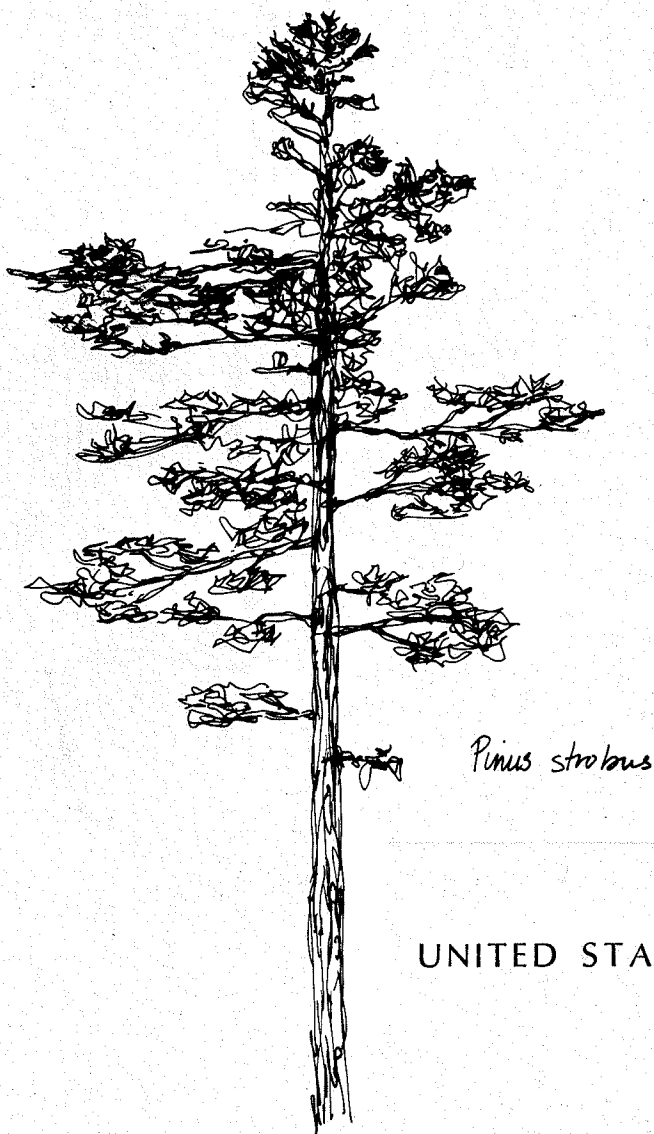


POTENTIAL IMPACTS OF ACID PRECIPITATION AND POOR AIR QUALITY ON THE RESOURCES OF ACADIA NATIONAL PARK

ACADIA NATIONAL PARK
RESOURCE MANAGEMENT DIVISION



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ACADIA NATIONAL PARK
RESOURCE MANAGEMENT DIVISION

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ON THE RESOURCES OF ACADIA NATIONAL PARK

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INTRODUCTION

Air pollution affecting Acadia National Park (ANP) may be composed of a variety of oxides (especially sulfur and nitrogen), halogens, heavy metals, particulate matter, ozone, ethylene, occasionally PAN (peroxyacetylnitrate), and, perhaps, others. At this time, the precise concentration and accumulation of many of these pollutants at ANP are not known. While there is relatively little industry and low levels of naturally occurring toxic substances on Mount Desert Island (MDI), several major sources of SO_2 and other pollutants are within 100 km of ANP. Such sources include paper mills and cement factories. In addition, there is a small oil-fired substation (capable of generating up to 8 mw) on MDI. Long-distance transport may be the major source of other pollutants (Rice, et al., 1981), although concentrations of lead have been roughly correlated with localized gasoline consumption (Lazrus, et al., 1970). Pollutants may be deposited wet or dry, though major problems exist concerning the techniques used in monitoring dry deposition (Droppo, 1976; Interagency Task Force, 1982).

The northeastern portion of the United States has been rated for sensitivity to pollution. Root, et al. (1980) rates Maine as "sensitive" to wet and dry acid deposition based on its soil characteristics, ecological systems such as forests with poorly buffered soils, and high precipitation levels. The alkalinity of surface water on MDI has been ranked as low (<200 eq/l) which is one indication of their sensitivity to acidic deposition (Omernik

and Powers, 1983). Hendrey, et al. (1980) give ratings to specific counties in the U. S., including Hancock County, which are based upon the ability of the geology to buffer or neutralize acidity. Type I rock has low to no buffering capacity; Type II has medium to low buffering capacity. Hancock County is rated as 50 to 59% Type I and 30 to 39% Type II; therefore the area is considered "highly sensitive" to acid deposition. McFee (1980) rates sensitivity based on soil types. Under this system MDI is rated as "slightly sensitive" to acidic deposition. Natural resources associated with surface waters, soils, and bedrock of low buffering capacity are more vulnerable to acidic deposition than those in areas where buffering capacity is higher. This report will discuss the possible effects, both direct and indirect, of acid precipitation and gaseous air pollutants on the flora and fauna of ANP.

The concentration of various air pollutants is important because some of these agents may be harmless or fertilizing at low concentrations. Smith (1974) gives three classes of how the forest ecosystem reacts to air pollution. Class I: undetectable or fertilizing; Class II: may cause a reduction in productivity, a shift in species composition, or an increase in morbidity or a reduction in vigor; and Class III: results in gross simplification of the community, disruption of hydrology or nutrient cycling, severe erosion, changes in microclimate, or overall reduction in stability. Class III situations are easily observed, but Class I and II effects are difficult to detect in the field.

Smith (1974) lists six research difficulties in categorizing Class I and II reactions to air pollution. First, there is variation within a species in response to pollutants and variation between similar species in the same habitat (see Daye and Garside, 1977; Grande, et al., 1976; and Pough and Wilson, 1976 in the Aquatic Organisms Section). Second, other environmental factors influence a species' and an individual's response to a pollutant (see Lister and Lemon, 1976 in the Terrestrial Plants Section). Third, pollutants are usually mixed, (see Evans and Miller, 1975 in the Terrestrial Plants Section) so determining which pollutant is causing which effect can be difficult. Some pollutants are synergistic--causing more than the independently additive effect--and some combinations of pollutants are antagonistic (see Carlson, 1977; and McCune and Weidensaul, 1978 in the Terrestrial Plant Section). Fourth, when the effects of pollutants are studied in experiments, doses are usually in excess of ambient air or soil levels, or they are at more acidic levels than found in surface waters. Fifth, most experimental research is done in controlled environments that are necessarily less complex than field conditions. Sixth, most species have not been evaluated for reactions to air pollutants. This is especially true for the understory plants and terrestrial wildlife of ANP. The most adequate information is available for fish and tree species found within the park.

This report is composed of three sections that discuss the effects of air pollutants on Terrestrial Wildlife, Terrestrial Plants, and Aquatic Organisms. Literature relevant to the flora and/or fauna of ANP is listed at the end of each section. The

final section, Research Needs, presents ideas and possible directions for future research on the consequences of air pollution and acid deposition on the flora and fauna within Acadia National Park.

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TERRESTRIAL WILDLIFE

Little information is available on the effects of air pollutants on terrestrial wildlife. More information is available on the effects of pollutants on domestic or laboratory animals (Amdur, 1978; Glass, 1978). In Newman's (1979) review of the literature available on wildlife, he reports that approximately one-fourth of the more than 100 episodes involving animals included vertebrate wildlife. Most wildlife studies have been done on deer and small birds because these are the managed species, or they are those of public interest. Of course, these are not the only species affected by pollution.

Air pollutants can cause direct or indirect effects on wildlife. Direct effects can be acute or chronic (Newman, 1980). In all cases, the variables depend upon the pollutant(s), ambient concentration, pathway of exposure, duration of exposure, and the animal's age, sex, reproductive condition, nutritional status, and overall health. Wildlife most commonly contract the pollutants by inhalation, or by absorption or ingestion. Dvorak, et al. (1978) states that noise can also adversely affect wildlife.

Newman (1980) reports that the respiratory system is most commonly affected, followed in order of importance by the central nervous, pulmonary, and gastrointestinal systems. Newman also reports that abnormal behavior is often the first sign of a problem. Other responses to air pollutants may be changes in any of the following: population numbers, spatial distribution, appearance, birth rates, or death rates (Newman, 1975).

Wildlife have been used as indicators for accumulation of pollutants. Several of these species occur in ANP, including deer (Odocoileus sp.), hare (Lepus sp.), beaver (Castor canadensis), raccoon (Procyon lotor), otter (Lutra canadensis), muskrat (Ondatra zibethica), woodchuck (Marmota monax), deer mouse (Peromyscus sp.), vole (Microtus sp.), and red fox (Vulpes fulva) (Gordon, 1969; Karstad, 1967; 1970; Kay, 1975; Newman and Yu, 1976; Wren, et al., 1980). Newman (1980) reports that wildlife can absorb some pollutants directly or ingest the pollutants during preening or from eating contaminated plants or animals such as with deer consuming contaminated lichens (Nieboer and Richardson, 1981). Contaminated food is the probable source of lead, cadmium, and zinc found in small roadside animals (Quarles, et al., 1974). Shrews, with a higher metabolic rate, consume more food (and lead) than mice. Voles (Clark, 1979) consume a higher percentage of insects than do other small rodents (Goldsmith and Scanlon, 1977). Concentrations of lead in small mammals and song birds have been reported to be well below toxic levels by Getz, et al. (1977a; 1977b), but the levels found by Clark (1979) were high enough to cause physiological problems (especially renal) in small mammals. Lead levels can be related to traffic volume, nearness to roadway, and age of the animal (Goldsmith and Scanlon, 1977; Quarles, et al., 1974; Schlesinger and Potter, 1974; Welch and Dick, 1975; Williamson and Evans, 1972). Dvorak, et al. (1978) and Peakall (1979) state that most wildlife were affected from shifts in their habitats rather than from being affected directly.

Fish-eating birds such as loons (Gavia immer), mergansers (Mergus merganser), and kingfishers (Megaceryle alcyon) will disappear if fish disappear from lakes, ponds, and streams (Linthurst, et al., 1983). Goldeneye ducks (Bucephala clangula), however, prefer acidic lakes with large insect populations (Eriksson, 1979). Major shifts in the vegetation patterns of over- or understory plants may destroy nesting or breeding habitats as well as protective cover, food supply, and so on. Nilsson and Nilsson (1978) recorded a decrease in breeding and species richness in acidic lakes. Though this work was conducted in Sweden, many of their research species occur in ANP, including red-breasted merganser (Mergus serrator), common merganser (M. merganser), osprey (Pandion haliaetus), black-backed gull (Larus marinus), herring gull (Larus argentatus), Canada goose (Branta canadensis), mallard duck (Anas platyrhynchos), green-winged teal (Anas crecca), common tern (Sterna hirundo), and the rarely seen European widgeon (Anas penelope) and black-headed gull (Larus ridibundus).

Soil animals, such as burrowing and litter-inhabiting forms, are sensitive to soil contamination. Decomposition of the litter may be slowed or altered, which can alter the habitat and food supplies of wildlife (Newman, 1980). Some species of earthworms have been shown to be sensitive to soil pH and, to a lesser extent, calcium supply (Hagvar, 1980; Stachell, 1955). Earthworms have been shown to redistribute trace metals in the soil and accumulate cadmium and zinc, but not lead (Van Hook, 1974). Earthworms, however, could be used to reflect the lead levels in soil (Goldsmith and Scanlon, 1977) and both lead and zinc may

accumulate to levels high enough to be lethal to earthworm eating animals (Gish and Christensen, 1973). Other soil invertebrates, including spiders, mites, beetles, and springtails, are affected by pH (Hagvar, 1980; Hagvar, et al., 1980). Various species within each group increased with acidification and reduced with liming, increased with liming and decreased with acidification, or showed no effect with changing soil pH.

Literature reviews of the effects of air pollutants on insects indicate that most research has been restricted to honeybees and pests of forests (Alstad, et al., 1982; Bromenshenk, 1976). Bee populations are adversely affected by pollutants, but many pests of trees such as aphids, scale, bark beetles, and foliar insects may increase in polluted areas. Such increases may be due to stress placed on the host species by the pollutants or a reduction in predators and parasites (especially parasitic Hymenoptera) (Alstad, et al., 1982; Bromenshenk, 1976). For example, predatory ground beetles were reduced near a paper mill in Canada (Hastings and Freitag, 1973). Some pollutants (including dust) may provide better webbing sites for mites. Dust apparently disrupts the epicuticular wax on an insect's exoskeleton and rapidly causes dessication (Alstad, et al., 1982). This could affect predators as well as pests. Roadside insects accumulate lead and higher concentrations are found in the higher trophic levels (such as predators) (Price, et al., 1974). A pollutant (or group of pollutants) may alter the sensitivity of an insect to pesticides (Bromenshenk, 1976). Alstad, et al., (1982) warn that pollutant-insect interactions

must be examined on a case-by-case basis as there are a variety of possible responses with different hosts and pests. There is virtually nothing published that studies hosts and pests that are both found in ANP, but increases in spruce budworm (Choristoneura orae) injury has been associated with SO₂ and other pollutants near an aluminum smelter (Alstad, et al., 1982).

A series of reports beginning in 1976 (Lewis and Lefohn, 1976) discuss the impact of a coal-fired power plant on a Montana habitat. The initial report lists many of the same species of wildlife (and plants) found in ANP, but later reports have not included species found in the park.

The effects of air pollutants on wildlife in ANP need to be researched. Wildlife in less than optimal habitats may be more sensitive to pollutants (Newman, 1980). It is important to have these populations identified and monitored. By the time wildlife begin to show the effects of pollutants, habitats may be so altered that they cannot be corrected. Early detection will be important in protecting the natural resources of ANP.

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TERRESTRIAL PLANTS

The literature reports that a variety of anthropogenic pollutants have a potential impact (either harmful or beneficial) on some of the terrestrial plants found in ANP. A recent survey provides an accurate listing of the vascular plants found in ANP (Feely and Drury, 1980). Literature citations can be found for most of the tree species with relation to at least one pollutant, but far less information is available for the understory and epiphytic plants of ANP. Non-vascular plants, such as lichens, liverworts, mosses, and fern gametophytes, are especially sensitive to air pollution (Evans and Bozzone, 1978; Hansbrough, 1967; Wielgolaski, 1975). Lichens have often been used as monitors of air pollution (Nieboer and Richardson, 1981; Wielgolaski, 1975). Effects of air pollutants on understory plants indicate that shifts in the population may occur, especially near pollution sources where infrequently found species such as red-berried elder (Sambucus pubens) and fringed bindweed (Polygonum cilinode) may come to dominate (Treshow, 1968). Shifts also occur in the fungal flora when fungi are subjected to air pollution; foliage pathogens such as the tar spot fungus (Rhytisma acerinum) decline as pollution increases (Greenhalgh and Bevan, 1978); declines in white pine blister rust (Cronartium ribicola) and wood rotting fungi have been observed near SO₂ sources (Heagle, 1973); and weak parasites like the shoe string fungus (Armillaria mellea) increase if host plants are weakened (Smith, 1981; Treshow, 1968; 1980).

Generally, evergreens are more sensitive to air pollution than are deciduous species, though variation occurs both between species and within a species. Eastern white pine (Pinus strobus) is especially variable (Hansbrough, 1967). Smaller leaves (needles) are generally more efficient collectors than larger leaves. Since needles remain year round, there is more potential for evergreens to accumulate pollutants (Smith, 1981). Though a particular species may be stressed by a pollutant, other species may be more stressed, and the first species will become dominant, leading to a shift in the population structure (West, et al., 1980).

Air pollutants, especially in areas close to the emission source, may be toxic enough to kill individual plants, may cause a chronic problem that leads to decline and eventual death from a secondary pathogen, may cause sublethal deterioration, or may have no impact at all (Saunders, 1971; Smith, 1980). Minor physiological or morphological effects, such as a lowered photosynthetic rate or reduced foliage (needle) size, may affect individual plants (McLaughlin, et al., 1982). Reduced fruit or cone production may lead to a change in species composition resulting in changes at the population level (Houston and Dochinger, 1977). Changes at the community level, such as reduced growth, and changes at the ecosystem level, such as decreased productivity, biomass, or diversity, may all result from air pollutants (Smith, 1980).

Physiological changes induced by air pollutants often occur before symptoms become visible (Kozlowski, 1980). This is especially true for long-term, chronic injury. When gaseous air

pollutants are absorbed through stomates, the amount of chlorophyll and protein in leaves decreases and enzyme activity increases (Rabe and Kreeb, 1979). This leads to a reduced rate of photosynthesis, an increased rate of respiration, and altered patterns of translocation (McLaughlin, et al., 1982). For white pine trees, needle size and longevity are decreased following absorption of air pollutants and eventually growth rate, as measured by the size of annual growth rings, declines (McLaughlin, et al., 1982). Acute injury includes stomatal blockage, loss of integrity of the cell membrane, and collapse of the cell structure (Kozlowski, 1980; Percy and Riding, 1978). The resulting chlorosis (yellowing) or necrosis (death) may be limited, only visible upon close inspection, or more widespread. In either of these cases, reduced quality and reduced growth of forest tree species would be expected (Kozlowski, 1980). The actual appearance of a plant species varies with the pollutant and each major pollutant will be discussed below.

Many references that concern species found in ANP concern laboratory or simulated field conditions rather than observations of natural field conditions where mixed pollutants usually exist. This is especially true for the literature concerning acidic deposition, which has only recently received attention in the northeastern United States. No research has been found that documents the effects of air pollutants within ANP; the literature reviewed here concerns research which was conducted elsewhere, on species found in ANP.

Ozone

There are few references on the effects of ozone (O_3) on lichens (Ferry, et al., 1973; Nash and Sigal, 1979). Nash and Sigal (1979) found no visible symptoms of injury even after 12 hours of O_3 fumigation, but the photosynthetic rate decreased after 1.5 to 3 hours of fumigation. Similar lag times are not observed with higher plants, but with lichens it may take time for the O_3 to penetrate to algal cells. Algal cells turn yellow-green to amber after fumigation (Nash and Sigal, 1979). Mosses are also reported as being sensitive to O_3 (Nash, 1973a), but more information is available on vascular plants.

Symptoms of ozone on broad-leaf species are red-brown to bleached flecking or stippling on the upper surface of the leaf. Affected areas are small and irregular, but may coalesce. Recently matured leaves are more sensitive than foliage at other stages; and because the palisade parenchyma cells (upper layer of photosynthetic cells) are the most sensitive tissue in a leaf, injury may be confined to the upper leaf surface (Applied Science Associated, 1978; Daines, et al., 1967; Nash, 1973a). Stippling and flecking on conifer needles are first silvery and turn purple. Areas may coalesce to form a band that may lead to death of the distal portion of the needle (tip necrosis). Whole needles can be affected and needles can appear bleached. Semi-mature needle tissue is the most sensitive to O_3 and the mesophyll (photosynthetic) cells adjacent to stomates are injured first (Applied Science Associated, 1978; Costonis, 1970; Daines, et al., 1967; Linzon, 1966; 1967; Nash, 1973a; Smith, 1981). Stomates must be open for ozone injury, and injury is more severe

when humidity, soil moisture, light intensity, and nitrogen levels are high and when the temperature is near 30 C (Daines, et al., 1967; Heck, 1968; Heck, et al., 1965; Nash, 1973a). High carbon dioxide (CO_2) levels, drought, or high temperatures can reduce sensitivity (Heck, 1968). Ozone-tolerant species either close their stomates when air pollutants are present or their pattern of stomatal opening and closing avoids exposing internal tissues to pollutants (Harkov and Brennan, 1979). Ozone injured trees have lower photosynthetic rates when compared with healthy trees. For maple, oak, and ash trees, photosynthetic rates are reported even lower from O_3 injury than reduction caused by sulfur dioxide (SO_2) (Carlson, 1979). With O_3 injury, chloroplasts aggregate and there is a breakdown of their membrane system (Kozlowski, 1980). Additional changes in the cellular chemistry are reported that lead to cell collapse (Horsman and Wellburn, 1976; Kozlowski, 1980).

Native understory species reported as being highly resistant to O_3 include columbine (Aquilegia sp.), lamb's quarters (Chenopodium album), and ragweed (Ambrosia artemisiifolia). On the other hand, currants (Ribes spp.), black bindweed (Polygonum convolvulus), chickory (Cichorium intybus), ox-eye daisy (Chrysanthemum leucanthemum), milkweed (Asclepias syriaca), and crunchweed (Brassica kaber) are reported as being sensitive to O_3 (Kohut and Krupa, 1977; Skelly, 1980).

Table I and II list the species of woody plants sensitive and tolerant to O_3 . Slower growing tree species tend to be less susceptible to O_3 injury than those trees that grow more rapidly

(Harkov and Brennan, 1979). Jack pine (Pinus banksiana) is the most sensitive pine to O₃ and red pine (Pinus resinosa) is the most tolerant pine to O₃ (Berry, 1971; Skelly, 1980).

Table I. Woody Species Susceptible to Ozone

Ash, white	<u>Fraxinus americana</u>
Hemlock, eastern	<u>Tsuga canadensis</u>
Pine, eastern white	<u>Pinus strobus</u>
Pine, jack	<u>Pinus banksiana</u>
Pine, pitch	<u>Pinus rigida</u>

Sources: Davis and Wilhour, 1976; Davis and Wood, 1972.

Table II. Wood Species Tolerant to Ozone

Cedar, northern white	<u>Thuja occidentalis</u>
Fir, balsam	<u>Abies balsamea</u>
Laurel, mountain	<u>Kalmia latifolia</u>
Maple, sugar	<u>Acer saccharum</u>
Oak, northern red	<u>Quercus rubra</u>
Pine, red	<u>Pinus resinosa</u>
Spruce, white	<u>Picea glauca</u>

Sources: Davis and Wilhour, 1976; Davis and Wood, 1972

Eastern white pine, however, has been studied more than any other forest tree species for its response to air pollution in general and to ozone in particular (Skelly, 1980). Needles are sensitive from the time of emergence until they are fully elongated and hardened-off (usually in June and July of their first year). The most sensitive tissue is near the needle sheath. Tip necrosis follows death of tissue closer to the base of the needle (Costonis and Sinclair, 1969; Gerhold, 1970). Variation in sensitivity between white pines to oxidant pollution has been reported in the literature (Berry, 1973; Berry and Ripperton, 1963; Dochinger, et al., 1965; McLaughlin, et al., 1982; Skelly,

1980). Though environmental conditions and pollutant concentration influence the degree of injury, genetic variation also explains some of the differences in susceptibility (Gerhold, 1977). Elevated ozone levels and oxidant damage on some white pine were reported for ANP during the summer of 1983 (C. Schell, personal communication). If elevated ozone levels become commonplace in ANP, some tree mortality and a loss of genetic variability will result.

Both O_3 and SO_2 are involved with the diseases called chlorotic dwarf and semi-mature needle blight of eastern white pine (Dochinger and Heck, 1969; Dochinger, et al., 1970; Linzon, 1966; 1967; Mann, et al., 1980). Though some symptoms are the same, chlorotic dwarf disease and semi-mature needle blight are not the same disease (Gerhold, 1977). There is widespread variability within eastern white pine in response to O_3 and SO_2 tants. Symptoms on susceptible trees include needle flexing, tip necrosis, stunted needles, chlorophyll breakdown, premature needle cast, and reduced shoot elongation (Blanchard, et al., 1979; Dochinger, et al., 1970; Hayes and Skelly, 1977; Linzon, 1966; Mann, et al., 1980; McLaughlin, et al., 1982). In susceptible trees, reduced biomass of needles leads to reduced carbohydrate supplies, which may affect the root system, thus reducing nutrient uptake and growth (Dochinger, et al., 1970; Mann, et al., 1980; Linzon, 1967a; McLaughlin, et al., 1982). With reduced pollutants, trees can recover (Berry and Ripperton, 1963; Dochinger, et al., 1965; McLaughlin, et al., 1982), but sensitive trees become more susceptible to other pressures and pathogens

(McLaughlin, et al., 1982). Though O_3 usually affects hosts more than pathogens (especially decay and root pathogens), Lophodermium needle cast is reduced by O_3 (Treshow, 1980).

On the East Coast, O_3 is usually episodal in occurrence, but it is recognized as one of the most important air pollutants affecting white pine (Skelly, 1980). Peroxyacetylnitrates (PAN), another common photochemical oxidant, is rarely reported for the eastern United States (Skelly, 1980). However O_3 and SO_2 often occur together, and the interaction of these two gases is more damaging than the additive effects of the two gases separately (Dochinger, et al., 1970).

Sulfur

Several sulfur compounds are known to be toxic to plants. Of these, sulfur dioxide (SO_2) is the most important and the first gas widely recognized as an air pollutant. Sulfur dioxide is produced by the burning of fossil fuels (coal, oil, and so on), the burning of wood, manufacturing processes using sulfur and sulfuric acid, and smelting of ores (Oglesby and Blosser, 1980; Smith, 1970). Sulfur may also come from decomposition processes and weathering of rock (Eaton, et al., 1978; Koterba, et al., 1979; Rice, et al., 1981).

Acute SO_2 symptoms occur when SO_2 levels exceed 0.50 ppm, and chronic injury can occur at levels as low as 0.03 ppm for susceptible species (Smith, 1970). Conifers are most sensitive to SO_2 in the spring and early summer when new needles are elongating (Daines, et al., 1967). Affected needles have bands of necrotic tissue with needle tips often turning a reddish-brown.

There may be premature needle abscission. For broadleaf trees, chronic injury may lead to chlorosis or to necrosis of marginal and interveinal leaf tissue. Discolored leaf tissue will be ivory or reddish-brown in color (Applied Science Associates, 1978; Linzon, 1978; Nash, 1973a; Smith, 1970). The young, fully expanded leaves of broadleaf species are most sensitive to SO_2 injury (Nash, 1973a). Though foliar injury is the most commonly observed response, reduced vigor and growth rates have been associated with SO_2 injury (McCune and Weidensaul, 1978; Linzon, 1978). Also, a plant's age, nutritional status, and water status will affect sensitivity to SO_2 (McCune and Weidensaul, 1978; Unsworth, et al., 1976).

More injury from SO_2 results when stomates are open than when they are closed, and environmental conditions that encourage open stomates (adequate light, warm temperatures, and high humidity) lead to greater SO_2 injury (Daines, et al., 1967; Garland and Branson, 1977; Kozlowski, 1980; Legge, 1976; Severson, 1975; Unsworth, et al., 1976). Uptake is much faster and injury is greater when moisture is present, which is at least partly due to SO_2 being quite water soluble (Daines, et al., 1967; Garland and Branson, 1977). In addition, dry deposition of SO_2 is important. The rate of deposition depends upon several physical characteristics including surface resistance and area of the leaf (Shreffler, 1978). Sheltered areas receive less SO_2 deposition than exposed areas (Saunders, 1971).

At low concentrations, SO_2 may be fertilizing to plants because it is a necessary element in proteins (Guderian and Kueppers, 1980; Severson, 1975). But at higher concentrations,

SO₂ results in decreased photosynthetic rates due in part to inhibition of an enzyme (ribulose diphosphate carboxylase) that is necessary for CO₂ fixation (Legge, 1976; Severson, 1975). Plants from polluted environments have higher activity of this enzyme than plants from "clean" airsheds (Wellburn, et al., 1976). Sulfur dioxide also alters the structure of the chloroplasts (Wellburn, et al., 1972) and can finally lead to the collapse of photosynthetic cells. Death of epidermal cells follows, and finally death of vascular tissues (Costonis, 1970; Kozlowski, 1980). Physiological changes were observed for maples, birches, spruce, and fir following SO₂ fumigation, but different changes were observed in the different species. For example, pentose phosphate (a sugar) increased in maples and decreased in birches (Horseman and Wellburn, 1976). An increase in a sulfur-containing compound glutathione, is observed when plants are fumigated with SO₂, and this may cause a disruption in fundamental cellular processes of the plant (Grill, et al., 1979).

Many reports on the effects of SO₂ on forest species concern observation near smelters, coal-fired power plants, or other large point sources of SO₂. The effects observed near a point source will differ from those observed farther from the source. Such observations, however, can provide information on the relative sensitivity of plant species found in ANP as well as on potential effects if a large point source is ever proposed for construction near the park. Generally, species diversity decreases near point sources and, depending upon the concentra-

tion of the pollutants, all plant species may be killed for several miles downwind of the source (Freedman and Hutchinson, 1980; Gordon and Gorham, 1963; Linzon, 1975; Winner and Bewley, 1978). A species may be given a competitive advantage in a polluted area if it is less harmed by the pollutant than other species in the area (Guderian and Kueppers, 1980). Species that are tolerant of SO_2 , such as fringed bindweed and red-berried elder, increase in number even though they are infrequent in the unaffected forest (Gordon and Graham, 1963; Guderian and Kueppers, 1980; Linzon, 1975). Red maple (Acer rubrum), red oak (Quercus rubra), and sugar maple (Acer saccharum) are tolerant tree species (Jensen and Kozlowski, 1975; Linzon, 1975), while white pine is the most sensitive tree species (Linzon, 1975). Tables III and IV list the relative susceptibility for some herbaceous and woody plant species of ANP.

Most non-vascular plants are more sensitive to SO_2 than are vascular plants because they absorb SO_2 over their entire surface (McCune and Weidensaul, 1978; Linzon, 1978). Sulfur is a well-known fungicide and may reduce the white pine blister rust fungus (Cronartium ribicola), the tar spot fungus (Rhytisma acerinum), and other forest pathogens, but soil-born fungi that attack weakened trees may have an advantage when trees are stressed by SO_2 (Bevan and Greenhalgh, 1976; Heagle, 1973; McCune and Weidensaul, 1978; Saunders, 1970; Treshow, 1980). In one study, bracken ferns (Pteridium aquilinum), most lichens, and bryophytes (liverworts and mosses) were absent near a smelter (Freedman and Hutchinson, 1980). Bryophytes are especially sensitive and may be absent when SO_2 exceeds 0.01 ppm (Gilbert, 1968; McCune and

Table III.

Relative Susceptibility of Herbaceous Plants to SO₂

A. Sensitive

Aster, large leafed	<u>Aster macrophyllus</u>
Fern, bracken	<u>Pteridium aquilinum</u>
Pondweed	<u>Potamogeton epihydrus</u> v. <u>nuttallii</u>
Sarsaparilla, wild	<u>Aralia nudicaulis</u>

B. Intermediate

Clover, red	<u>Trifolium pratense</u>
Mayflower, Canada	<u>Maianthemum canadense</u>

C. Tolerant

Bindweed, fringed	<u>Polygonum cilinode</u>
Lily of the Valley	<u>Convallaria majalis</u>
Madder, wild	<u>Galium mollugo</u>
Speedwell, common	<u>Veronica officinalis</u>

Sources: Gordon and Gorham, 1963; Guderian and Kueppers, 1980; Linzon, 1975; McCune and Weidensaul, 1978; Scale, 1980.

Table IV.

Relative Susceptibility of Woody Plants to SO₂

A. Sensitive

Apple	<u>Malus sp.</u>
Ash, white	<u>Fraxinus americana</u> ¹
Aspen, large-toothed	<u>Populus grandidentata</u> ²
Aspen, trembling	<u>Populus tremuloides</u> ²
Birch, grey	<u>Betula populifolia</u>
Birch, white	<u>Betula papyrifera</u>
Birch, yellow	<u>Betula alleghaniensis</u> ¹
Blueberry, low-bush	<u>Vaccinium angustifolium</u> ¹
Blueberry, sour-top	<u>Vaccinium myrtilloides</u>
Hazelnut, beaked	<u>Corylus cornuta</u>
Honeysuckle, bush	<u>Diervilla lonicera</u>
Pine, eastern white	<u>Pinus strobus</u>
Pine, jack	<u>Pinus banksiana</u>
Pine, red	<u>Pinus resinosa</u> ¹
Serviceberry, running	<u>Amelanchier stolonifera</u>
Sumac, staghorn	<u>Rhus typhina</u>
Willow, black	<u>Salix nigra</u>

¹Also rated as intermediate because researchers reported different levels of sensitivity.

²Also rated as tolerant because researchers reported different levels of sensitivity.

B. Intermediate

Ash, white	<u>Fraxinus americana</u>
Basswood	<u>Tilia americana</u>
Birch, yellow	<u>Betula alleghaniensis</u> ³
Blueberry, low-bush	<u>Vaccinium angustifolium</u> ³
Cherry, choke	<u>Prunus virginiana</u>
Current, red	<u>Ribes sp.</u>
Elm, American	<u>Ulmus americana</u>
Fir, balsam	<u>Abies balsamea</u>
Maple, mountain	<u>Acer spicatum</u>
Maple, red	<u>Acer rubrum</u> ⁴
Pine, red	<u>Pinus resinosa</u> ³

³Also rated sensitive because reseachers reported different levels of sensitivity.

⁴Also rated tolerant because researchers reported different levels of sensitivity.

Table IV. (Continued)

C. Tolerant

Ash, green	<u>Fraxinus pennsylvanica</u>
Aspen, large-toothed	<u>Populus grandidentata</u> ⁵
Aspen, trembling	<u>Populus tremuloides</u> ⁵
Bearberry	<u>Arctostaphylos uva-ursi</u>
Blueberry, highbush	<u>Vaccinium corymbosum</u>
Cedar, northern white	<u>Thuja occidentalis</u>
Elder, red-berried	<u>Sambucus pubens</u>
Hazel, witch	<u>Hamamelis virginiana</u>
Juniper, common	<u>Juniperus communis</u>
Maple, red	<u>Acer rubrum</u> ⁶
Maple, sugar	<u>Acer saccharum</u>
Poplar, balsam	<u>Populus balsamifera</u>
Oak, northern red	<u>Quercus rubra</u>
Spruce, black	<u>Picea mariana</u>
Spruce, white	<u>Picea glauca</u>

⁵Also rated as sensitive because researchers reported different levels of sensitivity.

⁶Also rated as intermediate because researchers reported different levels of sensitivity.

Sources: Berry, 1971; Brennan, et al., 1970; Davis and Wilhour, 1976; Gordon and Gorham, 1963; Hansbrough, 1967; Linzon, 1975; 1978; Jensen and Kozlowski, 1975; McCune and Weidensaul, 1978; Scale, 1980.

Weidensaul, 1978; Winner and Bewley, 1978). Sphagnum moss species vary in susceptibility to sulfur pollutants (Ferguson, et al., 1978; Ferguson and Lee, 1978; 1980). Injury due to bisulphite (HSO_3^-) and SO_2 is pH-dependent, and lower photosynthetic rates occur at low pH levels. But, low concentrations of HSO_3^- stimulate photosynthesis. At high concentrations, HSO_3^- , SO_2 , and $\text{SO}_4^{=}$ (sulfates) all reduce CO_2 fixation rates (Ferguson and Lee, 1978; Ferguson, et al., 1978).

The effects of SO_2 on lichens are well-known. Their sensitivity is at least partially explained by the fact that lichens absorb over their entire surface, accumulate up to 1000 times the concentration of SO_2 as in the atmosphere, and do not shed toxic-laden foliage (Hansbrough, 1967; Linzon, 1978; Nieboer and Richardson, 1981; Puckett, et al., 1973). Injury is most severe when lichens are moist (Nieboer and Richardson, 1981), but injury varies with habitat and species (Gilbert, 1970; Pearson, 1973; Tuerk and Wirth, 1975). Variation in amount of shelter and nutrients, pH levels, and buffering capacity of the substratum can modify the effects of SO_2 (Gilbert, 1970). Generally fruticose (shrubby or hair-like) lichens are more sensitive than foliose (leaf-like) which are more sensitive than crustose (crust-like) (Gilbert, 1970; Hill, 1971). There exceptions, and Andrews (1975) believes there are too many exceptions to consider this relationship valid. Highly resistant lichen species are usually rapid-growing species (Gilbert, 1970). Spores and colonies at the establishment stage are more vulnerable to pollutants than are established colonies (Gilbert, 1973; Saunders, 1970). Some lichens can survive in polluted areas, but they are

unable to reproduce (Saunders, 1971). Usnea and Lobaria are two genera reported to be very sensitive to SO₂, and Cladonia furcata and Physcia millegrana are reported to be fairly tolerant of SO₂ (Nash, 1973b; Nieboer, et al., 1976; Olkkonen and Takala, 1975).

Symptoms of acute SO₂ injury on lichens include a whitening of the thallus, then necrosis, and finally detachment from the substrate. Symptoms of chronic injury include a curling of the margin of the thallus and a discoloration of the tissue. The change in color varies with the species (Saunders, 1970). Waxy deposits may be secreted onto the surface of the lichen (Nieboer, et al., 1976). Internal symptoms include plasmolysis of the algal cells (cell membrane shrinks away from the cell wall), production of more vegetative propagules (soredia), lost chlorophyll, and inhibited photosynthesis (Andrews, 1975; Deruelle, 1978; Ikonen and Kaerenlampi, 1976; Nieboer, et al., 1976; Nieboer and Richardson, 1981; Pearson, 1973; Puckett, et al., 1973; Rao and LeBlanc, 1966; Saunders, 1970; Tuerk and Wirth, 1975). Some researchers report that respiration is decreased (Andrews, 1975; Deruelle, 1978; Nieboer and Richardson, 1981), increased (Pearson, 1973), or there is no change following exposure to SO₂ (Ikonen and Kaerenlampi, 1976). The algal cells (phycobiont) are affected before the fungal cells (mycobiont) (Eversman, 1978; Saunders, 1970). The stronger the dependency of the fungal cells on the algal cells, the more sensitive the lichen will be to air pollution (Beltman, et al., 1980).

Sulfur dioxide is most toxic at low pH levels (Sundstrom and

Hallgren, 1973; Tuerk and Wirth, 1975), and it is more toxic when the lichen is moist or SO₂ is dissolved in water than when SO₂ is dry-deposited (Puckett, et al., 1973; Rao and LeBlanc, 1966; Richardson and Puckett, 1973). Bisulfites and sulfates are also toxic to lichens (Rao and LeBlanc, 1966; Sundstrom and Hallgren, 1973).

Some metals can protect lichens from SO₂ injury. Those metals that prefer ionic interactions provide the most protection, such as Sr (strontium), Ni (nickel), and Zi (zinc), but metals that prefer covalent interactions, such as Ca (calcium) and Mg (magnesium), provide no protection. The metals protect the lichen by changing the membrane permeability of the cells (Nieboer, 1979a; 1979b; Richardson, et al., 1979).

In the field, lichen-free zones have been observed around SO₂ point sources (Deruelle, 1978; Nieboer, et al., 1976; Saunders, 1970). Outside the "lichen desert" there is a zone of transition with a reduced number of species, colonies spread farther apart, and a decline in fertility before normal lichen populations resume (Andrews, 1975; Deruelle, 1978). The lichens occupying these zones vary from region to region, and it is difficult to predict what will occur in any area that has not been studied (Andrews, 1975; Hoffman, 1974). Field studies have transplanted lichens into polluted environments to measure their reaction (Hoffman, 1974) or changes in lichens have been photographed over time (Lulman, et al., 1980). Similar studies would be needed in ANP to accurately measure the impact of a major SO₂ source on the lichens and other flora.

Nitrogen Oxides

Nitrogen is an essential element for plant growth, but nitric oxide (NO) and nitrogen dioxide (NO₂) can cause injury to plants (Daines, et al., 1967; Smith, 1970). Lesions on foliage resemble SO₂ injury (Brandt, 1967); generally NO₂ and SO₂ injury result in additive effects on foliage (McCune and Weidensaul, 1978). Though sources of nitrogen oxides (NO_x) include automobile exhaust and combustion of fossil fuels, the major threat to the flora of ANP may come from reaction of these oxides with atmospheric moisture, which contributes to decreased pH values in precipitation.

Acidic Deposition

While air pollutants such as SO₂ and O₃ have long been recognized to cause injury to plant tissue, concerns over possible phytotoxicity from acidic deposition have recently come to the forefront of air pollution research (Linthurst, et al., 1983). This deposition includes all forms of wet precipitation as well as dry-deposited materials. Deposition may contain a wide range of pollutants including heavy metals, but acidification is primarily attributed to the reaction of SO₂ and NO_x in the atmosphere (Smith, 1980).

In contrast to the volume of literature concerning the effects of SO₂ on lichen vegetation, there is little information on the effects of acidic deposition on lichens. Lechowicz (1982) reports that the photosynthetic capacity of caribou lichen (Cladina stellaris) is reduced following treatment with simulated

acidic precipitation. It is speculated that acidic precipitation may reduce the growth rate of this lichen. Since there is so little information on acidic deposition and lichens, it is unlikely that lichens can be used as a monitor for acidic deposition as they are for SO₂ pollution.

Few reports exist on the effects of low pH on other non-vascular plants. The growth of sphagnum moss is not affected by low pH, but high pH levels (circum-neutral) with high calcium levels can be lethal (Clymo, 1973). The photosynthetic rate of some mosses decreases as pH levels decrease. This is associated with a loss of chlorophyll a, but no effect on respiration has been observed (Sheridan and Rosenstreter, 1973). Bracken fern (Pteridium aquilinum) populations may be reduced by acidic precipitation. The sporophyte (vegetative) leaves are more sensitive than leaves from vascular plants (Evans, 1979; Evans and Curry, 1979). The gametes, especially the spermatozoids, were very sensitive to low pH and to elevated levels of chlorine, nitrate, and sulfate. Fertilization rates were decreased as a result of reduced sperm motility (Evans, 1979a; Evans and Bozzone, 1977; 1978; Evans and Conway, 1980; Evans and Curry, 1979). Though bracken fern is a clonal plant and spreads easily vegetatively, a reduction in fertility rate reduces genetic recombination and diversity within the species (Evans and Bozzone, 1977).

Fungi and bacteria are also affected by low pH. Bacteria usually cannot survive at low pH levels, and a decrease in bacterial-induced plant diseases would be expected (Shriner and Cowling, 1978). While fungi are relatively resistant to low pH,

it is more difficult to predict that a particular plant pathogen would have an advantage over its host due to acidic precipitation. Dissemination, germination, and penetration may each be affected. A thinner cuticle or more wounds to a host would ease penetration of a host by a fungus, but if nutrients on the leaf surface were leached, germination of a fungal spore might be slowed (Shriner and Cowling, 1978). Generally, severity of plant disease is expected to increase if the pathogens arrive on the host after acid precipitation and to decrease if the pathogens arrive before acid precipitation (Shriner, 1978).

Other forest microorganisms may also be affected by pH levels. Nitrifying bacteria are sensitive to acidic conditions (Raynal, et al., 1980). There is some indication that decomposition may be reduced by very low pH levels (Hovland, et al., 1980; Ishac and Hovland, 1976). But the conditions tested in these experiments were extreme compared to the pH of rainfall recorded in field conditions.

Conditions used to test the effects of acidic precipitation on forest vegetation rarely match true field conditions. The pH levels tested are often far below observed pH values of rain water reported for ANP, and simulated rainstorms do not vary in intensity of rainfall or variation in pH levels, as do natural rainstorms (Evans and Hendrey, 1979; Shriner, 1979). Rainstorms of long duration may be more phytotoxic than shorter ones (Jacobson and Van Leuken, 1977).

A cause-and-effect relationship has not been documented by field studies to date (Jacobson, 1978; Jones and Noggle, 1980;

Linthurst, et al., 1983; McLaughlin, 1984). But observations of declining and dying trees have been made in Europe and the northeastern United States in areas receiving acidic deposition and other transported pollutants. McLaughlin (1984) states:

A major deficiency in the investigative process to date has been the failure to demonstrate the primary physiological mechanisms by which acid rain has altered forest growth under field conditions. The problem of identifying primary causative factors in the forest environment is extremely difficult, because of the complex interactions between biotic and abiotic variables which ultimately regulate forest growth. Nevertheless, there is a growing body of evidence which can be used to formulate and test hypotheses regarding the causes, potential magnitude, and future direction of recent forest declines.

Changes in growth rate have been studied in an attempt to determine effects of acidic deposition on forest trees. Beneficial effects may occur from the addition of nitrogen and, in some cases sulfur to the ecosystem (Abrahamsen, 1980a; Jacobson, 1978; Linthurst, et al., 1983; Tamm and Cowling, 1976). Simulated acid rain stimulated the seed germination of eastern white pine, eastern red cedar (Juniperus virginiana), and yellow birch (Betula alleghaniensis) while inhibiting germination of staghorn sumac (Rhus typhina) (Lee and Weber, 1979). At very low pH levels (3.0 and less), however, germination of yellow birch and red maple seed is inhibited, but sugar maple is not inhibited (Raynal, et al., 1982a).

The stages of germination and seedling establishment have been shown to be the most sensitive to low pH levels (Abrahamsen, et al., 1976; Lee and Weber, 1979; Raynal, et al., 1982b). Detrimental effects may be caused by direct injury to the emerging seedling, by indirect effects from changes in soil nutrient

levels, or by increased susceptibility to soil pathogens (Raynal, et al., 1982a).

No significant effect on top growth of young seedlings has been reported for species occurring in ANP (Lee and Weber, 1979). White pine seedlings were exposed to simulated acid rain from pH 4 to 2.3. Though some injury to the needles is reported over the course of a 20-week experiment, the productivity of the plants was increased when they were treated with the simulated acid rain. The increase is assumed to be due to the nitrate in the rain. But necessary ions such as potassium, magnesium, and calcium were nearly depleted at the lowest pH level, and nutrient deficiencies may have resulted if the experiment had continued (Wood and Bormann, 1976).

Work with older trees has predominately centered on tree-ring analysis. There is a suggestion that observed decreases in tree growth may be related to a non-climatic environmental response, but this cannot be absolutely linked to acidic deposition (Abrahamsen, et al., 1976; Cogbill, 1975; Cogbill, 1976; Johnson and Siccama, 1983; Puckett, 1982; Siccama, et al., 1982; Whittaker, et al., 1974). Recent work in the Northeast has concentrated on the decline of red spruce, but declines have also been observed for balsam fir (Abies balsamea), white pine, pitch pine (Pinus rigida), beech (Fagus grandifolia), white birch (Betula papyrifera), yellow birch, and sugar maple. Drought, heavy metals (especially lead), gaseous air pollutants such as SO₂ and O₃, and increases in plant pathogens may all be contributing to the decrease in growth (Johnson and Siccama, 1983; Siccama, et al., 1982; Whittaker, et al., 1974).

Decreased growth is also reported for cranberry (Vaccinium macrocarpon) with pH levels of 3 and 4 (Medappa and Dana, 1970). There are few reports on the effects of acidic deposition on vascular understory plants occurring in ANP.

The most frequent direct responses of plants to simulated acidic precipitation are lesions on foliage near stomates, near areas of pooling (natural depressions), near veins and vein tips, at the base of plant hairs, hydrathodes, and glandular hairs, or at wounds where the leaf cuticle is thin or broken (Evans, 1979a; 1979b; Jacobson, 1978; Shriner, 1979). Developing leaves are more sensitive than mature leaves (Haines, et al., 1980). Necrotic spots occur at pH 2 and lower, on red maple, black locust (Robinia pseudoacacia), and pilewort (Erechtites hieracifolia), but necrotic spots on white pine are not observed above pH 1.0 (Haines, et al., 1980). Generally, the cell cytoplasm resists large pH changes, perhaps because of the proteins in the cell membrane. Photosynthetic capacity decreases at low pH levels, however, due to sulfur binding to the ribulose diphosphate carboxylase enzyme. This prevents the uptake of CO₂ from occurring (Ziegler, 1979). Gall formation has been reported for Populus sp. (Evans, 1979b). But Scholz and Reck (1976) report that white pine is intermediate in sensitivity to low pH (other species in the study are not found in ANP). The surface pH of leaves along a highway is lower compared to leaves away from the road, but pH levels increase if dust is deposited on the leaves (Fluckinger-Keller, et al., 1979). Levels of pH of tree bark also decrease due to acidic precipitation (Staxang, 1969).

The major indirect effects of concern are leaching of nutrients from the soil and the forest canopy and an accumulation of heavy metals in the soil. Most nutrients are lost in the spring or following heavy rain storms (Bormann, et al., 1969) and deficiencies may develop when Ca, Mg, and K are leached from the forest soils and vegetation (Abrahamsen, 1980a; Cronan, et al., 1980; Lee and Weber, 1980; Wood and Bormann, 1975). Under deciduous trees such as beeches and maples, the amount of nutrients is greater in the stemfall and throughfall than from open areas; ions move faster through the forest canopy so nutrient cycling is accelerated; and the pH of precipitation increases as it moves through the canopy (Eaton, et al., 1973; Helvey and Patric, 1965; Lee and Weber, 1980; Olson, et al., 1981; Voight, 1960). Under conifers, similar observations are made regarding throughfall and stemfall except that the pH of the precipitation decreases as it moves through the canopy, and more rain is intercepted by a conifer canopy than by deciduous species (Abrahamsen, 1980b; Cronan, et al., 1980; Nihlgard, 1970; Olson, et al., 1981). More nutrients are found in conifer stemfall than from deciduous stemfall; overall, more materials are leached from deciduous trees than from conifers (Olson, et al., 1981; Voight, 1960). In the balsam fir forests of New Hampshire, more water was collected from throughfall and stemfall than from collection of rainfall only. Cloud droplets and fog account for the difference in volume of moisture because conifers are especially efficient at trapping small particles, such as fog droplets. The clouds scavenge ions from the atmosphere, and the concentration of ions in the stemfall and throughfall is not decreased (Olson, et al., 1981).

The possibility that clouds and fog bring more acidic moisture to the forest must be considered for ANP, where fog and clouds conditions are considerable. It is possible that ions in sea fog may actually increase the pH of moisture received by the trees of ANP. In any case, it would be expected that ANP receives high amounts of ions (such as sodium) associated with marine environments (Grahn, et al., 1974). Possible effects of increased leaching include reduction in productivity, reduction in species diversity, reduction in regeneration rates, and changes in nutrient cycling (Eaton, 1979; Olson, 1975).

The second indirect effect of acidic precipitation comes from the accumulation of aluminum (Al) and a further reduction of already low soil pH levels in forested areas of the Northeast (Frink and Voight, 1976; Hanson, 1980; Hanson, et al., 1982; Likens, et al., 1967; McFee and Cronan, 1981). Nitrogen fixation and nitrogen mineralization are two bacterial processes that are inhibited by reduced soil pH (Tamm, 1976). Acidic precipitation brings more sulfate ions to forest soils (Cronan, 1979; 1980); and in non-calcaerous soils (low Ca content), Al may become mobilized. Aluminum can then be transported to surface waters or accumulate in soils where it may reach levels toxic to plant roots (Bartlett and Riego, 1972; Cronan, 1979; 1980; Cronan and Schofield, 1979; Krug and Frink, 1983; McFee and Cronan, 1981; Norton, et al., 1980; Tamm, 1976). Land use patterns and vegetational succession may cause more changes in soil than acidic precipitation (Krug and Frink, 1983), but increases in some heavy metals such as zinc and

lead are correlated with areas receiving acidic precipitation (Hanson, et al., 1982).

Heavy Metals

The increase in heavy metals in forest vegetation and in soils comes partly from atmospheric transport and partly from mineral weathering (Heinricks and Mayer, 1980). Lead (Pb) accumulates near roads, but it has also been found far from major highways (Elias and Croxdale, 1980; Lagerwerff and Specht, 1970; Siccama and Smith, 1978). In addition, cadmium (Cd), nickel (Ni), and zinc (Zn) may accumulate on vegetation and in soil near roads. Levels of these elements decrease with distance from the road and with soil depth (Lagerwerff and Specht, 1970).

Some of the transported elements can be beneficial to plants. These include iron (Fe), chlorine (Cl), manganese (Mn), copper (Cu), molybdenum (Mo), and Zn. Phytotoxic elements include fluorine (F), thallium (Tl), vanadium (V), mercury (Hg), Al, Cd, Ni, and Cu in high amounts (Foy, et al., 1978; Smith, 1981). Roots, especially those of seedlings, may be stunted and distorted if Al levels are high in strongly acidic soils. These roots are no longer efficient in absorbing nutrients and water. Though symptoms and sensitivity vary with species, affected plants are often stunted, and deficiencies in Ca, Fe, or P may occur (Foy, et al., 1978). Though no studies have been done on the tree species of ANP, root injury due to Al has been reported for European beech (Fagus sylvatica) and Norway spruce (Picea abies) in Germany (Hasbrouck and Cronan, 1984; Tomlinson, 1981).

Symptoms of heavy metal toxicity resemble those of nutrient deficiency (Nash, 1973a). But if levels are low, foliage accumulates these metals without symptom expression. Yarrow (Achillea millefolium) is reported to accumulate chromium (Cr), Hg, and Pb (Pilegaard, 1976); and spruce needles accumulate Cr, Cu, Fe, Pb, and Zn (Nilsson, 1972). These metals may then accumulate in the needle litter and may suppress decomposition (Nilsson, 1972; Ruhling and Tyler, 1973).

In a bog habitat, needles of black spruce (Picea mariana) accumulate the highest level of metals. Sphagnum moss accumulates the next highest amount, followed by Cladina, a lichen. Smaller amounts were found in the woody-bog species leather-leaf (Chamaedaphne calyculata) and Labrador tea (Ledum groenlandicum) (Glooschenko, et al., 1981).

When Pb is measured, however, lichens and mosses had the largest accumulation. Smith and Siccama (1981) list the following ratio for lead accumulation: lichens > mosses > tree twigs > roots > bark > leaves > bracket fungi > wood. Accumulations of approximately 1000 ppm lead have been reported in lichen thalli. Lead can suppress growth of young thalli though this response has not been observed in large thalli (Lawrey and Hale, 1979). Lichens have a differential affinity for various metals, and the following ratio is reported for selective uptake; Fe >> Pb > Cu > Ni = Zn > Co (cobalt) (James, 1973). Sphagnum is reported to accumulate Hg (mercury) (Madsen, 1981), and mosses and other bryophytes are reported to accumulate Cd, Pb, Cu, Ni, Cr, and Zn (Folkeson, 1979; Groet, 1976).

Respiration and photosynthesis are disrupted by heavy metals. Cadmium affects respiration by destroying enzyme activity (Bond, et al., 1976). Chlorophyll molecules are altered by Cu and Hg; photosynthesis is stopped entirely by Hg. For lichens, Puckett (1976) reports that Hg halts CO₂ fixation; Co reduces CO₂ fixation by 40%; Cu and Cd have no effect on CO₂ fixation; and Ni, Pb and P increase CO₂ fixation.

Halogens

The major pollutants in this group are fluorine and chlorine compounds. Currently there is little danger of fluorine toxicity to plant species in ANP. Sources for airborne fluorine are aluminum smelting, steel manufacturing, phosphate fertilizer manufacturing, refineries, ceramic and glass manufacturing, brick plants, and burning of low-grade coal (Amundson and Weinstein, 1980; Gilbert, 1973). If any of these sources were to be introduced near the park, it should be noted that hydrogen fluoride is very toxic to plants.

Fluorides are reported to affect lichen flora (Ferry, et al., 1973; Skye, 1979), and to cause breakdown of chlorophyll, plasmolysis of algal cells, and bleaching or color change of the thallus (Gilbert, 1973; LeBlanc, et al., 1971; Nash, 1971). Lichens may accumulate fluorides especially if the thallus is moist (Nash, 1971).

Trees may be as sensitive to fluorides as lichens (Gilbert, 1973), and they can also accumulate fluorides (Smith, 1970). Fluorides are reported to reduce photosynthesis in white pine, red maple, and larch (Larix deciduca) (Amundson and Weinstein, 1980).

Fluorides also interfere with respiration and carbohydrate metabolism, protein synthesis, cell-wall formation, energy balance, and nucleic acid synthesis (Nash, 1973a). Generally, symptoms resemble SO_2 injury with marginal and interveinal lesions on broad-leaf trees and red-brown necrotic tips on conifer needles (Smith, 1970). As with SO_2 , spongy mesophyll cells and lower epidermis cells collapse first followed by palisade parenchyma cells, upper epidermal cells, and finally vascular tissue (Smith, 1970).

Chlorides cause the same symptoms and biochemical alterations except necrosis is interveinal rather than marginal (Daines, et al., 1967). Fully expanded leaves and elongated needles of pine are most sensitive to chloride toxicity (Daines, et al., 1967). Toxicity can also be attributed to hydrochloric acid which may be a component of acidic deposition.

Ethylene

Ethylene, a by-product of fuel combustion, is a plant hormone causing maturation and senescence. Ethylene can cause retarded growth and early abscission of needles and leaves (Brandt, 1967). Little in the literature is relevant to the affects of ethylene on the flora of ANP.

Particulate Matter

Moreover, there is little in the literature specifically regarding the effects of particulate matter on the flora of ANP. Stomates may be blocked or wedged open; thereby particles inter-

fere with gas exchange and photosynthesis (Dochinger, 1980; Saunders, 1971). Few references concern the physiological effects on trees or their surface pathogens due to heavy metals and so on, which may be components of the particulate matter (Smith, 1970; Staskawicz and Smith, 1977). Increased research efforts in this area will probably be directed toward dry deposition in relation to acidic deposition.

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AQUATIC ORGANISMS

The literature addressing the effects of pollutants on aquatic organisms concerns acidification and related processes. Aquatic organisms are affected by chemicals in the water rather than by pollutants suspended in the air. Because there has been no research measuring the biological effects of acidic deposition at ANP, this study included extensive matching of research conducted elsewhere with aquatic organisms found in ANP. There is a limited taxonomic record of the invertebrates, fungi, algae, and bacteria of the park though there is an old survey of the insect fauna of MDI (Procter, 1946). Thus the research on specific small organisms often cannot be applied directly to related organisms found in ANP. Unless research conducted elsewhere was on a particular genus and species known to occur in ANP, the reference was excluded from this report. When the research discussed more general groupings, such as a whole order or family known to occur in the park, the reference was included. The surface waters in ANP are of low alkalinity (Omernik and Powers, 1982), and most lakes would be classified as low-elevation oligotrophic lakes with limited buffering capacity (Davis, et al., 1978). Generally, pH and alkalinity in Maine lakes have declined slightly over the last fifty years, presumably due to acidic deposition (Davis, et al., 1978; Galloway, et al., 1980; Haines and Akeilaszek, 1983), though comparison of current water chemistry data with historical data must be made with caution because techniques have been revised and the old

colorimetric measurements generally overstated pH (Haines, 1981). Historic records for Long Pond, Upper Hadlock Pond, Bubble Pond, Jordon Pond, and Echo Lake indicate a small depression in pH between 1942 and the late 1970s (Norton, et al., 1981).

There is little information on water chemistry for streams, especially first, second, and third order streams, as found in ANP. Streams with low alkalinity (unbuffered) can oscillate as much as 2 pH units (100 times the H^+ concentration) in a few days (Galloway, et al., 1980). In general, precipitation is most acidic from May to September and least acidic from December to February, but this is not always reflected in stream pH levels. At Hubbard Brook Forest in New Hampshire, pH levels in streams remain relatively uniform throughout the year (Hornbeck, et al., 1976). Depressed pH levels, however, are usually reported during spring snowmelt and following heavy autumn rains (Driscoll, et al., 1980; Haines, 1981). The majority of the acids and metals in the snowpack are released early in the snowmelt (Glass, 1980; Johannessen, et al., 1980) while high aluminum concentrations and low pH levels have been measured following autumn rainstorms (Driscoll, et al., 1980). Increased lead concentrations are assumed to come from atmospheric sources, while zinc may be atmospherically transported or leached from the soil along with magnesium, calcium, and aluminum (Galloway, et al., 1980; Cronan and Schofield, 1979).

Fish

The ability of fish to tolerate acidic waters depends upon the species. The relative tolerance of common fish species at

ANP ranging from most tolerant to least tolerant are lake trout (Salvelinus namaycush), brook trout (Salvelinus fontinalis), white sucker (Catostomus commersoni), brown trout (Salmo trutta), Atlantic salmon (Salmo salar), and rainbow trout (Salmo gairdneri) (Grande, et al., 1976; Haines and Schofield, 1980; Johansson, et al., 1977). Atlantic salmon are also more sensitive to zinc than brook trout (Herbert and Wakeford, 1964) and white suckers are more sensitive to aluminum than brook trout (Baker and Schofield, 1980).

Fish within the same species, but at different stages of growth vary in tolerance to acidic waters. Adult fish are most resistant to low pH levels, although there are reports of mortality in adult fish. Usually reduction in the population of adult fish occurs from recruitment failure (Haines, 1981). Theoretically, even an increase of 5% in the mortality in the 0 age group will affect total population (Jensen, 1971). The LD-50 (50% mortality) for yolk-sac fry is pH 4.0 and 4.6 for 96 hours exposure for brown and rainbow trout respectively (Brown and Rogers, 1978). Research indicates that eggs (embryos) though affected by low pH (Carrick, 1979), are more tolerant to low pH than fry and alvein stages for brown trout, rainbow trout, and Atlantic salmon (Brown and Rogers, 1978; Daye, 1980; Daye and Garside, 1977; 1979; and 1980; Fritz, 1980; Johansson, et al., 1977; and Menendez, 1976). Sensitivity to low pH decreases with age (Baker and Schofield, 1980; Kwain, 1975). For Atlantic salmon, chloionase, the hatching enzyme is inhibited at pH 4.0 to 5.5, and lower activity in the egg when exposed to low pH

water, may result in failure of the outer membrane to rupture (Haya and Waiwood, 1981; Peterson, et al., 1980a; 1980b). The hatchability of brook trout embryos decrease with decreasing pH (Menendez, 1976). Spawning may also be restricted. Brook trout avoid sites with low pH levels; Creaser (1930) reported that brook trout would spawn at pH levels as low as 4.1, while Johnson and Webster (1977) reported that female brook trout avoid water with pH 4.0 and 4.5, but not 5.0. Trout, as well as other fish, are found in poor spawning streams if pH levels are near normal, but not in acid streams (Linthurst, et al., 1983). Female white suckers will not release their eggs in acidified lakes (Beamish, 1976). The lack of egg release is believed to be due to low levels of calcium in the blood (Beamish, et al., 1975). The presence of limestone improved the survival rate of the eggs and fry of rainbow trout (Gunn and Keller, 1980), though alkaline waters can also be detrimental to developing fish (Bishal, 1960; Krishna, 1953). Although fish become more tolerant of acidic waters as they age, sublethal pH levels have been associated with reduced growth rates for brook trout (Menendez, 1976), rainbow trout (Gjedrem, 1980), and white suckers (Beamish, 1974). No effect on growth rate has been reported for brown trout (Jacobsen, 1977).

In laboratory tests, Beamish (1972) observed that white suckers did not feed below pH 4.5 even though food supplies were adequate. Growth rate for some species may increase as food availability increases because of lack of competition (Linthurst, et al., 1983; Haines, 1981). The growth rates of yellow perch (Perca flavescens) increase in acidic waters for age classes of

one to three years, but decrease for older fish (Rayan and Harvey, 1980). Skeletal deformities associated with acidic waters have been reported only for white suckers (Beamish, 1972; Beamish, et al., 1975; Tojnar, 1977a).

Experiments have shown that rainbow trout and Atlantic salmon are unable to acclimate to acidic waters (Daye, 1980; Lloyd and Jordan, 1964). But brook trout can acclimate to sub-lethal pH waters, especially if hatched at low pH (Trojnar, 1977b; Vaala, et al., 1971); and if the brook trout are first acclimated to elevated zinc levels, they are more likely to survive transfer from a hatchery to an acidic lake (Schofield, 1965). Certain strains and individuals of brook trout are more tolerant of acidic water (Dunson and Martin, 1973; Robinson, et al., 1976; Schofield, 1976; Swarts, et al., 1978). This characteristic is inherited, and it is not related to parental exposure to acidic waters (Menendez, 1976) nor to sex (Swarts, et al., 1978).

Several factors have been associated with survival of fish that may contribute to or help alleviate acid stress. Higher water temperatures reduce fish survival at low pH (Kwain, 1975; Robinson, et al., 1976); but increases in water hardness (increases in calcium, sodium, magnesium, and potassium) can increase tolerance to low pH (Lloyd and Jordan, 1964; Muniz and Leivestad, 1980). Calcium is especially important in moderating the effects of low pH (Leivestad, et al., 1980; McDonald, 1980; McWilliams, et al., 1980; Nelson, 1982). For rainbow trout, reduction in the dissolved oxygen concentration of water in-

creases the toxicity of copper, lead, and zinc (Lloyd, 1961).

Fish in acidic waters may accumulate heavy metals that are more mobile at lower pH levels. Elevated mercury concentrations in fish from acidic waters have been reported for Scandinavia and northeastern North America, including Maine (Akielaszek and Haines, 1981; Drummond, et al., 1974; Linthurst, et al., 1983; Jernelov, 1979; Tomlinson, et al., 1980). The concentration of mercury in yellow perch is correlated with the hydrogen ion concentration. For most species, though, accumulation is related to alkalinity (Suns, et al., 1980). There is little evidence of any adverse effects from mercury on fish (Linthurst, et al., 1983) except for an increase in cough response of brook trout (Drummond, et al., 1974). Copper has also been shown to cause an increase in this cough response in brook trout as well as altering feeding behavior, locomotor activity, and possibly creating some long-term problems (Drummond, et al., 1973; McKim and Benoit, 1971). Copper also affects the swimming performance of rainbow trout (Waiwood and Beamish, 1978). Elevated chromium, fluoride, zinc, and nitrate have been shown to adversely affect rainbow trout in laboratory tests (Neuhold and Sigler, 1960; Russo, et al., 1981; Sinley, et al., 1974; Van Der Putte, 1981). Pumpkinseeds (Lepomis gibbosus) accumulated more lead in slightly acidic water (pH 6.0) than in circum-neutral waters (Merlini and Pozzi, 1977). Atlantic salmon are more sensitive to zinc than are trout (Herbert and Wakeford, 1964), and white suckers are more sensitive to aluminum toxicity than are brook trout (Baker and Schofield, 1980).

While aluminum found in surface waters is primarily organic,

it is the inorganic forms of aluminum that are most toxic to fish (Driscoll, et al., 1980; Muniz and Leivestad, 1980; Tollan, 1980). Inorganic aluminum accounts for aluminum fluctuation in streams, and it has been observed that inorganic aluminum levels increase when pH levels drop (Driscoll, et al., 1980). Direct deleterious effects of aluminum have been demonstrated for brook trout, rainbow trout, and white suckers (Baker and Schofield, 1982; Freeman and Everhart, 1971; Muniz and Leivestad, 1980; Schofield and Trojnar, 1980). Aluminum is especially toxic to brook trout fry at pH 5.2 to 5.4 (Baker and Schofield, 1980; Schofield and Trojnar, 1979), and it is reportedly more important to survival than pH level or calcium concentration in acidic snowmelt runoff (Schofield and Trojnar, 1979). In laboratory tests, aluminum over 1.5 ppm caused reduced growth, behavioral changes, and physiological aberrations as well as mortality (Freeman and Everhart, 1971).

The toxic effects of aluminum were not initially recognized in research conducted on fish in acidified waters, and misleading data occur in the literature that indicate survival rates at pH levels lower than fish may tolerate in the natural environment (Muniz and Leivestad, 1980). Such contradictory data exist for several salmonoid species found in ANP. For example, Daye and Garside (1975) report pH 3.5 to be lethal for fingerlings of brook trout (Salvelinus fontinalis), while Baker and Schofield (1980) report pH levels as high as 5.4 to be toxic when inorganic aluminum is present. The exception to aluminum toxicity is at the egg stage when aluminum is actually beneficial to egg sur-

vival in acidic waters (Baker and Schofield, 1980).

The mode of action of most toxicants is still unclear, but physiological studies help explain the complex phenomenon called acid stress. Death in acidic waters is often attributed to failure of the gill membrane (Daye and Garside, 1980; Fromm, 1980; Schofield and Trojnar, 1979). Oxygen and ion regulation (especially sodium and chloride) are disrupted when the gill epithelial cells are damaged, become more permeable to sodium, or the gills become blocked by mucus (Daye and Garside, 1976; 1980; Fritz, 1980; Leivestad, et al., 1980; McWilliams and Potts, 1978; McWilliams, et al., 1980; Packer, 1979; Packer and Dunson, 1972; Plonka and Neff, 1969; Reynolds and Haines, 1980; Rosseland, 1980; Vaala, et al., 1969). Increased calcium content in the water reduces sodium loss and helps elevate acid stress (McWilliams and Potts, 1978; McWilliams, et al., 1980). Reduced concentration of dissolved oxygen or increased concentration of carbon dioxide in acidic water also affects survival because blood has a higher affinity for carbon dioxide than for oxygen (Alabaster, et al., 1957). Rainbow trout have the ability to extract oxygen over a wide pH range (Wiebe, et al., 1934), perhaps due to compensation in the red blood cells (Plonka, et al., 1971; Waiwood, 1980). In acidic waters, if decrease in oxygen tension in the blood occurs, brook trout can at least partially compensate by an increase in the number of red blood cells (Neville, 1979; Vaala, et al., 1971), buffering of fluids, decrease in metabolism, or decrease in locomotor activity (Dively, et al., 1977; Eddy, 1976). But without erythrocyte or other compensation, lower oxygen tension can be fatal (Vaala, et al.,

1969). Decreases in blood pH and serum sodium concentrations associated with reduced oxygen tension in the blood (Packer and Dunson, 1970; 1972; Vaala and Mitchell, 1970) may be fatal to fish (Fromm, 1980; Jonas, et al., 1972; Packer and Dunson 1970; 1972). It should be noted, however, that the pH levels used in these experiments were very low, usually between pH 2.0 and 3.5, which is certainly well below the pH levels of surface waters in ANP.

An increase in muscle lipids has been observed in brook trout exposed to acidic conditions (Dively, et al., 1971), but no changes were observed in liver histones when brook trout were exposed to low pH for four days (Crissman and Therrien, 1969). Changes have been observed in the renal (kidney) system of brook trout that resulted in inhibition of DNA replication and RNA synthesis and a reduction in diameter of the nucleus (Mudge, et al., 1977).

Results from laboratory experiments should be cautiously applied to the field situation. The field situation is far more complex than can be duplicated in the laboratory and cause-and-effect relationships between acidification and loss of fish population are difficult to establish. In this regard, Linthurst, et al. (1983) state:

In regions of the United States, other than the Adirondack Mountain area of New York State, no adverse effects of acidic deposition and/or acidification on fish have been definitely identified.

This should not be interpreted to mean that fish resources in ANP have not been or will not be affected. A large body of research has shown that many fish species found in ANP are sensitive to

acidification and its secondary pollutants such as an increase in the inorganic aluminum concentrations in aquatic ecosystems. The use of lime to neutralize acidity may harm fish species also. An abrupt pH change can stress fish and the pH increase may release toxic materials into the water that were inactive precipitates at low pH levels (Almer, et al., 1978).

Amphibians

Low pH levels and increases in heavy metals associated with spring snowmelt coincide with the breeding season for many aquatic animals. Amphibians are especially sensitive to acidic waters during their breeding period. Temporary pools are common breeding sites for salamanders. Spotted salamander (Ambystoma maculatum) can tolerate pH levels from 6 to 10, but egg mortality exceeds 60% when pH levels drop below 6.0 (Pough, 1976; Pough and Wilson, 1976). Adult spotted salamanders prefer neutral waters, but they are not as sensitive to low pH levels as are the egg and juvenile stages (Mushinsky and Brodien, 1975).

Fertility of the northern leopard frog (Rana pipiens) is also reduced in waters below pH 6.5 (Schlichter, 1981). Healthy embryos were only found above pH 6.0, and no eggs developed below pH 4.8 due to sperm motility failure. Though fertilization was the most sensitive stage, the stages of activation and cleavage were decreased below pH 6.3. Acidic waters may stress adult R. pipiens by upsetting the sodium ion transport regulated by the animal's skin (Fromm, 1981). In addition, populations of spring peepers (Hyla crucifer) and bullfrogs (Rana catesbeiana) are

reported to be sensitive to acidic waters (Linthurst, et al., 1983).

Wetland Plants

Acidification generally results in simplified aquatic communities with decreased species diversity and a few tolerant species dominating (Hendrey, 1978; 1980; Tollan, 1980). Changes in the ecosystem structure occur at all trophic levels, including altered animal habitats, inhibition of decomposition, and a delay in nutrient cycling (Conway and Hendrey, 1981; Foree and McCarty, 1970; Grahn, et al., 1974; Hendrey and Barvenik, 1978; Hendrey and Vertucci, 1980; Hendrey, et al., 1976; Linthurst, et al., 1983; Tollan, 1980). It has been observed that as pH levels in lakes drop, Sphagnum species replace water lobelia (Lobelia dortmanna) and quillwort (Isoetes sp.). There is also an increase in bladderworts (Utricularia spp.), oaks pondweed (Potamogeton oakesianus), and white buttons (Eriocaulon septangulare), while spike rush (Eleocharis acicularis) decreases (Conway and Hendrey, 1981; Grahn, 1976; Grahn, et al., 1974; Hendrey and Vertucci, 1980; Hendrey, et al. 1980; Jackson and Charles, 1981). Sulfates can reduce the population of common bladderworts (Utricularia vulgaris) and pondweed (Potamogeton epihydrus) even if waters stay above pH 6. Spike rush seems to be tolerant of sulfates (Gorham and Gordon, 1963), but the toxicity of sulfur to Chlorella vulgaris, a common green alga, is pH-dependent (Spedding, et al., 1980). Species found in acidic lakes also occur in non-acidic waters, but in lower numbers. The acid-tolerant species have a competitive advantage when pH drops

(Jackson and Charles, 1981), and macrophyte biomass may even increase when water clarity increases and more sediments can be colonized (Hendrey, et al., 1980).

Changes also occur in algal populations as a result of acidification. In non-acidic oligotrophic lakes the golden-brown algae, diatoms, and green algae dominate, but with declining pH, there are fewer species, and often dinoflagellates dominate and golden-brown algae are important (Conway and Hendrey, 1981; Hendrey, 1980; Linthurst, et al., 1983). In some cases blue-green algae dominate (Conway and Hendrey, 1981), but it is difficult to generalize because there are many differences reported (Linthurst, et al., 1983). Benthic (bottom) organisms may have an advantage over pelagic (near the surface of the water) algae because more phosphorus would be available in the sediments (Conway and Hendrey, 1981; Hendrey, et al., 1980; Jones, 1977) or these algae may be able to better tolerate the low level of carbon dioxide in acidic waters (Moss, 1973; Nielson, 1944). Algal and moss mats have been observed overgrowing bottoms of acidified lakes (Grahn, et al., 1974; Linthurst, et al., 1983), and such mats prevent normal mineral cycling by absorbing and binding metals (Hendrey and Vertucci, 1980; Linthurst, et al., 1983). Shifts in diatom flora represent changes in lake pH levels (Merilainen, 1967), and the diatom record in sediments is used to estimate historic pH levels of lakes and ponds (Davis, et al., 1983). Diatoms associated with acidity are observed to be primarily benthic (Davis, et al., 1978; Norton, et al., 1981).

Bacterial decomposers are replaced by fungi as pH levels

drop, and detritus accumulates on acidic lake bottoms as decomposition is slowed or inhibited (Hendrey and Barvenik, 1978; Hendrey, et al., 1976). Though decomposition is inhibited below pH 5.0, changes in normal chemical processes, such as oxygen uptake or concentration of organic matter, may explain this phenomenon rather than reduced bacterial counts (Linthurst, et al., 1983). Alterations in the numbers of detritus-eating invertebrates may also account for the accumulation.

Though dramatic changes can occur in acidified lakes and streams, it is unclear that primary productivity is affected by acidification (Linthurst, et al., 1983). Generally, levels of productivity will vary between different levels in the lake due to available nutrients, season of the year, chemistry, and species present (Jones, 1977; Muller, 1980). Phosphorus often limits growth in oligotrophic lakes and is more closely correlated to biomass than is pH. Aluminum reacts with dissolved phosphorus to form a precipitate that is unavailable to plant life (Almer, et al., 1978; Cronan and Schofield, 1979; Hendrey, et al., 1980). Shifts in species composition toward a predominance of inedible species may occur if invertebrate grazers increase when fish population declines (Linthurst, et al., 1983). Changes such as this are hypothetical, but they exemplify that alterations in any one trophic level affect organisms in other trophic levels.

Invertebrates

Aquatic invertebrates vary in their response to pollutants. Molluscs and large crustaceans are very sensitive to decreases in

pH as their calcareous shells are highly soluble in acidic water (Wiederholm and Eriksson, 1977). Fresh-water snails help keep aquatic plant surfaces clear of algae, and they serve as food for vertebrates, but few species can be found below pH 6.0 (Burton, et al., 1981; Hart and Fuller, 1974; Tollan, 1980). Filter feeders, such as clams, also accumulate heavy metals (Hart and Fuller, 1979; Raddum, 1980).

Taxonomic records for invertebrates of ANP should be updated to provide an accurate listing using correct scientific nomenclature. New surveys should be made because these organisms are important indicators of water quality (Linthurst, et al., 1983). Studies from other areas show that species richness, diversity, and total biomass of insects decrease with decreasing pH (Friberg, et al., 1980; Wiederholm and Eriksson, 1977; and others). Though fish predation is reduced in acidic conditions, food sources may also be reduced as mentioned above, and emergence of many insects occurs in the spring coincident to the pH depression during spring snowmelt (Bell, 1971). Insects are most sensitive to pH at emergence, but sensitivity varies widely. Mayflies (such as Baetis and Ephemerella funeralis) are especially sensitive to low pH waters while dragonflies (order Odonata) and midges (family Chironomidae) tolerate wide ranges in pH and temperature (Curry, 1965; Stickney, 1922). Libellula pulchella (dragonfly) nymphs can survive for several hours in pH 1.0 water (Stickney, 1922). Species of stoneflies (order Plecoptera) and caddisflies (order Trichoptera) vary in their response to acidic waters (Bell, 1971; Linthurst, et al., 1983).

Emergence of Leuctra and Nemoura (stoneflies) and Sialis (alderfly) increased in a stream acidified to pH 4.0 while Ephemeralla (mayfly) and Isoperla (stonefly) decreased (Hall and Likens, 1980a). In general, midges, and black flies (family Simuliidae) are well adapted to acidic waters (Linthurst, et al., 1983; Singer, 1981; Henrikson, et al., 1980) and often midges such as Chironomus riparius are the predominate insects on acidic lakes (Havas and Hutchinson, 1982). In contrast to gilled insects, air breathing Hemiptera (true bugs) and Coleoptera (beetles) are not usually affected by low pH and may be abundant if fish predators have been eliminated (Eriksson, et al., 1980; Henrikson, et al., 1980; Raddum, 1980; Singer, 1981; Vangenechten and Vanderborght, 1980). Also insects living in the sediment are less affected by pH than those in the water (Linthurst, et al., 1983). Invertebrate predators eat smaller prey than fish do, so there is a shift in lower trophic orders (Eriksson, et al., 1980; Henrikson, et al., 1980). When insects are grouped by feeding methods, shredder species increase and scrapper species decrease in an acidic stream compared with a reference stream (Friberg, et al., 1980). In an artificially acidified stream in New Hampshire (emulating snowmelt), scrappers were nearly lost from the population, collectors were decreased by 70%, and total emerging aerial adults were decreased by 37%. In the first week after acidification the number of predators, grazers, and collectors decreased, but there was no decrease in the number of shredders (Hall and Likens, 1980a; b).

Population changes also occur in zooplankton communities where biomass and species diversity decrease and there is a shift

in species composition though no new species occur in acidic water that are not found in circum-neutral water (Diehl, 1972; Groeterund, 1972; Hendrey, et al., 1980; Malley, et al., 1981). Zooplankton may accumulate heavy metals from phytoplankton, and this could lead to inhibition of growth or sexual development (Hart and Fuller, 1979).

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SUMMARY OF POTENTIAL IMPACTS ON PARK RESOURCES

There is little information available on the effects of pollutants on wildlife of ANP. It appears at present that the most important effects of pollution would be indirect. Shifts in habitat such as loss (or gain) of a food source when insects instead of fish dominate surface waters, will affect bird and other wildlife. Shifts in habitat are hard to convert back to the original condition.

Though major point sources occur within 100 km of ANP, there are no sources currently close enough to eliminate sensitive vascular plant species. However, new construction of point sources such as fossil fuel generating plants, paper mills, smelters, etc. within 100 km should be reviewed carefully as there would be an increase in pollutants arriving at ANP and if a proposed site was 40 km or less upwind from ANP, adverse effects on the natural resources could be expected. Plants do have a capacity to adapt to certain levels of pollution, but there is a variation in tolerance between species and within a species. Generally evergreen vascular plants are more sensitive than deciduous species, but non-vascular plants are more vulnerable than vascular plants to some pollutants. Some pollutants, such as the nitrogen in acidic deposition, may be fertilizing the flora of ANP, but eastern white pine has begun to show effects from pollutants transported long distances. If elevated O_3 and SO_2 levels become the norm in ANP the more sensitive members of this species will die and some genetic diversity will be lost.

As ANP is receiving acidic precipitation and is rated sensitive to acidic deposition, it is probable that the biotic and abiotic components of small streams and ponds are being stressed especially during snowmelt or after heavy rainfalls. Elevated Al may affect fish as well as other types of aquatic organisms. Reproductive stages are particularly vulnerable and losses at the 0 age group may not be observed for several years. Changes are reported at all trophic levels as a result of acid stress and often result in shifts in species composition and a reduction in species diversity. Research is still needed to determine what effects acidic precipitation may be having on the aquatic life of ANP.

RESEARCH NEEDS

The goals for management of Acadia National Park are different from those of a commercial forest. Scenic beauty, bird watching, berry picking, fishing, and even the bugs are part of the experience. Visitors expect a high quality and safe experience. In order to protect the public interest, park managers must predict possible impacts of various air pollutants and acidic deposition on the flora and fauna of ANP. Lower timber productivity due to air pollution is of less concern than are shifts in species composition or the presence of obvious damage from effects caused by humans.

Shifts in species composition, a potential ecological consequence of pollution, can only be recognized after a thorough survey of which species currently dwell in the park. Several groups are poorly surveyed at ANP. For example, little is known about ANP's non-vascular plants. Though a lichen survey was conducted in 1983, taxonomic surveys on bryophytes, fungi, and algae should be done. Non-vascular plant surveys are particularly important. These plants are useful early indicators of air pollution damage because they are more sensitive than most vascular plants to certain pollutants. In aquatic habitats, shifts in species composition of invertebrates and plants usually occur before fish populations are lost. More details are needed concerning the species of aquatic insects, aquatic plants, and plankton occurring in ANP. This information would make possible

better utilization of research conducted elsewhere on species identified as indicators of pollutants.

Even for composition changes in the well-understood species, more knowledge is needed about the causal pollutants at ANP. In addition to the monitoring of O₃ and of wet and dry deposition, SO₂ should be monitored at ANP. Roadside contamination by heavy metals should be periodically checked by analyzing soil invertebrates such as earthworms. Heavy metals transported long distances can be analyzed using litter and duff. Metals (such as Al) mobilized in aquatic ecosystems should be analyzed periodically along with pH and other water quality factors of surface waters within ANP. After baseline data is established for the presence of heavy metals, future monitoring could be at five-or-ten year intervals, or even longer depending upon the rate of deposition and any changes occurring in the rate of deposition.

An extensive baseline water quality project is just being completed which monitored 23 lakes or ponds and 26 streams over a two year period. Only Sargent Mt. Pond had a pH below 5.0 and a negative alkalinity. In the spring melt acidic pH levels were found for a few first-order streams (S. Norton, J. S. Kahl, J. Andersen, unpublished). Surface water monitoring should be repeated about every five years.

Research is inadequate regarding the effects of pollutants on wildlife populations. To predict the impact of point-source pollutants near ANP, research on shifts in habitats of mammals would be useful. Also, the consequences to mammals (such as beavers) of changes in aquatic habitats could be valuable to ANP.

Regarding plant life, research on the effects of pollutants on predominant understory and epiphytic plants of ANP should be conducted. Experiments conducted with bracken fern reproduction indicate that research on other species of native fern would be justified. Acidic deposition may benefit the many acid-loving, understory species in the park and beneficial as well as detrimental impacts should be measured. Research on the effects of pollutants should be conducted on species of mosses, liverworts, and lichens commonly found in ANP. Research on soil and litter flora (and fauna) would determine if changes in the rates of decomposition or nitrification can be expected. Alteration of host-pathogen or host-pest relationships should be studied for at least a few major parasites, such as white pine blister rust, spruce budworm, and gypsy moth. Little is known about the effects of pollutants on most of the vascular and non-vascular plants found in ANP. There is much work to be done before the possible impacts of pollution on the flora of ANP will be known.

Research conducted at ANP on the effects of pollution on plant species that have been tested elsewhere will provide a measurement of how well data from other locations can be applied to situations in ANP. There are several reasons to suppose that results are not completely transferable. For example, the mountains of ANP are often covered by fog or clouds. The same conditions in the Green Mountains of Vermont result in the forest ecosystem receiving more and lower pH precipitation than areas out of the cloud belt. But the seaside location of ANP may mean that the situation is substantially different because the ocean

fog may be acting as a buffer to the acids in the atmosphere. Thus, before the Vermont results are applied to ANP, research should be conducted on the chemistry of cloud and fog moisture in ANP the impact it may have on the forest ecosystem. An initial study on ocean fog on MDI will be conducted in the summer of 1984 (J. Andersen and D. Cass, personal communication). Of course, not all research must be repeated, but care must be taken when directly applying results from work conducted elsewhere to coastal Maine. Continuation in the FORAST (Forest Responses to Anthropogenic Stress) program will help park personnel evaluate the applicability of research conducted elsewhere by comparing growth rate changes attributable to pollution at ANP to changes reported for the same species receiving the same amount of pollution at another location. The FORAST project is designed to quantify the relative importance of acidic deposition on growth of several eastern forest tree species. Wherever possible the same species are being used throughout the eastern United States. Dendroecological techniques are employed to determine and separate the influence of climate and site factors from the impacts of air pollution. This study will provide an historical perspective as well as a comparative study of the impacts of air pollutants on different geographic areas.

The reproductive and seedling stages of trees are especially sensitive to pollution. More work should be done on the reproduction of tree species common in ANP. The areas to be researched should include fertilization, germination, and initial development when the plant is especially fragile.

Research on nutrient leaching from the forest should also be

conducted. The shallow soils common to many of the mountain habitats in ANP make these areas susceptible to the effects of nutrient leaching. Long-term laboratory and field studies using native soils should be conducted. Forest soils also serve as a sink for pollutants, and more research is needed in this area.

In aquatic habitats, fish populations should especially be monitored for reproductive failure. Shifts in aquatic plant species should be recorded, and more research on the effects of pH on primary productivity is needed. A survey of aquatic plants in surface waters of similar size but with different pH levels could be an initial study in this area. Similar work could then be conducted on aquatic insects. Since amphibians are especially sensitive to low pH, more work should be conducted on the species of this group found in the park. The effect of aluminum is not known for many aquatic organisms found in ANP. Though much is known about the effects of acidity and other pollutants on the physiology of fish, more work remains to be done on the effects of pollutants on other aquatic organisms. Less is known about the impacts of pollution on the aquatic ecosystem as a whole.

Research on possible interactions between aquatic and terrestrial habitats would be valuable. Major research projects on ecosystem interactions should be conducted only after baseline studies have been completed. Other major efforts are needed concerning the effects of interactions between two or more pollutants on the flora and fauna of the park as well as research to determine mechanisms for protecting the resources of ANP. Research techniques, such as those associated with dry deposition

and simulating acid rain, will improve. ANP should continue to stay abreast of research conducted elsewhere because the results will provide information valuable to understanding the impacts of air pollution and acid deposition on the flora and fauna within the park.