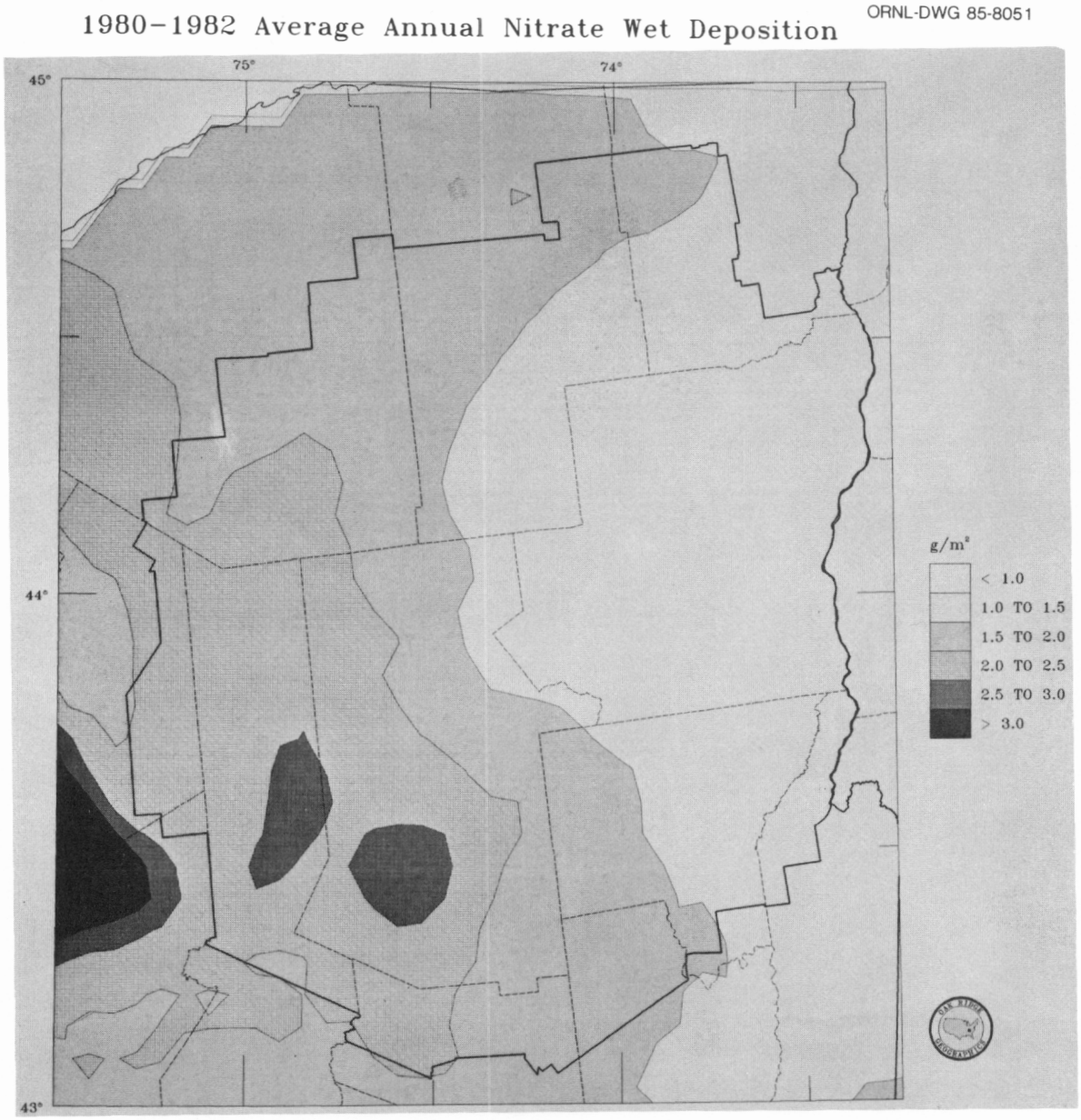

Fig. A-9 Wet nitrate deposition - Adirondack region of New York.

APPENDIX B. SINGLE FACTOR SOILS MAPS.

ORNL DWG 85-15729

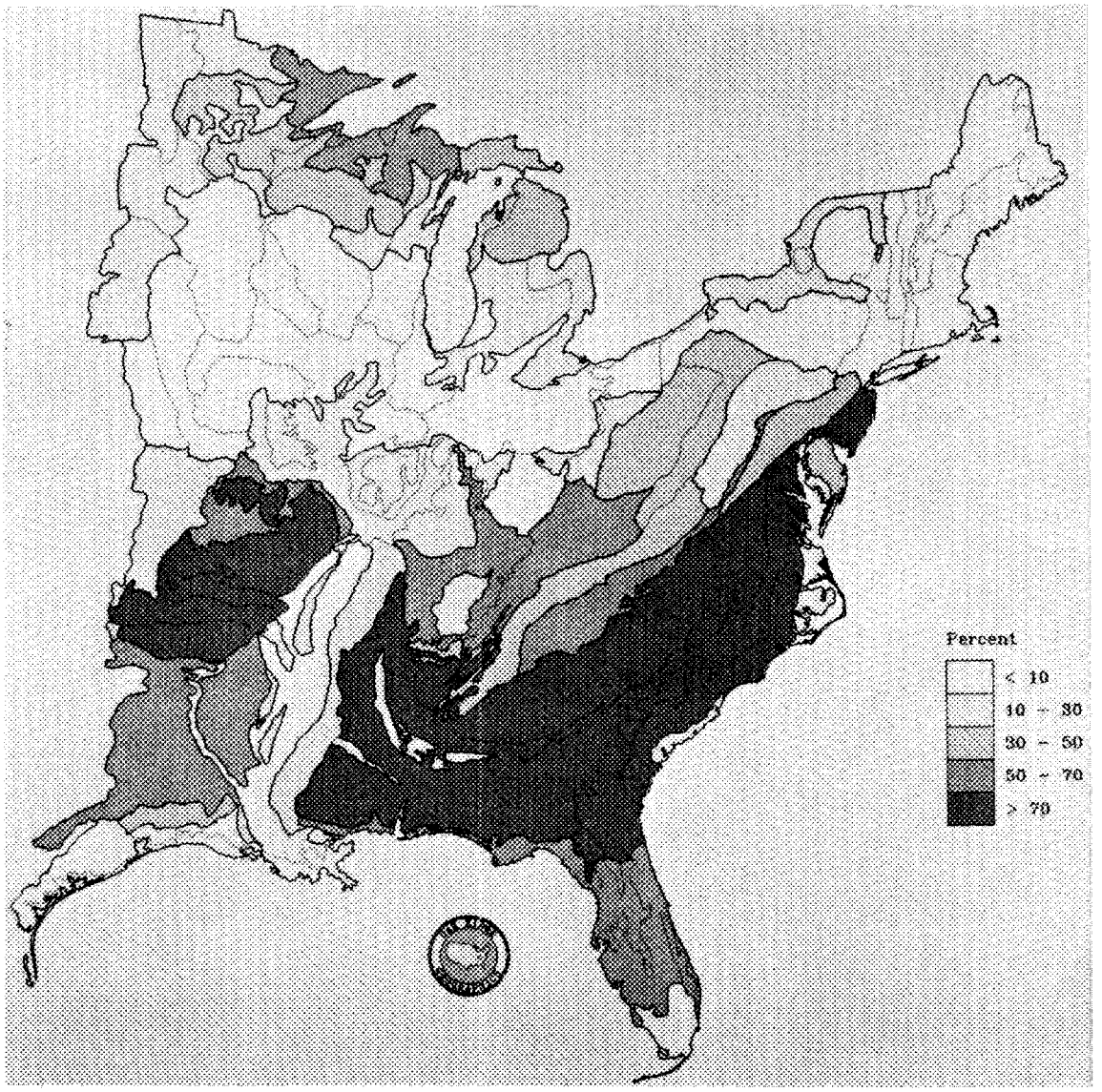

Fig. B-1 Percentage of area in MLRAs of the eastern U.S. having forested surface soils with cation exchange capacity <10 cmol(+)/kg.

ORNL-DWG 86-6949

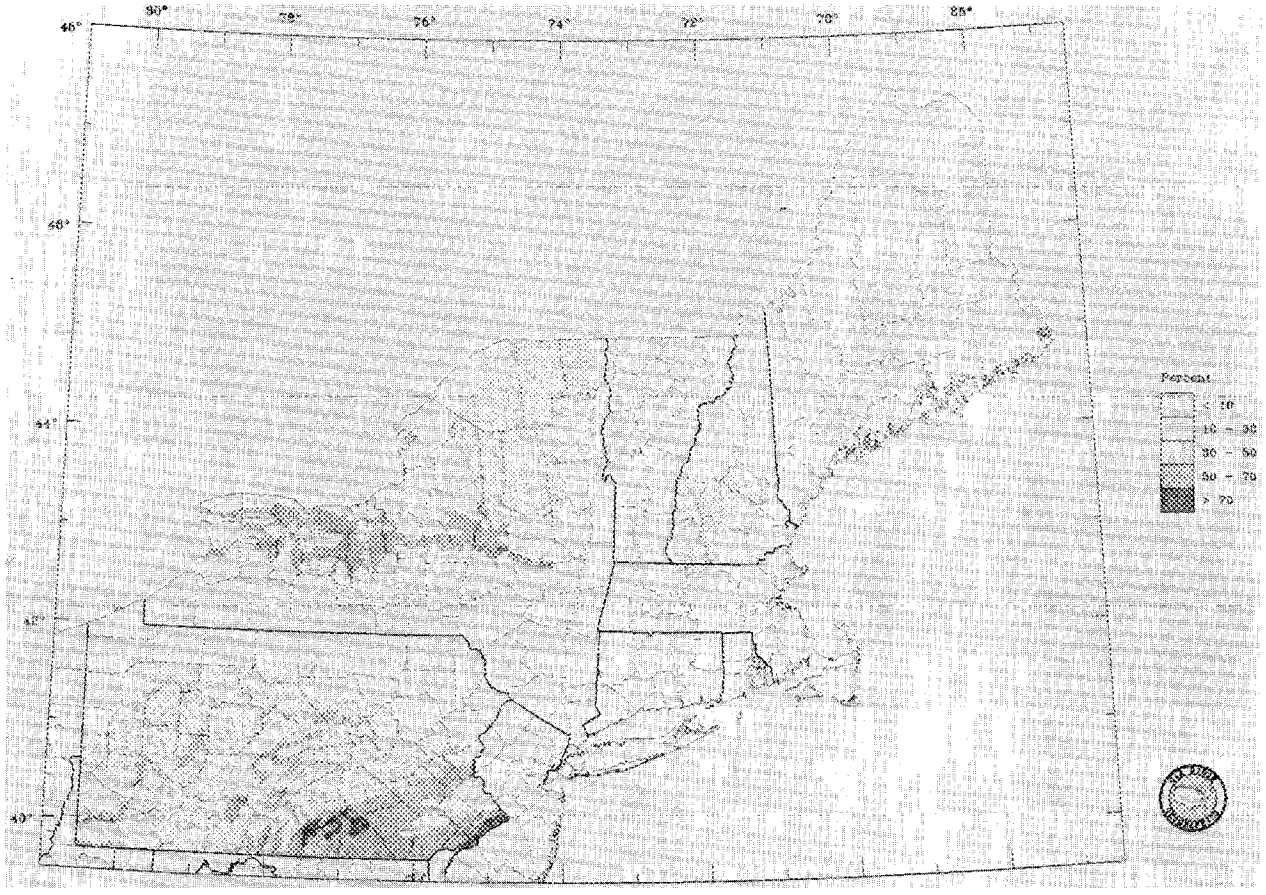

Fig. B-2 Percentage of forested surface soils in the northeastern United States with cation exchange capacity <10 cmol(+)/kg.

ORNL-DWG 86-6950

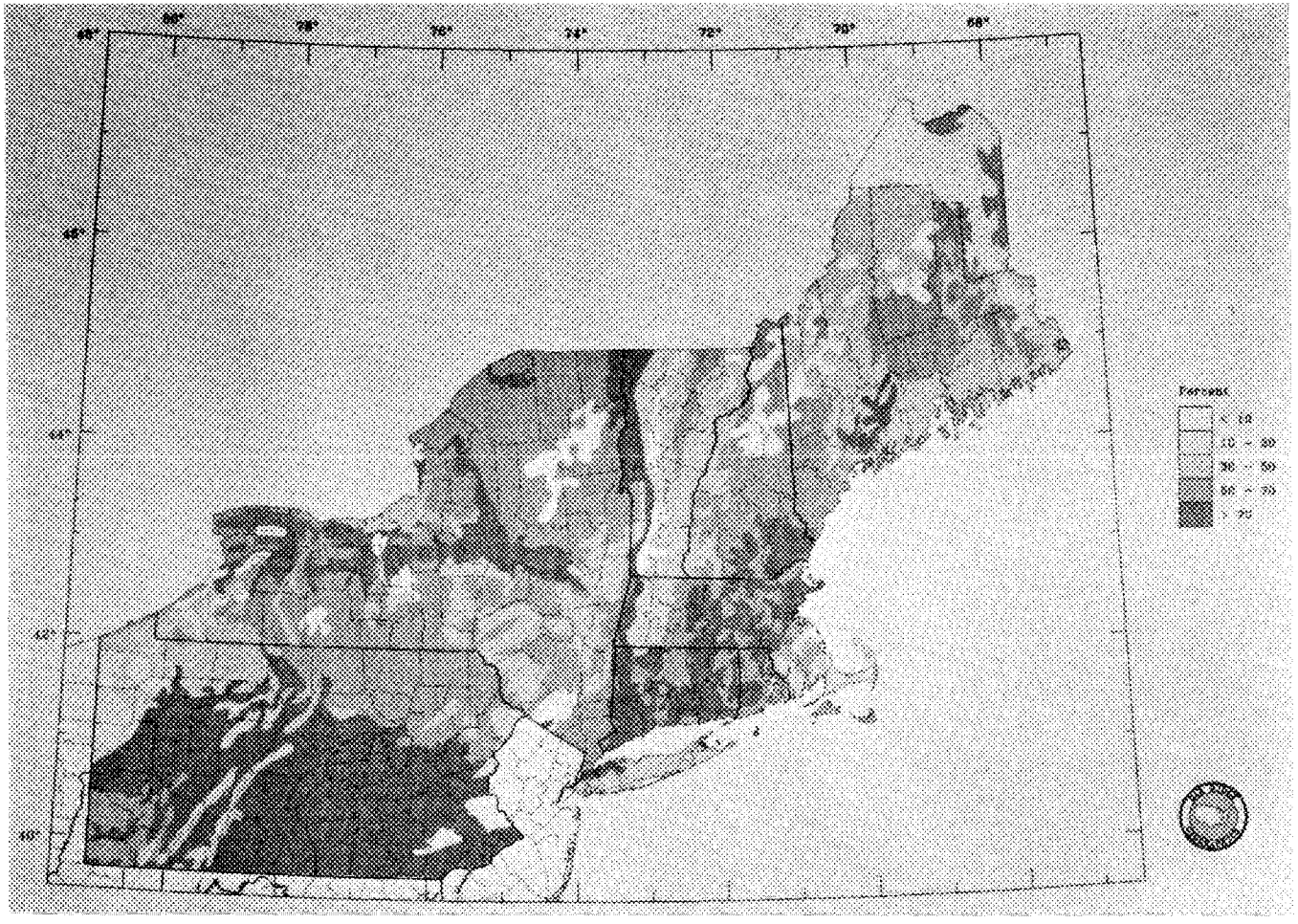

Fig. B-3 Percentage of forested surface soils in the northeastern United States with cation exchange capacity <15 cmol (+)/kg.

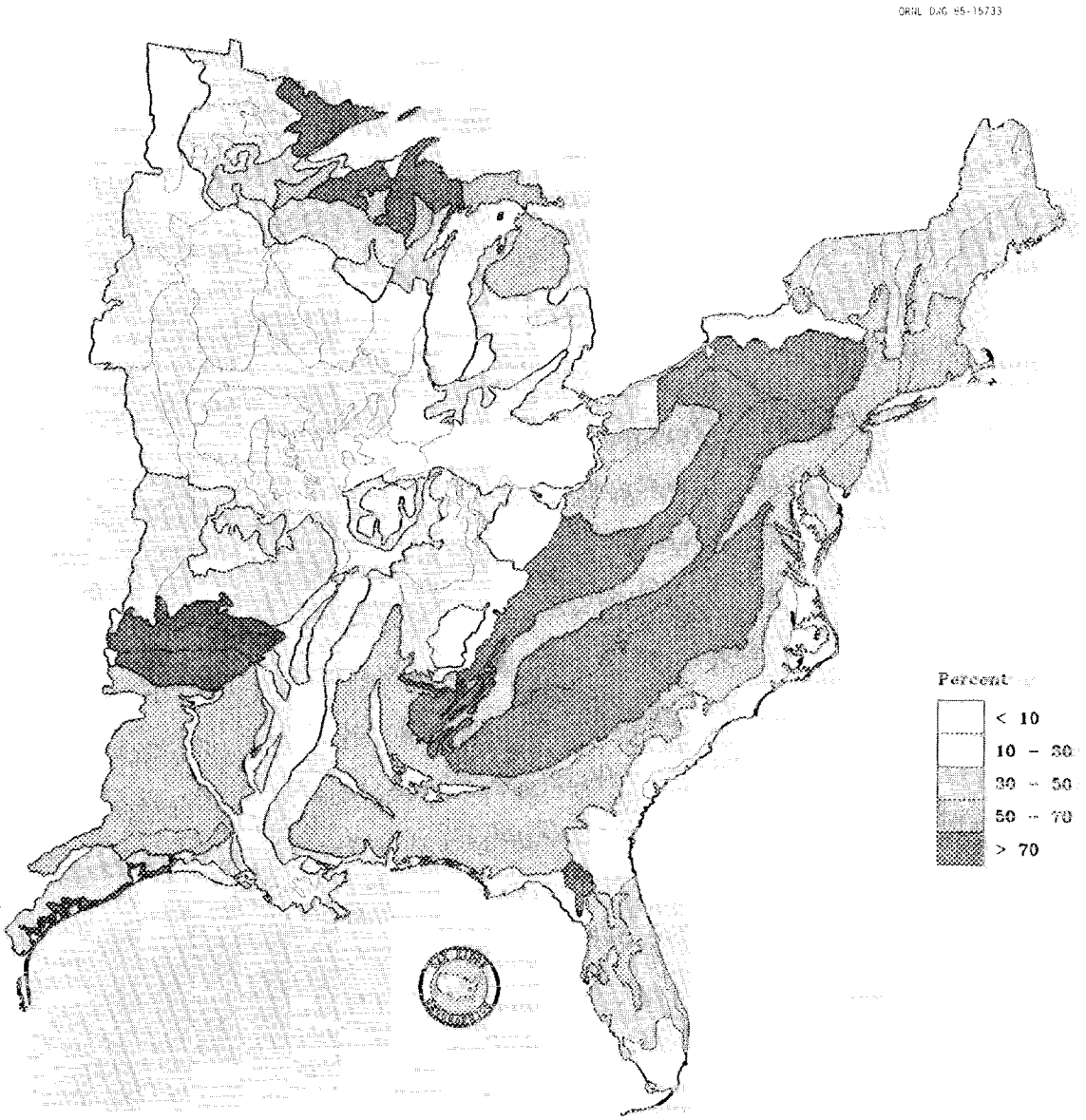

Fig. B-4 Percentage of area in Major Land Resource Areas of the eastern United States having forested surface soils with base saturation 20% – 60%.

ORNL-DWG 85-6951

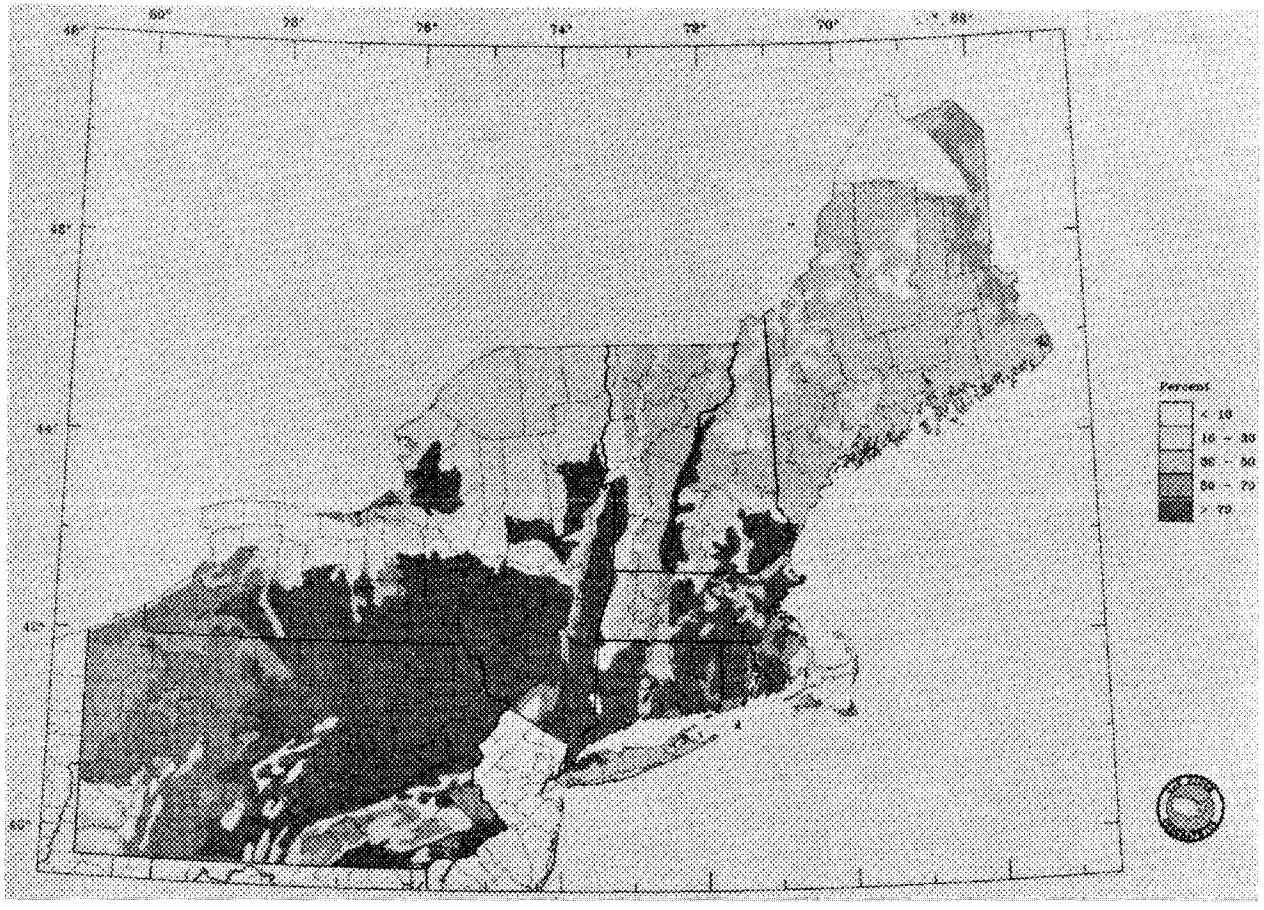

Fig. B-5 Percentage of forested surface soils in the northeastern United States with base saturation 20% - 60%.

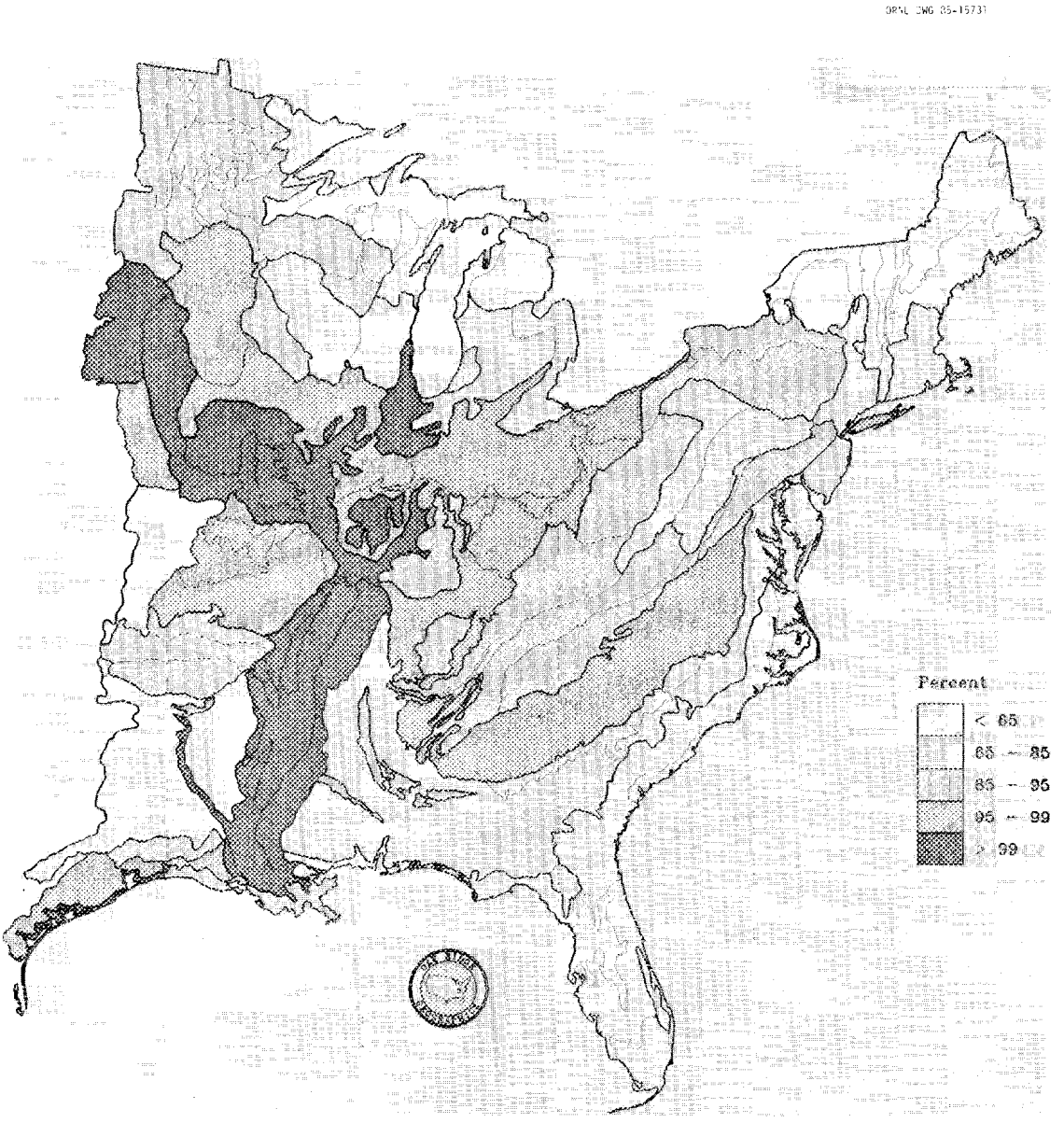

Fig. B-6 Percentage of area in Major Land Resource Areas of the eastern United States having forested surface soils with pH >5.

ORNL-DWG 86-6959

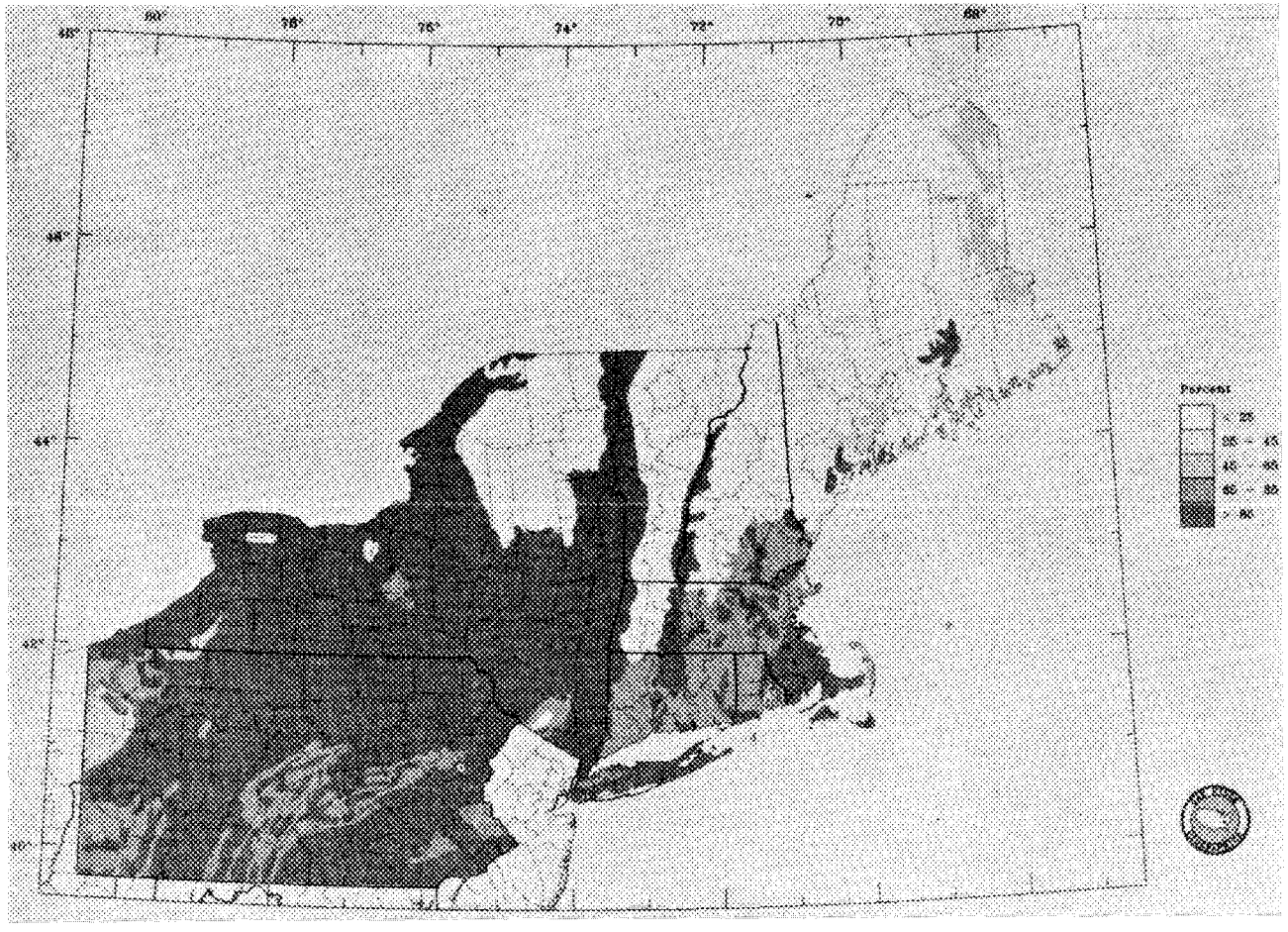

Fig. B-7 Percentage of forested surface soils in the northeastern United States with pH >5.

ORNL/TM-9917

ORNL-DWG 86-6953

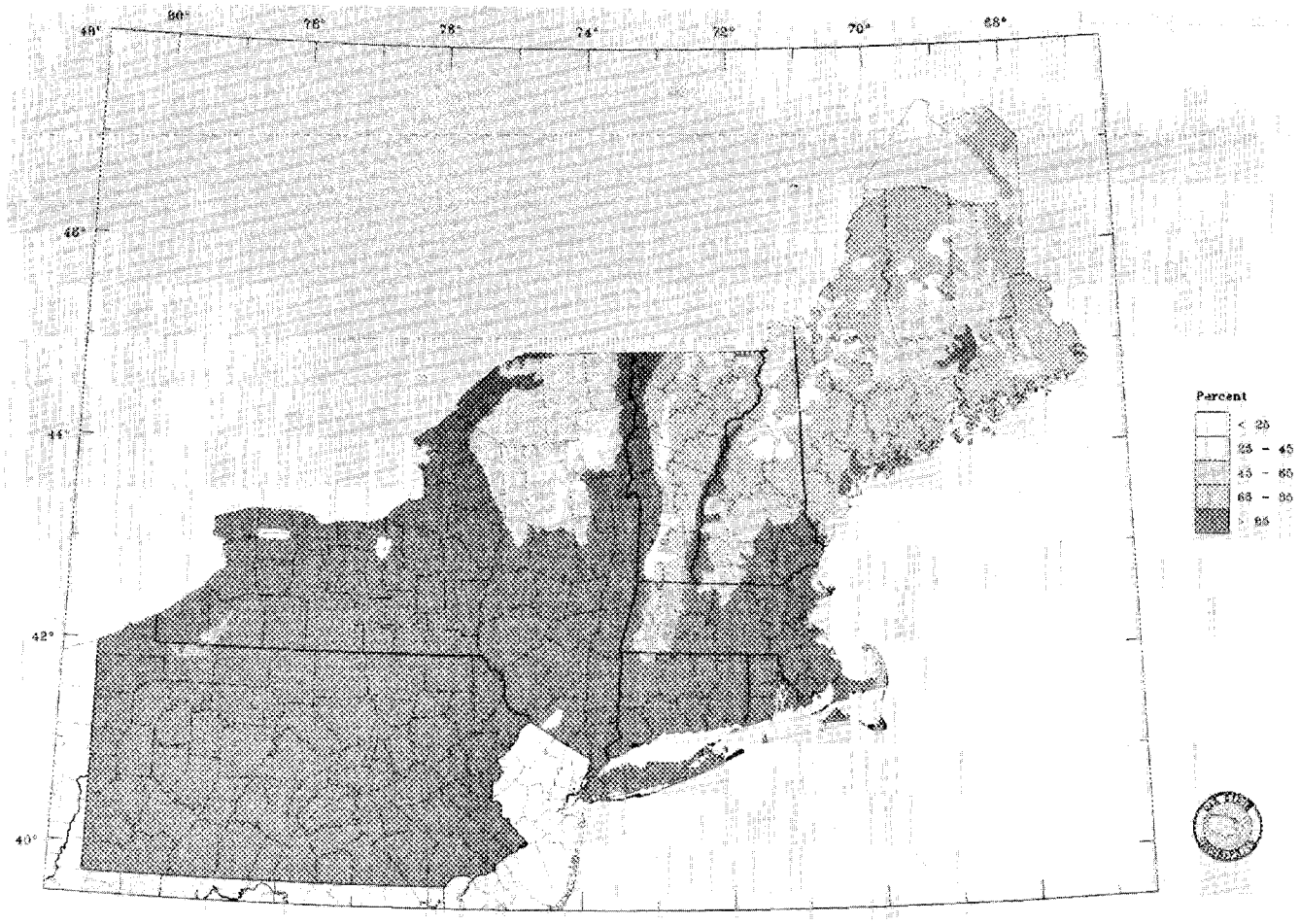

Fig. B-8 Percentage of forested surface soils in the northeastern United States with pH > 4.5.

ORNL DWG 85-15736

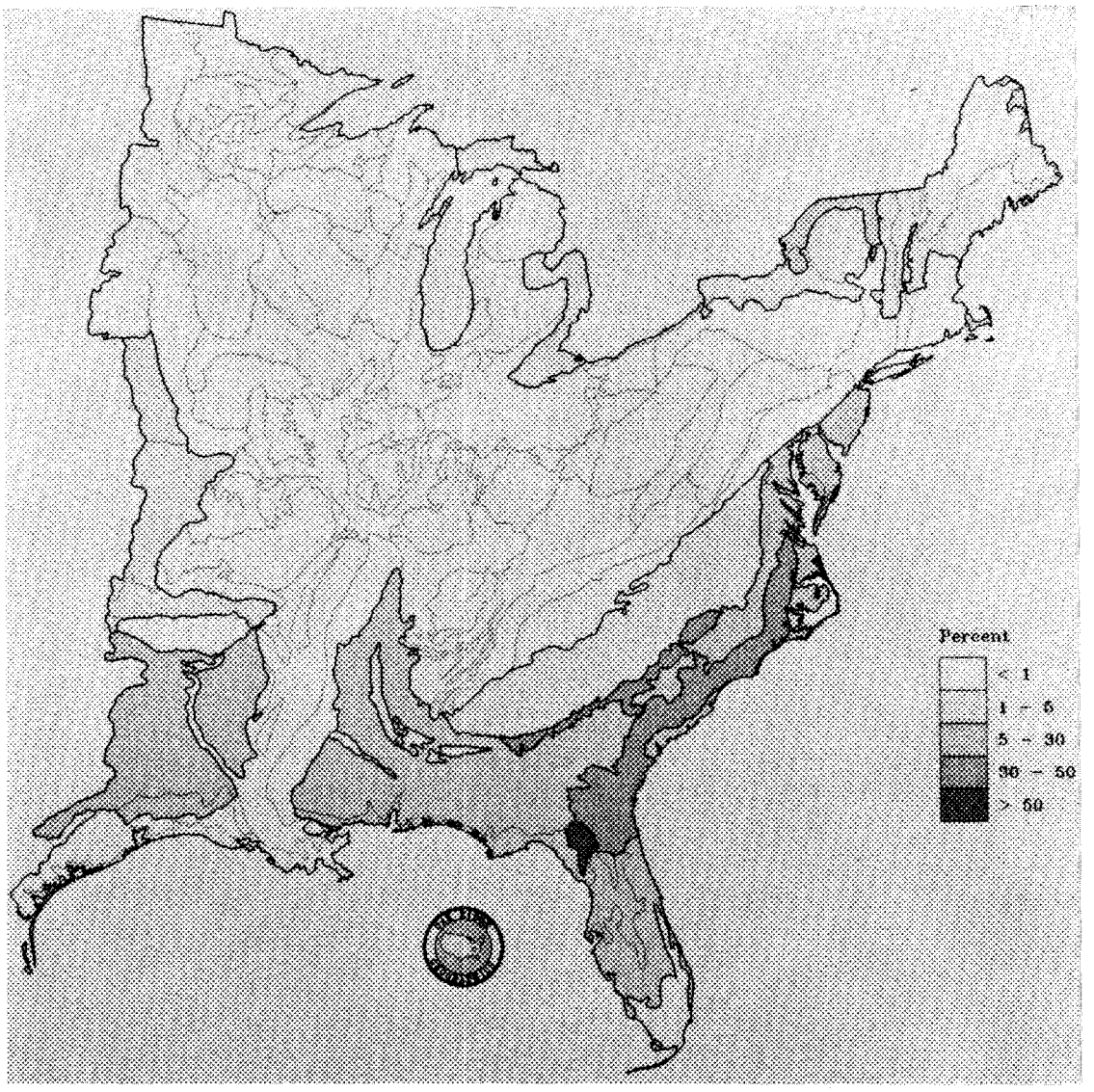

Fig. B-9 Percentage of area in Major Land Resource Areas of the eastern United States having forested surface soils with exchangeable bases <1 cmol(+)/kg.

ORNL-DWG 86-6957

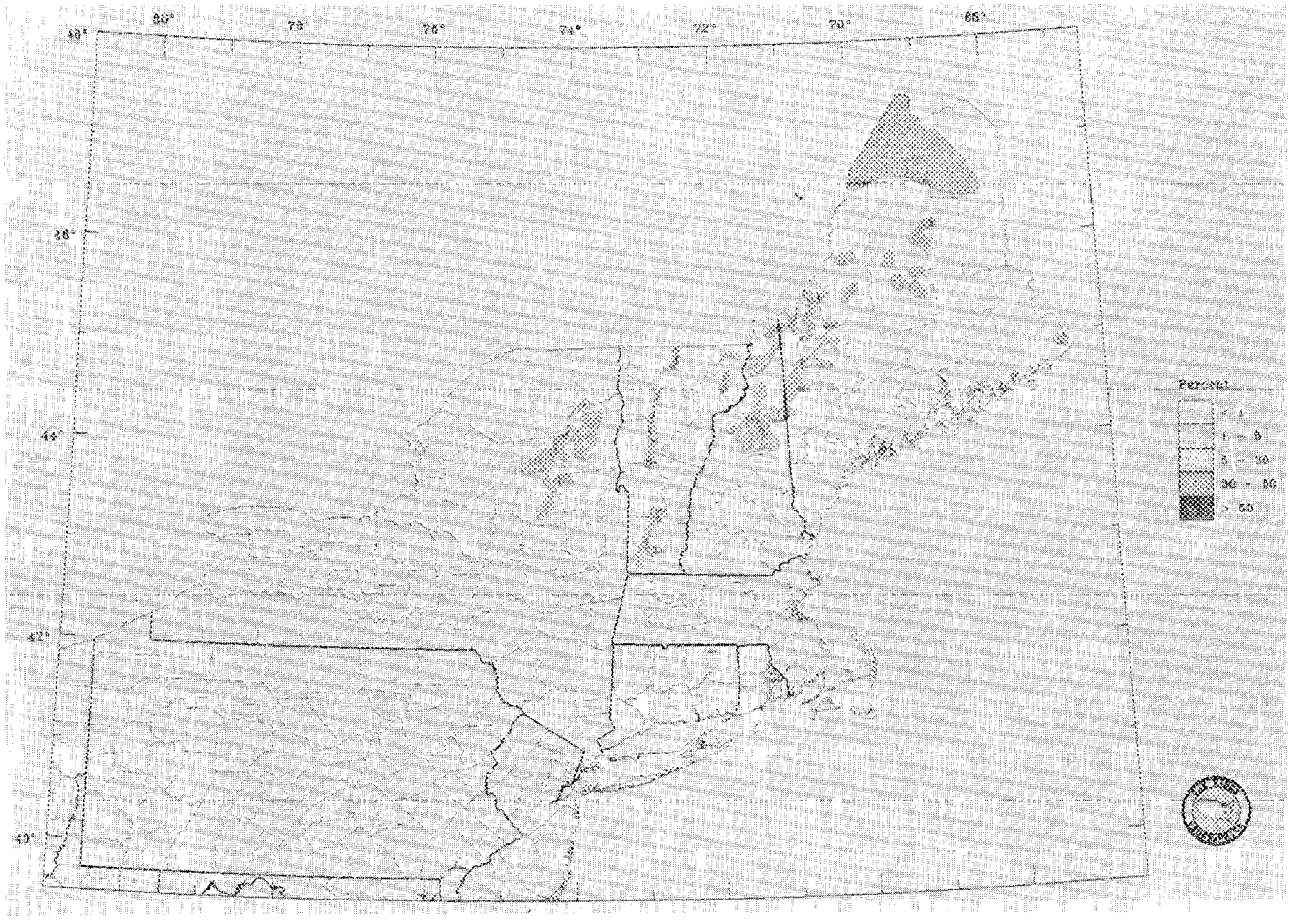

Fig. B-10 Percentage of forested surface soils in the northeastern United States with exchangeable bases <1 cmol(+)/kg.

ORNL DWG 85-15732

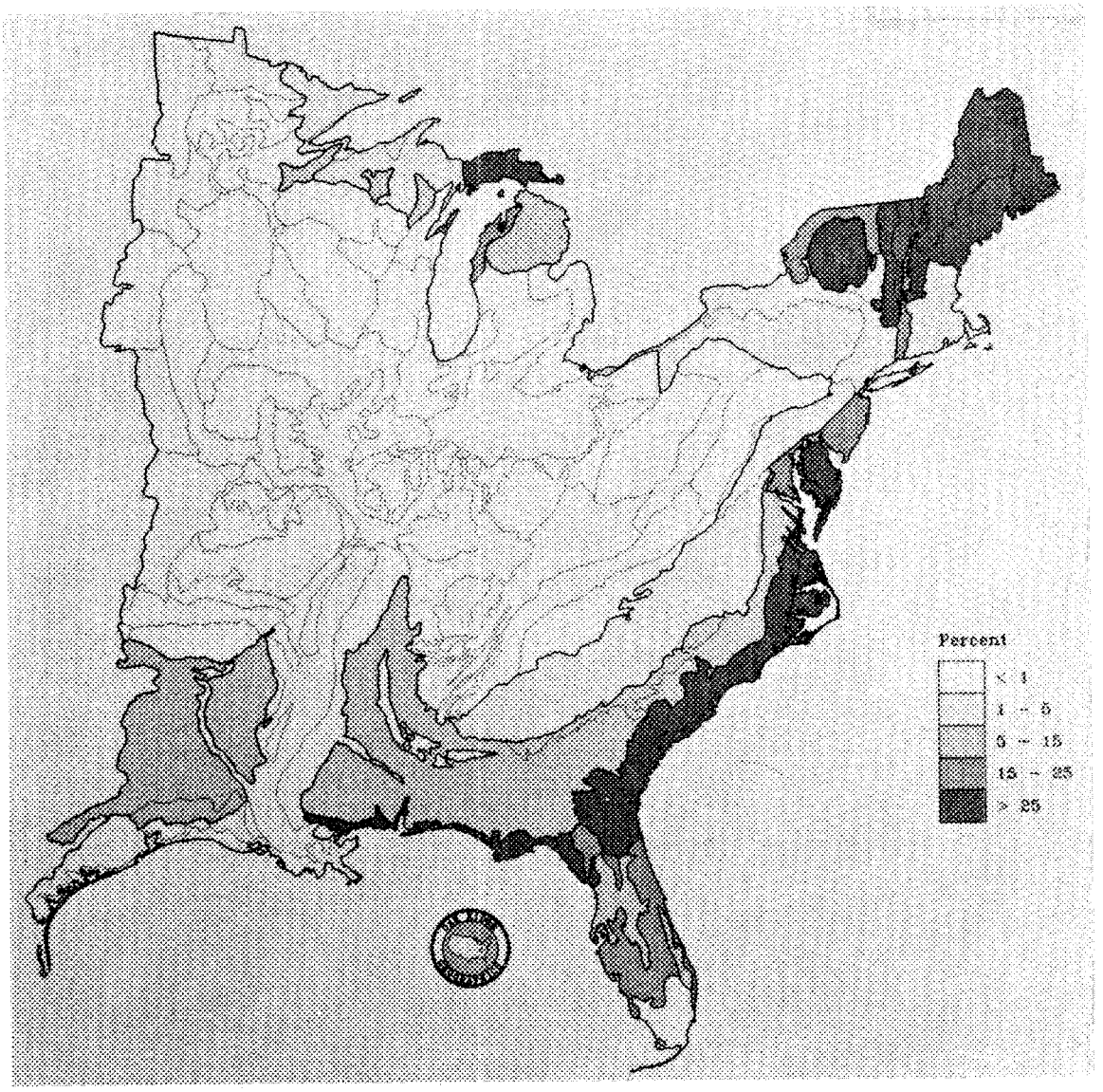

Fig. B-11 Percentage of area in Major Land Resource Areas of the eastern United States having forested surface soils with base saturation <20%.

ORNL-DWG 86-6952

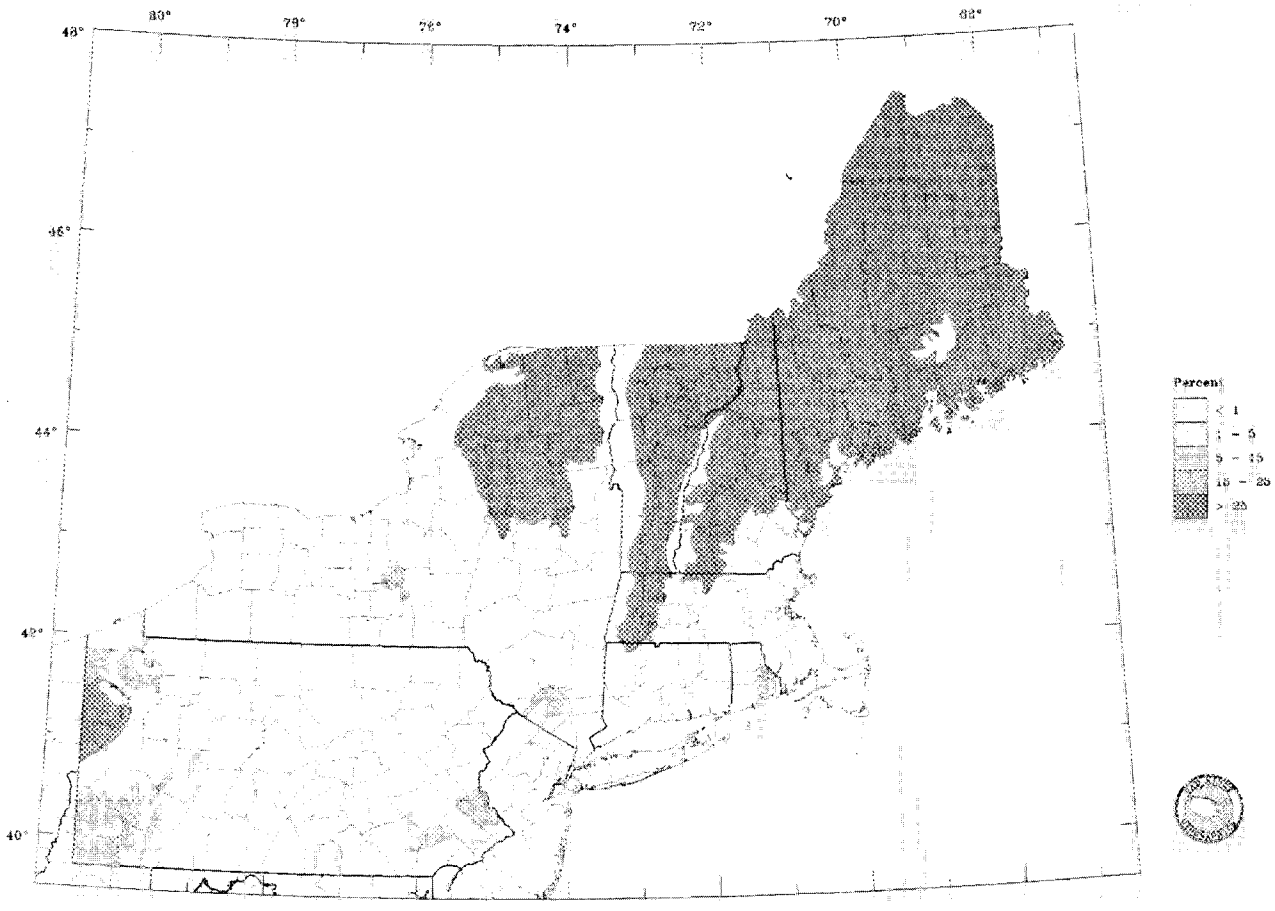

Fig. B-12 Percentage of forested surface soils in the northeastern United States with base saturation <20%.

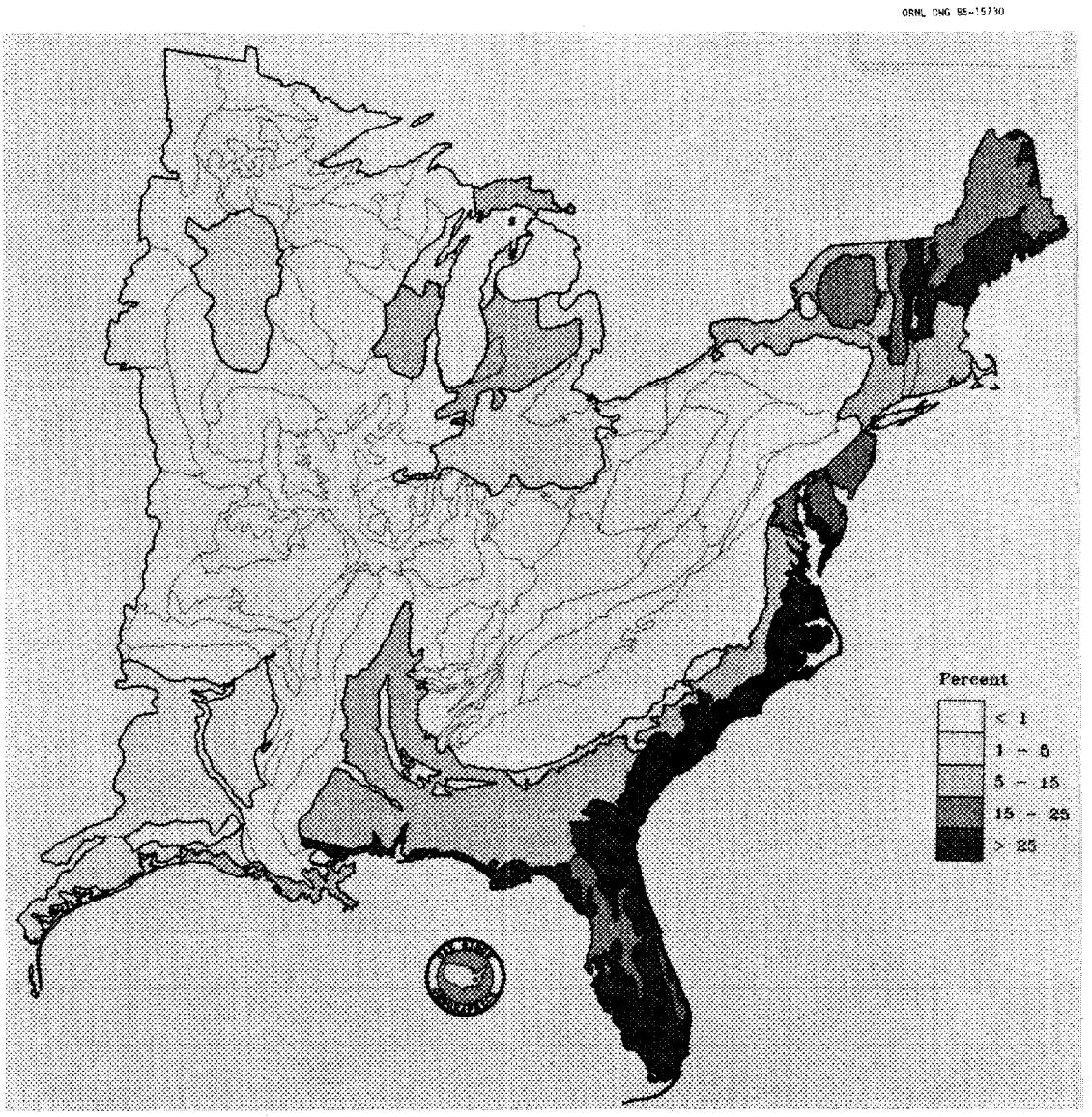

Fig. B-13 Percentage of area in Major Land Resource Areas of the eastern United States having forested surface soils with pH <4.5.

ORNL-DWG 86-6954

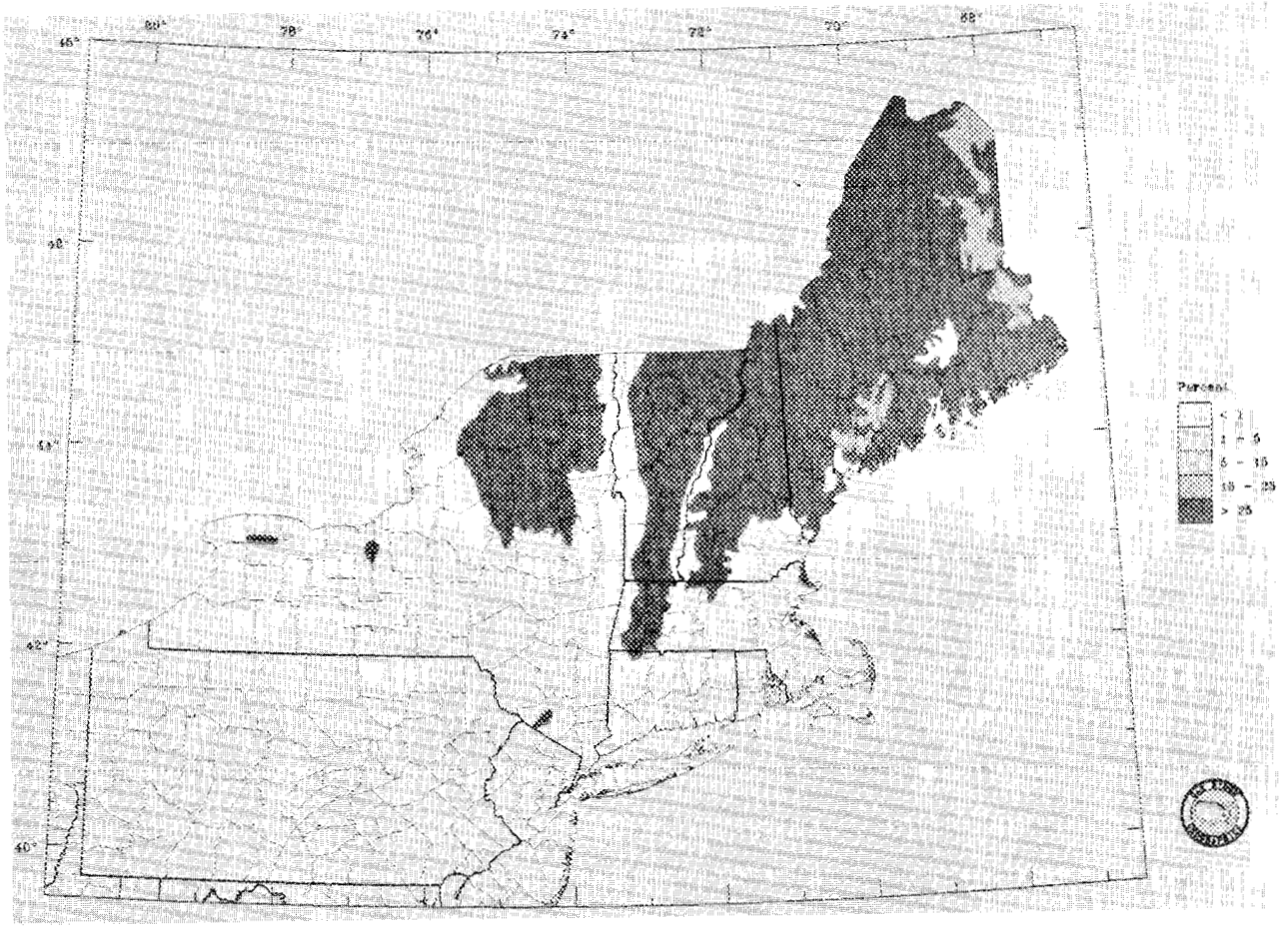

Fig. B-14 Percentage of forested surface soils in the northeastern United States with pH <4.5.

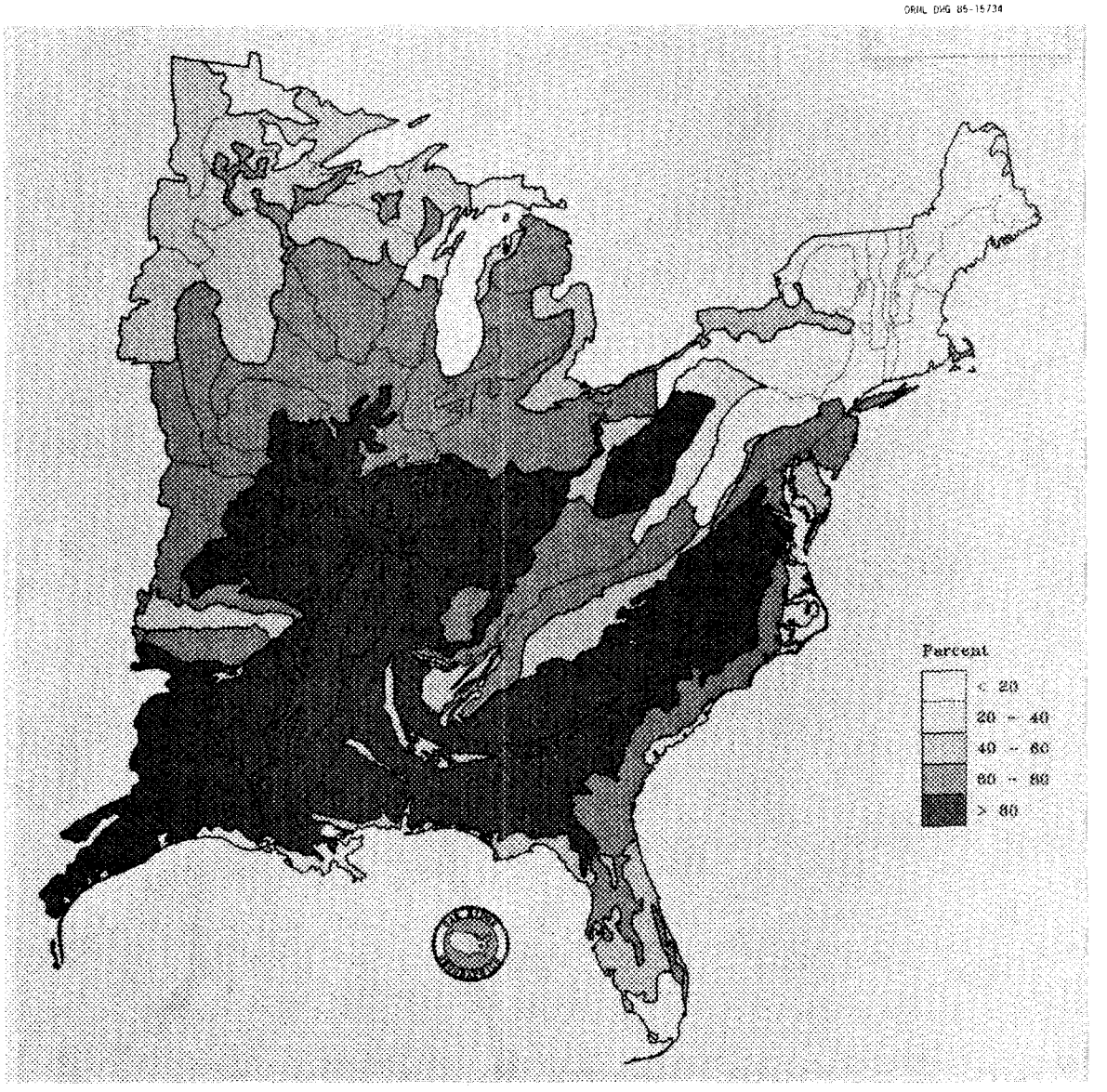

Fig. B-15 Percentage of area in Major Land Resource Areas of the eastern United States having forested surface soils with organic carbon <2%.

ORNL-DWG 86-6958

Fig. B-16 Percentage of forested surface soils in the northeastern United States with organic carbon <2%.

ORNL-DWG 86-6955

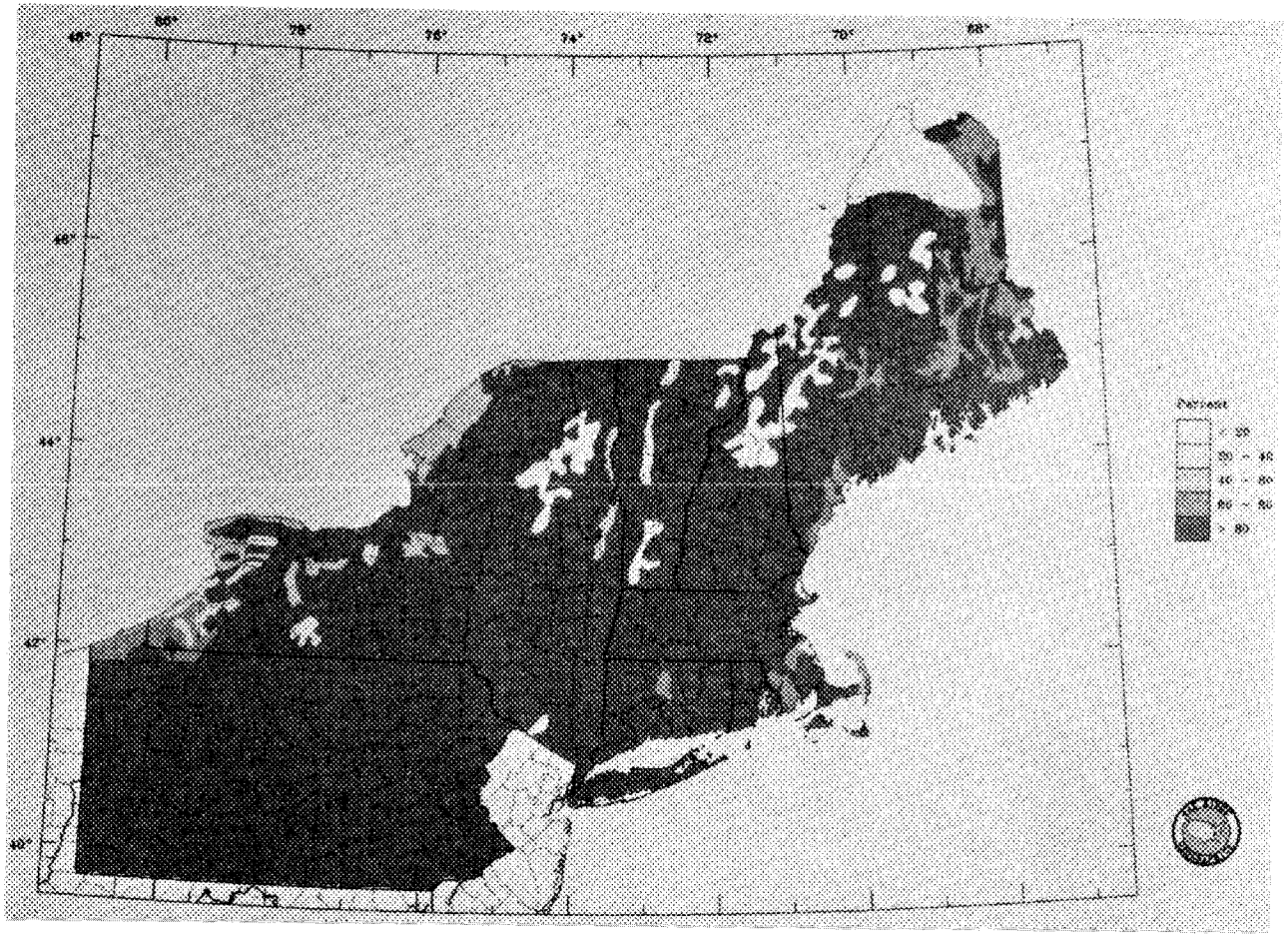

Fig. B-17 Percentage of forested surface soils in the northeastern United States with organic carbon <5%.

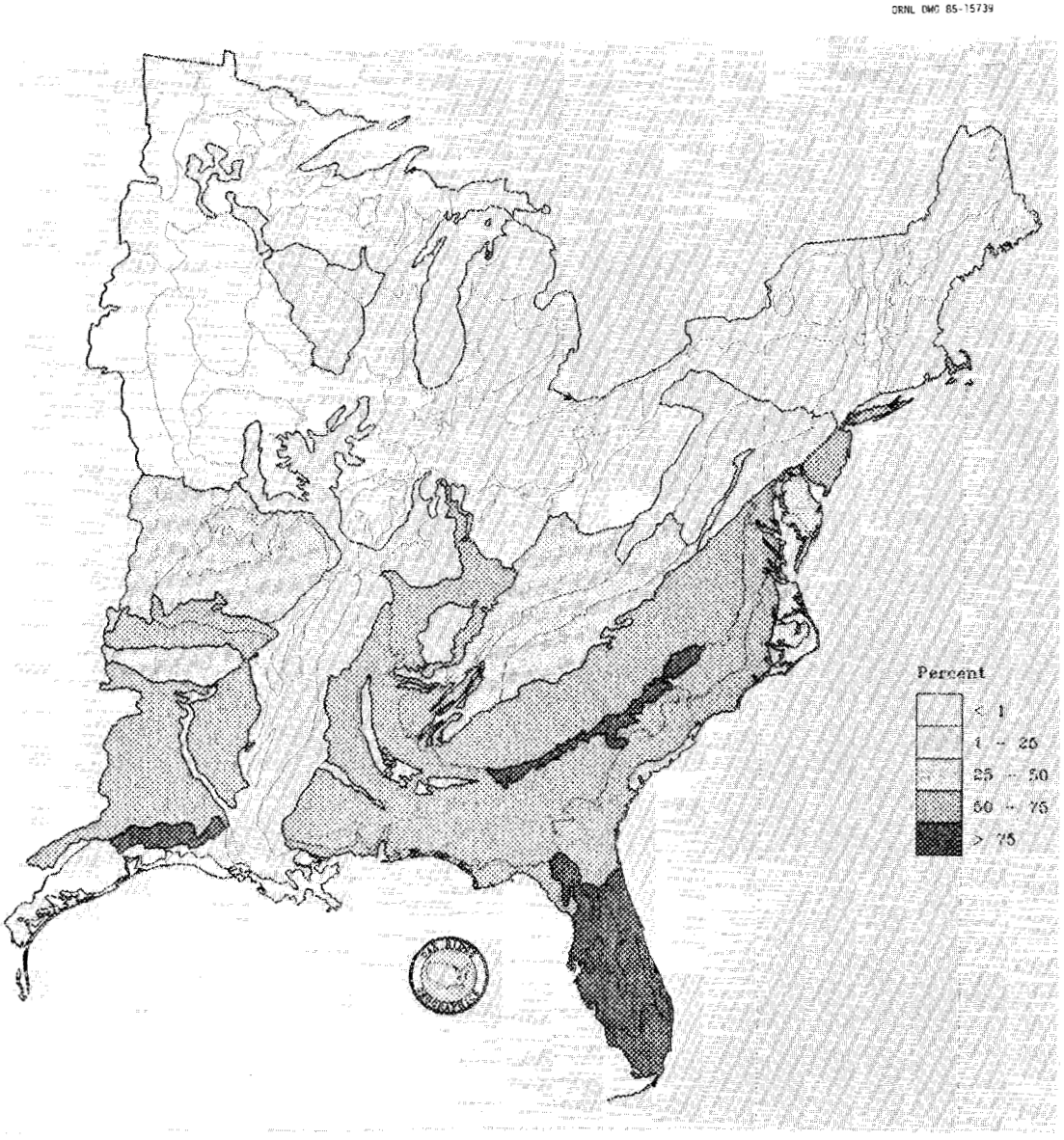

Fig. B-18 Percentage of area in Major Land Resource Areas of the eastern United States having forested soils with low weathering potential based on family mineralogy.

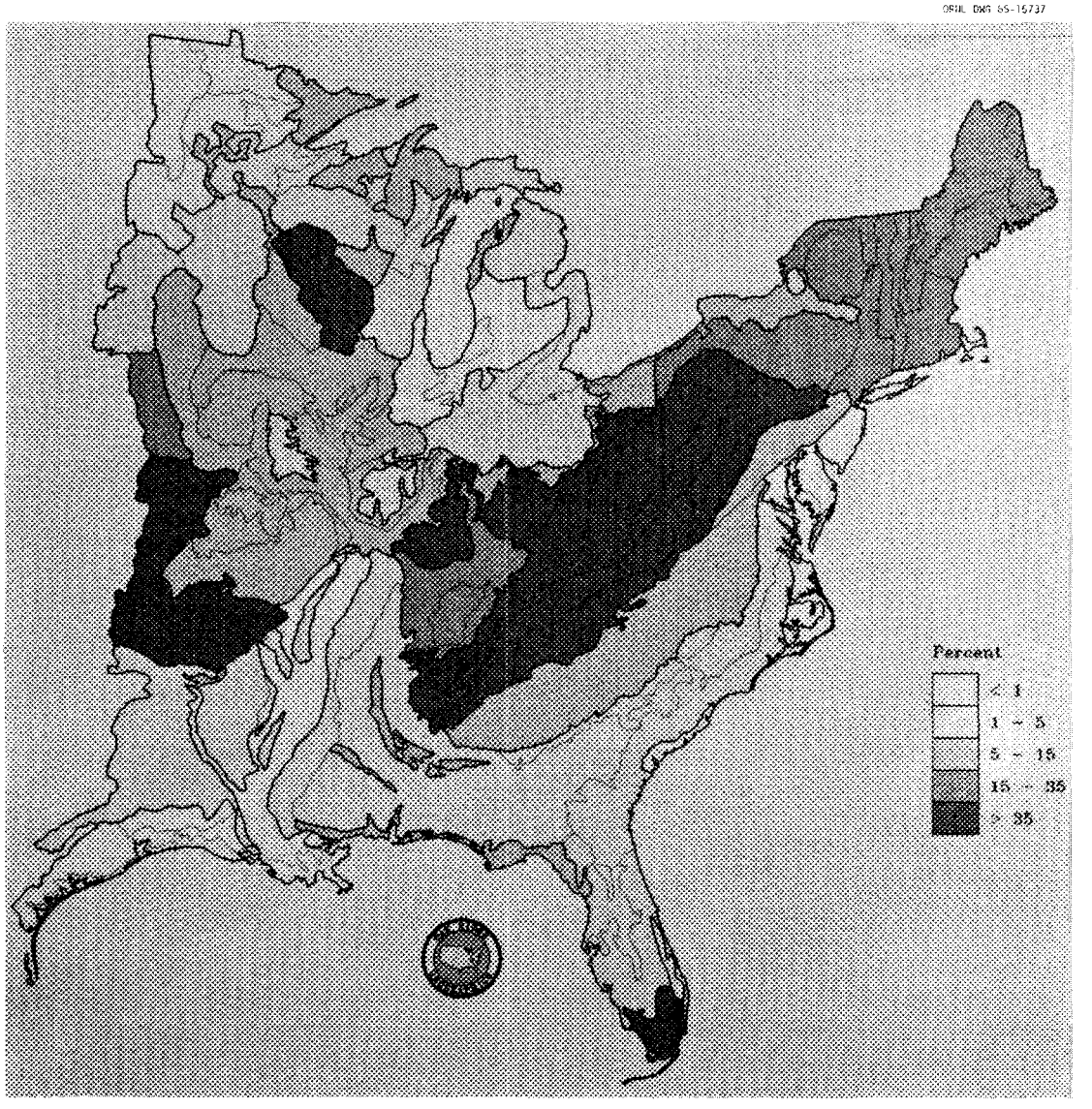

Fig. B-19 Percentage of area in Major Land Resource Areas of the eastern United States having forested soils with depth to bedrock to <50 cm.

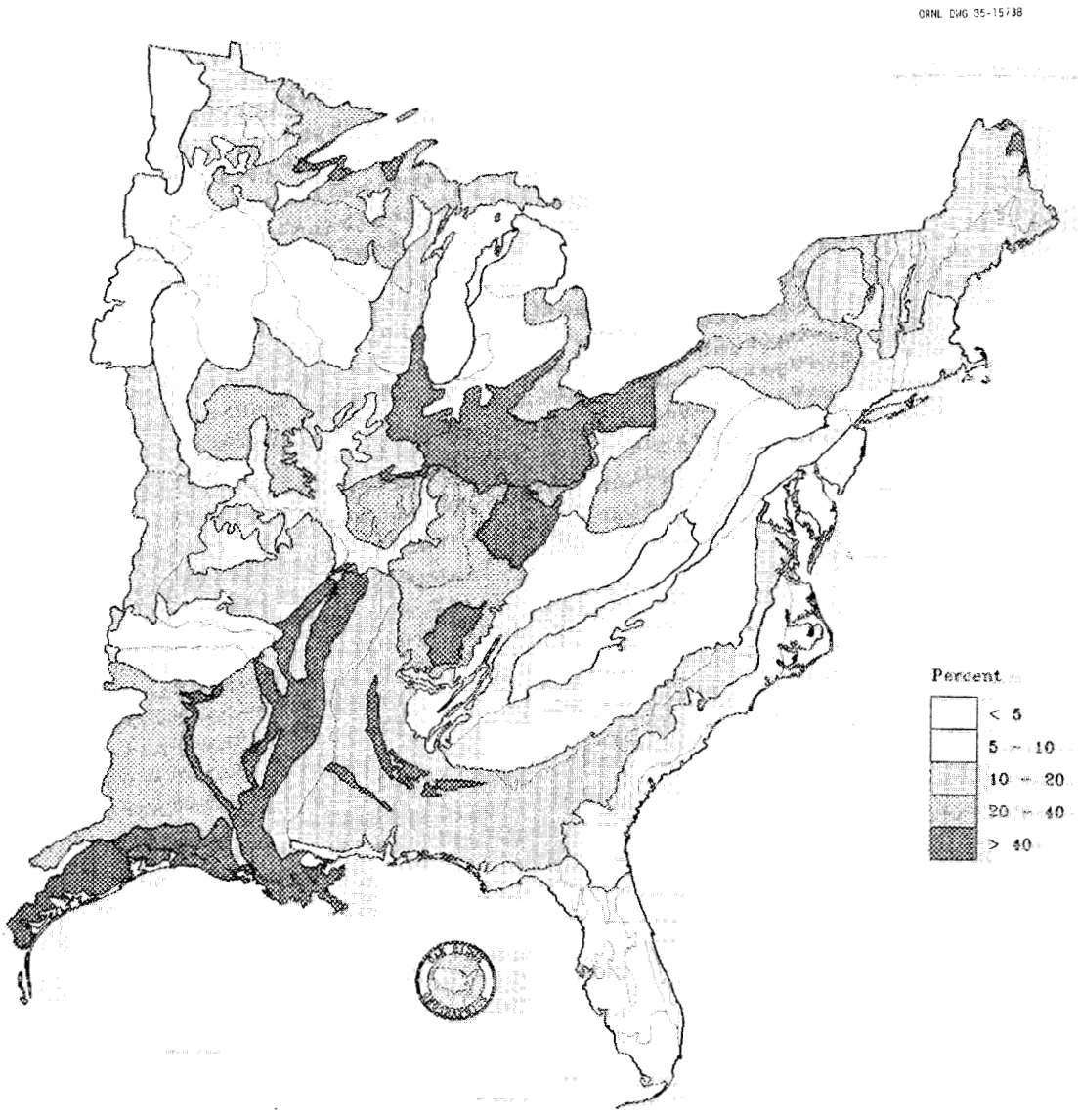

Fig. B-20 Percentage of area in Major Land Resource Areas of the eastern United States having forested soils with depth to a root restrictive layer <50 cm.

APPENDIX C. AREA HAVING SELECTED SOIL CHEMICAL CHARACTERISTICS, BY FOREST TYPE AND SOIL TAXONOMIC SUBGROUP, FOR SEVERAL REGIONS OF THE EASTERN UNITED STATES.

ORNL-DWG 86-6947

MLRA's of the Eastern U.S.

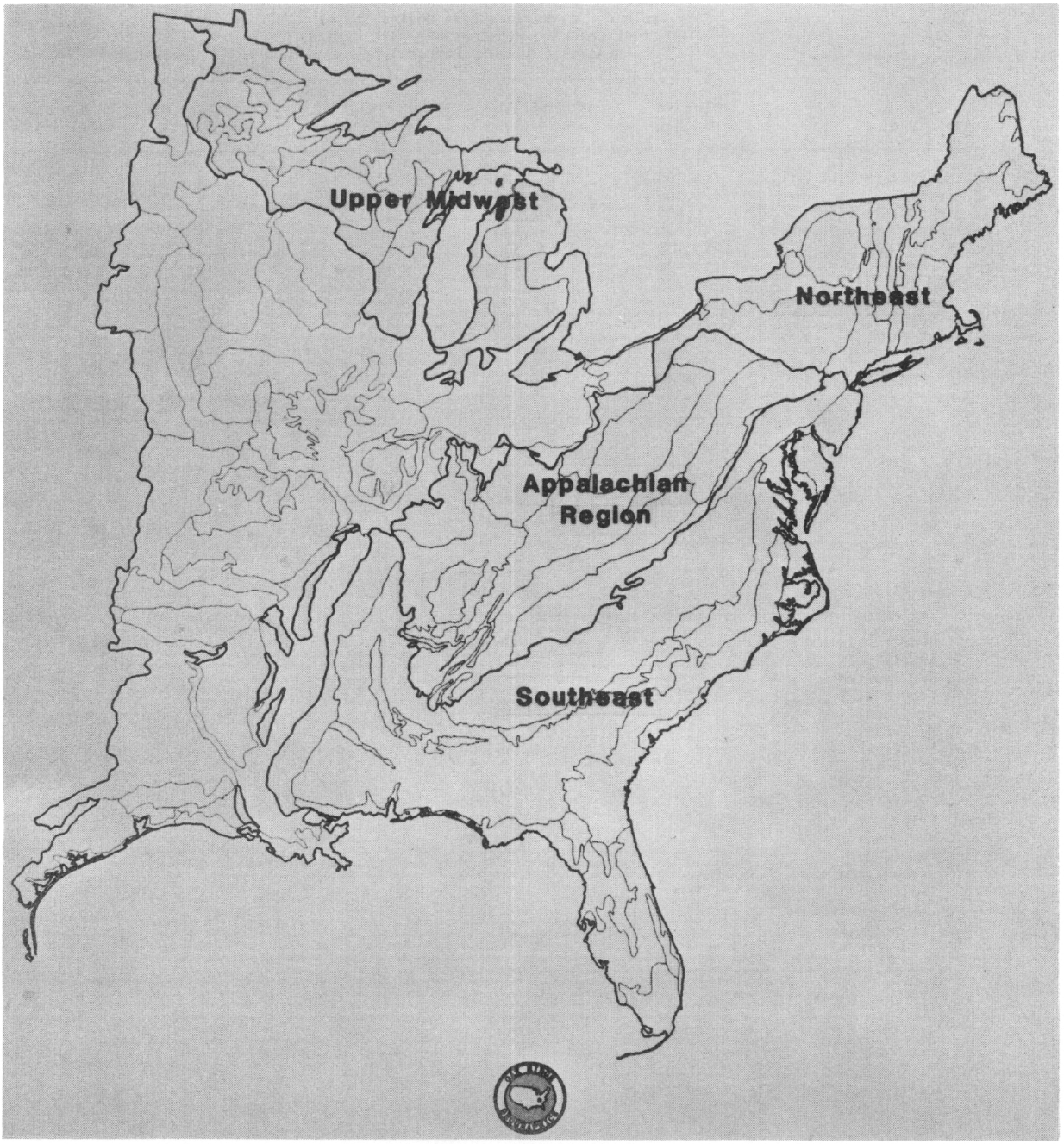

Fig. C-1 MLRAs in the Northeast, Appalachian Region, Southeast, and Upper Midwest that were used in calculating the regional summary statistics in this Appendix.

Table C-1.a Northeastern United States
Nutrient Leaching Hypothesis[a] - A/E Horizon
Sensitive Area by Forest Type

Forest Type	Sensitive area (ha)	Forest type area (ha)	Percentage of forest type	Percentage of total forest	Percentage of total region
White-Red-Jack Pine	666,700	2,433,600	27.4	3.3	2.1
Spruce-Fir	985,700	4,943,400	19.9	4.9	3.1
Longleaf-Slash Pine	800	2,300	34.8	0.0	0.0
Loblolly-Shortleaf Pine	12,700	62,400	20.4	0.1	0.0
Oak-Pine	441,100	1,183,300	37.3	2.2	1.4
Oak-Hickory	1,013,900	2,417,400	41.9	5.0	3.2
Oak-Gum-Cypress	200	13,000	1.5	0.0	0.0
Elm-Ash-Cottonwood	137,800	922,900	14.9	0.7	0.4
Maple-Beech-Birch	1,670,300	6,646,400	25.1	8.3	5.2
Aspen-Birch	222,300	1,126,300	19.7	1.1	0.7
Low Productivity	51,100	283,900	18.0	0.3	0.2
Nonstocked	16,100	89,700	18.0	0.1	0.1
	5,218,700	20,124,600		26.0	16.4

Table C-1.b Nutrient Leaching Hypothesis[a] - A/E Horizon
Sensitive Area by Soil Taxonomic Subgroup

Classification	Sensitive area (ha)
Typic Quartzipsamment	6,000
Typic Fragiaquept	163,500
Typic Dystrochrept	2,798,300
Typic Fragiaquult	2,700
Typic Fragiudult	42,800
Typic Hapludult	46,100
Aeric Fragiaquept	1,166,700
Aquentic Haplorthod	1,300
Aquic Fragiudalf	27,900
Aquic Quartzipsamment	200
Aquic Fragiochrept	4,100
Aquic Hapludult	700
Entic Haplaquod	38,800
Entic Fragiorthod	99,000
Entic Haplorthod	693,700
Entic Lithic Haplorthod	91,600
Humic Fragiaquept	35,300
	5,218,700

[a]Cation exchange capacity <15 cmol(+)/kg, BS 20-60%, pH >4.5

Table C-1.c Northeastern U.S.
Aluminum Toxicity Hypothesis* - A/E Horizon
Sensitive Area by Forest Type

Forest type	Sensitive area (ha)	Forest type area (ha)	Percentage of forest type	Percentage of total forest	Percentage of total region
White-Red-Jack Pine	200,900	2,433,600	8.3	1.0	0.6
Spruce-Fir	399,100	4,943,400	8.1	2.0	1.3
Oak-Pine	80,800	1,183,300	6.8	0.4	0.3
Oak-Hickory	58,000	2,417,400	2.4	0.3	0.2
Elm-Ash-Cottonwood	14,000	922,900	1.5	0.1	0.0
Maple-Beech-Birch	584,500	6,646,400	8.8	2.9	1.8
Aspen-Birch	109,800	1,126,300	9.8	0.6	0.3
Low Productivity	5,900	283,900	2.1	0.0	0.0
Nonstocked	600	89,700	0.7	0.0	0.0
	1,453,600	20,046,900		7.3	4.5

Table C-1.d Aluminum Toxicity Hypothesis* - A/E Horizon
Sensitive Area by Soil Taxonomic Subgroup

Classification	Sensitive area (ha)
Aeric Haplaquod	112,100
Humic Lithic Cryorthod	15,400
Lithic Cryorthent	9,400
Lithic Haplorthod	1,316,700
	1,453,600

*BS <20%, pH <4.5, Organic carbon <5%.

Table C-2.a Appalachian Region
Nutrient Leaching Hypothesis* - A/E Horizon
Sensitive Area by Forest Type

Forest type	Sensitive area (ha)	Forest type area (ha)	Percentage of forest type	Percentage of total forest	Percentage of total region
White-Red-Jack Pine	299,000	536,000	55.8	1.2	0.7
Spruce-Fir	14,800	32,400	45.7	0.1	0.0
Longleaf-Slash Pine	18,300	30,700	59.6	0.1	0.0
Loblolly-Shortleaf Pine	912,800	1,534,000	59.5	3.7	2.0
Oak-Pine	1,794,600	3,160,200	56.8	7.3	4.0
Oak-Hickory	9,738,200	16,808,500	57.9	39.8	21.8
Oak-Gum-Cypress	23,500	88,800	26.5	0.1	0.1
Elm-Ash-Cottonwood	73,500	283,900	25.9	0.3	0.2
Maple-Beech-Birch	964,500	1,571,900	61.4	3.9	2.2
Aspen-Birch	37,700	74,900	50.3	0.2	0.1
Low Productivity	40,000	138,500	28.9	0.2	0.1
Nonstocked	103,000	198,400	51.9	0.4	0.2
Larch	1,400	1,400	100.0	0.0	0.0
	14,021,300	24,459,600		57.3	31.4

Table C-2.b Nutrient Leaching Hypothesis* - A/E Horizon
Sensitive Area by Soil Taxonomic Subgroup

Classification	Sensitive Area (ha)
Typic Glossaqualf	400
Typic Quartzipsamment	15,500
Typic Fragiaquept	2,200
Typic Dystrochrept	5,031,700
Typic Fragiaquult	82,500
Typic Fragiudult	581,200
Typic Hapludult	6,795,700
Typic Rhodudult	54,100
Aeric Fragiaquept	5,300
Aquic Fragiudalf	45,700
Aquic Quartzipsamment	900
Aquic Hapludult	533,800
Aquic Paleudult	4,300
Entic Haplorthod	65,700
Fragiaquic Paleudult	3,400
Fragic Paleudult	37,100
Glossic Fragiudult	104,600
Humic Hapludult	64,100
Lithic Hapludult	258,700
Ochreptic Hapludult	185,500
Plinthic Paleudult	900
Psammentic Paleudult	2,200
Rhodic Paleudult	45,200
Ruptic-Lithic-Entic Hapludult	100,600
	14,021,300

*CEC <15 cmol (+)/kg, BS 20-60%, pH > 4.5

Table C-2.c Appalachian United States
Aluminum Toxicity Hypothesis* - A/E Horizon
Sensitive Area by Forest Type

Forest type	Sensitive area (ha)	Forest Type area (ha)	Percentage of forest type	Percentage of total forest	Percentage of total region
Oak-Hickory	500	16,808,500	0.0	0.0	0.0

Table C-2.d Aluminum Toxicity Hypothesis* - A/E Horizon
Sensitive Area by Soil Taxonomic Subgroup

Classification	Sensitive area (ha)
Arenic Paleaquult	500

*BS <20%, pH < 4.5, Organic carbon < 5%.

Table C-3.a Southeastern United States
Nutrient Leaching Hypothesis* - A/E Horizon
Sensitive Area by Forest Type

Forest type	Sensitive area (ha)	Forest Type area (ha)	Percentage of forest type	Percentage of total forest	Percentage of total region
White-Red-Jack Pine	40,100	50,100	80.0	0.1	0.1
Spruce-Fir	2,600	6,200	41.9	0.0	0.0
Longleaf-Slash Pine	2,927,700	5,469,300	53.5	6.9	3.5
Loblolly-Shortleaf Pine	6,525,500	10,804,900	60.4	15.3	7.9
Oak-Pine	6,478,700	10,612,900	61.1	15.2	7.8
Oak-Hickory	4,847,300	9,324,000	52.0	11.4	5.8
Oak-Gum-Cypress	329,100	4,920,500	6.7	0.8	0.4
Elm-Ash-Cottonwood	18,500	244,200	7.6	0.0	0.0
Maple-Beech-Birch	17,000	40,400	42.1	0.0	0.0
Aspen-Birch	900	8,600	10.5	0.0	0.0
Low Productivity	18,500	92,400	20.0	0.0	0.0
Tropical	1,300	12,300	10.6	0.0	0.0
Nonstocked	530,200	1,057,200	50.2	1.2	0.6
	21,737,400	42,643,000		50.9	26.1

Table C-3.b Nutrient Leaching Hypothesis* - A/E Horizon
Sensitive Area by Soil Taxonomic Subgroup

Classification	Sensitive area (ha)
Typic Glossaqualf	208,400
Typic Quartzipsamment	1,288,800
Typic Dystrochrept	466,300
Typic Haplohumod	76,800
Typic Fragiaquult	28,900
Typic Fragiudult	446,600
Typic Hapludult	11,149,600
Typic Rhodudult	470,600
Albic Glossic Natraqualf	35,100
Alfic Arenic Haplaquod	8,200
Aquic Fragiudalf	37,100
Aquic Quartzipsamment	248,900
Aquic Hapludult	999,800
Aquic Paleudult	463,300
Aquic Arenic Paleudult	143,800
Arenic Glossaqualf	58,800
Arenic Haplaquod	68,000
Arenic Haplohumod	20,200
Arenic Paleudult	506,600
Arenic Plinthaquic Paleudult	185,600
Arenic Ultic Haplaquod	3,200
Arenic Ultic Haplohumod	3,400
Entic Haplaquod	15,300
Entic Haplohumod	24,900
Fragiaquic Paleudult	198,300
Fragic Paleudult	3,300
Glossic Fragiudult	82,500
Grossarenic Haplaquod	49,600
Grossarenic Paleudult	1,598,900
Grossarenic Entic Haplohumod	56,300
Grossarenic Plinthic Paleudult	79,200
Haplaquodic Quartzipsamment	49,500
Humic Hapludult	1,000
Lithic Hapludult	40,000
Ochreptic Hapludult	205,600
Plinthic Paleudult	579,200
Psammentic Hapludult	104,600
Psammentic Paleudult	188,500
Rhodic Paleudult	362,800
Ruptic-Lithic-Entic Hapludult	29,800
Spodic Quartzipsamment	88,100
Spodic Paleudult	14,400
Ultic Haplaquod	607,900
Ultic Haplohumod	7,700
Vertic Paleudalf	413,900
Vertic Hapludult	18,100
	21,737,400

*CEC <15 cmol (+)/kg, BS 20-60%, pH >4.5.

Table C-3.c Southeastern United States
Aluminum Toxicity Hypothesis* - A/E Horizon
Sensitive Area by Forest Type

Forest Type	Sensitive area (ha)	Forest type area (ha)	Percentage of forest type	Percentage of total forest	Percentage of total region
White-Red-Jack Pine	1,300	50,100	2.6	0.0	0.0
Spruce-Fir	3,200	6,200	51.6	0.0	0.0
Longleaf-Slash Pine	1,173,800	5,469,300	21.5	2.8	1.4
Loblolly-Shortleaf Pine	820,700	10,804,900	7.6	1.9	1.0
Oak-Pine	801,700	10,612,900	7.6	1.9	1.0
Oak-Hickory	442,300	9,324,000	4.7	1.0	0.5
Oak-Gum-Cypress	604,600	4,920,500	12.3	1.4	0.7
Elm-Ash-Cottonwood	7,000	244,200	2.9	0.0	0.0
Low Productivity	10,900	92,400	11.8	0.0	0.0
Nonstocked	150,900	1,057,200	14.3	0.4	0.2
	4,016,400	42,581,700		9.4	4.8

Table C-3.d Aluminum Toxicity Hypothesis* - A/E Horizon
Sensitive Area by Soil Toxonomic Subgroup

Classification	Sensitive area (ha)
Typic Paleaquult	1,190,000
Aeric Haplaquod	699,400
Aeric Paleaquult	408,400
Arenic Paleaquult	657,700
Arenic Plinthic Paleaquult	187,600
Arenic Umbric Paleaquult	278,400
Grossarenic Paleaquult	319,100
Plinthic Paleaquult	70,700
Umbric Paleaquult	205,100
	4,016,400

*BS <20%, pH <4.5, Organic carbon <5%

Table C-4.a Upper Midwest
Nutrient Leaching Hypothesis* - A/E Horizon
Sensitive Area by Forest Type

Forest type	Sensitive area (ha)	Forest type area (ha)	Percentage of forest type	Percentage of total forest	Percentage of total region
White-Red-Jack Pine	531,200	1,674,800	31.7	3.3	1.4
Spruce-Fir	263,300	2,836,500	9.3	1.6	0.7
Oak-Pine	77,500	225,600	34.4	0.5	0.2
Oak-Hickory	414,200	1,923,200	21.5	2.6	1.1
Elm-Ash-Cottonwood	170,100	1,435,800	11.9	1.1	0.5
Maple-Beech-Birch	1,164,100	2,652,700	43.9	7.3	3.1
Aspen-Birch	1,478,400	5,003,000	29.6	9.2	3.9
Nonstocked	46,800	251,500	18.6	0.3	0.1
	4,145,600	16,003,100		25.9	11.0

Table C-4.b Nutrient Leaching Hypothesis* - A/E Horizon
Sensitive Area by Soil Taxonomic Subgroup

Classification	Sensitive area (ha)
Typic Glossaqualf	56,200
Typic Quartzipsamment	62,000
Typic Dystrochrept	220,900
Aeric Glossaqualf	114,100
Alfic Fragiorthod	610,900
Alfic Haplorthod	856,000
Aqualfic Haplorthod	68,800
Aquic Glossoboralf	175,900
Aquic Fragiochrept	65,700
Aquic Arenic Hapludalf	16,800
Arenic Glossaqualf	15,300
Entic Haplaquod	302,700
Entic Haplorthod	1,526,800
Entic Lithic Haplorthod	50,700
Lithic Glossoboralf	2,800
	4,145,600

*CEC <15 cmol (+)/kg, BS 20-60%, pH >4.5

Table C-4.c Upper Midwestern United States
Aluminum Toxicity Hypothesis* - A/E Horizon
Sensitive Area by Forest Type

Forest Type	Sensitive area (ha)	Forest type area (ha)	Percentage of forest type	Percentage of total forest	Percentage of total region
White-Red-Jack Pine	3,800	1,674,800	0.2	0.0	0.0
Spruce-Fir	4,600	2,836,500	0.2	0.0	0.0
Elm-Ash-Cottonwood	300	1,435,800	0.0	0.0	0.0
Maple-Beech-Birch	1,300	2,652,700	0.1	0.0	0.0
Aspen-Birch	2,800	5,003,000	0.1	0.0	0.0
Nonstocked	1,400	251,500	0.6	0.0	0.0
	14,200	13,854,300		0.0	0.0

Table C-4.d Nutrient Leaching Hypothesis* - A/E Horizon
Sensitive Area by Soil Taxonomic Subgroup

Classification	Sensitive area (ha)
Aeric Haplaquod	14,200

*BS <20%, pH <4.5, Organic carbon <5%

Table C-5.a Eastern United States
Nutrient Leaching Hypothesis* - A/E Horizon
Sensitive Area by Forest Type

Forest Type	Sensitive area (ha)	Forest type area (ha)	Percentage of forest type	Percentage of total forest	Percentage of total region
White-Red-Jack Pine	1,550,800	4,785,400	32.4	1.2	0.5
Spruce-Fir	1,266,400	7,824,200	16.2	1.0	0.4
Longleaf-Slash Pine	3,190,200	5,931,100	53.8	2.5	1.0
Loblolly-Shortleaf Pine	9,736,700	16,155,800	60.3	7.6	3.0
Oak-Pine	11,909,300	20,814,900	57.2	9.2	3.7
Oak-Hickory	17,987,700	42,049,100	42.8	13.9	5.6
Oak-Gum-Cypress	559,500	7,221,900	7.8	0.4	0.2
Elm-Ash-Cottonwood	425,000	4,125,500	10.3	0.3	0.1
Maple-Beech-Birch	3,827,600	11,088,500	34.5	3.0	1.2
Aspen-Birch	1,742,300	6,377,900	27.3	1.4	0.5
Low Productivity	144,900	607,000	23.9	0.1	0.1
Tropical	1,300	12,300	10.6	0.0	0.0
Nonstocked	887,100	2,023,900	43.8	0.7	0.3
Larch	1,400	1,400	100.0	0.0	0.0
	53,230,200	129,018,900		41.3	16.6

Table C-5.b Nutrient Leaching Hypothesis* - A/E Horizon
Sensitive Area by Soil Taxonomic Subgroup

Classification	Sensitive area (ha)
Typic Glossaqualf	1,612,200
Typic Quartzipsamment	1,483,400
Typic Fragiaquept	166,100
Typic Dystrochrept	8,782,300
Typic Haplohumod	76,800
Typic Fragiaquult	134,600
Typic Fragiudult	1,431,200
Typic Hapludult	21,149,100
Typic Rhodudult	524,700
Aeric Glossaqualf	147,200
Aeric Fragiaquept	1,175,100
Albic Glossic Natraqualf	48,200
Alfic Fragiorthod	610,900
Alfic Haplorthod	856,000
Alfic Arenic Haplaquod	8,200
Aqualfic Haplorthod	69,400
Aquentic Haplorthod	1,300
Aquic Glossoboralf	176,600
Aquic Fragiudalf	191,300
Aquic Quartzipsamment	250,000
Aquic Fragiochrept	69,800
Aquic Hapludult	2,125,200
Aquic Paleudult	808,900
Aquic Arenic Hapludalf	18,300
Aquic Arenic Paleudult	143,800
Arenic Glossaqualf	74,100
Arenic Haplaquod	68,000
Arenic Haplohumod	20,200
Arenic Paleudult	674,400
Arenic Plinthaquic Paleudult	227,200
Arenic Ultic Haplaquod	3,200
Arenic Ultic Haplohumod	3,400
Entic Haplaquod	356,800
Entic Haplohumod	24,900
Entic Fragiorthod	99,000
Entic Haplorthod	2,286,200
Entic Lithic Haplorthod	142,300
Fragiaquic Paleudult	246,200
Fragic Paleudult	40,400
Glossic Fragiudult	323,000
Grossarenic Haplaquod	49,600
Grossarenic Paleudult	1,752,200
Grossarenic Entic Haplohumod	56,300
Grossarenic Plinthic Paleudult	85,900
Haplaquodic Quartzipsamment	49,500
Humic Fragiaquept	35,300
Humic Hapludult	65,900
Lithic Glossoboralf	2,800
Lithic Hapludult	546,300
Ochreptic Hapludult	391,100
Plinthic Paleudult	1,161,600
Psammentic Haludult	104,600
Psammentic Paleudult	247,500
Rhodic Paleudult	408,800
Ruptic-Lithic-Entic Hapludult	130,400
Spodic Quartzipsamment	88,100
Spodic Paleudult	14,400
Ultic Haplaquod	607,900
Ultic Haplohumod	7,700
Vertic Paleudult	756,300
Vertic Hapludult	18,100
	53,230,200

*CEC <15 cmol(+)/kg, BS 20-60%, pH > 4.5

Table C-5.c Eastern United States
Aluminum Toxicity Hypothesis* - A/E Horizon
Sensitive Area By Forest Type

Forest type	Sensitive area (ha)	Forest type area (ha)	Percentage of forest type	Percentage of Total forest	Percentage of Total region
White-Red-Jack Pine	206,000	4,785,400	4.3	0.2	0.1
Spruce-Fir	406,900	7,824,200	5.2	0.3	0.1
Longleaf-Slash Pine	1,174,200	5,931,100	19.8	0.9	0.4
Loblolly-Shortleaf Pine	865,100	16,155,800	5.4	0.7	0.3
Oak-Pine	918,800	20,814,900	4.4	0.7	0.3
Oak-Hickory	509,800	42,049,100	1.2	0.4	0.2
Oak-Gum-Cypress	606,700	7,221,900	8.4	0.5	0.2
Elm-Ash-Cottonwood	21,300	4,125,500	0.5	0.0	0.0
Maple-Beech-Birch	585,800	11,088,500	5.3	0.5	0.0
Aspen-Birch	112,600	6,377,900	1.8	0.1	0.0
Low Productivity	16,800	607,000	2.8	0.0	0.0
Nonstocked	153,900	2,023,900	7.6	0.1	0.1
	5,577,900	129,005,200		4.4	1.7

Table C-5.d Nutrient Leaching Hypothesis* - A/E Horizon
Sensitive Area by Soil Taxonomic Subgroup

Classification	Sensitive area (ha)
Typic Paleaquult	1,279,500
Aeric Haplaquod	825,700
Aeric Paleaquult	408,400
Arenic Paleaquult	661,900
Arenic Plinthic Paleaquult	187,600
Arenic Umbric Paleaquult	278,400
Grossarenic Paleaquult	319,100
Humic Lithic Cryorthod	15,400
Lithic Cryorthent	9,400
Lithic Haplorthod	1,316,700
Plinthic Paleaquult	70,700
Umbric Paleaquult	205,100
	5,577,900

*BS <20%, pH <4.5, organic carbon <5%

APPENDIX D. EXAMPLE CALCULATION OF WORST-CASE LEACHING RATES AND CHANGES.

APPENDIX D

Calculation of Worst-Case Leaching Rates and Changes

A simple calculation of worst case leaching rates and changes in soil nutrient status can be made. This is a worst-case example because it assumes high acidic deposition, no base cation deposition with the acidic deposition, no weathering replacement of base cations, no cycling of base cations from depth by the vegetation, and no anion retention in the soil. It underestimates soil nutrient losses, however, in that it assumes no accumulation in vegetation.

Assume:

(1) 1 kmol/ha/yr acidic deposition,
(2) A typical "sensitive" soil with CEC = 10 cmol(+)/kg = 0.1 mol(+)/kg, BS = 50%, and bulk density = 1.3 g/cc = 1.3 x 10^3 kg/m^3.

$$\text{Exchangeable bases/ha} = 0.50 \times \frac{0.1\ \text{mol(+)}}{\text{kg soil}} \times \frac{1.3 \times 10^3\ \text{kg soil}}{m^3} \times \frac{10^4 m^2}{\text{ha}}$$

$$\times \frac{1\ \text{kmol}}{10^3 \text{mol}} \times .25\text{m of soil.}$$

$$= 160\ \text{kmol/ha (in top 25 cm of soil).}$$

Worst-case leaching rate = acidic deposition rate = 1 kmol/ha/yr
= 0.63% of soil exchangeable bases per year.

In 30 years, approximately 20% of the pool of exchangeable bases could be leached, reducing base saturation to 40%. In 80 years 50% of the pool could be leached. It is unknown what magnitude of decline in available nutrients will affect forest productivity.

In actuality, the assumptions listed above are never met. To calculate soil nutrient changes in a forested stand over time, a model

would have to account for, and better quantify, at least the following processes:

1. Deposition inputs: strong and weak acid anions; <0.1 - >1 kmol(-)/ha/yr; base cations; <0.1 - 0.8 kmol(+)/ha/yr.

2. Weathering replacement of base cations within the soil profile: 0 - >1 kmol(+)/ha/yr.

3. Cycling of base cations from depth by the vegetation (deep weathering): ? kmol(+)/ha/yr.

4. Accumulation of base cations by vegetation: 0.2 - 1.6 kmol(+)/ha/yr.

5. Anion (and cation) retention in or leaching from the soil: 0 - >2 kmol (±)/ha/yr.

6. Changes in the above parameters over the life of the stand.

Clearly the output of such a model would be highly uncertain, and few or no experimental data are available to verify it.

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