the Acid Deposition Research Program

BIOPHYSICAL RESEARCH

THE EFFECTS OF ACID DEPOSITION ON FORESTS

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February, 1987

PRIME RESEARCH CONTRACTOR:
The Kananaskis Centre for Environmental Research
The University of Calgary
Calgary, Alberta, Canada

Acid Deposition Research Program

The Acid Deposition Research Program is funded and administered by the Province of Alberta, the Canadian Petroleum Association, Alberta's electrical utilities and the Energy Resources Conservation Board.

A distinctive feature of the ADRP is the development and funding of research in two major areas, biophysical and human health.

Biophysical Research Prime Contractor:

Kananaskis Centre for Environmental Research
The University of Calgary

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prepared for:

Kananaskis Centre for Environmental Research
The University of Calgary
Calgary, Alberta T2N 1N4

for submission to:

The Acid Deposition Research Program
1500, 633 Sixth Avenue S.W.
Calgary, Alberta, Canada, T2P 2Y5

February 1987

This publication may be cited as:

Mayo, J.M., 1987.
The Effects of Acid Deposition on Forests. Prep for the Acid Deposition Research Program by the Department of Biology, Emporia State University. ADRP-B-09/87. 80 pp.

ISBN 0-921625-11-1 (Volume 9)

ISBN 0-921625-02-2 (Set of 11)

EXECUTIVE SUMMARY

There is a general decline in forest productivity and health over much of Europe, the eastern United States, and eastern Canada. The problem is particularly acute in West Germany where forests are plainly dying. There has been a vast and bewildering array of meetings, symposia, and research papers produced on this subject and there are a number of hypotheses as to the exact cause of the decline. Regardless of the proposed causes, they all centre upon pollutants produced in our industrial society. One is tempted to draw an analogy with the effects of tobacco upon human health; there is an obvious relationship, but there are so many effects that the exact sequence of events leading to deterioration is difficult to unravel. Nevertheless, the following hypotheses concerning forest decline have been proposed with considerable supporting data:

1. Direct effects. The direct effects of pollutants, acid forming or otherwise, have been proposed. It is well established that ozone damages membranes and inhibits photosynthesis. This has been shown for pine in California, and ozone levels in parts of Germany can reach quite high levels (e.g., levels as high as 788 $\mu g\ m^{-3}\ h^{-1}$). It has also been clearly shown that SO_2 can cause partial stomatal closure and reduce photosynthesis, and chlorophyll reduction has been proven. Thus, it seems quite possible that the cause of forest decline could be through reduced photosynthesis. Recently, chloroethenes have been implicated as a direct cause of forest decline in Germany. It is also possible that cuticular damage and nutrient leaching from leaves could be due to acid deposition.
2. Indirect effects due to soil acidification.
   a. Soil acidification can result in base cation leaching and increased availability of potentially toxic aluminum and manganese. It has been shown that in some areas such as Solling, West Germany, this may be the case. Soils potentially sensitive to acidification are noncalcareous and have low cation exchange capacity, low percent base saturation, and are near neutral in reaction (pH 5 to 6).
   b. Soil acidification may reduce nitrogen fixation because the infection process that produces the symbiosis between plant and bacterium is inhibited by low pH. Thus, the nitrogen budget may be altered. It is interesting that one hypothesis holds that nitrogen fertilization by ammonia is the cause of forest decline.
   c. Soil acidification may decrease decomposition and reduce nitrification. If this happens, nutrient deficiencies may result because of reduced nutrient cycling. The literature on the effects of soil acidification on nitrification seems equivocal. Some references say that nitrification is decreased and some state that there is considerable nitrification at low pH. This needs clarification.

There are a number of possible ways in which soil acidification could cause forest decline. It is possible that there are multiple causes. It also must be emphasized that there are several soil scientists who feel that pollution-caused soil acidification is not likely to be a problem except in special cases. They question the general hypothesis of acid deposition-caused soil acidification except near point sources of pollution or perhaps due to nitrogen fertilizers used in farm practices (e.g., Krug and Frink 1983; Tabatabai 1985).

3. Land use change and secondary succession have been proposed as major causes of soil acidification. There are several authors who feel that secondary succession following farm abandonment is the cause of much soil acidification. It is well established that secondary succession leading to a coniferous forest does result in lowered soil pH. This has been proposed as a cause of soil acidification in Norway and in the north-eastern US.
4. Fertilization. It has been proposed that fertilization, particularly nitrogen fertilization by NOx and NH_3, may have stimulated forest growth leading to imbalances in other nutrients. Stresses such as failure to become cold hardy and reduced pest resistance then lead to forest decline. It seems clear that in agricultural situations, nitrogenous fertilization has led to soil acidification. Whether or not this is happening in the forests is not clear, but it seems unlikely.

It seems likely that some combination of these various effects will prove to be the cause of forest decline in different situations and that there will not be one general cause. The effects of acid deposition may be very site specific, requiring different mitigative treatments.

The combined effects of acid deposition and forest harvest have been studied and it appears that the recent forestry practice of whole tree harvest on a shortened rotation, plus the potential for acid leaching of base cations, may prove very damaging to the survival of forests. If cations (Ca, Mg, K) are removed rapidly, then the soil will acidify. This, coupled with erosion losses, may result in forest decline. It seems likely that forests will not be able to sustain some of the proposed biomass-for-energy management much discussed during the oil shortages of the 1970's. Forest soils simply do not generate cations rapidly enough through weathering.

Acid deposition is a real phenomenon in Alberta, although it is not as apparent as in Quebec and Ontario. Forests are being affected near point sources such as sour gas plants and the Alberta oil sands extraction plants. Some soils in Alberta are also potentially susceptible to negative acid deposition impacts because of their noncalcareous nature, or low cation exchange capacity. These areas of the province should receive further study. It is important that integrated studies be carried out to determine the potential for serious forest decline before it occurs.

Major goals should be:

1. To understand, more fully, forest ecosystem processes in unpolluted areas. It is essential to understand the basic processes of: canopy exchange; photosynthesis, carbon allocation and growth; reproductive biology; soil microbial processes, including decomposition, nitrogen fixation, nitrification, denitrification, and the role of mycorrhizae; nutrient cycling, basic weathering rates; and effects of community changes upon soil.
2. To distinguish the direct and indirect effects of pollutants, especially sulphur gases, upon the forest ecosystem processes described above. It is important that sites for these studies be carefully selected to include sensitive soils likely to show the effects of acid deposition.
3. To separate and determine the relative contributions to soil acidification of: industrial pollution such as that emitted by sour gas plants; and non-point source agricultural inputs such as ammonium fertilizers. The selection of a site for this type of study is very important in that it requires the presence of an appropriate industrial activity adjacent to a farmed, sensitive soil, all of which are within a forested area.

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ACKNOWLEDGEMENTS

This literature review was funded by the Acid Deposition Research Program. I wish to thank the staff at the Kananaskis Centre who were so helpful and patient with the author. Special thanks are given to Dr. Allan Legge for his encouragement and for the use of his extensive library of environmental documents. I would like to acknowledge Julie Lockhart as being an invaluable aid in researching and editing this review, and Della Patton for her patience in typing the manuscript.

Dr. Joseph Shinn and Ms. Margaret Torn of The Lawrence Livermore National Laboratory, Livermore, California, searched the Department of Energy (DOE) data base for relevant literature, and their help is acknowledged here.

1. INTRODUCTION

The purpose of this paper is to review the recent literature concerned with acid deposition effects upon forests. The world-wide literature is reviewed and its relevance to the Alberta situation discussed.

The discussion is confined to current material (late 1970's to present), using the older literature as necessary to complete the information in selected subject areas. There are several recent, excellent reviews that discuss much of the earlier work, and these will be referred to as needed (Krug and Frink 1983; Morrison 1984; and McLaughlin 1985).

Some of the topics in this report, such as effects on soils and microorganisms, and numerical models of air pollutant exposure and vegetation response are only cursorily covered because they are being reviewed in depth by other members of ADRP (Turchenek at al. 1987; Visser et al. 1987; Krupa and Kickert 1987a).

Literature data bases searched include: Biological Abstracts, Forestry Abstracts, the US Department of Energy (DOE), and the Kananaskis Centre Library. A bibliography of more than 500 references was assembled and reviewed.

The effects of ozone on forests have been included even though ozone is not an acid former. This was done because recent articles have suggested that it may be involved with the problems in European forests, either as the primary pollutant or as a contributing factor (Skärby and Sellden 1984).

1.1 BIOGEOCHEMICAL CYCLES

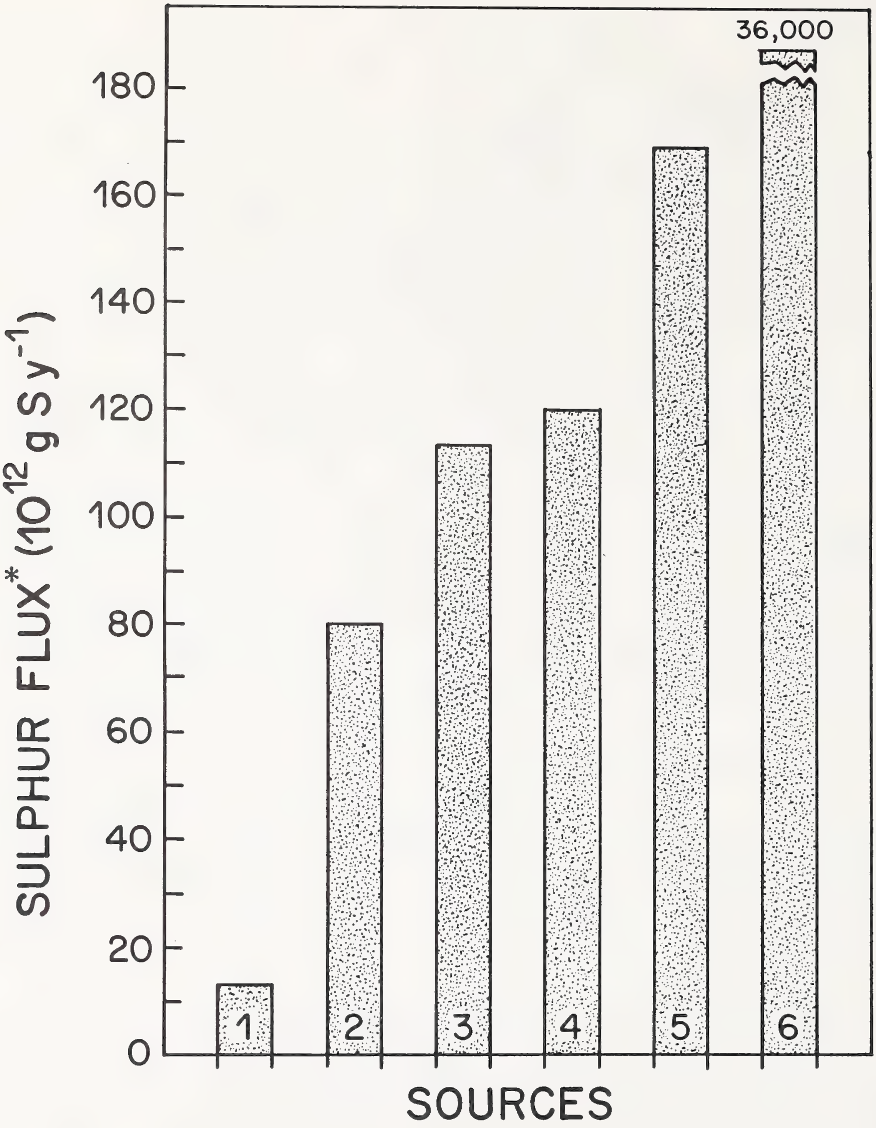
Carbon, nitrogen, and sulphur have naturally occurring world-wide cycles involving terrestrial ecosystems, the atmosphere, and oceans. These are only poorly understood (Smith 1981). Very large amounts of compounds of these elements are cycled by natural processes. Anthropogenic effects must be evaluated as augmentations to these natural sources.

1.1.1 Sulphur Cycle

The global sulphur cycle has received considerable attention in recent years and the reader is advised to see the reviews edited by Ivanov and Freney (1983), and Bolin and Cook (1983). Smith (1981) has summarized various estimates of biogenic release of hydrogen sulphide ranging from 58 to 110 x 10^6 T y^{-1}. He estimated global rates of S transfer to be as follows: biogenic (H_2S), 90 x 10^6 T y^{-1}; sea spray 43 x 10^6 T y^{-1}; and human activity, 50 x 10^6 T y^{-1}. Human activities thus account for a considerable fraction of the total.

Figure 1 summarizes various estimates of S fluxes. The estimates of anthropogenic S flux to the atmosphere tend to agree (80 to 113 Tg S y^{-1}). As can be seen, the anthropogenic flux has changed considerably since 1900 (Smith 1981; Ivanov 1983; and Ryaboshapko 1983). McLaughlin (1985) has estimated that SO_2 emissions in the mideastern US have increased seven-fold between 1900 and 1980. His estimates appear to be in agreement with those shown in Figure 1. The amounts of S extracted by mining (169 Tg S y^{-1}), as shown in Figure 1, constitute an ever increasing potential for increased release of S to the atmosphere.

Estimates of biogenic S flux to the atmosphere do not agree. Ivanov (1983) estimated it to be 25 Tg S y^{-1} whereas Smith suggested that it is approximately



* 10^{12} g = Tg

1. Anthropogenic emissions, 1900 A.D. (Ryaboshapko 1983)
2. Anthropogenic emissions, current (Smith 1981)
3. Anthropogenic emissions, current (Ivanov 1983)
4. Weathering (Ivanov 1983)
5. Extracted by mining activities (Ivanov 1983)
6. Release from forest soils (Smith 1981)

Figure 1. Sulphur fluxes from various sources compared to forest soils.

36,000 Tg S y^{-1} (Figure 1). The latter estimate is, in this author's opinion, too high. Smith himself calls the figure only a "guesstimate", and perhaps this illustrates the general lack of knowledge.

Regardless of the relative importance of anthropogenic versus natural sources, anthropogenic S flux will still constitute a major source of atmospheric S. Postel (1984 b,d) has summarized sulphur dioxide emissions for the United States, Canada, and West Germany (Figure 2). These are 24.1, 4.77, and 3.54 million tonnes y^{-1}, respectively (tonne = 10^3kg). Most of these emissions are from electric power plants in the US and Germany and from smelting and other industrial activities in Canada. Postel (1984d) estimated that the increases in SO_2 emissions between 1980 and 2000 will be +10% for the US, +2% for Europe and -5% for Canada. These are shown as the hatched portions of Figure 2. Eastern Europe and the Soviet Union will probably experience 36% increases during the same time period (Postel 1984d). Overrein (1977) stated that only about 10% of the sulphur compound emissions in Europe are due to natural processes, the rest being anthropogenic.

MacDonald and Sandhu (1975a) reported that 1974 sulphur emissions from Alberta gas plants were approximately 1,000 tonnes per day and these were dispersed over a wide area of the province. They further noted that development of the Alberta oil sands, which contain approximately 4.5% sulphur, could add to the total (MacDonald and Sandhu 1975a). Sulphur dioxide emissions were 1600 tonnes/day in 1981 and could approach 2000 tonnes/day by 2000 (Colley and Poon 1982). Licensed sulphur emissions from industrial sources in Alberta were 1470.3 tonnes/day in 1981; however, observed emissions were only 766.7 tonnes/day (Alberta Environment 1982).

Sulphur is an essential nutrient for plant growth, and sulphur deficiencies are widespread in regions such as the US Great Plains and adjacent Canadian Prairies (Brady 1984). Thus, sulphur is not only an acidifying pollutant but is also a very necessary element for plant growth and crop yield.

In view of the very large amounts cycled both naturally and anthropogenically, its role as a plant nutrient, and its acid forming potential, it is not surprising that considerable research attention has been given to sulphur. This is true of Alberta because sulphur gas emissions are the largest single industrial pollutant in the province. The potential effects of S-deposition and acidification will be discussed further under appropriate headings.

1.1.2 Nitrogen Cycle

The global nitrogen cycle is more complex than that of sulphur (Smith 1981). Pertinent aspects of the nitrogen cycle are shown in Figure 3. While estimates of total biological nitrogen fixation vary, the quantities are large. Maximum estimates are from 259 x 10^{12} g y^{-1} to 330 x 10^{12} g y^{-1} (Smith 1981; Rosswall 1983). Smith (1981) estimated that forest nitrogen fixation is approximately 40 x 10^{12} g y^{-1}, and in the same publication he estimated that nitrogen losses from forest soils are more than four times the amount fixed. If this is true, forests are experiencing a rapid net nitrogen loss. Smith's estimate of nitrogen loss from forest soil also greatly exceeded estimates of the contribution from industrial combustion and fossil fuel use (Figure 3). These estimates of nitrogen fluxes are tentative, but do indicate the relative importance of forests to the nitrogen cycle.

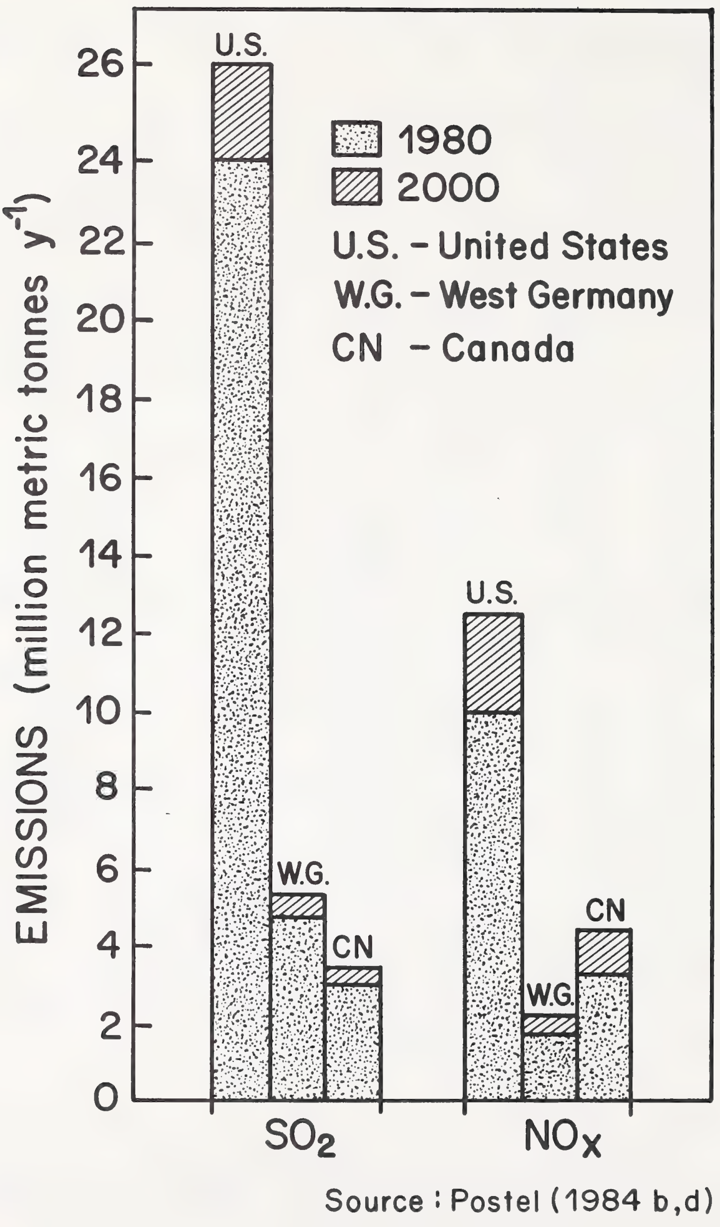
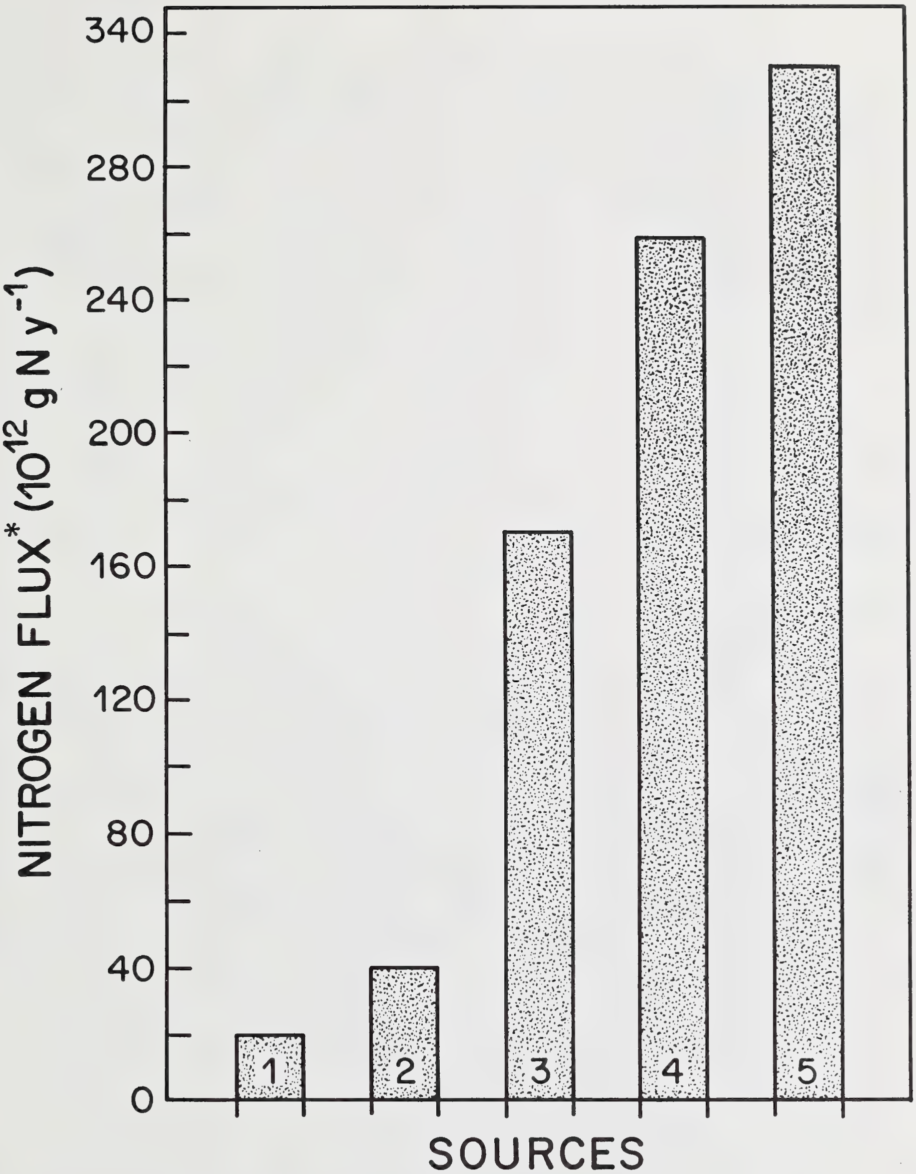


Figure 2. Total anthropogenic emissions of SO_2 and NOx for the US, West Germany, and Canada.



*10^{12} g = Tg

1. Industrial combustion and fossil fuel use (Rosswall 1983)
2. Forest nitrogen fixation (Smith 1981)
3. Release from forest soils (Smith 1981)
4. Maximum total biological nitrogen fixation (Smith 1981)
5. Maximum total biological nitrogen fixation (Rosswall 1983)

Figure 3. Biological nitrogen fixation and release from various sources compared to estimates of those for forests (global).

In addition to the natural nitrogen cycle, very large quantities of nitrogen oxides are emitted from anthropogenic sources, primarily from the transportation sector and electrical generating plants. The US, Canada, and West Germany emit, respectively, 19.3, 1.83, and 3.0 million metric tons of nitrogen oxides annually; approximately 75% of these are due to transportation and utilities (Postel 1984b). McLaughlin (1985) estimated that NOx emissions in the northeastern US have risen from 3,180 kg $km^{-2}y^{-1}$ in 1950 to 8,650 kg $km^{-2}y^{-1}$ in 1980. Thus, nitrogen oxides have increased considerably. Postel (1984b) estimated that nitrogen oxide increases between 1980 and 2000 will be 25% for the US, 33% for Canada, and 21% for the European economic community (Figure 2).

Legge et al. (1980) have reviewed the effects of nitrogen oxides on plants and the possible consequences in Alberta, Canada. It can be seen (Figure 3) that large quantities of nitrogen are naturally cycled annually, but it is important that anthropogenic sources also be considered in estimates of the nitrogen cycle and its implications.

Nitrogen, like sulphur, is an acid forming pollutant as well as an essential plant nutrient (Brady 1984). Thus, the potential effects on forests range from fertilization to soil acidification.

1.1.3 Carbon Cycle

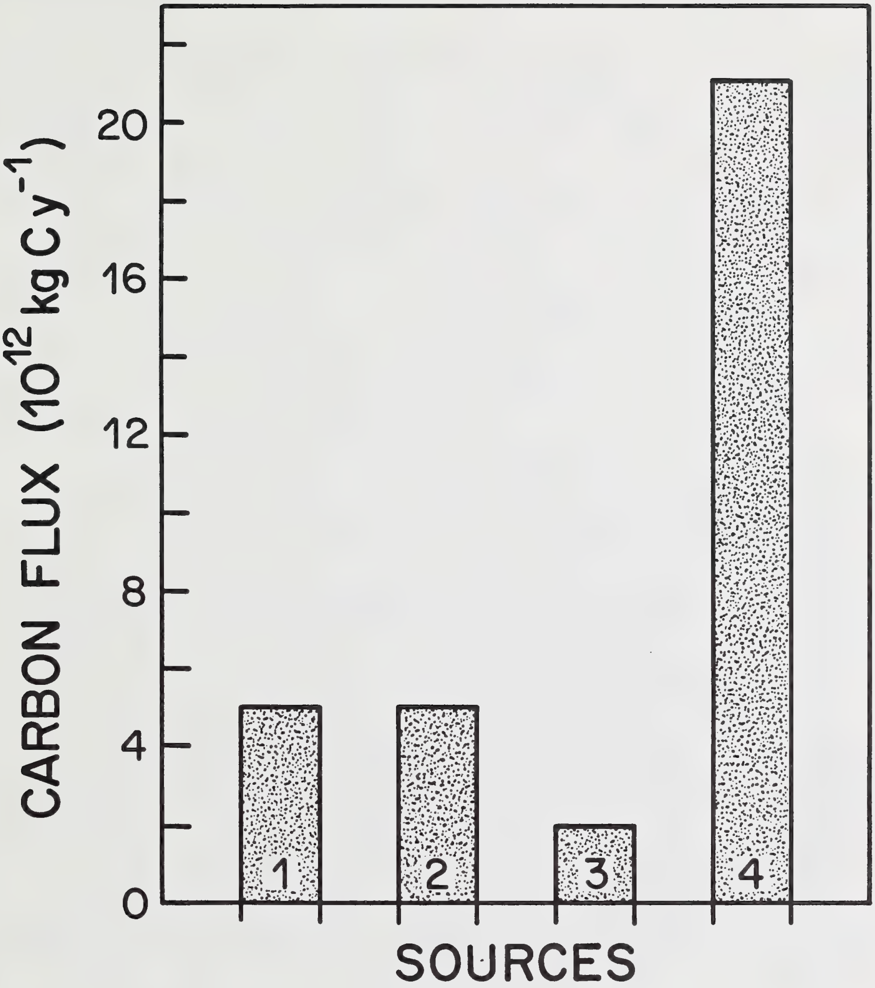
Atmospheric CO_2 concentrations have been increasing at about 3% per decade in recent years (Smith 1981). Between 1958 and 1978, worldwide atmospheric CO_2 increased from 315 ppm to 335 ppm, largely due to the activities of man (Oeschger et al. 1980). The level of carbon dioxide in pre-industrial times may have been as low as 265 ppm (Bolin 1983). There is concern that CO_2 increases will cause a general warming of the earth such that the biosphere will be affected (Bach et al. 1980).

Figure 4 shows some estimates of important carbon fluxes related to forests. Bolin (1983) estimated the fossil fuel carbon flux to be 5×10^{12} kg y^{-1}. This is equal to that originating in the soil. Smith (1981) estimated (and he cautioned against "absolute faith" in his numbers) that the combined flux due to forest destruction, burning, and humus oxidation is considerably greater than that from fossil fuel use (Figure 4). Although Bolin's (1983) estimate of the C flux due to deforestation was much less, the combined fluxes from soil and deforestation do exceed that of fossil fuel burning. Although these numbers are estimates, forests are believed to be major contributors to the worldwide carbon balance. Therefore, anything impacting a forest system would also be expected to have an impact on the carbon cycle.

Atmospheric carbon dioxide can form carbonic acid, thus adding potential acidity to the soil. Because it is a weak acid, it represents potential acidity and can act as a buffer. This is in contrast to strong acids such as nitric and sulphuric which dissociate completely (Krug and Frink 1983).

In terms of positive effects, carbon dioxide, like nitrogen and sulphur, is essential for plant growth because it is a raw material of photosynthesis. It is well known that plants can respond to elevated CO_2 by increasing the rate of photosynthesis and growth (Kramer 1981).

Only a few studies have examined the interactions of elevated CO_2 and other pollutants (Carlson 1982).



1. Fossil fuel burning (Bolin, 1983)
2. Soils, worldwide (Bolin, 1983)
3. Deforestation (Bolin, 1983)
4. Forest destruction, humus oxidation, and burning (Smith, 1981)

Figure 4. Estimated carbon fluxes from fossil fuel burning, soils, and forests.

In summary, atmospheric carbon dioxide, which is increasing, can have beneficial effects on photosynthesis, detrimental effects through acid formation, and potentially great effects upon climate.

1.1.4 Ozone

Ozone is not an acidifying compound, but it is included in this review for the following reasons:

1. It has been shown to cause damage and reduced growth in pine in the San Bernardino Mountains of California (Coyne and Bingham 1977), and it has been implicated in white pine emergence tip burn in West Virginia and Tennessee (Berry and Ripperton 1963; Berry and Hepting 1964).
2. Ozone levels in California have a direct correlation with CO_2, i.e., as CO_2 increases so does O_3 (Coyne and Bingham 1977)
3. Increasing ozone may have effects upon climate. For example, if the worldwide atmospheric ozone concentration were to double, temperatures could rise by $1^{\circ}C$ (Lacis et al. 1981).
4. Ozone has been shown to have synergistic interactions with other air pollutants (Ormrod 1982).

Recently, more attention has been given to ozone as a possible contributor to observed forest decline in Europe and the eastern United States (Blank 1985). Skärby and Selden (1984) reported that ozone concentrations of 100-250 $\mu g\ m^{-3}$ are frequent in Europe, and these often exceed the World Health Organization (WHO) limits of 120 $\mu g\ m^{-3}$ for one hour as well as the United States' recommended limits of 240 $\mu g\ m^{-3}$ for one hour. McLaughlin (1985) has summarized the maximum hourly concentrations from various places in the US and Germany. These vary from 255-911 $\mu g\ m^{-3}$ in Ohio to 666-784 in California. Maximum hourly concentrations in Germany vary from 788 $\mu g\ m^{-3}$ in the Black Forest to 410 $\mu g\ m^{-3}$ in the Ruhr. Postel (1984b) reported concentrations of 500 $\mu g\ m^{-3}$ in the Netherlands, West Germany, and Belgium. Ozone levels in Europe are therefore approaching those in the United States. Ozone may be a more important pollutant than was previously believed; currently, a widely held hypothesis promotes it as the cause of much forest decline (Blank 1985).

1.1.5 Summary of Biogeochemical Cycles and Ozone

In summary, it should be noted that carbon dioxide, nitrogen, and sulphur all are implicated in acid deposition -- especially the strong acid formers NOx and S-gases. It also must be recognized that in appropriate amounts all three are necessary for plant growth and under appropriate conditions can stimulate growth of plants. All three occur naturally in biogeochemical cycles involving the atmosphere, oceans, and terrestrial biosphere. Anthropogenic additions through fossil fuel combustion and fertilizer practices are additions to these cycles and may have far-reaching climatological and ecological effects. Lefohn and Brocksen (1984) have summarized potential hypotheses for separating natural and anthropogenic effects of acid deposition. Ozone, while not an acid former, is included because of its anthropogenic origins, known direct effects on plants, and interaction with other pollutants.

1.2 FOREST PROBLEMS ASSOCIATED WITH ACID DEPOSITION

1.2.1 Acid Deposition

It is evident that acid deposition, either as acid rain (wet deposition), pollutant gases (dry), or particulate matter (dry) is occurring over wide areas (McLaughlin 1985). Postel (1984b) stated that "...precipitation in many industrial areas is 10 to 30 times more acidic than would be expected in an atmosphere free of humanity's pollution". Krug and Frink (1983) have discussed acid precipitation fully and state that water in equilibrium with CO_2 has a pH of 5.6; therefore, precipitation values of less than 5.6 are considered acidic. They further pointed out that the involvement of a weak acid such as carbonic acid means that precipitation differences of 1 pH unit do not mean 10 times more acidity, drawing an analogy with reserve acidity in soils, because not all the carbonic acid will be dissociated. Nevertheless, it has been shown that low pH rainfall (pH 4 to 4.5) occurs over much of Europe (Postel 1984b) and eastern North America (Brady 1984).

Considerable effort has gone into estimating acidic deposition. Abrahamsen (1980) has stated that SO_4^{2-} deposition in Europe has increased two to three percent per year during the last twenty years, while NO_3^- deposition has increased five percent. He stated that H^+ deposition over Europe and North America varies from .1 to 1 kg H^+ $ha^{-1}y^{-1}$ and wet and dry deposition of SO_4^{2-} is from 10 to 120 kg S $ha^{-1}y^{-1}$. There is little doubt that anthropogenic pollutants are the major cause of acid deposition in Europe and North America.

1.2.2 Forest Decline

Many forests in Europe and North America appear to be undergoing degradation that includes reduced productivity, dieback, and death. Excellent reviews include those of Abrahamsen et al. (1976), Binns (1984), Morrison (1984), Postel (1984a,b,c,d,e,f), Postel (1985), and McLaughlin (1985).

The decline of European forests has been well documented by a number of workers in several countries. Overrein et al. (1980) have summarized the effects of acid deposition on Norwegian forests. Paces (1985), in an excellent study of element budgets for the Elbe River basin, has documented the probable cause of forest decline in central Europe. Van Breeman (1985), commenting on the previous references, stated that only pollution control can reverse forest decline. Postel (1984a) quoted Pravda as stating that forests along the Volga River near the city of Togliasti may "...soon resemble a wasteland", presumably due to atmospheric pollutants. O'Sullivan (1985) has reviewed the European situation and the attempts to reduce sulphur emissions by 30% by 1993.

The West German forests represent the best documented case of decline. Postel (1984a) reported that 76% of the fir, 41% of the spruce, and 43% of the pine showed damage. In the case of spruce, this involves 1,194,000 hectares. Together, these three species constitute two-thirds of West Germany's forests. Damage has also been found in the hardwoods. McLaughlin (1985) reported increases in moderate or heavy damage in one year (1983-84) of 86% in spruce, 87% in pine, 18% in fir, 168% in beech, and 327% in oak. Binns and Redfern (1983) have documented the forest decline in West Germany, including observation trips, disease outbreaks, and soil types. Ulrich and his co-workers have written numerous papers concerning acid deposition effects on West German forests (Ulrich et al. 1979, 1980; Ulrich 1982; 1983a,b,c,d; Ulrich and Matzner 1983; and Ulrich and Pankrath 1983). Roberts (1983) has reviewed the West German situation for Bioscience. More recently, Blank (1985) has reviewed the decline situation in Germany.

McLaughlin (1985) has reviewed the unprecedented decline of red spruce growth in the eastern United States, as reported by Siccama et al. (1982). Siccama et al. (1982) have presented good evidence of this large decline. For instance, in the Camels Hump boreal zone, the density of stems of $\leq$10 cm diameter declined by 52% between 1964 and 1979, and basal area declined by 44%. While these seem to be very large changes, statistical treatment is lacking, leaving the reader to judge the meaning of these results. Linthurst (1984) has edited a review of the problem in the US that suggests that there is indeed reduced productivity of eastern forests. However, there is no consensus as to the cause. In the preface, Linthurst (1984:xiii) stated: "To date, few studies exist that support beliefs that long-term acidic deposition will negatively impact plant productivity". Johnson and co-workers have documented reductions in growth rates of forest trees in Vermont, New Hampshire, and New Jersey and postulate a link with acid precipitation (Johnson, A.H. et al. 1981, 1982; Johnson 1983; and Johnson et al. 1984). They state that a regional stress is indicated and that red spruce (*Picea rubens*) is clearly experiencing dieback. Lefohn and Brocksen (1984:1007) have also reviewed the problem and concluded: "At present, scientific results support the conclusion that no direct evidence exists that acidic deposition currently limits forest growth in North America". Hibbard (1982) suggested that not enough is known about the problem. He placed great blame on local sources, such as oil furnaces in New York. Legge et al. (1978) reported reduced production in a lodgepole x jack pine forest downwind from a sour gas plant in Alberta, Canada, and attributed reduced production to the sulphur emissions.

1.3 INTRODUCTION SUMMARY

It has been shown that anthropogenic increases in carbon dioxide, oxides of nitrogen, sulphur gases, and ozone are occurring in industrialized regions. Forests downwind from these industrialized areas are being exposed to a variable mixture of pollutants as well as to acid deposition from those pollutants. In these same areas, regional declines in forest production and, in some cases, forest dieback have been observed. The process involved is unclear and there are a number of hypotheses that attempt to explain the effects of pollutants upon forests. As can be seen in Table 1, there is a range of possible effects of acid deposition on forests that can act singly or in combination. The references listed in Table 1 are examples of research showing altered responses due to acid deposition or acidic pollutants. The remainder of this paper will discuss these.

Table 1. References to various effects of acid deposition on soil, plants, forests, and ecosystems.

Effects as: Direct or Indirect	Effect	Reference
Direct Effects	Stomatal or mesophyll resistance	Black (1982)
	Photosynthesis	Carlson and Bazzaz (1982)
	Metabolism	Heath (1980)
	Hormones	Reid (1985)
	Membranes	Skärby and Sellden (1984)
	Growth	Higginbotham et al. (1985)
Indirect Effects	Canopy leaching	Foster and Morrison (1976)
	Soil Acidification:	
	-nutrient leaching	Morrison (1983)
	-aluminum and manganese	Ulrich et al. (1980)
	-phosphorus	Cook (1983)
	-weathering	Johnson et al. (1982b)
	-decomposition	Coleman (1983)
	-mycorrhizae	Patten (1983)
	-nitrogen fixation/ nitrification	Belser (1979)
	Fertilizer effects:	
	-sulphur	Smith (1981)
	-ammonium	Nihlgard (1985)
	Harvest technique	Johnson and Richter (1983)
	Forest reproduction	Cox (1983)
	Land use change and/ or succession	Krug and Frink (1983)

2. DIRECT EFFECTS OF POLLUTANTS ON FORESTS

2.1 STOMATAL PHYSIOLOGY, TRANSPIRATION, AND PLANT-WATER RELATIONS

The effects of pollutants on stomatal diffusive resistance (R_L), or stomatal conductance, and transpiration are varied and seem to be related to the kind of pollutant, exposure, and plant species (Table 2).

Majernik and Mansfield (1970) found that broad bean (Vicia faba) stomata opened more rapidly and to a greater degree when exposed to SO_2 at concentrations between 0.25 and 1.0 ppm. Beckerson and Hofstra (1979) found that: 0.15 ppm SO_2 stimulated opening of the stomata in radish, cucumber, and soybean; O_3 caused closure; and a mixture of SO_2 plus O_3 each at 0.15 ppm resulted in a higher R_L value than O_3 alone. Bytnerowicz and Taylor (1983) found that SO_2 increased diffusive resistance of bean, and that adding O_3 increased the resistance even more. Biggs and Davis (1982) found no effect of 0.12 ppm SO_2 on poplar leaf conductance while 0.25 ppm increased conductance (i.e., lowered diffusive resistance). Noland and Kozlowski (1979) found that 1.0 ppm SO_2 lowered diffusive resistance in elm and that 2.0 ppm raised it. Carlson (1982) reported that with C_3 and C_4 successional species, there were no effects of SO_2 at 0.25 ppm but that elevated CO_2 concentrations caused closure. Majernik and Mansfield (1972) have reported a similar response in broad bean. Caput et al. (1978) found SO_2 to increase diffusive resistance in three species of pine, as did Farrar et al. (1977) with Pinus sylvestris. Biggs and Davis (1982) found lowered water potential in older poplar leaves but little effect of SO_2 on young leaves. Mayo et al. (1986) found no difference in xylem pressure potential in Pinus contorta x P. banksiana hybrids with increasing distance from a S-gas source. Higginbotham et al. (1985) found no effect of elevated CO_2 on pine grown for five months. Kelly et al. (1984) reported that artificial acid rain (pH 5.7, 4.5, 4.0, or 3.5) applied for 30 months had no effect on transpiration rates of tulip poplar, white oak, and Virginia pine. Their results were obtained with a steady state porometer, yet no diffusive resistance or conductance values were given, making them impossible to evaluate and compare with other data.

Clearly, there are effects of atmospheric pollutants on stomata and water status. The variation in effects is likely due to very different exposure regimes, particularly those involving varying relative humidity and pollutant concentrations. As well, short-term laboratory experiments may give responses different from those found in the field. Black (1982) emphasized that well designed field and laboratory experiments are needed because stomata respond to light, CO_2, soil moisture, and relative humidity as well as to pollutants.

The ability to predict stomatal response is confounded by the interactive effects of SO_2, CO_2, and O_3. Tingey and Taylor (1982) emphasized that ozone (O_3) is less soluble than most pollutants but is highly reactive, and therefore diffusive resistance studies involving O_3 may be misleading. Ozone's major impact is probably on membrane permeability (Tingey and Taylor 1982). Elevated ozone concentrations always seem to favour stomatal closure, probably because of membrane degradation and loss of K^+ from guard cells. Sulphur dioxide, at low levels, may stimulate opening; however, at higher concentrations, SO_2 can cause closure in certain species, perhaps because of CO_2

Table 2. The effects of pollutants on stomatal diffusive resistance and water status.

Species	Pollutant	Concentration	Comparison of relative diffusive resistance, treatment control [1]	Water status	Reference
Vicia faba	SO_2	0.25 - 1.0 ppm	Treated < Control	-	Majernik & Mansfield (1970)
Radish, cucumber, soybean	SO_2	0.15 ppm	Treated < Control	-	Beckerson & Hofstra (1979)
Radish, cucumber, soybean	O_3	0.15 ppm	Treated > Control	-	Beckerson & Hofstra (1979)
Radish, cucumber, soybean	$SO_2 + O_3$	0.15 ppm each	Treated >> Control	-	Beckerson & Hofstra (1979)
Phaseolus vulgaris	SO_2	520 $\mu g\ m^{-3}$	Treated > Control [2]	-	Bytnerowicz & Taylor (1983)
Phaseolus vulgaris	$SO_2 + O_3$	520 $\mu g\ m^{-3}$ + 390 $\mu g\ m^{-3}$	Treated >> Control [2]	-	Bytnerowicz & Taylor (1983)
Hybrid poplar	SO_2	0.25 ppm	Treated < Control (5-10%)	Older leaves, lower water potential	Biggs & Davis (1982)
Hybrid poplar	SO_2	0.12 ppm	No Difference	Older leaves, lower water potential	Biggs & Davis (1982)
Ulmus americana	SO_2	1.0 ppm	Treated < Control	-	Noland & Kozlowski (1979)
Ulmus americana	SO_2	2.0 ppm	Treated > Control	-	Noland & Kozlowski (1979)
Pinus picea, *P. nigricans*, *P. sylvestris*	SO_2		Treated > Control	-	Caput et al. (1978)

continued...

Table 2 (Concluded).

Species	Pollutant	Concentration	Comparison of relative diffusive resistance, treatment control [1]	Water status	Reference
Ginko biloba	SO_2	2.0 ppm	Treated < Control	-	Noland & Kozlowski (1979)
Pinus sylvestris	SO_2		Treated > Control	-	Farrar et al. (1977)
C_3 and C_4 Successional Species [3]	CO_2 SO_2	300, 600, 1200 ppm 0.25 ppm	Treated > Control No effect of SO_2	-	Carlson (1982)
Pinus contorta [4]	CO_2	1000 and 2000 ppm	No effect on R_L minimum	Minimum xylem pressure potential; no effect	Higginbotham et al. (1985)
Pinus contorta x P. banksiana	S-Gas	Concentration gradient downwind from sour gas plant	Generally higher adjacent to the gas plant	No measurable difference	Mayo et al. (1986) Legge et al. (1981)

[1] Relative diffusive resistance either as resistance measured, conductance, or inferred from transpiration.
[2] Returned to control values after fumigation.
[3] C_3 Chenopodium, Datura, Polygonum; C_4 Amaranthus and Setaria.
[4] Long term growth experiment seedlings grown 5 months at elevated CO_2.

buildup that results from reduced photosynthesis (see Section 2.2). Carbon dioxide can reduce transpiration in short-term experiments (Carlson 1982) but seems to have little effect under long-term experiments (Higginbotham et al. 1985). The results with tree species seem to suggest clearly that SO_2 favours stomatal closure (with the exception of Ginkgo) once a relatively high SO_2 level (e.g., 2 ppm in elm) is reached (Table 2). Further work should improve our ability to predict responses.

2.2 PHOTOSYNTHESIS AND CARBON ALLOCATION

Recent data concerning the effects of pollutants on photosynthesis and carbon allocation (or growth) are shown in Table 3. The review by Natr (1975) is included to emphasize the fact that nutrients, especially nitrogen, can affect photosynthesis. This illustrates how photosynthesis can indirectly be affected by acidification impacts on soil nutrients.

Photosynthesis and carbon allocation are reflected in growth and productivity more than any other physiological parameter. The results summarized in Table 3 were observed under a variety of conditions ranging from laboratory controlled exposure to field conditions using various techniques. All suggest that a major effect of acid deposition is upon the photosynthetic process either through stomatal closure (Table 1) or through direct effects on the mesophyll cells, as discussed below.

Sulphur dioxide has been shown to reduce photosynthesis in lichens (Nieboer et al. 1976), C_3 and C_4 successional herbs (Carlson and Bazzaz 1982), various species such as peas (Black 1982), and Pinus contorta x Pinus banksiana hybrids (Legge et al. 1986). The response of lichens has implications for the pine-lichen woodlands and must be considered as serious. The leakage of K^+ ions in Cladonia (Table 3) may be due to acid dissolution of membranes. Various species of pine are clearly sensitive to SO_2 with substantial reductions in photosynthesis, and, according to Legge et al. (1986), with reductions in photosynthetic capacity. It is unclear if the SO_2 effect is direct or, rather, if it occurs indirectly through soil acidification. The latter possibility is discussed in more detail below.

Higginbotham et al. (1985) showed that carbon dioxide increases photosynthesis of lodgepole pine, and growth to a degree, but at higher concentrations both are reduced, suggesting limits on the ability to assimilate carbon. This was a controlled-environment experiment in which none of the other factors such as light, nutrients, or water was limiting. Biomass increases were greatest in the roots, suggesting increased carbon allocation to the roots at high CO_2 levels.

Carlson and Bazzaz (1982) found that SO_2 caused only slight reductions in the photosynthesis of C_3 plants and slightly greater ones in C_4 successional herbaceous plants. The reductions in growth were more pronounced, indicating altered carbon allocations.

To a considerable degree, increasing CO_2 tends to ameliorate the effects of SO_2 (Black 1982). This is also suggested in Carlson and Bazzaz's (1982) work: higher CO_2 concentrations increased photosynthesis, with or without SO_2.

Ozone and other oxidants reduce photosynthesis. Miller et al. (1969) reported sharp reductions in photosynthesis when ponderosa pine was fumigated with O_3 in the laboratory. They found other chemical changes as well. Coyne and Bingham (1982)

Table 3. The effects of pollutants on photosynthesis and carbon allocation [1].

Species	Pollutant	Concentration	Effect on Photosynthesis (Ps)	C-allocation	Other effects	Reference
Wheat & Barley	Nitrogen	Increasing	Linear increase in photosynthesis with nitrogen regardless of the source	Ca affects translocation	K^+ deficiency reduces Ps	Natr (1975)
Lichens; <u>Cladonia</u>	SO_2	0.75-75.0 ppm	5% to 100% reduction in photosynthesis		Large K^+ leakage	Nieboer et al. (1976)
<u>Pinus ponderosa</u>	O_3	0.15 ppm (30 d) 0.30 ppm (30 d) 0.45 ppm (30 d)	Reduced Ps 10% Reduced Ps 70% Reduced Ps 85%	Lower needle polysaccharide level	Increased ascorbic acid content of needles	Miller et al. (1969)
<u>Pinus contorta</u>	CO_2 CO_2 CO_2	300 $\mu L\ L^{-1}$ 1000 $\mu L\ L^{-1}$ 2000 $\mu L\ L^{-1}$	6.2 mg $dm^{-2}\ h^{-1}$ Net Ps 12.3 mg $dm^{-2}\ h^{-1}$ Net Ps 8.9 mg $dm^{-2}\ h^{-1}$ Net Ps	Maximum growth at 1000 ppm mostly as root growth		Higginbotham et al. (1985)
Alfalfa, bean, grass	O_3, NOx SO_2, PAN	Various - review	Reduced photosynthesis 0.5 ppm $\geq$ 1 h		Interaction with membranes, loss of Ca from cell wall	Heath (1980)
C_3 and C_4 [2] successional herbs	SO_2 and CO_2	300, 600, 1200 ppm CO_2; 0.0 or 0.25 ppm SO_2	SO_2 caused a slight reduction in Ps of C_3's and greater reduction in C_4's	SO_2 decreased growth in all but C_3 at 300 ppm		Carlson & Bazzaz (1982)

continued...

Table 3 (Continued).

Species	Pollutant	Concentration	Effect on Photosynthesis (Ps)	C-allocation	Other effects	Reference
Pinus ponderosa	Oxidant fumigations; O_3 San Bernardino Mts.	3 injury classes: I-Slight, II-Moderate, III-Severe Concentration implied from injury class: I-slight; II-moderate; III-severe	2 y needles class III lowest Ps; current needles, class I highest Ps; lowest = 1 mg CO_2 g^{-1} h^{-1} highest=12 mg CO_2 g^{-1} h^{-1}	CO_2 assimilation efficiency ratio highest in current class I	Current needles class II had very high conductances	Coyne & Bingham (1982)
Pinus strobus	Oxidant-ozone sensitive, intermediate and tolerant trees studied	125 to 339 h y^{-1} $\geq$ 0.08 ppm	Ps capacity of sensitive trees 4.5 mg CO_2 g^{-1} h^{-1} and resistant 4.6, i.e., no difference	Growth less in sensitive trees; less export of C to boles & roots in sensitive trees	Respiration higher in sensitive trees	McLaughlin et al. (1982)
Various - review	SO_2	Concentration that causes reduced photosynthesis varies with species;	Pea - .2 ppm causes 50% reduction; Vicia - .4 ppm causes 40% reduction; Atriplex - 1.4 ppm causes 15% reduction; Heteromeles - 1.8 ppm causes 50% reduction		ABA enhances inhibition of transpiration by SO_2 (2 ppm); high CO_2 results in less reduction of Ps by SO_2	Black (1982)
Pinus contorta x P. banksiana hybrid (3)	S-gas emissions, SO_2, H_2S, etc.	Gradient downwind from S-gas plant	Current foliage 2.8 mg g^{-1} h^{-1} at a distance; 2 y old needles 2.0 mg g^{-1} h^{-1} close to plant and 2.4 mg g^{-1} h^{-1} further from plant	Growth reduced by proximity to S source	Maximum leaf resistances highest close to S source	Legge et al. (1986)

continued...

Table 3 (Concluded).

Species	Pollutant	Concentration	Effect on Photosynthesis (Ps)	C-allocation	Other effects	Reference
Pinus strobus	SO_2	5 pphm for 2 h	Sensitive clones reduced Ps 27%; resistant clones 10%		No difference in needle length	Eckert & Houston 1980
Tomato (greenhouse)	NOx	2000 ppb	Low nutrient solution nitrogen 43 ppm results in + 3% yield of fruit; 170 ppm N in solution results in -13% yield; infers greater photosynthesis with low N		CO_2 + NOx gives greater yield than no CO_2 enrichment	Law & Mansfield 1982
Soybean	$SO_2 + O_3$ or acid rain	0.1 µl L^{-1}+ 0.5 µl L^{-1} or pH 3.4, 4.2, 5.0	Inhibited by $SO_2 + O_3$ but not by acid rain	Growth reduced by $SO_2 + O_3$ but not by acid rain	Nitrogen fixation reduced by $SO_2 + O_3$ but not by acid rain	Norby et al. (1985)
Liriodendron tulipifera, *Quercus alba*, *Pinus virginiana*	Acid rain	pH 5.7, 4.5, 4.0, 3.5	No reduction or enhancement	-	No effect on respiration	Kelly et al. (1984)

[1] Ps used to indicate some measure of photosynthesis, including infrared gas analysis, CO_2 incorporation, etc.
[2] C_3 plants include *Chenopodium*, *Datura*, *Polygonum*; C_4 plants *Amaranthus* and *Setaria*.
[3] Data from cut branches, therefore values are photosynthetic capacity.
[4] Data based upon Warburg manometric measurements - not comparable to infrared gas analysis or $^{14}CO_2$ methods.

reported greatly reduced photosynthesis in ponderosa pine in the San Bernardino mountains of California with older, obviously injured needles having photosynthetic rates of less than 10% of current, little injured needles. Clearly there is a relationship between age, degree of injury, and photosynthetic capacity. Interestingly, Coyne and Bingham (1977) found a high positive correlation between ozone and CO_2 atmospheric levels in California; whether the increased CO_2 can to some degree ameliorate the effects of ozone is not clear. McLaughlin et al. (1982), studying white pine, found that sensitive and resistant trees had the same photosynthetic capacity but that altered carbon allocation patterns in the sensitive trees resulted in reduced growth. This study is the only one reported in Table 3 in which photosynthesis was not reduced. Eckert and Houston (1980) found photosynthesis in sensitive clones of white pine to be inhibited by 27% compared with only 10% in tolerant clones. The work of Kelly et al. (1984) suggested that acid *per se* does not reduce photosynthesis; the effects of acid formers such as SO_2 may be directly upon the mesophyll. They used Warburg manometry to determine photosynthesis of leaf discs, a technique not comparable to infrared gas analysis or CO_2 analysis. The latter methods are more desirable because they measure photosynthesis of an intact plant under more natural conditions.

The results shown in Table 3 also suggest considerable variation in species sensitivity, with lichens being very sensitive, as has long been known. These data also suggest considerable variation in sensitivity within a species, making selection of resistant clones a possibility. Still, the overwhelming impression is that proximity to an acid- or oxidant-source can be expected to reduce forest photosynthetic capacity, which is likely to be reflected in growth.

2.3 PLANT BIOCHEMISTRY

There is ample evidence to suggest various biochemical mechanisms by which acid deposition and ozone can effect plants (Table 4). Sulphur dioxide can cause chlorophyll destruction, reduced ATP formation, and reduced carbohydrate formation (Malhotra 1977; Harvey and Legge 1979; and Wellburn 1982). Ozone inhibits nitrate reductase and reduces carbohydrate levels (Miller et al. 1969; Tingey and Taylor 1982). Sulphur dioxide can stimulate ethylene production which, combined with the effects listed above, results in senescence. Ozone also disrupts membranes, resulting in K^+ efflux (Tingey and Taylor 1982). Sulphur dioxide will also competitively inhibit Rubisco, the carboxylating enzyme in C_3 species, as well as PEP carboxylase, a major carboxylation enzyme in C_4 plants (Black 1982). All of these biochemical effects suggest that direct impacts on photosynthesis are possible.

Recently, Frank and Frank (1985) and Frank (1985) presented evidence that suggests chloroethenes may be involved with forest decline in Germany. They provide the following evidence: (1) Chloroethenes and other halogenated chlorocarbons have increased over the last ten years, particularly in rural areas of industrialized countries; (2) Chloroethenes are lipophilic with high partition coefficients between lipids and air; thus, they will tend to be taken up efficiently by thick waxy cuticles such as those found in conifer needles; (3) UV light can activate chlorocarbon transformations to highly toxic species such as phosgene; (4) needles of *Picea abies* exposed to tri- or tetrachloroethene for five hours exhibited HPLC (high pressure liquid chromatography)

Table 4. Biochemical effects of pollutants.

Species	Pollutant	Concentration	Lab=1 Field=2	Effect on Photosynthesis	Relation to Physiology, Growth, etc.	Reference
Pinus contorta x Pinus banksiana	SO_2	0-250 ppb 10-207 ppb	1 2	Linear reduction of ATP with increasing SO_2 - approx. 80% reduction at 225 ppb. Field foliage has 450 ng ATP g^{-1}. Lab cultured foliage 1460 ng ATP g^{-1}	Lab reduction of photosynthesis as ATP declined	Harvey and Legge (1979)
Pinus contorta	SO_2 aqueous		1	Chlorophyll destruction		Malhotra (1977)
Review	Ozone			Inhibition of nitrate reductase changes membrane permeability particularly K+ efflux	N metabolism	Tingey and Taylor (1982)
Dactylis glomerata Poa pratensis Lolium, Phleum	SO_2, NO_2 SO_2 + NO_2		1	Glutamate dehydrogenase increase with SO_2 (indicates ammonia assimilation) SO_2-no effect on nitrite reductase. NO_2-great increase nitrite reductase. SO_2 + NO_2-no increase nitrite reductase	SO_2-decreases photophosphorylation. SO_2 + NO_2-decreases photophosphorylation. NO_2-great increase in photophosphorylation.	Wellburn (1982)
Pinus ponderosa	O_3	0.30 ppm for 30 days	1	Higher ascorbic acid in injured needles. In one-year-old needles sugars reduced by O_3, polysaccharides decreased by 40%	Reduced sugars and polysaccharides related to reduced Ps and early needle drop.	Miller et al. (1969)

continued...

Table 4 (Concluded).

Species	Pollutant	Concentration	Lab=1 Field=2	Effect on Photosynthesis	Relation to Physiology, Growth, etc.	Reference
Various - review	SO_2	various	1	SO_2 enhances ABA induced stomatal closure- 2 ppm SO_2 on radish degrades chlorophyll - chloroplasts and mitochondria swell. Rubisco and PEP carboxylase competitively inhibited by SO_2, sulphite and bisulphite.	Membrane integrity organelle ultra-structure effect on stomata and various processes reduce photosynthesis.	Black (1982)
Review	Ozone, SO_2	various		O_3 at 0.05 $\mu L\ L^{-1}$ & SO_2 at 0.3 $\mu L\ L^{-1}$ can stimulate ethylene production (acid ppt) 50 ppb to 200 ppm for 1 h threshold	Ethylene causes a wide variety of responses- slow cell division, leaf & flower senescence, resin formation in pine, slow leaf expansion.	Reid (1985)
<u>Picea</u> <u>omorica</u>	Chloroethenes	tree exposed to trichloroethenes	2	Chlorophyll destruction	Developed symptoms of German dieback	Frank & Frank (1985)
<u>Picea</u> <u>abies</u>	Tri- or Tetra-chloroethene	14 $\mu g\ m^{-3}$ for 5 h	1	HPLC[1] of pigments very similar to that of naturally affected needles.		Frank (1985)

[1] HPLC = High Pressure Liquid Chromatography

chromatograms of leaf pigments very similar to naturally affected needles; and, (5) Picea omorica exposed in the field developed dieback symptoms similar to those found in German forests.

2.4 CUTICULAR DAMAGE

Morrison (1984) has reviewed the voluminous literature on the effects of artificial acid rain on the cuticle of a wide range of forest tree species. These experiments, mostly on seedlings exposed to very acid solutions in the pH range of 2.0 to 3.0, do indicate that the cuticle can be damaged and necrosis develop. However, these are drastic treatments and do not explain the deleterious effects of much less acidic depositions. Undoubtedly, if exposed to such acid precipitation for extended periods, many of the effects discussed in the previous sections would also occur. Morrison (1984) concluded that reports of positively identified injury of this type in field grown trees are "lacking".

2.5 SUMMARY OF DIRECT EFFECTS

Tables 2 through 4 provide abundant documentation that acid precursors such as SO_2, NOx, and ozone can cause stomatal closure, affect water relations, reduce photosynthesis, and change carbon allocation through a variety of effects acting upon phosphorylation, chlorophyll, carboxylation, hormone balance, and membrane integrity. In no case was ozone shown to be beneficial. Frank (1985) and Frank and Frank (1985) also provided evidence for pigment destruction by UV activated chloroethene in German forests. However, it is possible that under some conditions, very low levels of nitrogen oxides or sulphur can act as fertilizers. It is equally clear that the effects of these pollutants interact with elevated carbon dioxide and in many cases they are less damaging or slightly stimulatory. It is also possible that some of the direct effects discussed in these studies are, in fact, indirect effects brought about through soil acidification. The work of Legge et al. (1986) is a case in point.

3. INDIRECT EFFECTS ON FORESTS

3.1 INTRODUCTION

The following topics are included under indirect effects: canopy-pollutant interactions; soil acidification; nutrient leaching; weathering; and effects on microbial activity. The reader is advised of several excellent, recent reviews covering most of the same information. These include those of Abrahamsen et al. (1976), Binns (1984), Morrison (1984), and McLaughlin (1985).

3.2 CANOPY-POLLUTANT INTERACTIONS

Forests can remove large amounts of materials from the atmosphere, thus changing considerably the concentration of wet and dry deposition reaching the soil. Smith (1981) estimated that a 1-ha model forest can remove the following amounts of pollutants (t/y): O_3 - 9.6 x 10^4; SO_2 - 748; CO - 2.2; NOx - 0.38; PAN - 0.17. Binns (1984) noted that 30 to 40% of incoming precipitation can be intercepted and re-evaporated from the canopy. Thus, there is considerable opportunity for canopy-pollutant interaction, and deposition on the soil may be considerably different from that of the ambient atmosphere. Granat (1983), studying conifers in Sweden, estimated that wet deposition of sulphur compounds (SO_4^{2-} and SO_2) was greater than dry deposition. The yearly average in grams m^{-2} was: SO_2 (dry), 0.22; SO_4^{2-} (dry), 0.10; and SO_4^{2-} (wet), 0.90. Nevertheless, dry deposition was 0.32 g m^{-2} y^{-1}. Höfken (1983) reported considerably more dry deposition of SO_4^{2-} and NO_3^- in spruce forests. The wet deposition rates of SO_4^{2-} were 660 mg m^{-2} $month^{-1}$ in winter and 330 in summer, whereas the dry deposition was 2,600 mg m^{-2} $month^{-1}$ in winter and 1,400 in summer. His results for NO_3^- were similar, although the absolute amounts were less. These data indicate considerable amounts of dry deposition of acidic ions which can react with the leaves and at a later time be washed from the plant.

Table 5 summarizes some of the recent studies of the effects of the forest canopy on precipitation. In general, hardwood canopies tend to raise pH of the throughfall (Abrahamsen et al. 1976; Hoffman et al. 1980; Miller 1983; and Mollitor and Raynal 1983). Miller (1983) found that a young scots pine stand raised pH as well. In general, conifers lowered pH (Table 5), although Miller (1983) reported that sitka spruce raised the pH from 4.6 to 5.4. All of the references in Table 5 indicated that stemflow was more acidic than throughfall. Hoffman et al. (1980) reported that total acidity of the throughfall was approximately the same as incident precipitation but that weak acids increased by 20-40% in the throughfall while strong acids decreased by a like amount, suggesting an exchange of weak acids, perhaps organic in nature, for the strong acids in precipitation.

Hardwood canopies decreased the hydrogen ion concentration in the throughfall relative to precipitation (Table 5). Cation concentrations increased in the throughfall, suggesting H^+-cation exchange. This agrees with the results of Eaton et al. (1973) who suggested H^+-cation exchange. Sulphate concentration increased in the throughfall as well, thus providing a mobile anion to accompany leached cations. Nitrate and ammonium were both less in the throughfall, suggesting differential absorption (i.e., fertilization).

Table 5. Canopy-precipitation interactions: pH, hydrogen ion exchange, and various cation exchanges.

	Incident Precipitation		Throughfall								
Species	pH	H^+	pH	H^+	SO_4^{-2}	NO_3^-	K^+	Ca^{2+}	Mg^{2+}	Other	Reference
Quercus prunus	4.0-4.6		4.6-5.7							Weak acid increase 20-40%. Strong decreased throughfall.	Hoffman et al. (1980)
Sitka spruce		440-970 eq $ha^{-1}y^{-1}y$		130-390 eq ha^{-1} y^{-1}						Young stands increase pH but older conifers reduce pH. Stem flow is more acidic.	Binns (1984)
Old scots pine	4.6		3.9							Stem flow pH = 3.1	Miller (1983)
Young scots pine	4.6		4.7							Stem flow pH = 3.5	
Sitka spruce	4.6		5.4							Stem flow pH = 3.9	
Japanese larch	4.6		4.8							Stem flow pH = 3.7	
Birch	4.6		4.8							Stem flow pH = 3.9	
Open	4.3	0.03 g m^{-2}	-	-	1.6 g m^{-2}	0.8 g m^{-2}	0.1 g m^{-2}	0.1 g m^{-2}	0.1 g m^{-2}	No PO_4^{3-} NH_4^+ 0.2 g m^{-2}	Abrahamsen et al. (1976)
Betula	4.3	0.03 g m^{-2}	4.5	0.02 g m^{-2}	2.1 g m^{-2}	0.4 g m^{-2}	0.8 g m^{-2}	0.4 g m^{-2}	0.2 g m^{-2}	No PO_4^{3-} NH_4^+ 0.1 g m_{-2}	
Pinus	4.3	0.03 g m^{-2}	4.0	0.08 g m^{-2}	3.3 g m^{-2}	0.3 g m^{-2}	0.9 g m^{-2}	0.6 g m^{-2}	0.3 g m^{-2}	No PO_4^{3-} NH_4^+ 0.1 g m_{-2}	
Picea	4.3	0.03 g m^{-2}	4.0	0.05 g m^{-2}	3.6 g m^{-2}	0.1 g m^{-2}	1.5 g m^{-2}	0.6 g m^{-2}	0.2 g m^{-2}	No PO_4^{3-} No NH_4^+	
Precipitation	4.06	88	-	-	66μeq L^{-1}	31μeq L^{-1}	6μeq L^{-1}	13μeq L^{-1}	5μeq L^{-1}	K^+,Ca_2^+,Mg_2^+ were increased by hardwood and greatly increased by conifer.	Mollitor & Raynal (1983)
Hardwood forest	4.06	88	4.18	66μeq L^{-1}	196μeq L^{-1}	34μeq L^{-1}	52μeq L^{-1}	139μeq L^{-1}	25μeq L^{-1}		
Conifer forest	4.06	88	4.02	96μeq L^{-1}	157μeq L^{-1}	44μeq L^{-1}	80μeq L^{-1}	175μeq L^{-1}	40μeq L^{-1}		

continued...

Table 5 (Concluded).

Species	Incident Precipitation		Throughfall								Reference
	pH	H^+	pH	H^+	SO_4^{-2}	NO_3^-	K^+	Ca^{2+}	Mg^{2+}	Other	
Not stated	5.0		4.8							Stemflow always lower. pH at control site raised in throughfall.	Baker et al. (1977)
Above canopy	-	0.071 mgL^{-1}	-		1.24 mgL^{-1}		0.65 mgL^{-1}	0.21 mgL^{-1}	0.48 mgL^{-1}		Foster and Nicholson (1983)
Acer saccharum			-	0.052 mgL^{-1}	2.20 mgL^{-1}	0.604 mgL^{-1}	2.68 mgL^{-1}	1.04 mgL^{-1}			
Open	-	240 eq ha^{-1}	-		260 eq ha^{-1}					HCO_3= 420 eq ha^{-1}	Cole and Johnson (1977)
Douglas fir	-	-	-	70 eq ha^{-1}	320 eq ha^{-1}					HCO_3= 570 eq ha^{-1}	
Control sites (lodgepole pine)	6.1										Baker (1977)
Impinged sites (lodgepole pine)	5.8-6.1										

Several of the conifers tended to increase the H^+ and cations of the throughfall; however, sitka spruce and Douglas fir decreased H^+ in the throughfall. Sulphate, K^+, Ca^{2+}, and Mg^{2+} increased considerably in the throughfall, suggesting leaching. However, cation leaching was not always accompanied by hydrogen ion uptake. It is interesting that sulphate ions were increased in the throughfall since sulphate accumulation often occurs (Legge and Bogner 1983). Legge and Bogner (1983) reported values of approximately 1,000 ppm sulphate-sulphur in lodgepole x jack pine foliage close to a S-gas emission source, which they attributed to direct foliar uptake as well as absorption from the soil. Baker (1977) also studied the effects of proximity to gas extraction plants in Alberta and found that rainfall pH near S-gas sources varied from 5.8 to 6.1 while pH at his control site was 6.1. He also found that needles had higher sulphur levels (447 ppm near the source compared with 364 ppm at the control site). Aluminum content in needles was 490 ppm near the plant compared with 260 ppm at the control site. The percentages of Mg, Ca, K, P and N in foliage were all less in the S-gas impingement areas, suggesting that these elements were leached.

3.3 SOIL ACIDIFICATION

The effects of acid deposition on soil have received a great deal of attention in recent years. The effects of low soil pH on nutrient availability, aluminum toxicity, manganese toxicity, iron availability, and soil microbiology have long been known (Brady 1984). The relationship between anthropogenic acid formers and soil pH is not so well established and there is considerable debate as to its importance.

There are several very good recent reviews on the subject, but the reviewers do not reach the same conclusions. Ulrich (1983c) has reviewed the nature of soil acidity, and the ranges of soil buffering due to calcium carbonate (pH >8 to 6.2), silicate (pH 5 to 4.2), cation exchange (pH 5 to 4.2), aluminum (pH 4.2 to 2.8), and iron (pH 3.8 to 2.4). He has discussed the various measures of soil acidity including pH and titration curves. He also discussed the characteristics of a soil that is sensitive to acid.

Prenzel (1983) discussed the ways in which acid can be stored in soils rather than showing up in groundwater seepage. One of these is as $AlOHSO_4$, which is sparingly soluble and could release H^+ and Al for years after anthropogenic acid input ceased, thus maintaining acidity and keeping the exchange complex devoid of base cations.

Ulrich is convinced that forest decline in Germany is due to acid deposition-caused soil acidification (Ulrich et al. 1979; Ulrich 1982, 1983a,b,d; and Ulrich and Matzner 1983). Krug and Frink (1983) have written an excellent review on the effect of acid precipitation on soils, in which they go to great lengths to show that not all soil acidification is due to air pollutants. The detailed review by Tabatabai (1985) on the effect of acid rain on soils is also highly recommended. Other recommended reviews on the subject are those of Binns (1984) and Morrison (1984). These will be mentioned within the various topics discussed below.

There are a number of researchers who believe anthropogenic acid deposition causes soil acidification and is the primary cause of forest decline. Ulrich (1983a) described a highly stable ecosystem as one with many species structured in layers that are deeply rooted. Decomposers are very active, especially earthworms, and the soil is in the silicate or carbonate buffer range. He then described the destabilizing effects of acid deposition leading to humus disintegration, loss of stable soil aggregates, buildup of litter, and finally, aluminum toxicity.

Ulrich et al. (1980) found that in the Solling region of West Germany, pH declined by 0.5 units between 1966 and 1979, exchangeable aluminum increased from 0.9 to 1.5 mg L^{-1}, stored carbon and nitrogen increased, and productivity declined. They attributed all these changes to acid deposition. However, in a later paper, Ulrich (1983b) stated that forest dieback cannot be attributed solely to soil acidification because it also occurs on soils that have not been acidified.

Baker (1977) and Baker et al. (1977) have documented lowered pH, elevated exchange acidity, elevated exchangeable aluminum, and depressed calcium and magnesium levels in the soils near natural gas treatment plants in Alberta. Legge et al. (1981) have documented the same responses in the West Whitecourt area of Alberta. It should be noted that although these gas treatment plants are operated within the provincial standards, they will generally have SO_2 concentrations near the plant which are higher than in other rural areas. For instance, the 1/2-hour maximum concentration in Bavaria, an industrial region, is 1900 $\mu g\ m^{-3}$ and in rural Tennessee is 165 (McLaughlin 1985). During the 1976 study period at the West Whitecourt intensive site, 1.5 km distance from the source, 71% of the 63 study days had SO_2 events of .01 ppm (26 $\mu g\ m^{-3}$) or higher which lasted a median time of 12 minutes (Lester et al., 1986). This illustrates the greater possibilities for acidification near a point source such as a sour gas plant. Thus, it seems that some of the clearest evidence of soil acidification may be found in Alberta.

It is by no means clear that soil acidification is always due to pollution. Krug and Frink (1983) suggested that much of the soil acidification in the northeastern US is due to changed land use patterns which have allowed cut-over forests and abandoned farmland to revert to coniferous forests by secondary succession, a process that naturally acidifies the soil. They reported that fields cropped to corn and tobacco which were abandoned about 1900 have become more acidic as red pine occupied the sites. pH had dropped to 4.4 by 1929 and to 3.8 by 1944. Rosenqvist (1978) has suggested that acidification of rivers in Norway may be due to secondary succession to forest and heath, timber harvesting that removes 10-60 meq bases m^{-2} annually, and extensive cropping practices that remove bases. Tabatabai (1985) suggests that much of the soil acidification in the midwestern US is due to the extensive use of nitrogenous fertilizers such as ammonia, which upon nitrification add protons to the soil. Thus, acidification may be due to farm practices rather than to industrial pollution. Overrein et al. (1980) have also suggested that the acidification and loss of fish in Norwegian lakes may be due to land use changes. Where there is no farming (i.e., forests), 78% of the lakes lack fish; in areas with abandoned farms (i.e., some stages of secondary succession), 60% of the lakes are barren; and, where watersheds are farmed only 30% of the lakes have no fish. They conclude that reforestation may enhance acidification. However, Wright and Hendriksen (1980), in a survey of 72 lakes in southwest Scotland, found no relationship between the percent of catchment reforested and lake pH. Nilsson (1983) stated that there is no unequivocal evidence of soil acidification caused by atmospheric deposition. He believes that tree species replacement and ion accumulation in plant biomass and humus are the most important causes. Nihlgard (1985) has postulated that forest dieback results from the extensive use of ammonia fertilizers which contribute an estimated 10-25 Kg N $ha^{-1}y^{-1}$ to bulk precipitation in central Europe and the central US. This compares with <1 Kg N $ha^{-1}y^{-1}$ in clean areas. He feels that this fertilization of the forest can lead to reduced frost hardiness and lowered disease resistance which can result in forest decline. Skeffington (1983) studied soil properties along transects out from the trunks of trees and found that pH was much lower nearest the tree and concluded that the trees were affecting the soil. However, as mentioned earlier, the pH of stem flow is always lower than that in throughfall or bulk precipitation.

Nilsson et al. (1982) also cited forest growth as a cause of acidification; when NH_4^+ is absorbed, a proton is released. This, along with heavy cation uptake during the early portions of a rotation (after harvest), can acidify the soil.

From the discussion above, it is apparent that there is evidence of soil acidification that can lead to forest decline and there are compelling reasons to suspect natural acidification. Lefohn and Brocksen (1984) have stated that a major research goal should be to separate natural from anthropogenic acidification. It may well be that forest decline is due to various causes. It seems reasonable to expect that in some cases, soil acidification is the cause of forest decline.

There has been a considerable effort to define the characteristics of soils that might be sensitive to acid deposition. Some of these characteristics are: non-calcareous, coarse textured, low cation exchange capacity (less than 12 meq/100g), 30 to 50% base saturation, intermediate pH (>5.5 to ~6), and intermediate fertility (Krug and Frink 1983; McFee 1983; Ulrich 1983c; Morrison 1984; and McLaughlin 1985). In the discussion that follows, the effects of acidification on various processes will be summarized. It is assumed that in soils with characteristics similar to those listed above, these problems could arise.

3.3.1 Nutrient Leaching

Cook (1983) has reviewed the effects of acid deposition on nutrient cycles, and the reviews by Morrison (1984) and McLaughlin (1985) discuss the effects of acid deposition on leaching. Much of the experimental work has centred on the effects of simulated acid rain on nutrient leaching.

If there is to be nutrient leaching, the deposited acid must exchange a proton for a cation on the soil exchange complex and provide a mobile anion to accompany the exchanged cation (Cook 1983), a complex process. Mobile anions include Cl^-, NO_3^-, SO_4^{2-}, $H_2PO_4^-$, HPO_4^{2-}, and HCO_3^- (Cook 1983). Chloride is the most mobile anion and may be important near coastal areas. Nitrate is of interest because of its mobility and the fact that it is an important macronutrient often limiting plant growth. Sulphate may be a major mobile anion participating in the leaching process. Richter et al. (1983) found a much greater SO_4^{2-} concentration at depth than Cl^- or HCO_3^- in two Tennessee forest ecosystems, suggesting that SO_4^{2-} from atmospheric deposition is the major mobile anion. Singh (1984) showed that in some soils, SO_4^{2-} is strongly adsorbed at low pH (<5.0) and is highly resistant to acid leaching. Under these conditions, inputs of sulphuric acid would have little effect on cation leaching. He also found that ease of sulphate desorption increased with pH, thus agreeing with the characteristics of acid sensitivity. Johnson et al. (1977) found that HCO_3^- was the major mobile anion in the tropics, presumably due to elevated soil respiration, but it was less important further north or in alpine situations. That the situation is complex is perhaps an understatement. Krug and Frink (1983) stated that H_2SO_4 can actually decrease the aluminum in the leachate. Ugolini et al. (1977) stated that the assumption that stream water will reflect the chemistry of the solum may not be valid because effects may originate in the lithological substrate below the root zone or at the surface as a result of erosion; the path of water and its residence time is quite important. With this in mind, the following discussion will cover simulated acid rain and the evidence for leaching under natural conditions.

Morrison (1981, 1983) leached reconstructed jack pine soil profiles with simulated acid rain (H_2SO_4) of pH 2, pH 3, and pH 4. After 4½ years of such treatment, only the pH 2 treatment showed any effects. In that treatment, soil pH declined, sulphate in the leachate increased, and major cations in the leachate increased and then declined. The results suggested that SO_4^{2-} was the major mobile anion (as expected) and that leached cations were from the exchange complex and not due to increased weathering. It should be noted that after 4½ years, pH 3 and pH 4 "rain" had no measurable effect. Abrahamsen et al. (1976) reported that simulated rain of pH 4 did lower soil pH and reduce the extractable calcium, magnesium, and manganese in podzol soils. In some cases, NO_3^- losses occurred. Thus, simulated acid rain can cause cation leaching under experimental conditions. Some of the treatments (e.g., pH 2) may have been far more acidic than that occurring in precipitation.

Vitousek et al. (1979) found NO_3^- losses $\geq$100 μeq L^{-1} in 19 different forest ecosystems when disturbed by trenching and vegetation cutting. Thus, disturbance alone tends to result in loss of NO_3^-, a mobile anion. They made no mention of cations that accompanied the anion, but presumably basic cations could do so. This is interesting from a nutritional standpoint, because Krajina et al. (1973) reported that of five coniferous species studied, only Douglas fir preferred nitrate as a nitrogen source.

Cronan et al. (1978) compared the soil solution and groundwater seepage in a New Hampshire subalpine forest with that from unpolluted areas in Minnesota. Sulphate was the dominant anion in the New Hampshire forest, presumably because of acid deposition, whereas in the unpolluted areas, either organic anions or carbonic acid dominated. In New Hampshire, the lowest pH of 3.64 had 747 μeq L^{-1} cations compared with only 159 in pH 4.70 water from a spring. These results suggested that sulphur pollution can supply protons in exchange for cations and SO_4^{2-} as the mobile anion. Richter et al. (1983) found SO_4^{2-} to be the major anion at depth in a Tennessee forest. However, they felt that weathering and deep-rooted trees would supply the cations lost via leaching. Matzner (1983) did an element balance in ecosystems impacted by acid rain and found that soils with pH's near the aluminum buffer range lost nitrogen through humus disintegration and showed marked reductions in the stored cations Ca, Mg, and Al, and had increased H^+ storage (i.e., reserve acidity). This work clearly suggested leaching effects from acid deposition. Van Breeman et al. (1984) concluded that in some cases, acid deposition rates exceed internal proton generation and toxic aluminum is released. Van Breeman (1985), in a brief review, concluded that atmospheric deposition of anthropogenic sulphur and nitrogen is a major contributor to soil acidification, and that only a reduction in emissions of these pollutants will solve the problem. Paces

(1985), in a very detailed study of small watersheds in central Europe, a region of high industrial pollution levels, concluded that the dry deposition of SO_4^{2-} and fertilizers has acidified the soils and water of the Elbe River basin, which may lead to forest decline. Legge et al. (1981) and Legge et al. (1986) provided evidence that acidification and perhaps leaching have occurred downwind from a sour gas plant in Alberta.

In conclusion, it appears that although soil acidification and nutrient leaching are necessarily complicated because of the complexity of soils, simulated acid rain can cause cation leaching. In recent years, there have been a few studies in Europe and North America that suggest this is happening. Krug and Frink (1983), on the other hand, concluded that in the northeastern US, southeastern Canada, and Scandinavia, acid rain will not measurably enhance leaching of nutrient cations. Thus, in some cases, acidification, cation leaching, and aluminum toxicity are plausible causes of forest decline and are questionable causes in others. The storage mechanisms discussed by Prenzel (1983) are important in that H^+ and Al stored during acid deposition may have acidifying effects for years after acid input ceases, preventing adsorption of base cations on the exchange complex. Thus, the effects of acidification may be a long-term condition not easily ameliorated, especially in forests. More research is needed to clarify this issue.

3.3.2 Iron, Aluminum, and Manganese Solubilization and Phosphorus Deficiency

Krug and Frink (1983) have called attention to the fact that cation exchange sites of silicaceous soils at <pH 5 are nearly saturated with aluminum and that podzols, peaty soils, and oxisols all have aluminum problems. They cited the considerable literature concerning aluminum in naturally acidic soils. It seems clear that the occurrence of soils with soluble aluminum at pH <5 is not evidence for pollution-caused acidic deposition. This is especially true for the Peace River region of Alberta and British Columbia. Hoyt and Nyborg (1972) have shown that soils in that region may be naturally acidic (pH 4 to 5.6) and have elevated exchangeable Mn and Al levels that relate to yield reductions of cultivated barley. For example, a humic Gleysol of pH 4.1 and exchangeable Mn of 48.1 ppm and Al of 20.6 ppm, has only 29% of the yield of the same soil when limed. They stated that a knowledge of pH, Al, and Mn is necessary to predict plant response, i.e., measurement of only one element is not sufficient.

Cook (1983) reported that high aluminum in the soil may precipitate phosphorus as $AlPO_4$. Thus, phosphorus deficiency may in turn be a symptom of elevated aluminum. He further mentioned that in calcareous soils, PO_4^{3-} may increase in groundwaters; one gets very different responses depending upon the soil type. Van Breeman (1985) stated that aluminum toxicity can result from soil acidity. Grant et al. (1979) found that exposure of soils to 10 ppm SO_2 for 5 days increased solubility of iron and manganese but that exposure to NO_2 had no such effects. Thus, pollutants such as SO_2 can increase solubility of aluminum, iron, and manganese, and these can be toxic, perhaps via phosphorus precipitation.

Abrahamsen (1983) sprayed trees with artificial rain from pH 6 to pH 2.0. pH 2 "rain" removed Ca, Mg, and solubilized Al but did not cause any ill effects on Norway spruce, scots pine, and silver birch. His leaching results were similar to those of Morrison (1981). Mayo (personal observation) has noted that birch is a dominant tree in the Peace River region acid soils described by Hoyt and Nyborg (1972), demonstrating that some trees can withstand acid soils with elevated aluminum, resulting in community changes. Ulrich et al. (1980) have documented the following soil changes in a Fagus silvatica community in the Solling area in West Germany. Between 1966 and 1979, pH at the soil surface has fallen from 3.5 to 3.0. Exchangeable aluminum has increased from 0.9 mg L^{-1} to 1.5 mg L^{-1}, and exchangeable iron has also increased, although not so markedly. They attributed these changes to acid deposition and cited them as a cause of forest decline. Johnson and Todd (1984) irrigated a mixed deciduous forest in Tennessee with H_2SO_4 and HNO_3 at 2 times and 10 times the current annual H^+ input and found no effects on extractable phosphorus or aluminum. The results of these studies on acid irrigation are conflicting. It should be noted that Abrahamsen (1983) worked with a Typic Udipsamment, which is a sandy Entisol, i.e., a soil with no developed pedogenic horizons, that has many of the characteristics of a sensitive soil described earlier. On the other hand, Ulrich's work was on a Typic Dystrochrept, i.e., an inceptisol only slightly more developed than an Entisol (also an infertile soil with a very weak B horizon). Both of these soils might be considered sensitive to acid deposition. Johnson and Todd (1984), on the other hand, worked with a Typic Fragiudult, i.e., an Ultisol with a fragipan slowly permeable to water, and having clay accumulation in the B horizon below an E layer which is already leached. It is perhaps due to the differences in soil type, or the residence time of leach water, that the researchers obtained the different results noted above.

Bruce and Riha (1984) measured concentrations of labile and total Al in soil extracts of six forest soil organic horizons acidified with HNO_3. They found that decreases in solution pH of 0.1 to 0.2 units between pH 4.5 and 2.0 caused increases and decreases in concentrations of labile and total Al, respectively. They concluded that a knowledge of pH alone was not enough to predict the response of Al solubility to acidification.

In conclusion, it is possible to acidify and leach cations from some soils and it is possible that soluble aluminum, manganese, and iron can cause problems in some situations, but it is far from clear whether this is the cause of forest decline in every case. Perhaps on sensitive soils it may be, but it does not seem to be a universal cause. Much more attention must be given to soil characteristics.

3.3.3 Acid Effects on Weathering

Johnson et al. (1982b) stated that increased weathering due to acid deposition could offset leaching losses of basic cations. Likens et al. (1977) reported that normal weathering produces 2 Keq $ha^{-1}y^{-1}$ in the US eastern deciduous forest, and 1 Keq $ha^{-1}y^{-1}$ in Norwegian studies, but did not indicate any enhancement of weathering. Morrison's (1983) study of simulated acid rain would suggest that there is little enhancement of weathering in reconstructed jack pine profiles. In that study only pH 2 "rain" had any effect and the abrupt drop in K^+, Ca^{2+}, and Mg^{2+} in the leachate after an initial rapid rise would suggest a clearing of the exchange complex with little enhanced weathering. This study suggested that weathering would not offset leaching losses. Weathering at depth would not offset leaching losses unless deep-rooted species could bring the cations back into the major root zone. This is what happens in hardwood forests where each year's leaf fall adds cations, particularly Ca^{2+}, to the soil surface (Johnson and Richter 1983).

3.3.4 Acid Effects on Microbial Activity

The effects of acid deposition on microbial activity have been reviewed extensively by others in the Acid Deposition Research Program (ADRP) (Visser et al. 1987); the following discussion will therefore be brief. There has been considerable work on the effects of acidification, either simulated or in the field, on soil microorganisms. Much of this work has centred on organism number estimates, decomposition (or respiration), nitrogen fixation, and nitrification. What follows is a brief review of a topic of considerable importance in the whole field of acid deposition research.

3.3.4.1 Effects of Acidification on Populations of Soil Organisms. Wood et al. (1984) reported that 60% of the bacteria and fungi are in the upper 10 cm of the forest soil, with the greatest numbers in the Ao horizon, the surface layers. Therefore, the soil layers most likely to be affected first by acidification also contain the most microorganisms. Leetham et al. (1982) found that in a northern prairie soil, nematodes and rotifers were found in the surface soil layer (0-10 cm) and tardigrades were restricted to the 0-2 cm layer. If forest soils are similar, then these organisms would also be affected by surface layer acidification. Leetham et al. (1982) found that SO_2 fumigations of <1 pphm reduced tardigrade populations significantly. Nonstylet nematode populations were also reduced, whereas stylet bearing nematodes and rotifers were unaffected.

Firestone et al. (1984) found that pH 2 acid "rain" inhibits *Aspergillus flavus* spore germination, which could affect the population of organisms. In a greenhouse experiment, Patten (1983) watered 3 soils (a silt loam and two loams) with pH 2.5, 4.0, and 5.0 for 20 or 52 weeks, and then did plate counts to determine the relative abundance of bacteria and fungi. After 20 weeks there were fewer bacteria and more fungi in the pH 2.5 treatment than in the 5.0. This was to be expected since bacteria are generally less tolerant of acidic conditions than fungi (Alexander 1977). However, after 52 weeks of treatment there were no differences in numbers of fungi, bacteria, or spores of mycorrhizal fungi even though the pH 2.5 treatment lowered soil pH by 0.8 pH units. Patten (1983) concluded that the organisms had adapted to acid conditions. This is important since adaptation under field conditions may be quite different from short-term laboratory experiments. It appears that microorganism populations may be affected by acid deposition. However, the results are often contradictory, indicating the need for much more work on this important problem.

3.3.4.2. Effects of Acidification on Decomposition, Respiration, and Soil Enzymes. Abrahamsen et al. (1976) stated that microbial activity, including that of decomposers, is less at pH levels <5.0. Cook (1983) stated that reduced decomposition due to acid deposition would result in increased organic matter retention. Thus, reduced decomposition is indicated by organic matter buildup, reduced soil respiration, reduced decomposition of study substrates, or altered enzyme activity. Abrahamsen et al. (1976) reported previous work in which "acid rain" (pH 5.6 to 3.0) did not influence the decomposition rate of withered pine needles, cellulose sheets, or aspen match sticks after one to two years of exposure. In 1978, Abrahamsen et al. (1978) reported that decomposition of lodgepole pine needles was increased. It should be noted that a buildup of organic matter would hardly be detected by looking just at needles alone. For example, Tamm et al. (1976) observed reduced CO_2 evolution from artificially acidified forest soil and concluded that microbial activity was reduced. Killham et al. (1983) found that pH 2.0 simulated rain inhibited both respiration and several enzymes such as urease, phosphatase, dehydrogenase, and aryl sulphatase. Rain of pH 3.0 and 4.0 stimulated respiration and caused varied enzyme responses. Thus, the pH of artificial acid rain is quite important, and only very acidic rain inhibited soil respiration. Bääth et al. (1980) showed a significant increase in organic matter when a pine forest was artificially acidified. Both needle and root litter decomposition were less under acid treatment. Fungal hyphae length ($m\ g^{-1}$) was reduced as was bacterial biomass. The springtail population increased significantly. Chang and Alexander (1984) found that in general, simulated acid rain of pH 3.5 reduced decomposition and organic matter leaching from soils of three watersheds. There were some exceptions in which specific treatments stimulated decomposition when organic matter leaching was also stimulated. All of the work listed above was experimental, using artificially acidified systems. It shows that under certain (often quite harsh) conditions, decomposition can be reduced.

Data from acid impacted forests are perhaps more definitive. Ulrich et al. (1980) found that between 1966 and 1979, carbon stored in the organic layer increased from 14,800 to 22,300 kg ha^{-1} and stored nitrogen increased from 809 to 1010 kg ha^{-1}. This occurred while soil pH decreased from 3.5 to 3.0. They concluded that acidification of the organic layer had reduced decomposer activity and caused the increase. It should be noted that the Solling, West Germany, soil studied is considered to be sensitive to acidification. Matzner (1983) found that there was reduced nitrogen storage in the humus layer of beech and spruce forests due, presumably, to humus breakdown. Cook (1983) stated that nitrogen loss would be expected when a soil is acidified because of NO_3^- leaching and increased N_2O flux to the atmosphere. It seems clear that decomposition can be inhibited by acid deposition. The Solling experience (Ulrich et al. 1980) demonstrated that. Variation in factors such as precipitation pH, substrate materials, or soil conditions might negate this effect, however. Careful monitoring of soil organic matter over long periods of time is necessary, because effects on decomposition may be difficult to detect and may be site specific.

3.3.4.3 Effects of Acid Deposition on Nitrogen Fixation. Because nitrogen levels are generally low in many forest soils and nitrogen fixing activity is also often low (Abrahamsen et al. 1976), anything which reduces nitrogen fixation could have profound effects on the ecosystem. This is especially true if nitrogen fixation plays a key role in the nitrogen cycle and subsequent plant productivity (Jones and Gay 1985; Cronquist-Jones 1985). Table 6 summarizes four major nitrogen-fixing systems that can supply nitrogen to forests and gives example of references suggesting acid effects, with one exception. The work of Norby et al. (1985) shows that SO_2 + O_3 will inhibit N-fixation

Table 6. Types of nitrogen fixation and the effects of acid deposition.

Type of Nitrogen Fixation	Evidence for the Effect of Acid Deposition	Reference
Symbiotic nodule-bacteria	Poor legume growth in acid soils	Tabatabai (1985)
	High Al and Mn, and low Mo in acid soils	Alexander (1980)
	Inhibition of nodulation in kidney bean and soybean	Shriner (1976)
	Inhibition of infection by NH_4^+ nitrogen	Dazzo & Brill (1978)
	Low N-fixation in Norwegian forests.	Hovland (1976)
	N-fixation inhibited in soybean by SO_2+O_3 but not acid *per se*	Norby et al. (1985)
Free-living N-fixing bacteria	The lack of *Azospirillum* in naturally pH 3.8 soils	Giberson (pers. comm.)
	Low N-fixation in Norwegian coniferous soils	Hovland (1976)
	Review of nonsymbiotic nitrogen fixers in acid soils. *Azotobacter* is intolerant *Beijerinckia* is tolerant	Jurgensen & Davey (1970)
Cyanobacteria	Reduced algal N-fixation in forest soils	Chang & Alexander (1983)
Lichen	Acid "rain" reduces N-fixation by *Lobaria oregana*	Denison et al. (1977)
	Direct effects of SO_2 on lichens	Nieboer et al. (1976)

but acid deposition *per se* has no effect. They did not say whether the artificial acid rain had any effect on soil pH. Thus, it is not clear whether soil acidification actually occurred. Not all of the references in Table 6 are for forest species. Nitrogen fixation is a high energy-requiring process, and anything that reduces photosynthesis will reduce nitrogen fixation. Phillips (1980) has reviewed the efficiency of nitrogen fixation by legumes, and although quite variable, it takes approximately 6 g carbon (which must come from photosynthesis) to fix 1 g N. Thus, anything that affects carbon fixation, directly or indirectly, will affect N-fixation. The depressing effects of acid deposition on photosynthesis have been discussed previously. Shriner (1976) has shown that legume nodule formation itself can be inhibited by acid deposition. Dazzo and Brill (1978) have shown that nodule formation, an infection process by *Rhizobium*, is very sensitive to the form of nitrogen present. Using the clover-*Rhizobium trifolii* system, they showed that 1 mM NH_4^+ reduces infection to zero, whereas maximum infection occurs with 5 mM NO_3^- and does not reach zero until 15 mM. This suggests that the effect of acid deposition may be through its effect on nitrification rather than directly on nitrogen fixation.

Alder stands in Oregon have pH 3.9 soils with 5 ppm NH_4^+ and 67 ppm NO_3^-, whereas adjacent conifer stands have pH 5.3 soils with 25 ppm NH_4^+ and 24 ppm NO_3^- (Bollen and Lu 1968). This supports the work of Dazzo and Brill (1978) in suggesting that nodule formation (which is a characteristic of alder) requires more NO_3^- than NH_4^+ and may be related to nitrification. It is also possible that alder preferentially absorbs NH_4^+. This has been reported for some trees, especially conifers (Miller 1983). However, as shown later in Table 8, nitrification under alder can proceed rapidly when the soil is amended with NH_4^+. This also helps to explain why the H^+ production associated with nodule nitrogen fixation does not inhibit the infection process (Tabatabai 1985).

Chang and Alexander (1983) showed that soil algal nitrogen fixation is reduced by pH 3.5 "rain", perhaps through reduced photosynthesis because CO_2 fixation is also reduced. Lichens are notoriously sensitive to pollutants (Nieboer et al. 1976). Thus, it is not surprising that acid deposition reduces lichen nitrogen fixation (Denison et al. 1977). This could have serious consequences for the pine-lichen woodlands found in northern Alberta.

Although the mechanism is unclear, there is little doubt that acidification can reduce nitrogen fixation. The effect of too much nitrogen from pollution will be discussed under fertilizer effects in Section 4.1.

3.3.4.4. Effects of Acid Deposition on Nitrification and the Form of Soil Nitrogen. The potential effect of nitrification upon nitrogen fixation was mentioned in the previous section. This section will deal with the effects of acid deposition on nitrification. There is some question about the occurrence of nitrification in some forest soils, even in the absence of anthropogenic acid deposition. Belser (1979) has reviewed the population ecology of nitrifying bacteria and concluded that nitrogen cycling is quite efficient in climax forest ecosystems. He also discussed the hypothesis of Rice (1974) and Rice and Pancholy (1972) that nitrification may be inhibited in climax ecosystems and concluded that it is an unproven, although important, hypothesis. Ammonium as a cation is more resistant to leaching than nitrate (Belser 1979); thus, inhibition of nitrification would conserve nitrogen. However, Belser (1979) also stated that a relatively high ammonium concentration in the soil could be due to rapid nitrate uptake or to leaching. Indeed, much emphasis has been placed upon measuring nitrification potential (Robertson 1982; Vitousek et al. 1982) rather than nitrate and ammonium pools. This is usually done using dried, ground, and sieved soil samples; thus, it is not surprising that the results correlate poorly with factors known to affect nitrification such as pH, C:N ratio, oxygen, moisture, and temperature (Robertson 1982; Vitousek et al. 1982). No attempts have been made to evaluate nitrification in situ. Belser (1979) also emphasized that different responses to adverse conditions could affect the form of nitrogen in the soil. High soil temperatures inhibit nitrification more than ammonification although such high temperatures (40°C) are not likely in many forest soils. Low soil temperatures also reduce nitrification more than ammonification, resulting in higher ammonium levels. This is more likely in forest soils, especially in northern regions. Regardless of the causes, there exist considerable differences in the amounts of NH_4^+ and NO_3^- found in various ecosystems. Table 7 summarizes the relative amounts of these nutrients found under various conditions. Note the data for red alder, a nitrogen fixer. Bollen and Lu (1968) showed more NO_3^- than NH_4^+, whereas Vitousek et al. (1982) reported just the opposite. It may be that nutrient poor sites (Morrison and Foster 1977, Table 7) simply do not support nitrifiers for reasons other than pH.

It is equally true that many species are selective as to the nitrogen form required for growth. Ellenberg (1977) reported that _Mecurialis_, _Campanula_, and _Carex alba_ die when grown on only NH_4^+, and _Calluna_ and _Vaccinium_ do very poorly on NO_3^-. Krajina et al. (1973) found that _Pinus contorta_ has greater root and shoot growth on NH_4^+ than on NO_3^- or a mixture of the two, Douglas fir does best on NO_3^-, and western red cedar does much better on NH_4^+. Alexander (1983) stated that _Dicea sitchensis_ does best on NH_4^+. That plants vary in their response to ammonium has been known since 1935 (Pardo 1935). If nitrification is severely limited by acid deposition it could have undesirable effects on communities of Douglas fir and western red cedar. Since several of the communities have more soil NH_4^+ than NO_3^- (Table 7) and the dominant species such as pine may have a requirement for NH_4^+, it does not seem likely that preferential absorption of NO_3^- explains the relative amounts in the soil. It has also been reported that NO_3^- may inhibit mycorrhizal infection of some species. Alexander (1983) showed great reductions of root tips converted to mycorrhizas in _Pseudotsuga menziesii_ and _Picea sitchensis_ when grown with NO_3-N compared with NH_4^+-N. Thus, all of the nutritional aspects, such as P absorption, associated with mycorrhizae may be affected by the form of nitrogen. It is also important to remember that with some systems, NH_4^+ is inhibitory to symbiotic nodule formation, thus explaining why symbioses are associated with certain vegetation communities (Table 6). Clearly, the form of nitrogen is important and much work needs to be done to sort out species requirements, the natural causes of amounts of NH_4^+ and NO_3^-, and the effects of acid deposition.

As stated earlier, the amounts of NH_4^+ and NO_3^- in the soil may not reflect the rates of nitrification since mineralization, uptake, leaching, and denitrification may affect pool sizes. Table 8 gives references to nitrification in various soils and plant communities. Not all of these are forests. The following points can be found in Table 8: Clearly nitrification can take place at low soil pH (Weber and Gainey 1962; Bollen

Table 7. Ammonium and nitrate nitrogen pools in forest soils.

Forest Community Location	Amount or Relative Amount NH_4^+	NO_3^-	Conditions	Conclusions	Reference
Alder - Oregon Conifer - Oregon	5 ppm 25 ppm	67 ppm 24 ppm	pH 3.9-samples in March pH 5.3-samples in March	rapid nitrification slow nitrification	Bollen & Lu (1968)
Calluna - Scotland *Pinus* & *Picea* *Betula* - Scotland	90-100% 75-100% 50-95%	0-10% 0-25% 5-50%	samples in spring, summer, autumn, winter	75% of the nitrogen used by trees in Scotland is ammonium	Miller (1983)
Jack pine - Canada	large amts NH_4-N	insignificant NO_3-N	fertilizer experiment 300 kg urea/ha 3rd year	insignificant nitrate found - accounted for ~ 70% applied urea-N	Morrison & Foster (1977)
Pine - Sweden	140 µeq L^{-1}	0	fertilization in the field, control concentration given	no nitrate was detected in any soils	Söderström et al. (1983)
Ponderosa pine, spruce, subalpine fir, Pacific silver fir	high	no increase in lysimeter concentration <100µeq L^{-1}	disturbance does not result in increased nitrification	NH_4^+ pool is greater in low fertility sites	Vitousek et al. (1979)
Alder Ponderosa pine Indiana maple Northern hardwoods	43 ppm 28 ppm 90 ppm 81 ppm	12.5 ppm 1.4 ppm 11.9 ppm 7.0 ppm	FF horizon FF horizon FF horizon FF horizon	NH_4^+ pool large NH_4^+ pool large NH_4^+ pool large NH_4^+ pool large	Vitousek et al. (1982)
Review			NH_4-N common form in acidic soils		Rorison (1980)

Table 8. Rates of nitrification and conditions affecting rates.

Forest Community/Location	Reference to Rate of Nitrification	Conditions	Conclusions	Reference
Alder - Oregon	0.3%	F layer (=Oe) nitrification over 28 days unamended	pH of alder 3.5	Bollen & Lu (1968)
Conifer - Oregon	0.8%	F layer (=Oe) nitrification over 28 days unamended	pH of conifer 4.3	
Alder - Oregon	293.5%	F layer amended with ammonium sulphate + $CaCO_3$	Rapid nitrification in alder: greater than conifer	
Conifer - Oregon	34.0%	F layer amended with ammonium sulphate + $CaCO_3$ No pollution		
Northern hardwood Hubbard Brook	0 to 600 kg ha^{-1}	Increased NO_3^- export due deforestation	Loss from the watershed may not be increased nitrification but shows considerable NO_3^- loss	Likens et al. (1978)
Spodosols - New York	nitrification markedly reduced	Acid rain pH 3.5, 8x natural	Acid rain <u>can</u> reduce nitrification	Novick et al. (1984) Klein et al. (1984)
Jack pine - Ontario	"negligible nitrification"	300 kg/ha urea - soil pH 4.4 to 4.6	Very little nitrification potential	Morrison & Foster (1977)
Grassland - Kansas	significant nitrification	Applied AlH_4SO_4 to silty clay loam down to pH 3.7	Significant nitrification in grasslands at pH 3.7	Weber & Gainey (1962)
Conifers - Great Britain	"nitrification is infrequently important"	Not given except unpolluted areas	Little nitrification	Malcolm & Garforth (1977)

continued...

Table 8 (Concluded).

Forest Community/Location	Reference to Rate of Nitrification	Conditions	Conclusions	Reference
Tropical Rain Forest	10 to ~70 ppm NO_3^-	Wet season, soil incubated 20 days in the field	Increased NO_3^- with succession due to increased N high nitrification	Lamb (1980)
Grassland - Georgia	low nitrification	Ammended soil with NH_4SO_2 pH 4.6 to 4.8. Aluminum as high as 273 ppm	Low pH and soluble Al inhibit nitrification	Brar & Giddens (1968)
Hardwood forests Wide range of locations	$\geq$100 μeq L^{-1} in lysimeters	Trenched plot experiments. Measurements of NO_3^- in lysimeters. Many soils pH 3.1 to 4.0	Disturbance increases nitrification	Vitousek et al. (1979)
Ponderosa pine Spruce-subalpine fir Pacific silver fir	<100 μeq L^{-1} in lysimeters	Disturbance did not result in increased nitrification	NH_4^+ pool is greater and nitrification potential is less in less fertile sites	Vitousek et al. (1979)
Elm, Sycamore, Hackberry, red oak white oak - Missouri	5 to 16 ppm NH_4^+ 2.00 ppm NO_3^- up to 4m from tree	Samples taken in spring prior to growth season	Difference is due to reduced nitrification not growth or loss of NO_3^-	Lodhi (1977)

and Lu 1968; and Vitousek et al. 1979). It is equally true that in some soils, acidification results in reduced nitrification (Brar and Giddens 1968; Novick et al. 1984). Disturbance tends to result in increased nitrification (Likens et al. 1978; Vitousek et al. 1979), and quite a few forest soils have naturally low nitrification rates (Bollen and Lu 1968; Brar and Giddens 1968; Lodhi 1977; Malcolm and Garforth 1977; Morrison and Foster 1977; Klein et al. 1984; and Novick et al. 1984). The whole area of nitrogen-fixation and nitrification is important and needs further work to sort out the importance of nitrogen form, rates of natural nitrification, and the effects of acid deposition.

3.3.4.5 Effects on Mycorrhizae. This important subject has been reviewed in depth by others in the ADRP (Visser et al. 1987), and therefore, little will be said here. Mycorrhizae play a very important role in nitrogen uptake (Fogel 1980), phosphorus uptake (Pearson and Tinker 1975), disease resistance, and detoxification (Alexander 1983). Alexander (1983) has reviewed the role of mycorrhizae in forest ecosystems and concluded that it is highly critical. Patten (1983) found that after 52 weeks of treatment with acid rain of pH 2-5 there was no reduction in mycorrhizal spore production. She felt that they had adapted to acid conditions. In view of the very important role of mycorrhizae, it is essential to do more research on these important organisms.

3.3.5 Summary of Indirect Effects

The possibilities for indirect effects of acid deposition on forests are almost limitless. From the moment that acidic materials come into the canopy layer a variety of effects is possible. The canopy itself exchanges and modifies precipitation in a variety of ways (as given in Table 5), such that whatever reaches the ground may be different in pH, acid form, and ionic concentration from the bulk precipitation. Upon entering the soil (including the organic layer) precipitation is subjected to what is, in effect, a large, variable and reactive exchange column. The soil can be acidified with solubilized Al, Mn, and Fe, increasing to toxic levels. Cations may be leached, and microbiological processes may be altered. The whole nitrogen budget may be altered. Nodule formation may be inhibited, nitrogen fixation may be altered, and nitrification may be inhibited. The size and proportions of the nitrate-ammonium pools may change. Mycorrhizal relations may be altered. The inescapable conclusion is that there is a variety of possibilities, that these are site- and situation-specific, and that each individual situation may require specific research. There is little doubt that these kinds of effects can take place. Whether or not they take place under specific conditions, and their importance, remain to be clarified.

4. INTEGRATIVE EFFECTS OF ACID DEPOSITION

4.1 FOREST NUTRITION AND GROWTH

There is no doubt that forests in many regions of the world are growing poorly or are suffering serious decline (Smith 1981; McLaughlin 1985). The previous sections have discussed the various ways that pollutants and acid deposition can affect forests. These vary from direct effects upon photosynthesis to indirect effects such as cation leaching, aluminum toxicity, and reduced microbial activity. The fact remains that although these responses should result in reduced forest growth and productivity, there is evidence that this may not happen. In fact, SO_2 and NOx can act as fertilizers if the forest should be limited in available S and/or N. There is also the interaction between the effects of acid deposition and forest harvest techniques. These effects have to be evaluated with respect to nutrient cycles that normally occur in forest ecosystems. There are several recent reviews on nutrient cycling, fertilization, and harvesting interactions. Abrahamsen (1980) has reviewed the potential relationships between acid deposition and plant nutrition. He reminds us of the law of the minimum which states that only one factor can be limiting at a given time and that tree growth is usually limited by nitrogen. Thus, adding sulphur will not promote growth and may cause problems even though it is essential to plants. Also, fertilization with one element may cause deficiencies in others. Nihlgard (1985) proposed that such is the case with NH_4^+, suggesting that it is the cause of forest dieback. Other reviews of nutrient supply and nutrient cycling include those of Khanna and Ulrich (1984), Miller (1984), and Gosz (1984). They discussed the factors involved with nutrient cycling and the problems of managing forests. In spite of these data, there exists doubt as to the actual cause of forest decline. For example, Tabatabai (1985) concluded that the buffering capacity of soils minimizes pH changes, that N and S additions are beneficial, and that experiments with simulated acid rain "...will not provide the information needed". He feels that acid deposition near point sources of pollution deserves further investigation. Table 9 is a summary of specific references to neutral or beneficial responses to pollutants, and Table 10 references specific detrimental effects. While these tables undoubtedly do not represent all of the numerous articles on the subject, they do illustrate the fact that there is disagreement. The fact that there are numerous articles referring to little effects or even stimulation effects indicates just how site-specific forest response to acidic deposition can be.

4.2 HARVESTING AND ACID DEPOSITION EFFECTS

Management can have a considerable effect on nutrient cycles within the forest ecosystem, and the interaction with acid deposition could be important. Variables include the type of harvest, such as whole tree or bole only, and the length of rotation between harvests. An 80 year rotation with only saw log removal will be very different from a short rotation whole-tree harvest biomass system of management. The act of harvest disturbance can itself result in nutrient loss from the ecosystem. Vitousek et al. (1979) have shown that disturbance results in increased nitrification and nitrate leaching.

Table 9. References to no effect or improved plant response due to acid deposition.

Species or Community	Location	Nutrient or Pollutant	Response	Treatment or Condition	Conclusion	Reference
Ponderosa pine	Oregon	S	Increased growth 50%	224 kg ha^{-1} S	Pumice soils are S deficient	Will & Youngberg (1978)
Corn	US	N	Beneficial	3-10% of N necessary for corn crop	Nitrogen deposition fertilizers	Tabatabai (1985)
Several crops	US	S	Reduced deficiencies	2-20 kg ha y^{-1} average	Probably prevents S deficiency	Tabatabai (1985)
Mixed hardwood	Eastern US	N	Fertilizer	40% annual requirement		Lindberg et al. (1986)
Mixed hardwood	Eastern US	S	Fertilizer	100% annual requirement	S excess may be detrimental	Lindberg et al. (1986)
Picea abies	Norway	-	All treatments grew well	3 y irrigation with pH 5.4, 4.0, 3.0, 2.5 "rain"	Clones varied responses N limiting factor	Ogner & Teigen (1980)
Crops - wheat - alfalfa	US	SO_2	Increased growth	Low soil S	Beneficial if soil S is low	Cowling & Koziol (1982)
Sunflower, corn	US	NO_2, NH_3	Increased growth	Low soil N	Beneficial if soil S is low	Cowling & Koziol (1982)
Norway spruce Scots pine Silver birch	Norway	Acid rain	Little effect on height or diameter growth	Sprayed with pH6 to 2.5 artificial rain	Trees did not suffer aluminum toxicity	Abrahamsen (1983)

continued...

Table 9 (Concluded).

Species or Community	Location	Nutrient or Pollutant	Response	Treatment or Condition	Conclusion	Reference
Pinus strobus	US	HNO_3 "acid rain"	Increased seedling growth K^+, Mg^{2+}, Ca^{2+}, leached	20 week greenhouse experiment. pH 5.6 to 2.3	Growth increased but leaching suggests it could not be maintained	Wood & Bormann (1977)
Liriodendron tulipifera	US	"Acid rain"	No effect on growth	30 month microcosm expt. with restructured forest soil-profile-treated with acid pH 5.7 4.5, 4.0, 3.5	No effect on tree physiology but some cation leaching	Kelly et al. (1984)
Quercus alba	US		No effect on growth			
Pinus virginiana	US		No effect on growth			

Table 10. References to negative tree response due to acid deposition.

Species/Community	Location	Nutrient or Pollutant	Response	Treatment or Condition	Conclusion	Reference
Picea abies, Larix	Bavaria	Mg, Ca, K, Zn	Low Mg & relatively low Ca, K, Zn in needles	S concentration in toxic range, up to 238 ppm Al in needles	Mg deficiency from acid deposition affects trees	Zech et al. (1985)
Forests	Europe	NH_4	Soft growth	Ammonia from agriculture, industry, etc.	Decreased frost hardiness. Increased insect & disease attack. Nitrogen stress	Nihlgard (1985)
Lodgepole pine	Canada	S-gas	Reduced growth	Downwind from a sour gas plant approx. 15 y	S-gas has reduced photosynthesis and growth	Legge et al.(1976) Legge et al.(1978) Legge (1980)
Red spruce	US	S, Al	Poor foliage condition	S content of poor foliage .12%, while good condition foliage is .10%	Higher aluminum at Camels Hump site	Johnson et al.(1982)
Forest ecosystem	Germany		Dieback		Acidification of the soil and aluminum toxicity	Ulrich - numerous articles - see references cited
Green ash, Paper birch, Red pine	Greenhouse	SO_2	Relative growth rate (RGR) of green ash not affected. RGR of birch lowered. RGR of pine roots lowered	Greenhouse fumigation at different temperatures	Resistance varied with temperature	Norby & Kozlowski (1981a)

continued...

Table 10 (Concluded).

Species/Community	Location	Nutrient or Pollutant	Response	Treatment or Condition	Conclusion	Reference
Green ash Eucalyptus *Pinus resinosa*	Greenhouse	SO_2	Growth reduced except for green ash	SO_2 at 0.0, 0.2, 0.35 or 0.5 ppm for 30 h at 22°C	Post fumigation temperature had no effect	Norby & Kozlowski (1981b)
Scots pine	UK	O_3 and acid mist	O_3 reduced fine root biomass & increased senescence	56 day exposure	O_3 caused nutrient leaching from leaves, not likely a cause of forest decline	Skeffington & Roberts (1985)
Scots pine, Norway spruce	Sweden	S deposition	Reduced growth	Field soil susceptible to acid deposition	Conclude no reason to attribute reduced growth to anything other than acid deposition	Jonsson (1977)
Seedlings of Jeffrey pine, Monterey pine, Shore pine, Lodgepole pine, Sugar pine, Western white pine, Ponderosa pine, Douglas fir	Oregon	O_3	Only ponderosa & western white pine growth reduced	O_3 exposure chambers outdoors for 22 week to seedlings	Increased seedling emergence, necrosis in all species	Wilhour & Neely (1977)

Johnson and Richter (1984) have reviewed the effects of harvesting and pollution on several forest ecosystems in the US, and in Solling, West Germany. Commercial clear-cutting results in increased NO_3^- and Ca^{2+} loss by leaching. They also reported that whole tree harvesting generally removes more N, S, Ca, K, and Mg from the ecosystem than bole-only removal. In other words, whole tree removal depletes nutrients more than does taking only saw logs. They also showed that sulphur input in areas receiving pollution usually exceeds leaching and harvest removal, even when the whole tree is taken. This is not always the case with nitrogen; leaching and harvest can greatly exceed inputs, and whole tree harvesting is, in this case, much more detrimental than bole-only removal (Johnson et al. 1982a; Johnson and Richter 1984). They also concluded that both harvesting and acid deposition can result in base cation loss and that harvesting would have the greater impact. These kinds of data raise serious doubts as to the sustainability of short rotation biomass energy plantations such as have been proposed in recent years, and show that rotations must be tailored to the rate at which nutrients become available through atmospheric inputs, biological activity, and weathering.

4.3 EFFECTS OF ACID DEPOSITION ON REPRODUCTION

If acid deposition affects pollen germination, seed set, and seed germination, it could have profound effects on forest reproduction and community structure. While few references to reproductive biology were found, the ones cited here suggest the need for further work in this important area. These are summarized in Table 11.

Generally, pollutants reduce pollen germination and tube growth (Houston and Dochinger 1977; Bonte 1982; Cox 1983; DuBay and Murdy 1983; Cox 1984; and Van Ryne and Jacobson 1984). However, DuBay and Murdy (1983) found that a reduction of 50% in pollen germination did not reduce seed set. This was a study of _Lepidium_, which is a crucifer and not a tree. If this were true for trees, then reduced pollen germination would be of little significance as far as forest reproduction is concerned. As shown in Table 11, there is considerable evidence for pollution-caused reduction in cone size, cone weight, seeds/cone, and seed weight (Scheffer and Hedgecock 1955; Miller 1973; Smith 1981; and Bonte 1982). Seed germination has been inhibited by high pollution levels (SO_2) and by acid rain in red pine (Bonte 1982) and _Acer rubrum_ (Raynal et al. 1982). Seed germination and/or emergence has been stimulated by simulated acid rain in Douglas fir (Lee and Weber 1979) and in white pine (Raynal et al. 1982). Seedling emergence has been stimulated in Ponderosa pine by O_3 (Wilhour and Neely 1977). Seed emergence has been inhibited in sumac, _Acer rubrum_, and in _Betula lutea_ by acid precipitation (Lee and Weber 1979; Raynal et al. 1982). Finally, acid "rain" (pH 3.0) had no effect upon seed germination of _Acer saccharum_ and _Tsuga canadensis_ (Raynal et al. 1982). It appears that acidic deposition as well as pollutants such as SO_2 and ozone can affect reproduction; however, these effects are quite varied. Further study of the reproductive processes and possible community effects would prove fruitful. Until these are understood, the effects of acid deposition on population and community ecology will be difficult to ascertain.

4.4 ACID DEPOSITION EFFECTS ON PLANT COMMUNITIES

Most of the acid deposition research has centred on tree responses individually or as a forest. While few studies of forest plant communities were found, studies of

Table 11. Effect of pollutants on reproductive biology.

Species	Location	Pollutant	Response	Condition	Conclusion	Reference
White pine		high pollution (SO_2?)	55 seeds/cone. Avg. seed wt 1.532 g. Pollen germination 60%	Area with high pollution	Areas with high pollution had fewer seeds/cone, lower average seed weight, lower % pollen germination	Houston & Dochinger (1977)
		low pollution	67 seeds/cone. Avg. seed wt 1.850 g. Pollen germination 83%	Area with low pollution		
Red pine		high pollution (SO_2?)	Avg. cone length, 44 mm Avg. seed wt 0.666 g Filled seed 50% Seed germin. 50% Pollen germin. 65% Avg. pollen tube length 92 mm	Area with high pollution	Areas with high pollution had fewer seeds/cone, lower avg. seed weight, lower % pollen germ.	Houston & Dochinger (1977)
		low pollution	Avg. cone length 47 mm Avg. seed wt .805 g Filled seed 68% Seed germ. 66% Pollen germ. 84% Avg. pollen tube length 113 mm	Area with low pollution		
Scots pine	Russia	SO_2	Avg. seed wt 4.85-5.75 g Cone length 32.1-35.6 mm Cone wt 2.39-3.03 g	Near copper smelter	Near copper smelter (high SO_2), lower cone length, cone weight, and avg. seed weight	Smith (1981)
			Avg. seed wt 5.47-6.67 g Cone length 39.5-42.4 mm Cone wt 3.81-5.91 g	Clean (low SO_2)		
Ponderosa pine		O_3	Reduced cone size			Miller (1973)
Ponderosa pine	British Columbia	SO_2	Reduced cone size	Near copper smelter		Scheffer & Hedgecock (1955)

continued...

Table 11 (Continued).

Species	Location	Pollutant	Response	Condition	Conclusion	Reference
Douglas fir	Oregon	"acid rain" pH 3 to 5.6	Seed emergence stimulated	Field conditions ambient rain excluded	Stimulation of Doug. fir emergence and growth emergence of hickory, maple, white pine, cedar, & birch	Lee & Weber (1979)
Sumac	Oregon	"acid rain" pH 3 to 5.6	Seed emergence inhibited			
Dogwood, Black birch, Yellow birch, Sugar maple	NewYork	pH 5-2.6 "rain"	Reduced pollen germination below pH 4.2	Lab study	Acid rain may affect reproductive processes	Van Ryne & Jacobson (1984) Van Ryne et al.(1986)
Oenothera	Canada	pH 5.6-2.6	pH $\leq$3.6 inhibited pollen germination and tube growth. Reduced stigmatic receptivity	Field plants		Cox (1984)
White pine <u>Acer rubrum</u> <u>Betula lutea</u>	New York	pH 5.6,4.0,3.0	Stimulated seed germination. Inhibited seed germination at pH 4	Greenhouse	Response varies from stimulation to inhibition of seed germination	Raynal et al. (1982)
<u>Acer saccharum</u>			No effect			
<u>Tsuga canadensis</u>			No effect			
Ponderosa pine	Oregon	O_3	Seedling emergence stimulated			Wilhour & Neely (1977)
<u>Lepidium virginicum</u>	-	SO_2	Reduced pollen germination 50% <u>but</u> not seed set		.6 ppm SO_2	DuBay and Murdy (1983)

continued...

Table 11 (Concluded).

Species	Location	Pollutant	Response	Condition	Conclusion	Reference
Ontario		"acid rain"	50% inhibition in pollen growth		Conifers most resistant, deciduous trees intermediate	Cox (1983)
Acer saccharum		3.9				
Populus tremuloides		3.7				
Betula papyrifera		3.6				
Pinus strobus		3.6				
P. banksiana		3.2				
Tsuga canadensis		2.95				

populations and ecotypes exist, and the changes in reproductive biology described in the previous section would suggest that varied responses leading to community changes are possible. Law and Mansfield (1982), in a greenhouse study of NOx, concluded that variable uptake rates of nitrogen may have been due to different varietal resistance. Garsed and Rutter (1982) screened populations of _Pinus_ _sylvestris_, _P_. _nigra_, and _P_. _contorta_ and found wide differences in sensitivity to SO_2 among species as well as within species, suggesting that pollution could exert a strong selection pressure. McClenahen (1978) found that chronic pollution reduces the diversity index in deciduous forests, i.e., it simplified the community. All of these observations suggest that chronic low level pollution could affect forest community diversity and perhaps even dominance (i.e., change the climax). Pielou (1982) tested the species composition along potential pollution gradients in the Athabasca River valley (northeastern Alberta near the SUNCOR oil sands plant) and found no evidence of changes in species composition.

Considerably more work needs to be done on low level pollution. We know that wherever dieback is occurring, the forest community is changing rapidly (Postel 1984a-d).

It has already been mentioned that there are those who feel that secondary succession is an acidifying stage and that changing land use involving succession back to conifers is a major cause of soil acidification (Rosenqvist 1978; Overrein et al. 1980; and Krug and Frink 1983). Thus, instead of acid deposition altering succession, succession may be the cause of soil acidification.

4.5 SUMMARY

In conclusion, it is apparent that forests are declining in parts of Europe and eastern North America for a number of possible reasons. This phenomenon appears to be an integrated response at the population and community levels of organization. Although the author is not a modeller, it is suggested that the only way to understand what is happening is to adopt a hierarchical modelling approach and then test the models in the real world. Mathematical models exist for a variety of plant processes and could be adapted for pollution research. For example, Thornley (1976) described the modelling of light interception, photosynthesis, growth, and photosynthate partitioning. A nutrient model (Reuss 1980) was developed specifically for acid leaching, and it seems possible to test such a model. More appropriate models likely exist, however. Adaptation of various process models, accompanied by field testing, appears to be a good approach for resolving some of the confusion about the effects of acid deposition on forests. An excellent review and analysis of numerical models of air pollutant exposure and vegetation response has been prepared by Krupa and Kickert (1987a,b). Table 12 lists examples of various models that might be adapted. It should be noted that most of the research reviewed in Sections 2 and 3 is such that it does not lend itself to modelling. This is to say that if modelling is to be undertaken, then appropriate research must be carried out. The fragmentary nature of most of the research to date does not provide the mathematical relationships demanded by a modelling approach. Continuous long-term monitoring of the forests and variables regulating growth will eventually provide answers.

Table 12. Simulation models used or adaptable to acid deposition studies.

Process Modelled	Model Characteristics and Comments	Reference
Nutrient loss due to acid rain	Takes into account lime potential, equilibrium between CO_2 and H^+ and HCO_3^-. Computes ionic composition in leachate H^+, Ca^{2+}, Al^{3+}, SO_4^{2-}, Cl^-, and HCO_3. The model is for noncalcareous soils.	Reuss (1980)
Resource based competition	Written as a predator-prey competition model but could be adapted for insect attack predator competition based on interference.	Hsu (1981)
Model of mortality in a self-thinning plant population	A mortality function is based on size distribution. If the size distribution changes due to pollution, then the mortality function changes, resulting in a stand change.	Hara (1985)
Partitioning of photosynthate during vegetative growth	Depends upon substrate concentration in plant parts and simple phenomenological relations. If pollution affects photosynthate supply, partitioning and growth change.	Thornley (1969)
Transpiration rate based upon stomatal function	Model predicts transpiration rate based on leaf conductivity. Could be adapted by knowing pollutant effect on conductivity.	Penning-deVries (1972)
Effects of acid precipitation on soil nitrogen and productivity	Simulation model of the effect of acid rain upon N availability and forest production.	Aber et al. (1982)
Growth (basal area increment)	Empirical model to relate ambient SO_2 and changes in the rate of stem growth	Nosal (1984a,b) Nosal (1986)

5. ACID DEPOSITION IN ALBERTA

Alberta has long recognized the potential for environmental problems from acid-forming emissions (Sandhu and Blower 1986). Sanderson (1984) has reviewed the Alberta situation and reported the following for 1981: Alberta is the largest producer of acid-forming emissions in Western Canada; gas plants produce 44.6% of the total sulphur dioxide emissions (249,368 tonnes/y); Oil Sands plants produce 25.3% (141,328 tonnes/y); power plants 17.8% (99,353 tonnes/y); sour gas production facilities 5.3% (29,930 tonnes/y); flaring gas plants 2.5% (13,797 tonnes/y); and fertilizer plants, refineries, pulp and paper plants, and heavy oil recovery plants produce the remaining 4.5% (25,915 tonnes/y) for a total of 559,691 tonnes/y of SO_2. Sanderson reported that there are also 377,700 tonnes of NOx produced annually in Alberta, making an annual total for 1981 of 937,391 tonnes of acid-forming emissions, an annual figure that is considerably less than that of Quebec (1,457,100 tonnes) or Ontario (2,146,000 tonnes). He also stated that dry deposition greatly exceeds wet. Another interesting fact is that the annual contribution to soil acidity in terms of $CaCO_3$ equivalents for neutralization (kg ha^{-1}) is 2 kg ha^{-1} for atmospheric deposition, and 51 kg ha^{-1} for fertilizer-caused soil acidity. This implies that fertilizers contribute more to potential soil acidity than do industrial sources. It should be noted from the above that Alberta has many point sources of acid-forming emissions and that these are the areas that Tabatabai (1985) has said should receive our attention.

Because of this, Alberta has an active program of research with meetings held at regular intervals (MacDonald and Sandhu 1975b; Sandhu and Nyborg 1977; Sandhu et al. 1982; and Sandhu et al. 1987). The major emphasis has been the emission of sulphur gases.

Alberta also has areas with soils that would be considered sensitive to acid deposition. The characteristics of Alberta soils have been reviewed in detail by others in the ADRP program (Turchenek et al. 1987). They have stated that approximately 21,000 km^2 of Brunisolic soils in Alberta are highly sensitive to acidic deposition (Holowaychuk and Fessenden 1987). The Dystric Brunisols found in the Whitecourt area (Legge et al. 1986) are believed to be undergoing acidification because of proximity to a S-gas plant. The pine lichen woodland north of Fort McMurray is on similar sandy non-calcareous soils. These soils (e.g., Héart Series) have low cation exchange capacity (CEC) and pH's in the range of 5.5 to 6. Thus, they are good candidates for acidification. There are natural gas or oil sands sulphur sources in this region as well. There are Solonetzic soils in Alberta that can also be sensitive to acid deposition. These soils are, in some cases, becoming more acidic. This is apparently due to fertilizer practices (Perl et al. 1982). An important consideration is the relative effect on sensitive soils of S-gas from industrial sources compared with nitrogenous fertilizers. This should receive consideration in the future. Finally, there are naturally acidic soils, due apparently to sulphur in the parent material. These soils contain soluble aluminum and manganese that reduce yields when broken for crops (Hoyt and Nyborg 1972). The native forest communities on these soils offer opportunities for studying the effects of long-term exposure to soluble aluminum and manganese as well as for the selection of resistant clones (Giberson and Mayo, personal communication).

5.1 ACID DEPOSITION RESEARCH IN ALBERTA

The considerable amount of acid deposition research in Alberta is varied and detailed. Table 13 gives examples of the research dealing with acid deposition on forests that has appeared in journals in the open literature. There is considerably more in various reports funded by industry and government. No attempt has been made to summarize these. However, as these industry and government papers appear in journals, they will eventually constitute a large body of information. As can be seen in Table 13, and as noted earlier, the major emphasis has been on the effects of S deposition near point sources such as sour gas plants along the Rocky Mountain foothills and the Alberta Oil Sands near Fort McMurray.

There is evidence of soil acidification with aluminum and manganese becoming soluble (Baker et al. 1977; Nyborg et al. 1977; Addison and Puckett 1980; and Addison 1984). However, Lore (1984) found no evidence of soil pH change downwind from a sour gas plant near Pincher Creek, Alberta. This study was on loam to clay loam soils not likely to be sensitive to acid deposition. Clearly, there has been a reduction in photosynthetic capacity of the major tree species near these gas plants (Legge et al. 1978; Legge 1980; Legge et al. 1981; Legge 1982; Addison et al. 1984; Amundson et al. 1986; and Legge et al. 1986), and there have been effects measurable at the biochemical level such as chlorophyll destruction (Malhotra 1977; Harvey and Legge 1979). It appears that forest growth may also have been reduced (Legge, 1980; Legge et al. 1984; and Legge et al. 1986). These data and many more that have appeared in the various reports cited earlier clearly indicate that Alberta has a problem caused by industrial pollution sources as well as by non-point deposition from farming activities. This clarification of whether these effects are direct or indirect depends on a better knowledge of basic forest processes; further research is required.

5.2 RECOMMENDATIONS FOR RESEARCH

The following are recommendations for research directions in Alberta.

5.2.1 Integrated Studies and Modelling

Integration is necessary and it is recommended that more integrated studies involving soil scientists, soil microbiologists, plant physiologists, ecologists, and atmospheric scientists be undertaken. A modelling approach would be desirable. It should be understood that modelling cannot be just an attachment to existing research programs, but must be the goal from the outset. This requires that funding be sufficient for the modelling effort, including the research necessary to provide model input, and the means with which to validate the model in the real world. Too often all of these steps are not undertaken, the result being a model that has not been fully tested. An integrated, model-oriented study of a forest ecosystem on an acid sensitive soil near a known source of acidic pollution is recommended. This also requires that a good control site, free from pollution, be selected, and that fundamental research into basic ecosystem functioning must be supported. The Oil Sands region of northeastern Alberta or the Whitecourt area of west-central Alberta are two possible locations for such a study. In view of the proposed future development of the Oil Sands, it is recommended that such research begin now, rather than after expansion has begun.

Table 13. Alberta research references on the effects of pollutants on forest ecosystems.

Species or Soil	Research	Location	Reference
Not stated	Sampled precipitation, throughfall, and stemflow and soil solution. Soil acidity and extractable Al increased nearer the source. Precipitation acidified.	Rocky Mountain House and Fort McMurray	Baker et al. (1976) Baker et al. (1977)
Pinus contorta forest	Effect of elemental S dust from natural gas plant stock-pile. Sites <200 m had elevated soil S in the upper 20 cm. Lowered pH and Mg, K, and Mn in LFH horizon. Mosses eliminated and herb cover reduced.	Rocky Mountain House	Addison et al. (1984)
Forest	Dry deposition of S may be more important than wet fallout.		Nyborg et al. (1977)
Lichens	Transplanted lichen-covered black spruce branches showed reduced cover over 3 years up to 8.3 km along a steep SO_2 gradient.	Fort McMurray	Addison (1984)
Hypogymnia	High levels of Al (3000-5000 mg Al kg^{-1}) and (200-300 mg Mn kg^{-1}) in area with high SO_2.	Fort McMurray	Addison & Puckett (1980)
Aspen, Willow, Green alder, Paper birch, Jack pine, White spruce, Black spruce, Labrador tea	Fumigation with 15.2 μmol m^{-3} (0.34 ppm) SO_2 reduced net photosynthesis in all species and produced visible symptoms in 2 to 20 days. Response was less rapid in the conifers. Conifers on tailings sand responded more rapidly than on the native soil.	Plants from Fort McMurray	Addison et al. (1984)
Lodgepole x jack pine hybrids	Foliage ATP levels decreased as SO_2 increased.	Trees from Whitecourt	Harvey and Legge (1979)
Pinus contorta	SO_2 (aq) destroys chlorophyll	Edmonton	Malhotra (1977)

continued...

Table 13 (Concluded).

Species or Soil	Research	Location	Reference
Lodgepole x Jack pine hybrids, White Spruce and Aspen	Photosynthesis rates of the trees were lower than expected and corresponded to high SO_4-S levels in the leaves.	Whitecourt	Legge et al. (1977) Amundson et al. (1986)
Boreal forest	Describes a 4 year case study involving air monitoring, isotopic studies, woody plant water relations, pine tree physiology, plant mineral nutrition, and plant biochemistry	Whitecourt	Legge et al. (1981) Legge (1982) Krouse et al. (1984) Mayo et al. (1986) Amundson et al. (1986)
Lodgepole x Jack pine hybrids	Analysis of tree rings from ecological analogs downwind from a sour gas plant by PIXE demonstrates that rings have a historical record of elemental composition differences, presumably due to S gases.	Whitecourt	Legge et al. (1984)
Forest	Studied forest composition along pollution gradients away from the SUNCOR oil sands plant. No differences were found.	Fort McMurray	Pielou (1982)
Loam to clay loam	Soil pH measured for 9 years at locations downwind from a sour gas plant. There were no changes in pH.	Foothills near Pincher Creek	Lore (1984)

5.2.2 Effects on Soils

Work is being done to locate the potentially sensitive soils within Alberta (Holowaychuk and Fessenden 1987). The effects of acid deposition on soils and research recommendations have been made by other reviewers in the ADRP (Turchenek et al. 1987). However, a few recommendations seem appropriate here:

1. soils research should be integrated with other disciplines such as plant responses and microbial studies;
2. the rates of acidification of sensitive soils near pollution sources needs to be determined unequivocally;
3. the extent of NO_3^- and base cation leaching should be determined;
4. the extent of aluminum, manganese, and iron solubilization should be determined;
5. rates of weathering should be determined; and,
6. an integrated study of nutrient cycling in sensitive soils should be undertaken.

5.2.3 Effects of Acid Deposition on Microbial Activity

Research on the effects of acid deposition on microbial activity should be increased. The effects of acid deposition on nitrogen fixation and nitrification need to be studied under Alberta conditions. As noted earlier, there are conflicting reports about the effects of acid deposition on these activities. The effects of acid deposition on mycorrhizae and decomposition require further study. These have been reviewed elsewhere in the ADRP (Visser et al. 1987).

5.2.4 Effects on Photosynthesis and Growth

Studies on photosynthesis and growth should be carried out to separate the direct effects of pollutants from the indirect effects through soil acidification. Since the soils near Whitecourt (Legge et al. 1978: Legge et al. 1986) are becoming acidic, the long-term effects on photosynthetic capacity and growth should be monitored. In particular, the effects of soluble aluminum should be determined.

5.2.5 Effects of Emissions on Reproductive Biology and Community Structure

The areas of reproductive biology and community structure as affected by acid deposition need increased work. It remains to be seen whether or not the effects on photosynthesis and growth will be reflected in community structure (although changes in the lichen and bryophyte communities have already been demonstrated). Studies of pollutant effects on lodgepole and jack pine seed production should be carried out. These trees have aerial seed banks which may be exposed for years prior to cone opening. The effects of high aluminum concentration in tissue should be studied, particularly with regard to pollen viability, seed size, and seed germination.

5.2.6 Combined Effects of Acidic Deposition and Timber Harvesting

The combined effects of acidic deposition and timber harvesting need attention. Where trees are being harvested and acid deposition occurs, the combined effects which deplete base cations and increase NO_3^- leaching are of obvious concern. Perhaps a site near a sour gas plant or in the oil sands area could be located for such a study.

5.2.7 Direct and Indirect Effects of Atmospheric Pollutants

Sorting out the importance of direct and indirect effects of atmospheric pollutants would seem to be uniquely possible in Alberta. The combination of a sensitive soil (near Whitecourt, for example), a point source of S-emissions (relatively free from other pollutants such as O_3, chloroethenes, and NOx,), and a relatively simple plant community would offer a good opportunity to see the relative importance of each. To date, results of the various studies discussed in this report have not clarified the issue.

5.2.8 Naturally Resistant Species

Selection of trees, grasses, and microorganisms tolerant of low soil pH, high aluminum, and manganese would be possible in the Peace River region. As reported by Hoyt and Nyborg (1972), the Peace River region has naturally acidic mineral soil. Native vegetation, species, and perhaps soil organisms would offer a unique gene pool for resistance to some of the problems created by sulphur emissions and acidity.

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ISBN 0-921625-11-1 (Volume 9)
ISBN 0-921625-02-2 (Set of 11)