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**Pacific Southwest
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General Technical Report
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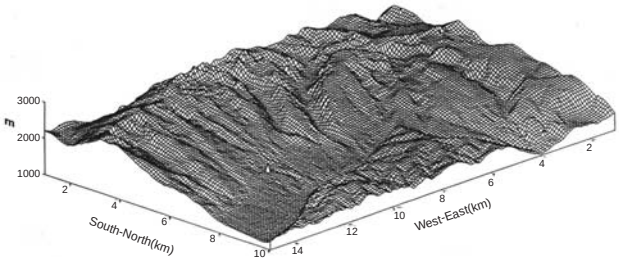
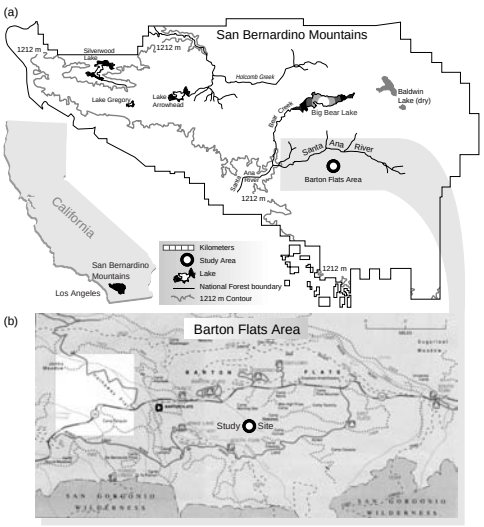
Proceedings of the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems

February 5-9, 1996 **Riverside, California**





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Abstract

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The purpose of the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems held February 5-9, 1996 in Riverside, California, was to summarize results of several years of scientific collaboration between the United States and European specialists about air pollution and climate change effects in Central European and North American forests. More than 40 technical presentations given by scientists and forest managers from the United States and Central Europe were divided into 5 sessions discussing mechanisms of air pollution and climate change effects on plants; air pollution status of forests; results from several gradient and case studies on the effects of air pollution/climate change on forests; information on forest health in several Central European countries; and information regarding photosynthetic active radiation measurements and their use in forest health monitoring.

Retrieval Terms: effects on forests, monitoring, nitrogen deposition, ozone, sulfur dioxide

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Proceedings of the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems

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Andrzej Bytnerowicz Michael J. Arbaugh Susan L. Schilling
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Preface

Industrial air pollution has been identified as one of the primary causes of severe damage to forests of central Europe in the past 30 to 40 years. The mountain forest ecosystems have been affected considerably, resulting in extensive areas of severely deteriorated forest stands (e.g., the Krusne Hory of the Czech Republic or the Izerske and Sudety Mountains along the Polish and Czech border). In addition, the increase of motor vehicles in central Europe has caused a higher amount of nitrogen oxide and hydrocarbon emissions. Under particular weather conditions (high solar radiation, high temperatures, temperature inversions), these compounds may undergo complex chemical transformations resulting in the formation of photochemical smog and a build-up of potentially phytotoxic ozone concentrations. Ozone has already been responsible for injury to ponderosa pine stands in the San Bernardino Mountains in southern California and to forest vegetation in the eastern United States and western Europe. Elevated concentrations of nitrogenous components in photochemical smog and the emissions of ammonia from agricultural activities have caused eutrophication of natural ecosystems, including forests, both in Europe and North America. In addition, changing climatic factors, especially the amount of precipitation, temperature and solar radiation (including ultraviolet-B), modify responses of plants to air pollution, and must be considered when these effects are evaluated.

To successfully prevent additional deterioration of forest health both in central Europe and in the United States, well-coordinated research activities are needed in the areas of air pollution and forest health monitoring, investigations of the effects of air pollution and climate change on forest functioning, and selection and implementation of proper response strategies. These studies may result in stricter air pollution standards in the forested areas and more effective control strategies. Reduction of air pollution and the simultaneous improvement in understanding forest functioning and the application of research results should help prevent future forest decline and rebuild the already damaged forests. Cleaner air also results in the preservation of valuable biological resources and healthier living conditions of people.

Since 1991, U.S. agencies such as the USDA Forest Service (FS), USDA International Cooperation and Development (ICD), and U.S. Environmental Protection Agency (EPA), and the European institutions such as the Polish Academy of Sciences have sponsored a collaborative scientific program entitled "The Effects of Atmospheric Deposition and Climate Change on Forest Ecosystems in Central and Eastern Europe and the United States." This program has resulted in several cooperative studies between U.S. and Polish scientists, introduced the European cooperators to the forest health monitoring procedures used in the U.S., and provided training in using plants as bioindicators of air pollution phytotoxicity. Other cooperative studies, sponsored mainly by the FS, ICD, and European institutions, have been conducted in Bulgaria, Poland, the Czech Republic, and Ukraine. Thanks to the financial support provided by the U.S. State Department, FS, and EPA, and governments of the European countries, the European participants have received advanced training in the U.S. Forest Health Monitoring methodologies, and selected countries have implemented this system. In addition, funds received from the USDA FS Pacific Global Change Research Program and the Southern California Edison Company made possible collaborative research between Polish, U.S., and Canadian scientists on the effects of elevated levels of nitric acid vapor (an important component of photochemical smog) on California tree species.

Partial presentation of the developing plans for this cooperative research and some of the preliminary results were presented at the international conference "Climate and Atmospheric Deposition Studies in Forests" organized by the Polish Academy of Sciences and the USDA Forest Service in October 1992 in Nieborow, Poland. In addition, at the Nieborow meeting, Ann Bartuska of the FS suggested that the Pacific Southwest Research Station, USDA FS organize an international symposium in Riverside, California in 1995/1996. The purpose of such a meeting would be to present and discuss the continuing progress of the U.S.-European research cooperation.

Thanks to the financial support of the USDA ICD, the USDA FS, and a grant from the Southern California Edison, the International Symposium on Air Pollution and Climate Change Effects of Forest Ecosystem was held in the recently renovated historic Mission Inn in downtown Riverside on February 5-9, 1996. The primary goal of the meeting was to discuss the effects of air pollution and climate change and modification of these effects by other abiotic and biotic factors on forest ecosystems. Seventy scientists, forest managers, and decision makers from Belarus, the Czech Republic, Estonia, Hungary, Latvia, Lithuania, Poland, Russia, Slovakia, Ukraine and the United States participated in the meeting.

The participants of the symposium were greeted by the Mayor of the City of Riverside, Ronald Loveridge, who is also a member of the South Coast Air Quality Management District, and James Space, Director of the Pacific Southwest Research Station, USDA Forest Service. In his opening remarks James Space stated, "The issues of global climate change and human-made air pollution are closely interwined and have been the subject of major inter-governmental research programs, both locally and worldwide. The implications are vital to the long-term sustainability of our forest ecosystems." William T. Sommers, Director of the Forest Fire and Atmospheric Sciences Research Staff, USDA Forest Service, outlined the domestic and international activities of the FS related to air pollution and climate change effects on forests and the U.S. forest health monitoring activities. Whetten Reed, Director of the USDA International Cooperation and Development, discussed international cooperative research sponsored by the USDA in central and eastern Europe.

These introductory speeches were followed by five scientific sessions in which more than 40 papers by European and U.S. forest scientists and managers were presented. Session 1 included the mechanisms of the effects of oxidant air pollutants on plants, the use of plants as bioindicators of air pollutants, and the effects of air pollution and climatic factors on mountain forests of southern Poland and the San Bernardino Mountains and Sierra Nevada of California. At Session 2 papers were presented on air pollution status in European and U.S. forest sites, effects of fire on the Siberian boreal forests, and deposition of nitrogen air pollutants to forests and its effects on the health and ecology of natural ecosystems. At Session 3 the results of case studies addressing air pollution and climate change on forests, both in Europe and the United States, were discussed. Session 4 included information on the status of forests in the individual countries, based on the results from the Forest Health Monitoring networks. Session 5 dealt with measurements of photosynthetic active radiation (PAR) and its use for forest health monitoring efforts; and a video, "Integrating Remote Sensing with GIS," produced by the USDA FS, was shown. In general, the meeting provided an update on forest health and air pollution status in forests, mechanisms of air pollution and climate change on forest vegetation, and an evaluation of the possible ecological effects of air pollution and climatic factors in forests of the represented countries.

After 2 days of sessions, three field trips were offered to the symposium participants: (1) in the San Bernardino National Forest and Joshua Tree National

Park in California, the air pollution monitoring efforts, effects of air pollution on natural resources of forest and desert ecosystems, and programs that involve school children in nature preservation were presented; (2) at the Harry Reid Center for Environmental Studies, University of Nevada, Las Vegas, software used for forest health monitoring activities in the United States and some of the central European countries was demonstrated, and the participants received hands-on training of the presented methodologies; and (3) at the Pacific Southwest Research Station, USDA FS in Riverside visitors were acquainted with scientific activities related to the effects of air pollution and climate change on forests, various aspects of research and management related to fire presence in forests, meteorology and fire severity forecasting, and wildland recreation and the urban culture research, and visited the Botanic Garden of the University of California, Riverside and natural history collections of the Riverside Municipal Museum.

Future research cooperation between the U.S. researchers and their European colleagues may include climatic gradient studies in Poland, Belarus, and Russia or cooperation between the Czech Republic, Poland, Slovakia, Ukraine and the United States on ozone distribution studies and phytotoxicity in forests of the Carpathian Mountains. In addition, a similar meeting may be organized in a few of years in one of the European countries to discuss and evaluate progress of the ongoing cooperative work.

The Organizing Committee for the Symposium on February 5-9, 1996 consisted of Alicja Breymeyer, Polish Academy of Sciences; Andrzej Bytnerowicz, USDA Forest Service; William Manning, University of Massachusetts; and Reginald Noble, Bowling Green State University. The Local Arrangements Committee consisted of Michael Arbaugh, Linda Burkholder, Andrzej Bytnerowicz, Paul Miller, Mark Poth and Susan Schilling, Pacific Southwest Research Station, USDA Forest Service.

Andrzej Bytnerowicz,
Technical Coordinator

SESSION I

Air Pollution/Climate Change Effects on Plants: General Topics

William J. Manning and Stefan Godzik, Chairs

Biochemical Reactions of Ozone in Plants[1]

J. Brian Mudd[2]

Abstract

Plants react biochemically to ozone in three phases: with constitutive chemicals in the apoplastic fluid and cell membranes; by forming messenger molecules by the affected constitutive materials (ethylene); and by responding to the messenger molecules with pathogenic RNAs and proteins. For instance, plant reactions with ozone result in constitutive molecules such as the ozonolysis of ethylene in the gas spaces of the leaf, and the reaction with ascorbic acid in the apoplastic fluid. Formation of messenger molecules include the stimulation of ethylene production. Responses to the messenger molecules include the formation of the pathogen related proteins and their mRNAs. Reactions of ozone with biological molecules also frequently result in classical ozonolysis of double bonds, with the production of various aldehydes and peroxides, such as ethylene, isoprene, fatty acids, tryptophan, and some phenylpropenoic acids. Some reactions of ozone with biological molecules do not fit the classical ozonolysis mechanism, such as the oxidation of methionine and some phenylpropenoic acids.

Introduction

Although ozone has toxic effects on both plants and animals, the mechanisms by which the toxic effects are elicited are inadequately understood. However, plants react biochemically to ozone in three phases: with constitutive cells and materials; by generating messenger molecules; and by responding to the messenger molecules.

Reactions with constitutive cells and materials result in substances in the gas phase, (e.g., ethylene) that are dissolved in the apoplastic fluid, (e.g., ascorbic acid) and that react with the lipids and proteins of the plasma membrane of cells surrounding the sub-stomatal cavity. The concentration of ozone in the leaf spaces inside the leaf is zero (Laisk and others 1969). This indicates that ozone passing through the stomata is immediately consumed by oxidizable substances lining the leaf spaces. Because lesions characteristic of ozone damage occur in the palisade parenchyma, palisade cells are part of the sink for ozone.

Reactions of messenger molecules cause stimulation of the synthesis of molecules such as ethylene, while the response of messenger molecules results in the formation of messenger RNA and the proteins for which they provide the genetic code. However, because these three stages of response to ozone are simply an analysis of the problem, a practical method of protecting plants needs to be developed.

This paper discusses the biochemical reactions of plants to ozone, focusing on the reactions of constitutive cell materials, the synthesis of messenger molecules, the responses of the messenger molecules, and various methods to protect plants from ozone damage.

Discussion

Ozone Reaction with Constitutive Materials

Ethylene

Ethylene has been known as a "ripening hormone." The arrival of the ripening is coincident with a "climacteric" in ethylene production. In addition to this

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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function in normal development, ethylene is known to be produced under various forms of stress. The biosynthesis of ethylene has posed a problem that has consumed many years of research, but it is now clearly understood as the following function:

S-adenosyl-methionine ----> ACC (1-aminocyclopropane-1-carboxylate)
ACC -----> C2H2

The two critical enzymes are ACC synthase (which is the rate-limiting enzyme), and ethylene synthetase.

The reaction of ozone with ethylene in the gas phase involves a classical ozonolysis *(fig. 1)*. The degradation products include formaldehyde, hydroxymethylhydroperoxide (HMHP), and hydrogen peroxide. Mehlhorn and Wellburn (1990) and Hewitt and Kok (1991) have suggested that ozone toxicity is a consequence of the reaction of ozone with ethylene. Mehlhorn and others (1990) attribute radicals detected after exposure of plants to ozone to the reaction of ozone with ethylene. Are these reactions taking place in the atmosphere? Hellpointer and Gab (1989) measured the peroxide content in rainwater collected on different days (*table 1*). The major peroxide usually found was hydrogen peroxide. Small amounts of HMHP were also detected. These measurements show that the reaction of ozone with ethylene in the atmosphere is a clear possibility. In the gas phase reaction it was found that hydroxyl radicals could be detected by the conversion of cyclohexane to cyclohexanol and cyclohexanone (Atkinson and others 1992). The yield of radicals was 0.12 per mole of ethylene reacted.

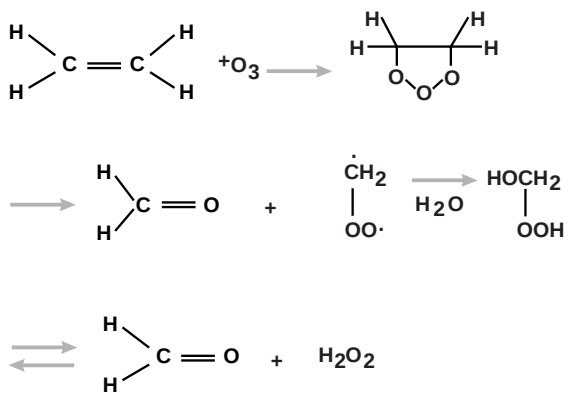


Figure 1 — Reaction of ozone with ethylene.

Table 1 *— Peroxides in rainwater (Hellpointer and Gab 1989).*[1]

H	H_2O_2 [2]	HMHP [3]	HEHP [4]	CH_3OOH [5]
5.9	*7*	*0.2*	*0.1*	*0.2*
6.7	*63.1*	*nd*	*nd*	*0.3*
6.4	*110.6*	*nd*	*nd*	*0.4*

[1] Data are expressed as µmoles/liter.

[2] H_2O_2: hydrogen peroxide.

[3] HMHP: hydroxymethyl hydroperoxide.

[4] HEHP: hydroxyethyl hydroperoxide.

[5] CH_3OOH: methyl hydroperoxide.

Isoprene

Isoprene is the basic member of the group of plant natural products known as isoprenoids. Important classes of these compounds are formed from multiples of the basic five carbon unit: monoterpenes (C10), sesquiterpenes (C15), and diterpenes (C20). These compounds are important articles of commerce because of

characteristic odors. Many terpenoids are volatile and give rise to the characteristic odors associated with some plants.

Another volatile compound produced in sometimes copious amounts by plants is isoprene, which can react with ozone (*fig 2*). Theoretically one molecule of isoprene can give rise to two molecules of HMHP. The HMHP can degrade to formaldehyde and hydrogen peroxide. Results of different laboratory studies indicate in one case that the major product is hydrogen peroxide, and in another case that the major product is HMHP. These compounds are in equilibrium:

HMHP <-----> formaldehyde + hydrogen peroxide.

Hydroxyl radicals are also formed in the reaction of ozone with isoprene. The yield is 0.27 radicals per isoprene reacted. (Atkinson and others 1992).

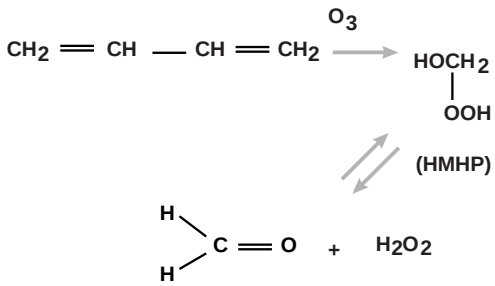


Figure 2 — Reaction of ozone with isoprene.

Fatty acids

Fatty acids are most frequently found in plant tissues esterified with the trihydric alcohol glycerol. Triglycerides are the most important storage fat of seeds. More importantly, when glycerol is esterified with two fatty acids and one polar head group such as phosphorylcholine, the compound formed is amphipathic, which has a lipid soluble moiety and a water soluble moiety. The property of amphipathicity is essential for the formation of the lipid bilayer of the plant membranes. Any oxidation of the fatty acids in the plant membranes will destroy the integrity of the membrane.

The reaction of ozone with fatty acids has a long history — the reaction was used to determine the position of double bonds for many years. Using oleic acid as an example, the reaction is exactly analogous to the reaction of ozone with ethylene (*fig.3*). The double bond is broken with the formation of keto groups on the two nine carbon fragments of oleic acid. The hydroxyhydroperoxide is an intermediate, similar to the case of ethylene. The very significant difference in the ozonolysis of

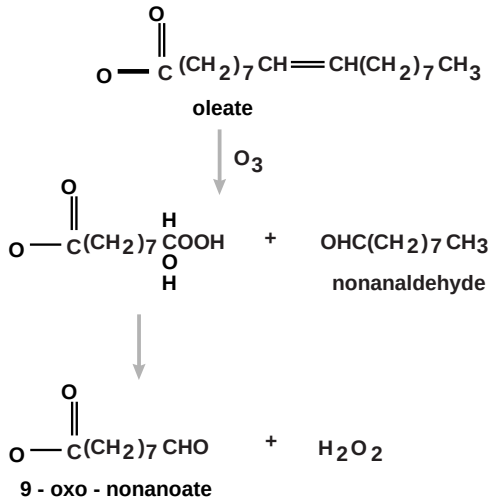
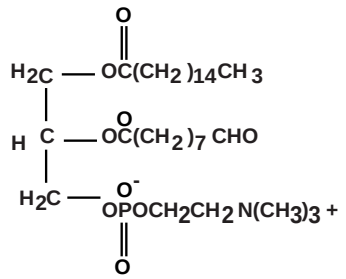


Figure 3 — Reaction of ozone with oleic acid.

fatty acids is that they are most often esterified to form phospholipids, which are a universal feature of biological membranes, forming a lipid bilayer. After ozononlysis the product of phospholipid oxidation has only a nine-carbon chain (1-acyl-2-(9-oxononanyl) sn-glycerophosphocholine (Santrock and others 1992) (*fig. 4*). This compound has the capability to lyse biological membranes.

Even though these reactions can be measured in solutions of phospholipid, it is doubtful that ambient ozone has access to the double bonds in the phospholipid bilayer of the biological membrane.



1-palmitoyl-2-(9-oxononanyl)-sn3-glycerophosphorylcholine
(PN_1PC)

Figure 4 — 1-palmitoyl-2 (9-oxononanyl)-sn3-glycerophosphorylcholine.

Amino Acids and Proteins

All parts of plant cells contain proteins. They may be structural in nature, or they may be catalytic. They may be soluble or membrane-bound. The plasma membrane is close to 50 percent lipid and 50 percent protein. The proteins of the membrane are responsible for various catalytic activities, including pumps for sodium, potassium, and calcium. The proteins of the plasma membrane are probably also responsible for the reception of chemical stimuli (natural and unnatural). The reaction of ozone with membrane proteins can be visualized as having a devastating effect.

Reaction of ozone with a few amino acids and proteins has been well documented (Mudd and others 1969, Previero and others 1963) The reactive amino acids include tryptophan, histidine, and methionine (*fig. 5*). The product of tryptophan ozonolysis is N-formylkynurenine (Kuroda and others 1975, Previero and others 1963), which is consistent with a classical ozonolysis breaking the double bond. The product of ozonolysis of histidine, when it is in peptide linkage, is aspartic acid which also implies a classical ozonolysis of the double bond of the imidazole ring (Berlett and others 1996). The product of oxidation of methionine is methionine sulfoxide (Johnson and Travis 1979, Mudd and others 1969) showing a pathway of ozone oxidation that is clearly not a classical ozonolysis. Some other amino acids that are oxidized by ozone include cysteine, tyrosine, and phenylalanine. Proteins that are inactivated by ozone include lysozyme (Kuroda and others 1975), glyceraldehde-3-phosphate dehydrogenase (Knight and Mudd 1984), and glutamine synthase (Berlett and others 1996).

Ascorbic Acid

Ascorbic acid (vitamin c) is a widely distributed constituent of plant tissue. It has long been known that fruits can prevent and correct the results of vitamin c deficiency in humans. Ascorbic acid is an outstanding antioxidant. It is fundamental important in the detoxification of superoxide produced during the reactions of photosynthesis. Ascorbic acid is also an important cofactor in hydroxylation reactions such as the formation of hydroxyproline. The presence of ascorbic acid in apoplastic fluid can be viewed as a protection against ozone, but it is incomplete since the effects of ozone on plant cells is obvious.

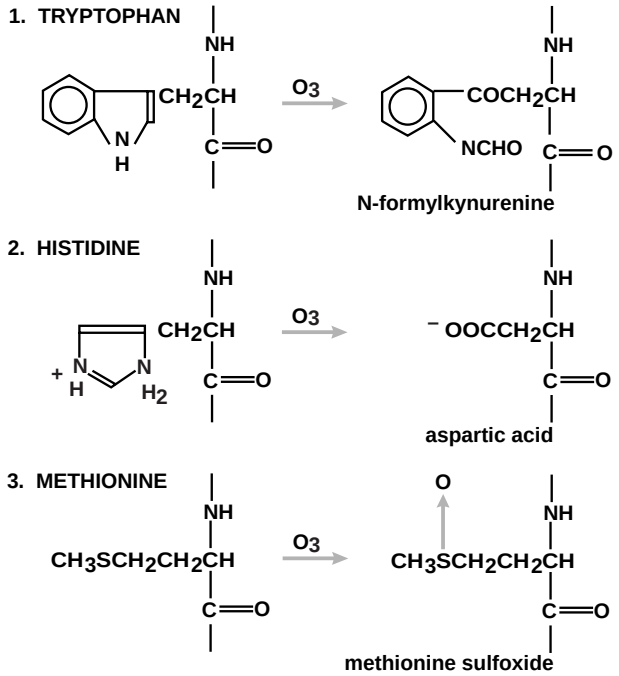


Figure 5 — Reaction of ozone with amino acids.

Foliar spray of ascorbic acid can prevent ozone damage to plants (Freebairn 1957). Ascorbic acid in the apoplastic fluid of plant leaves has also been shown to be oxidized by ozone (Luwe and others 1993). The reaction of ozone with ascorbic acid is stoichiometric, and the reaction is more rapid at pHs above the first pK (pH 4.7) (*fig. 6*) (Giamalva and others 1985). The reactivity of ascorbic acid may also be caused by the cleavage of the double bond in the ring (Chameides 1989), and the oxidation product may be an epoxide formed at the double bond (Kanofsky and Sima 1991). There are precedents for both reactions. An examination of the product of oxidation of ascorbic acid by ozone using nuclear magnetic resonance, mass spectrometry, and enzymic re-reduction using DHA reductase (*fig. 7*) has shown beyond doubt that the product is dehydroascorbic acid. Since dehydroascorbate reductase is common in plants, the oxidation of ascorbic acid by ozone in the apoplastic fluid is not a suicide-protection mechanism but rather a renewable form of defense. This defense does depend on the ability of the plant to take up DHA, re-reduce it in the cytoplasm and then export AA to the apoplastic fluid.

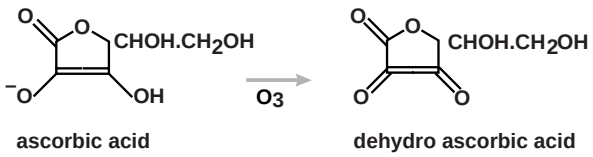


Figure 6 — Reaction of ozone with ascorbic acid.

Phenylpropenoids

The phenylpropenoids are unique plant natural products. They have been recognized and studied for many years because of their abundance and their characteristic reactions. The phenylpropenoids are synthesized by way of shikimic acid and phenylalanine. Phenylalanine is the direct precursor of the simplest phenylpropenoic acid: cinnamic acid, catalyzed by phenylalanine-ammonia lyase (PAL). PAL has received a great deal of attention because it responds to many stimuli, including irradiation, pathogen infection, and ozone. Phenylpropenoic acids are common in plants. They comprise cinnamic acid coumaric acid, caffeic

acid, ferulic acid and sinapic acid. These compounds are important because they are precursors of lignin and flavanoids. They may also have a role in protection of plants from the damaging effects of ultraviolet-B radiation.

The phenylpropenoic acids are susceptible to oxidation by ozone.[3] In some cases the major pathway of oxidation is ozonolytic cleavage of the double bond in the propenoic acid side chain. In other cases the oxidation seems to be directed to the phenolic ring. Pryor (1974) has suggested a mechanism for the attack of ozone on the phenolic compounds that involves radical attack on the phenolate ion. However, the reaction of ozone with the propenoic acid moiety is clearly important.

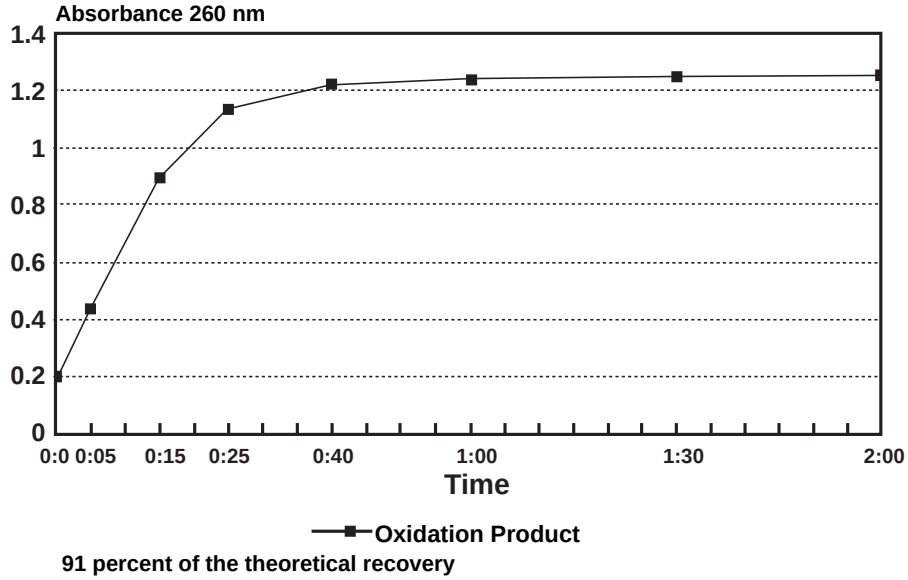


Figure 7 — Re-reduction of ozone-oxidized ascorbic acid by DHA reductase.

Messenger Molecule Synthesis

Ethylene

The production of ethylene after the exposure to ozone is one of the fastest responses known. The speed of the response is consistent with the role of ethylene as a messenger molecule in the succession of events after ozone exposure (Langbartels and others 1991). The messenger RNA for aminocyclopropane carboxylic acid synthase has also been shown to rapidly increase after exposure to ozone (Schlagnhaufer and others 1995). Gunderson and Taylor (1988) have shown that ethylene causes stomatal responses similar to those when plants are exposed to ethylene. Other messenger molecules could be examined such as jasmonic acid and salicylic acid. In the case of salicylic acid the response is too slow to account for responses to ozone (Yalpani and others 1994).

Responses to the Messenger Molecules

Clear responses to messenger molecules are the formation of chitinase and b-glycosidase (Schraudner and others 1992). The formation of these proteins may be a beneficial response to the challenge of ozone, or merely a response to the stress by production of proteins analogous to "pathogen related proteins." It is probable that the late production of these proteins are an indication that damage has been done.

Protection from Ozone

In the early 1950's the protection of plants from ozone was accomplished by application of ascorbic acid (Freebairn 1957). But the protection was short-lived because of the ready oxidation of ascorbic acid in the atmosphere. The protection afforded by ascorbic acid points to a critical distinction among protectant chemicals:

[3] Unpublished data on file at the Department of Botany, University of California, Riverside.

those that are simply sacrificed and lower the effective dose of ozone to the plant, such as foliar applications of ascorbic acid, and those that alter the plant physiologically and biochemically in such a way as to protect the plant from ozone. There are no convincing examples of the second class of protectants at present. Many compounds have been tested for protection, and several have been found effective. The most interesting and most effective of these compounds is N-[2 2-oxo-1-imidazolidynil)ethyl]-N′-phenylurea (EDU). EDU was first synthesized and tested for protection of plants against ozone by chemists at DuPont Chemical Company (Carnahan and others 1978). EDU has been found to be effective both as a foliar spray and as a soil drench (Kostka-Rick and Manning 1993). It has been stated that EDU does not react with ozone (Pitcher and others 1993), and it is therefore a candidate for effectiveness by changing the physiology and biochemistry of the plant. Lee and Chen (1982) have reported that EDU has cytokinin-like activity, which could be related to its activity as an antiozonant. Lee and Bennett (1982) found that the application of EDU increased the activities of superoxide dismutase, catalase, and peroxidase, but these findings have been contradicted by Pitcher and others (1993). The mode of action of EDU is therefore still controversial and worthy of much greater research effort.

Another method for plants to resist the effects of ozone is to develop resistant varieties. Resistant varieties of several agricultural crops include tobacco (Heggestad 1993), alfalfa (Graumann 1972), and maize (Cameron and others 1970). These plants can be studied to understand the reasons for resistance, and breeding programs can be initiated to introduce the desirable trait into all commercially available lines. One method to detect differences in resistant and susceptible plants is to analyze the DNA by restriction fragment length polymorphism (RFLP). Neale and Harry (1995) have discussed the uses of these technologies with particular attention to coniferous trees. The identification of DNA fragments associated with resistance could be useful to breeders, especially since the DNA testing for resistant crosses can be made on seedlings rather than mature plants.

Conclusion

The reaction of ozone with many types of compounds found in plants have been studied in vitro. It is difficult to connect these known chemical reactions to reactions which are responsible for the succession of events in the plant which eventually result in characteristic lesions of ozone damage. How are we going to make the connection between chemical reactions and visual symptoms? One approach is to compare the DNA of genetically related plants that are either ozone-resistant or ozone susceptible. Methods of analysis include the use of DNA probes for known enzymes that may be involved in ozone resistance. Another method is the comparison of DNA fragments derived by action of restriction endonucleases. Techniques of molecular biology offer the most promise for understanding the differences between ozone-resistant and ozone susceptible plants.

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Oxidant Induced Alteration of Carbohydrate Production and Allocation in Plants[1]

Robert L. Heath[2]

Abstract

Urban air basin produced oxidants, notably ozone, induce a decline in productivity in plants. This loss of productivity is manifested by slower growth, hindered development, lower reproduction rates, impaired ability to resist disease, and other stresses. While many metabolic events have been linked to oxidant exposure, three major shifts have been well-studied: increased production and more rapid turnover of antioxidant systems; production of symptoms similar to a mechanical wounding of the tissue, especially ethylene production; and decline in photosynthesis. Although these processes may be linked metabolically at a fundamental level, the mechanisms leading to a decline in photosynthesis have been shown to directly lower plant productivity. There are two distinct changes in physiology which can directly alter the photosynthetic rate by leaves: a closure of the stomata limiting CO_2 concentration, and a decline in the ability to fix CO_2 within the chloroplast. In many studies it is difficult to discriminate which is more critical and which triggers various effects because in the final analysis, both limit carbon assimilation. The mechanisms of stomatal closure may be linked to the loss of membrane permeability and transport because of oxidation of membrane channels and transport proteins and/or oxidation to an increased sensitivity of the stomata to closure signals, such as internal Ca^{2+} levels or abscisic acid. On the other hand, impairment of the processes of photosynthesis is probably not caused by changes in the light gathering photosystems, but rather by a loss of CO_2 fixing ability induced by a decline in Ribulose 1,5-bisphosphate carboxylase/oxygenase (Rubisco) or by an alteration in normal metabolite flow via changes in ionic balance. It is not yet clear which of these mechanism is more critical to each individual plant species under varied environmental conditions. However, the loss of photosynthetic products is only the first step in the loss of productivity. Much evidence indicates that translocation of the fixed carbon is altered by oxidants. Because translocation is very sensitive to energy and carbohydrate status as well as membrane function of the leaf, phloem loading would also be at risk by oxidant exposure. A decline in carbohydrate levels to roots and growing shoot tips would have profound effects upon the plant's ability to grow and respond normally to the integrated effects of the remainder of the environment.

Introduction

Many studies that describe the attack of ozone upon the tissues and physiological processes of green plants have used agriculturally important crops (Heagle 1989; Heath 1980, 1994a, [In press]; Heck and others 1988; Karpinski and others 1993). While native plants in the ecosystem may respond differently to ozone than crop plants, certain fundamentals seem to be constant between species. Thus, some generalities of physiology can be stated with certain confidence. To be sure, we have much more to learn, but the concepts and ideas from basic research adds to our "more practical" knowledge base. Understanding how ecosystems can be protected from pollutants requires the foundations of basic research.

Definitions of Forest Decline

"Forest decline" can be defined as the negative changes within forest ecosystems induced by a degeneration in air quality. Two examples of this decline include the process of sinking to a "weaker" or inferior condition, or a diversion from the "normal" development process. In both cases we are forced to define what is meant by the terms "weaker" and "normal." Studies of "normal" ecosystems are few; thus, normal is not inadequate when defined experimentally. Currently, a "normal" ecosystem is that with very low levels of air pollutants. Finding such an ecosystem is not a simple task (Sandermann and others [In press]). A "weaker" ecosystem is one which is not resistant to attack by biotic or abiotic stresses. The loss of trees in

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the San Bernardino Mountains in southern California is a good example because ozone causes an early loss of foliage, seemingly giving the tree less resources to fight a later attack by bark beetle which ultimately kills the tree (Miller 1992).

The observational definitions of plant "changes" caused by air quality include: [1] visible injury, [2] loss of productivity, [3] inability to withstand other stresses, [4] accelerated senescence, and [5] changes in metabolic pathways. Visible injury is often used as the primary observation of forest decline and includes loss of chlorophyll, necrosis, "water-logging" in deciduous leaves, and production of anthocyanin-containing "spots" (Jacobson and Hill 1970). The loss of productivity has been most easily measured by a decline in timber production in a forest. The inability to withstand other stresses has been found in forests attacked by bark beetles, and an inability to compensate for a lowering of nutrient and water availability are examples of other natural abiotic stress. Accelerated senescence is most easily observed as early loss of needles and crown thinning but can include an altered leaf or root development. Changes in metabolic pathways are the most difficult to observe in the field, yet these observations can most clearly indicate a plant's ability to respond ultimately to the ozone attack.

Plant Physiology and Ozone

Various hypotheses have been proposed to explain the interaction of plant physiological processes and ozone attack (*table 1*). Although the hypotheses overlap at the fundamental biochemical level, each hypothesis is an independent process, and individual research groups tend to focus on only one at a time. The antioxidant protection hypothesis involves both the amount of each antioxidant present and the ability to produce antioxidants which serve to eliminate ozone so that ozone cannot move into any cellular site where it will induce damage. Ozone

Table 1 *— Various hypotheses that explain the modification of plant physiological processes induced by ozone.*

General	Specific	Detailed
Antioxidant protection		
	Superoxide:	
		Superoxide dismutase
	Hydrogen peroxide	
		Peroxidases
	Hydroxyl radical	
		Ascorbate, glutathione
		Tocopherol
Wounding response		
	Wounding proteins or pathogen response proteins	
		Chitinase
		β -glucanase
	Ethylene production	
Loss of photosynthetic capacity		
	Inappropriate stomatal response	
	Photosynthetic processes	
		Photo-inhibition of photosystems
		Loss of carboxylation
		Slowing of translocation
Membrane dysfunction		
	Loss of ion channels (K^+ , Ca^{2+})	
	Loss of permeability	
		Channels
		Membrane structure
	Loss of signal transduction receptors	

itself is not expected to be stable within the cell or its wall because of its high reactivity. Three major transformation species are currently believed to be formed: superoxide (O_2^-), hydrogen peroxide (H_2O_2), and hydroxyl radical ($HO^\cdot$) (Grimes and others 1983, Möller 1989, Mudd [In press]). Other possibilities, such as ozonide radical ($O_3^\cdot$) and peroxyl radical ($HOO^\cdot$) have been described (Heath 1986), but their reactions are currently poorly understood. Data seem to indicate that many oxidative species from ozone can be detoxified easily by reactions with ascorbic acid or glutathione (Benes and others 1995, Bors and others 1989, Chameides 1989, Gupta and others 1991, Guzy and Heath 1993, Kangasjärvi and others 1994, Luweand others 1993, Polle and others 1995, Willekons and others 1994).

If interception by antioxidants is unsuccessful, then ozone (or its oxidative products) will alter the membrane function through oxidation of critical sulfhydryls of ion channels or pumps and induce an increase in general membrane permeability via inhibition of channel closure or alterations in the membrane structure (Heath 1988, 1994a). Another, but unproved, mechanism is the false triggering of the signal transduction receptors by a direct or indirect chemical modification. Both effects would lead to a non-natural balance of ions across all membranes. The most critical of these ions is Ca^{2+}; its plasmamembrane efflux pump is inhibited and its general permeability is increased by ozone (Castillo and Heath 1990). Many metabolic processes are in turn activated as the Ca^{2+} concentration within the cell rises.

One major response of ozone exposure mimics a general wounding, either mechanical or pathogen induced. Since one of the initial triggers of wounding responses is a rise of Ca^{2+} level within the cell, wounding fits well with what is known about ozone attack. This rise in Ca^{2+} level, in turn, triggers a series of metabolic cascades, ultimately generating ethylene and the production of pathogen-response (PR) proteins (Fengmeier and others 1994, Kärenlampi and others 1994, Langebartels and others 1991, Schraudner and others 1992, Sharma and Davis 1994). It is not clear that ozone directly produces these responses; some oxidative product of ozone may be the key chemical. For example, the prevention of ethylene release has been shown to prevent the induction of visible injury (Mehlhorn and Wellburn 1987, Mehlhorn and others 1991), and the visible injury ultimately has been considered a result of the chemical interaction of ozone and the double bond of ethylene producing a toxic product (Gunderson and Taylor 1991, Taylor and others 1988, Tingey and others 1976).

A clear, but mechanistically confusing, plant response to ozone is the loss of photosynthetic capacity. Stomata generally partially close during ozone exposure but, under some conditions, can open further (Heath 1994b). Under many conditions the level of the primary enzyme of CO_2 fixation (ribulose 1,5-bisphosphate carboxylase/oxygenase, Rubisco) declines (Dann and Pell 1989, Nie and others 1993, Pell and others 1994). Yet one clear visible injury pattern, the loss of chlorophyll, signals a problem within the photosystems of the chloroplast (Heath 1989). Photoinhibition can be induced if CO_2 fixation becomes too slow for a given rate of photon capture. Many observed events suggest that while carbon assimilation within the leaf declines, translocation of carbon is inhibited even more so such that growing points of the plant are inhibited and root/shoot ratios are altered (Dugger and Ting 1970, Tjoelker and others 1995).

Examination of the changes in photosynthetic capacity suggests that a multiple series of declines are at the heart of a loss in productivity and resource accumulation of the plant. In general, stomata partially close during ozone exposure. The apertures of stomata are governed by a dynamic balance of water loss and internal CO_2 concentration (which is fixed by a balance of gas flow through the stomata and carbon assimilation) (Farquhar and Sharkey 1982). We do not know if a membrane imbalance first leads to a loss of osmotically accumulated water from the guard cell or if the imbalance is caused by an inhibition of CO_2 fixation. In fact, there are many possible interactive mechanisms: stomata closure can occur with an influx of Ca^{2+} into the guard cell in which its higher concentration changes the sensitivity of the

guard cell to abscisic acid (ABA) transported to the leaf from the roots or produced within the leaf due to a lowered cell water potential (Atkinson and others 1990). These interactive mechanisms make understanding of the full process difficult.

A poor balance between light gathering (photosystems) and CO_2 fixation by Rubisco seems to lead to photoinhibition (Farage and others 1991). Changes in chlorophyll fluorescence patterns and in kinetics of protein production (e.g., the D1 protein of photosystem II) (Barber and Anderson 1992, Osmond 1981) suggest that this is a real mechanism with the possibility that it is not expressed in all plants (Nie and others 1993, Pino and others 1995). The loss of Rubisco is difficult to measure as Rubisco is present in very high concentration. Its level declines in senescing leaves much faster after ozone exposure, suggesting a role in early senescence. An easier method to follow Rubisco changes is by measuring the concentration of its m-RNA leading to the production of the protein *rbc*S (the message for its small subunit is nuclear transcribed). According to studies by Pell and others (1994) and Reddy and others (1993), a rapid, but not complete, loss of the protein message occurs within an hour or so of exposure but recovers within a day after the exposure ceases (*fig. 1*).

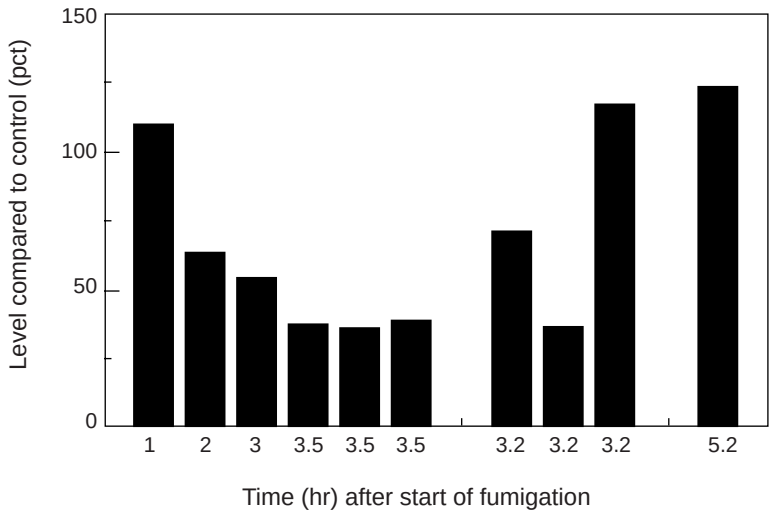


Figure 1 — Changes in the production of m-RNA for the small subunit of Rubisco. Tomato plants, grown in a growth chamber for 3 weeks, were fumigated with 0.33 ppm ozone (produced within O_2 stream) at an air temperature of 28 degrees C and a relative humidity of 25 percent for 3 hours. No visible injury could be seen even after several days, but a slight (ca. 40 percent) stomatal closure could be observed towards the end of the fumigation period (*fig. 3*). The message RNA was probed (Cohen and Bray 1990) and quantified on the slot blot membrane by the phosphor Imager System (Molecular Dynamics). [3]

The full scheme for the interaction between membrane dysfunction and Ca^{2+} changes and the other events observed during ozone exposure is not easily decoded. However, the current data indicates that metabolic pathways are regulated and mutually dependent. When ozone alters one metabolic event many others far removed from that initial site of interaction are changed.

We have used a dual label isotope porometer (Johnson and others 1979) to measure the stomata conductance of water vapor (using inward flowing ^3H_2O as an analog for normal outward flow of water vapor from the leaf). Assimilation is measured simultaneously by $^{14}CO_2$ fixation. The gas stream passing over the leaf has both isotopes (*fig. 2*). Knowing the stomata conductance for H_2O vapor allows the calculation of the conductance for O_3 and with the known external level of ozone, a dose of ozone within the leaf can be calculated (Heath 1994b). The inhibition of stomata conductance is greater than the inhibition of assimilation, but under the low light intensity of these experiments, the stomata can close a great deal without inhibiting assimilation (Heath 1994b). We are currently using the dual label porometer in the single label mode (using only $^{14}CO_2$ at high specific activity) to label photosynthetically- fixed carbon in a leaf in order to follow its movement through translocation. Preliminary results suggest ozone exposure dramatically slows carbon movement.

Another method to continuously measure stomata conductance involves the use of leaf temperature as a probe of evaporative water loss through transpiration.

[3] Mention of trade names or products is for information only and does not imply endorsement by the U.S. Department of Agriculture.

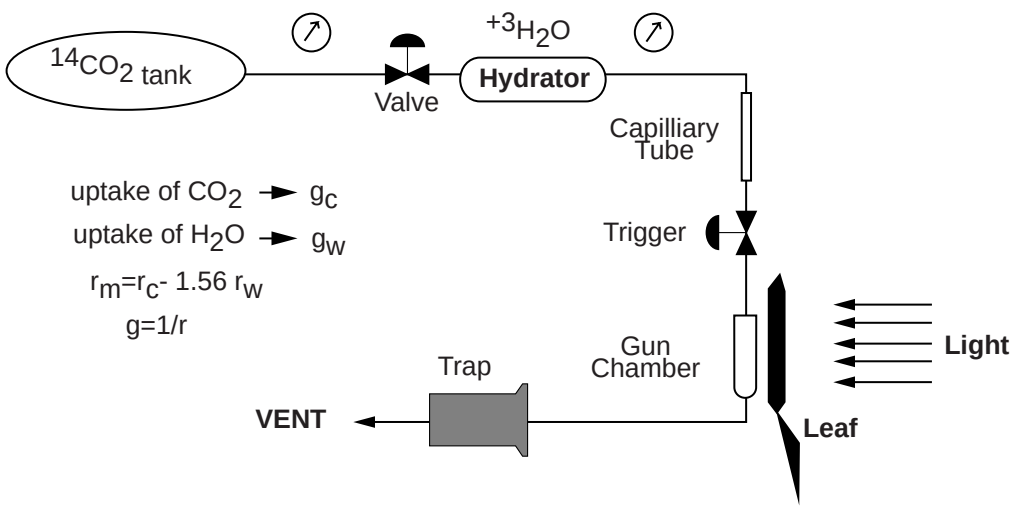
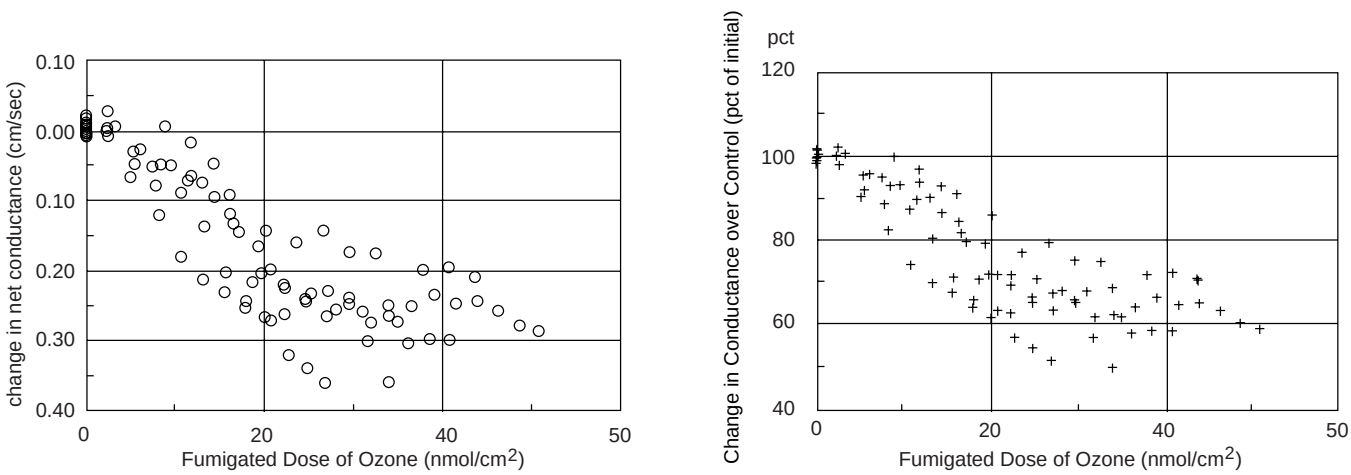


Figure 2 — Dual isotope label porometer (Johnson and others 1979). This machine allows the measurement of water conductivity (measured inversely by the flow of tritiated water from the gas stream into the leaf) and carbon assimilation (measured by the flow of ^{14}C carbon dioxide into the leaf). The area labeled is a circle of diameter of 0.55 cm.

By using two thermocouples wired in series but with legs reversed (the same metal joined at a common junction) and one thermocouple touching the leaf and the other in the air just below the leaf, the differential temperature ($dT = T_{leaf} - T_{air}$) can be continuously recorded. Calibrations allows dT to be used as a measure of transpiration rate. By using the air temperature and relative humidity, the stomata conductance can be calculated, and the ozone dose can be "measured" continuously during exposure. A plot for tomato exposed to ozone was measured by using this technique (*fig. 3*). The change in conductance is measured against the actual delivered dose of ozone to six leaves. Each leaf begins with a slightly different conductance (as they are different leaves on the two plants and are at different developmental ages, which affects their conductance); thus, the total time of exposure at which stomata begin to close varies. However, converting time into accumulated dose for each gives clearer results; the dose for the beginning of closure is about the same as 8 nmole cm^{-2}-leaf area. The stomata close to about 50-60 percent of the initial level at a dose of about 30-35 nmole cm^{-2}-leaf area. Higher doses do not seem to close the stomata any further.



Conclusion

Figure 3 — Changes of stomatal conductance during fumigation of tomato plants with ozone. The data were collected and dose was calculated by the amount of ozone in the air multiplied by the stomatal conductance for ozone each 210 seconds. This is an average of thermocouple measurements for six leaves (2 on 2 plants, 1 on 4 plants). Several plants were destructively removed during the course of the experiment for m-RNA sampling.

Results of the various studies discussed in this paper have made sizable advancement in the understanding of ozone induced plant changes. Several testable hypotheses regarding the mechanism of ozone induced injury have been formulated (Bors and others 1989, Castillo and Heath 1990, Eckardt and Pell 1994, Langebartels and others 1991, Pino and others 1995) and cultivars and mutants of varied species have been generated which differ in their sensitivities to ozone (Guzy and Heath 1993, Sharma and Davis 1994). The understanding of the basic processes of plants has made this progress possible. Our comprehension of how plants respond to the environment continues to grow. In the future, better technology to allow us to measure physiological events, and a more complete understanding of genetics and the use of mutants should make it possible to understand how ozone exposure puts a plant at risk and how we can aid the plant in its attempt to protect and repair itself in the face of that risk.

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The Use of Plants as Bioindicators of Ozone[1]

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Abstract

*A variety of vascular plant species exhibit typical foliar injury symptoms when exposed to ambient ozone, making them useful as bioindicators of relative air quality for a particular location or region. They are quite useful in areas where mechanical ozone monitors are not available. Bioindicators are often introduced plant species known as sentinels. They are known to be sensitive to ozone and will respond rapidly if they are given special care to ensure ozone uptake and injury. Sentinels are usually genetically-uniform, rapid growing herbaceous annuals. Their rapid and well-characterized response to ozone exposure has made them quite useful. Bel-W3 tobacco (*Nicotiana tabacum *L.) is a sentinel plant bioindicator for ozone that is used all over the world. Detector bioindicators are plant species that are found growing naturally in an area and known to be sensitive to ozone only when conditions are appropriate for ozone uptake and plant injury. Detectors are often slow-growing, determinate perennial plants, shrubs, or trees that respond slowly to ozone, with symptoms occurring quite late in the growing season. Populations of detectors are not genetically uniform and only part of a population may show ozone injury symptoms. Black cherry (*Prunus serotina *L.) is a common detector bioindicator for ozone in North America. A comparison of surveys of sentinel and detector bioindicators in the same area often show different results. From an ecological perspective, visible injury on a detector bioindicator is more significant than visible injury on a sentinel bioindicator. When using plants as bioindicators, careful consideration needs to be given to the nature, requirements, and utility of sentinels and detectors in relation to the relevance and utility of the results obtained.*

Introduction

For more than 50 years, certain plant species have been used as bioindicators because they are sensitive to ozone under ambient conditions (Middleton and others 1950, Noble and Wright 1958). Sensitive individuals exhibit typical foliar injury symptoms when exposed to ambient ozone under conditions appropriate for ozone uptake. These symptoms are considered to be diagnostic or typical as they have been verified in exposure/response studies under experimental conditions (Krupa and Manning 1988). These plants are considered to be reliable biological indicators or bioindicators for ambient ozone. The subjective determination of the intensity or extent of foliar injury of bioindicators is used as an index of relative air quality for ozone for a particular location or region. Ozone has become an air pollution problem in most industrialized nations, resulting in an increased interest in using bioindicator plants on a world-wide basis. According to Guderian and others (1985), injury on Bel-W3 tobacco (*Nicotiana tabacum* L.) is usually the first indication a county or region has developed an ozone problem. The history and use of plants as ozone bioindicators has been extensively reviewed elsewhere (Arndt and others 1987, Burton 1986, deBauer 1972, Feder and Manning 1979, Guderian and others 1985, Heck 1966, Hernandez and deBauer 1989, Manning 1991, Manning and Feder 1980, Posthumus 1982, Stuebing and Jager 1982, Tonneijck and Posthumus 1987, Weintstein and Laurence 1989).

This paper discusses the use of introduced species or sentinels and naturally occurring plants or detectors as bioindicators of ozone exposure.

Sentinels and Detectors

From an ecological perspective, Spellerberg (1991) recognizes two types of plant bioindicators useful in studies designed to detect gaseous air pollutants (*table 1*). Sentinels are non-indigenous plant species, consisting of well-defined selections,

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cultivars or clones, that exhibit diagnostic reliable foliar symptoms when exposed to ambient ozone. Sentinels are grown in charcoal-filtered air and then introduced into areas, usually for short time periods. They are quite useful in identifying areas where ozone concentrations and exposures are suspected to be high enough and of a long enough duration to cause foliar injury (Ashmore and others 1980, Bytnerowicz and others 1993, deBauer 1972, Heck and Heagle 1970, Hernandez and deBauer 1989, Jacobson and Feder 1974, Kelleher and Feder 1978, Lipa and Votapkova 1995, Noble and Wright 1958, Oshima 1974, Posthumus 1982). Examples of sentinel bioindicators for ozone include Bel-W3 tobacco (Heck and others 1964, Heck and Heagle 1970, Heggestad 1991) and morning glory (*Ipomoea purpurea*) (Manning 1977, Nouchi and Aoki 1979). Detector plants are indigenous to an area that may exhibit typical foliar injury symptoms to ozone exposure *in situ*. They are useful in assessing the long-term or cumulative effects of ozone. Examples include the sensitive individuals in populations of black cherry (*Prunus serotina*) (Davis and others 1982), Ponderosa (*Pinus ponderosa*) and Jeffrey (*P. Jeffreyi*) pines (Stolte and others 1992), and milkweed (*Asclepias syriaca*) (Duchelle and Skelly 1981).

Table 1 *— Types of indicator plant species used to assess relative air quality for ozone (Spellerberg 1991).*

Sentinels	Well-defined plants known to be sensitive to ozone are introduced into an area to serve as early warning devices or checks on the efficiency of abatement practices. Examples—Bel-W3 tobacco (*Nicotiana tabacum*), Morning glory (*Ipomoea purpurea*)
Detectors	Plants that naturally occur in the area of interest that may exhibit typical foliar responses to ozone. Examples—Black cherry (*Prunus serotina*), Milkweed (*Asclepias syriaca*)

Sentinels are the most commonly used bioindicators of ambient ozone. They are usually well-defined, genetically-uniform, herbaceous annuals that grow rapidly. Their response time to ozone is rapid, serving as an early warning of ozone presence. Rapid response, however, may only occur in the early stages of their growth cycles, necessitating frequent re-introduction of new plants. They must be grown in charcoal-filtered air until they are old enough to move to the field. To minimize edaphic factors, they are usually grown in pots of a uniform artificial growing medium. They require water, fertilizer, shading, and often pesticide applications, on a regular basis. Protection from animals and vandals may also be required. If sentinels do not receive the special care they require, they will not respond well to ozone in a typical or relevant manner.

Detectors are native plants selected *in situ* and usually are not given any special cultural care. Usually they are determinate perennial plants, trees, or shrubs that respond slowly and often fairly late in the growing season. Only the sensitive individuals in a population of a detector bioindicator will respond to ozone and only when they experience appropriate edaphic and tropospheric conditions coupled with ozone exposures and concentrations sufficient to cause foliar injury. The distribution of ozone-sensitive genotypes (where there is little phenotypic variation) in a population of a detector bioindicator is usually not well-known. This adds uncertainty in interpreting results from detectors. When large numbers of individuals are present, confidence coefficients increase (Stolte and others 1992).

Bioindicator Plant Response to Ozone

Sentinels and detectors evaluated in the same area often show different results: sentinels may respond but detectors may not. Results from ozone plant bioindicator studies must be interpreted on the basis of a number of interacting biological, cultural, and physical factors (*table 2*). Foliar ozone injury symptoms are only an indication of previous exposure to ozone, when the concentration was high enough for a certain time period, and environmental conditions were conducive to ozone uptake and cellular injury. The visible response is a relative index of air quality for ozone during a previous exposure period. It is not possible to use this response to quantitatively assess air quality for ozone. This is because we do not yet truly understand the relationship between the occurrence and magnitude of plant response and the environmental conditions and concentrations and durations of ambient ozone present when the plant injury response was initiated. If data from a co-located or nearby mechanical ozone monitor is available, however, periods when monitored ozone concentrations have potential biological significance or meaning can be indicated, as reflected by frequency, duration, and intensity of plant responses.

Table 2 — *Biological, cultural, and physical factors affecting responses to ozone (Manning 1991).*

Biological and Cultural		
Factors		***Responses***
Biological		
	Genetic diversity	Homogeneous plants give uniform responses; species, clones, cultivars, and provenances react differently to O_3
	Stage of plant development	Plant developmental stages and leaf age affect responses to O_3
Cultural		
	General cultural practices	Optimal or usual practices result in "typical" responses to O_3
Growth media		Soilless media allow uniformity and reproducibility, but are less relevant than natural soils.
Nutrients		Optimal levels usually result in optimal injury from O_3.
Pesticides		Variable effects, ranging from none to protection, from injury to reduced tolerance to joint effects with O_3.
Physical		
Factors		***Responses***
Air movement		Must be sufficient to alter boundary layer resistance to allow O_3 uptake.
Light		
	Intensity, photoperiod, and quality	Ideal value varies for each plant; less or more than ideal value for each plant reduces O_3 sensitivity
Temperature		Injury increases in a range from 3 to 30 °C.
Water		
	Relative humidity	Controls stomatal opening exchange; uptake of O_3 and injury should increase as relative humidity increases.
Soil moisture tolerance		Water stress increases O_3 due to stomatal closure

Previous research has determined some well-defined bioindicators for ambient ozone (*table 3*). Each of these bioindicators will exhibit typical ozone injury symptoms (*table 4*). Tobacco cultivar Bel-W3 is the best-defined, best-described, and most widely-used bioindicator for ambient ozone (Heggestad 1991). It has become the standard sentinel bioindicator for ozone and has been used all over the world to detect ambient ozone in areas ranging from local sites to whole countries and continents. Heck and Heagle (1970) used Bel-W3 tobacco to determine incidence of phytotoxic concentrations of ozone around Cincinnati, Ohio. Bel-W3 tobacco plants were used by Manning to demonstrate ozone in Warsaw, Krakow, and forested areas in southern Poland (Bytnerowicz and others 1993). Jacobson and Feder (1974) reported the results of a network of Bel-W3 tobacco plantings in the northeastern United States. Long-range transport of ozone from metropolitan New York to Nantucket Island was demonstrated by response of Bel-W3 tobacco plants on Nantucket (Kelleher and Feder 1978). Ashmore and others (1980) mapped phytotoxic ozone in the British Isles by using networks of Bel-W3 tobacco plants. Schoolchildren in many European countries use Bel-W3 tobacco to assess relative air quality for ozone (Lipa and Votapkova 1995).

Table 3 *— Selected bioindicators of ozone.*[1]

Plant	Latin Name	References
Sentinels		
Blue grass	*Poa annua*	Noble and Wright 1958
Bean	*Phaseolus vulgaris*	Oshima 1974, Sanders and others 1992
Clover	*Trifolium repens*	Heagle and others 1992
	T. subterraneum	Sanders and others 1992
Morning glory	*Ipomoea purpurea*	Manning 1977, Nouchi and Aoki 1979
Spinach	*Spinacea oleraceae*	Posthumus 1982
Tobacco	*Nicotiana tabacum*	Heck and Heagle 1970,
	Bel-W3 (sensitive)	Heaggestad 1991
	Bel-B (tolerant)	
Detectors		
Blackberry	*Rubus* spp.	Chappelka and others 1986, Manning 1991
Black cherry	*Prunus serotina*	Chappelka and others 1992, Davis and others 1982
Green ash	*Fraxinus pennsylvatica*	Davis and Wilhour 1976
Milkweed	*Asdepias sytiaca*	Duchelle and Skelly 1981
Quaking aspen	*Populas tremuloides*	Karnosky 1976, Keller 1988
Sassafras	*Sassafras albidum*	Chappelka 1992
Tulip poplar	*Liriodendron tulipifera*	Davis and Chappelka 1986, 1992; Davis and Wilhour 1976
White ash	*Fraxinus americana*	Chappelka 1992, Davis and Wilhour 1976

[1] Not all varieties or individuals in a species will be ozone-sensitive.

Several plant species have cultivars or clones that differ in their response to ambient ozone. The best defined incidence of this are the tobacco cultivars Bel-W3 (ozone-sensitive) and Bel-B (ozone-tolerant). Bel-W3 tobacco responds to ambient ozone at lower concentrations than does Bel-B. When grown together in an area, their comparative responses to ozone can provide a better description of relative air quality for ozone. If both cultivars do not respond, then the air is relatively clean and ozone concentrations are probably low. If Bel-W3 responds, but Bel-B does not, then ozone concentrations are intermediate. If both cultivars respond, then ozone concentrations are higher (Manning and Feder 1980).

Detectors are being used to assess relative long-term air quality in forested and wilderness areas. Study areas are selected and certain species are examined on an annual basis for incidence and severity of foliar ozone injury. In wilderness areas of New Hampshire and Vermont, Manning and others (1991) have surveyed black cherry, white ash, and milkweed. Chappelka and others (1986, 1992) have used black cherry, sassafras, white ash, yellow poplar, and blackberry in the southeastern United States. The USDA Forest Service's Forest Health Monitoring Program uses detector bioindicators in its forest surveys in the U.S. (Conkling and Byers 1993). Detector bioindicators are used in forested regions of the Carpathian Mountains in Poland, Ukraine, Czech Republic, Slovakia, and Rumania (Manning and others, unpublished paper).

EDU (ethylenediurea) is a chemical known to protect plants from ozone injury (Carnahan and others 1978). It can be used to verify the response of an ozone bioindicator in the field. This is especially useful if an ozone monitor is not readily available or there are questions about the nature of the response of the bioindicator. This approach was used in Poland (Bytnerowicz and others 1993). Bel-W3 tobacco plants were outplanted in a number of locations. Half of the plants were treated with EDU before outplanting. The treated plants did not develop any symptoms of foliar injury while the non-treated plants developed varying degrees of typical ozone injury symptoms, verifying that ozone was the cause.

Table 4 *— Common symptoms of foliar ozone injury (Krupa and Manning 1988).*

Acute injury	
Flecking	Small necrotic areas due to death of palisade cells, metallic or brown, fading to grey or white.
Stippling	Tiny punctate spots where a few palisade cells are dead or injured, may be white, black-red, or red-purple.
Chronic injury	
Pigmentation (Bronzing)	Leaves turn red-brown to brown as phenolic tan pigments accumulate.
Chlorosis	May result from pigmentation or may occur alone as chlorophyll breaks down.
Premature senescence	Early loss of leaves or fruit.

Methodology for Assessing Bioindicator Plants

Great care must be taken in assessing the response of bioindicator plants to ambient ozone. In most cases, assessment of bioindicators involves the subjective determination of the intensity or extent of acute ozone injury symptoms (*table 4*). Leaf injury evaluations should be made at regular intervals, often weekly, and by the same person. The use of a set of color photographs illustrating degrees or categories of severity of injury can help to standardize the evaluation process (Heck 1966; Heck and others 1966, 1969; Oshima 1976). If plants are left in the field for more than 1 week, then new injury on both older and new leaves needs to be estimated and recorded.

The observer should look at each leaf and visually integrate the injured areas of each leaf and then determine the percentage of the total leaf area that has been injured. Depending on the type of plant, extent of injury, and purpose of the study, the evaluation system can be quite simple and uncomplicated (*table 5*). With only six indices, this is a simple system to use. Where more precise information is required, an expanded system is more appropriate (*table 6*).

Table 5 *— System for evaluation of ozone injury for bean (*Phaseolus vulgaris *L.) (Manning and Feder 1980).*

	Rating Systems	
Injury rating	**Injury severity index**	**Percent leaf injury**
None	0	0
Slight	1	1-25
Moderate	2	26-50
Moderate-severe	3	51-75
Severe	4	76-99
Complete	5	100

Table 6 *— The Horsfall-Barratt Scale for assessing foliar injury (Horsfall and Barratt 1945, Horsfall and Cowling 1978).*

(Percent severity or class incidence)	
0	0
1	0-3
2	3-6
3	6-12
4	12-25
5	25-50
6	50-75
7	75-88
8	88-94
9	94-97
10	97-100
11	100

Data can be depicted graphically. Plotting weekly injury scores, or number of leaves injured, on a weekly basis against elapsed time gives a cumulative injury curve over time. If data are available from a nearby mechanical ozone monitor, ozone injury can be plotted against cumulative ozone or related to episodes in which thresholds are exceeded (Manning and Feder 1980).

As an alternative to using subjective determination of the intensity or extent of foliar injury as a measure of ozone effects, other investigators have developed bioindicator systems that use shoot biomass response from sequential harvests as a measure of ozone effects. Oshima and others (1976) developed a standardized pot culture system with alfalfa (*Medicago sativa* L.) along an ozone gradient in southern California. Heagle and others (1994) have developed an ozone bioindicator system by using sensitive (NCS) and resistant (NCR) clones of ladino clover (*Trifolium repens* L.) NCS and NCR plants are grown in standardized pot culture. Shoot biomass is removed at 28-day intervals and dry weights are obtained. Ozone impact is determined by obtaining dry weights of harvested shoot biomass and calculating the ratios of NCS to NCR. A ratio of less than one indicates that ambient ozone has had an adverse effect on foliar biomass of NCS.

Summary

Bioindicator plants for ozone can be extremely useful in assessing relative air quality, especially in areas where ozone monitors are not available. If they are not used properly, however, poor quality or misleading results will be obtained. Common factors that produce poor results are poor plant culture for sentinels, failure to adhere to regular evaluation schedules, careless and inaccurate evaluation of symptoms, and the use of more than one person to evaluate plant responses.

Care should be taken in interpreting responses of sentinel bioindicators. As they are well-watered and fertilized, they respond much more frequently than do detectors. Their response indicates the occurrence of periods of ozone exposure when detectors might respond if they were also growing under conditions of sufficient soil moisture and fertility. From an ecological perspective, response of detector bioindicators to ozone is more important than a response by a sentinel bioindicator. A response by a sentinel indicates what could happen under ideal conditions, while a response by a detector indicates what did happen under more realistic conditions.

There is an unfortunate tendency to confuse a bioindicator with a biomonitor. A bioindicator indicates that the system has been affected. A biomonitor should also indicate how much of the causal factor was present and caused the observed effect. Currently, bioindicators of ambient ozone cannot be used as biomonitors. It is not possible to make quantitative inferences about air quality for ozone on the basis of plant symptom expression alone.

With a clear understanding of the strengths and weaknesses of the use of plants as bioindicators of ozone exposure, they can be quite useful in assessing relative air quality, especially in programs like Forest Health Monitoring, in remote areas, and in emerging nations, where technology is lacking.

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State of Polish Mountain Forests: Past, Present, and Future[1]

Krystyna Grodzinska,[2] Grazyna Szarek-Lukaszewska[2]

Abstract

*Mountains occupy only 3 percent of Poland. They are the northern part of the European arc of the Carpathian and Sudety Mountains, extending about 700 km along the southern Polish border. They are of medium height (about 1,500 m., maximum 2,600 m. a.s.l.), and diversified in terms of climate, geology, soils, vegetation, and anthropogenic impacts. The forest vegetation of the Sudety and Carpathian Mountains forms three elevational zones. Forests occur in the foothills (as high as 600 m. a.s.l.), the lower mountain forest zone (as high as 1,250 m. a.s.l.) and the upper montane forest zone (as high as 1,500 m. a.s.l.). The original lower mountain forests consist mainly of the fir (*Abies alba *Mill.) and beech (*Fagus sylvatica *L.), while upper montane forests consist of spruce (*Picea abies *[L.] Karst.). Centuries of economic activity have changed the species composition of mountain forests. The share of the spruce has increased considerably, and the percentage of the fir and beech has decreased significantly. In the Sudety and western part of the Carpathian Mountains prevalence of spruce is the highest (83 percent), while in the eastern part of the Carpathians, it does not exceed 10 percent. The percentage of beech in the Sudety and western part of Carpathian forests is below 20 percent, but about 40 percent in the eastern part of the Carpathians. Severe weather conditions, frequently poor habitats and improper management and considerable air pollution occurring in the past 50 years, followed by infestations of primary and secondary insects and spread of parasite fungi, have led to considerable destruction of mountain forests. According to forecasts, the area of totally destroyed and severely damaged mountain forests will increase considerably through the year 2010. Deterioration of forest health will proceed from west to east. Protection of forest health against deterioration requires reduction of industrial emissions, changing existing forests into less sensitive habitats more compatible with their carrying capacity, and recultivation of contaminated forests.*

Introduction

Generally Poland is a lowland country. Mountains occupy only 3 percent of the country. Even though the area occupied by mountains is small, their landscape makes them the most beautiful areas of Poland, and they are valuable in terms of wildlife, with numerous endemic and rare plant and animal species, and specific biocenoses (Denisiuk 1995, Szafer and Zarzycki 1972, Zarzycki and others 1991).

Humans have affected the wildlife of the Polish mountains for centuries. Forms of human intervention have changed over the years, evolving from direct activity (cutting forests, raking duff, grazing) to air pollution, a much more dangerous indirect impact that is difficult to control. Human impacts, exacerbated by specific climatic, orographic and soil conditions, have seriously worsened the condition of mountain ecosystems, including forests, in at least part of the ranges.

General Characteristics of Polish Mountains

Polish mountains form the northern part of the European arc of the Sudety and Carpathian Mountains (*fig. 1*) extending about 700 km along the southern boundary of Poland (*fig. 2*). They are of medium height, with average elevation not exceeding 1500 m. a.s.l. (*fig. 3*). They are diversified in terms of climate, geology, soil and vegetation, and also vary with anthropogenic impacts (Fabiszewski and Jenik 1994, Paschalis 1995, Starkel 1991, Szafer and Zarzycki 1972, Zarzycki and others 1991).

The Carpathian climate is more continental (colder and drier) than that in the Sudety Mountains. Continentalism increases along the west-east transect. The Sudety Mountains are composed mainly of granite, gneiss and basalt, and the Carpathians are comprised of sandstone and shales (Carpathian flysch) with small

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areas occupied by granite and limestone rock (Starkel 1991). Thus, the Sudety have a poor substratum and strongly acidic soils, while the Carpathians have less acidic, more nutrient-rich substratum and soil.

The vegetation of the Sudety and Carpathians is diverse by elevation (*fig. 4*). The foothill zone, as high as 600-700 m. a.s.l., is occupied by deciduous and mixed forests (deciduous and coniferous). The lower montane zone (as high as 1,250 m. a.s.l.) is represented by mixed forests, usually beech and fir, and the upper montane belt (as high as 1,500 m. a.s.l.) consists of spruce. Above the montane forest zones in the higher mountain ranges, there is a dwarf mountain pine zone (up to 1,750 m. a.s.l.), and alpine (up to 2,300 m. a.s.l.) and subalpine zones (>2,300 m. a.s.l.) (*fig. 4*).

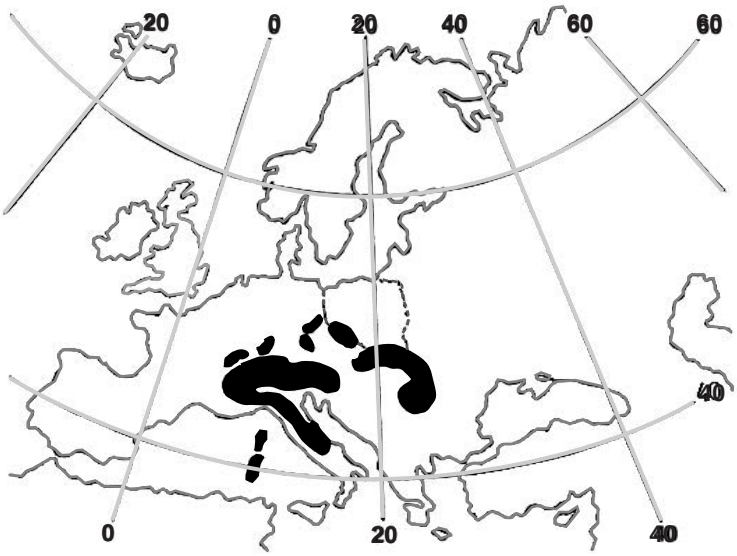


Figure 1 — Central European Mountains Province (Szafer and Zarzycki 1972).

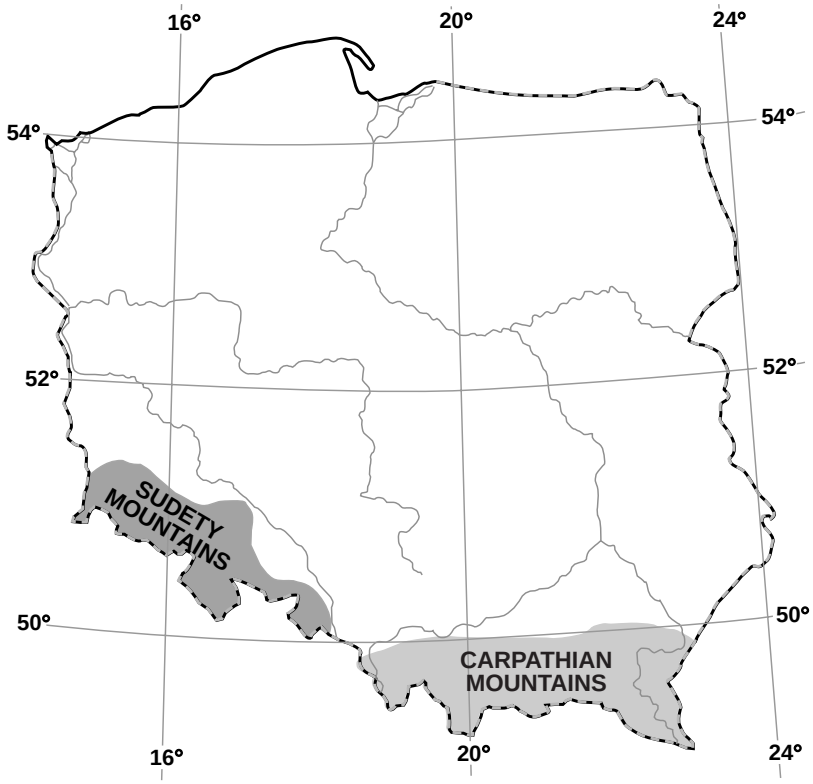


Figure 2 — Polish Mountains: the Sudety and Carpathians.

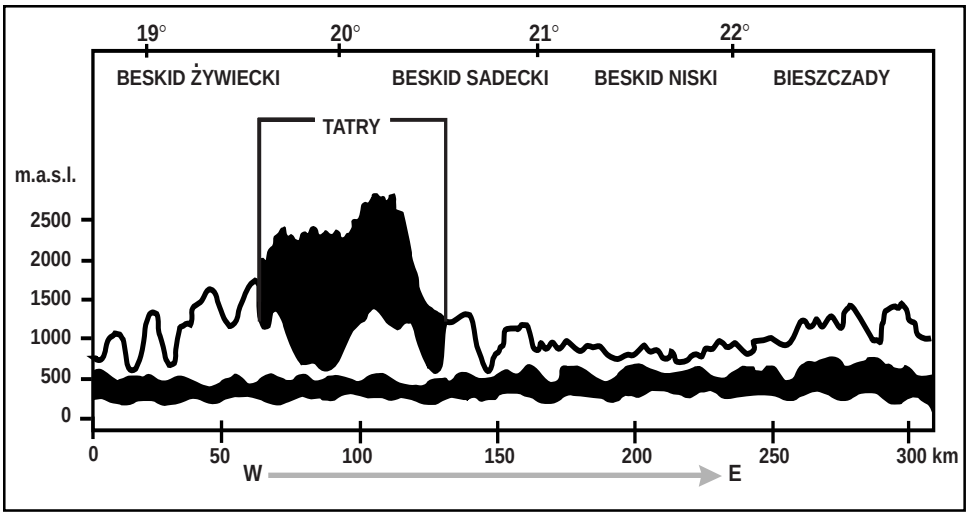


Figure 3 — The Carpathian Mountains (Warszynska 1995).

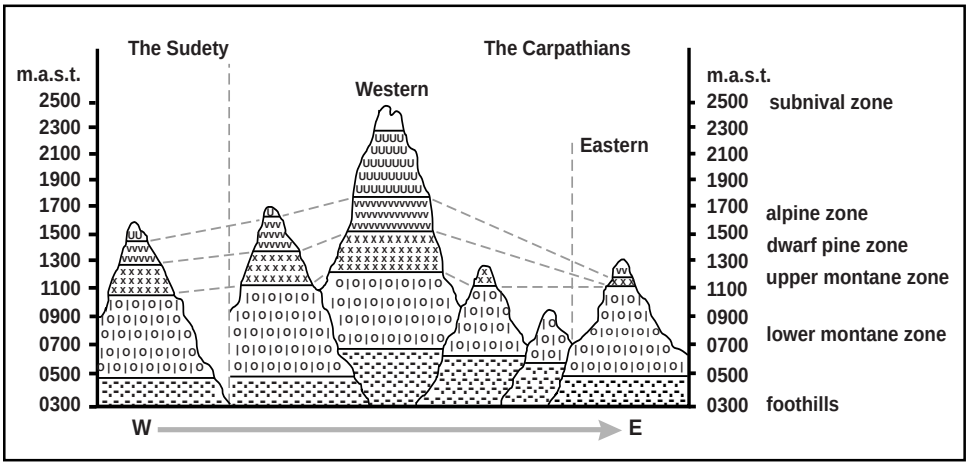


Figure 4 — Vegetation zones in Polish mountains (Denisiuk 1995, Fabiszewski and Janik 1994).

Polish Mountains Forests

The primeval lower montane forests are comprised mainly of the fir (*Abies alba* Mill.) and beech (*Fagus sylvatica* L.), while the upper montane forests have spruce (*Picea abies* [L.] Karst). The first two species are abundant in the Carpathians, and the third species is common in the Sudety. Generally, poor acidophilic forest communities occur in the Sudety, while the Carpathians had more eutrophic communities (Matuszkiewicz 1984).

The current species composition of the forests significantly differs from the primeval one because of centuries-long, often improper management. Currently, spruce stands occupy considerably larger mountain areas, and often the seed source is from non-native plantations. The percentage of the fir and beech in forests in Polish mountains has significantly decreased. In Sudety the current percentage of the spruce in forests is more than 83 percent (Capecki 1989) (*table 1*). In the Carpathians the average percentage of this species does not exceed 22 percent, and beech and fir are about 25 percent each (Fabijanowski and Jaworski 1995). The percentages of these three forest-forming species are different in various parts of the Carpathian arc. In the western part (Silesian Beskid Mountains, Zywiecki Beskid Mountains) the spruce dominates (> 70 percent), while in the eastern part (Beskid Niski and Bieszczady Mountains beech (27 and 41 percent) and fir (31 and 20 percent) dominate (Fabijanowski and Jaworski 1995) (*table 1*).

The average age of Carpathian forest trees is about 50 years, and their volume per hectare is as high as 200 m^3, with average annual growth 2.6 m^3/ha/year. In selected regions of the Carpathians there are considerably older tree stands (> 80 years), with

higher volume and valuable native populations (e.g., Beskid spruce) and ecotypes (Istebnianski spruce), but they occupy small areas.

Mountain forests, both in the Sudety and Carpathians, are affected by severe climatic conditions, including strong winds, low temperatures, frequent mists, and high snowfall. Tree stands, in habitats unsuitable for them and thus weakened, are more easily broken by wind and snow.

In the last 50 years, the severe climatic conditions have been accompanied by gaseous (SO_2, NO_x) and particulate (heavy metals) air pollutants. The Sudety, located closest to large industrial centers in the Czech Republic, Germany, and Poland, are among the most threatened and damaged Polish mountains. In the Sudety the average annual SO_2 concentration reaches 25 µg m^{-3}, fluctuating during the year between 16 and 45 µg m^{-3}; average annual NO_x reaches 32 µg m^{-3} (Godzik and Szdzuj 1994). The average 24-h ozone concentrations reaches in the spring month of May 100 µg m^{-3} (Zwozdziak and others 1994). Annual sulfur deposition is estimated at about 30 kg/ha^{-1}, and nitrogen deposition at 8-9 kg ha^{-1}. Atmospheric precipitation is acidic, below pH 4 (Stachurski and others 1994, Zwozdziak and others 1995). Critical ambient concentrations of sulfur and ammonia compounds are considerably exceeded in the Sudety forests. In the westernmost parts of the Carpathians (Silesian Beskid Mountains), air pollutant concentrations and deposition of these pollutants are nearly the same as those in the Sudety (Godzik and Szdzuj 1994). Air pollution is considerably lower eastwards (Beskid Niski and Bieszczady Mountains) because no large emission sources are found in these areas (Dmuchowski and Wawrzyniak 1994) (*figs. 5, 6*).

The decrease in the air pollution gradient from west to east (from the Sudety Mountains to the Carpathians) is confirmed by sensitive plant indicators (Dmuchowski and Wawrzyniak 1994, Zolnierz and others 1995). Concentrations of and toxic elements (cadmium and aluminum) in the needles of the spruce and pine (*Pinus silvestris* L.) are considerably higher in the Sudety than in the Carpathians (*figs. 7, 8*).

Mountain forests that have altered species composition, improperly managed and affected by industrial emissions are attacked by primary insect pests, such as *Zeiraphera griseana*, *Cephalcia falleni*, and *Lymantria monacha*, and then by secondary pests, such as *Ips typographus* and *Pityogenes chalcographus* as well as by parasite fungi (among others *Armillaria obscura* and *Heterobasidion annosum*). Insects of both groups inflict the greatest damage in the Sudety and western Carpathians (Capecki 1989, Godzik 1995, National Inspectorate for Environmental Protection 1993).

Table 1 *— Main tree species in forest stands in the Sudety and Carpathian Mountains.*

Forest Stand	*Picea abies*	*Abies alba*	*Fagus sylvatica*
	pct		
Sudety	83.4	-	5.0
Carpathians	21.7	25.0	25.3
Beskid Slaski	73.6	3.5	17.2
Beskid Zywiecki	74.8	8.8	12.1
Beskid Makowski, Wyspowy	12.9	43.9	17.2
Gorce, Beskid Sadecki	36.9	28.1	25.9
Tatry	77.6	2.8	2.2
Beskid Niski	3.7	31.0	27.0
Bieszczady	9.3	20.0	40.9

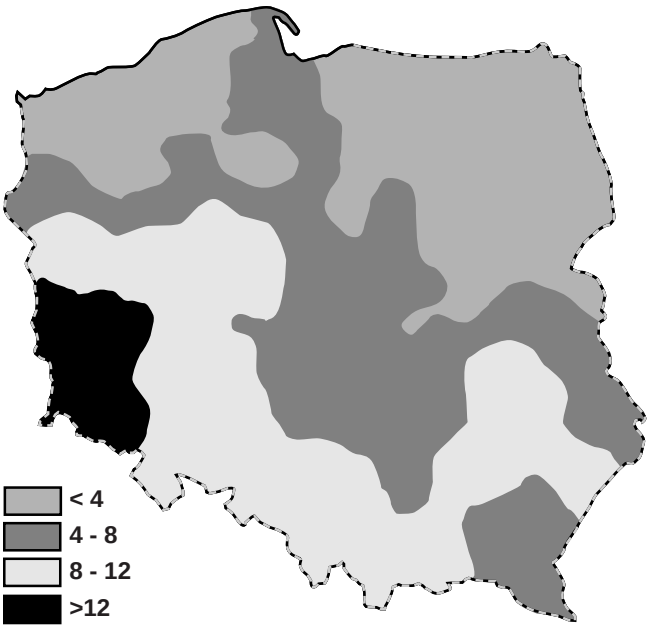


Figure 5 — SO_2 deposition index, average value for 1989-1991, summer period (mg/m²/24 h) (Dmuchowski and Wawrzoniak 1994).

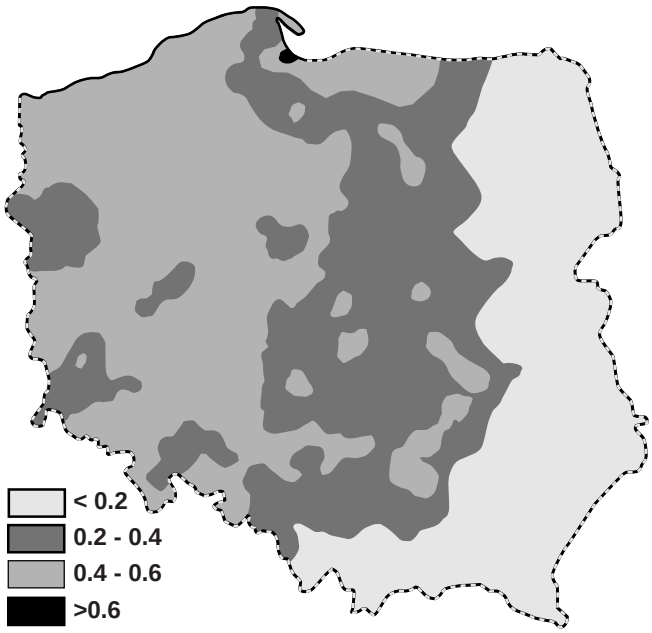


Figure 6 — NO_x deposition index, average value for 1989-1991, summer period (mg/m²/24 h) (Dmuchowski and Wawrzoniak 1994).

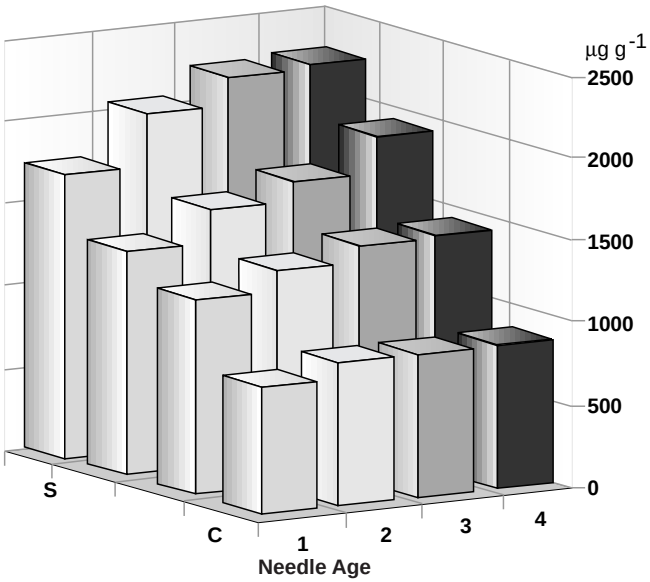


Figure 7 — Total sulfur concentration in spruce needles (Zolnierz and others 1995).

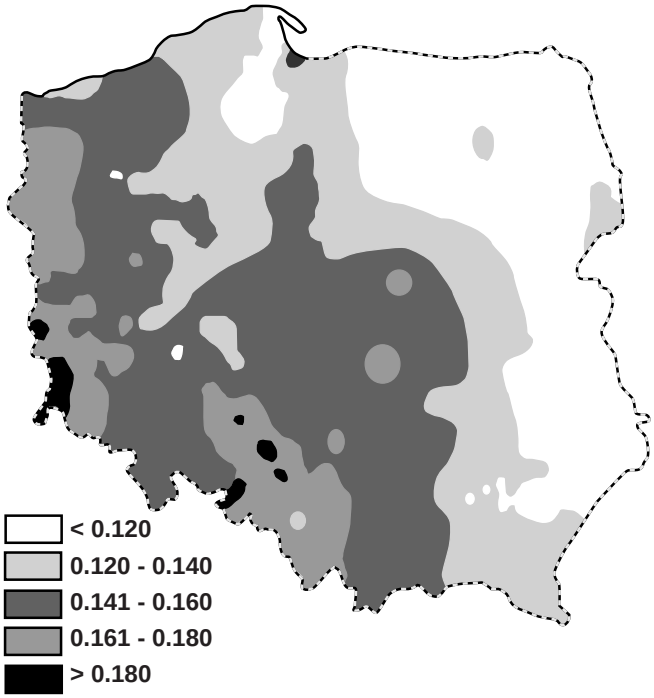


Figure 8 — Total sulfur concentration (percent in d.wt) in Scots pine needles (previous year growth) (Dmuchowski and Wawrzoniak 1994).

The Present Health of Mountain Forests in Poland

According to the general map of forest health by Paschalis (1995), based on data from the Institute of Forest Research, there were no healthy forests in Poland in 1995. The Sudety forests and westernmost Carpathian ranges are in a state of total disaster or very damaged (*fig. 9*). The forests in the central and eastern Carpathians have been classified as strongly damaged. Compared to 1990, the state of the forests has deteriorated considerably (Paschalis 1995).

Factors involved in damage to the mountain forests, which is highest in the western part and lowest in the eastern part of the Sudety-Carpathian arc include climatic, geological, and soil conditions; the intensity of anthropogenic impact; the level of air pollution from industrial emissions; the occurrence of insect pests and parasite fungi (*fig. 10*).

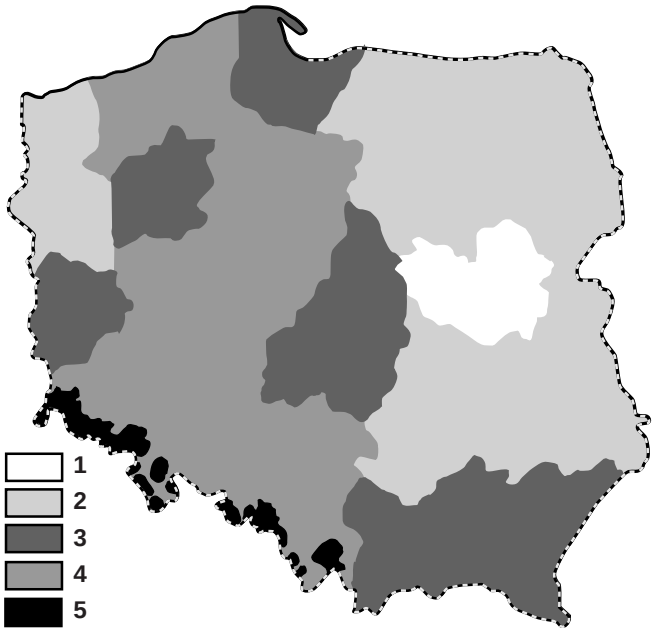
In addition to the horizontal east-west gradient of contamination of the mountain forests, a vertical gradient has been found. Upper montane forests, higher than 1,200 m. a.s.l., are exposed to greater amounts of wet deposited pollutants than lower ones (< 1,000 m a.s.l.). These pollutants are brought there together with orographic mists and clouds that occur more frequently at higher elevations than lower elevations (Zwozdziak and others 1994).

Locally, beautiful stands of forests can be still found in Poland. They are usually under protection from economic activity in national parks. The country's seven mountain national parks are located at regular intervals along the Sudeten-Carpathian arc. Forests occupy between 67 percent and 91 percent of their areas.

Future of Mountain Forests in Poland

According to forecasts of the Institute of Forest Research based on a large-area inventory of damage to tree stands, the area of completely and very damaged forests will increase considerably during the next 15 years (through the year 2010) (Paschalis 1995). Deterioration of forest health will proceed from west to east (*fig. 11*).

To counteract the deterioration of mountain forest health, managers and scientists should reduce both local and transboundary industrial emissions; change existing forests into less sensitive genotypes, more compatible with local habitats; and recultivate contaminated forests.



Fgure 9 — Forest damage in Poland. Damage degree:
1 = low,
2 = moderate,
3 = heavy,
4 = very heavy,
5 = deforestation (Paschalis 1995).

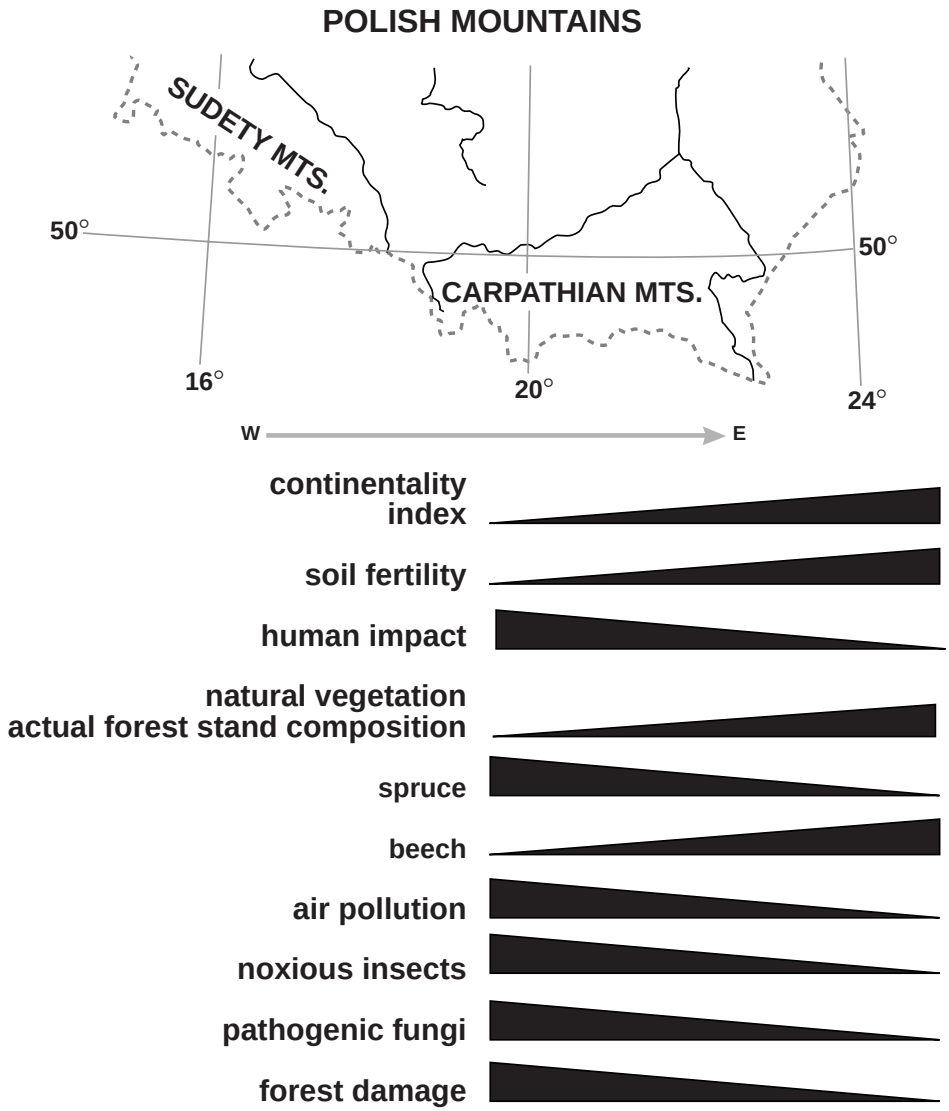


Figure 10 — Factors involving various degree of damage to mountain forests in Poland.

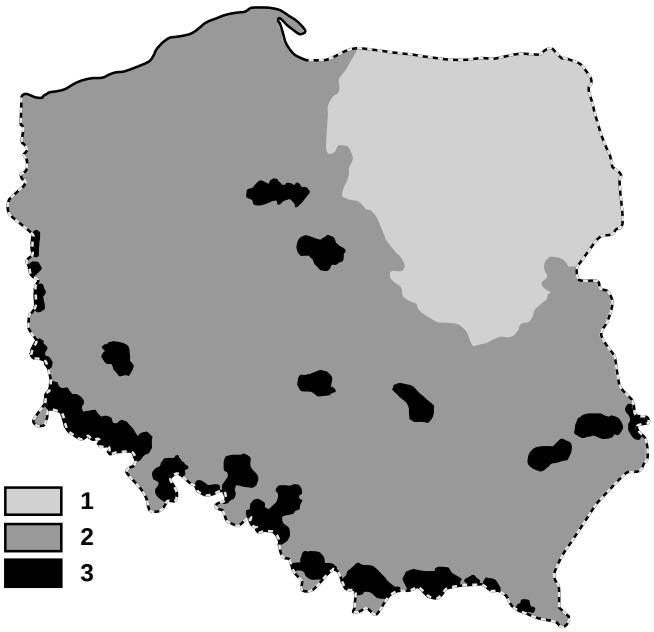


Figure 11 — Forest damage in Poland, prognosis 2010. Damage degree:
1 = moderate,
2 = heavy and very heavy,
3 =deforestation (Paschalis 1995).

The future of mountain forests is a primary concern of Polish ecologists. We are reminded of the consequences of total forest destruction on the environment by an admonition more than a dozen years ago by Keating who stated, "The fate of mountain ecosystems affects half the world's people" (The Earth Summit Agenda for Change 1983).

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Ozone Injury Responses of Ponderosa and Jeffrey Pine in the Sierra Nevada and San Bernardino Mountains in California[1]

Paul Miller,[2] Raleigh Guthrey,[2] Susan Schilling,[2] John Carroll[3]

Abstract

*Ozone injury was monitored on foliage of ponderosa (*Pinus ponderosa *Dougl. ex Laws.) and Jeffrey (*Pinus jeffreyi *Grev. & Balf.) pines at 11 locations in the Sierra Nevada and 1 site in the San Bernardino Mountains of southern California. Ozone injury on all age cohorts of needles on about 1,600 trees was surveyed annually from 1991 to 1994. A new method for describing ozone injury to whole tree crowns, the ozone injury index (OII), was field tested and improved. The OII ranged from 0 (no injury) to 100 (the maximum possible injury). The OII multi-year average was only 5 at Lassen Volcanic Park in rural northern California and gradually increased in a southward direction west of the Sierra Nevada crest to moderate amounts (28-41) in the Sequoia National Forest and Sequoia National Park. The OII multi-year average measured at the San Bernardino Mountain site was 46. An assessment of annual changes at the 12 individual sites indicated both increases and decreases in OII from 1991 to 1994. The two most responsive indicators of the annual increments of accumulated injury, contributing 40 percent each, were chlorotic mottle and needle fascicle retention (within remaining needle whorls). Data for these components were tested with quadratic and Weibull functions against several expressions of ozone exposure (including ozone exposure indices Sum 0, Sum 60, W126, and number of hours exceeding 80 ppb during the summer exposure periods). Sum 0 was a suitable exposure index having Weibull correlation coefficients of 0.57 with percent chlorotic mottle and 0.74 with percent fascicle retention. These results provide estimates of ozone injury responses across a range of annual accumulated ozone exposures and environmental conditions during four summers.*

Introduction

In 1990-91 two companion projects were begun at analogous locations in the Sierra Nevada of California: the Sierra Cooperative Ozone Impact Assessment Study (SCOIAS) and the Forest Ozone Response Study (FOREST). Funded by a contract with the California State Air Resources Board, the principal activity of SCOIAS was to monitor ozone and meteorological variables at six Sierra Nevada sites managed by the University of California, Davis. FOREST was developed as a companion project to SCOIAS through an agreement of intent between the USDA Forest Service, Pacific Southwest Region , and the California State Air Resources Board.

The FOREST agreement led to the establishment of forest vegetation plots within 3 miles distance and 500 ft elevation of SCOIAS monitoring stations for the purpose of annual assessments of ozone injury to ponderosa and Jeffrey pine populations. The Air Resources Management Staff of the USDA Forest Service's Pacific Southwest Region provided training of field crews and actual data gathering activities in six National Forests. Other participants included Yosemite, Sequoia-Kings Canyon, and Lassen Volcanic National Parks, and the Forest Service's Pacific Southwest Research Station. The participants used accepted procedures for instrument calibration and maintenance for ozone measurements and tree injury assessment (Miller and others 1996b). The Pacific Southwest Research Station provided data management, data analysis and reporting services for FOREST (Guthrey and others 1993, 1994).

This report describes the use of the ozone injury index (OII) to monitor ozone injury to ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) and Jeffrey pine (*Pinus jeffreyi* Grev. and Balf.) in the Sierra Nevada and San Bernardino Mountains in 1991-1994; and it provides results of tests that analyzed the relationship of annual accumulated ozone exposure expressed as Sum 0, Sum 60 and W126 to the intensity of chlorotic mottle and the retention of needle fascicles in all remaining needle whorls.

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[1] An abbreviated version of this paper was presented at the International Symposium: Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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Methods

Ozone monitoring stations established as part of SCOIAS were located at six Sierra Nevada sites including National Forests, a State Park, and private land (Van Ooy and Carroll 1995). Ozone data were also available from the National Park Service at Lassen Volcanic, Yosemite, and Sequoia-Kings Canyon National Parks and from the Forest Service at Barton Flats in the San Bernardino Mountains. At each site, three forest plots with at least 40-50 tagged ponderosa or Jeffrey pines were established within 5 km and 150 m elevation of the monitoring stations (*fig. 1*). Trees were sampled annually by pruning five branches each from about 1,700 trees distributed in 36 field plots. The ozone injury index (OII) was computed for each tree based on four weighted variables: number of annual needle whorls retained (40 percent), amount of chlorotic mottle (the ozone injury symptom) on needles of each whorl (40 percent), length of needles in each whorl retained (10 percent), and percent live crown (10 percent). The OII ranged from 0 (least severe injury) to 100 (most severe injury) (Miller and others 1996b).

Two of the attributes included in the computation of the OII, amount of chlorotic mottle (percent of the total needle surface) on needles in each whorl and the percent of the original complement of needle fascicles retained in each whorl, were considered the best indicators of the annual incremental increases of ozone injury to ponderosa and Jeffrey pine foliage. A schedule was established for computation of accumulated ozone exposure for each age cohort of needles

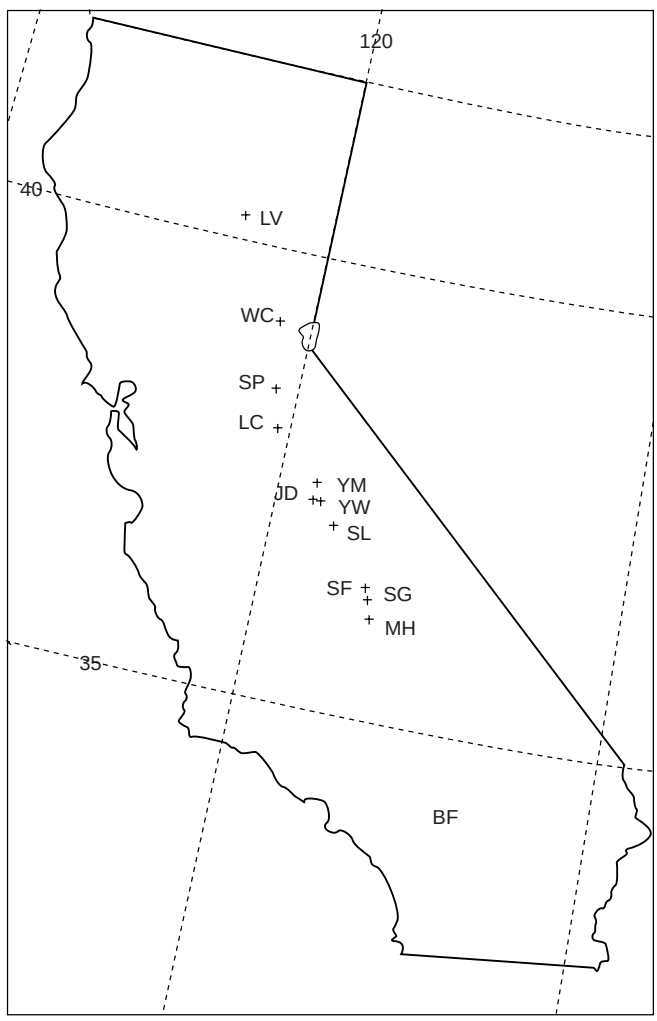

Figure 1 — Map locations of ozone monitoring sites and associated plots for monitoring ozone injury to ponderosa or Jeffrey pines.
LV = Lassen Volcanic,
WC = White Cloud,
SP = Sly Park,
LC = 5 mile Learning Center,
JD = Jerseydale,
YM = Yosemite Mather,
YW = Yosemite Wawona,
SL = Shaver Lake,
SF = Sequoia Giant Forest,
SG = Sequoia Grant Grove,
MH = Mountain Home,
BF = Barton Flats.

(*table 1*). Summer-season exposure was evaluated for each cohort, from August 1 to the injury evaluation date. August 1 was determined as the estimated date when current year needles had grown to maximum length and had measurable stomatal conductance (Temple and Miller, [this volume]). Since the SCOIAS sites were established in 1991, a full summer season of ozone data was available for only a few sites in 1991. A record of dates of injury evaluation was available for trees at each set of three plots near each monitoring station. Evaluation of all three plots ranged from 1 to 2 weeks. The beginning of the evaluation period was selected as the termination of the exposure period that had caused observed injury in that season. The exposure was initially calculated from August 1 to the first day of the foliar injury evaluation in the origin year of the needle whorl. Accumulation of ozone exposure for the cohort's second year was continued after evaluation and until October 31. Accumulation was resumed from June 1 each year until the next date of foliar symptom evaluation. This procedure was carried out for each annual needle age cohort (1991-94) until the beginning of the last period of the symptom evaluation in 1994, which had only the 1994 season of ozone exposure. The ozone exposure indices which were computed for each annual or consecutive annual time interval were Sum 0, Sum 60 (ppb), W126 (Lefohn 1992), and number of hours ≥ 80 ppb. The sum of the number of days in each seasonal exposure was included as a separate (dummy) variable. This variable simulates the extreme case in which daily ozone exposure is the same every day.

Table 1 *— The schedule for computation of accumulated ozone exposure and evaluation of needle injury (E) at 12 ozone-monitoring, tree plot locations in California, 1991-94.*

Origin year of accumulated	Periods of the summer each year when ozone exposure was needle whorl before and after injury evaluation (-E)			
	1991	***1992***	***1993***	***1994***
1991	8/1-E-10/31	6/1-E-10/31	6/1-E-10/31	6/1-E-10/31
1992		8/1-E-10/31	6/1-E-10/31	6/1-E-10/31
1993			8/1-E-10/31	6/1-E-10/31
1994				8/1-E-10/31

For each annual injury data set the amount of chlorotic mottle present was recorded as 0 (0 percent), 1 (1-6 percent), 2 (7-25 percent), 3 (26-50 percent), 4 (51-75 percent), and 5 (76-100 percent). For the regressions these ranges were represented by their midpoints (e.g., 0 = 0 percent, 1 = 3.5 percent, 2 = 16 percent, 3 = 38 percent, 4 = 63 percent, and 5 = 88 percent). Similarly, the 0 (0 percent), 1 (1-33 percent), 2 (34-66 percent), and 3 (67-100 percent) recorded values for percent of needle fascicles retained in each whorl were represented as 0 = 0 percent, 1 = 17 percent, 2 = 50 percent, and 3 = 83.5 percent.

The Weibull function was selected as the most appropriate model for relating the ozone exposure indices to foliar injury measurements (Rawlings and others 1988). The Weibull function was used successfully in the analysis of the ozone exposure of annual crop plants (U.S. Environmental Protection Agency 1986). It expresses the relationship between plant injury and ozone exposure as:

$$Y = \alpha \exp[-(x/\sigma)^c]$$

in which α = response at 0 O_3 exposure; x = accumulated O_3 exposure (in ppb-hrs); σ = accumulated ozone exposure at which α is reduced by 63 percent; and c = a dimensionless shape parameter.

Results

Ozone Monitoring from 1991-1994

Although average ozone concentrations varied at stations within a given sub-region, e.g., the central Sierra Nevada (Carroll and Dixon 1993, Van Ooy and Carroll 1995), average concentrations from north (Lassen Volcanic National Park) to south (San Bernardino Mountains) increased. Thus, we found an adequate exposure gradient to examine the relationship between ozone exposure and the injury response of pines.

OII Values Along a North to South Transect — 1992-1994

The least injury is present in the northern to central Sierra Nevada (*fig. 2a*) and the most in the south central Sierra Nevada and the San Bernardino mountains (*fig. 2b*). The observed injury amounts are generally proportional to seasonal ozone exposures reported by Van Ooy and Carroll (1995) and by Miller and others (1996a). The combined ozone and OII data sets are considered only marginally useful for providing adequate detail of the spatial distribution of injury because the locations of ozone monitors and tree plots could not be randomly selected. The change in yearly OII at any single site is considered a satisfactory indicator of exposure response because it was a repeated measurement of the same trees. Between 1991 and 1994 at the northern locations the OII rose slightly at four of six locations and was essentially unchanged at each of the two remaining locations (*fig 2a*). At the six southern locations OII declined slightly (less injury at all places where the record was complete) except at Barton Flats where it increased slightly (*fig. 2b*).

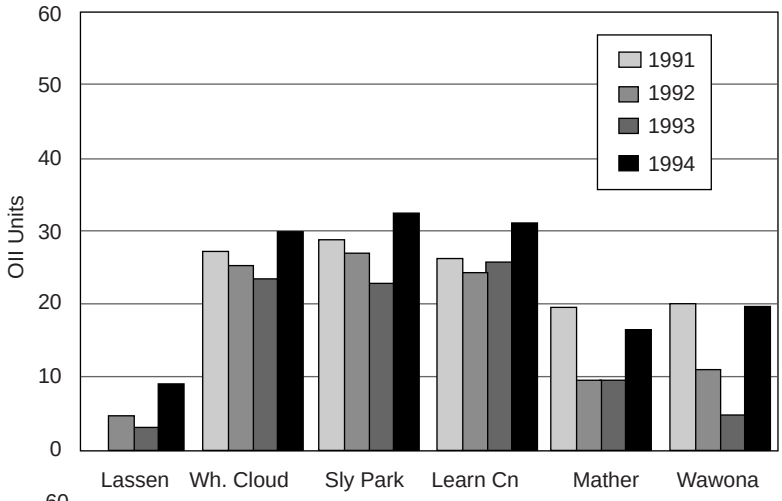

Figure 2a — Annual changes in the ozone injury index (OII) from 1991 to 1994 at six northern or central California sites: Lassen Volcanic , White Cloud, Sly Park, Learning Center, and Camp Mather and Wawona in Yosemite National Park.

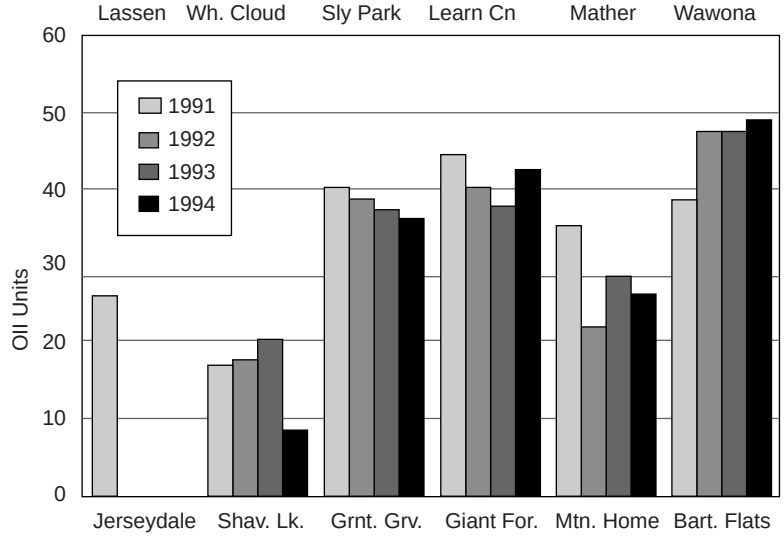

Figure 2b — Annual changes in the ozone injury index (OII) from 1991 to 1994 at five south-central Sierra Nevada sites and one southern California site: Jerseydale, Shaver Lake, Grant Grove and Giant Forest in Sequoia-Kings Canyon National Park, Mountain Home State Park, and Barton Flats in the San Bernardino National Forest. Data are incomplete for Jerseydale from 1992 to 1994 because of unusual amounts of bark beetle-caused mortality.

Correlation of Percent Chlorotic Mottle with Several Ozone Exposure Indices and Cumulative Days of Exposure

The year by year change of percent chlorotic mottle is illustrated as a function of accumulating ozone exposure received by current-year, 1-year-old, 2-year-old, and 3-year-old needles. The data were tested with a number of fitted curves, including quadratic and a Weibull function. The latter was selected for this application because it is a monotonically decreasing (or increasing) fit that more likely applies to the expected biological response in this case. The r^2 for both quadratic and Weibull fits were computed. The corrected regression coefficients for percent chlorotic mottle and each of the exposure indices were:

Index	*Quadratic*	*Weibull*
Sum 0	.583	.574
Sum 60	.592	——
W126	.596	——
Hrs ≥ 80	.553	——
Expos. Days	.433	.424

The r^2 for Sum 0 was not the highest value for quadratic fits but it was the only form of accumulated ozone exposure that converged to the required parameters needed for the Weibull fit. These results point to Sum 0 as an acceptable exposure index to apply in this situation. Therefore, the Weibull equation for percent chlorotic mottle (PCM) and Sum 0 was:

PCM = 100-100*EXP(-(Sum 0/1579000) 2.170)

During the 1991 to 1993 seasons the Sum 0 accumulated exposure ranged between 0 and 600,000 ppb-hrs (*fig. 3*). It was not feasible to observe a needle whorl for longer than three seasons (e.g., 1991 needles could not be used in 1994) because many needle fascicles would have abscised and those few remaining would have less chlorotic mottle. These results indicate that Sum 60, W126, and hrs ≥ 80 are not feasible to use because these data did not converge to the required parameters needed for the Weibull curve fit. Accumulated number of exposure days had the lowest coefficient.

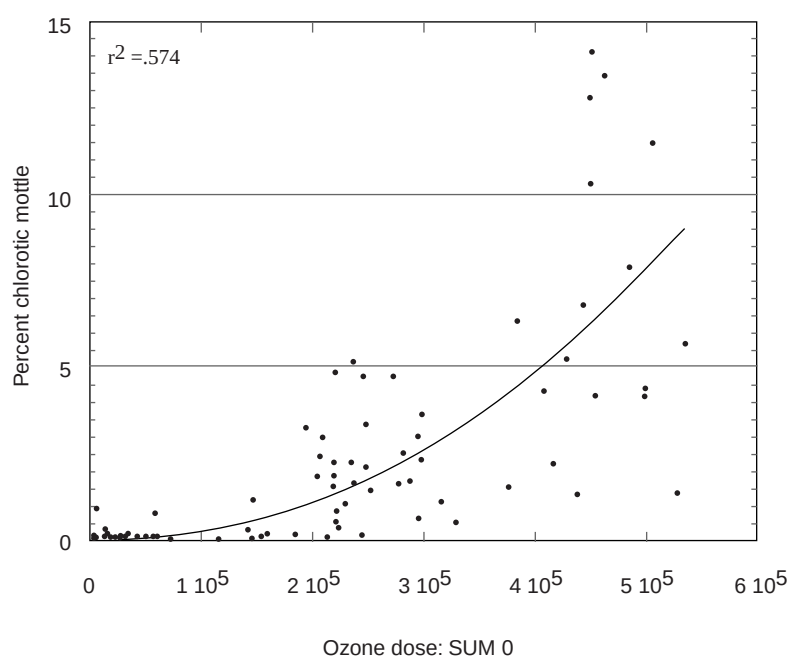

Figure 3 — Weibull curve fit of percent chlorotic mottle accumulated by 1991 to 1993 whorls in relation to Sum 0 ozone.

Correlation of Percent Fascicle Retention with Several Ozone Exposure Indices and Cumulative Days of Exposure

As chlorotic mottle develops on needle fascicles, the natural abscission of needle fascicles within each annual whorl increases thereby leaving whorls with fewer needle fascicles until eventually all needle fascicles abscise and the entire whorl is gone. The corrected r^2 values show Sum 0 to be the Weibull fits:

Index	*Quadratic*	*Weibull*
Sum 0	.759	.742
Sum 60	.675	.613
W126	.653	.609
Hrs ≥ 80	.443	.411
Expos. Days	.692	.669

Because the expected biological response corresponds to the monotonically decreasing form of the Weibull function, it was selected for relating percent fascicle retention (PFR) to the selected exposure indices, e.g., in the case of Sum 0:

PFR = 83.5*EXP(-(Sum 0/732600) 3.057)

For the Weibull fit of Sum 0 data versus percent fascicle retention (*fig. 4*) the maximum amount for Sum 0 exposure is 800,000 ppb-hr as compared to 600,000 ppb-hr for percent chlorotic mottle because it was feasible to use a longer time span (1991-94) for percent fascicle retention than for percent chlorotic mottle (1991-93).

Three Dimensional Display of the Sum 0 Exposure Index, and the Frequency of Sampled Trees in Original Classes of Chlorotic Mottle

Because the Weibull function correlations of Sum 0 versus percent chlorotic mottle involve transformed data, such as the midpoint value of each class, 1 = 1-6 percent (3.5 percent) and 2 = 7-25 percent (16 percent) etc., we examined the distribution of the untransformed data. We related the frequency of trees to the original chlorotic mottle classes as a function of Sum 0 exposure index (*fig. 5*). The threshold above which category 1 (1 to 6 percent) chlorotic mottle appears is about 150,000 ppb. The equivalent threshold values for category 2 (7 to 25 percent) chlorotic mottle could be more useful as a category to monitor because needle fascicle abscission begins at this level.

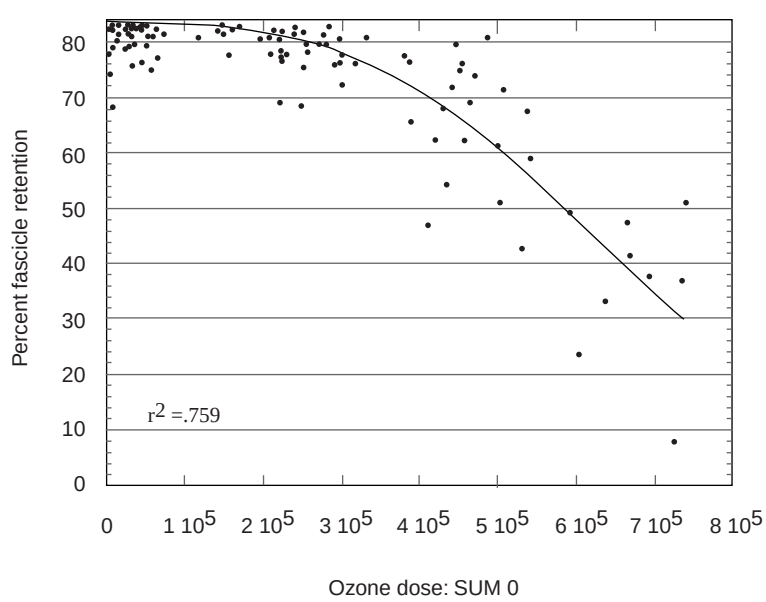

Figure 4 — Weibull curve fit of percent needle fascicle retention accumulated by 1991 to 1994 whorls in relation to Sum 0 ozone.

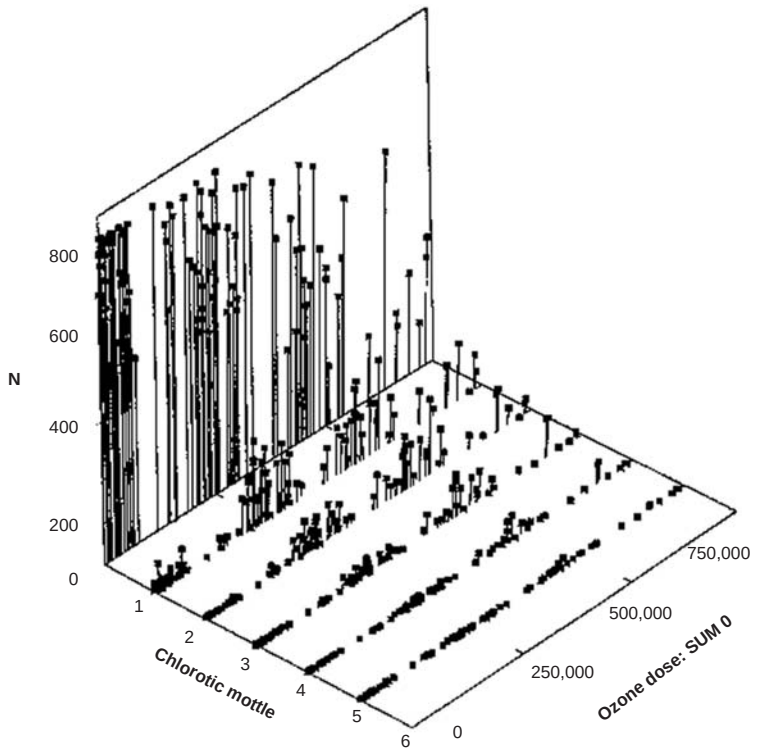

Figure 5 — Distribution of 1991 to 1994 annual whorls from all trees into original categories for chlorotic mottle as a function of Sum 0 ozone exposure index during the same period.

Discussion

The time resolution for examining pine needle response to ozone can vary from daily examination of physiological variables (Grulke and Lee, [In press]; Temple and Miller, [this volume]) to annual visual ratings of injury. For the purpose of evaluating annual increments of injury increase and comparing yearly changes, the visual rating method used in this study, the OII (Miller and others 1996b), appears to provide a reliable estimate. It could be improved by the inclusion of an estimate of the seasonal pattern of stomatal conductance and an estimate of ozone flux to pine foliage, but this improvement generally requires concurrent field measurements of stomatal conductance. Attempts to apply models of ozone flux that are parameterized primarily by atmospheric conditions (temperature and relative humidity) have realized very limited success as predictors (Fredericksen and others 1996) when compared to flux estimates based on actual conductance measurements.

Because of the relative simplicity of the OII, we recommend it as a suitable protocol for monitoring yearly injury. In our study we have used a large number of trees (1,700) to evaluate the visual change of individual age cohorts of needles on each tree exposed to ozone in their initial year and in successive years. Certainly the climate and soil-water availability conditions from 1991 to 1994 were variable and caused important yearly differences in ozone uptake and incremental changes of needle injury amounts. However, annual differences were detected by the chlorotic mottle element of the OII (Temple and Miller, [this volume]), and to a lesser degree by the OII itself. In further support of the visual estimation procedure of the OII, it was possible to include enough trees to represent the range of genetic resistance or susceptibility of ponderosa and Jeffrey pines as well as trees in different crown position classes (e.g. dominant, codominant, etc). Large sample numbers are usually difficult to achieve with physiologically-based response variables (Grulke and Lee [In press]). By using the OII or its most sensitive components, we can examine further the different versions of a possible air quality standard that may be appropriate for the task of protecting ozone-sensitive tree species.

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SESSION II

Air Pollution Status of Forest Sites

Paul R. Miller and Juri Martin, Chairs

Ozone Air Pollution in the Ukrainian Carpathian Mountains and Kiev Region[1]

Oleg Blum,[2] Andrzej Bytnerowicz,[3] William Manning,[4] and Ludmila Popovicheva[2]

Abstract

*Ambient concentrations of ozone (O_3) were measured at five highland forest locations in the Ukrainian Carpathians and in two lowland locations in the Kiev region during August to September 1995 by using O_3 passive samplers. The ozone passive samplers were calibrated against a Thermo Environmental Model 49 ozone monitor located at the Central Botanical Garden in Kiev. The 2-week long average concentrations in August to September at the Carpathian Mountains ranged from 27.4 to 51.8 ppb, and at the lowland forest location (Lutezh, near Kiev) ranged from 29.3 to 32.2 ppb. The 2-week long average concentrations in Kiev (Botanical Garden) ranged from 22.0 to 31.5 ppb. The highest diurnal average concentration in August 1995 in the Botanical Garden was 42.1 ppb while the highest 1-hr average concentration reached 84.4 ppb. Ozone-sensitive Bel-W3 tobacco (*Nicotiana tabacum *L.) plants at the Botanical Garden in Kiev were injured from exposure to ambient concentration of ozone. Ozone injury symptoms were found on native plants (e.g.,* Sambucus racemosa *and* Humulus lupulus*) elsewhere in Kiev and at some of the study sites in the Carpathians.*

Introduction

In general, elevated concentrations of sulfur and nitrogen oxides have been blamed for air pollution-related damage to forest vegetation in central and eastern Europe. However, increasing concentrations of ozone may also play an important role in the observed suite of symptoms of forest decline in that part of Europe. Ozone alone may affect plant health, while the effects of mixtures of ozone with other pollutants, especially with sulfur dioxide, may be synergistic (more than additive) in nature (Guderian 1985).

Recently, ozone concentrations have significantly increased across Europe mainly because of increased production of ozone precursors from combustion of gasoline and other fuels (Derwent and Jenkin 1991). Long range atmospheric transport may also be responsible for elevated concentrations of ozone in various forested areas of the region (Dovland 1987).

In Ukraine the annual atmospheric emissions of gaseous pollutants are: NO_x (nitrogen oxides) — 1.1 million tons; CO (carbon monoxide) — 8 million tons; C_xH_y (hydrocarbons) — 1.4 million tons (National Report 1994). As a result of photochemical transformation of these emissions, the level of surface background concentrations of ozone could theoretically increase two-fold compared to the pre-industrial pollution period (Nikolay Gurevich, personal communication).

Average 24-hour background O_3 concentrations in the unpolluted lower troposphere ranged from 10 to 40 ppb (Logan 1985). It has been estimated that in the countries of the former USSR the background O_3 concentrations ranged from 15 to 40 ppb (Israel 1984). Concentrations of ozone in European countries of higher latitudes, such as Finland (Laurila and Lattila 1993) or Lithuania (Girgzdiene 1991), are relatively low and do not pose a threat to vegetation. However, higher ozone concentrations, potentially phytotoxic, occur in the lower latitudes of western and central Europe (Emberson and others 1996, Semenov and Kouhta 1996).

Still relatively little is known about concentrations of ozone and potential toxic effects of ozone in forests in the former Communist countries of central and eastern Europe, particularly in Ukraine. Levels of ozone in central and western European countries of comparable climatic conditions such as Poland (Bytnerowicz and others 1993, Godzik 1996), Czech Republic (Bytnerowicz and others 1995), Austria

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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(Dovland 1987), Switzerland (Ballaman 1993), or France (Proyou and others 1991) often exceed permissible levels for that pollutant. For example, in forested areas of central and southern Poland 1-hour mean ozone concentrations can be as high as 105 ppb (Bytnerowicz and others 1993). During long periods of hot weather and high solar radiation, the highest 1-hour mean ozone concentrations monitored in Katowice (the Upper Silesia Region of Poland) have reached 115 ppb (Godzik and others 1994).

Information about the air pollution status of the Ukrainian Carpathians is essential for a better understanding of environmental stresses that affect valuable forest ecosystems of central Europe. Thus, level of ozone, one of the main components of photochemical smog and a strong phytotoxic agent, is a primary area of study.

Until recently, surface background concentrations of ozone in natural landscapes of Ukraine were not measured, with the exception of O_3 monitoring in the territory of Karadag State Reserve on a coast of the Black Sea of the Crimea. At that site, the 24-hour average O_3 concentrations in the spring-summer period of about 15 ppb and 1-hour average O_3 maximum concentrations of about 105 ppb have been recorded.[5]

This paper discusses results of a pilot study that used ozone-sensitive Bel-W3 tobacco and other plants to measure ozone concentrations in Ukraine.

Materials and Methods

Research Sites

In summer 1995 research sites were established in five forest locations in the Ukrainian Carpathians (*fig. 1*) in the forest complex of the northern Ukraine (Staropetrovskaya Forest Research Station, vil. Lutezh, near Kiev) and at two locations, one the northern and one in the southern part, in the Central Botanical Garden of the National Academy of Sciences in Kiev (50°24′47″ N, 30°34′14″ E; 190 m a.s.l.).

Forest sites were located at altitude 750-1,000 m a.s.l. along the Carpathian range from the west to south-west: 1 — Uzhoksky Pass (Sjanki village, 850 m a.s.l.); 2 — Synevir (National Park "Synevir," 1,000 m a.s.l.); 3 — Shiroky Lug (Carpathian Biosphere Reserve, Kuzij massif, peak Sokolyne Berdo, 750 m a.s.l.); 4 — Yablunitsa village, Yablunetski Pass (Carpathian National Park, 968 m a.s.l.); 5 — Kryvopilja village (Carpathian National Park, 975 m a.s.l.).

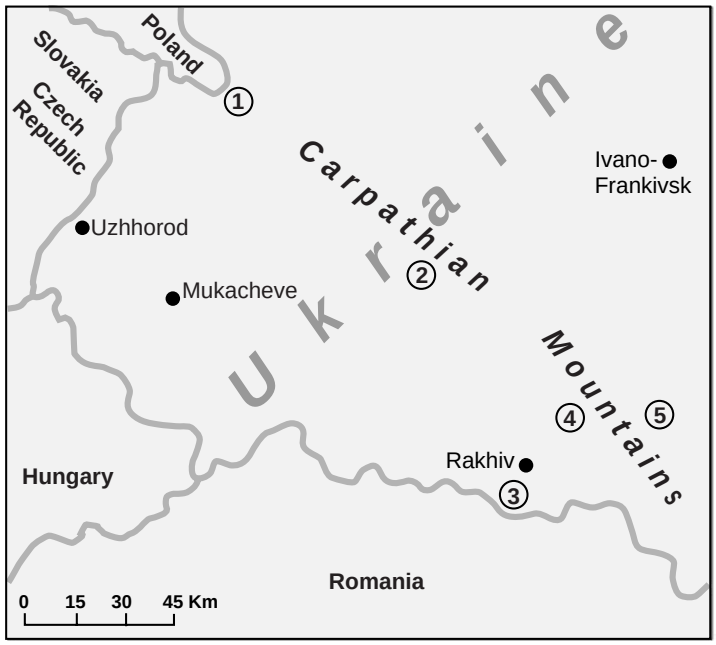

Figure 1 — Ozone monitoring sites in the Carpathian Mountains in Ukraine:
1 — Uzhoksky Pass,
2 — Synevir,
3 — Shiroky Lug,
4 — Yablunitsa,
5 — Kryvopilja.

[5] Unpublished data on file, Gas Institute, National Academy of Sciences of Ukraine

Ozone Monitoring

Average ambient concentrations of O_3 were determined with the Ogawa and Co., Inc., passive ozone samplers[6] (Koutrakis and others 1993). The samplers were exposed to ambient air for a duration of 7 to 14 days. Measurements were carried out in all forest locations from the beginning of August to the end of September.

In the Central Botanical Garden (Kiev) measurements of O_3 with passive samplers were accompanied by continuous monitoring with the Thermo Environmental Model 49 ultraviolet absorption instrument (Cambridge, MA). Results from passive ozone samplers were compared to those obtained from the Thermo Environmental instrument, which had been calibrated before its use.

Ozone Phytotoxicity

Cultivated and native plants were surveyed for symptoms of ozone injury at all five Carpathian forest locations and in Kiev. To determine the phytotoxicity of ambient O_3, two cultivars of tobacco (*Nicotiana tabacum* L.) were used: Bel-W3 (O_3-sensitive), and Bel-B (O_3-tolerant) (Heggestad 1991). Before field exposures, all seedlings were grown in charcoal filtered air to the 4-leaf stage of development in a special growing substrate (pH = 6.5), consisting of peat, perlite, and gypsum (1:1:1 by volume).

The ozone-free air chamber made of transparent plastic (1.1 m^3 volume) was constructed in order to grow tobacco plants. The chamber was located in the laboratory room. Two incandescent lamps (500 W) wrapped with aluminum foil (for darkening) and a thermo-relay kept constant temperature at about 32 - 35 °C. Air entering the chamber was pumped at a flow rate of 30 l/min. through a charcoal filter. Cuvettes with water placed at the bottom of the chamber provided the required humidity for proper growth of the tobacco plants. The plants were grown under natural and luminescent lighting (light/dark regime — 16:8 hours).

Injury of the Bel-W3 tobacco plants in the Central Botanical Garden was observed in two locations from early August to early September 1995. At one of the research sites, measurements with O_3 passive samplers and bioindicators were accompanied by photometric monitoring of ambient O_3 concentrations (near Building No. 5, Department of Plant Physiology). At another location, (i.e., the Palmetto Orchard) measurements were done by using only O_3 passive samplers and the bioindicator tobacco plants.

Results and Discussions

Ozone Concentrations

The 2-week long average concentrations in August to September in the Carpathian Mountains sites ranged from 27.4 to 51.8 ppb, while at Lutezh, the lowland forest location near Kiev, they ranged from 29.3 to 32.2 ppb. The 2-week long average concentrations in Kiev (Botanical Garden) ranged from 22.0 to 31.5 ppb (*table 1*). The highest 24-hour average concentration measured with the Thermo Environmental instrument in August 1995 in the Botanical Garden was 46.7 ppb, while the highest 1-hr average concentration reached 84.4 ppb. The lowest 1-hour average concentrations were as low as 7.0 ppb (*fig. 2*).

In Kiev, typically, the lowest O_3 levels occurred in the morning with highest concentrations found in the afternoon. Average concentrations of O_3 at night were about 25 ppb, while the maximum values reached about 47 ppb (*fig. 3*).

In general, O_3 concentrations in the five mountain forested locations in the Ukrainian Carpathians were low or only moderately elevated. The determined concentrations were similar or slightly lower than the values determined in the forested locations of central and southern Poland (Bytnerowicz and others 1993, Godzik 1996) and the Czech Republic (Bytnerowicz and others 1995). Ozone results obtained with the Ogawa passive samplers in areas of low ozone concentrations have about ± 20 percent sampling accuracy (Koutrakis and others 1993).

[6] Mention of trade names or products is for information only and does not imply endorsement by the U.S. Department of Agriculture.

Table 1 *— Concentrations of ozone determined with the Ogawa passive samplers in summer 1995 in the Ukrainian locations (ppb).*

Location	August 6-19	August 19-Sept. 4	September 4-18
1. Uzhoksky Pass	40.9	36.3	34.2
2. Synevir	39.5	38.3	27.4
3. Shiroky Lug	27.4	33.1	51.8
4. Yablumitsa	-	36.1	-
5. Kryvopilja	-	35.8	-
6. Kiev, Palmetto	22	31.5	-
7. Kiev, Bldg. No. 5	30	29.8	25.7
8. Lutezh	29.3	32.2	-

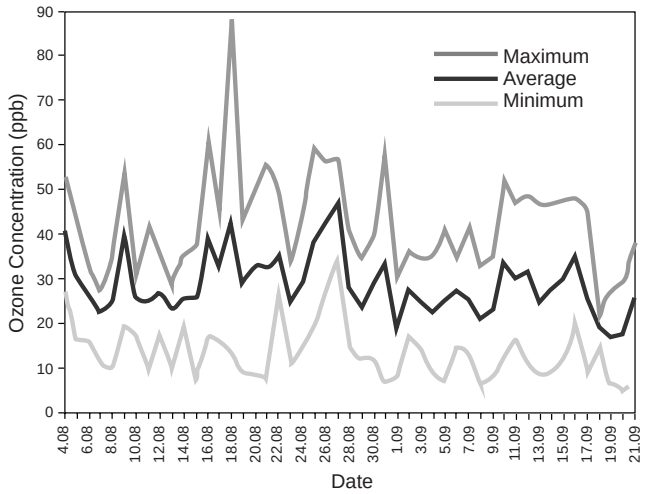

Figure 2 — Results of O_3 monitoring in the Botanical Garden in Kiev during the August 4 to September 21, 1995 period. Results are shown as 24-hour averages, and 1-hour maximum and 1-hour minimum values for every day during the monitoring period (ppb).

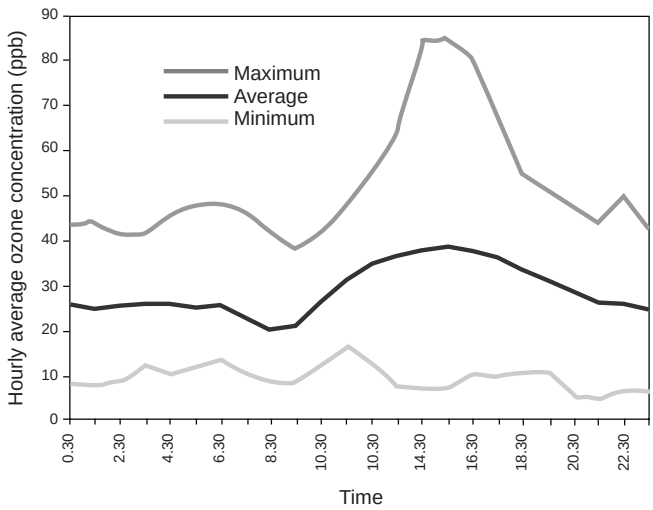

Figure 3 — Diurnal dynamics of O_3 concentrations in the Central Botanical Garden, National Academy of Sciences in Kiev, shown as averages for the August 4 to September 21, 1995 period. Results are shown as mean, maximum, and minimum values for every hour.

Ozone Phytotoxicity

Ozone injury symptoms were present on both Bel-W3 (ozone sensitive cultivar) and Bel-B (ozone tolerant cultivar) tobacco plants exposed to ambient air in the Botanical Garden (Kiev). The highest amount of O_3 injury was found on the Bel-W3 plants at the Palmetto Orchard and near the Building No. 5 (*table 2*) reaching 62 percent of the foliar area on the first leaf of plants after 14-day exposure. Moderate injury on the Bel-W3 plants (11-20 percent of the leaf area) occurred at the Building No. 5 location after 7-day exposure. Slight O_3 injury was also observed on the Bel-B tobacco plants (ozone tolerant cultivar, 1-4 percent of the leaf area) in the Building No. 5 location after 14-day exposure. Higher degree of injury on the Bel-B plants (up to 13 percent of the leaf area) occurred at the Palmetto Garden location after 14-day exposure.

Table 2 *— Injury of the Bel-W3 and Bel-B tobacco plants at two locations in the Central Botanical Garden, Kiev (percent of leaf area injured presented as means and standard deviation; n=4).*

Location	Leaf No.	7 days of exposure (Aug. 7-14)	14 days of exposure (Aug. 7-21)	7 days of exposure (Aug. 22-29)	14 days of exposure (Aug. 22-Sept. 6)
Bel-W3					
Palmetto Orchard	1	37 ± 9	40 ± 11	30 ± 12	62 ± 1
	2	32 ± 14	35 ± 15	23 ± 8	57 ± 11
	3	7 ± 3	25 ± 5	5 ± 3	11 ± 31
	4	0	4 ± 3	0	2 ± 6
Building No. 5	1	11 ± 5	18 ± 6	19 ± 9	39 ± 16
	2	20 ± 9	27 ± 8	17 ± 7	40 ± 8
	3	2 ± 1	6 ± 4	9 ± 5	18 ± 13
	4	0	1 ± 1	0	0.5 ± 0.9
Bel-B					
Palmetto Orchard	1	0	0	6 ± 2	13 ± 6
	2	0	0	1 ± 1	11 ± 7
	3	0	0	0	0
	4	0	0	0	0
Building No. 5	1	0	0	0	4 ± 2
	2	0	0	0	1 ± 1
	3	0	0	0	0
	4	0	0	0	0

Ozone injury was found on morning glory (*Ipomea purpurea*) leaves at a monastery complex in Kiev (*table 2*). Wild hop (*Humulus lupulus*) and native clematis (*Clematic vita alba*) showed possible ozone injury at the Botanical Garden in Kiev. At Uzoksky Pass, possible ozone injury was observed on leaves of red-fruited elderberry (*Sambucus racemosa*), a common forest inhabitant. At Synevir, elderberry also showed possible ozone injury. At Shiroky Lug, probable ozone injury was observed on blackberry (*Rubus hirtus*), hazelnut (*Corylus avellana*), wild clematis (*Clematis* hyb.), and vincetoxicum (*Vincetoxicum* sp.). At a forestry administration unit near Rakhiv, probable ozone injury was noted on cultivated clematis, wild hop, and elderberry.

Some native plants are potentially excellent detector indicators of ambient ozone in the Carpathian Mountains of Ukraine:

- Actual Injury — Bel-W3 tobacco (*Nicotiana tabacum*), morning glory (*Ipomoea purpurea*).
- Probable Injury — wild hop (*Humulus lupulus*).
- New Species — clematis (*Clematis vita alba*), clematis (*Clematis hyb.*), hazelnut (*Corylus avellana*), Vincetoxicum (*Vincetoxicum sp.*).
- Known Species — blackberry (*Rubus hirtus*), elderberry (*Sambucus racemosa*).

Their response to ozone needs to be verified so that they can be used as bioindicators.

In general, the results of our pilot study indicated that ambient O_3 in Kiev and some forest regions of Ukraine during some summer periods could injure plants. The predicted increase of automobile traffic in Ukraine and the neighboring countries and long-range transport of the air masses contaminated with

photochemical smog may cause further increase of O_3 concentrations in the forested areas of the Ukrainian Carpathian Mountains. Continuous monitoring of O_3 concentrations and its phytotoxic effects in the Carpathian forests and other parts of Ukraine is needed in order to signal the potential for deterioration of sensitive plant species in this part of Europe.

Acknowledgments

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Ozone Monitoring in the Krakow Province, Southern Poland[1]

Barbara Godzik[2]

Abstract

From June to mid-October in 1995, the concentration of tropospheric ozone in 18 localities in the Krakow Province of southern Poland was measured by using ultraviolet monitors and Ogawa passive samplers. At three active monitoring stations, tropospheric ozone was recorded in the downtown and western part of Krakow and in Szarow, 30 km to the east. The passive method was applied in a dozen or so localities distributed within and outside Krakow's city limits. In these locations two varieties of tobacco (Nicotiana tabacum *L.), Bel-W3 and Bel-B, were exposed. In the Krakow Province the mean 24-hour ozone concentration (from active monitors) was higher than 30 μg/m³ (24-hour Polish standard) in June through July and part of September. The highest concentration was recorded in early afternoons and the lowest between midnight and sunrise. The maximum 30-minute concentration of 205 μg/m³ occurred in August. However, great diversity of the tropospheric ozone concentration (measured with passive samplers) was recorded in all locations. The lowest average concentration was recorded in Krakow and areas located west, northwest and east of the city, whereas the highest average concentration was recorded in the north, northeastern, southeastern, and southwestern parts of the Province. The highest average concentration of 100 μg/m³ for a 2-week period of filter exposure was recorded in July and August from the sites Ratanica, located 40 km south, and Goszcza, 25 km north of Krakow. The amount of damage to the leaves of the ozone-sensitive variety of tobacco (Bel-W3) exposed at all sites was correlated (coefficient 0.69, p<0.000) with concentration results obtained with passive samplers. Greater damage was recorded in the sites located outside the city than elsewhere, whereas the damage that occurred within Krakow was markedly smaller.*

Introduction

Tropospheric ozone is believed to be an important phytotoxic air pollutant (Fuhrer and Achermann 1994, Guderian 1985, Stanners and Bourdeau 1995). Ozone levels recorded in western Europe, such as France (Proyou and others 1991), Switzerland (Ballaman 1993), or Austria (Dovland 1987), exceed the permissible concentrations for that pollutant. In the countries located in northeastern Europe, such as Finland (Laurila and Lattila 1993) or Lithuania (Girgzdiene 1991), concentrations of ozone are relatively lower than in central Europe (Simpson 1993). Southern Poland is among the most polluted areas of central Europe. The high input of sulphur, nitrogen oxides, and heavy metals has originated from local and remote sources (Nowicki 1993). Recent studies performed in southern Poland (Upper Silesia Region, Beskidy Mountains, Krakow Region) have shown that during the summer season a single 1-hour average ozone concentration may reach 120 ppb (Bytnerowicz and others 1993, Godzik and others 1994).

In the Krakow agglomeration, the main sources of pollution are a huge steel mill, a cement factory, an electric power station, privately-owned coal stoves and also rapidly increasing car traffic (Turzanski 1994). The highest concentration of air pollutants in the Krakow Province occurs centrally in the city of Krakow (Turzanski 1996).

This paper reports a study of ozone concentrations within the city of Krakow in southern Poland and outside its limits; identifies which areas have the highest ozone levels; and evaluates phytotoxic effects of ozone on the basis of responses of bioindicator tobacco (*Nicotiana tabacum* L.) plants.

Study Area and Methods

The Krakow Province covers an area of 3,254 km^2. In 1995 measurements of the ozone concentration were made at 18 sites: 7 among them were located within the

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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city limits, 11 were evenly distributed in the Krakow Province *(fig. 1)*. Two sites located within the city (Balice, Wieza Ratuszowa) and the one in Szarow (30 km east of the city) were equipped with continuously working monitors (Thermo Environmental Model 49 UV-photometric ambient O_3 analyzer, Franklin, MA).[3] In two stations within the city measurements were performed by the Voivodship Inspectorate for Environmental Protection in Krakow. The correlation coefficient for the conversion of ppb to $\mu g/m^3$ were 1.93 for T=20 °C. All sites were furnished with the Ogawa passive samplers (Ogawa and Company, Pompano Beach, FL) (Koutrakis and others 1993). Passive sampler exposure was done from early June to mid-October in 2- or 3-week intervals. The O_3 concentration in the Krakow Province was mapped by using Surfer 5.0 program. Isolines were made following the kriging method for the parameters: component 1 = linear, scale C = 133, length A = 40.7. Graphic description was made in CorelDraw 3.0 program.

In every monitoring site four plants of the Bel-W3 and four plants of the Bel-B tobacco *(Nicotiana tabacum* L.*)* varieties were exposed during two periods (07/20-08/08 and 08/11-09/06). Damage to leaves was measured at 1-week intervals according to procedures described by Manning and Feder (1980). Four leaves on each plant were evaluated for injury by the sum of percentage area showing symptoms on each leaf. The correlation between percentage of leaf damage and ozone concentration was analyzed by Spearman rank test.

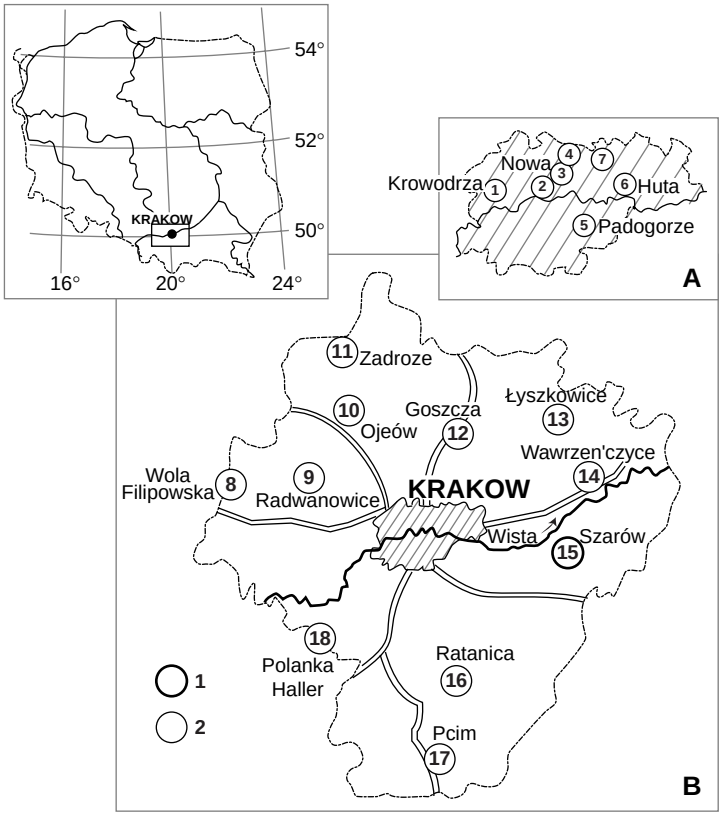

Figure 1 — Distribution of ozone-monitoring sites in the Krakow city (A) and the Krakow Province (B). 1—stations equipped with monitor and passive samplers, 2—stations equipped with passive samplers.

Results

Thirty-minute average concentrations of ozone measured with continuous monitors varied predictably during a 24-hour period (especially in July and August) and between months in all sites. The ozone concentration at night decreased to several $\mu g/m^3$, and it increased to 160 $\mu g/m^3$ during the day *(fig. 2)*. According to the Polish standard, the 30-minute concentration should not be greater than 100 $\mu g/m^3$. About 100 hours in June and July and more than 200 hours in August showed concentrations higher than the standard. In September excessive ozone concentrations occurred less frequently *(fig. 3)*.

[3] Mention of trade names or products is for information only and does not imply endorsement by the U.S. Department of Agriculture.

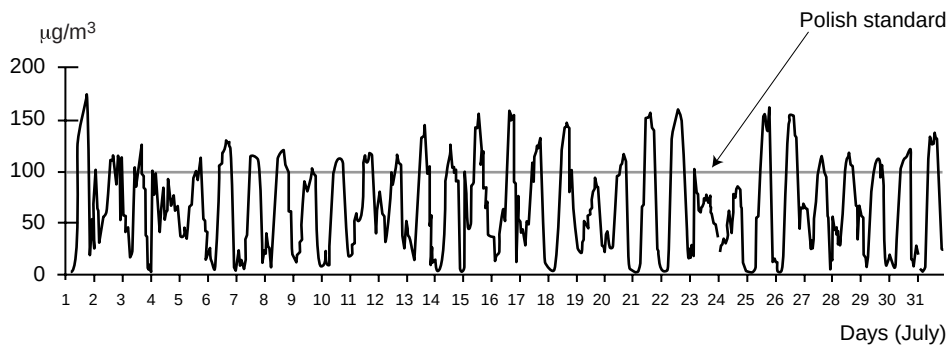

Figure 2 — Diurnal pattern of ozone concentration (µg/m³) in July 1995 at the Balice monitoring station (western part of the city). (1µg/m³ O_3= 0.51 ppb O_3 at 25ºC, 760 mm of Hg.)

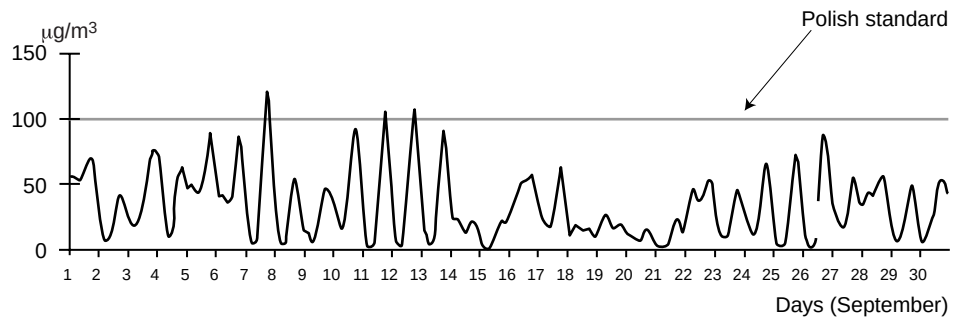

Figure 3 — Diurnal pattern of ozone concentration (µg/m³) in September 1995 at the Balice monitoring station (western part of the city).

Table 1 — *Highest 30-minute ozone concentration (µg/m³) at three sites in southern Poland, by month of measurement.*

	June	July	August	September
Balice	152	177	[1]159	120
Wieza Ratuszowa	148	189	205	126
Szarow	-	-	180	129

[1] *Data incomplete for a whole period*

The maximum 30-minute ozone concentration for subsequent months occurred in August *(table 1)*. The highest ozone concentration occurred in August. The lowest concentration was recorded between midnight and sunrise and the highest in early afternoons *(fig. 4)*.

In July and August the maximum mean 24-hour concentrations at the Balice research station were higher than in June. These values were markedly lower in September than in the previous months *(table 2)*. The mean 24-hour concentrations of ozone at the Wieza Ratuszowa station were similar in subsequent months to the concentrations recorded in the Balice station. In general, however, the minimum values for subsequent months were lower and the maximum values were higher than the ones recorded from the Balice station *(table 2)*. The mean 24-hour permissible ozone concentration, which follows the Polish standard, should not be higher than 30 µg/m³. In June, July, August, and part of September the mean 24-hour concentration in Krakow was markedly higher than the Polish standard.

During the period of measurements from June through September, the mean monthly concentrations measured by active monitor were higher at the Balice station (western part of the city) than at the Wieza Ratuszowa station (center of the city). The highest mean monthly value for August was recorded from the Szarow station (30 km east of Krakow) *(table 2)*.

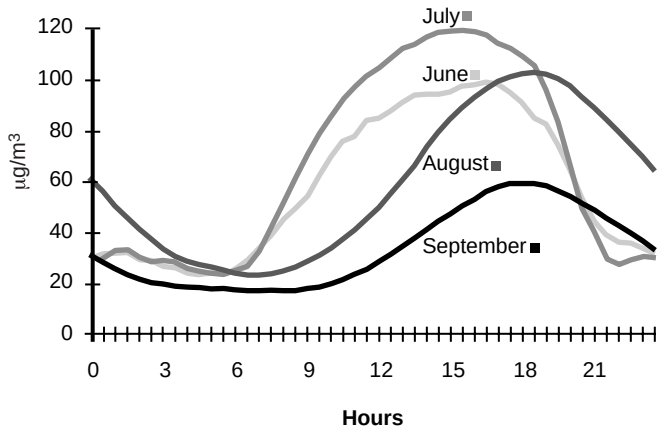

Figure 4 — Monthly averages of diurnal dynamics of ozone concentrations (µg/m³) at the Balice monitoring station.

Table 2 *— Average daily minimum (min) and maximum (max) and average monthly (x) ozone concentrations (µg/m³) at the Balice, Wieza Ratuszowa, and Szarow Research Station.*

Month		Balice	Wieza Ratuszowa	Szarow
	min	30.21	25.11	-
June	max	69.71	78.43	-
	x	57.95	55.06	-
	min	46.66	45.57	-
July	max	83.02	87.72	-
	x	66.46	65.28	-
	min	30.13	21.49	28.16
August	max	92.08	98.41	113.96
	x	58.98	53.86	68.68
	min	10.51	5.25	10.28
September	max	58.96	52.65	61.48
	x	34.06	27.64	30.41

In all periods of measurements made with passive samplers, the average ozone concentration was markedly different among selected sites and exposure periods *(table 3)*. In the first two 2-week periods of measurement (including June) and in the last two (September and October) the mean ozone concentration was low (table 3). In July and August it increased dramatically. The maximum values for a 3-week exposure (June, July) were recorded from Goszcza and Ratanica research site (about 100 µg/m³ *(table 3)*. During the same period, a high average ozone concentration occurred also in Szarow, Polanka Haller, and Trzyciąz. In all measurement periods the lowest average concentration of ozone was recorded from the sites located in the city center (Krasinski Avenue, Helclow Street) *(table 3)*. All sequences of measurements yielded a similar pattern of spatial differentiation of the ozone concentration in the Krakow Province. During all exposure periods the lowest average ozone concentration occurred in the center of the Krakow Province (Krakow city and the areas located west, northwest, and east of the city) and in its southern part *(figs. 5, 6, 7)*. The highest concentration was determined in the north, northeastern, southeastern, and southwestern parts of the Krakow Province *(figs. 5, 6, 7)*.

The amount of damage to the leaves of the ozone-sensitive variety of tobacco (Bel-W3) exposed at all sites was correlated with results obtained through passive samplers. This coefficient (=0.69, $p<0.000$) was statistically significant. During two exposition periods of tobacco *(Nicotiana tabacum* L.*)* the lowest injury to the leaves of the ozone-sensitive (Bel-W3) variety of tobacco was recorded at the sites located in the center of Krakow (Krasinski Avenue, Helclow Street)

and at the sites located west of the city (Radwanowice, Ojcow, Wola Filipowska) *(fig. 8)*. The most severe injury to the tobacco leaves were noted in sites located in the northern, southeastern, and southwestern parts of the Krakow Province (Trzyciąz, Goszcza, Szarow, Ratanica, Polanka Haller) *(fig. 8)*. No leaf injury was observed in the ozone-tolerant tobacco variety (Bel-B) at any of the study sites.

Table 3 — *Average ozone concentrations (μg/m³) during different exposure periods in the Krakow city and the Krakow Province.*

Station	Exposure periods						
	5/31 -6/13	6/14 -6/27	6/27 -7/17	7/17 -8/7	8/7 -8/30	8/30 -9/21	9/21 -10/17
Krakow city:							
1. Balice	54.5	43.3	72.9	78.5	55.8	32.1	32.2
2. Krasinskiego	-	-	19.3	14.7	17.1	12.7	10.8
3. Helclow	-	4.9	25.7	25.1	23.5	12.7	7.6
4. Strzelcow	-	-	50.2	57.9	45.0	28.5	18.5
5. Wallenroda	-	-	71.5	65.0	46.3	35.3	22.3
6. Zeromskiego	-	43.6	71.1	66.9	48.4	26.6	19.3
7. Zielone	-	-	57.2	49.3	38.7	23.2	17.4
Krakow Province:							
8. Wola Filipowska	44.0	30.3	55.1	49.8	39.1	24.9	19.5
9. Radwanowice	47.7	31.4	49.8	55.8	43.8	24.8	21.7
10. Ojcow	29.8	12.4	32.9	37.9	29.1	19.1	16.8
11. Zadroze	60.7	57.3	86.2	88.0	67.1	45.4	39.3
12. Goszcza	65.0	57.2	100.1	101.5	61.4	48.1	35.3
13. Lyszkowice	54.3	48.1	77.9	81.5	60.8	42.3	29.4
14. Wawrzenczyce	40.3	17.9	50.5	50.0	39.3	31.8	19.8
15. Szarow	58.8	51.1	81.5	89.1	70.5	39.1	45.9
16. Ratanica	69.5	52.6	98.1	107.6	73.3	56.2	32.1
17. Pcim	35.4	50.8	50.0	47.4	35.8	26.5	20.3
18. Polanka Haller	54.3	48.2	86.6	85.9	63.5	44.2	41.4

Discussion

Ozone has been recognized as a major phytotoxic constituent of urban and suburban air pollution (Stanners and Bourdeau 1995). Concentrations of ozone at average and high latitudes of the northern hemisphere doubled during the last century (Anfossi and others 1991, Volz and Kley 1988). Over the last decade an increase in ozone concentration has also been observed in the countries of north-eastern Europe. For example, rural Lithuania has been affected by the long-range transport from central Europe (Girgzdiene 1995).

Ozone causes the reduction of yield in crop species in North America and Europe (Fuhrer and Ascherman 1994), and these effects become more apparent as the phytotoxic concentrations of ozone increase in the rural areas (Colbeck and Mackenzie 1994). Ozone is also one of the factors causing recent forest decline on these continents (Ashmore and others 1985, Chevone and Linzon 1988).

Studies made in the Krakow Province found that the concentration of ozone in this area is markedly higher from the levels generally accepted as natural (Guderian 1985). During summer the concentrations of ozone in the area surrounding Krakow are higher than in unpolluted areas of central Europe (Girgzdiene 1991, Hakoala and others 1991), and they are similar to the ozone concentrations recorded from western European countries (Ballaman 1993, Dovland

Figure 5 — Spatial ozone distribution (μg/m³) obtained from passive samplers in the Krakow Province for the period 27 June to 17 July 1995.

Figure 6 — Spatial ozone distribution (μg/m³) obtained from passive samplers in the Krakow Province for the period 17 July to 7 August 1995.

Figure 7 — Spatial ozone distribution (μg/m³) obtained from passive samplers in the Krakow Province for the period 21 September to 16 October 1995.

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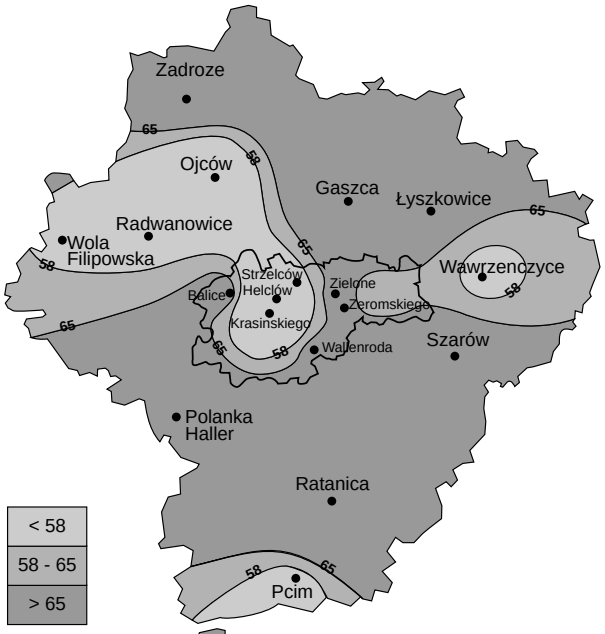

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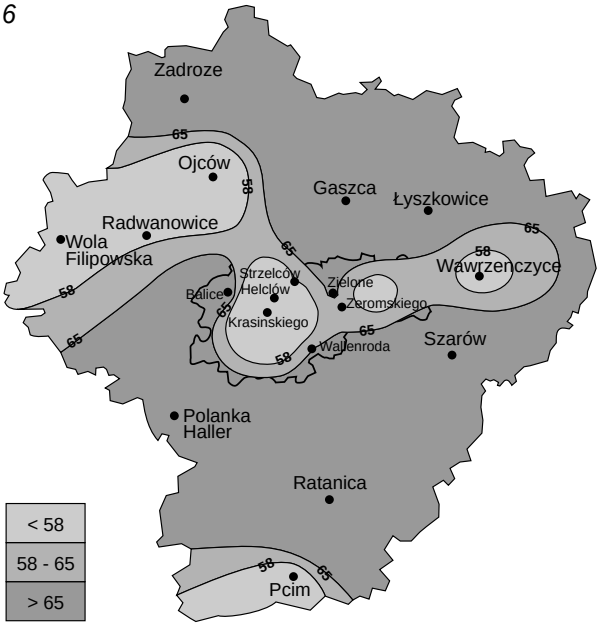

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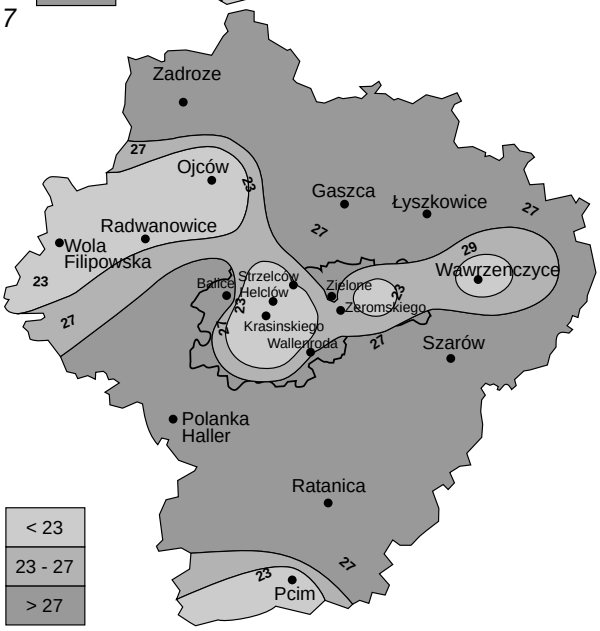

1987, Proyou and others 1991). The highest monthly ozone concentration in southern Poland occurred in July and in the beginning of August. In Upper Silesia (80 km west of Krakow) the ozone concentration recorded in 1994 was similar to the values recorded in Krakow and were higher than the values recorded from these areas in 1995 (Godzik and others 1994, Godzik and Grodzinska 1996). Although the highest concentration of ozone occurred in the city of Katowice, a high concentration was not recorded at that time in other sites in Silesia. This was probably caused by the presence of appropriately high concentrations of ozone precursors in the air (Godzik and others 1995). Ozone concentrations within the Krakow Province differed: higher concentrations were recorded in the northern and southern sites that are at higher elevations. In these localities, in contrast to the urban areas, small changes in the ozone concentration over 24 hours have been observed (absence of zero ozone concentration in the night) (Godzik and others 1995). It is impossible to give definite reasons for the occurrence of the highest ozone concentration in the northern and southern parts of the Krakow Province because data are not available on the proportion of nitrogen oxides and hydrocarbon or sun exposure regimes for the study sites.

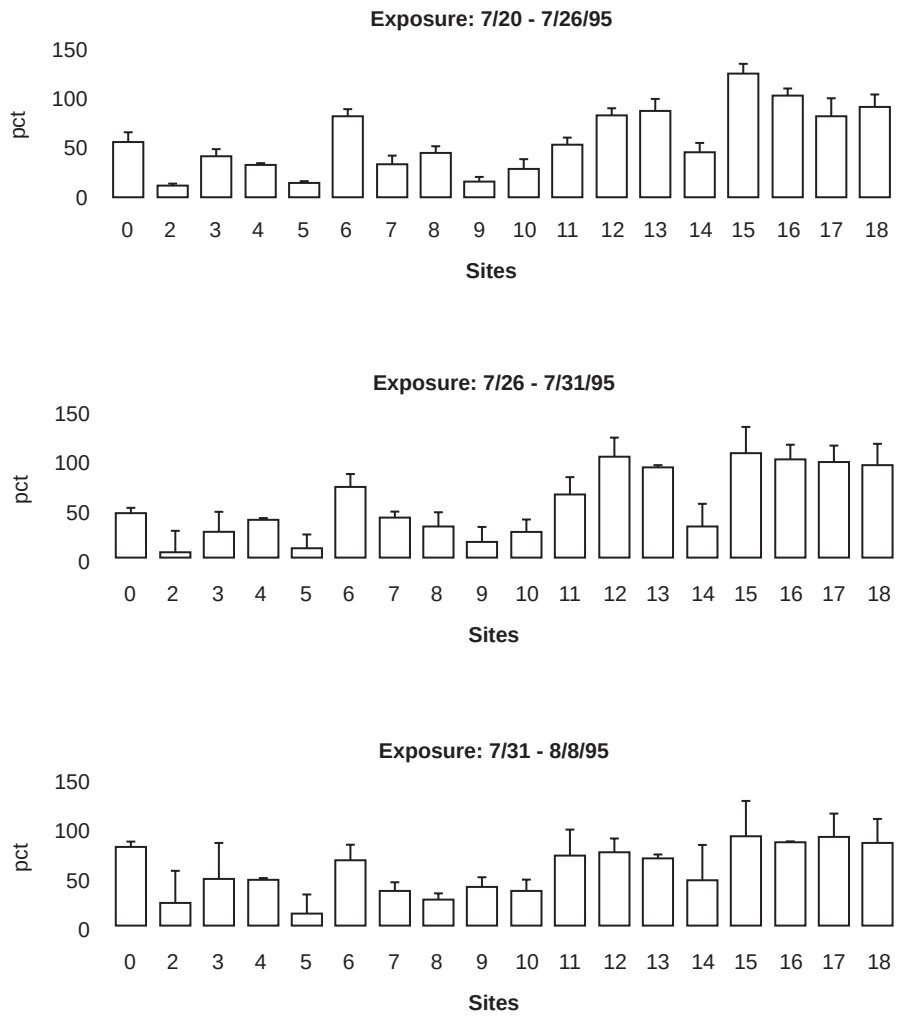

Figure 8 — Leaf injury to bioindicator plants (Bel-W3 cultivar of tobacco) exposed in Krakow (sites 1-7) and outside the city (sites 8-18) in the period 07/20-08/08/1995. The amount of injury is presented as the mean sum for four evaluated leaves from each of the four plants; data accumulated for subsequent exposition weeks. Research sites have been labelled as in *table 3*.

The ozone concentrations in the Krakow Province were high enough to cause leaf injury of the ozone-sensitive variety of tobacco (Bytnerowicz and others 1993, Spellerberg 1991). Similar concentrations in such amounts may also cause injuries in other plant species (Chappelka and Chevone 1992, Runeckles and Chevone 1992).

In the Krakow Province the levels of other pollutants are high, including sulfur dioxide, particulate pollution heavily contaminated with heavy metals, etc. These pollutants originate either from the local sources or from the transboundary transport (Nowicki 1993). Synergistic effects of these compounds and ozone may result higher damage of plants than by single pollutants (Guderian 1985). The results of ozone measurement concentration in southern Poland (Upper Silesia, Krakow region) and evaluation injury of the test-plants have shown that the co-occurence of elevated ozone concentration and high physiological potential of native plants can cause their injury. Ozone injury to native plants in Beskid Sląski of southern Poland has been observed but has not been verified in controlled ozone exposures (Godzik and others 1995). Similarly, the occurrence of ozone injury described in the literature has been recorded in the area surrounding Krakow. Among the affected species are: *Ulmus scabra, Ulmus laevis, Sambucus racemosa, Rubus hirtus*.[4] However, the above observations need to be confirmed in the controlled ozone exposures.

The number of motor vehicles has rapidly increased since 1990 in the Krakow region (Turzanski 1994) and has resulted in increased emission of nitrogen oxides and hydrocarbon. These compounds, known as ozone precursors, can cause further increase in ozone concentrations in the future.

[4] Unpublished data on file at the Polish Academy of Sciences.

Acknowledgments

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Ozone in the Upper Silesia Region — Concentration and Effects on Plants[1]

Stefan Godzik[2]

Abstract

*In the Beskidy Mountains at Brenna, Poland, and several other locations in the Katowice administrative district, plants were used as bioindicators to determine ozone concentration measurements in 1994 and 1995. Results showed that ozone is the only gaseous air pollutant significantly exceeding the permissible concentrations and causing foliar injury to both test and some native plant species. For all of the tests using plants, the most severe ozone injuries, were found for the Brenna location. The first signs of leaf injury were found on beans (*Phaseolus vulgaris *L.). Tobacco (*Nicotiana tabacum *L.) Bel W3 also showed leaf injury at this time. Less sensitive plants included the tomato (*Lycopersicum esculentum*) and soy bean* (Glycina max). *More severe injuries to the same plant species* (cultivars) *were not found with increased elevation. In the Brenna region only, foliar injury was also found on field-grown plant species:* Achillea *sp.,* Asceplias syriaca *L.,* Impatiens parviflora *DC.,* Sambucus nigra *L., S.* racemosa *L.,* Fagus silvatica *L.,* Prunus avium *L., and* Rubus *sp. The mean ozone concentrations at Brenna were 85.2 and 77.4 $\mu g/m^3$ in the summers of 1994 and 1995, respectively. For the locations in the Katowice area, mean concentrations were 50.3 and 44.5 $\mu g/m^3$. AOT40 calculated on a 24- hour or day-hours (9.00 - 17.00) basis exceeded the critical values of 5,300 ppb.h and 1 ppm.h recommended by the United Nations ECE for crops and forest tree species. Thus, on the basis of these results ozone is a contributing factor to poor forest health conditions in the Upper Silesia Region of Poland.*

Introduction

Leaf injury to native or introduced plant species, such as horse-chestnut (*Aesculus hippocastanum* L.), rose (*Rosa* sp.), and some others, was caused by air pollutants of photochemical origin in the vicinity of coke ovens in the mid-1960's (Godzik and Piskornik 1966). However, injury of tobacco (*Nicotine tabacum* L.) Bel W3 was not the same as that caused by ozone. Nonetheless, air pollutants of photochemical origin can cause leaf injury in locations where sulfur dioxide, nitrogen oxides, and particulate matter were seen as the only "real" problem to the environment.

Exposure started later for the Bel W3 and Bel B tobacco, and for radish (*Raphanus sativa* cv Cherry Bella) with and without ethylene diurea (EDU) treatments in the mountain region (Godzik and others 1991). Injuries to Bel W3 and differences in the response of radish with no EDU treatments have shown that ozone has caused adverse effects, including injury to test plants. Injury to some native or introduced field grown plant species has been observed simultaneously.

A 6-month series of measurements using passive samplers was performed in 1990 for the Rozdroze Izerskie (Sudety Mountains) and Brenna area (Beskidy Mountains). These measurements were completed by 1-week continuous measurements of ozone in these two locations.[3] Data have shown that concentrations are similar to those found in other European countries.

First, short-term ozone concentration measurements using ozone analyzers were done in Brenna in 1990.[3] The highest 1-hour mean concentration was 110 $\mu g/m^3$. Next, more regular measurements of ozone concentration and experiments with several test plants were done. In 1994 an extensive air pollution monitoring program started in the Katowice area by using test plants as bioindicators, including tobacco (*Nicotiana tabacum* L.), Bel W3 and Bel B, bean (*Phaseolus vulgaris* L. cv Lit), clover (*Trifolium subtereanum* cv crimsoni), tomato (*Lycopersicum esculentum* L. cv Tiny Tim), and soy bean (*Glycina max*). These were grown in containers with standardized soil and in the local soil (Godzik and others 1995, 1996 a, b).

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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[3] Unpublished data supplied by the author.

The objective of this study was to assess the effect of ambient ozone concentration in the Upper Silesia Region of Poland by using plants as bioindicators exposed at locations that differed in climate and air pollution.

Materials and Methods

In the growing seasons of 1994 and 1995 ozone concentrations were measured by using analyzers, including Environics series 300, MLU and Thermo Environmental.[4] Before each measuring season analyzers were calibrated. Seeds of all but one test plant (tobacco) were provided by the International Cooperative Program (ICP) Crops Coordination Center of Nottingham University, England. Seeds of Bel W3 and Bel B tobacco were provided by the University of Massachusetts.

Clover and bean in standardized soil and containers were exposed in Brenna, Katowice, Kuznia Nieborowicka, and Slawkow. Before 1994 the number of test plants was limited to clover and tobacco only. Radish, treated and non-treated with EDU, was used during two growing seasons.

Bean, clover, and Bel W3 and Bel B tobacco were grown in standardized soil, in self-watering containers. Procedures were based on guidelines elaborated by the ICP Crops, under the UN ECE Convention on Long Range Transboundry Air Pollution. At Brenna and Katowice, beans, tomato (*Lycopersicum esculentum* L.), and soy bean (*Glycina max*) were also grown in local soils. Foliar injury of both test and native plant species were compared with those given in the literature and photographs[5] (Manning and Feder 1980, Sanders and Benton 1995).

Results and Discussion

Mean concentration of ozone for Brenna in 1994 was 85.2 $\mu g/m^3$, and for 1995 was 77.4 $\mu g/m^3$ *(table 1)*. Mean concentrations for the same period for Katowice region were 50.3 $\mu g/m^3$ and 44.5 $\mu g/m^3$, respectively. The highest ozone concentration was measured in Katowice, June 1994 — 232 $\mu g/m^3$. Other locations did not show high concentrations (Godzik and others 1995, 1996) compared to the data for the years 1994 and 1995. According to the national standards, this finding suggests

Table 1 *– Concentration of sulfur dioxide, nitrogen dioxide, and ozone at some monitoring stations in the Upper Silesia Region, Poland.* [1]

Pollutant and year	Katowics	Kuznia Nieb.	Slawkow	Brenna	Standard value D_{30} D_{24} D_a
Nitrogen dioxide					
					500
1994	50.1	22.7	12.1	3.1	150
1995	23.5	17.0	16.8	5.3	50
Sulfur dioxide					
					600
1994	36.7	14.1	27.8	6.5	200
1995	27.6	10.4	21.5	24.0	32
Ozone					
					100
1994	45.9	50.3	45.8	85.2	30
1995	47.3	44.5	-	77.4	-

[1] Mean values for the period: May 5 – Oct. 31, 1994 and May 5 – Aug. 31, 1995 in comparision to national standards ($\mu g/m^3$). Calculations are based on 24-h mean values (Godzik and others 1996b).

[4] Mention of trade names or products is for information only and does not imply endorsement by the U.S. Department of Agriculture.

[5] Unpublished data supplied by the author.

that ozone concentrations found in the Brenna region and the Katowice area are high enough to cause foliar injury to plants (Council Directive 1992).

In the Brenna and Katowice locations, ozone concentrations set by the UN ECE convention as critical levels were exceeded (Ashmore 1994, Benton and others 1995, Fuhrer and Achermann 1994). These levels were defined as accumulated dose over the threshold of 40 ppb (AOT40), which is the sum of hours during the day of ozone concentration above 40 ppb (Benton and others 1995, Fuhrer and Achermann 1994, and Legge and others 1995).

The patterns of daily ozone concentration differ at locations in the Katowice area and Brenna. Peak day values are higher for the Katowice area, but mean concentrations are higher at the Brenna site. This difference is caused by higher ozone concentrations at night at the Brenna site compared to the Katowice area measurements. On most occasions the concentration at Brenna during the night are in the range of 15 µg/m^3 *(figs. 1-2, table 1)*. Both for 1994 and 1995 the highest AOT values are for Brenna, not for Katowice, where the highest 1-hour mean values were measured *(figs. 3,4)*.

Foliar injury of the test plants—clover, tobacco, bean, soy bean, tomato, and native plant species—corresponded to that described in the literature (Arndt and others 1987, Manning and Feder 1980, Sanders and Benton, 1995). The first signs of injury were found at the Brenna location and were noted for Bel W3 and bean in both local soil and standardized soil. They appeared after about 2 weeks of

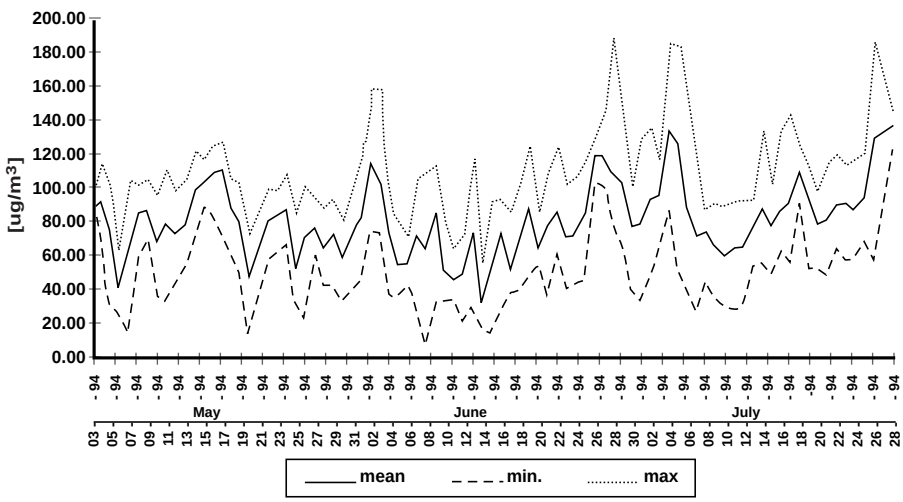

Figure 1 — Daily pattern of ozone concentrations at the Brenna, Poland monitoring station, summer 1994. (1µg/m^3 O_3= 0.51 ppb O_3 at 25ºC, 760 mm of Hg.)

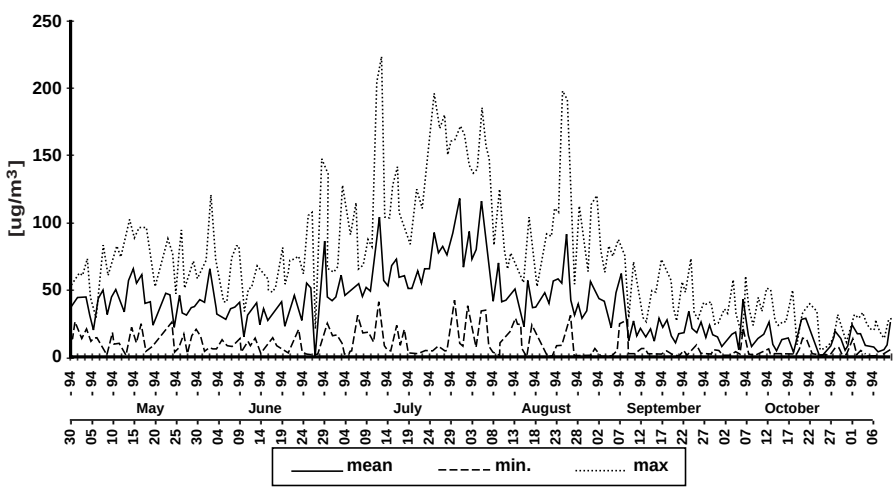

Figure 2 — Daily pattern of ozone concentrations at the monitoring station of the Institute for Ecology of Industrial Areas, Katowice, 1994.

exposure. Intensity of injuries increased during the growing season. In the Brenna location all test plants—species or cultivars—showed foliar injury. Statistically significant differences were not found in dry mass production of test plants treated and non-treated with EDU and grown in standardized soil at different locations. The number and weight of pods of bean cultivar Lit grown in local soil in both Brenna and Katowice was lower than cultivars of Stella and Grophy. Crops of pods of beans cultivars Stella and Grophy were very similar (Godzik and others 1996b).

Differences were found in plant responses in 1994 and 1995 between elevations in the Brenna region and the Katowice area. At Salmopol less severe injuries were observed in 1995. The opposite was found for the Katowice area. The reason for these differences was undetermined. These 2 years differed in climatic conditions, with air temperatures higher for 1994; but they also differed in concentrations of other air pollutants (Godzik and others 1996a, 1996b; table 1). Air temperatures close to and above 30 °C at the Brenna location were measured in June and July in 1994 for several days. A similar phenomenon was not found in the same period of 1995. Temperatures in the Katowice area for the same days were higher by a few degrees Celsius (Godzik and others 1996a, 1996b).

Results of other investigations performed in the Brenna region in forests suggest that climatic parameters are important factors to include when determining severity

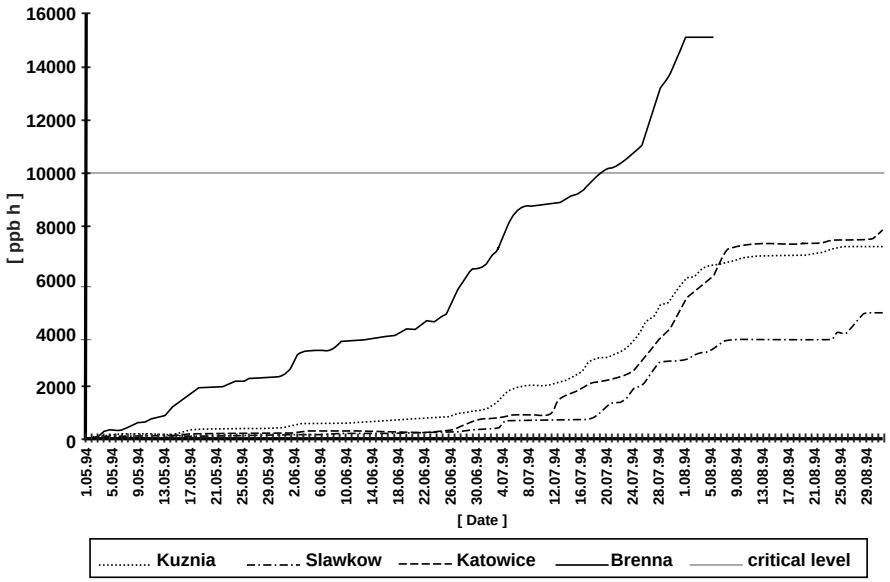

Figure 3 — Accumulated dose over the threshold of 40 ppb (AOT40) for locations in Brenna and Katowice district. Values for the daily hours (9:00 – 17:00) were used for calculations.

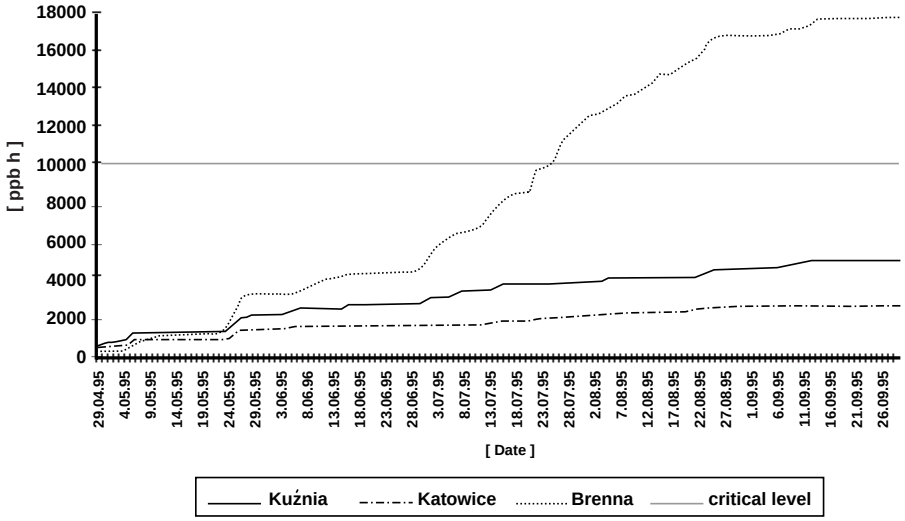

Figure 4 — Accumulated dose over the threshold of 40 ppb (AOT40) for Brenna and Katowice growing season, 1995. Twenty-four hour values were used for calculations.

of injury (Godzik and others, this volume). Ozone-like foliar injuries have been frequently found on field-grown plant species: *Achillea* sp., *Asceplias syriaca* L., *Impatiens parviflora* DC., *Sambucus nigra* L., S. *racemosa* L., *Fagus silvatic*a L., *Prunus avium* L., and *Rubus* sp. Except *Asceplais syriaca* grown in the Katowice area, all other species mentioned have shown signs of injury in the Brenna region only. These findings have not been confirmed by controlled O_3 fumigations.

Foliar injury on all test plants appeared in late June and increased during the growing season. Foliar injury on *Impatiens parviflora, Achillea* sp., *Asclepias syriaca* and *Rubus* sp. were noted from mid-July. On other tree or shrub foliage, signs of injury were found in August.

Conclusion

Severe ozone injury has been observed for tobacco Bel W3 and beans grown in the Brenna and Salmopol locations. Injuries of other plant species such as soy bean and tomato were less severe. Ozone injury to the same plant species grown in the Upper Silesia Industrial Region (USIR) were observed less frequently and differed between 1994 and 1995.

Ozone concentration during the day was the highest in the USIR, but the mean concentration was the highest in the Brenna location. The accumulated dose over the threshold of 40 ppb (AOT40) was also much higher at the Brenna. Ozone of photochemical origin is likely the only air pollutant in southern Poland causing foliar injury to both test and some field grown native or introduced plants, including some broad leaf tree species.

The effects of climatic parameters and other air pollutants, such as sulfur dioxide and nitrogen dioxide, to plant injury are not yet known. The role of air pollutants on crop of test plants used in our experiments remains uncertain. Data from experiments performed were not consistent with ozone concentration. For a few locations in the Katowice area, dry mass production was the highest where $N\text{-}NH_4$ in bulk deposition was found to be the highest. However, more data are needed to confirm these results.

Acknowledgments

Investigations on ozone effects on plants performed at the Beskidy site—Brenna and Salmopol—were funded by the National Science Committee, Poland, by grant 4 S401 014 05, and the USDA Foreign Agriculture Service, International Cooperation and Development grant "Effects of Atmospheric Deposition and Climate Change on Forest Ecosystems in Poland and the United States."

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Ozone Concentrations at the Bulgarian Govedartsi Ecosystem Site in Early Summer of 1994 and 1995[1]

Evgeny Donev,[2] Karl Zeller,[3] and Christo Bojinov[4]

Abstract

Ozone measurements are presented from the Bulgarian Govedartsi ecosystem study site, Ovnarsko No. 3. The site is located on the south slope of the Govedartsi Valley in the northwestern part of the Rila Mountain area. The Rila is the highest mountain in the Balkan peninsula and is representative of rural conditions in that part of Europe. The experimental site, Ovnarsko, is situated at 1,600 m above sea level and about 10 km west of the village of Govedartsi. There are no anthropogenic sources for ozone precursors for at least 30 km in all directions. Ozone concentrations within the Govedartsi Valley are strongly influenced by thermally driven local winds from late spring to summer that occur during high pressure synoptic weather conditions. In these cases a thin stable cold layer of downslope wind is experienced during the night, separating surface air from the higher boundary layer air. This helps the reduction of ozone concentrations from dry deposition measured at the Ovnarsko site. During the 1994 and 1995 measurement periods no high ozone concentration episodes were detected. This may be because of either an absence of significant anthropogenic sources of ozone precursors in this part of the Balkan peninsula or to the shielding of the Valley during high pressure conditions. In general, ozone concentrations at Ovnarsko were below 35 ppb.

Introduction

In September 1991, an air-pollution study was launched at the Govedartsi in the highest mountain range in the Balkans. The site—designated as Ovnarsko No. 3—is located on the south side of the Govedartsi Valley on the northwestern part of the Rila Mountains (Zeller 1991). The Valley is bordered on the south and west by high ridges (~2,600 m), and on the north by a lower and more fractional ridge (highest point ~1,800 m). The Valley opens to the northeast with a narrow mountain pass, through which the Iskar river flows toward the Samokov area. The experimental site Ovnarsko is situated at 1,600 m above sea level and about 10 km west of the village of Govedartsi, which is on the less steep part of the northern slope of Rila Mountain. The terrain is higher in the western part and lower in the central part of Govedartsi Valley (*fig.1*). The greater part of Rila mountain lies to the south of the Valley area.

There are no anthropogenic sources for ozone precursors for at least 50 km in the south direction and for about 30 km in the west direction where the mountain ridge is a high obstacle for west-to-east transport of air pollutants. Weekly average measurements of SO_4^{2-}, NO_3^-, and some metals taken in 1991 (Zeller and others 1992) and 1992 demonstrated increasing air pollutant concentrations after each well-marked western air outbreak (strong winds accompanied by rain during cyclonic synoptic weather). However, during high-pressure weather systems (low wind and clear sky), decreased SO_4^{2-} and NO_3^- concentrations were observed. Under the latter weather conditions mountain-valley gradient wind flows predominate, allowing local natural sources of trace gases to determine the measured air quality. During such periods the Govedartsi Valley is probably screened from anthropogenic air pollutants from outside. In these cases the mountain ridges, surrounding the Valley, are mostly above the height of the planetary boundary layer causing the measured wind speeds at the experimental site to be low and wind directions to follow a diel cycle of mountain-valley circulation.

This paper presents data that the Ovnarsko experimental site is not impacted by ozone or its precursors of anthropogenic origin under high-pressure weather. This phenomenon is associated with so called episodes of high ozone concentrations

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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in rural areas in other regions of the world (Cox and others 1975, Logan 1985). Many studies (Attmannspacher and others 1984, Guicherit and van Dop 1977, Mueller 1994, Prior and others 1981, Vukovich and others 1977) relate the high ozone levels to the presence of warm and slowly moving synoptic-scale high pressure systems, which was not observed at the Ovnarsko site.

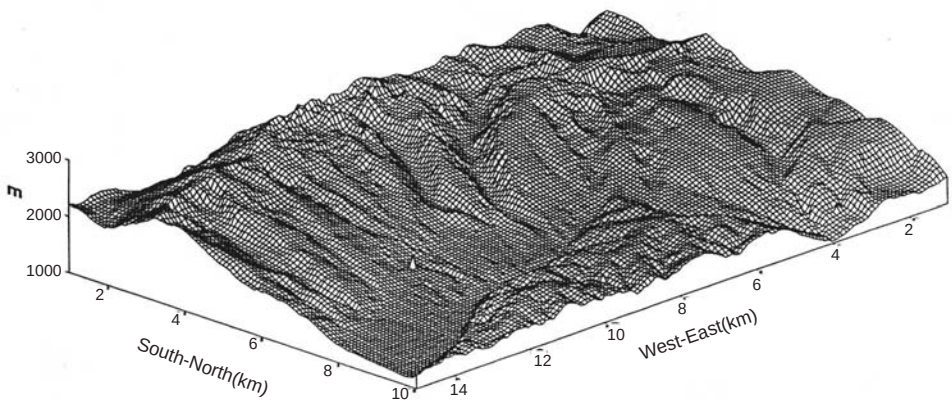


Figure 1 — The Govedartsi Valley from the northeast on a 100 m grid. The x axis, west-east, is 15 km and the y axis, south-north, is 10 km.

Data and Instrumentation

The reason for monitoring ozone at the Ovnarsko site was to determine the extent to which upwind anthropogenic sources influenced Govedartsi Valley ozone concentrations and whether the Ovnarsko site was representative as a background station for this part of Europe. The site location allows for separating synoptic from mesoscale meteorological wind regimes, isolating source regions for measured air pollutants. Because of difficult site access during winter and limited technical support after late summer, ozone concentrations were only measured from late spring to summer: May 28 to July 9, 1994 (Julian Days 148 to 190) and June 12 to July 30, 1995 (Julian Days 163 to 212). Wind speed and direction, air temperature and humidity at 5 m height, total solar radiation, wetness, precipitation, soil temperature and soil heat flux were measured through autumn. Hourly mean and standard deviations of the measured quantities were recorded with a Campbell Scientific 21x data logger (*table 1*).[5] Ozone was sampled through a teflon filter at 2 m height. The ozone instrument was calibrated in the United States before shipment to Bulgaria. It appeared to function normally; however, a post calibration was not made. Because of the inherent stability of the ultraviolet adsorption technique, the ozone concentrations were within ±5 ppb. The teflon filter used to remove foreign objects in the sample air was changed approximately every 15 days.

Table 1 *— Parameters measured hourly at the Ovnarsko site.*

Sensor	Parameter	Units
Teco 49	O_3	ppb
Metone	Wind speed	m s^{-1}
Metone	Wind direction	degree
Eppley	Solar radiation	watt m^{-2}
MRI Inc.	Precipitation	mm
Metone	Temperature	°C
Metone	Relative humidity	pct
Rad. energy	Soil heat flux	watt m^{-2}
thermocouple	Soil temperature	°C
thermocouple	Soil temp Gradient	°C
Campbell Sc.	Leaf wetness	wet/dry

[5] Mention of trade names or products is for information only and does not imply endorsement by the U.S. Department of Agriculture.

Results and Discussion

A difference in diurnal ozone concentration cycle patterns was observed after sorting the data by meteorological phenomena. Air laden with ozone and ozone precursors is frequently transported from urban areas to cleaner mountain and rural areas during periods with high atmospheric pressure and low wind speeds (Cox and others 1975, Vukovich and others 1977, Wolff and others 1977). The days with such weather are categorized: "days with clear skies and local winds" for this paper. On the other hand, cyclonic weather conditions with strong winds, and turbulent boundary layer air cause intense vertical mixing and potential dilution of polluted air. In the Govedartsi Valley, however, this same vertical mixing appears to cause downward ozone transport from higher ozone-laden layers of the troposphere. In remote areas higher mean ozone concentrations are often experienced under these more turbulent conditions when air, transported from distant upwind sources, is mixed downward (Zeller and others 1977). For Ovnarsko the data were separated into the two types of weather by using wind speed and direction and solar radiation as the main criteria. Precipitation, air, and soil temperatures were used as additional criteria when necessary.

Meteorology

The hourly and 24-hour mean wind direction and 24-hour wind speed sums for the 1994 and 1995 measurement periods were obtained through summation of mean values for each hour (*fig.2a,b*). There are two distinct meteorological regimes affecting wind flow as measured at Ovnarsko: locally generated winds and synoptic winds. During local wind days regional synoptic pressure gradients are weak and local mountain-valley circulation predominates. On such days the wind direction follows

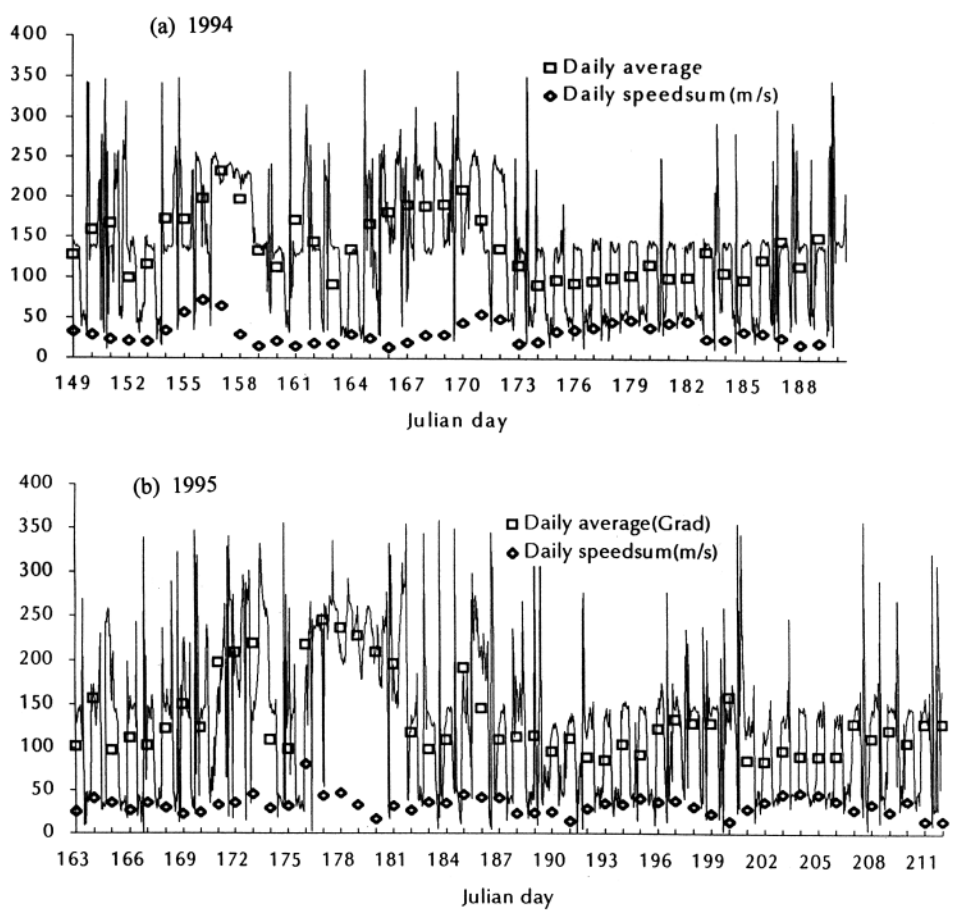


Figure 2 a, b — Wind direction for each hour (line), mean wind direction (squares), and velocity for each day (diamonds) for the two measurement periods in 1994 (a) and 1995 (b).

a typical diurnal cycle: southeast at night (mountain catabatic winds) and northeast during the day (upslope winds). On days influenced by synoptic weather the diurnal cycle of mean wind direction does not follow a predictable course but varies from south to west demonstrating the effect of upwind regional air intruding into the Govedartsi Valley. The strongest of these winds are from southwest and west during such days and tending to return to southeast at night demonstrating the surface layer decoupling from upper synoptic winds on most but not all synoptic nights.

The total solar radiation was calculated for each hour in addition to the daily mean (*fig. 3a,b*). Days with high solar radiation (indicating few or no clouds) are typically days when local winds occur ("days with clear skies and local winds") and vice versa: lower solar radiation (cloudy days) are the days with stronger westerly winds. The 1994 study days determined as "days with clear skies and local winds" are: Julian days 152-154 and 174-182, 12 days out of the 42-day measurement session. For 1995 the study days are the Julian days 163, 183, 184, 194, 195 and 202-207—11 days out of the 50-day measurement session.

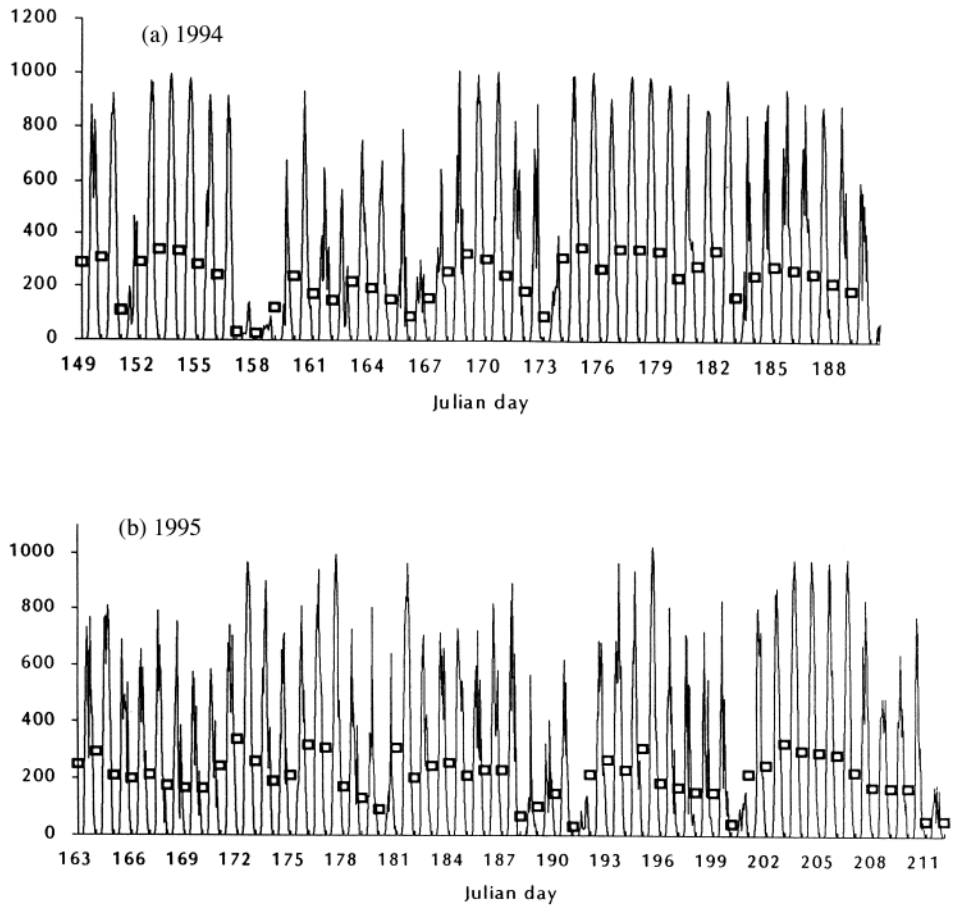
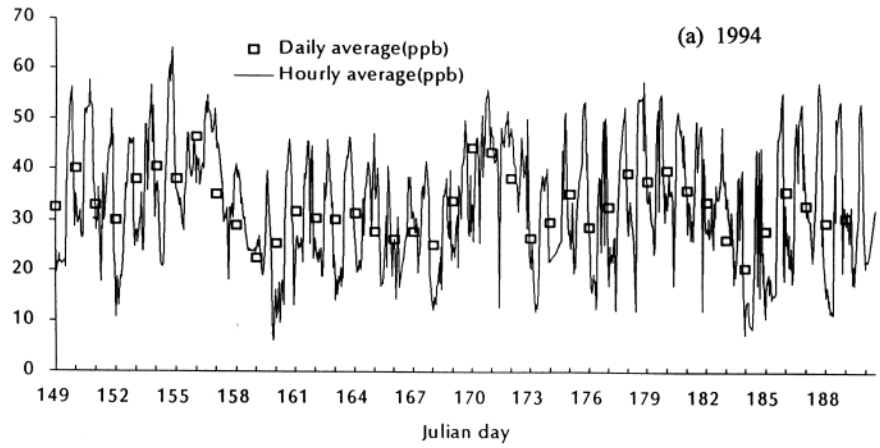


Figure 3 a, b — Total solar radiation (watts/m^2) for each hour (line) and daily mean values (squares) for the two measurement periods in 1994 (a) and 1995 (b).

Ozone Concentration

Hourly and daily mean ozone concentrations from the 1994 and 1995 measurement periods were plotted (*fig. 4a,b*). These values, adjusted to mean sea level, rarely exceeded 35 ppb during the day and drop to 15 to 20 ppb at night. This diurnal swing in concentration is similar to summertime results taken in a spruce-fir forest in Snowy Range, Wyoming above 3,000 m elevation at 3 m above ground level but below canopy height (Wooldridge and others 1994, 1995). Ozone concentrations taken at 30 m above ground (13 m above canopy height) at the same Wyoming site only vary 2 to 3 ppb from day to night (Zeller 1995). At Ovnarsko, higher nighttime ozone concentrations were measured during the synoptic wind days when air from Central or Southern Europe apparently entered the Valley (e.g., Julian days 156

and 170 from 1994 and days 176 and 186 in 1995). There were no high ozone concentration episodes during either classification period in 1994 and 1995. With the exception of Julian days 154 in 1995 and 195 in 1996, periods when outer air masses penetrated into the Valley (i.e., days without clear skies and local winds) tended to be associated with similar daily maximum ozone concentration; however, the nighttime minimums on such days were much higher, causing larger daily averages. These data suggest that strong exchange of air within the whole boundary layer during these periods enhanced vertical ozone transport, advected from distant upwind sources, and maintained an above ridge-height ozone source preventing the Valley from experiencing lower nighttime ozone concentrations. The lower nighttime minimums during days with clear skies and local winds were from ozone deposition to surfaces and the lack of a transport mechanism to vertically transport the upper ozone to maintain higher surface concentrations. This explains the strong influence of the local winds (mountain-valley circulation) on the diurnal cycle of ozone concentrations in the Govedartsi ecosystem.



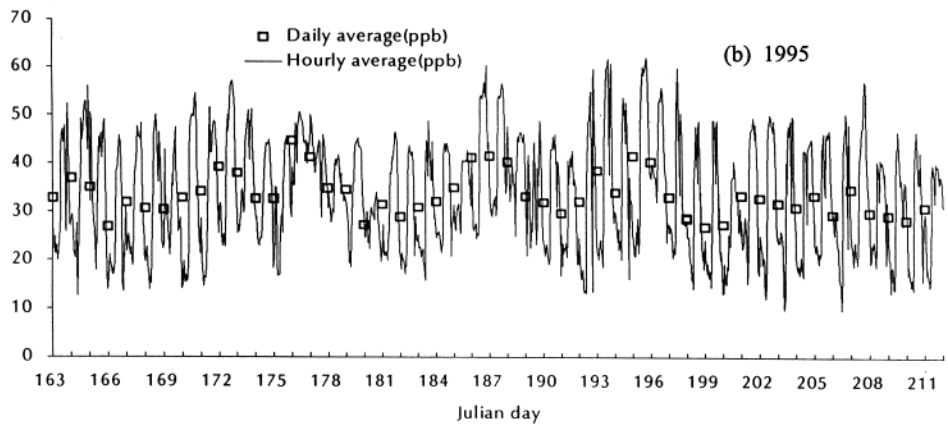


Figure 4 a, b — Hourly mean ozone concentration (line) and daily mean values (squares) for the two measurement periods in 1994 (a) and 1995 (b).

The existence of well-manifested thermally driven local wind system in the Valley around the Ovnarsko site gives us the opportunity to compare the influence of upslope and downslope winds on the diurnal variations of ozone concentration. The mean diurnal cycle of ozone concentration, temperature, wind speed, and direction (degree/10) was calculated for the chosen days with clear skies and local winds. The thermally driven mountain-valley circulation is clearly seen especially during the night when the cold downslope/down valley winds along the north-northwest slope are constantly east-southeast to east in direction from 22:00 local standard time until 8:00 LST. Northeast upslope winds begin after 8:00 LST until 20:00 to 21:00 LST. Diurnal ozone concentrations show a typical diurnal cycle for the clear skies days in rural areas (Bottenheim and others 1994, Meagher and others

1987, Prevot and others 1993): minimum values in the morning until upslope conditions take over, and maximum values in the afternoon. During a typical afternoon the east to northeast wind passes through the agrarian Govedartsi village 10 km E (upwind at this time) of Ovnarsko. There is little automobile traffic and few farm vehicles that are not horse-drawn. It is possible some anthropogenic ozone precursors come from Samokov, a larger city about 15 km farther east, however, in general there is a lack of so called high ozone concentration episodes, frequently experienced in the eastern part of the USA and in Western Europe under similar weather conditions (Aneja and Li 1992, Aneja and others 1994, Davies and others 1992, Mueller 1994, Trainer and others 1993).

In mountainous terrain, Brother and Gydax (1985) found an increase of ozone during the day caused by upslope air transport with photochemical ozone production. During the night ozone concentrations decreased on the slope due to dry deposition. In another case Prevot and others (1993) concluded that thermally driven winds defined the origin of air masses of the Reuss Valley in the summer during high pressure systems. During the night the thickness of the downslope wind layer experiencing efficient dry ozone deposition seemed to be very thin on the slopes. They found that ozone concentrations at 2 m height were significantly lower than those at 15 m height during the night similar to Wooldridge and others (1994, 1995). The nocturnal concentrations measured at Ovnarsko at the same 2 m height during downslope winds were rather low (*fig. 5a, b*) for a site at 1,600 m elevation. The low nighttime Ovnarsko ozone values may be a result of dry deposition of ozone to surfaces, including reactions with soil emitted NO (Brother and Gydax 1985) within the cold stable downslope flow. The downslope flow concentrated within a thin layer near the earth will not interact with the air mass above, where ozone concentration is most likely higher and not effected by dry deposition mechanisms. Mueller (1994) showed a clear tendency for diurnal ozone amplitudes to diminish at rural mountain sites as site elevation increases. For some mountain experimental sites a tendency for a reversed diurnal cycle of ozone

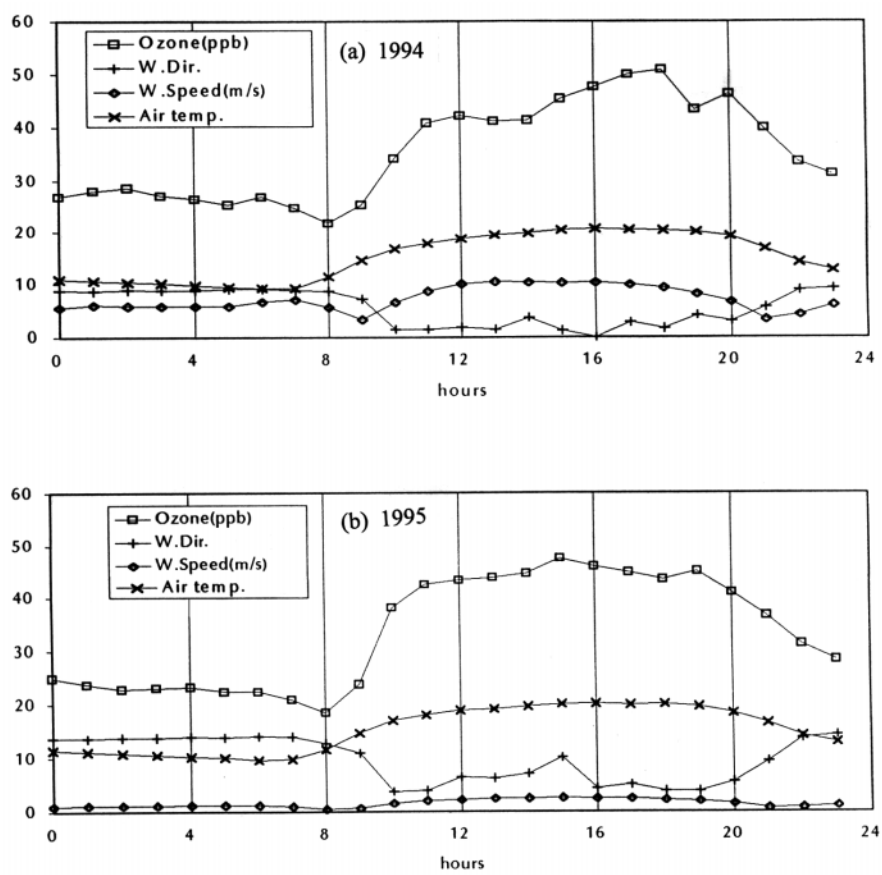


Figure 5 a, b — Mean diurnal cycle of ozone concentration (squares: parts per billion), wind speed (diamonds: m/s), wind direction (+: degree x 10^{-1}), and air temperature (x: °C) for the two measurement periods in 1994 (a) and 1995 (b).

concentration has been detected when maximum values are measured during the night (Aneja and Li 1992). This is not the case for sites under the influence of a mountain-valley circulation.

Apparently ozone concentrations are lower in this rural part of Bulgaria compared to more populated areas. For instance, within southeast Sofia during the end of the summer 1-m height, 10-day daily mean ozone concentrations were between 40 and 50 ppb, with maximum values reaching 70 to 80 ppb. These Sofia values are similar to those in metropolitan Denver, Colorado (Zeller 1995). Ozone values found in forested areas of Poland are predominantly above 60 ppb and often exceed 80 ppb (Bytnerowicz and others 1993), unlike the Ovnarsko results.

Conclusion

Ozone concentrations within the Govedartsi Valley are strongly influenced by thermally driven local winds from late spring to summer during high pressure systems. In these cases, a thin stable cold layer of downslope wind is experienced during the night, which separates surface air from the higher boundary layer air causing reduction of ozone concentrations from dry deposition measured at the Ovnarsko site. During the two measurement periods, no high ozone concentration episodes were detected. This condition may be a result of the absence of significant anthropogenic sources of ozone and its precursors in this part of the Balkan peninsula or from the shielding of the Valley during high pressure conditions where the surrounding mountain ridges are above the boundary layer height and the air in the valley flows as thermally-driven local circulation.

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Ozone and Modeled Stomatal Conductance at a High Elevation Subalpine Site in Southeastern Wyoming[1]

Robert C. Musselman, Karl F. Zeller, and Nedialko T. Nikolov[2]

Abstract

Ozone concentrations have been monitored at the Glacier Lakes Ecosystem Experiment Site (GLEES) in the Snowy Range of the Medicine Bow Mountains 55 km west of Laramie, Wyoming, USA. The site is located at 3,186 m elevation in a large subalpine meadow of a mature subalpine forest near timberline. Continuous ozone and meteorological monitoring are a part of the GLEES research to determine the effects of atmospheric deposition on alpine and subalpine ecosystems. Ozone monitoring has shown specific summer and winter diel ozone distribution patterns, typical of remote, rural, high elevation sites. The data show relatively high background ozone concentrations with little diel variation in winter. Stratospheric intrusions contribute to the ground-level ozone concentrations in the spring. Summer ozone patterns show diurnal photochemical peaks and significant night time concentrations suggesting lack of scavenging. The relationship of modeled leaf stomatal conductances, boundary layer conductances, canopy conductances and photosynthesis rates to ambient ozone concentrations and measured ozone flux was examined by using regression analysis. Ozone concentrations were also compared with wind, radiation, temperature, and humidity conditions at the site by using regression analysis. The data provide information on ozone concentration and environmental conditions at a remote, high elevation site typical of many wilderness areas of the western United States.

Introduction

The Glacier Lakes Ecosystem Experiment Site (GLEES) is a 600-ha research site in southeastern Wyoming, where atmospheric deposition and its effects on terrestrial and aquatic ecosystem processes are being studied (Musselman 1994). The site is located at 3,100 to 3,400 m asl elevation in the Snowy Range of southeastern Wyoming, about 40 km west of Laramie. The research site is in the Medicine Bow National Forest at the alpine/subalpine ecotone. Much of the atmospheric monitoring is conducted at the Brooklyn Lake site, a forest stand dominated by Engelmann spruce (*Picea engelmannii* (Parry) Engelm.) and subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.). The Brooklyn site is occupied by patchy forest with large subalpine meadow openings.

This paper discusses the results of studies to determine the characteristics of ozone deposition in alpine and subalpine ecosystems in southeastern Wyoming.

Methods

The Brooklyn meteorological and air quality monitoring site at the GLEES is located in a small subalpine forest opening (30-m diameter). Electric power is available at the site. Standard meteorological measurements are monitored continuously at 10 m and 29 m above the ground and recorded on a data logger. Parameters measured at the site include wind speed and wind direction, soil and air temperature, radiation, wetness, and relative humidity. Precipitation is monitored at 2 m above the ground with an Alter shielded Belfort rain gauge.[3] Air quality measurements include ambient ozone and dry and wet deposition.

Dry deposition is monitored as part of two national networks, the National Dry Deposition Network (NDDN) Centennial site, and the Interagency Monitoring of Protected Visual Environments (IMPROVE) Brooklyn Lake site. The NDDN site is part of a United States Environmental Protection Agency-sponsored national network of dry deposition monitoring stations. Protocols for both dry deposition monitoring networks use a filter pack for collection of dry particles and gases, but

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside California.

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[3] Mention of trade names or products is for information only and does not imply endorsement by the U.S. Department of Agriculture.

they each conduct different analyses. The NDDN site is located in a larger opening about 120 m southwest of the Brooklyn meteorological monitoring. The IMPROVE monitor is located at the Brooklyn monitoring site meteorological tower.

Wet deposition is monitored as a part of the U.S. National Atmospheric Deposition Program (NADP) national network, site Brooklyn Lake, WY95, which monitors wet chemistry of precipitation. The NADP wet deposition station is also located at the Brooklyn Lake Monitoring site, about 80 m north of the NDDN site.

Ozone has been monitored at 3,180 m asl at the GLEES Brooklyn Lake NDDN monitoring site since 1989. Monitoring and calibrations follow standard NDDN protocols. The station monitors ozone continuously at 3-m height above the ground. Ozone flux measurements were also made at the site by using an eddy correlation system (Zeller and Hehn 1996).

The FORFLUX model (Nikolov 1995) was used to estimate canopy photosynthesis and stomatal CO_2 conductance by using meteorological (March to June 1990) data from the site. FORFLUX estimates canopy fluxes by numerical integration of LEAFC3 model (Nikolov and others 1995) over stand leaf area index. Leaf area index of 3.5 was used for this study. The modeled photosynthesis and CO_2 conductances were compared with values of ambient ozone using regression analyses. The relationship was examined between the modeled canopy conductance and (1) ambient ozone concentration or (2) ozone flux measurements. The ambient ozone concentration data were also compared to measured meteorological data by using regression analysis.

Results and Discussion

Typical ozone distribution patterns at the GLEES are characteristic of high elevation remote sites. These patterns show low diel variation in ozone concentration, a result of minimal daytime increase from photochemical production and minimal night time decrease due to scavenging. The site also shows evidence of stratospheric intrusion of ozone (Wooldridge and others 1996). Characteristics of the ozone deposition at the GLEES have been described elsewhere (Wooldridge and others 1996, Zeller and Hehn 1995). There are three main characteristics of the ozone distribution for this remote site. First, concentrations of ozone are relatively high at the site (*table 1*). Concentrations above 60 ppb occur frequently during the spring and summer (Wooldridge and others 1996). Concentrations above 50 ppb are common.

Table 1 *— 1990-1994 average monthly ozone concentration at the GLEES Brooklyn monitoring site (Wooldridge and others 1996).*

Month	*Mean ozone concentration, ppb*
January	44.6
February	47.5
March	50.5
April	53.9
May	54.6
June	49.8

A second characteristic is the diel variation of ozone concentration shows very little amplitude. Diel concentrations vary as little as 1 to 3 ppb in January and February (44 to 47 ppb). Diurnal variation and peak daily ozone concentration increase monthly from March (minimum 48 ppb, maximum 51 ppb) to May (minimum 51 ppb, maximum 58 ppb), with typical afternoon photochemical peaks. The diel pattern in May and June is more pronounced, typical of increased photochemical generation of ozone during the day and greater amount of ozone scavenging by nitrogen oxides and surfaces at night. This difference indicates the

importance of the lack of night time nitrogen oxide scavenging at this site during the winter. In addition, there is very little daily photochemical generation of ozone in winter. The lack of a diurnal peak is likely because of the limited amount of precursors, and the very low temperatures that are not conducive for chemical reactions. Temperatures at the site in January average less than -10 °C. The lack of night time scavenging is likely due to the snow cover. Snow has been shown to be a poor sink for ozone (Stocker and others 1995, Zeller and Hehn 1995).

The third unique characteristic is relatively high ozone concentration occurring at this site even during midwinter, considering its remoteness, and the change in average ozone concentration as the season progresses (Wooldridge and others 1996) (*table 1*). Monthly average ozone concentration increases from January to May, then decreases in June. There is evidence that stratospheric intrusion causes the April/May ozone peak (Wooldridge and others 1996). Examination of the data indicates high concentration ozone episodes with levels above 60 ppb that can last for several days. These episodes are associated with very low (less than 25 percent) relative humidity, suggesting downward intrusion of air from the stratosphere or upper troposphere. The episodes occur frequently during April and May coincident with passage through the area of cutoff low pressure centers. Ozone concentration levels, during times of the month when these episodes occur, show almost no evidence of diurnal photochemical ozone generation.

The June pattern has the highest diel variation (41 ppb to 52 ppb), yet the average ozone concentration is reduced as the weather patterns for intrusions are greatly reduced. Diurnal photochemical generation of ozone is more likely to occur during June, and lack of a snow cover increases night time scavenging. Yet, average low concentrations in June do not fall below 41 ppb, indicating considerably less scavenging than in urban areas.

Output from the FORFLUX simulation model and ozone concentration was examined, and no significant relationships were found between ambient ozone concentration and modeled canopy photosynthesis or CO_2 conductance (*fig 1*). Variability in the modeled parameters was high. There was no significant relationship between solar radiation, temperature, wind speed, and relative humidity and ambient ozone concentration. We have previously shown a relationship between low humidity and high ozone concentrations (Wooldridge and others 1996), and ambient ozone has been shown to be related to temperature (Flaum and others 1996). There did appear to be a slight trend for increasing ozone concentration with increasing solar radiation and temperature. These relationships would reflect meteorological conditions favorable to ozone production at the site, and would be expected to be stronger with summer data than with the spring data presented here.

Plants often have a closer relationship to ozone flux than to ambient ozone concentration (Musselman and others 1994). Modeled canopy CO_2 conductance showed some relationship to measured ozone flux at GLEES site (*fig. 2*), in contrast to the lack of relationship between conductance and ambient ozone concentration.

There are several reasons for the lack of a relationship between modeled physiological parameters and ambient ozone. First, the model was not specifically designed to simulate physiological response to ozone. Our objective was simply to examine any possible relationship. Second, ozone concentrations are near the lower threshold where plant response should occur; thus, the impact of ozone on plant physiological response was limited. Third, these data represent early season meteorological data. Greater effects should be expected later in the growing season when plants are physiologically more active. CO_2 uptake and conductance increase as the growing season progresses (*fig. 1*). Although the ambient changes in ozone concentration are real, the magnitude of the change as the season progresses is comparatively small and cannot account for the changes in canopy photosynthesis and conductance.

A relationship was found between canopy conductance and measured ozone flux, although the r^2 value was low indicating high variability in the data. We

expected that a greater relationship would be found between ozone flux and conductance than between ambient ozone and conductance.

The lack of a significant relationship between ambient ozone and meteorological factors is not surprising. The interaction of diurnal photochemical production of ozone with the longer duration episodes of stratospheric input of ozone and subsequent mixing complicate the dynamics of ambient ozone concentration at any given time, particularly when examined for only 1 year. Multi-year data were necessary to determine the relationships previously reported (Wooldridge and others 1996). Lack of significant soil and canopy surface sinks and atmospheric chemical sinks at the site also tend to stabilize ozone concentration during changing meteorological conditions. The relationship between ozone concentration and meteorological parameters would be expected to be greater in the summer when soil and canopy sinks are more pronounced and conditions for photochemical production and atmospheric chemical scavenging are greater.

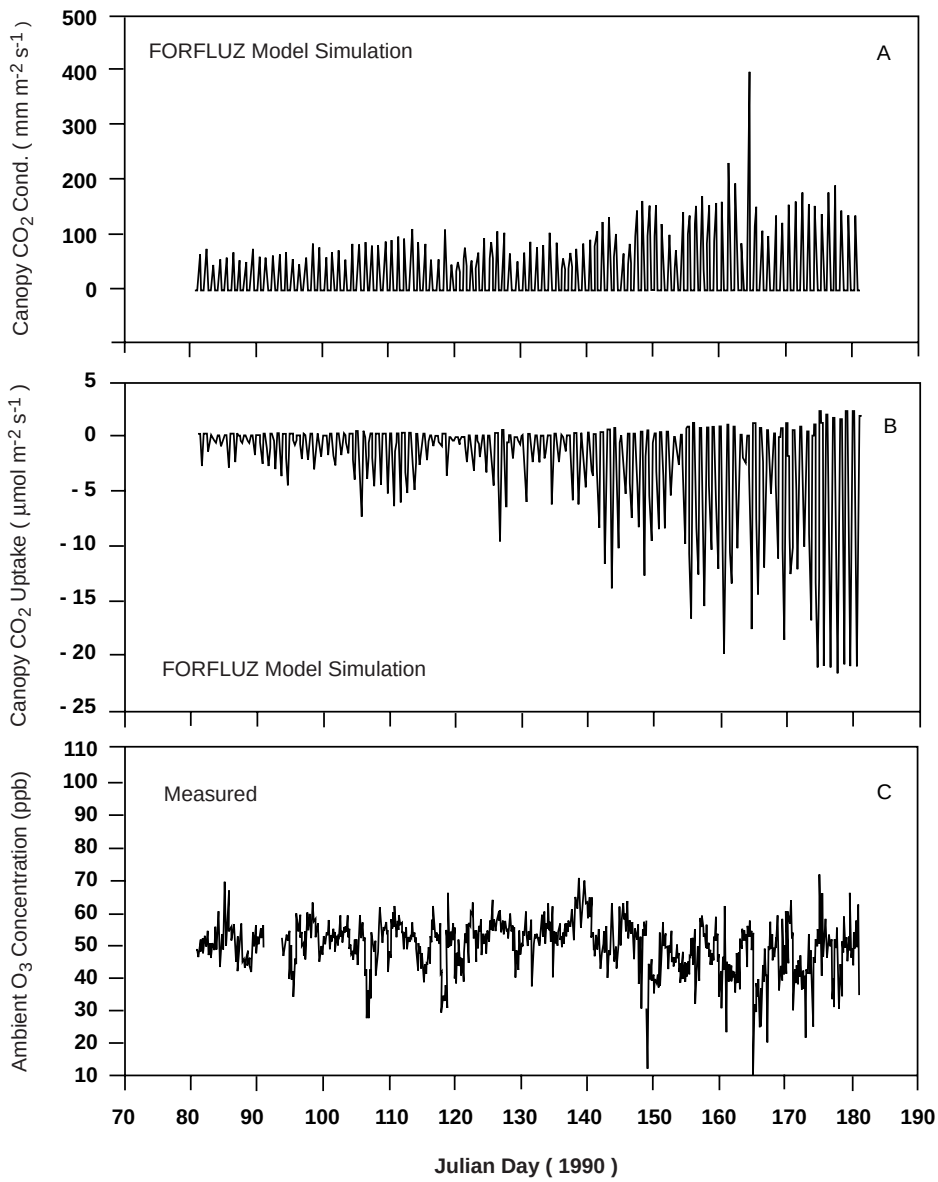

Figure 1 — FORFLUX model simulation for GLEES, canopy CO_2 conductance from mid-March through June, 1990 (A); FORFLUX model simulation for GLEES, canopy CO_2 uptake from mid-March through June 1990 (B); and ambient ozone concentration from mid-March through June 1990 (C).

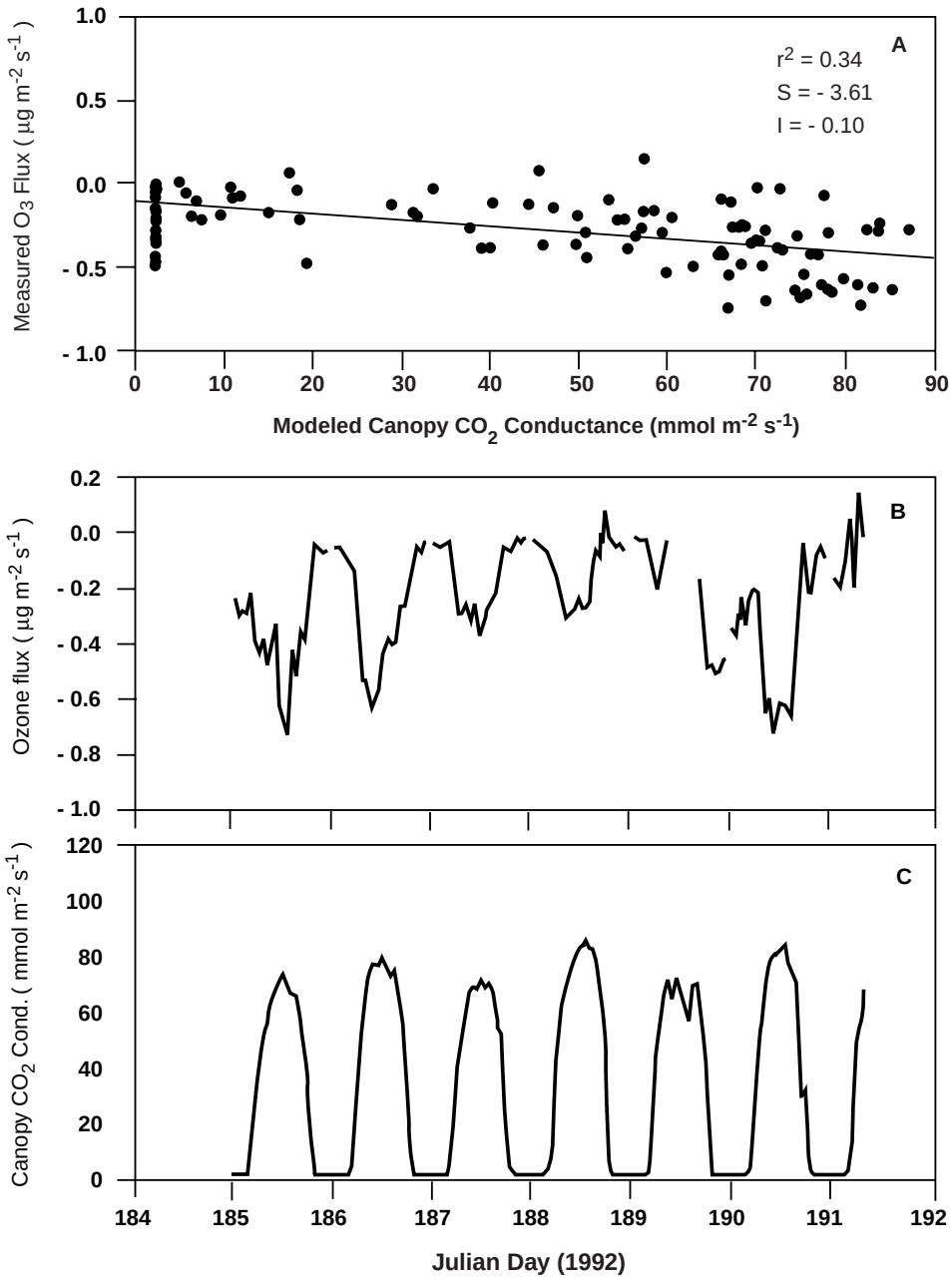

Figure 2 — Regression analysis (A) between measured O_3 flux (B) and modeled canopy CO_2 conductance (C).
r^2 = correlation coefficient;
S = slope, and
I = intercept.

Conclusions

Ozone monitoring at a relatively remote high-elevation Rocky Mountain site showed high concentrations of ambient ozone. Diel variation was slight during winter months indicating little diurnal photochemical generation of ozone; but, diel variation increased as the spring season progressed. Little night time scavenging of ozone was evident when snow cover was present. There was strong evidence of stratospheric intrusion of ozone during spring months, particularly during April and May, associated with the passage of upper level low pressure troughs. Modeled canopy physiological responses were not related to ozone concentration, but modeled canopy CO_2 conductance was related to ozone flux. There was no significant relationship between meteorological parameters and ambient ozone concentration for the 1990 data.

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Bulgarian Rila Mountain Forest Ecosystems Study Site: Site Description and SO_4^{2-}, NO_3^- Deposition[1]

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Abstract

*Bulgaria's forest ecosystems (31 percent of the country's area) are considered vulnerable to dry and wet pollution deposition. Coniferous forests that cover one-third of the total forest land are particularly sensitive to pollution loads. The USDA Forest Service, Sofia University, and the Bulgarian Forest Research Institute (FRI) established a cooperative program to study a Rila Mountain forest ecosystem site, its climatology, air quality, surrounding watershed, hydrology, and soil characteristics. The cooperative program was initiated in September 1991 by upgrading an existing climatology site: Ovnarsko no.3 in the Govedartsi Valley on the north slope of the Rila Mountains. The dominant tree species surrounding the site include Scotch pine (*Pinus sylvestris *L.), black pine (*Pinus nigra *Arn.), Norway spruce (*Picea abies *Karst.), white fir (*Abies alba *Mill.), beech (*Fagus sylvatica *L.), and several oak species (i.e.,* Quercus petraea *Liebl,* Q. pubescens *Willd.,* Q. conferta *Kit.,* Q. cerris *L., and* Q. robur*). Weekly concentrations of nitrate, sulfate, sulfur dioxide, calcium, sodium, and potassium were measured using a filter-pack technique. Data for 18 weeks between October 1991 and November 1993 were collected. Annual acidic aerosol deposition values for 1992 and 1993 were estimated for SO_4^{2}, which was slightly above 3 kg ha^{-1} yr^{-1}, and for NO_3^-, which decreased from 1.3 to 0.3 kg ha^{-1} yr^{-1}.*

Introduction

The Bulgarian Higher Institute of Forestry (HIF), Govedartsi Valley forest meteorological monitoring site, Ovnarsko no. 3, was upgraded in September 1991 (Zeller 1991). The manual recording protocol was enhanced to include continuous hourly measurements of standard meteorology and a filterpack system to determine pollutant dry deposition by direct measurements (Zeller and others 1992). A memorandum of understanding, MU RM-93-164, entitled "Atmospheric Interaction with Natural Ecosystems in Bulgaria and the U.S.A." was executed in July 1993 between the Bulgarian Forest Research Institute (FRI), Sofia University Department of Meteorology (SUDM), and the USDA Forest Service's Rocky Mountain Forest and Range Experiment Station (RMFRES) to formalize the cooperative effort.

This paper describes the Rila Mountain forest ecosystem site in Bulgaria and its surrounding watershed, location, forest, hydrology, geology, soils, and climate by using land manager's records and the European International Food and Agriculture Organization (FAO) system (Bojinov 1981, FAO 1989); and it presents sulfate and nitrate measurements resulting from the cooperative effort.

Location and Methods

Bulgaria is located in the southeastern part of the Balkan peninsula south of Romania, east of the Black Sea, north of Greece and west of Serbia. Forests cover about 33,300 km^2, 31 percent of the country's area. About 37 percent of the forested area is occupied by coniferous forests, 19 percent by seed-originated broad-leaved stands, and 44 percent by deciduous coppices of low productivity. The dominant tree species include Scotch pine (*Pinus sylvestris* L.), black pine (*Pinus nigra* Arn.), Norway spruce (*Picea abies* Karst.), white fir (*Abies alba* Mill.), beech (*Fagus sylvatica* L.), and several oak species (i.e. *Quercus petraea* Liebl, *Q. pubescens* Willd., Q. *conferta* Kit., *Q. cerris* Lq., and *Q. robur*).

Ovnarsko no. 3 (elevation 1,600 m) is located 4 km west of Govedartsi, Bulgaria, situated on the north slope of the Rila Mountains (see site map in Donev and others, this volume). Govedartsi (cattle place) is a small, agrarian village that sports

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a small commercial ski lift, Maljovica, 13 km southwest. Govedartsi is about 70 km south of Sofia, the capital city of Bulgaria, and 14 km southwest of Samokov, a small city. The climate discussion herein is specific to Ovnarsko no. 3. The forest, hydrology, geology and soils discussions are for the general area on the north slope of the Rila Mountains and Govedartsi Valley.

The northern slopes of the Rila Mountains are located in the south of Sofia County. The forested areas are managed by the Samokov and Borovets forest districts and are typical Bulgarian forest regions. In addition to Govedartsi are the small villages Madgare, Mala Tsarkva, Borovets, and Beli Iskar. The Borovets meteorological station location is also a famous all-season national resort of the same name. The general study area described here includes three parts of four north slope Rila Mountain morphographic spurs: Malyovishki (northwestern); Skakavishki (central); and Moussalevsni (eastern). The area has the shape of a leaf with its long fingers about 26 km long and about 12 km wide covering latitudes: 42°06' - 42°17' N; and longitudes 23°12' - 23°40' E. The terrain is very rugged, with clearly outlined hillsides and deeply incised rivers, streams, and ravines. Elevation of the area described is 1,050 - 2,200 m.

Ovnarsko Site Instrumentation Upgrade

Wind speed and direction, air temperature, and humidity sensors are installed at 5 m height; total solar radiation, wetness (actually psuedo-wetness: an electronic plate that simulates a leaf surface and records the value 1 when wet and 0 when dry), precipitation, soil temperature, and soil heat flux measurements are also collocated. Hourly means and variances of the measured quantities were recorded with a Campbell Scientific 21x data logger[6] at 0.1 hertz. Ozone was sampled during the spring and summer months of 1994 and 1995 (Donev and others[this volume]) at 2 m height (*table 1*).

Table 1 *– Parameters measured at the Ovnarsko site.*

Sensor	Parameter	Period	Units
Filter Pack	SO_4^{2-}	week	$\mu g\ m^{-3}$
Filter Pack	NO_3^-	week	$\mu g\ m^{-3}$
Filter Pack	Mg^{2+}	week	$\mu g\ m^{-3}$
Filter Pack	Ca^{2+}	week	$\mu g\ m^{-3}$
Filter Pack	Na^+	week	$\mu g\ m^{-3}$
Filter Pack	K^+	week	$\mu g\ m^{-3}$
Dasibi 1000H	Ozone	hour	ppb
Pump flow	Sample volume	hour	liter pm
MetOne, Inc	Wind direction	hour	Degree, Deg^2
MetOne, Inc	Wind speed	hour	$m\ s^{-1}$, m^2s^{-2}
Eppley	Solar radiation	hour	watt m^{-2}
MRI Inc.	Precipitation	hour	mm
MetOne, Inc.	Temperature	hour	°C
MetOne, Inc.	Relative humidity	hour	pct
Rad Energy	Soil heat flux	hour	watt m^{-2}
thermocouple	Soil temperature	hour	°C
thermocouple	Soil °C gradient	hour	°C
Campbell Sc.	Pseudo wetness	hour	wet:1;dry:0

[6] Mention of trade names or products is for information only and does not imply endorsements by the U.S. Department of Agriculture.

Particulate and Gaseous Ambient Concentration

Pollutant concentration measurements were made by drawing ambient air at the rate 2.0 ± 0.2 liters per minute (24 hr day^{-1}, multi-day $period^{-1}$) through a three-stage filter pack mounted on the meteorology mast at 4 m above ground level. The filter pack system was designed and supplied by the U.S. National Biological Service/ Environmental Science and Technology Center (ESTC). The filter pack inlet is 0.95 cm in diameter, which allows for the collection of aerosols smaller than about 10 mμ diameter when oriented downward. The filter pack itself was mounted under a 20.3 cm diameter polyvinyl chloride (PVC) rain shield. Air flow is volumetrically controlled with a diaphragm pump, pelton-wheel flow meter. Filter packs were exchanged periodically as indicated in the record.

Filter packs provide a convenient, low-cost method for measuring acidic air pollutants (Anlauf and others 1985). Various acid gases and particles are retained on the stacked filters. Wet chemistry extraction of the filtered material yield pollutant masses. The filter pack chemistry used at the Ovnarsko site is the same used for the U.S. Environmental Protection Agency National Dry Deposition Network (NDDN) (Edgerton and others 1989). Three filters are placed in series within an inert Teflon holder: first, teflon "Zeflour" 2μm filter traps suspended particulate aerosols; next, a nylon "Nylasorb" 1μm filter absorbs acidic gases such as nitric acid and sulfur dioxide; and last, two Whatman 41 ashless cellulose filters infuse a 25 percent K_2CO_3 solution and glycerol SO_2 is oxidized to SO_4^{2-} when passed over K_2CO_3, and retained as SO_4^{2-} on the Whatman filters (Zeller and others 1996).

Filter pack preparation, extraction, and analysis was accomplished by a Bulgarian Laboratory under the supervision of the FRI as outlined in the "NDDN Laboratory Operations Manual" (Hunter/ESE 1989). Filter specific laboratory species weights are divided by the measured air volume that passed through the filter pack to provide average concentrations: SO_4^{2-} (teflon filter SO_4^{2-} mass) and NO_3^-(teflon filter NO_3^- mass).

Record Data

Forest

Rila Mountain northern slope forests are mostly indigenous conifers and are managed for timber production. Human occupations that have influenced forest health and use in the Rila Mountain area include iron and gold mining, range management, logging, agriculture, and in the past charcoal production associated with mining. The study area can be examined by forest activity, forest type, and density (*table 2*) (Bojinov 1981). Total timber production area is 16,845 hectares.

Table 2 *— Forest type, density, and area of the Rila Mountain site.*[1]

Type	Coniferous	Broadleaf	Total	Percent
Density 0.4-1.0				
Indigenous	13,641	556	14,197	73.2
Crop	590	50	640	3.3
Density 0.1-0.4				
Indigenous	1,696	30	1,726	8.9
Crop	51	0	51	0.3
Density 0.0				
Seeded	230	0	230	1.2
Dwarf pine	97		97	0.5
Bare lands	2,442		2,442	12.6

[1] ha

The majority of timber production occurs on slopes between 10 to 30 percent steepness (*table 3*). Bulgarian forests are also classified by slope orientation exposure: sunny (SE, S, SW, and W) 4,498 hectares (ha); and shady (NW, N, NE and E) 12,348 ha. *Table 4* gives a breakdown by elevation.

Table 3 — *Rila Mountain forest by area and slope.*

Slope					
(percent)	0-4	5-10	11-20	21-30	>30
Area (ha)	152	977	6,081	7,901	1,735

Table 4 — *Rila Mountain forest by area and elevation.*

Evevation					
Elevation(m)	<1,300	1,301-1,500	1,501-1,700	1,701-1,900	>1,900
Area (ha)	4,354	5,294	3,439	2,717	1,042

The main species composition of the study area include: Scots pine (*Pinus sylvestris*), Norway spruce (*Picea abies*), silver fir (*Abies alba*) and Macedonian pine (*Pinus peuce*). There are, however, 23 different coniferous species, including 10 *Pinus* species, and 54 different broad-leaved species including 8 *Acer* (Delkov 1988, Stefanov and Ganchev 1958). Sensitivity of these Bulgarian tree species to human activities has been expressed as the resistance of each tree species to air pollutants (Prokopiev 1978). *Table 5* (Bojinov 1981) gives the coverage area by main species and age.

Silver fir is most sensitive to environmental factors. It mostly inhabits cool sites where winter temperatures are moderate. It is particularly sensitive to extreme temperatures, especially spring frosts. Silver fir only occupies a small part of the study region: shady and semi-shady sites with mean annual temperatures between 4.5 and 7.5 °C.

Scots pine is a species with almost no sensitivity to particular environmental factors. It tolerates low tundra-like temperatures to high steppe-like temperatures. It withstands extreme temperatures and quickly colonizes bare areas.

Macedonian pine expresses itself as a native species. It demands high and permanent moisture, particularly high humidity. Macedonian pine trees usually grow in a narrow strip above 1,800 m where environmental conditions are favorable to their development or in areas with mean annual temperatures around 4 °C.

The range of Norway spruce coincides with that of silver fir. In the lower portions of the mountains, it begins at regions with mean annual temperatures of 6.5 °C and grows to elevations with mean annual temperatures of about 2.5 °C.

Table 5 — *Rila Mountain forest by age and coniferous type.*

Type / Age (yrs.)	0 - 40	40 - 80	80 - 120	120 - 160	Percent
Scots pine (ha)	922	3,792	1,112	50	37
Norway spruce	593	1,629	2,796	329	31
Silver fir	25	297	824	21	7
Macedonian pine	185	669	1,117	99	13
Mixed coniferous	124	651	989	93	12

Hydrology, Geology and Soils

The headwaters of the Iskar and the Marica, Bulgaria's two longest rivers, are located within the study area. The Iskar flows north to the Duna (Danube) and the Marica flows southeast to Turkey and eventually the Aegean Sea. Numerous mountain lakes are situated deeply within the Rila alpine at elevations above the study site.

The Rila Mountains are composed of intrusive and metamorphic rocks. South Bulgarian granite is a representative for the first group and is the major component of the mountain core. This granite is found mostly in the south half of the study area just beneath the soil surface. Metamorphic rocks are found in the north half of the study area and are represented mainly by gneiss and schists. Sandstones and conglomerates are found to a lesser extent and lime rocks as well as mechanical sediments represented by gravels and sands also occur. The Rila Mountains were formed during the Paleozoic era (the Hercynian folding) and were strongly denuded in the Mesozoic era. In the beginning of the Tertiary, the study area experienced vertical movements that caused the formation of the surrounding kettle, such as the inner Govedarska Kettle Hole (Bojinov 1981).

The soils in the study area are humic, entric and distric cambisols (*table 6*). Humic cambisols, which are the deepest and most stony, are distributed in the form of a belt in the south portion of the study area above the granite. These soils occur mainly under stands of Macedonian pine and Norway spruce, and pure and mixed Norway spruce stands that comprise 5,027 ha or 30 percent of the study area timber. Entric cambisols, the next deepest and moderate to slightly stony, occupy 9,815 ha (58 percent of the timber area) and are located in the north and lower elevations of the study area above the gneiss. These soils are dominated by Norway spruce and silver fir with some Macedonian and Scots pines. Distric cambisols, which are only slightly stony, are situated mainly on south-facing slopes in the NW portion of the study area and total 2,004 ha (12 percent) of the timber with mostly Scots pine stands.

Moisture reserves in all north slope soils are favorable for tree vegetation and growth (*table 6*). These soils are also well-supplied with nutrients (humus and total nitrogen), especially the humic and entric cambisols (*table 6*). The acidity of distric cambisols varies within the narrowest range (5.2 to 5.4); humic cambisols (3.6 to 5.2) are similar but slightly lower compared to the entric cambisols (4.2 to 5.7).

Table 6 *— Average characteristics of soil types on the north Rila Mountain slopes (Bojinov 1981).*

Type	Depth/humus Horizon (cm)	Water capacity Total/active (mm)	Nutrient Humus/total N (t/ ha)	pH
Humic cambisol	90/35	433/382	442/17	4.4
Eutric cambisol	>80/25	393/359	426/20	5.0
Dystric cambisol	<80/<20	146/139	83/3	5.3

Climate

The climatology of Ovnarsko was recorded between 1961 and 1970 (Serafimov 1978) before the site was upgraded (*table 7*). Other climate stations in the study area include: Dolna Bania (42° 18′ N, 23° 45′ E; 635m); Samokov (42° 19′ N, 23° 34′ E; 1,030m); Borovets (42 ° 15′ N, 23 ° 37′ E; 1346m); Sitniakovo (42° 14′ N, 23° 37′ E; 1,742m); Yazavor Beli Iskar (42° 09′ N, 23° 16′ E; 2390m); Hija Musala (42° 12′ N, 23° 26′ E; 2,392m); and Vrah Musala (42° 11′ N, 23° 35′ E; 2,925m).

Table 7 *— Ovnarsko no. 3 Climatology, 1961-1970.*

Parameters	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
2m temperatures within stand (℃)	- 5.4	- 3.4	- 1.4	3.1	8.2	11.7	13.5	13.1	9.9	5.6	2.7	- 2.2
2m temperature in open (℃)	- 4.9	- 2.8	- 0.6	4.4	9.5	13.0	14.8	14.1	10.4	6.1	3.2	- 2.0
2m max temperature within stand (℃)	- 1.7	0.4	1.8	6.9	12.2	15.3	17.4	17.3	14.1	9.6	5.6	0.8
2m max temperatures in open (℃)	- 0.2	2.7	3.6	9.1	13.9	16.8	18.9	19.3	16.2	11.8	7.4	2.5
2m min temperature within stand(℃)	- 8.0	- 6.3	- 4.2	- 0.1	4.1	7.2	8.7	8.6	6.4	2.6	-0.5	- 6.0
2m min temperature in open (℃)	- 8.7	- 7.5	- 4.6	- 0.5	3.9	6.9	8.2	8.2	5.7	2.0	- 0.9	- 5.6
2m wind speed stand (m s^{-1})	0.25	0.31	0.25	0.28	0.17	0.16	0.16	0.16	0.20	0.26	0.25	0.28
2m wind speed open (m s^{-1})	0.73	0.85	0.77	0.81	0.59	0.53	0.53	0.55	0.69	0.79	0.90	0.91
Precipitation (mm)	71	61	66	82	106	120	90	63	57	69	72	70
2m RH in stand (pct)	86	83	82	78	77	79	76	75	76	75	79	85
2m RH in open (pct)	84	81	79	72	72	73	70	71	73	72	76	84

Precipitation chemistry was sampled in 1989 and pH was below 4.5 (*table 8*). Because there were no nearby sources causing low pH precipitation and an exact record does not exist, it was expected that the Govedartsi valley should be clean of air pollutants. The low pH measurements (*table 8*) were a surprise at the time (Serafimov 1978) and remain unexplained. The concentrations of nitrates are low. Concentrations of chlorides and sulfates and concentrations of heavy metals are indicative of those naturally occurring.

Table 8 *— Precipitation chemistry composition of 1989 samples with < 4.5 pH.*

Ion	NO_x	Cl	SO_x	Pb	Cu	Mn	Ni	Cd
mg L^{-1} (FAO 1989)	0.58	0.44	1.4	11.5	1.6	12.5	2.0	0.8

Sulfate (SO_4^{2-}) and Nitrate (NO_3^-) Concentrations and Deposition

A total of 15, multi-day filter pack samples exposed during summer and autumn periods between May 6, 1992 to November 6, 1993 were recorded (*table 9*). The three initial samples taken between the period September 11, 1991 to October 27, 1991, as reported by Zeller and others (1992) were atypical and not repeated here. Deposition in kilograms per hectare per year (kg ha^{-1} yr^{-1}) was calculated by assuming 0.4 cm s^{-1} deposition velocity (Bytnerowicz and others 1987). If the 1992 and 1993 data are extrapolated for an entire year, NO_3^- deposition would be 1.3 and 0.3 kg ha^{-1} yr^{-1} respectively. Deposition for SO_4^{2-} would be 3.6 and 3.3 kg ha^{-1} yr^{-1} respectively. These last deposition estimates give an indication of annual values only and should be used with caution because measurements were never taken for the entire year, nor were any measurements taken during winter and spring seasons.

Sulfate concentrations are similar to those found in other parts of Europe (Ruijrok and others 1995) but below those in more polluted regions (Zeller and

others 1996). The projected annual SO_4^{2-} aerosol deposition is similar to results for higher polluted regions (Zeller and others 1996), where deposition velocities were individually modelled and typically below 0.4 cm s^{-1}, which might explain the comparison difference between concentration and deposition results. Nitrate concentration and deposition at Ovnarsko are lower than those found in more polluted regions.

Table 9 *— Filter pack concentrations and calculated deposition.*[1]

Days exposed	Date On	*Concentration* ($\mu g\ m^{-3}$) NO_3^-	SO_4^{2-}	*Deposition* kg $ha^{-1}\ yr^{-1}$ NO_3^-	SO_4^{2-}
32	05/06/92	0.17	4.30	0.21	5.42
8	06/06/92	0.26	2.03	0.33	2.56
7	06/14/92	0.38	2.17	0.48	2.74
7	06/21/92	0.27	1.48	0.34	1.87
7	06/28/92	0.18	1.80	0.23	2.27
7	07/05/92	4.89	5.20	6.17	6.56
16	06/13/93	0.39	4.19	0.49	5.29
17	06/29/93	0.59	5.25	0.74	6.62
20	07/16/93	0.14	2.13	0.17	2.69
17	08/05/93	0.20	1.31	0.25	1.65
17	08/22/93	0.17	1.17	0.21	1.48
17	09/08/93	0.22	3.09	0.27	3.90
15	09/25/93	0.14	1.78	0.17	2.25
13	10/10/93	0.16	2.38	0.20	3.00
14	10/23/93	0.22	2.29	0.27	2.89

[1] 0.4 cm s^{-1} deposition velocity.

Conclusion

The cooperative Bulgarian Rila Mountain forest ecosystems study site, Ovnarsko no. 3, has been described with initial dry deposition results. Although data were not collected for the winter and spring seasons, results indicate SO_4^{2-} and NO_3^- aerosol concentrations are similar to other less polluted regions of Europe, while deposition of NO_3^- is also low and deposition of SO_4^{2-} may be similar to higher polluted areas. This result is dependent on modelled deposition velocity and needs further study.

Acknowledgments

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Fire in Siberian Boreal Forests — Implications for Global Climate and Air Quality[1]

Susan G. Conard[2] and Eduard P. Davidenko[3]

Abstract

Boreal forests and woodlands comprise about 29 percent of the world's forest cover. About 70 percent of this forest is in Eurasia, mostly in the Russian Federation. Boreal forests contain about 45 percent of the world's growing stock and are an increasingly important part of global timber production. Fire impacts large areas of boreal forest annually in Russia, and in excess of 60 percent of these fires are believed to be caused by human activity. Because of the large extent of boreal forest, the large amount of fire activity, and the expected sensitivity of carbon cycling in these systems to fire and climate patterns, there is justifiable concern about the potential effects of boreal fire regimes on atmospheric chemistry, global carbon cycling, and global climate, as well as potential feedback effects of global climate change on fire regimes and carbon cycles in these systems. Accurate estimates of these impacts are hampered by a lack of good data on the area burned and on fire severity. We estimate that the average area burned annually in Russian boreal forests is around 7.3 million ha, an order of magnitude greater than indicated by official fire statistics. This discrepancy has major implications for estimates of emissions of CO_2 and other greenhouse gases. In severe fire seasons, there are also frequent long-term episodes of air pollution caused by wildfire smoke. Because of the interactions between fire, landscape patterns, air quality, and climate in boreal forests, fire regimes and fire management in Siberia have potentially large impacts on regional air quality and on the global environment.

Introduction

Boreal forests and woodlands cover a large area of the earth's surface between 45 and 70° north latitude. Boreal forest and woodland cover an estimated 1.2 billion ha, about 900 million ha of this in closed boreal forest (FIRESCAN 1994). This represents about 30 percent of the world's forested area. More than 50 percent of the world's forests are in North America and Eurasia (Nyejlukto 1994). Russia alone contains about 620 million ha of boreal forest (Tchebakova and others 1994), about 520 million ha of which is in closed (stocked) boreal forest (Alexeyev and others 1995).

Because of their extent, and the large carbon reserves they contain, boreal forests have long been recognized as critical to the earth's carbon balance (e.g., Bonan 1991, Crutzen and others 1979, Levine 1991). Estimates vary, but suggest carbon storage in boreal forest worldwide in the range of 66-127 Pg C in plant material, and 135-247 Pg C in soils and downed woody material and litter (Apps and others 1993, Bolin 1986, Kauppi and others 1992, Olson and others 1993, Tans and others 1990.) This represents perhaps 10 to 17 percent of the total global terrestrial carbon reserves in soils and plant materials (2,100-2,200 Pg C; Bolin 1986). A recent comprehensive analysis of carbon in Russian ecosystems (Alexeyev and Birdsey 1994) estimates that the boreal forests in Russia contain 99.8 Pg C, with 25 percent of this in vegetation and the remainder in soils and forest floor material.

This paper discusses the potential effects on carbon cycling, global climate, and air quality of fires in the boreal forests of Russia, which comprise about two-thirds of the world's boreal forest.

Extent of Fire in Russia's Boreal Forests

Fire is one of nature's primary carbon-cycling mechanisms. It is a critical disturbance factor in Russia's boreal forests, where many fire scars are visible across the landscape and vegetation patterns are often dominated by evidence of past fire history. During the summer fire season, one can hardly fly across central Siberia without observing smoke plumes from the often numerous wildfires. Although data are somewhat

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sparse, dendrochronological studies have estimated typical fire return intervals of 25 to 50 years for *Pinus sylvestris* forest types in central Siberia (Furyaev 1996, Swetnam 1996). Longer fire return intervals (90-130 years) appear more typical for forests dominated by *Larix* species (Furyaev 1996, Valendik [In press]) and by *Abies siberica* and *Picea obovata*. Fire return intervals also decrease from north to south (they may be as short as 10 years on some sites), and are longer on the most well-watered sites along rivers or in areas where forest stands are isolated by wet bogs (FIRESCAN 1997). As a reflection of the difference in dominance and in fire return intervals of the various forest types, about 70 percent of the fires in Russia occur in light coniferous pine and larch-dominated stands (*Pinus sylvestris* or *Larix*), and only about 15 percent in dark coniferous boreal forest (primarily *Abies*, *Picea;* based on Korovin 1996), although the latter cover about 31 percent of the boreal forest area (Tchebakova and others 1994). About 80 percent of the area typically burns in surface fires (Furyaev 1996, Korovin 1996), but patchy crown fires are common, and in severe fire seasons, crown fires may dominate. The Russian Forest Service (Nyejluktov 1994) reports a range of 12,000 to 34,000 wildfires a year in Russia during the period from 1974-1993, with a long-term average of about 17,000 fires per year. Most of these fires occur in a 4-month fire season that extends from May through August (*fig.1*).

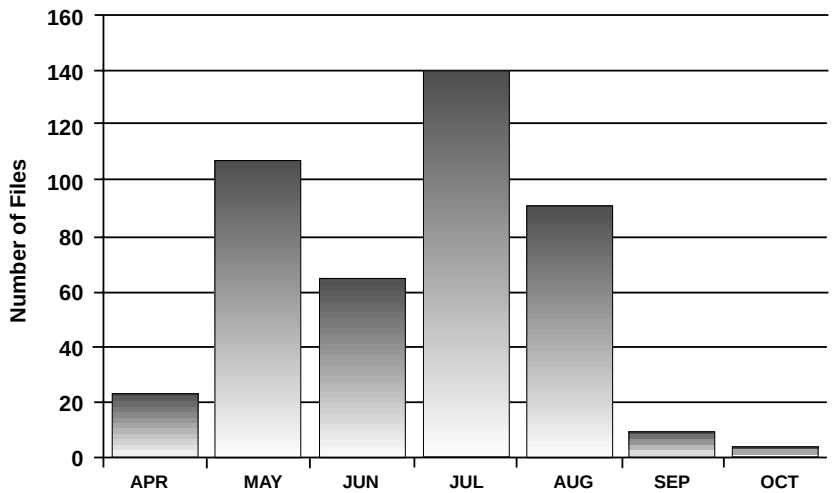

Figure 1 — Monthly distribution of fire frequency during the fire season in Russia. Source: Unpublished fire statistics on file at Russian Aerial Forest Protection Service, 20 Gorkova Street, Pushkino, Moscow Region 141200, Russia.

Estimates of the extent of fires in Russia's boreal forest vary over an order of magnitude (*table 1*); those made by Russian government agencies are among of the smallest. Some of the variation in these estimates undoubtedly results from differences in areas covered. The official Russian fire statistics cover only areas that receive fire protection. The fire protection area includes most of European Russia, but is more limited in Siberia and the Far East. Overall, about 66 percent of the forested area of Russia was monitored for fires in the early 1990's (Dixon and Krankina 1993), most of this by airborne pilot-observers. With recent budget decreases, this area is now considerably less. While the monitored area in the past included most of the closed boreal forest, some boreal forest fires, especially in the more northern regions, were not reported. Other estimates have been based on extrapolations from North American fire regimes (Kasischke and others 1993, Stocks 1991). On the basis of satellite data for three areas of Siberia in 1992, Stocks and others (1996a) estimated close to 1 million acres had burned in the study areas, and suggested this might represent 30 to 40 percent of the area burned in Siberia that year. This would lead to an estimate of 2.5 to 3.3 million acres of burned area for all of Siberia in 1992. On the other hand, analysis of satellite data at the time of the 1987 China fires led to an estimate of about 14 million ha burned in southern Siberia and China (mostly in Siberia) during May and June of 1987 (Cahoon and others 1994). Emissions from this amount of burned area were estimated to represent about 4 percent of the annual global emissions from biomass burning. Although progress is being made in analysis of current satellite data, it is likely to

be a number of years before adequate information will be available from satellite data to allow better long-term estimates. At this point the uncertainty in the area burned seems one of the greatest barriers to accurate estimates of the effects of boreal forest fires in Russia on global biogeochemistry.

Table 1 *— Average area (ha) of boreal forest burned annually in Russia and worldwide.*

Russia	Worldwide	Period	Source
——	1.3×10^6	——	Crutzen and others 1979
3.0×10^6	5.6×10^6	1980-1989	Stocks 1991
4.5×10^6	8.5×10^6	1940-1991	Kasischke and others 1993
1.9×10^6	——	1971-1986	Dixon and Krankina 1993
7.4×10^6	——	1987-1991	Dixon and Krankina 1993
2.9×10^6	——	1971-1991	Dixon and Krankina 1993
0.3×10^6	——	1979-1988	Nyejlukto 1994
0.8×10^6	——	1989-1994	Nyejlukto 1994

In the light of this wide range in estimates of fire area, we decided to take another approach to obtaining a reasonable independent estimate of area burned in Russian boreal forests based on the relationship between fire-return interval and area burned per year. By using a relatively conservative estimate of 500 million ha of closed boreal forest in Russia, we can predict (*fig. 2*) the average burned area per year for different fire return intervals. For a fire return interval of 60-80 years, which seems reasonable based on the literature, we would estimate between 6.25 and 8.3 million ha of burned area per year. This is similar to a recent estimate of 7.4 million ha/yr for the 1988-1991 period by Dixon and Krankina (1993), but considerably higher than their longer-term estimate of 2.9 million ha/yr (*table 1*), which would imply a fire return interval of 172 years (267 years if you use their estimate of 774 million ha of total forest area). Some of the lower estimates of burned areas (*table 1*) would yield fire return intervals in excess of 1000 years. This has clearly not been the pattern over the last several hundred years, or in the 1900's, as revealed by landscape patterns and dendrochronology data (FIRESCAN 1994, 1997).

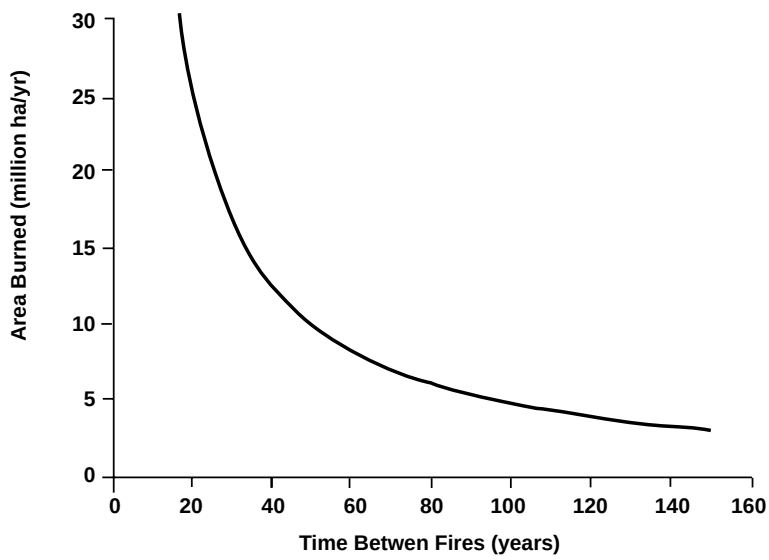

Figure 2 — Relationship between annual area burned in forest fires and the average fire return interval. This graph uses a conservative estimate of 500 ha of closed boreal forest in Russia and assumes no variability in fire return intervals.

We recognize the possibility of some error in this approach, as it does not consider possible recent impacts on fire regime of fire suppression activities and changes in ignition patterns because of human activities. Although evidence from Ontario, Canada, suggests an 89 percent decrease in area burned annually (increase in fire return interval from 65 to 580 years) in areas under fire suppression (Ward and Tithecott 1993), we find no compelling evidence for a similar trend in Russia

(Antonovski and others 1992, Korovin 1996, FIRESCAN 1997). Many fires in Russia, especially large fires and those in remote areas, are little affected by fire suppression. In fact, Russian Forest Service statistics suggest a possible trend toward increasing areas burned over the past 10-15 years (Nyejlukto 1994), which may be associated with increased human ignitions. Despite the great interannual variability of area burned and the unreliability of fire size data for Siberia, it would certainly be possible to detect a major trend of decreased burned areas or increased fire return intervals. In the absence of such evidence, we believe that our estimate is a reasonable one until more accurate ones are available.

Emissions from Boreal Forest Fires

Obviously, wide variations in estimates of area burned will lead to similar variations in estimates of the contribution of emissions from boreal forest fires to the global carbon balance, and this is reflected in the literature. We can assume, for example, that the estimates by Crutzen and others (1979) of 40 to 60 Tg dry matter (18-27 Tg C) consumed per year are probably about an order of magnitude too low, as they are based on an estimate of 1.3 million ha burned annually worldwide (including North America and Eurasia). A better combined global estimate might be about 10 million ha per year (0.2 million ha per year for Alaska and Scandinavia, and 2.4 million ha per year for Canada [Stocks 1991]; 7.3 million ha per year for Russia (our estimate)). On the basis of various sources of Russian fire statistics for the period 1971 to 1991 (see *table 1*), Dixon and Krankina (1993) estimated an average of about 47 Tg C emissions per year after fires. If we adjust this upward based on our estimate of an average burned area of 7.3 million ha per year, we can project annual average C emissions from fire in Russia of about 118 Tg C per year, and 292 to 702 Tg C/yr from biogenic sources. Adding in a rough estimate for North American forests would increase these values by about 30 percent (to about 153 Tg carbon/yr from fire emissions, and about 380 to 913 Tg C/yr from postfire biogenic emissions). This direct amount from fire emissions represents about 4 percent of the estimated 3,900 Tg C/yr released globally from biomass burning (Andreae 1991). If our estimates are reasonable (and there is every reason to believe they are conservative), then annual contributions of boreal fire to global carbon release may have been vastly underestimated by many authors.

Even if we have a good estimate of the annual extent of boreal forest fires in Russia, extrapolation to impacts on emissions or on carbon cycling is complicated by the wide variability in fire intensity across the landscape. Perhaps only 20 percent of the burned area in Russia burns in crown fires (Furyaev 1996, Korovin 1996). In the other 80 percent of the area, low levels of combustion of needles and other fine aerial fuels will result in lower emissions of carbon compounds in smoke per unit area. On the basis of data on a *Pinus sylvestris*/lichen forest in central Siberia (FIRESCAN 1997) we can estimate that fuel consumption in a surface fire in this vegetation would be about 80 percent of that in a crown fire (assuming 4.059 kg/m^2 consumed in crown fire; 3.252 kg/m^2 consumed in a surface fire). So estimates of emissions based on assumption of crown fires will be about 20 percent too high. Dixon and Krankina (1993) recognized this concern and assumed that an average of 10 percent of the biomass was consumed in boreal forest fires in Russia. In other studies it is not always clear how this issue was dealt with.

An even greater potential source of error is the amount of combustion of the litter/forest floor and soil layers, which contain about 75 percent of the carbon stored in Russia's boreal forests (Alexeyev and Birdsey 1994). Depending on the seasonality and intensity of fire and on the vegetation type, combustion of these layers in North American boreal forests can range from about 16 to 90 percent, with an average of perhaps 40 to 60 percent (Kasischke and others 1995). Furthermore, fire characteristics also have generally poorly understood, but substantial, effects on the magnitude of postfire increases in biogenic emissions,

which have been estimated to be 70 to 85 percent of the total fire-related emissions (Dixon and Krankina 1993). Still another source of error is that tree mortality in areas of surface fire can vary widely, and may occur over several years, affecting photosynthesis and respiration, as well as temporal and spatial patterns of postfire decomposition and biogenic emissions. Accurate estimates of the extent and severity of fire in boreal Russia and their effects on combustion patterns and biogenic emissions are desperately needed.

Fire and Global Climate Change

Several authors suggest that, for boreal forest areas in North America, there will be increased fire weather severity (Flanigan and Van Wagner 1991, Stocks 1993), increased length of fire season (Street 1989, Wotton and Flannigan 1993), and increased lightning (Fosberg and others 1990) with CO_2-induced global climate change. Changes in temperature are expected to be considerably higher in boreal regions than the global averages. Stocks (1993) discussed the possible impacts of a doubled CO_2 scenario on fire danger for Canada, based on predictions of the Canadian Climate Center global climate model and the Canadian fire danger rating system. This model predicts a 4-5 °C increase in summer temperatures and a 8-10 °C increase in winter temperatures, along with decreased soil moisture availability throughout central Canada. Wotton and Flannigan (1993) have predicted an increase in the length of the fire season by 30 days, and Flannigan and Van Wagner (1991) predicted a 46 percent increase in seasonal fire severity across Canada under a double CO_2 scenario. Stocks and Lynham (1996) used the Monthly Severity Rating from the Canadian Forest Fire Danger Rating System and the Russian Nesterov Index to compare fire weather severity between Canada and Russia. Both of these indices represent the fire danger if an ignition occurs, based on weather patterns and fuel moisture conditions. Their results suggest that Russia may be even more at risk of increased fire activity in response to climate change than Canada, as Siberia currently has about three times as much area in extreme fire danger categories as Canada (based on Monthly Severity Rating for July). Increases in fire occurrence can also be expected to cause positive feedbacks with atmospheric CO_2, thereby exacerbating effects of global warming (Kurz and others 1994).

Pollution

Direct impacts of fire on air pollution are also cause for concern. It is not uncommon in severe fire seasons for multiple fires to burn over long periods, casting a pall of smoke over large areas. This is by no means a recent problem. Shostakovich (1925) reported on extensive fires in Siberia in 1915. These fires burned for 50 days, and burned about 14 million ha, but the smoke pall from the fires covered 680 million ha, and severely reduced visibility at ground level over a large percent of the area. Extensive smoke plumes were visible on Advanced Very High Resolution Radiometer (AVHRR) imagery in 1987, during the period of the great China fires (Cahoon and others 1994). Given the large number of fires that burn in Russia every summer (an average of perhaps 140 starts per day over the 4-month fire season, and many more at the peak of the season), and the large areas burned, it is not surprising that smoke intrusion over extensive areas occurs.

In addition to the obvious effects of such extensive smoke on visibility and other aesthetic values, prolonged wildfire smoke episodes in western North America in recent years (particularly northern California in 1987, Yellowstone in 1988) have raised concerns over potential negative impacts of wildfire smoke exposure on human health, as well as short-term toxicity from elevated carbon monoxide levels (Reinhardt and others 1995). The main components of concern for human health in wildfire smoke include carbon monoxide, aldehydes, benzene, and particulate matter (Reinhardt and others 1994). Carbon monoxide has been

recognized as a potential hazard to wildland firefighters for a number of years (Jackson and Tietz 1979), but specific information is scant. Studies on prescribed fires in the Pacific Northwest have observed smoke levels exceeding established exposure limits (Reinhardt and others 1994). In a recent study on levels of smoke exposure for firefighters on wildfires in northern California grass and light brush fuels (Reinhardt and others 1995), however, peak exposures never reached recommended ceiling limits of CO of 200 ppm, and only 2 of 23 peak exposures to respiratory irritants exceeded exposure limits. Reinhardt and others (1995) point out that these fires were all in relatively light fuels, and experienced firefighters involved in the incidents rated the smoke levels as low to average. Furthermore, exposures in these fires were typically of short duration. Other research (Materna and others 1992) has found high CO exposure among engine crews on wildland fires in California. Reinhardt and others (1995) concluded that extended exposure to forest fire smoke could exceed exposure guidelines for both CO and respiratory irritants, and recommended that more extensive studies be conducted under a wide range of fuel and burning conditions. A recent review (Reinhardt and Ottmar 1997) of published and unpublished research on exposure of wildland firefighters to smoke and potential health hazards reported mostly on studies conducted in the western U.S. The authors stated that few data on smoke exposure of wildland firefighters are available from other parts of the world. Considering the periodic exposure of rural and urban populations in Siberia to wildfire smoke over prolonged periods, further research on firefighter smoke exposure and on the effects of long-duration smoke exposure from wildfires on human health is vitally needed.

Human Influences on Fire Regimes

Statistics over a recent 3-year period for Russia show that more than 65 percent of wildfires are human-caused, and about 17 percent are confirmed to be caused by lightning (*fig. 3*). Newer information, however, suggests that the importance of lightning ignitions may have been underestimated. Of the 17 percent of fires where cause is unknown, evidence from recently-installed lightning detection systems suggests that many of these may be lightning fires. And the importance of lightning as an ignition source varies greatly geographically (*table 2*), with the highest percent of lightning ignitions in relatively unpopulated areas such as the Krasnoyarsk and Yakutia regions.

Nonetheless, even in these regions, a large percentage of fires is human-caused. Furthermore, it appears possible, based on tendencies visible in the fire statistics of the Russian Forest Service (Nyejlukto 1994) and those reported by Korovin (1996) that there has been an increase in fire numbers and area burned in the past 10 years. After some decrease in the number of fires and areas burned between the early 1970's and about 1983, the trend shows a gradual increase through 1992-93. Average burned area for 5-year periods reported by the Russian Forest Service (Nyejlukto 1994) increased from 0.23 million ha in the 1979-1983 period to 1.03 million ha in 1989-1993, although levels still are not as high as they were in the early 1950's (as high as 2.5 million ha in 1953 [Korovin 1996]). The large influence of human populations on fire occurrence is well-illustrated by Korovin (1996), who shows that about 68 percent of fires occur within 5 km of a road, and 60 percent of fires occur within 10 km of a populated area. Human-caused ignitions may be having a short-term net effect of increased fire in many parts of Russia, despite a strong fire suppression program. Changes in area burned, whether resulting from ignition patterns, weather cycles, or fire management strategies, have particularly important implications for global climate change, as alterations in fire return interval can have significant impacts on carbon storage (which increases with stand age). Model estimates for stored carbon in boreal forest (Kasischke and others 1995) show strong sensitivity to fire return interval in the range of 50 to 150 years, with projected increases in carbon storage of about 10 to 17 percent as fire return interval increases from 50 to 100 years, and increases of about 25 to 40 percent as fire return interval increases from 50 to 150 years. This

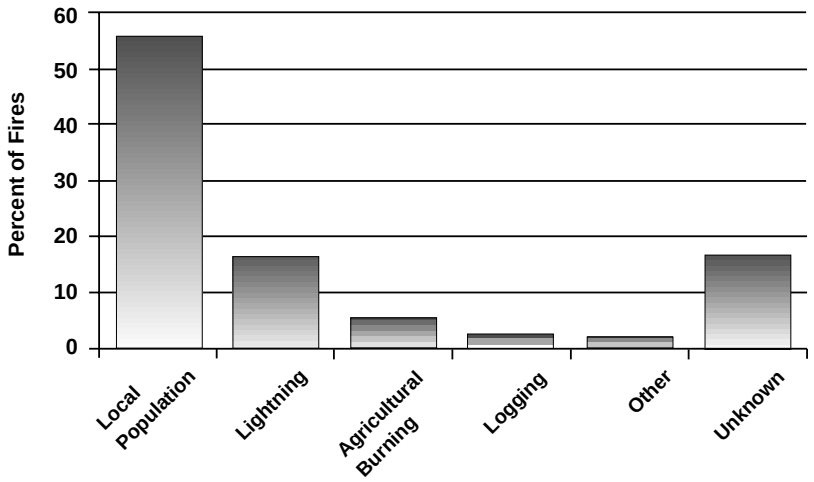

Figure 3 — Frequency of wildfire causes in Russia for the period 1985 through 1994. Source: Unpublished fire statistics on file at Russian Aerial Forest Protection Service, 20 Gorkova Street, Pushkino, Moscow Region 141200, Russia.

Table 2 *— Percent of fires caused by lightning in Russia's major aerial forest protection regions over the 3-year period 1993-1995.*[1]

Protection Region	1993	1994	1995	Average
		Percent of total fires		
Krasnoyarsk	42	35	43	40
Irkutsk	16	12	23	17
Far East	27	20	27	25
Yakutia	53	42	69	55
Arkhangelsk	47	2	17	22
Perm (Europe)	3	0.7	6	3

[1] Source: Unpublished fire statistics on file at Russian Aerial Forest Protection Service, 20 Gorkova Street, Pushkino, Moscow Region 141200, Russia.

sensitivity in the range of current fire return intervals suggests the great potential for human impacts on the fire regime to significantly affect carbon storage, as well as the importance of fire management in managing fire regimes and carbon cycling. An active fire suppression program will be a key element in maintaining desirable fire return intervals in the face of increased human ignitions.

Conclusions

Fire is an important factor in boreal forests of Russia. Despite the paucity of accurate data, it is clear that boreal forest fires in Russia contribute significantly to global CO_2 emissions. Increased CO_2 is likely to substantially increase fire danger and occurrence in boreal forests, leading to possibilities of positive feedbacks to global CO_2 and temperatures (Kurz and others 1994) and significant ecosystem impacts if fire suppression and other fire protection and prevention approaches are not able to compensate. Extensive fires can cause smoke pollution over large areas, which may result in hazards to human health and other negative impacts. The large percent of human-caused fires in Russia's boreal ecosystems suggests the critical importance of fire management and fire suppression for maintaining desirable fire regimes and preventing decreases in fire return interval and loss of carbon storage in these ecosystems.

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Nutrient Fluxes in Forests of the Eastern Sierra Nevada: Comparisons with Humid Forest Systems[1]

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Abstract

*Preliminary results of studies on nutrient fluxes in forests of the eastern Sierra Nevada were compared to those from more humid and polluted ecosystems. Snowmelt, soil solution, soil, and streamwater were collected from Jeffrey and lodgepole pine (*Pinus jeffreyii *[Grev. and Balf.] and* Pinus contorta *Dougl.) stands in Little Valley, Nevada, and from California red fir (*Abies magnifica *A. murr.) and Jeffrey pine/white fir (*Pinus jeffreyii/Abies concolor *[Gord and Glend.] Lindl.) stands at Sagehen, California. Snowmelt, soil solutions, and streamwaters from both sites were circumneutral and dominated by base cations and bicarbonate. The red fir stand at Sagehen had high NO_3^- concentrations (approximately 30-100 μmol /L) in both snowmelt and soil solution during the relatively dry 1993-4 water year. The Little Valley sites had substantially lower NO_3^- concentrations in both snowmelt (5-20 μmol /L) and soil solution (0.5-3 μmol /L) in both wet and dry years. At both sites, a pulse of streamwater NO_3^- (from 0.5 to 20-40 μmol /L) was detected during dry years but not wet years. The Andic soils at the Sagehen site have trace levels of available P in soils, whereas the Entisols and Inceptisols in Little Valley have 10 to 100 times greater levels. The results suggest a hypothesis that the greater mobility of NO_3^- in the Sagehen red fir site was caused by the amount and timing of NO_3^- release from snowmelt (e.g., too concentrated and early for dormant biota to respond to), and that P may also be limiting at Sagehen. Compared to other forest ecosystems in more humid and polluted environments, N, S and H+ deposition and fluxes at the Little Valley site are extremely low, but HCO_3^- and base cation fluxes are high. This reveals that the Little Valley site is relatively pristine, and that ionic fluxes at this site are dominated by natural carbonic acid leaching and weathering reactions.*

Introduction

In comparison to more humid regions, information about nutrient cycling in forests of arid and semi-arid regions is scant. Some nutrient cycling work has been done on ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) (Hart and Firestone 1989, Klemmedson 1975) and lodgepole pine (*Pinus contorta* Dougl.) ecosystems (Fahey 1983, Fahey and Knight 1986, Schimel and Firestone 1989), but these studies are relatively fewer than those in more humid regions (Cole and Rapp 1981, Johnson and Lindberg 1991). Except for Stark's (1973) study on Jeffrey pine (*Pinus jeffreyi* Grev. and Balf.), little is known about the cycling of nutrients in forests of the eastern Sierra Nevada.

Scientific knowledge of nutrient cycles in forests of the eastern Sierra Nevada should be increased for several reasons. First, this knowledge would allow intelligent assessments of forest management practices such as harvesting, burning, and site preparation as well as the effects of exogenous influences such as fire, air pollution, and climate change on these forests. Second, these forests act as filters for nutrients, especially nitrogen, which might otherwise enter the sensitive surface waters of the region. The long-term deterioration of water suality of Lake Tahoe has been clearly documented (Goldman and others 1983). This deterioration is thought to be caused by increasing nutrient inputs from land development, atmospheric deposition, and N fixation by riparian mountain alder (*Alnus tenuifolia* Nutt.) (Byron and Goldman 1989, Coats and others 1976, Leonard and others 1980). With N and H+ inputs increasing in the Tahoe Basin (Byron and others 1991), it is important to gain some knowledge of the capacity of forest ecosystems in the basin to filter these inputs. Galloway and others (1995) forecast that anthropogenic N fixation (by a combination of energy production, fertilizer production, and cultivation of N-fixing crops) will increase by about 60 percent by the year 2020. They predict that about two-thirds of this increase will take place in Asia, which

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by 2020 will account for over half of the global anthropogenic N fixation. Thus, there is the potential for increased N deposition in western North America from both local and long-range sources. At the same time, the capacity of forests to retain N has been reduced because of increased insect- and drought-induced mortality and from fires.

This paper reports progress to date on studies of nutrient cycling in forests of the eastern Sierra Nevada, and compares these results with those from a variety of forest ecosystems in more humid climates by using the Integrated Forest Study database on forest nutrient cycling (Johnson and Lindberg 1991).

Sites and Methods

The Little Valley site is located about 30 km southwest of Reno, Nevada in the eastern Sierra Nevada (*fig. 1*). It ranges from 2,010 to 2,380 m elevation. Vegetation is dominated by lodgepole pine in riparian areas, Jeffrey pine at lower elevations and white fir (*Abies concolor* [Gord and Glend.] Lindl.) at higher elevations. Mean annual temperature (MAT) near the valley floor is 5 °C and mean annual precipitation (MAP) is 550 mm, about 50 percent of which falls as snow. Site 1 in the northern end of the Valley is located on the Corbett series soil, a typic frigid Xeropsamment derived from outwash of decomposed granite. Overstory vegetation consists of Jeffrey pine with an occasional white fir. Understory vegetation is absent. Site 2 is located about 5 km south of site 1 on Marta series soil, a sandy, mixed Aquic Cryumbrept derived from colluvium of decomposed granite. Overstory vegetation in site 2 consists of lodgepole pine with occasional Jeffrey pine. Understory vegetation consists of sage (*Artemesia tridentata* Nutt.) and various grasses and forbs.

The Sagehen Creek watershed is located about 10 km north of Truckee, California (*fig. 1*). Elevation ranges from 1,830 to 2,500 m elevation. MAP and MAT at the Sagehen field station (elevation = 1,830 m) are 870 mm and 4.8 °C, respectively. More than half of total precipitation falls as snow. Site 1 is located on Waca series soil, a loamy skeletal, mixed, frigid Andic Xerochrept derived from andesitic lahar and tuff. Overstory vegetation consists of 80-160-year-old California red fir (*Abies magnifica* A. murr.) with occasional white fir and an understory of pinemat manzanita (*Arctostaphylos nevadensis* Gray) with *Ribes* and *Wyethia* spp. Elevation at the site is 2,100 m. Site 2 is located on Fugawee series soil, a fine-loamy, mixed frigid Ultic Haploxeralf derived from andesitic lahar and tuff. Overstory vegetation consists of 80-160-year-old Jeffrey pine and white fir. Understory consists primarily of occasional Ribes and squaw carpet (*Ceanothus prostratus* Benth.).

Over a 6-year period (1989 - 1995), we collected snowmelt, soil solutions, and streamwater from various sites. Because of a combination of fiscal and logistical limitations, we could not collect solutions from all sites simultaneously. Thus, solutions were collected from Site 1 in Little Valley during the relatively droughty 1989-90 water year, from Site 1 at Sagehen site during the relatively droughty 1993-94 water year, and from Site 2 at Little Valley during the relatively wet 1994-95 water year. We attempted to collect from Site 1 at Sagehen during 1994-95, also, but the deep snowpack (as much as 5 m) there during that year precluded consistent water collections.

Snowmelt collectors and tube-type tension lysimeters were located in six randomly located positions within a 0.05-ha plot at each site. Snowmelt collectors consist of buried collection bottles to which the open collectors are attached. The open collectors sit on the ground surface partially buried and consist of the cut-off tops of polypropylene bottles fitted with a drainage tube that is attached to the buried collection bottle. These devices collect rain or throughfall (depending upon their location) during the snow-free seasons and snowmelt during the snow season. The collection bottle is fitted with tubing that is run-up poles so that collections of snowmelt solutions are possible from beneath snowpack without disturbance.

Tube-type lysimeters are fitted with an extra tube which, along with a vent tube, allows removal of soil solutions from under snowpack without disturbance (Johnson 1995). All solutions were analyzed for conductivity, pH and alkalinity (titration to pH 5.0), cations (Ca^{2+}, Mg^{2+}, K^{+}, Na^{+}, NH_4^{+}) by Dionex ion chromatography,[6] anions (Cl^{-}, NO_3^{-}, ortho phosphate, SO_4^{2-}) by Dionex ion chromatography, and Si by ICP spectroscopy.

We used the chloride balance method (Lindberg and Johnson 1989) to estimate nutrient fluxes at Little Valley. (Because of an insufficient number of reliable snowmelt water volume collections at the other sites, these calculations were not made). Chloride fluxes were measured directly from water volumes and concentrations in the snowmelt collectors beneath the forest canopy. We used these rather than the open collectors in order to better incorporate dry deposition of Cl^{-} into the calculations. It was then assumed that Cl^{-} fluxes through the soil were equal to amounts from snowmelt (i.e., that Cl^{-} was a conservative tracer), and soil water fluxes were calculated from this. These water fluxes were then used with soil solution concentrations to estimate soil leaching rates for the various ions.

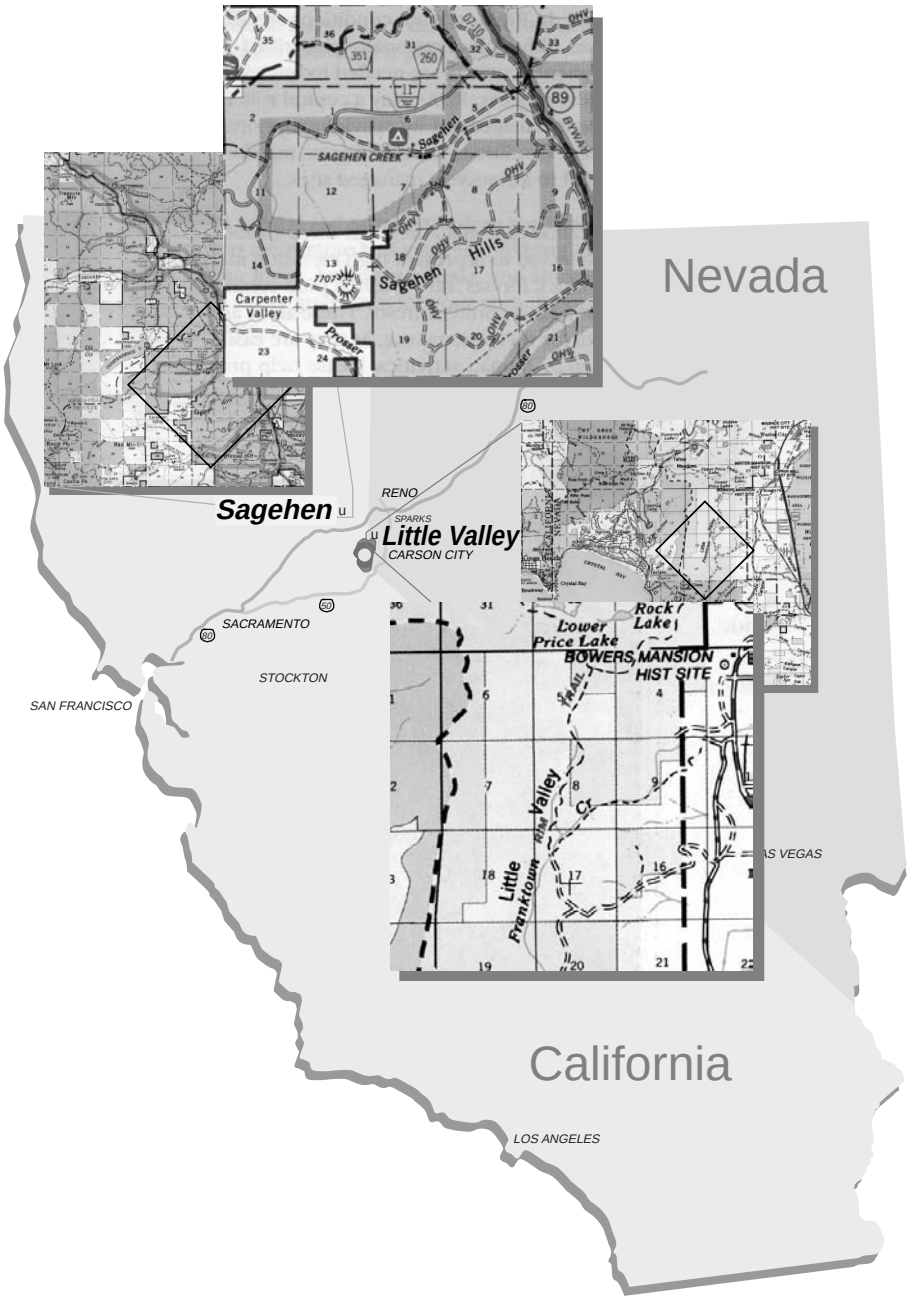

Figure 1 — Research sites were located at Little Valley, Nevada, and Sagehen, California.

[6] Mention of trade names or products is for information only and does not imply endorsement by the U.S. Department of Agriculture.

Results and Discussion

In previous studies in Little Valley (Site 1), we followed up the studies of Coats and others (1976) in the Tahoe Basin and investigated the effects of N fixation by both mountain alder (*Alnus tenuifolia* Nutt.) and snowbush (*Ceanothus velutinus* Dougl.) on soils and soil solutions in comparison to Jeffrey pine. (We know of no previous studies have been conducted on soil solution chemistry beneath snowbush.) In contrast to the results of Coats and others (1976), we found no evidence of excess NO_3^- beneath either mountain alder or snowbush during the 1989-90 water year. Soil solution NO_3^- concentrations were near trace levels under all three species (*table 1*; Johnson 1995). We found a sizable NO_3^- peak in streamwaters from nearby Franktown Creek during snowmelt in 1990 (a dry year) (*fig. 2*) but not in 1995 (a wet year). Sulfate, HCO_3^-, total cation and total anion concentrations in Site 1 in Little Valley were quite high. Sulfate and Cl^- concentrations in soil solutions were two orders of magnitude greater than in nearby streamwaters (Johnson 1995). More recent studies (1994-95 water year) in a lodgepole pine stand growing on more weathered soil lower in the valley (Site 2) showed similarly low soil solution NO_3^- concentrations, but also much lower SO_4^{2-}, Cl^-, total cation and anion concentrations than in the original site. At this stage, we do not know the reasons for the differences in soil solutions between these two sites, but possibilities include the age of the parent material and the amount of water flux. The parent material in Site 1 consisted of alluvium from nearby Ophir Creek and is noticeably less oxidized than that in Site 2. Research is underway to determine the reasons for these differences.

Snowmelt and soil solutions from the red fir stand (Site 1) at Sagehen were similar to those from Site 2 in Little Valley in terms of pH, HCO_3^- , SO_4^{2-} , total

Table 1 *— Concentrations of selected ions in soil solutions and streams from Little Valley, Nevada, and Sagehen, California.*[1]

	pH	HCO_3^-	NH_4^+	SO_4^{2-}	Cation	Anions
		$\mu mol_c\ L^{-1}$				
Little Valley:						
Site 1[2]						
Soil solution						
Ceanothus	6.7 ± 0.1	306 ± 47	3 ± 3	107 ± 15	1107 ± 129	614 ± 113
Pine	7.1 ± 0.1	417 ± 94	<0.5	273 ± 46	1098 ± 200	842 ± 75
Mt. Alder	7.0 ± 0.2	688 ± 192	<0.5	181 ± 59	1084 ± 315	1017 ± 296
Stream	7.1 ± 0.2	373 ± 15	3 ± 2	3 ± 0.2	373 ± 12	396 ± 15
Site 2						
Snowmelt	6.0 ± 0.3	102 ± 88	4 ± 6	15 ± 18	337 ± 203	143 ± 104
Soil solution						
15 cm	6.4 ± 0.4	260 ± 257	0.2 ± 0.6	10 ± 25	514 ± 328	292 ± 255
30 cm	6.6 ± 0.4	281 ± 138	0.2 ± 0.5	7 ± 6	501 ± 109	328 ± 125
Stream	6.8 ± 0.4	389 ± 30	0.5 ± 1.0	11 ± 12	579 ± 77	419 ± 48
Sagehen Creek:						
Site 1						
Snowmelt	6.8 ± 0.2	125 ± 75	64 ± 60	13 ± 8	340 ± 120	231 ± 93
Soil solution						
15 cm	7.2 ± 0.7	300 ± 111	64 ± 48	19 ± 12	693 ± 297	408 ± 129
30 cm	7.3 ± 0.3	407 ± 283	48 ± 33	20 ± 16	702 ± 329	505 ± 301
Stream	7.4 ± 0.4	977 ± 440	19 ± 21	2 ± 1	1280 ± 219	1006 ± 429

[1] Standard errors are given.
[2] Johnson (1995).

cations and total anions, but were considerably higher in both NO_3^- and NH_4^+ (*table 1*). (Snowmelt and soil solution data from Site 2 at Sagehen are being collected in the 1995-96 season and are not yet available.) Sagehen Creek showed a peak in NO_3^- during snowmelt in a drought year (1993-94) but not in a wet year (1994-95) (*fig. 3*). At both Little Valley Site 2 and Sagehen Site 1, snowmelt NO_3^- peaked during the spring of 1994 and 1995 (*fig. 4*). But the two sites differed: the peak NO_3^- concentrations at the Sagehen site were an order of magnitude greater than at Little Valley Site 2, and the pulse of NO_3^- at Sagehen penetrated the soil whereas this was not the case in Little Valley Site 2. The same patterns were apparent in NH_4^+ release from snowpack and mobility at these two sites (not shown). Differences in snowpack amounts, snowmelt NO_3^- and NH_4^+ concentrations, the timing of NO_3^- and NH_4^+ release from snowpack NH_4^+ , and site conditions could contribute to these differences in NO_3^- and NH_4^+ mobility at the two sites. At Sagehen Site 1, peak snowmelt NO_3^- was much higher and occurred earlier (March-May) than at Little Valley Site 2 (May-June). Only in June, when snowmelt NO_3^- concentrations decreased and biological N uptake presumably commenced, did soil solution NO_3^- at Sagehen Site 1 drop to levels typical of N deficient ecosystems. The same patterns were apparent for NH_4^+ (not shown).

Currently, we are entertaining two alternative hypotheses to explain the differences in N^- mobility in the Little Valley and Sagehen sites. The first hypothesis is that there is a temporal discoupling of N release from melting snowpack and

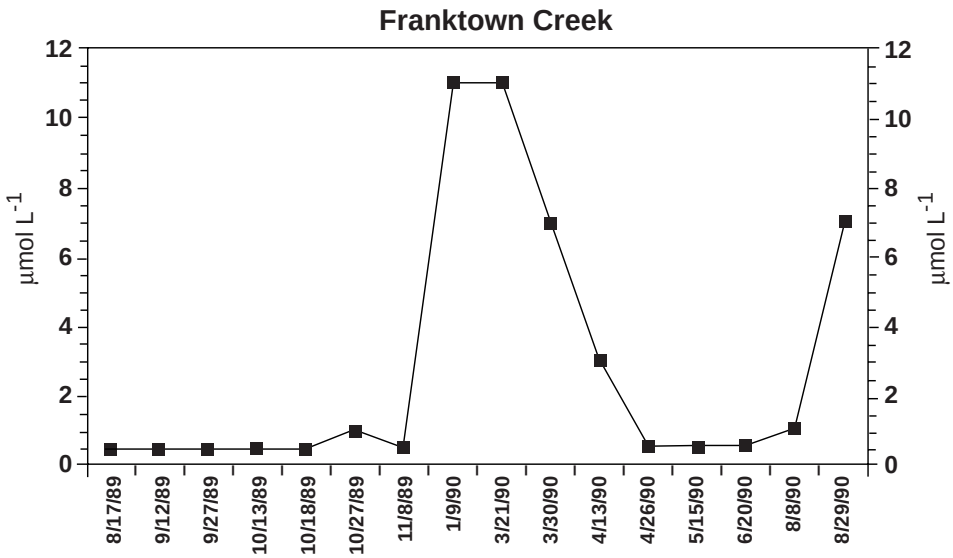

Figure 2 — Temporal patterns in NO_3^- concentration in Franktown Creek, Little Valley, Nevada.

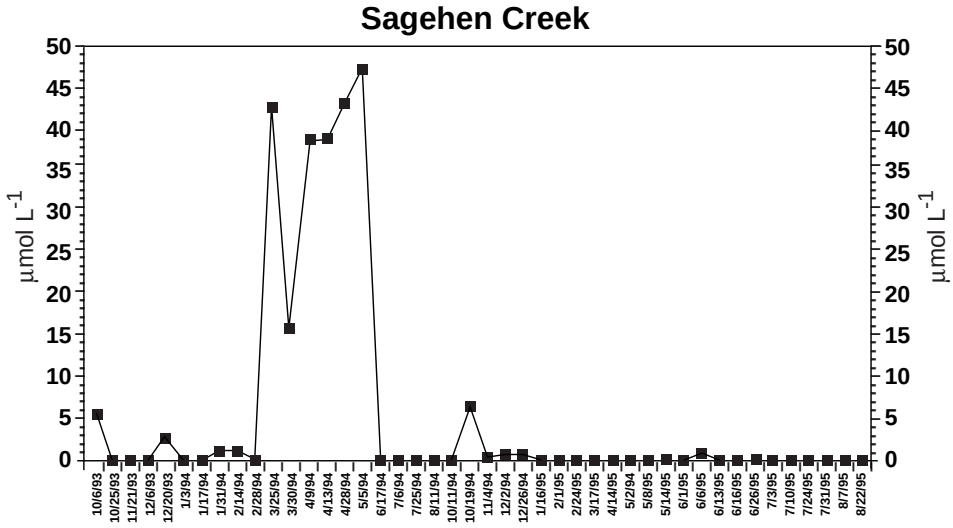

Figure 3 — Temporal patterns in NO_3^- concentration in Sagehen Creek, California.

the activation of biological uptake in these systems. In Mediterranean ecosystems like these, when soil moisture conditions are most favorable for biological activity (winter/spring), the soil and air temperatures may be limiting; when soil and air temperatures are most favorable (summer) for biological activity, soil moisture conditions are limiting. Thus, only during brief periods (e.g., after the snow pack melts and in the fall period before snow pack accumulates) is biological activity not severely inhibited by temperature and moisture conditions. We hypothesize that amount and timing of N release during snowmelt is critical to the mobility of N in the system. If snowmelt N release is concentrated and early, as apparently was the case at Sagehen in the droughty spring 1994, biota may not have broken dormancy sufficiently to take up N as it passed by. If snowmelt N release is more diluted and later, such as was the case in the wet spring 1995, there is a better chance for biotic uptake of N. We are currently testing this hypothesis by making simultaneous collections in the low-elevation pine-dominated stands (Site 2) at both Little Valley and Sagehen, where we are comparing and contrasting N mobility during wet and dry years.

The second hypothesis to explain the difference in N mobility in these two watersheds relates to the soil parent material and its effects on P status. The available P status of the Sagehen soils, which nearly fall into the Andisol order (known to have high phosphate adsorption capacity), is low compared to the soils in Little Valley. Soils from Site 1 in Little Valley have extractable levels that are an order of magnitude greater than those typical of more humid region forest soils (Johnson and Lindberg 1991) (*table 2*). In the more weathered and oxidized soils in Site 2 in

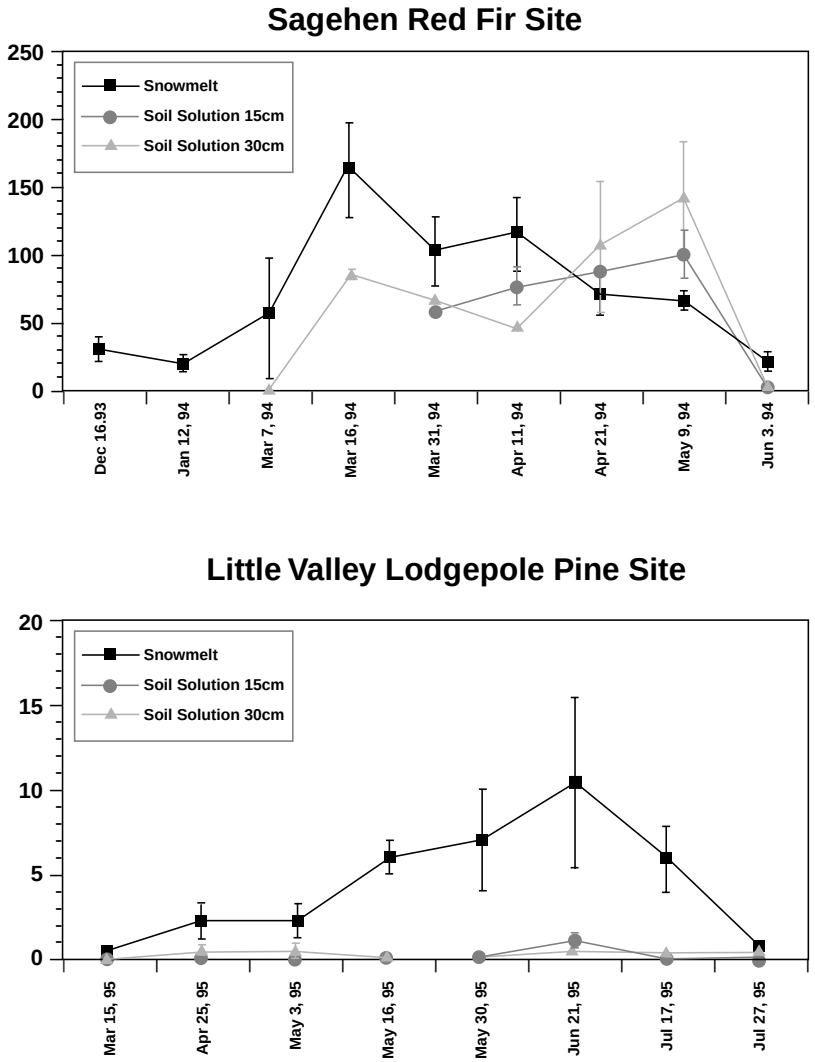

Figure 4 — Snowmelt and soil solution NO_3^- in Sagehen Site 1 (top) and Little Valley Site 2 (bottom).

Little Valley, available P levels are in the range typically encountered in more humid forest soils. In contrast, extractable P in the Sagehen soils from two different sites are below detection limits (*table 2*). We hypothesize that high P fixation capacity on non-crystalline materials (e.g., ferrihydrite, allophane, imogolite, Al and Fe humus complexes) in the Andic soils at Sagehen has caused a potential P limitation. In contrast, we hypothesize that P levels in Little Valley Entisols and Inceptisols are high because of low contents of non crystalline materials and low P fixation capacity. We hypothesize that P limitation in the Sagehen sites limits the capability of these forests to take up N released during snowmelt. We are currently testing this hypothesis in part by performing single-tree N and P fertilization studies at both Little Valley and Sagehen. These fertilization trials and their associated vector analysis of foliage response have proven to be extremely reliable indicators of potential nutrient limitation in other coniferous forests (Timmer and Stone 1978).

We do not believe that the differences in N in the two sites are related to air pollution inputs. Chemical analysis of air (Bytnerowicz and Riechers 1995) was performed in August 1994 at the Sagehen and Little Valley sites (*table 3*). Concentrations of ammonia (NH_3), nitric acid vapor (HNO_3), and particulate nitrate (NO_3^-) were about three times higher at the Little Valley site compared with the Sagehen site. The concentrations at the Little Valley site were similar to the values at the mixed coniferous forest locations near Kings Canyon, western Sierra Nevada (Bytnerowicz and Riechers 1995). HNO_3 concentration at Sagehen was about two times higher than the average summer concentration at the clean site of the eastern Sierra Nevada (Miller and Walsh 1991).

Table 2 *— Soil extractable P (NH_4F/HCL) in Little Valley, Nevada, and Sagehen, California soils.*[1]

Little Valley [2] Horizon and depth (cm)	extr. P ($\mu g\ g^{-1}$)	Sagehen depth (cm)	Horizon and extr. P ($\mu g\ g^{-1}$)
Site 1		***Site 1***	
A (0-20)	101 ± 21	A (0-10)	<0.5
BA (20-40)	106 ± 36	B (10-18)	<0.5
BC (40-60)	69 ± 12	BC (18-45)	<0.5
Site 2		***Site 2***	
A (0-5)	10 ± 3	A (0-10)	0.3 ± 0.3
BA (5-20)	8 ± 3	B (10-18)	0.1 ± 0.1
BC (20-50)	4 ± 2	BC (18-45)	0.1 ± 0.1

[1] Standard errors are given.

[2] Johnson (1995)

Table 3 *— Concentrations of selected nitrogenous air pollutants at two Sierra Nevada sites ($\mu g\ m^{-3}$).*[1]

Location	NH_3	HNO_3	NO_3^- particulate
Sagehen, California	0.347	0.636	0.149
	(0.044)	(0.060)	(0.060)
Little Valley, Nevada	1.142	2.019	0.506
	(0.099)	(0.212)	(0.039)

[1] Standard errors are given. Nitrogenous compounds were measured with annular denuder system (Bytnerowicz and Reichers 1995).

Comparison of Little Valley to Other Forest Ecosystems

Deposition and soil leaching fluxes of N, S, H+, and Ca^{2+}, Mg^{2+}, and HCO_3^- in Site 2 at Little Valley were compared to fluxes in a variety of forest ecosystems from the Integrated Forest Study (IFS) (*figs. 5, 6*). (Because of logistical problems and funding limitations, we have not been able to collect sufficient data to calculate fluxes for Site 1 in Little Valley or the Sagehen site as yet.)

Although the pulses of NO_3^- in the Little Valley and Sagehen sites are of concern in terms of water quality in the sensitive surface waters of this region, comparisons with the IFS sites clearly reveal that the Little Valley (LV) site is relatively pristine. The fluxes of H+, N, and S in LV are lower than those in the other systems, including the relatively unpolluted systems in the Pacific Northwest (DF and RA). To some extent, this comparison may reflect the lack of inclusion of dry deposition to the LV site, since deposition fluxes include only ionic fluxes via rain and snowmelt. However, it seems quite clear that deposition at LV will remain at the very low end of the scale even if annual dry deposition increases inputs estimates by several-fold. If flux data were available for Sagehen Site 1, the N outputs would be substantially greater, judging by the tenfold higher soil solution NO_3^- concentrations.

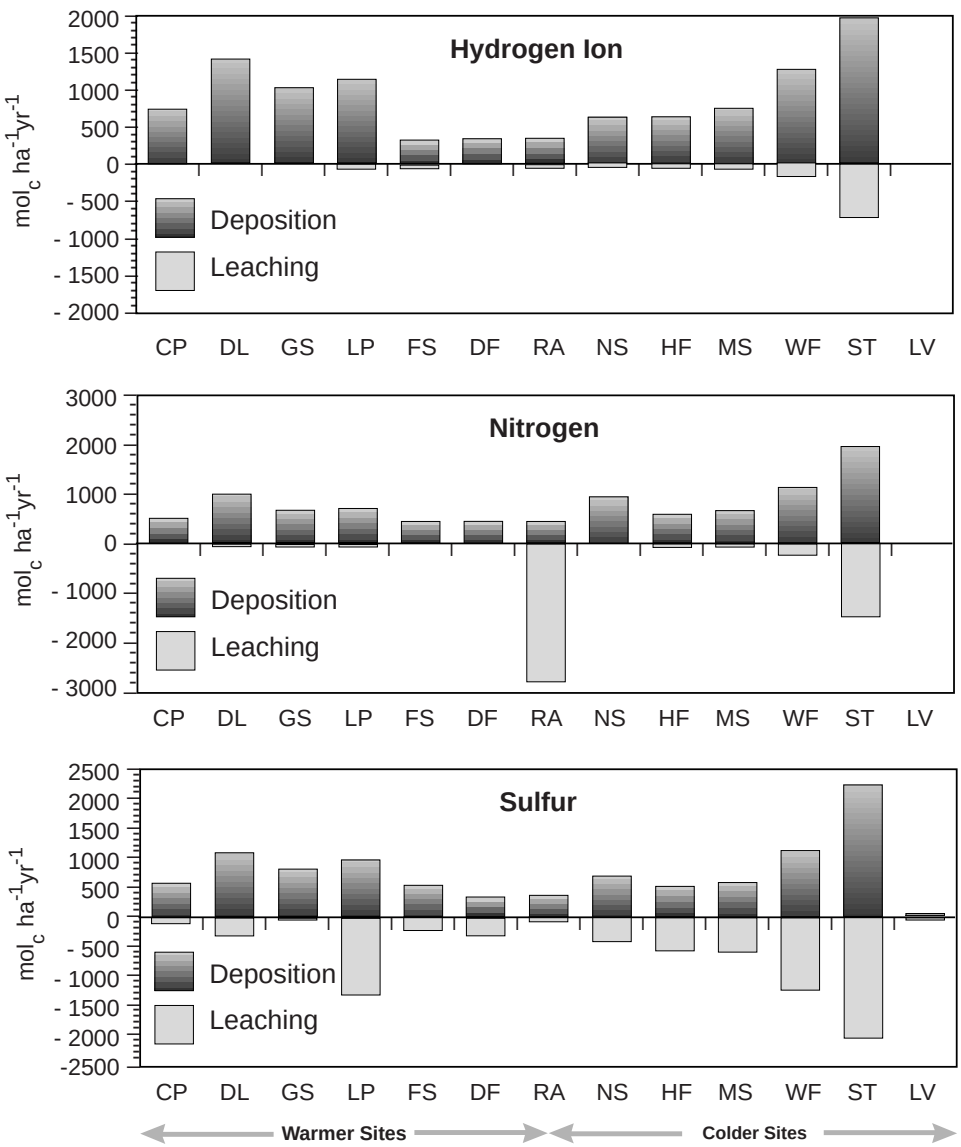

Figure 5 — Atmospheric deposition and leaching of H^+, S, and N in the Integrated Forest Study sites (Johnson and Lindberg 1991) and Site 2 in Little Valley, Nevada (LV). Stands: CP (*Pinus strobus*, Coweeta, NC); DL (*Pinus taeda* stand at Duke, NC); GS (*Pinus taeda*, B.F. Grant Forest, GA); LP (*Pinus taeda*, Oak Ridge, TN); FS (*Pinus eliottii*, Bradford Forest, FL); DF (*Pseudotsuga menziesii*, Thomspson, WA); RA (*Alnus rubra*, Thompson, WA); NS (*Picea abies*, Nordmoen, Norway); HF (northern hardwood, Huntington Forest, NY); MS (*Picea rubens*, Howland, ME); WF (*Picea rubens*, Whiteface, NY); ST (*Picea rubens*, Clingman's Dome, NC); LV (*Pinus contorta/P. jeffreyii*, Little Valley, NV). Deposition fluxes at LV include bulk precipitation during the snow-free season and snowmelt inputs.

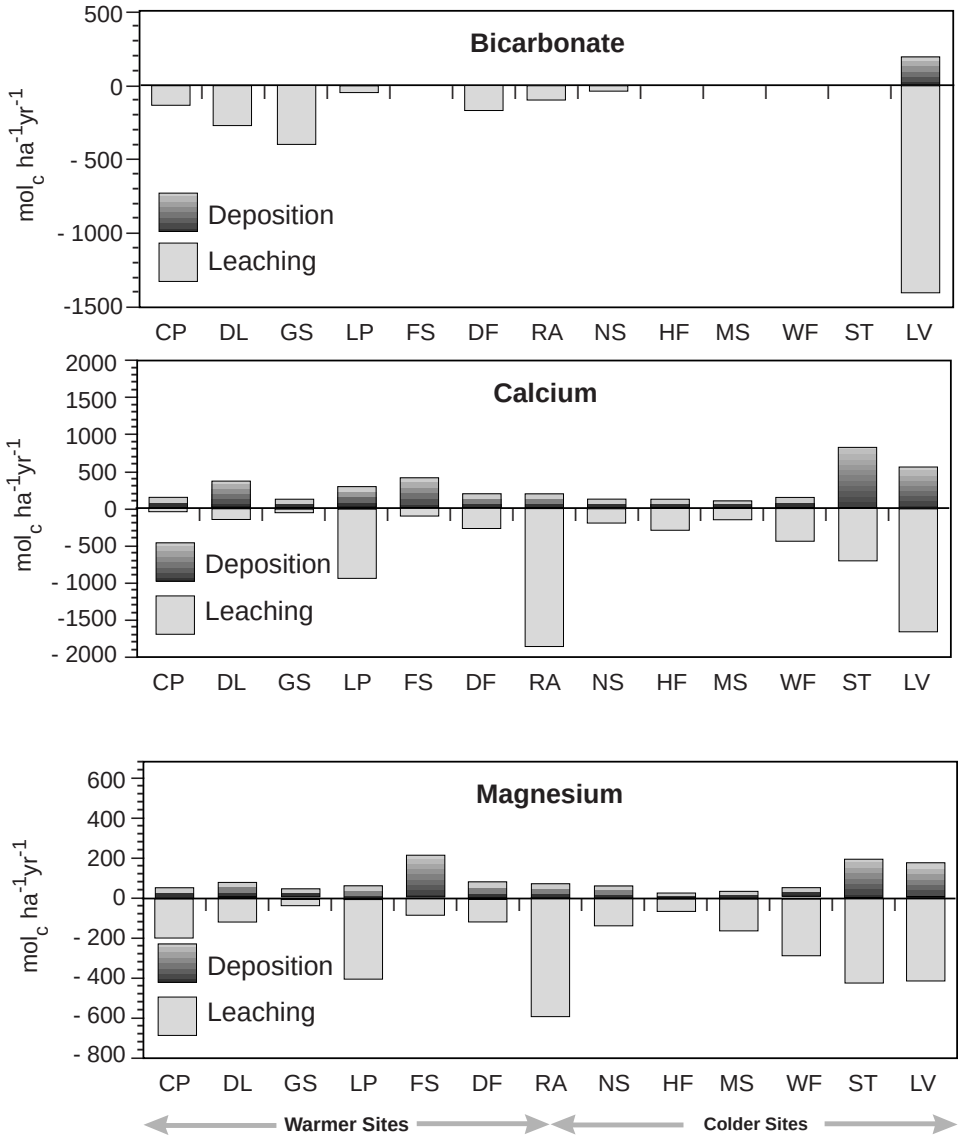

Figure 6 — Atmospheric deposition and leaching of HCO_3^-, K, and Mg in the Integrated Forest Study sites (Johnson and Lindberg 1991) and Site 2 in Little Valley, Nevada (LV). See *figure 5* for legend. Deposition fluxes at LV include bulk precipitation during the snow-free season and snowmelt inputs.

In contrast, the LV site had was in the high range for HCO_3^- fluxes and in the moderate to high range for Ca^{2+} and Mg^{2+} fluxes compared to the other IFS sites. The high inputs no doubt reflect high inputs of dust to the LV site from nearby desert systems, and the high outputs reflect the relatively high base saturation of the soils, high weathering rates of primary minerals, and high rates of HCO_3^- leaching in these systems. The LV site seems to be experiencing net losses of Ca^{2+}, Mg^{2+}, and K+ during the 1994-95 water year, but it should be noted that this was a very wet year (>200 percent of normal snowpack), and leaching fluxes in drier years are almost certainly lower. We will continue to monitor this site in both dry and wet years to develop long-term average flux values.

Conclusions

Studies to date reveal a wide range in N and P status in the four sites investigated in the eastern Sierra Nevada, with no clear patterns yet emerging. In one low-P high-elevation red fir site at Sagehen, NO_3^- released during snowmelt readily leached through the soil profile, suggesting the possibility of a temporal disconnection between snowmelt N release and biological uptake. In lower-elevation pine sites in Little Valley, snowmelt NO_3^- release was later and less concentrated, and did not penetrate through the soil profile. Streamwater NO_3^- pulses occurred during dry years but not wet years in both sites, however, suggesting that snowmelt NO_3^- shunts

around the biological system in at least part of each watershed during dry years. The differences in NO_3^- mobility among these sites may also be related to soil parent material. A large range in available soil P was evidenced among the different soil parent materials, and the degree of weathering varied. The possibility of P deficiency in the Andic soils at Sagehen is being explored.

Compared to other forest ecosystems in more humid and polluted environments, N, S, and H+ deposition and soil leaching fluxes at the Little Valley site are extremely low, but HCO_3^- and base cation fluxes are high. These differences reveal that the Little Valley site is relatively pristine, and that ionic fluxes at this site are dominated by natural carbonic acid leaching and weathering reactions.

Acknowledgments

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Concentrations and Deposition of Nitrogenous Air Pollutants in a Ponderosa/Jeffrey Pine Canopy [1]

Andrzej Bytnerowicz,[2] Mark E. Fenn,[2] Michael J. Arbaugh[2]

Abstract

*Nitrogenous (N) air pollutant concentrations and surface deposition of nitrate (NO_3^-) and ammonium (NH_4^+) to branches of ponderosa pine (*Pinus ponderosa *Dougl. ex. Laws.) seedlings were measured on a vertical transect in a mature ponderosa/Jeffrey (*Pinus jeffreyi *Grev. & Balf.) pine canopy in the mixed conifer forest stand in the San Bernardino Mountains of southern California during the 1992 to 1994 summer seasons (mid-May to mid-October periods). In general, concentrations of nitric acid vapor (HNO_3), nitrous acid vapor (HNO_2), ammonia (NH_3), particulate NO_3^- and NH_4^+ were elevated and not affected by the position within the canopy. In contrast, surface deposition of NO_3^- and NH_4^+ to ponderosa pine seedling branches strongly depended on position in the canopy. The highest deposition of the two ions occurred at the canopy top, while at the canopy bottom deposition was often three to four times lower. The range of deposition fluxes of NO_3^- and NH_4^+ were similar in each of the three seasons. A comparison of the results of this study with other sites in California indicated that deposition fluxes of the studied ions were elevated. In addition to the deposition of NO_3^- and NH_4^+, conductance (K_1) of total NO_3^-, HNO_3 vapor, and particulate NH_4^+ to pine branch surfaces depended on the position in the canopy. The highest values were always measured at the canopy top.*

Introduction

Little is known about concentrations of nitrogenous (N) pollutants in forest locations in California (Bytnerowicz and Fenn 1996). Annular denuder systems (Possanzini and others 1983) allow for effective separation and precise determination of gaseous and particulate N pollutants. These systems have been successfully used in chaparral stands of the San Gabriel Mountains (Grosjean and Bytnerowicz 1993), mixed conifer forests in the central Sierra Nevada (Bytnerowicz and Riechers 1995) and coastal sage scrub communities in southern California (Allen and others, this volume). Short-term, preliminary measurements of N pollutants in mixed conifer forests in the San Bernardino Mountains have been reported (Fenn and Bytnerowicz 1993), but detailed information on their temporal and spatial distribution is lacking.

In the arid climates of the California mountains, dry deposition is one of the major mechanisms of ion transfer from the atmosphere to forests (Rundell and Parsons 1977). Foliage rinsing techniques have been successfully used for determinations of atmospheric deposition of various ions to trees (Bytnerowicz and Fenn 1996, Bytnerowicz and others 1987, Lindberg and others 1986). This technique is especially valuable in the dry climates of California where precipitation (which would allow for throughfall collection) is scarce in the summer season.

This paper presents results of a study that monitored concentrations of gaseous and particulate N air pollutants; measured deposition of nitrate (NO_3^-) and ammonium (NH_4^+) to branches of ponderosa pine seedlings; and calculated conductance to branch surfaces (K_1) of total nitrate, nitric acid vapor, and particulate ammonium on a vertical transect in a canopy of mature ponderosa/Jeffrey pines within the mixed conifer forest in the San Bernardino Mountains in southern California during the mid-May to mid-October periods of 1992 to 1994.

Methods

The study was performed in Barton Flats in the San Bernardino Mountains at an elevation of 2,100 m (*fig. 1*). A 30 m tall tower was located in the mixed conifer

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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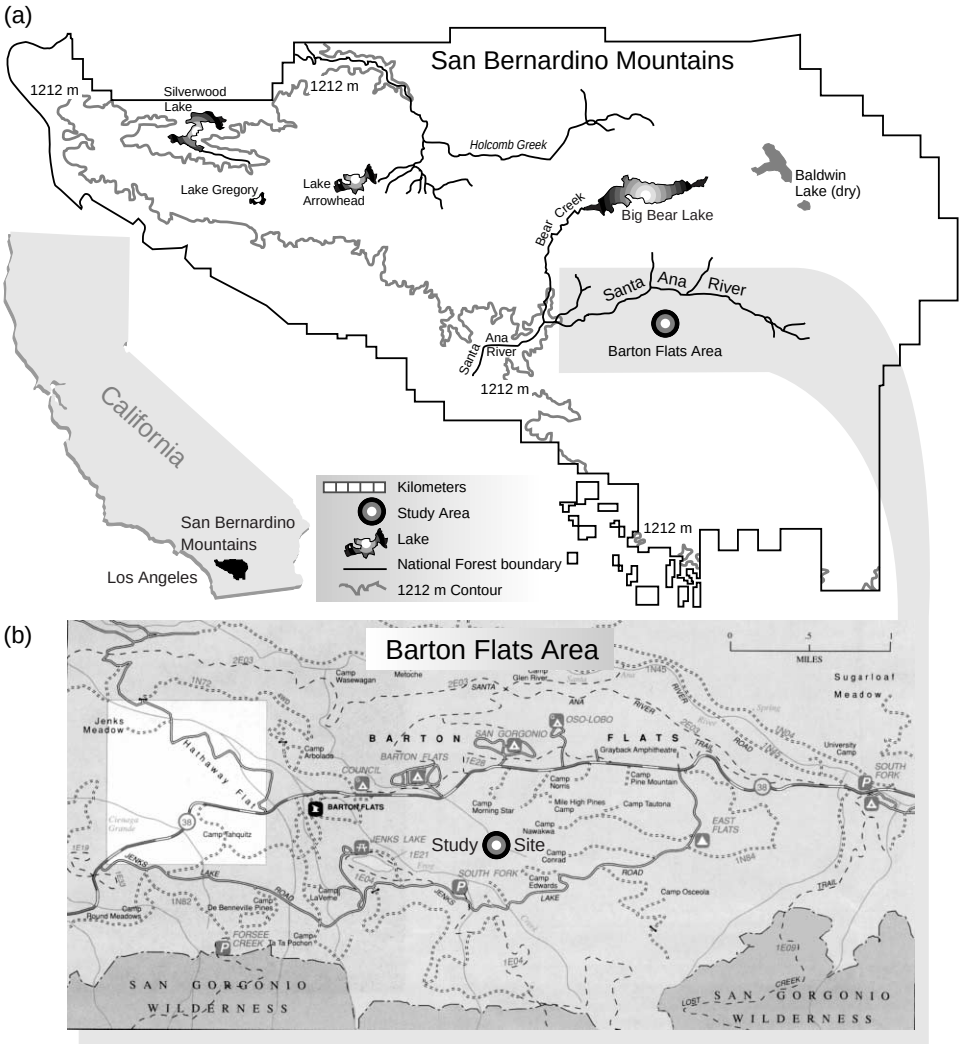

Figure 1 — The study site, Barton Flats, is located in the San Bernardino Mountains in southern California (a), south of State Highway 38 (b).

forest stand of mature ponderosa/Jeffrey pines (Arbaugh and others, this volume). The forest stand was open with about 67 percent of its ground area under tree canopy (Fenn and Bytnerowicz 1996). Single annular denuder systems were placed at each of four different levels of the tower: 29 m (canopy top), 24 m and 16 m (canopy middle), and 12 m (canopy bottom) (*fig. 2*). Average 24-hour concentrations of nitric acid (HNO_3) vapor, nitrous acid (HNO_2) vapor, ammonia (NH_3), particulate nitrate (NO_3^-), and particulate ammonium (NH_4^+) were determined with an annular denuder system (Peake and Legge 1987, Possanzini and others 1983). Constant air flow of 17 L min^{-1} through the systems was provided by steady flow pumps. This allowed for quantitative determinations of gaseous compounds deposited inside annular denuder tubes and the fine particle fraction (< 2.2 □m in diameter) collected on teflon and nylon filters. Concentrations of NO_3^- and NH_4^+ in the annular tube extracts and filter extracts were determined with ion chromatography (Dionex 4000i chromatograph).[3] Concentrations of N pollutants on the tower were measured six times during the 1992 summer season, eight times during the 1993 summer season, and four times during the 1994 summer season.

Deposition fluxes of NO_3^- and NH_4^+ to branches of ponderosa pine seedlings were determined on each of the four levels of the tower in the vicinity of the annular denuder systems. Four 2-year old seedlings were located at each of the tower levels. Seedlings were about 40-50 cm tall and were grown in 6 L pulp pots filled with 2:1:1 mixture of Oakley sand, peat, and fir bark. The seedlings were watered with tap water as needed (typically once a week). At the beginning of each collection

[3] Mention of trade names or products is for information only and does not imply endorsement by the U.S. Department of Agriculture.

period, seedling branches about 10 cm long were thoroughly rinsed with double deionized water. The branches were re-rinsed at the end of 1-week long collection periods with 100 mL of double deionized water and the rinses were collected in 250 mL Nalgene bottles. Bottles were placed on ice, immediately transferred to the laboratory and placed at -18 °C. Concentrations of NO_3^- were determined with ion chromatography (Dionex 4000i ion chromatograph) and concentrations of NH_4^+ by colorimetry (TRAACS, Model 800 instrument) (Bytnerowicz and others 1987). Deposition was measured four times during the 1992 season, six times during the 1993 season, and five times during the 1994 photochemical smog season.

Conductances (K_1) of total NO_3^- (K_{NO3}), HNO_3 vapor (K_{HNO3}) and particulate NH_4^+ (K_{NH4}) to surfaces of pine branches have been calculated (Bytnerowicz and others 1996). Conductance of these compounds is analogous to the deposition velocity at a canopy level (v_d) and is also expressed as cm s^{-1} (Hanson and Lindberg 1991). A general equation for conductance calculations is as follows:

$K_1 = F / c$
in which:
F - deposition flux to branch surface (µg m^{-2} h^{-1})
c - pollutant concentration (µg m^{-3})

K_{NO3} was derived by dividing the NO_3^- deposition flux to branch surfaces by a sum of ambient concentrations of particulate NO_3^- and gaseous HNO_3. K_{HNO3} was calculated by dividing the deposition flux of NO_3^- to branches that could be attributed to HNO_3 by the ambient concentrations of HNO_3. Deposition fluxes of NO_3^- attributed to HNO_3 were obtained by subtracting the portion of particulate NO_3^- deposition from the total NO_3^- deposition (calculated by multiplying ambient concentrations of particulate NO_3^- by the literature value of particulate NO_3^- deposition velocity to conifer foliage) (Davidson and Wu 1990). K_{NH4} was calculated by dividing NH_4^+ deposition to branches by the ambient concentrations of particulate NH_4^+.

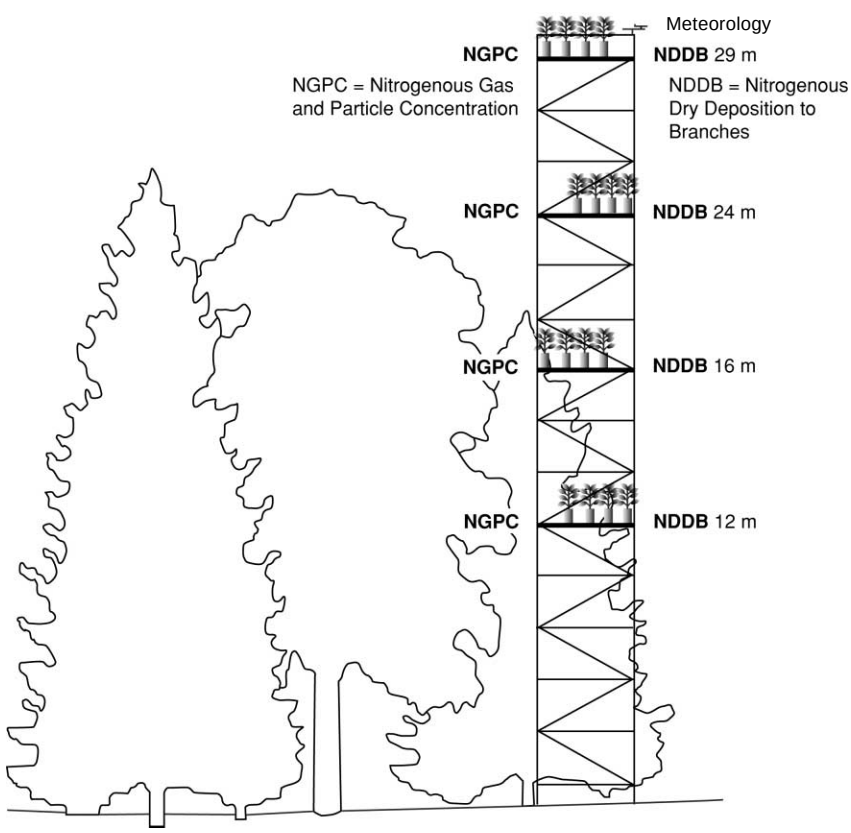

Figure 2 — Schematic of the tower in the mature ponderosa/Jeffrey pine canopy indicating the four levels at which concentrations of N pollutants and their deposition to branches were measured.

Significance of the effects of position on the tower and a period of measurement on concentrations, deposition fluxes, and K_1 values of N pollutants during the 1992 to 1994 summer seasons were evaluated with a two-way analysis of variance and a Bonferroni t-test (SigmaStat 1994).

Results

Concentrations of N pollutants

Seasonal concentrations of HNO_3 vapor, NH_3, particulate NO_3^- and NH_4^+ were averaged at different heights in the canopy for the 1992 to 1994 seasons (*table 1*). In the 1993 season the highest HNO_3 vapor concentrations occurred at 24 m and were significantly higher than the concentrations at 12 m. In 1992 and 1994, position within the canopy had no significant effect on HNO_3 concentrations. Position within the canopy did not affect the concentrations of NH_3, NO_3^-, or NH_4^+ during the entire study.

During the 3 years of study, the highest concentration of HNO_3 vapor (9.7 µg m^{-3}) was measured in June 1994 and the lowest (1.3 µg m^{-3}) in May 1993. The highest concentration of NH_3 (2.6 µg m^{-3}) was recorded in July 1992, while the highest concentrations of particulate NO_3^- (9.5 µg m^{-3}) and particulate NH_4^+ (2.0 µg m^{-3}) were determined in June 1992. The lowest concentrations of NH_3, particulate NO_3^- and particulate NH_4^+ were determined in September 1992 (1.0, 0.35, and 0.2 µg m^{-3}, respectively). Concentrations of HNO_2 vapor during the 1992 season ranged from 0.10 to 0.42 µg m^{-3}, from 0.00 to 0.35 µg m^{-3} in the 1993 season, and from 0.00 to 0.12 µg m^{-3} in the 1994 season. In general, the season significantly affected the concentrations of all studied N species, but no consistent trends were evident (Miller and others 1996).

Table 1 *— Average 24-hour concentrations of N pollutants in the ponderosa/Jeffrey pine canopy during three summer seasons (1992-1994) of investigations (µg m^3) Different letters assigned to the concentration numbers indicate statistically significant differences between the canopy levels.*

Canopy level	1992	1993	1994
		HNO_3	
29 m	3.65	3.24 ab	6.48
24 m	3.33	3.69 a	6.74
16 m	3.05	3.32 ab	6.78
12 m	3.00	3.14 b	5.86
P value	0.084	0.039	0.517
		NH_3	
29 m	1.43	1.51	1.6
24 m	1.43	1.54	1.56
16 m	1.49	1.41	1.56
12 m	1.49	1.35	1.56
P value	0.841	0.119	0.963
		NO_3^-	
29 m	3.86	2.37	2.33
24 m	3.3	2.44	2.33
16 m	3.32	2.72	2.37
12 m	2.97	2.92	1.91
P value	0.250	0.661	0.187
		NH_4^+	
29 m	1.08	0.75	0.69
24 m	0.99	0.64	0.69
16 m	0.98	0.63	0.7
12 m	0.71	0.68	0.5

Surface Deposition of Washable Nitrate and Ammonium

Deposition of NO_3^- to branches of the ponderosa pine seedlings was significantly affected by position within the canopy (*table 2*). The highest deposition flux values were always determined at the canopy top (29 m) and approached about 200 µg m^{-2} h^{-1} during individual 1-week long determination periods (Bytnerowicz and others 1996). Deposition at 24 m was consistently lower than at the top but nearly always higher than at 16 m and the canopy bottom (12 m). The highest deposition fluxes occurred in July and August in each of the three years. Nitrate deposition values were similar during the three years (Bytnerowicz and others 1996).

Ammonium deposition was also strongly affected by position within the canopy (*table 2*). The highest deposition flux values occurred at the canopy top and were sometimes as high as 20 µg m^{-2} h^{-1} during individual determination periods (Bytnerowicz and others 1996). Deposition at 24 m was higher than at 16 m and at 12 m. Although significant differences in deposition fluxes between the individual collection periods occurred, no clear seasonal trend was observed. The values of NH_4^+ deposition flux were similar in all three seasons (Bytnerowicz and others 1996).

Table 2 — Average seasonal deposition fluxes of NO_3^- and NH_4^+ (µg $m^{-2}h^{-1}$) to seedling branches within the ponderosa /Jeffrey pine canopy during three summer seasons (1992-1994). Different letters assigned to the deposition numbers indicate statistically significant differences between the canopy levels.

Canopy level	1992	1993	1994
		NO_3^-	
29 m	126.2 a	118.6 a	132.6 a
24 m	67.6 b	79.8 b	84.0 b
16 m	40.7 c	51.9 c	62.9 bc
12 m	35.4 c	52.4 c	54.7 c
P value	<0.0001	<0.0001	<0.0001
		NH_4^+	
29 m	13.29 a	14.57 a	12.00 a
24 m	9.77 b	10.01 b	9.02 b
16 m	6.61 c	8.13 c	8.28 b
12 m	6.33 c	7.02 c	5.74 c
P value	<0.0001	<0.0001	<0.0001

NO_3^-, HNO_3 and NH_4^+ Conductance at a Branch Level

The highest conductance (K_1) values for total NO_3^-, HNO_3 vapor, and particulate NH_4^+ (*table 3*) were recorded at the canopy top and the lowest at the bottom of the canopy. For total NO_3^- and HNO_3 vapor these differences were strongly significant during three seasons. For particulate NH_4^+ a significant effect of the position within the canopy was seen only in 1993. Large differences were found between individual periods of measurements within each season, i.e., during the 1993 season the range of calculated K_{HNO3} values for the canopy top was between 0.4 cm s^{-1} and 3.1 cm s^{-1} (Bytnerowicz and others 1996). Significant differences between the $K_{NO3\ total}$ and K_{HNO3} values during individual collection periods occurred in 1993 and 1994, and between the K_{NH4} values only in 1993.

Discussion

Position within the canopy did not affect concentrations of the studied pollutants probably because of the openness of the canopy upwind from the scaffold (with

Table 3 *— Average conductance values (K_1) of N species to single ponderosa pine branches during 3 years (1992-1994) of investigations (cm s^{-1}). Different letters assigned to the K_1 values indicate statistically significant differences between canopy levels.*

Canopy level	1992	1993	1994
		NO_3^- total	
29 m	0.50 a	0.61 a	0.42 a
24 m	0.31 ab	0.45 ab	0.27 ab
16 m	0.16 b	0.29 bc	0.21 b
12 m	0.16 b	0.26 c	0.19 b
P value	<0.002	<0.000	<0.002
		HNO_3	
29 m	0.83 a	1.26 a	0.57 a
24 m	0.48 ab	0.74 ab	0.36 ab
16 m	0.30 b	0.55 b	0.26 b
12 m	0.25 b	0.52 b	0.26 b
P value	<0.010	<0.003	<0.011
		NH_4^+	
29 m	0.28	0.66a	0.66
24 m	0.22	0.50 ab	0.29
16 m	0.16	0.40 b	0.25
12 m	0.18	0.34 b	0.29
P value	0.099	0.012	0.160

the exception of significantly higher concentrations of HNO_3 at 24 m than at 12 m during the 1993 season). In the westerly direction, from which pollutants were transported, no nearby canopy inhibited vertical mixing. In more dense forest canopies, such as in the eastern United States, a vertical gradient of pollutants with diminishing concentrations from canopy top to bottom is typical (Meyers and others 1989).

Concentrations of HNO_3 vapor at Barton Flats were moderately high (seasonal means 3.0 to 6.8 µg m^{-3}) compared to the summer average concentrations in forest locations in Tennessee (3.0 µg m^{-3}) and near Gottingen, Germany (5.8 µg m^{-3}) (Lindberg and others 1990). At Tanbark Flat in the San Gabriel Mountains of southern California, a mountain site which is considered one of the most polluted in the United States (Bytnerowicz and others 1987), the average 24-hour concentrations during the 1988 to 1991 summer seasons ranged from 6.7 to 11.7 µg m^{-3} (Grosjean and Bytnerowicz 1993). At various locations in the Sierra Nevada the average 24-hour summer concentrations were below 1 µg m^{-3} (Ashbaugh and others 1991).

Nitrous acid vapor concentrations (seasonal means 0.00 to 0.42 µg m^{-3}) were lower than the summer average concentrations at the Tanbark Flat site (0.55 to 0.65 µg m^{-3}) (Grosjean and Bytnerowicz 1993). However, they were higher than the annual mean concentration of 0.05 µg m^{-3} reported at the remote location in the Canadian Rockies (Legge and Krupa 1989).

Ammonia concentrations (seasonal means 1.35 to 1.6 µg m^{-3}) were elevated and similar to the seasonal means 1.1 to 1.85 µg m^{-3} determined at Tanbark Flat (Grosjean and Bytnerowicz 1993) and lower than the estimated seasonal mean 3.0 µg m^{-3} determined near Gottingen, Germany (Lindberg and others 1990). The measured concentrations were several times higher than the annual mean 0.26 µg m^{-3} value measured in the Canadian Rockies (Legge and Krupa 1989)

and the estimated seasonal mean 0.2 µg m^{-3} in Tennessee (Lindberg and others 1990). However, concentrations of NH_3 in the Netherlands and Great Britain may be several times higher than the reported values reaching 15 µg m^{-3} (Draaijers and others 1989, Keuken and others 1988).

Concentrations of particulate NO_3^- (seasonal means 1.9 to 3.9 µg m^{-3}) were high and similar to the values determined at Tanbark Flat (seasonal means 1.75 to 2.9 µg m^{-3}) (Grosjean and Bytnerowicz 1993). These values greatly exceeded the annual mean 0.13 µg m^{-3} reported for the Canadian Rockies (Legge and Krupa 1989), and seasonal mean 0.19 µg m^{-3} determined in Tennessee (Lindberg and others 1990). The reported values were similar to 2.1 to 5.1 µg m^{-3} measured in the San Bernardino Mountains (Fenn and Bytnerowicz 1993) and 3.3 µg m^{-3} measured in the Los Angeles Basin, California (Sickles and others 1988). Mean NO_3^-concentration in Claremont, Los Angeles Basin (7.9 µg m^{-3}) (Pierson and Brachaczek 1988) was higher than the concentrations determined in this study.

Concentrations of particulate NH_4^+ (seasonal means 0.5 to 1.1 µg m^{-3}) were similar to the values determined at Tanbark Flat (seasonal means 0.65 to 0.85 µg m^{-3}) (Grosjean and Bytnerowicz 1993) and other locations in the San Bernardino Mountains (0.1 to 0.9 µg m^{-3}) (Fenn and Bytnerowicz 1993). Ammonium concentrations were also similar to the mean concentration of 1.0 µg m^{-3} determined in the urban Los Angeles area (Sickles and others 1988). Ammonium concentrations reported for the forest locations in Tennessee and Germany (seasonal means 2.4 and 5.5 µg m^{-3}, respectively) (Lindberg and others 1990) were higher than in this study.

Nitrate fluxes to branches were elevated; the highest values during individual collection periods (approaching 200 µg m^{-2} h^{-1}) were similar to the value of 180 µg m^{-2} h^{-1} determined for mature ponderosa pine in Camp Paivika of the San Bernardino National Forest (Fenn and Bytnerowicz 1993). Higher NO_3^- deposition values were determined at Tanbark Flat; deposition to hoaryleaf ceanothus (*Ceanothus crassifolius*) ranged from 88 to 455 µg m^{-2} h^{-1} (Bytnerowicz and others 1987). Deposition of nitrate to native pines in other parts of the state were much lower: about 4 to 6 µg m^{-2} h^{-1} for Eastern Brook Lake in the eastern Sierra Nevada (Bytnerowicz and others 1992); 27 to 30 µg m^{-2} h^{-1} for Emerald Lake in the western Sierra Nevada (Bytnerowicz and others 1991); and 6 to 31 µg m^{-2} h^{-1} for Whitaker Forest in the western Sierra Nevada (Bytnerowicz and Riechers 1995).

Deposition fluxes of NH_4^+ were also elevated at Barton Flats; the highest values determined during individual collection periods approached 20 µg m^{-2} h^{-1} (Bytnerowicz and others 1996). These values were similar to deposition of 21 µg NH_4^+ m^{-2} h^{-1} determined at Camp Paivika, a high pollution site in the western San Bernardino Mountains (Fenn and Bytnerowicz 1993). Deposition of NH_4^+ to hoaryleaf ceanothus at Tanbark Flat (17 to 46 µg m^{-2} h^{-1}) was higher than at Barton Flats (Bytnerowicz and others 1987). Deposition to native pines at some other forest sites in California were lower than in the present study: at Eastern Brook Lake ranged from 0.8 to 1.6 µg m^{-2} h^{-1} (Bytnerowicz and others 1992); Emerald Lake, 6.0 to 9.0 µg m^{-2} h^{-1} (Bytnerowicz and others 1991); and Whitaker Forest, 1.0 to 1.5 µg m^{-2} h^{-1} (Bytnerowicz and Riechers 1995). The results of this study have clearly indicated that NO_3^- and NH_4^+ deposition fluxes in a mixed coniferous forest strongly depend on the position of the branches within the canopy. Ion deposition increased with increasing branch height. Because concentrations of gaseous and particulate pollutants did not depend on canopy position, the observed distribution of deposition fluxes within the forest canopy could not be related to the air pollution concentration differences. Difference in deposition fluxes could be attributed to varying wind speed occurring along the vertical transect within the canopy because the highest deposition values occurred at the canopy top where wind speeds were highest (Arbaugh and others, this volume). The HNO_3 conductance (k_{HNO3}) values at the canopy top reached 3.1 cm s^{-1} (Bytnerowicz and others 1996) and were similar to the k_{HNO3} values of 0.6 to 3.4 cm s^{-1} determined for loblolly pine in Tennessee (Hanson and Garten 1992). In general, the conductance of a pollutant to a branch

multiplied by the ambient pollutant concentration provides an estimate of deposition at the branch level. Branch level deposition multiplied by the canopy leaf area index (LAI) provides an estimate of canopy level deposition. Therefore, theoretically, the K_{HNO3} and ambient HNO_3 concentration values can be used for an estimate of surface NO_3^- deposition to ponderosa pine canopies from atmospheric HNO_3. Similarly, information on conductance to branches of total NO_3^- and particulate NH_4^+ can be used for estimates of surface dry deposition of total NO_3^- and NH_4^+ if information on ambient concentrations of HNO_3 vapor + particulate NO_3^- or particulate NH_4^+ are provided. On the basis of the results of this study, and other related companion studies, empirical models for estimating N atmospheric deposition to the mixed conifer forest stands in California are being developed (Arbaugh and others, this volume).

Total N deposition to the forest stand in Barton Flats was estimated based on the presented results, information on deposition to other dominant tree species, and LAI for the overstory species (Bytnerowicz and others 1996). The estimated forest stand deposition for the 1993 season was 5 to 9 kg ha^{-1} yr^{-1}, and occurred mostly as dry deposition to surfaces of plants and soil (Bytnerowicz and others 1996). Judging from the results of this and other studies (Fenn and Bytnerowicz 1993, Grosjean and Bytnerowicz 1993), Barton Flats can be characterized as a site with elevated N deposition located in the middle to low end of the west to east air pollution gradient in the San Bernardino Mountains. Chronic N deposition in the more polluted regions of the San Bernardino and San Gabriel Mountains is associated with symptoms of N excess as described by Aber and others (1989), which are especially evident as high NO_3^- concentrations in soil solution and streamwater (Fenn and others 1996, Riggan and others 1985). Long-term effects of N deposition on plant community dynamics in these forests are as yet unknown.

In summary, concentrations of the measured N species at the Barton Flats site were elevated compared to the pristine mountain areas in California and elsewhere. No differences in concentrations of the measured N species were found on the vertical transect in the mature pine canopy. However, deposition fluxes of NO_3^- and NH_4^+ as well as conductance (K_1) of gaseous HNO_3, total NO_3^-, and particulate NH_4^+ were always the highest at the canopy top.

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Predicting Nitrogen Flux Along a Vertical Canopy Gradient in a Mixed Conifer Forest Stand of the San Bernardino Mountains in California[1]

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Abstract

*A 3-year study of nitrogenous (N) air pollution deposition to ponderosa pine (*Pinus ponderosa *Dougl. ex. Laws.) seedlings along a mature tree vertical canopy gradient was conducted in the mixed conifer forest of the San Bernardino Mountains of southern California. Concentrations of nitric acid vapor (HNO_3), particulate nitrate (NO_3^-), and ammonium (NH_4^+) were measured, as well as dry surface deposition of NO_3^- and NH_4^+. By using this data, along with meteorological information, a series of simple models were developed that predict the vertical gradient of foliage-rinse surface deposition for NO_3^-, NH_4^+, and total N. Individual models for NO_3^- and NH_4^+ were calculated by using deposition data, air concentrations at the top of the canopy, and wind speed. These models explained 80 percent of the variation between deposition values at the lower canopy positions. Examination of model coefficients indicated that the two models were not significantly different, and a single model was developed to estimate total N deposition. This model explained less variation than the individual models ($R^2_N = 0.69$) but is a simpler description of the system. All models have two parameters (a_0 and a_1) that are estimated by nonlinear regression. Independent data indicates that the a_1 parameter depends only on the rate of decline in wind speed vertically down the canopy. The a_0 parameter reflects the difference between foliage rinse surface deposition at the top of the canopy and that at lower canopy positions. It indicates that about 20 percent less dry surface deposition occurs at lower canopy positions than expected from wind speed alone for all nitrogenous compounds, most likely because decreased turbulence within the canopy results in increased quasi-laminar boundary layer resistance at lower canopy positions.*

Introduction

Dry deposition is one of the major mechanisms of nitrogenous (N) transfer from the atmosphere to plant surfaces in arid California mountains (Bytnerowicz and Fenn 1996, Rundell and Parsons 1977). High dry deposition fluxes of N pollutants have been reported in exposed areas of the San Gabriel and San Bernardino Mountains (Bytnerowicz and others 1987, Fenn and Bytnerowicz 1993), and in lesser amounts in the southern and western portions of the Sierra Nevada Mountains (Bytnerowicz and others 1991).

Foliage rinsing techniques (Bytnerowicz and others 1987, Lindberg and Lovett 1985) have been successfully used for determinations of atmospheric deposition of various ions to trees. The technique is especially valuable in dry climates of southern California where in summer and fall periods precipitation is scarce, which limits collecting throughfall measures of deposition.

Extensive foliage rinse sampling requires considerable time and expense to analyze. Seedlings must be obtained and stored in greenhouses before field exposure. A scaffold tower must be erected at the remote site, and periodic watering and rinsing must be done for groups of seedlings at different tower heights to obtain sufficient information to quantify the vertical deposition profile. This extensive sampling reduces the number of sample locations and times that can be directly measured using this approach.

Expanding limited foliage rinse measurements to larger areas and longer time periods has been accomplished in some systems by using inferential models that have been developed over the last decade (Hicks and Meyers 1988, Meyers and Baldocchi 1988). These models estimate deposition flux (F_d) from air concentration measurements (C) and deposition conductance (K_1) by using air concentrations and micrometeorological information. To use inferential models, aerodynamic transport, boundary layer transport, and physiochemical surface interactions must be estimated for model input. This model approach works best when ecosystems

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Figure 1 — The Barton Flats Tower site in a mixed conifer forest in the San Bernardino Moutnains in southern Califronia photographed aerially to illustrate the diverse canopy structure of the study site. Canopy diameters vary and are interspersed with large open areas. The arrow points to the location of the scaffold tower where data was gathered for this study.

have uniform canopy and vegetation and when atmospheric conditions are stable and predictable.

The mixed conifer forest of the San Bernardino Mountains is quite different (*fig. 1*) from these systems. This forest type is comprised of several species including, ponderosa pine (*Pinus ponderosa* Dougl. ex. Laws) or Jeffrey pine (*Pinus jeffreyi* Grev & Balf.) (in transition zones both species are found), sugar pine (*Pinus lambertiana*), white fir (*Abies concolor*), incense cedar (*Calocedrus decurrens*), California black oak (*Quercus kelloggii*), and a herbaceous or shrubby understory. The forest often has a discontinuous canopy that results in increased atmospheric mixing and edge effects that seldom result in a significant vertical air pollutant concentration gradient (Bytnerowicz and others, this volume). However, equal deposition does not occur over the canopy. These conditions reduce the ability to apply inferential models to this system, and can result in excessive estimates of N by inferential models (Taylor and others 1996) compared with empirical estimates.

In this study we measured atmospheric concentrations and foliar deposition in the San Bernardino Mountains to determine relationships between atmospheric concentrations, meteorological factors, and deposition. The primary data consists of annular denuder and foliage-rinse data gathered at several heights on a scaffolding tower erected between the canopies of three trees (Miller and others 1996). Filter pack data (Chow and Watson 1996) that continuously monitored air pollution and meteorology at a location about 5 kilometers from the tower location was used to corroborate the results of the annular denuder data.

Our goal was to develop a predictive model for foliage rinse flux along a vertical canopy profile that minimizes the number of flux measurements needed to estimate N deposition flux to mature pine canopies. The model developed is intended to reduce the amount of time and expense to determine deposition vertically along the canopy, and enable estimates of total N dry deposition to the forest.

Methods

Modeling and Analysis Data

As part of a multi-disciplinary study (Miller and others 1996), foliar rinse data were gathered to determine the effect of canopy position on deposition of nitrogenous and sulfurous pollutants, to compare deposition to branches of mature trees and seedlings, and to compare deposition to three tree species (ponderosa/Jeffrey pine, white fir, and California black oak). Measurement of nitrate, ammonium, and sulfate deposition to branches of seedlings and mature trees of ponderosa pine at various levels of the canopy were performed during three photochemical smog seasons (1992, 1993 and 1994). Fluxes of NO_3^- and NH_4^+ to branches of ponderosa pine seedlings were determined on a tower located in a stand of mature ponderosa/Jeffrey pines. Four seedlings were located at each of four different levels of the tower (29 m at the top of the canopy; 24 m and 16 m in the middle of the canopy; and 12 m at the canopy bottom).

In the beginning of each collection period branches about 10 cm long were thoroughly rinsed with double deionized water. The branches were re-rinsed at the end of the collection period with about 100 mL of double deionized water and the rinses were collected in 250 mL Nalgene bottles. Bottles were placed on ice, immediately transferred to the laboratory and placed at -18 °C. Concentrations of nitrate were determined with ion chromatography (Dionex 4000i ion chromatograph)[3] and concentrations of ammonium colorimetrically with a Technicon TRAACS Autoanalyzer.

The filter pack dry deposition sampling system (Chow and others 1993) was equipped with an acid-washed PFA (perfluoralkoxy) sampling surface with PFA Teflon-coated Bendix-Sensidyne Model 240 cyclone. Particles and gases were drawn through the Teflon-coated $PM_{2.5}$ inlet at a constant flow rate of 113 l/min. A total of

[3] Mention of trade names or products is for information only and does not imply endorsement by the U.S. Department of Agriculture

62 l/min was drawn through four filter packs simultaneously. In routine operation, air was drawn through the filter pack for NO_2 sampling at a flow rate of 2 l/min and through the remaining three filter packs at a flow rate of 20 l/min apiece. Four additional ports were provided for field blanks that were used to evaluate filter loadings during passive sampling periods and during filter handling.

Daytime (0600 to 1800 PST) and nighttime (1800 to next day 0600 PST) measurements were taken every sixth day between 11/02/91 and 09/28/93 and between 06/01/94 and 08/30/94 for ammonia (NH_3) and nitric acid (HNO_3) gases on absorbing substrates, and ammonium (NH_4+) and nitrate (NO_3-) particulates on filter substrates.

Model Development

Empirical relationships are subject to spurious correlations and relationships, especially when limited numbers of locations and sampling times are available. To reduce this effect, we based our model upon simplified theoretical relationships rather than observed ones. We chose the equilibrium relationship between aerosol and gaseous concentrations as the basis for model formulation:

$$NH_4NO_3 <=> HNO_3 + NH_3$$

If it is assumed that all particulates are deposited as NH_4NO_3, then it can be assumed that:

$$K_{NO3p} = K_{NH4p} = K_{NH4NO3}$$

in which:

K = conductance of N to the plant surface (cm s^{-1})

NO_{3p} = N from particulate NO_3-

NH_{4^+p} = N from particulate NH_4+

Expressing atmospheric concentrations (C) as µg m^{-3} N, then

$$C_{NO3p} = C_{NH4p}$$

These relationships implicitly assume that NH_4+_p deposition to plant surfaces is caused by NH_4+ rather than NH_3. Several studies that have linked diurnal patterns of NH_3 deposition to stomatal aperture (Aneja and others 1986, Hutchinson and others 1972, Rogers and Aneja 1980) found that little NH_4+ is absorbed cuticularly (Van Hove and others 1987), and that NH_3 deposition contributes little to surface deposition (Van Hove and others 1987).

The equilibrium relationship is also only valid if the majority of particulates are in the range of 0.1 µm to 2 µm diameter. Wall and others (1988) found nitrate and ammonium particulate sizes vary between 0.52 µm and about 2.0 µm in a study conducted in Claremont, California. Similar results were found by Lundgren and others (1979) in a study conducted in Riverside, California, which is about 50 km from our study site.

Both the annular denuder and filter pack data were used to examine model assumptions. High significant correlations were found by both systems between particulate NO_3^- and particulate NH_4^+ concentrations (r=0.92 and r=0.93, respectively) during foliage wash periods, indicating that the majority of particulates were in the form of NH_4NO_3. Average NO_3^- concentrations were 1.80 mg N/m^3 ± 0.98 mg N/m^3 compared to 1.98 mg N/m^3 ± 0.74 mg N/m^3 estimated from NH_4^+. The two estimates are not significantly different (paired t-test, p = 0.05).

By using these relationships, flux (F) and atmospheric concentration (C) data can be expressed as:

$$F_{NO3p} = F_{NH4p}$$

and

$$F_{HNO3} = F_{NO3t} - F_{NH4p}$$

in which F is flux (µg N m^{-2} s^{-1}).

Fluxes at lower canopy positions can be estimated for nitrate deposition as:

$$F_{NO3h} = a_0 e[a_1(1- H_h/H_t)](F_{HNO3t} + F_{NO3t}) + e_h$$

for ammonium deposition as:

$$F_{NO3h} = a_0 e[a_1(1- H_h/H_t)](F_{NH4t}) + e_h$$

or for either type of deposition as:

$$F_{Nh} = a_0 e[a_1(1- H_h/H_t)](F_{Nt}) + e_h$$

in which:
h = vertical height along canopy (H)
t = max(h) = top of canopy
F_{Nt} = flux to leaves from either nitrate and nitric acid or ammonium
a_0, a_1 = constant terms estimated from nonlinear regression
e_h = independently and identically Gaussian distributed errors.

The model was fitted with a nonlinear regression procedure, NLIN (SAS 1990), by using the derivative-free multivariate secant method. The starting value of wind speed parameter a_1 = - 1.2 was estimated as:

$$\text{Wind speed} = e[a_1(1- H_t/H_h)],$$

from wind speed information gathered during 1994 at the tower location (Miller and others 1996). A grid search was used to identify a starting value of a_0. Graphical and analytical analyses of residuals were conducted to determine if any model assumptions were violated.

Results

Model results indicate that deposition estimated at the top of the canopy is strongly related to deposition at lower canopy positions for individual ion models (*table* 1). Nonlinear R^2 (analogous but not equal to linear R^2) was high for both pollutants, indicating that conductance for both compounds at lower levels of the canopy is also strongly linked to conductance at the top of the canopy (*figs. 2a, 3a*).

The observed decline in flux is dominated by declines in wind speed from the top to the bottom of the canopy. Neither NO_3^- or NH_4^+ model a_1 parameters were significantly different from a_1 developed from wind speed information. The a_1 parameters from both models are lower than those reported for other systems (Davidson and Wu 1990), indicating that wind speed declines more linearly in this open forest relative to closed canopy forests.

Table 1 *— Model parameters and summary statistics for individual ion and combined ion deposition models.*[1]

Model	n	Mean	Std [2]	a_0	(SE) [3]	a_1	(SE)	RMSE [4]	R^2_N
NO_3^-	33	6.79	0.98	0.761	(0.05)	-1.39	(0.17)	1.03	0.81
NH_4^+	33	6.25	0.77	0.853	(0.05)	-1.04	(0.15)	1.28	0.80
N_{either}	66	6.52	0.88	0.791	(0.04)	-1.26	(0.14)	1.27	0.69

[1] All models use the equation: $F_h=a_0e[a_1(1-H_t/H_h)](F_t) + e_{hj}$.
All ion values are expressed as mg N/m^2/hr.
[2] Standard deviation.
[3] Standard error.
[4] Root mean square error.

Coefficients were not significantly different (at $p = 0.05$) for the NO_3^- and NH_4^+ models, indicating that it might be possible to develop a combined model. A single model was parameterized using NO_3^- and NH_4^+ expressed as mg N. Model results indicate that the single model, although marginally poorer for explaining variation, may be a simpler representation of the system. The combined ion model a_1 parameter estimate was nearly identical to that estimated using wind speed information ($a_1 = -1.2$). Individual model differences (*table 1*) may be due to smaller data sets used in the analysis.

Interpretation of a_0 is more difficult for these models. The parameter reflects a proportional reduction in the estimated foliar deposition at lower canopy positions relative to the canopy top. This constant reduction does not depend on canopy position and is similar to the proportion of total deposition retained on foliage after rinsing or throughfall occurs (Friedland and others 1991, Lovett and Lindberg 1993). Model differences are not, however, the result of canopy retained N; rather, the parameter may reflect an additional resistance to deposition in the canopy not present at the more open areas of the canopy top. Although data is sparse, this resistance may result from reduced turbulence occurring in the canopy interior that results in increased quasi-laminar layer depth and stability, which increases diffusive and interception resistance. Detailed measurements are needed, however, to confirm this interpretation.

Residual distribution was best for the combined model (*figs. 2b, 3b, 4b*). No significant heteroscadisticity or serial correlation was found for the models, nor was there any apparent relationship between canopy position and residual distribution. Analysis of residuals did suggest that individual ion models may overestimate deposition at low air concentrations (*figs. 2b, 3b*). This may be due to reductions in the ion concentrations near the ground caused by reduced atmospheric mixing. This overestimation was lessened in the combined model (*fig. 4b*).

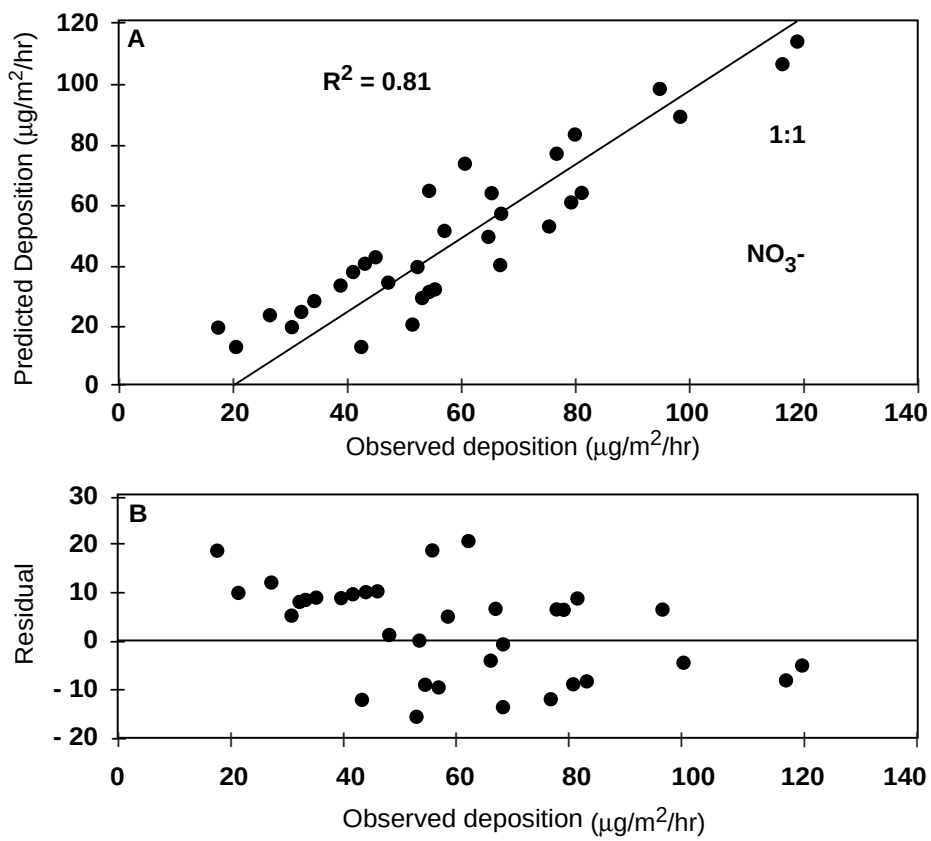

Figure 2 — Model results for predicting NO_3^- deposition at 24 m, 16 m, and 12 m on the canopy. Predicted deposition is compared with observed; the line represents a perfect model fit (1:1) and corresponds to deposition calculated at 29 m (A). Model residuals lack any pattern that would indicate poor model fit (B).

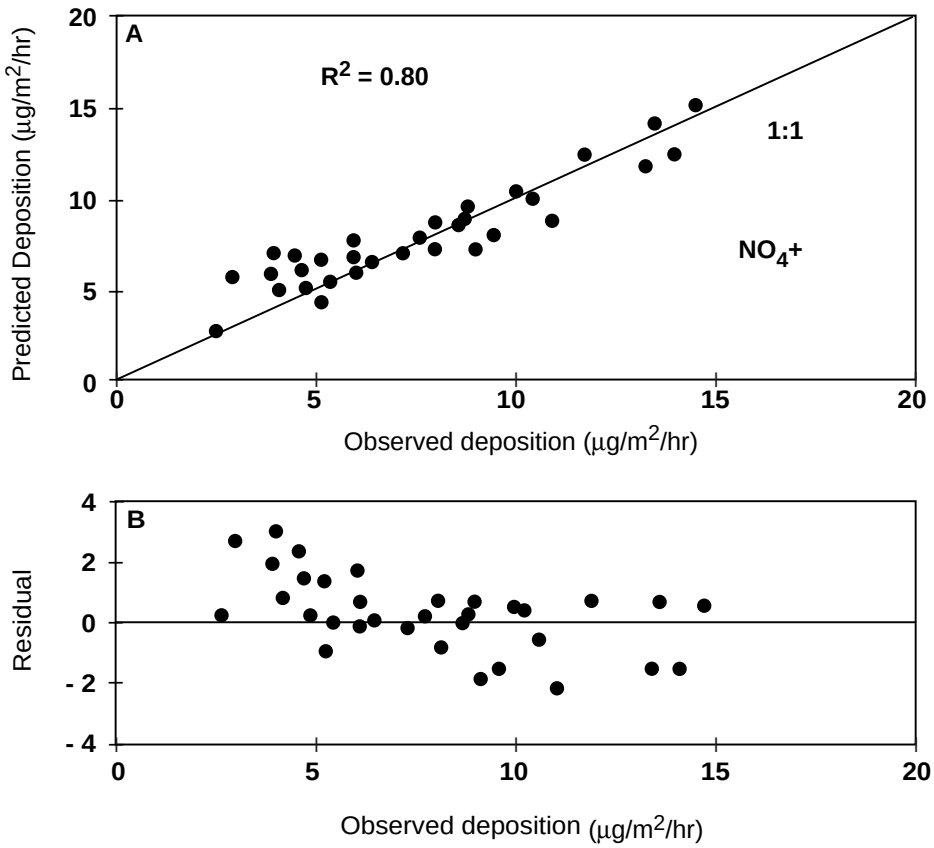

Figure 3 — Model results for predicting NH_4^+ deposition at 24 m, 16 m, and 12 m on the canopy. Predicted deposition is compared with observed; the line represents a perfect model fit (1:1) and corresponds to deposition calculated at 29 m (A). Model residuals lack any pattern that would indicate poor model fit (B).

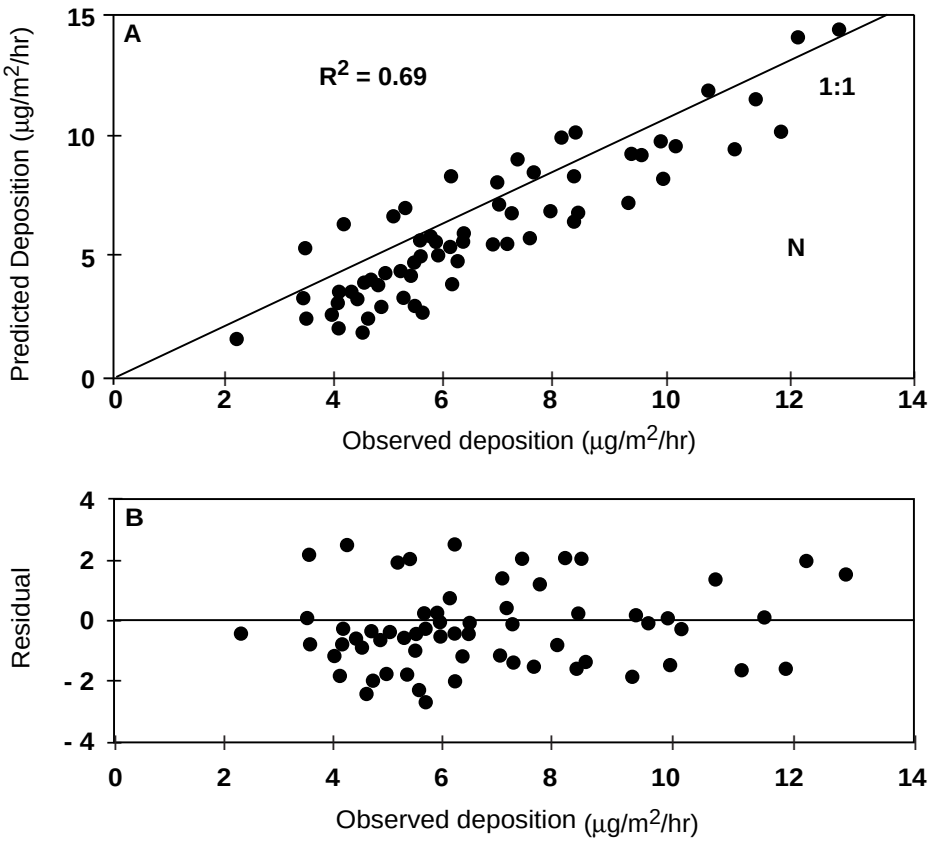

Figure 4 — Single model results for N deposition from ammonium and nitrate. Predicted deposition is compared with observed; the line represents a perfect model fit (1:1) and corresponds to deposition calculated at 29 m (A). Model residuals lack any pattern that would indicate poor model fit (B).

Conclusions

The estimators developed in this study have potential for reducing the necessary work in future studies to obtain estimates of whole canopy flux. The physically based models used in this study have high R^2, favorable residual properties, and interpretable parameters. Study results indicate that the models are applicable over a range of concentration values and pollution exposure seasons. The models need to be tested at other locations and for other tree species to verify their predictive ability. The scaffold tower data used for the analysis represents only a single point, and the foliage extraction information is most relevant for only two species, ponderosa and Jeffrey pine. Additional locations, especially in the western San Bernardino Mountains and the southern Sierra Nevada, need to be examined.

A single combined model may have application throughout the Transverse Ranges in southern California and the southern Sierra Nevada, since the chemical composition of N pollutants and forest types are similar for these areas. It may also have application to other open montane systems that lack vertical concentration gradients of pollutants and where the dominate form of aerosol N pollution is NH_4NO_3. Such systems may include coastal sage scrub, oak-grasslands, and chapparal.

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Indicators of Nitrogen Status in California Forests[1]

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Abstract

Indicators of ecosystem nitrogen (N) status are needed for monitoring and for identifying ecosystems that are at risk of becoming N saturated. The N chemistry of a number of plant, soil and hydrologic components were analyzed to assess the N status of mixed conifer forests across an N deposition gradient in the San Bernardino Mountains east of Los Angeles, California. All of the measured parameters at Camp Paivika on the western high-deposition end of the gradient indicated that the forest is N saturated, while forest stands on the eastern end of the gradient are N-limited. On the basis of these and other studies, the following parameters are recommended as indicators for monitoring forest N status: foliar nitrogen:phosphorus (N:P), foliar nitrate (NO_3^-), foliar growth response to N fertilization, soil carbon:nitrogen (C:N) ratio, NO_3^- in soil extracts or soil solution, and streamwater NO_3^-. However, foliar NO_3^- levels are more temporally sensitive than the other indicators mentioned, and foliar NO_3^- levels vary widely among plant species. Nitrate levels measured in KCl extracts or in soil saturation extracts were as effective as soil solution NO_3^- measurements for comparing soil NO_3^- concentrations among sites across the N deposition gradient. Extractable NO_3^- may be the preferred indicator of soil NO_3^- because it can be measured when soils are wet or dry, whereas soil solution cannot easily be obtained from soils with low water content. Elevated N trace gas emissions from soil (nitric oxide [NO] is dominant under the arid conditions of California) seem to be diagnostic of excess soil N, but field measurement of N trace gases (particularly NO) is not routine for most laboratories. Streamwater NO_3^- concentration is a highly useful integrative indicator of the net outcome of N cycling processes within a watershed. Temporal trends in streamwater NO_3^- are indicative of the N retentiveness of the watershed. Studying a combination of suitable indicators is recommended when evaluating forest N status.

Introduction

Indicators of ecosystem nitrogen (N) status are needed for monitoring and for identifying ecosystems that are at risk of becoming N saturated. The need for indicators is evidenced by the expanding geographic scope of forested areas over which elevated atmospheric N inputs are expected (Galloway and others 1994). Nitrogen deficiency or limitation is common in northern temperate forests (Vitousek and Howarth 1991), and undisturbed forests are generally considered to be highly conservative of N. However, many recent studies have shown that forests exposed to chronic N deposition often result in nutrient cycling processes that have been significantly altered by excessive inputs of atmospheric N. In many forests the levels of available soil N are in excess of biotic demand, and N losses from these forests can be high (Dise and Wright 1995, Riggan and others 1985). Such forests with available N over and above the biotic and abiotic retention capacities are considered N saturated (Aber and others 1989). Many current and recent studies have focused on the consequences of N deposition and enrichment on natural ecosystems that are adapted to N-limiting conditions. Water quality has deteriorated in many watersheds exposed to N air pollution as a result of high NO_3^-concentrations in streamwater. Although low-to-moderate N deposition may increase productivity, considerable evidence suggests that detrimental environmental effects will ensue in the long-term in forests with high levels of N inputs (Aber 1992).

Nitrogen deposition and ozone concentrations in the South Coast Air Basin (SCAB; Los Angeles, California, and surrounding counties) are among the highest in North America. Montane forests encompassing the SCAB are exposed to high concentrations of ozone and N pollutants (Bytnerowicz and Fenn 1996). Nitrogen and ozone air pollution are also on the rise in the western and southern portions of the extensive Sierra Nevada range, as a result of large urban and agricultural emission sources throughout California (Blanchard and others 1996). The effects

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of N deposition on forests within the SCAB may provide a preview of future scenarios in other western forests as urbanization and fossil fuel consumption continues to increase in western North America. The more extreme cases of air pollution exposure increase the likelihood that signals of air pollution effects will be more clearly discernible above the background variation inherent in natural ecosystems (Lang and others 1982).

In the San Bernardino Mountains (SBM) east of Los Angeles, California, a steep air pollution gradient spanning 55 km has been identified, with ozone and N deposition decreasing from west to east (Fenn and Bytnerowicz 1993, Miller and others 1986). Ozone injury to sensitive species has been observed for many years, while more recently the effects of chronic N deposition have also been documented (Fenn and others 1996). A number of nutrient pools and processes have been monitored at sites with high and low N deposition in the SBM to determine the impact of atmospheric N inputs on N cycling.

This paper reviews which nutrient cycling parameters may be useful as indicators of forest N status under the environmental conditions prevalent in California.

Selection Criteria for Indicators of Forest Nitrogen Status

Evaluating forest N status is challenging because N is a major constituent of all life forms and is cycled through a complex web of processes involving many biotic and abiotic mechanisms. Monitoring toxic pollutants or xenobiotics such as heavy metals or pesticides is in many ways a much simpler task. To evaluate the N status of a forest ecosystem, we recommend the use of a small set of reliable indicators that are easy to implement. Although numerous components of the ecosystem could be measured to provide information on forest N status in intensive research studies, the selection of indicators may be limited by practical considerations in monitoring programs. Suggested criteria for selecting suitable indicators for monitoring include the following:

- Accurate measure of site N status
- Lack redundancy
- Lack sensitivity to time of sampling or other environmental factors
- Sufficiently high signal-to-noise ratio
- Relatively standard sampling and analytical methods
- Cost effective (low cost/high information value)
- Definable range of "normal" values
- Clearly identifiable N excess and N deficiency.

Although selected indicators must accurately indicate site N status, evaluations of forest N status may be difficult to establish based on any one indicator. Ecosystem N status should be evaluated with a complementary set of nonredundant indicators so that the relative degree of N limitation, sufficiency, or saturation (excess) may be more firmly established. Vegetation-based indicators of N status are particularly useful because they function as biomonitors and allow site N status to be assessed by plant-environment interactions. Vegetation-based indicators help ensure that interpretations of site N status are biologically relevant. For some indicators, seasonal trends should be monitored to interpret indicator responses.

For sites at risk of N saturation, indicators should be included that can identify when available N exceeds biotic demand. Such indicators typically monitor parameters that are normally at background or low levels in N-limited systems. Sustained high activity levels or concentrations suggest that available N is in excess of biotic demand. Examples of indicators from the literature and from our studies (Fenn and others 1996) that seem to indicate excess plant-available N include foliar accumulation of nitrate (NO_3^-) (Stams and Schipholt 1990) or amino acids (Ohlson

and others 1995), elevated emissions of N trace gases from soil (Castro and others 1994, Sitaula and others 1995), high soil solution NO_3^- accompanied by high leachate losses of NO_3^- (Aber and others 1989), and above normal NO_3^- concentrations and fluxes in streamwater (Riggan and others 1985, Stoddard 1994). High levels of nitrate reductase activity (Norby and others 1989) may also be indicative of high NO_3^- availability or elevated atmospheric concentrations of nitric acid vapor (HNO_3). However, in the SBM, concomitant exposure to high ozone concentrations may interfere with enzyme activity, confounding the use of nitrate reductase activity as an indicator of excess N availability (Krywult and others [In press]).

However, the cardinal indicator or manifestation of N saturation in all ecosystem types, including California forests and chaparral, is increased and prolonged NO_3^- loss below the main rooting zone and in streamwater (Dise and Wright 1995). Changes in solution concentrations and fluxes as water passes vertically through forest ecosystems in precipitation, throughfall and stemflow, forest floor percolate, soil solution, surface runoff, and in streamflow have been referred to as solution or water chemistry profiles (Binkley and others 1982, Chorover and others 1994).

The N solution profile in a forest is highly indicative of the N status or degree of N saturation. In N-limited forests, nearly all of the solution N is retained within the vegetation or upper soil layers of the solution profile. However, in N-saturated forests, excess N is transported through the canopy and soil layers and exported mainly as leached NO_3^- and nitrogenous trace gases (Aber 1992).

Recommended Indicators in Arid Western Forests

Indicator selection in arid western forests is based on current understanding of N cycling in temperate forest ecosystems and on studies from air pollution gradients in the SBM (Fenn and others 1996) and San Gabriel Mountains (SGM) (Anderson and others 1988, Kiefer 1995, Riggan and others 1985). Valuable indicators of forest N status in the summer-dry conditions that predominate in much of western North America include:

- Soil NO_3^- (in soil solution or in soil extracts)
- Nitrate:ammonium (NO_3^-:NH_4^+) ratios in soil
- Soil carbon:nitrogen (C:N) ratios
- Nitric oxide (NO) emission levels from soil
- N-mineralization and nitrification rates
- Foliar nitrogen:phosphorus (N:P), C:N, and N:cation ratios
- Accumulation of NO_3^- in foliage of understory and overstory species
- Plant response to N fertilization
- Streamwater NO_3^- concentrations and fluxes
- Base saturation and soil pH.

Many of the indicators suggested for arid forests of the western United States have also been demonstrated to be effective in more humid temperate forests as well. However, indicator responses may differ depending on climate and ecosystem characteristics. For example, emission levels of nitric oxide (NO) are good indicators in western forests, while in more humid forests of the eastern U.S. or Europe, studies have shown that nitrous oxide (N_2O) emissions are much greater than NO emissions (Aber and others 1989). Temporal dynamics of streamwater NO_3^- fluxes may also be different in Mediterranean ecosystems (Riggan and others 1985). Nitrate in the soil solution can also build up to high concentrations during the dry season in sites with low or moderate N deposition as soil drying results in low volumes of soil solution with high ionic concentrations (Fenn and others 1996). As with all indicators, data from low-deposition sites or other reference data should be compared for interpreting indicator results.

Characteristics of Recommended Indicators

Soil Nitrate

In strongly N-limited systems, N cycling and plant uptake of N occurs mainly as ammonium (NH_4^+) (Aber 1992). As N saturation develops, plant-available NO_3^- increases in soil. Thus, the NO_3^-:NH_4^+ ratio in soil may be a useful indicator of site N status. Soil NO_3^- but not soil NH_4^+ increased with greater N deposition in the SGM (Kiefer 1995) and SBM (Fenn and others 1996). However, ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) trees prefer NO_3^- to NH_4^+ (Griffin and others 1995, Norton and Firestone 1996), contrary to the conventional belief that conifers prefer NH_4^+ (Adams and Attiwill 1982, McFee and Stone 1968). Soil NO_3^- and NH_4^+ can be measured in soil solution or in soil extracts (e.g., saturation or KCl extracts). Obtaining soil solution samples with the widely-used suction lysimeter method (Fernandez and others 1995) from the coarse-textured soils typical for these ecosystems can be very difficult during the dry summer growing season. We found that the soil centrifugation method (Soon and Warren 1993) worked well for obtaining soil solution samples from coarse-textured forest soils with water content near field capacity. However, in our studies in the SBM, NO_3^- levels measured in soil saturation extracts and in KCl extracts were equally effective as soil solution NO_3^- measurements for comparing soil NO_3^- concentrations at sites across the deposition gradient. Soil extracts can be obtained from soils collected at any time, regardless of soil water content or soil texture, and in that regard are probably more practical as an indicator in a monitoring program. Another recent study concluded that KCl extracts performed similarly to suction cup lysimeters in quantifying soil NO_3^- (Djurhuus and Jacobson 1995).

Streamwater Nitrate – a Watershed-Level Indicator

Streamwater ionic flux is a very useful integrator of overall ecosystem integrity and serves as a watershed-level indicator of ecosystem nutrient loss or conservation. Nitrogen saturated watersheds export above-normal amounts of NO_3^- once the capacity for N retention by the biota and soil chemical fixation mechanisms are exceeded (Stoddard 1994). Temporal or seasonal trends in NO_3^- export in streamwater reveal the N retention capacity at the watershed scale and are a consequence of the many nutrient cycling processes occurring within the watershed. A brief and relatively small NO_3^- loss from temperate forest watersheds is expected during the spring water flow from the watershed brought on by snowmelt. During this early spring season water flows are greatest and soil temperatures are still low, and plant and microbial nutrient demand are insufficient to retain all of the dissolved N. However, a state of N saturation is indicated when NO_3^- concentrations begin to increase over this "normal" background level and when losses are prolonged. Peak flows in N-enriched California ecosystems seem to occur during major winter storms or during subsequent snowmelt periods when more complete washout of the soil profile occurs (Riggan and others 1985).

Monitoring streamwater NO_3^- dynamics throughout the hydrologic year will provide the most information on watershed N processing and losses. However, with an understanding of the temporal dynamics of streamwater chemistry within a given set of environmental conditions and ecotypes, sampling windows may be identified when NO_3^- in streamwater is indicative of above-normal N losses or N saturation. For example, high streamwater NO_3^- concentrations during the summer when the terrestrial and aquatic biota are normally capable of retaining available N may suggest an N-saturated condition. When the objective is to compare annual N inputs and outputs for a watershed, streamwater flow volumes and NO_3^- concentrations should be measured.

Soil C:N Ratio

The soil C:N ratio is a useful indicator of N enrichment in forests. Soil C:N ratio is less temporally and environmentally sensitive than other indicators, such as NO_3^-

levels in soil and foliage, nitrate reductase activity, amino acid concentrations in foliage, trace gas emissions from soil, and streamwater NO_3^- concentrations. The relative stability of soil C:N enhances the utility of this indicator and reduces the odds of fluctuating environmental conditions or nutrient cycling processes confounding interpretation of indicator results. However, the fire history of monitored sites must be considered when implementing any N status indicator (Chorover and others 1994), including C:N ratio. Percent C and N losses from burned plant material are highly similar (Raison and others 1985), and soil C:N ratios appear to be little changed after fire (Carreira and others 1994, Raison 1979). However, total C and N pools can change dramatically in burned sites (Raison 1979), suggesting the need for caution when monitoring sites have burned recently.

Foliar Nitrate

Nitrate is stored in the vacuole (Granstedt and Huffaker 1982) but does not accumulate in foliage until the growth requirements for N are satisfied (Zhen and Leigh 1990) and the capacity for NO_3^- assimilation is exceeded (Lee and others 1986). Nitrate accumulation in plants can be a very useful indicator of N excess because NO_3^- does not accumulate in foliage unless plant demand for N has been met. Nitrate concentrations in foliage of pine, oak and fern were much higher at an N-saturated site compared to an N-limited site in the SBM (Fenn and others 1996). Nitrate accumulation was especially high in bracken fern (*Pteridium aquilinum* var. *pubescens* Underw.), a species that has been shown to accumulate NO_3^- under conditions of high N deposition (Stams and Schipholt 1990).

Amino acid levels in foliage may also be an effective indicator of N status or storage of excess available N (Ericsson and others 1993, Ohlson and others 1995), but this has not yet been investigated in western forests. However, amino acid analysis is not likely to be routine for many plant and soil analysis laboratories.

Foliar Nutrient Ratios

Foliar nutrient ratios are often more reliable and informative indicators of plant nutrient status than nutrient concentration alone (Ericsson and others 1993, Hüttl 1990, Mohren 1986, Zinke 1980). In three separate studies, foliar N:P ratio has been found to increase with increasing N deposition in the mountains of southern California (Fenn and others 1996, Poth and others 1991, Zinke 1980). Foliar C:N ratio is also a useful indicator of site N status (Fenn and others 1996). Nitrogen:cation ratios in foliage have not been sufficiently tested as N status indicators in California but may also be useful indicators of N status or of nutrient imbalances (Hüttl 1990).

Elaborate nutritional diagnosis systems have been developed based on a suite of nutrient ratios. The Diagnosis and Recommendation Integrated System (DRIS) is an example of a widely used system in agriculture (Walworth and Sumner 1987), which has also been applied to a number of forest species (Binkley and others 1995, Riiters and others 1992, Shumway and Chappell 1995). DRIS uses nutrient ratios in high-productivity sites as a guide to optimum ratios that could be achieved through fertilization of less productive sites. Before DRIS can be used for a particular plant species, however, DRIS "norms" have to be determined for that species based on nutrient data from a large survey of the species in question (Needham and others 1990). DRIS norms have not been developed for ponderosa pine or most other western forest species; but if they are developed, DRIS could prove to be a very effective tool for evaluating site N status and for identifying other nutrients that may be limiting or in excess. This could be particularly useful in cases where excess N may induce deficiencies of other nutrients (Houdijk and Roelofs 1993, Mohren and others 1986, Schulze 1989).

Plant Response to Fertilization

Studies of plant response to N amendment treatments is an effective technique for evaluating site N status, although this requires treatment application and

subsequent comparisons of untreated control plants and fertilizer-treated plants. Vector analysis is a simple, but elegant, graphical technique by which tree N status can be determined from physiological responses to fertilization under field conditions (Haase and Rose 1995, Timmer and Stone 1978, Weetman and Fournier 1982). Fertilization of individual trees with N before the growing season followed by end-of-season analysis of tree response in terms of foliar biomass, N content, and N concentration (vector analysis) can provide clear indications of the N status of forest trees. The value of the fertilization/vector analysis method, however, is that field grown trees can be used to biomonitor and interpret site N status — at least for the selected tree species. Vector analysis studies in the SBM and SGM suggest that ponderosa and Jeffrey pine (*Pinus jeffreyi* Grev. & Balf.) trees in stands with N deposition from ca. 6 to 16 kg ha^{-1} yr^{-1} are N deficient, while N was not limiting at sites with more than 30 kg ha^{-1} yr^{-1}, (Kiefer and Fenn [In press]).

Nitrogen Mineralization and Nitrification

In N-enriched sites, N-mineralization and nitrification rates are usually increased (Aber and others 1989, Van Miegroet and others 1992). Under conditions of excess available N, the need for N conservation is reduced, and canopy N turnover, net N-mineralization and net nitrification increase. Nitrogen mineralization can be measured in situ with the buried bag technique (Pastor and others 1984), or potential N-mineralization and nitrification can be assayed under controlled conditions. Both aerobic and anaerobic assays can be used, although nitrification rates are not determined in anaerobic incubations (Binkley and Hart 1989). The laboratory assays provide information on potential net N-mineralization and nitrification rates, which are measures of the soil capacity to supply N, while eliminating confounding environmental influences on soil N cycling. Assays performed under standard conditions allow for comparisons of soil N status for different sites and remove site environmental constraints (e.g., temperature and moisture limitations) that are constantly changing. N mineralization rates were higher in sites with greater N deposition in the SBM and SGM in southern California (Fenn and others 1996, Kiefer 1995).

Fluxes of N Trace Gases

Fluxes of trace gases from soil often increase with greater soil N supply (Castro and others 1994, Sitaula and others 1995). In forests in eastern North America and in Europe, N_2O emissions appear to be much greater than NO emissions (Aber 1992), although NO fluxes are frequently not measured. In California soils, NO fluxes are typically much greater than N_2O fluxes (Anderson and others 1988, Davidson 1993). NO emissions were much higher in a high deposition site compared to a low deposition site in the SBM (Fenn and others 1996). Nitric oxide appears to be diagnostic of excess available soil N in western coarse-textured soils. A major drawback of NO as an indicator of soil N status is the specialized equipment required to measure NO. Under conditions where N_2O emissions are sufficiently high (finer textured soils with more than 60 percent water-filled pore space) (Davidson 1993), simple flux chambers could be installed on the forest floor and headspace samples collected, transported to the laboratory, and N_2O quantified via gas chromatography. Although N trace gas fluxes seem to be indicative of soil N status, other indicators may be easier to implement for the general plant and soil chemical analysis laboratory.

Soil pH and Base Saturation

Decreased base saturation and soil pH in forests with high N inputs does not prove cause and effect because other stand development factors can also cause soil acidification and decreased base cation levels in soil. However, if other indicators provide evidence of N saturation, decreasing base saturation and soil pH may provide supporting data of the effects of N enrichment. Decreased base saturation

is a result of cation leaching as counter-balancing ions for leached NO_3^-. In the SBM soils in sites with high N deposition had lower base saturation and pH than soil in sites with low N deposition (Fenn and others 1996).

Summary

To assess the N status of a forest ecosystem several reliable and complementary indicators should be selected. Under the arid conditions of much of the western U.S., recommended indicators include the following:

- NO_3^- concentrations in soil and foliage
- Foliar nutrient ratios such as N:P
- Soil C:N
- Streamwater NO_3^- concentrations and fluxes
- Plant response to N fertilization or vector analysis.

The N status of a forest ecosystem can also be assessed in the context of solution chemistry profiles that consider the sequential processing of N in precipitation, throughfall and stemflow, forest floor percolate, soil solution, surface runoff, and streamflow. Nitrogen mineralization and nitrification rates and trace gas fluxes from soil (especially NO) are also potentially useful indicators of N status. With further research, amino acid levels in foliage and the Diagnosis and Recommendation Integrated System (DRIS) method could also be used in evaluating site N status. However, DRIS norms would need to be developed for the particular plant species.

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Nitrogen Deposition Effects on Coastal Sage Vegetation of Southern California[1]

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Abstract

*The coastal sage scrub (CSS) vegetation of southern California has been declining in land area and in shrub density over the past 60 years or more, and is being replaced by Mediterranean annual grasses in many areas. Although much of this loss is attributable to agriculture, grazing, urbanization and frequent fire, even protected areas have experienced a loss in native shrub cover. Nitrogen (N) deposition has not previously been examined as a contributor to CSS decline, but up to 45 kg/ha/yr are deposited in the Los Angeles Air Basin. Several mechanisms were examined by which atmospheric N deposition might affect the shrubs and promote growth of weeds. Field nitrogen fertilization studies at sites of high and low deposition showed that most of the abundant native and introduced species had increased growth after fertilization in the low deposition site, but in a high deposition site only one weedy species, small-podded mustard (*Brassica geniculata *(Desf.) J. Ball), responded to N fertilization. Greenhouse studies showed that both shrubs and weeds had high plasticity in their growth response to N fertilizer, an unexpected result for the shrubs. Preliminary competition studies indicated there was no change in the relative competitive ability of the shrubs or grasses after fertilization. However, negative effects of high N have been detected on the growth and survival of the shrubs. Greenhouse grown California sagebrush (*Artemisia californica *Less.) began to senesce at 6 to 9 months when fertilized with 50 μg N/g soil. This soil N concentration corresponds to extractable N levels in polluted sites, while levels are typically less than 10 μg/g in unpolluted sites. Another source of damage to plants can be cuticular lesions caused by nitric acid, but how nitric acid affects CSS leaves is unknown. Fumigation of pine needles with high ambient levels of nitric acid caused cuticular lesions and stomatal collapse, as well as modifications to nitrogen assimilation pathways. The preliminary evidence suggests that CSS vegetation may decline due to elevated nitrate levels in the soil, and additional studies are need to test effects of ambient nitric acid on CSS leaves.*

Introduction

The coastal sage scrub (CSS) vegetation of southern California has been declining in land area and in shrub density over the past 60 years or more, and is being replaced by Mediterranean annual grasses (Davis 1994, Freudenberger and others 1987, Minnich and Dezzani [In press], O'Leary and others 1992, Zink and others 1995). Although much of this loss is attributable to agriculture, grazing, urbanization and frequent fire, even protected areas have experienced a substantial loss in native shrub cover (Minnich and Dezzani [In press]). Nitrogen (N) deposition has not previously been examined as a contributor to CSS decline, but up to 45 kg/ha/yr are deposited in the Los Angeles Air Basin (Bytnerowicz and Fenn 1996). Nitrogenous compounds in polluted air affect even those tracts of land that have been set aside as reserves, which are in many cases surrounded by urbanization in southern California (O'Leary and others 1992). The coastal sage scrub is of particular interest to conservationists because it supports some 200 sensitive plant species and several federally listed animal species (DiSimone 1995, O'Leary 1989). We need to understand how to manage lands that are influenced by urban air pollution, and whether the biotic communities of these lands can be preserved.

Nitrogen deposition is known to cause vegetation type conversions in other countries, notably the Netherlands (Bobbink and Willems 1987), which have up to 85 kg/ha/yr of N deposited, the highest measured in the world. Nitrogen deposition to reserves is a conservation problem in the Netherlands, where heathlands and species-rich pastures are turning into species-poor grasslands (Asman and others 1989, Bobbink and Willems 1987). The high rates of N deposition in southern California have caused increased soil fertility and surface litter decomposition rates in mixed conifer forests (Fenn 1991), but less is known about

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vegetation change with elevated N deposition in this region. The coniferous forests of the San Bernardino Mountains exhibit symptoms of N saturation, such as high N:P ratios in leaf tissue, high soil N, high rates of N loss from the ecosystem, and others (Fenn and others [In press]). If the relatively productive forests of the Los Angeles Air Basin are N-saturated, then less productive vegetation such as CSS that is receiving similar levels of deposition would also surely be saturated. Less productive vegetation may become N saturated more rapidly than highly productive vegetation because the ratio of deposited N to plant biomass is greater, as suggested by Aber and others (1989, 1992) concept of anthropogenic nitrogen saturation. In addition, long-lived trees may respond less rapidly than shorter-lived shrubs, so we may expect to see a more rapid response in vegetation change in CSS than in nearby forests.

Here we examine the mechanisms by which deposited nitrogen might cause CSS shrubs to be replaced by Mediterranean annual grasses. We present three hypotheses with preliminary data to explain how nitrogen deposition may affect CSS species. Stated in the null form they are 1) CSS shrubs are equally plastic in their growth response to N as are introduced grasses, 2) CSS shrubs are equally competitive as the grasses after N fertilization, and 3) CSS shrub growth and mortality are not affected by high N levels.

Effects of N Deposition on Coastal Sage Scrub Vegetation

We have done series of field and greenhouse experiments on the effects of N deposition on CSS species. The field experiments were done along a gradient of atmospheric N concentrations, using one site with relatively high and another with relatively low N air concentrations for intensive measurements (*fig. 1*). The high deposition site is at Box Springs Mountain near the University of California, Riverside at an elevation of 670 m, and the low deposition site is some 60 km to the south at the Lake Skinner Reserve, elevation 540 m. Longterm climate data are not available at either site, but the city of Riverside, adjacent to Box Springs Mountain, receives 280 mm precipitation annually. We are still working out the actual deposition rates, which are likely lower than the estimated high values of 30 kg N/ha and more recently 45 kg N/ha at Camp Paivika on the western end of the

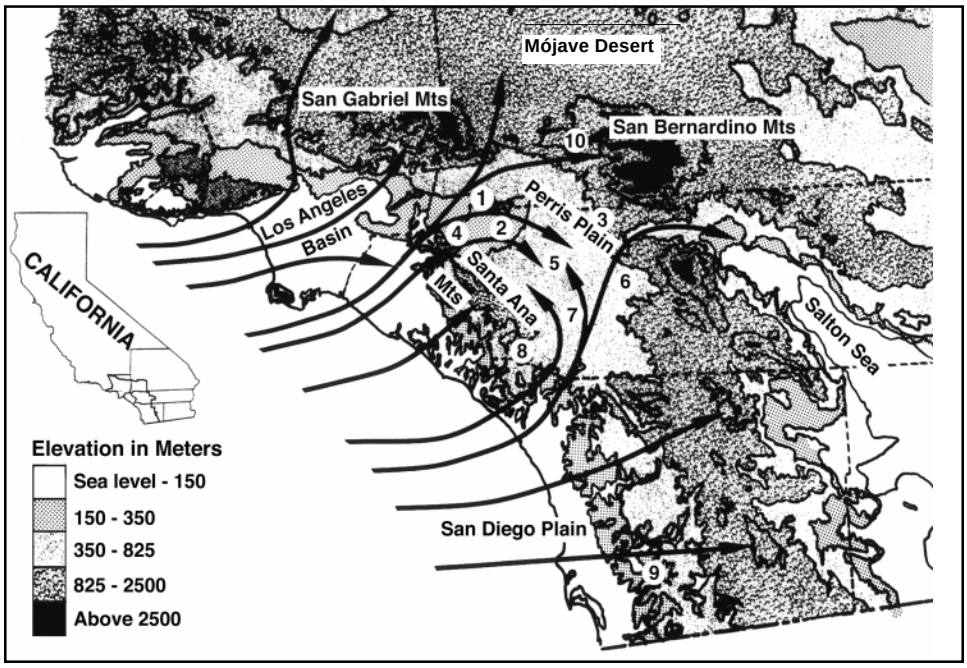

Figure 1 — Sample sites on the Perris Plain referenced in the text. The site of highest measured N deposition is Camp Paivika (10) and the site with lowest air concentrations is (9) at Mission Trails Park in San Diego. Arrows show air flow patterns, with polluted air flowing inland from Los Angeles and cleaner air flowing inland from other coastal sites. A convergence zone of polluted and clean air occurs in the middle of the Perris Plain.
1 = Jurupa Hills,
2 = Mockingbird Reservoir,
3 = Box Springs Mountain,
4 = Lake Matthews,
5 = Motte Rimrock Reserve,
6 = Simpson Park in Hemet,
7 = Lake Skinner,
8 = Santa Margarita Ecological Reserve,
9 = Mission Trails Park in San Diego,
10 = Camp Paivika, San Bernardino Mountains.

San Bernardino Mountains (Bytnerowicz and Fenn 1996, Fenn and Bytnerowicz 1993). Peak air concentrations were measured with an annular denuder system in August, 1994 as 31.0 µg/m³ NO_3^- plus 8.6 µg/m³ NH_4^+ at Box Springs Mt., and 13.9 µg/m³ NO_3^- and 3.9 µg/m³ NH_4^+ at the Lake Skinner Reserve (*fig. 2*). Sulfur was relatively low across the gradient, as is the case in other western air pollution measurements (Bytnerowicz and others 1987). The vegetation at both sites is CSS, with a higher proportion of introduced grasses on Box Springs Mountain than at Lake Skinner (Minnich and Dezzani [In press]). Both sites are on granitic soils. Additional sites were used for less intensive measurements.

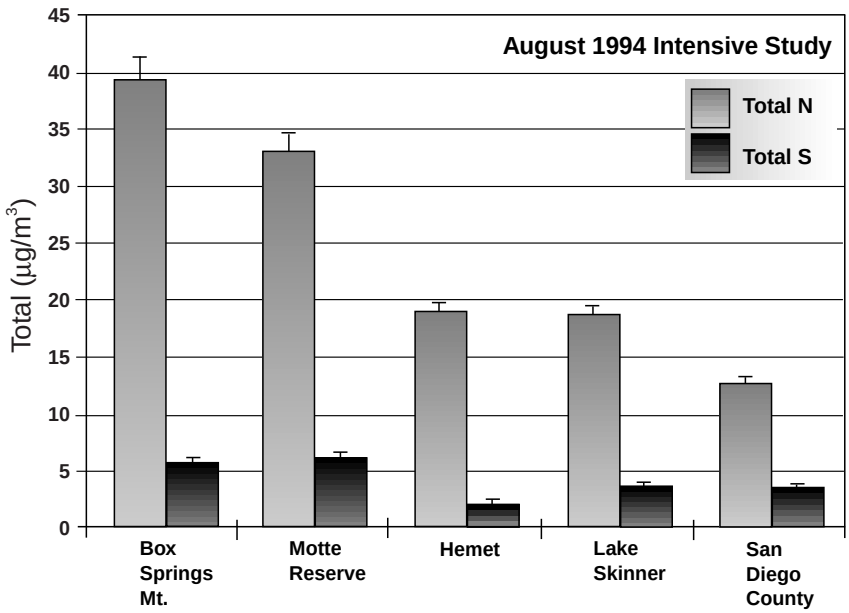

Figure 2 — Nitrogen (NO_3^--N and NH_4^+-N) and total S (SO_4^{2-}-S) concentrations in the atmosphere at five sites representing a gradient of air pollution from Box Springs Mountain in Riverside and southward to San Diego during August 1994. Location of sites is shown in *figure 1*.

In addition to air measurements, soil samples were taken along the gradient of pollution that also included sites to the northwest of Box Springs Mountain (*fig. 3*). The top 2 cm of soils of the Jurupa Hills had as much as 86 µg/g of extractable N in the form of nitrate plus ammonium (*fig. 3*), with nitrate predominating at the more polluted sites. Box Springs Mountain had 44 µg/g, and Lake Skinner had 8 µg/g of extractable N. These soils were collected in September during the dry season, when soil N accumulates because plants have senesced and are no longer taking up N. The soil N measurements confirm the gradient of N pollution by showing that the soils also accumulate N. We do not yet know if the soil N is higher because of accumulated N deposition, or because of increased mineralization that is induced by N deposition (Fenn and Dunn 1989). However, such high concentrations of N in the soil are likely to affect the plant community, which at each of these sites is CSS with an understory of annual grasses. At some sites, such as Box Springs Mountain and the Jurupa Hills, the annual grasses have become dominant with interspersed patches of shrubs. We explore the mechanisms to explain how the vegetation may change from shrubland to grassland after N deposition.

Plasticity Hypothesis

For one species to replace another in a high N soil, it must have a greater response to N, or in other words be more plastic in response to changes in resources (Jennings and Trewavas 1986). To test this hypothesis, we did N fertilization experiments in the field and the greenhouse. In the field, we fertilized plots at the Box Springs Mountain and the Lake Skinner sites. Both sites had burned in November 1993, and the fertilizer treatments included both burned and unburned vegetation. Each site had ten 5 by 5 m² plots and N was applied at a rate of 60 kg N/ha as NH_4NO_3 in two doses of 30 kg/ha each in February and March, 1994. Plant response was

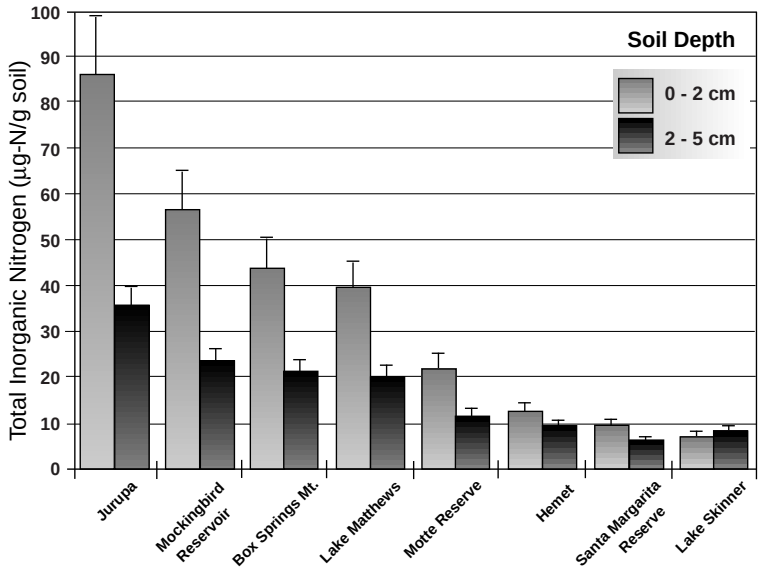

Figure 3 — Extractable soil nitrogen (NO_3^- -N and NH_4^+ -N) from sites on a nitrogen gradient in September 1995. Soil cores were divided into the upper 2 cm and lower 2 to 5 cm. Location of sites is shown in *figure 1*.

evaluated in May using non-destructive percent cover data. The unburned plots did not respond to fertilizer at either site with one exception described below, so results of fertilization trials are shown only from the burned plots (*table 1*).

At Box Springs Mountain only one introduced forb, small-podded mustard (*Brassica geniculata* [Desf.] J. Ball), responded significantly to N fertilizer on the burned plots, and this was also the only species that responded on the unburned plots (data not shown, but small-podded mustard increased from 17 to 29 percent after N fertilization on unburned plots). None of the other introduced grasses and none of the native species responded significantly at Box Springs Mountain in the burned plots (*table 1*). The introduced grasses included species in the genera wild oats (*Avena*), brome (*Bromus*), fescue (*Vulpia*) and split grass (*Schismus*), while the native species included a diverse mixture of some 70 species.[5] By contrast, at Lake Skinner almost all of the categories of species, both native and introduced, responded to fertilizer, with the exception of the introduced forbs (*table 1*). Small-podded mustard occurred infrequently at Lake Skinner, with < 5 percent cover. The total percent vegetative cover was greater in fertilized plots at both sites, but was higher overall at Box Springs Mountain. These results suggest that the plants are N deficient at Lake Skinner, and N saturated at Box Springs Mountain, with the exception of small-podded mustard, which continued to grow and take up N after fertilization. Leaf tissue N of small-podded mustard increased from 3.2 percent to 4.2 percent after fertilization. This suggests that small-podded mustard may be one of the "winners" in the high N deposition zones, as it can take advantage of additional N, even when soil N is already high. The results from the Lake Skinner site suggest that both native and introduced species are plastic in their responses to N, whereas our original hypothesis was that only the introduced species would be highly responsive to N, or "nitrophilous."

The responses in the field may have been influenced by a number of factors, especially competition in a complex community. To understand the responses of native and weedy plants under more controlled conditions, we performed greenhouse N fertilizer trials of monocultures of three introduced annuals (wild oats [*Avena fatua* L.], red brome [*Bromus rubens* L.], and small-podded mustard), and three native shrubs (California sagebrush (*Artemisia californica* Less.), brittlebush (*Encelia farinosa* Gray.) and California buckwheat (*Eriogonum fasciculatum* Benth.). The plants were grown in 3.5l pots in native soil amended with 0, 10, 50 and 100 µg/g of N as NH_4NO_3. The soil was collected from the Motte Reserve after scraping off the top 5 cm of soil. After 4 month's growth in the greenhouse, the native shrubs were just as plastic in their responses to N as were the introduced

[5] Unpublished data on file, Department of Botany and Plant Sciences, University of California, Riverside.

annuals (*figs. 4 and 5*). In fact, the annuals tended to saturate at 50 µg/g, while the shrubs had continued increased growth to 100 µg/g. This was an unexpected result, as native wildland species are typically thought to have low responses to nutrient additions compared to weeds and crop plants (Chapin 1980). Clearly the native shrubs are adapted to rapid growth during the brief 4 to 6 month rainy period, at which time they likely take up nutrients as rapidly as possible. The growth period is brief in Mediterranean climates, which have a moist winter and spring but the other seasons are dry. However, in an unpolluted situation, the loading of available N in the soil would not be as great as after N deposition. Our field observations showed that available N built up as high as 86 µg/g in the upper 2 cm of soil (*fig. 3*) during the dry season, an amount that would be available to newly growing seedlings at the beginning of the next rainy season. Both the native shrubs and the introduced grasses are apparently able to take advantage of this high soil N.

These results were unexpected and not only confound our notions of how plants behave in the wild (the paradigm states they should have slow rates of nutrient uptake and growth), the results also do not explain why N deposition would shift the vegetation from a shrub-dominated to a grass-dominated type. An alternative hypothesis is that competition for N occurs between the grasses and shrubs that is skewed in favor of the grasses.

Table 1 *— Percent cover of vegetation in nitrogen fertilized and unfertilized treatments in burned plots at the high N (Box Springs) and low N (Lake Skinner) deposition sites.*

Species	BOX SPRINGS N Fert.	BOX SPRINGS No N		LAKE SKINNER N Fert.	LAKE SKINNER No N	
Brassica geniculata	17.1	9.0	([1])	0.0	0.0	n.s.
Bromus rubens	15.6	13.5	n.s.[2]	4.5	2.0	P = 0.07
Other exotic grasses	20.1	14.7	n.s.	10.5	4.2	([1])
Other exotic forbs	27.5	26.3	n.s.	19.8	18.9	n.s.
Native forbs	20.7	19.2	n.s.	37.5	22.7	([1])
Native shrubs	0.0	0.1	n.s.	11.1	4.9	([1])

[1] Significantly different using a t-test at $P < 0.05$.

[2] n.s.=not significantly different.

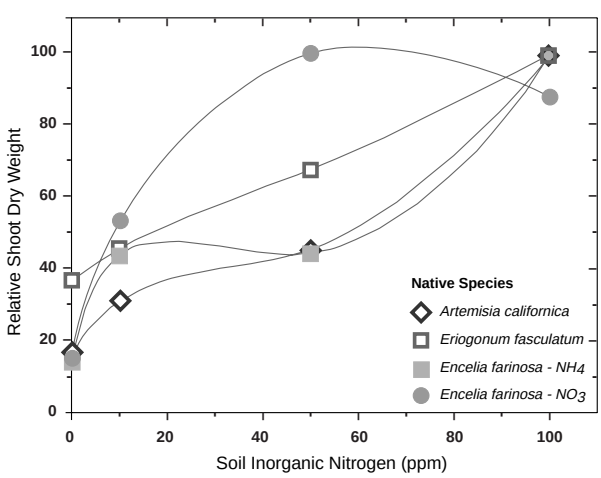

Figure 4 — Relative shoot dry weight of three native shrub species subjected to levels of 0, 10, 50, and 100 µg/g N as NH_4NO_3 in the soil after 4 months of growth. In the case of brittle- bush (*Encelia farinosa*), N was added either as NH_4Cl or as $Ca(NO_3)_2$.

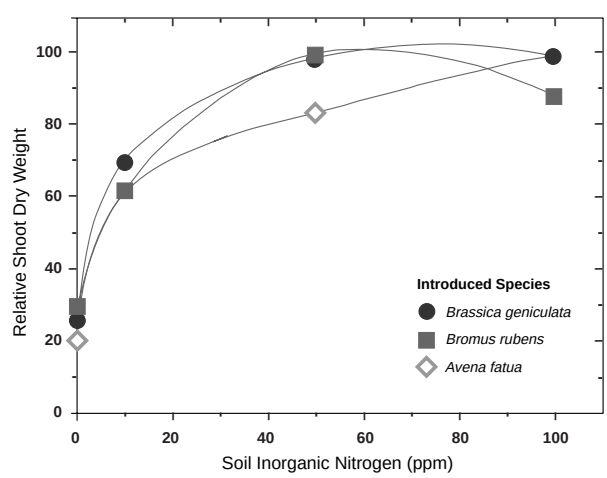

Figure 5 — Relative shoot dry weight of three introduced weed species subjected to levels of 0, 10, 50, and 100 µg/g N as NH_4NO_3 in the soil after 4 months of growth.

Competition Hypothesis

To determine whether N shifts the competitive balance between the grasses and shrubs, we initiated a competition experiment in a patch of introduced grassland that was once dominated by CSS species. This research was done at the Motte Rimrock Reserve about 24 km south of Riverside and intermediate in nitrogen deposition to the Box Springs and Lake Skinner sites. In a blocked experimental design we weeded grasses from ten, 1.2 m^2 plots and left an additional ten plots as controls dominated by the introduced grasses red brome and foxtail fescue (*Vulpia megalura* Rydb.) A few introduced forbs were also present, mainly species of storksbill (*Erodium*), but few native species. One-half of the cleared plots and one-half of the grassy plots were fertilized with two doses of NH_4NO_3 at the rate of 30 kg N/ha each time in spring 1995. Nine seedlings of California sagebrush were planted in each of the plots in a 2 by 2 factorial design to test two levels of N (0 and 60 kg/ha) and two levels of competition (with and without grasses) on the growth of California sagebrush. The seedlings were spaced 30 cm apart, so they did not interfere in aboveground growth during the first growing season. Because we did not wish to harvest seedlings during the first year, we did non-destructive measurements of height and width to calculate shrub volume. Volume was calculated by assuming that the shape of California sagebrush is spheroid.

The grass competition proved overwhelming for California sagebrush, which had only about one-ninth the volume in the grass plots compared to the cleared plots, with or without N fertilizer (*fig. 6*). Nitrogen did not shift the balance of competition in favor or against California sagebrush, it only increased the growth of California sagebrush with and without grass competition. Increased growth of California sagebrush after fertilization should be a benefit, if this means that it can survive competition from the grasses. But in fact most of the California sagebrush seedlings in grass plots died during the normal summer drought that followed these measurements, while many survived in the cleared plots. With or without N the seedlings that experienced grass competition were still tiny, and apparently not large enough, or with roots too shallow, to survive the summer drought. On the basis of this experiment, we cannot expect that areas with N deposition will have greater shrub seedling survival. In fact, we noted mortality of mature shrubs in areas of high N deposition, and proceeded to examine potential negative effects of N on shrub growth.

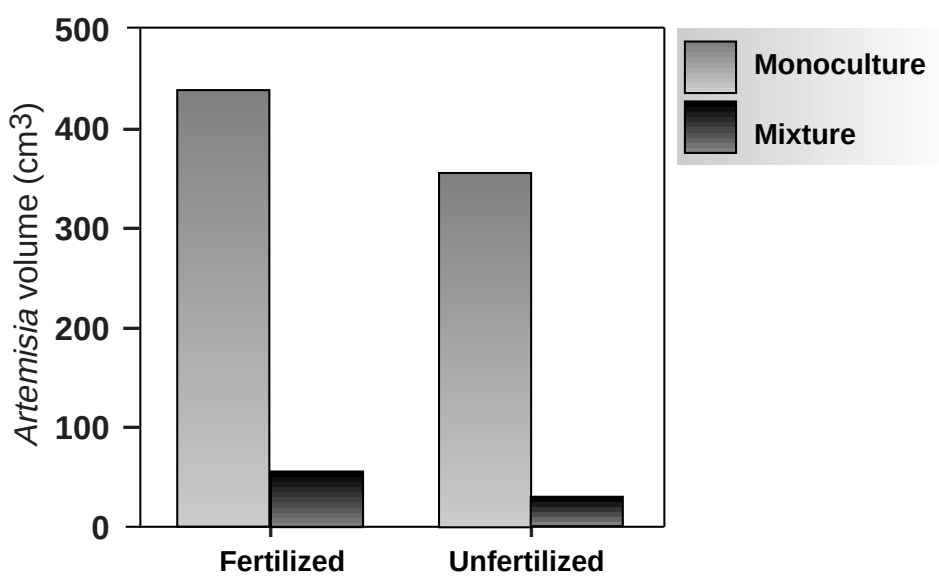

Figure 6 — Volume measurements of California sagebrush (*Artemisia californica*) in weeded (monoculture) and grassy (mixture) plots in the field after 3 months of growth, fertilized with 60 kg/ha N as NH_4NO_3 or unfertilized.

Negative Plant Response to Nitrogen Hypothesis

Because the plasticity and competition hypotheses did not explain why N might cause CSS decline, we continued to test alternative hypotheses. Shrubs in pollution-impacted urban areas have been dying, as can be seen by a walk through Box Springs Mountain County Park and other local reserves. Some 10 percent of the shrubs in the polluted areas that we marked for experimental purposes died during two growing seasons, but we did not experience shrub loss at the Lake Skinner site. We do not yet have an estimate of the rate of mortality on a larger scale, but Minnich and Dezzani [In press] have shown up to a 90 percent loss in shrub cover since the 1930's in urban areas of the Perris Plain (Riverside, California and southward). Shrub loss is lower in the more rural southern Perris Plain. They did their analyses from historic plots in CSS collected by the USDA Forest Service in the 1930's, which they resampled in 1993. Although they showed an increasing pattern of shrub loss toward the urban areas, some factors confounded the results, such as changes in soil type and CSS stand age since the most recent fire at the time of sampling.

Wellburn (1990) explained that N deposition may more frequently harm than fertilize plants, especially in the form of N oxides. We did a series of experiments to determine if elevated N levels might have negative effects on the shrubs. One was a greenhouse fertilization experiment with California sagebrush, where the seedlings were fertilized with 50 µg N/g as NH_4NO_3. Seeds were planted in a soil of low N, and after they were 3 to 4 months old they were fertilized with the 50 µg N/g soil. They were then fertilized every 2 months to maintain high soil N. During the first 6 months the seedlings grew rapidly, as was noted above. Senescence of individual branches at 6 months, and then complete mortality of all seven replicate plants between 6 and 9 months occurred. At 9 months, and even at 1 year, all of the plants that received low N fertilizer levels were still healthy. These negative effects are difficult to explain, as they appear to be a toxic effect. The soil N we maintained in the greenhouse was not as high as the highest levels we measured in the field, up to 86 µg/g in soils of polluted sites during the dry season. The greenhouse experiment was designed to maintain a concentration of 50 µg/g in the soil during the experiment, with two to three fertilizer additions during the 6 to 9 months. In the field, such a high level would build during the dry season and be maintained only until the beginning of the next rainy season. When vegetation begins to grow again in response to rain, the plants take up the available soil N, and our measurements showed that extractable soil N dropped to 10 µg/g even in the most polluted sites.[6] Leaching and denitrification might also remove some of the deposited N from the soil, although both of these are probably minor components of the N cycle in this semiarid shrubland. Such high levels of soil N were not maintained throughout the year, so high soil N would not cause as high a rate of shrub mortality in the field as in the greenhouse. But these results do suggest that, over time, high soil N may be a cause of shrub mortality.

The mechanisms by which shrub mortality occurred under high soil N is not known. Horticulturists have long known that native species from soils of low fertility will have short lifespans in a fertile garden, and advise that low fertilizer levels be used for native California shrubs (Keator 1994). We can discuss a number of alternative explanations for the adverse response of native shrubs to high N. It is likely that California sagebrush was taking up large quantities of N, because it has evolved only to take up this limiting nutrient in the soils, not to exclude it. The metabolic requirements for maintaining and detoxifying N in the tissues (NH_4^+ is toxic in high concentrations) may be expensive, and may lead to a shortened lifespan. Assimilation of NH_4^+ requires that carbon be shunted away from sugar synthesis into amino acid synthesis. This would leave a plant deficient in carbohydrates and consequently energy

[6] Unpublished data on file, Department of Botany and Plant Sciences, University of California, Riverside.

to carry out other biochemical functions. We plan to pursue research on this question, and currently are testing other species of CSS shrubs to determine if this direct effect is generalizable.

Atmospheric N not only causes increased soil N, it results in exposure of vegetation to elevated levels of gaseous and particulate forms of N that may be deposited on leaf surfaces and interact directly with them (Krywult and others 1996). Fumigation experiments have not yet been carried out on CSS species, but they have been done on ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) and black oak (*Quercus kelloggii* Newb.). Fumigation with ambient peak summer levels of nitric acid (50 $\mu g/m^3$) caused cuticular lesions and stomatal collapse. These adverse responses by leaf surfaces could result in reduced stomatal control and increased exposure of the leaf to other stresses. In addition, nitric acid fumigation caused induction of nitrate reductase and increases in amino acid levels in leaf tissue (Padgett and others 1995). These negative effects can only be postulated for CSS species until similar research is done. Because up to 90 percent of the leaves of CSS shrubs senesce in the summer, there are fewer opportunities for foliar interaction with airborne N at peak summer pollution levels. The CSS plants would be subject to lower spring airborne nitric acid levels.

Conclusions

The decline of CSS vegetation is caused by a combination of complex factors, beginning with direct destruction of CSS for agriculture and urban construction, possibly including past grazing, and also including increased fire frequency in lands that are adjacent to urban areas. But even when these effects are held constant by examining CSS natural reserves that have not been impacted, these shrublands are still declining. The lack of response of the vegetation at Box Springs Mountain to additional N fertilization suggests that it is already N saturated, one response that is cited by Aber and others (1989) as being an indicator of N saturation. However, two forms of air pollution, ozone and N compounds, are likely both central players in the Los Angeles Air Basin and the Perris Plain. We have not discussed ozone in this paper, although it may also play a role in CSS decline. Ozone is known to increase the mortality of conifers in the local mountain ranges (Bytnerowicz and Grulke 1992, Miller and others 1963). Westman (1990) hypothesized that ozone may also be a cause of CSS decline because it reduced the growth of well-watered seedlings in greenhouse experiments at simulated ambient summer concentrations (Stolte 1982). However, during peak summer air pollution conditions stomates of CSS shrubs are closed and many leaves have senesced. Spring ozone concentrations are relatively low, so ozone effects in the field are likely smaller than greenhouse experiments would suggest. The effects of ozone may not be as serious as previously thought in CSS vegetation. However, until our work began, all of the air pollution effects were attributed to ozone and none to N deposition on this vegetation type.

Our studies showed that the annual grasses that replace CSS vegetation are not more nitrophilous than the shrubs, as they have equal plasticity in response to N. However, one species we have tested to date, California sagebrush, suffers a 100 percent mortality rate in the greenhouse in fertilized soils with available N at levels that are no higher than field soils in polluted sites. We are currently conducting experiments with two additional shrub species, brittle-bush and California buckwheat. We have not seen similar results from studies on ozone that would show such a rapid mortality rate. Most studies on ozone report only decreased growth rates at ambient concentrations, not mortality. In addition, the study on toxic effects of nitric acid vapor on pine and oak leaves suggests that more studies of deposited N on CSS shrubs are

needed. Increased mortality of CSS shrubs in N-polluted areas may be a cause of CSS decline in the Los Angeles Air Basin. CSS may be the first vegetation type in the western United States that exhibits stage 3 symptoms of N saturation, which is defined as toxic effects on the vegetation (Aber and others 1989).

Acknowledgments

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SESSION III

Air Pollution/Climate Change Effects on Forests: Results of Gradient and Case Studies

Reginald Noble and Alicja Breymeyer, Chairs

Transect Studies on Pine Forests Along Parallel 52° North, 12-32° East and Along a Pollution Gradient in Poland: General Assumptions[1]

Alicja Breymeyer[2]

Abstract

The responses of pine forest to changing climate and environmental chemistry were studied along two transects following the pollution and continentality gradients in Poland. One axis begins on the western border of Poland, crosses the country along the 52nd parallel, and ends on the eastern border of Poland in the area of Bialowieza National Park, Biosphere Reserve. The second axis begins in industrial Upper Silesia in southwestern Poland and ends at the same point in the Bialowieza area on the eastern border of Poland. The west-east latitudinal transect follows the gradient of cooling and continentality, whereas the Silesian transect follows the pollution gradient. The extension of the continental transect to the west (Germany) and the east (Belarus) was possible after 2 years of the program. This "large transect" covers 20° of latitude, with a difference in mean annual temperature of 3.5 °C. The general assumptions of this program, e.g., philosophy of gradient/transect studies, selected global change scenarios, current climatic conditions, and air pollution exposure in central Europe, are reviewed. The International Geosphere Biosphere Program on Global Change predicted temperature increases of 1.5-2 °C in our climatic zone, which are close to the difference in thermoclimate registered along the transects in this study. In the description of transects, Polish systems of forest monitoring and procedures used for stand selection are introduced. The phytosociological characteristics of the studied stands and their position in the classification of vegetation has been determined, as well as some of the present changes in pollutant distribution.

Introduction

Because pollution in Silesia is clearly elevated but has steadily decreased in recent years, it is important to understand changes in local sources of contamination and their impact on Polish forests. Our transect studies are an attempt to relate the field test of pine forest responses to global warming. Because central European pine forests are seriously affected by air contamination, we try to include in the same program the influence of pollutants on forest behavior. Sites distributed along two-axis transects were studied (*fig. 1*). One axis begins on the western border of Poland, crosses the country along the 52nd parallel, and ends on the eastern border of Poland in the area of Bialowieza National Parks, Biosphere Reserve. The second axis begins in industrial Upper Silesia in southwestern Poland and ends at the same point in the Bialowieza area on the eastern border of Poland. The west-east latitudinal transect follows the gradient of cooling and continentality, whereas the Silesian transect follows the pollution gradient. The extension of the continental transect to the west (Germany) and the east (Belarus) was possible after 2 years of the program. This "large transect" covers 20° of latitude, with a difference in mean annual temperature of 3.5 °C.

This paper studies the phytosociological characteristics of pine stands, their position in the classification of vegetation, and some of the present changes in pollutant distribution in order to analyze the process of pollution transect selection and assumptions about its use as a research tool.

General Assumptions

The change in global climate anticipated and most probably occurring now is expected to produce changes in the functioning of terrestrial ecosystems. The very rich ecological literature dealing with this subject has resulted in the development of a variety of proposed methods to evaluate changes in ecosystems. These include

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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laboratory treatment of individual organisms with differing doses of CO_2 and different thermic regimes; experiments more closely approximating natural conditions (especially through the widespread "open chambers" in which various simulated climates are applied to plants or habitat fragments enclosed in plastic tubes open at both ends); studies of the reactions of whole ecosystems to artificially manipulated changing climatic conditions (e.g., through heating of tundra soil); or studies on natural gradients of rising mean monthly air temperatures. Studies on comparative analysis of the same type of ecosystem functioning over suitably long periods of time in different climates are probably the most accurate. Certainly the use of such methods avoids the problems linked with extrapolating results from laboratory or semi-laboratory conditions to the field, and reduces the risk that the ecosystem response to sharp experimental intervention may be unpredictable or chaotic.

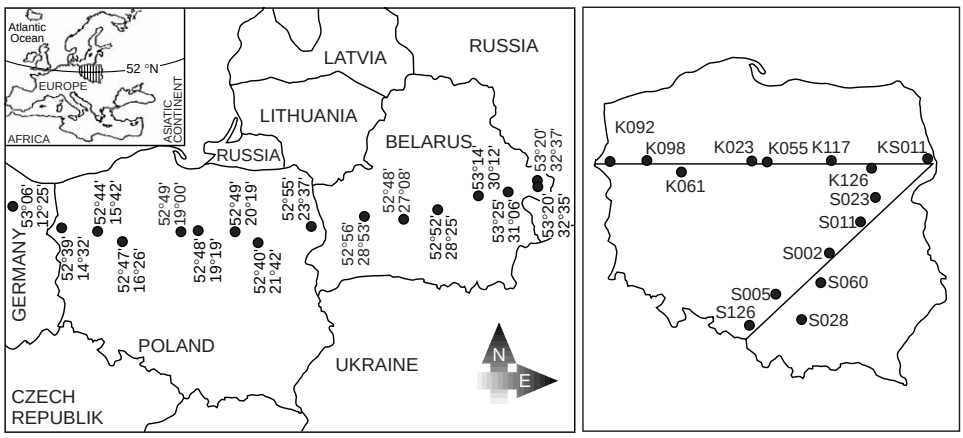

Figure 1 — (A) - "Large" climate transect along the parallel 52° N. Position of 16 stands measured by Geographic Positioning System (GPS) Geo Explorer, Trimble Navigation, Model 17 319. (B) - Two axes of climatic and pollution transects on the schematic map of Poland. Site names numbered according to Institute of Forest Research monitoring system; "K" = climatic transect, "S" = Silesian transect. Site "KS011" is common to both transects; it is the coolest site in the climatic transect and the "cleanest" site in the Silesian transect.

In gradient-related ecosystem research, stability is assumed in all environmental factors except the one studied, which varies gradually but sufficiently to modify the functioning of the ecosystem. The correlation between the changes in the factor being studied and observed departures in the functioning of the ecosystem are considered to express cause-effect relationships. Ideally, ecosystems that are selected carefully to be similar may also be treated like the same ecosystem subjected to gradually-changing intensities of a given factor. Gradients in space are thus considered, within limits, as substitutes of gradients in time, which would affect the ecosystem if it experienced the increasing action of the factor under consideration.

Published scenarios and considerations on global warming (*table 1*) have been quite consistent in their prognoses concerning "the geography of climate change." The most clear impact of warming is expected in the Northern Hemisphere in the zones of temperate and cool climate. Changes in atmospheric precipitation in these zones are difficult to predict, but the zones in question should experience a clear shift in the present ranges of the extensive deciduous and coniferous forest biomes.

Table 1 *— Predictions of temperature response to change in $C0_2$ concentration (Bolin and others 1986).*

Model	Globally averaged surface temperature response to CO_2 doubling
WPC (1981)	1.5 - 3.5 °C
CDAC (1983)	1.5 - 4.5 °C
EPA (1983)	1.5 - 4.5 °C
Clark and others (1982)	2 - 3 °C
Julich (1983)	1 - 3 °C
Present assessment	1.5 - 5.5 °C

Attempts to predict global changes over time have been made by Boryczka and Stopa-Boryczka (1992), who expect annual air temperatures to decrease between 1990 and 2050, and then to rise between the years 2050 and 2100. The overall rise in temperature between 1990 and 2100 is an anticipated 0.5 °C, a figure significantly lower than that suggested in the various computer simulations and experiments linked with the International Geosphere Biosphere Program and usually involving a doubling of carbon dioxide concentrations by 2030 and a consequent warming of 2-3 °C (*table 1*).

Climatic conditions in central Europe are determined by Arctic influences along the north-south axis (cooling in the direction of the Pole), as well as by oceanic influences of the Atlantic along the west-east axis (greater extremes of temperature and cooling towards the interior of Asia) (*fig. 2*). Smialkowski (1995) described the climatic characteristics from meteorological stations near research sites along the transects (*fig. 3*). The mean annual air temperature declined from 8.3 °C in the west to 6.7 °C in the east, while the total annual precipitation amounted to just over 500 mm throughout. The Upper Silesian transect was characterized by a fall in temperature from 8.1 (Silesia) to 6.7 °C at the northeastern end. The mean annual amplitude of air temperature (based on monthly average temperatures), a measure of the continentality of climate, increased from 18.8 °C in western Poland to 22.3 °C near the border with Belarus.

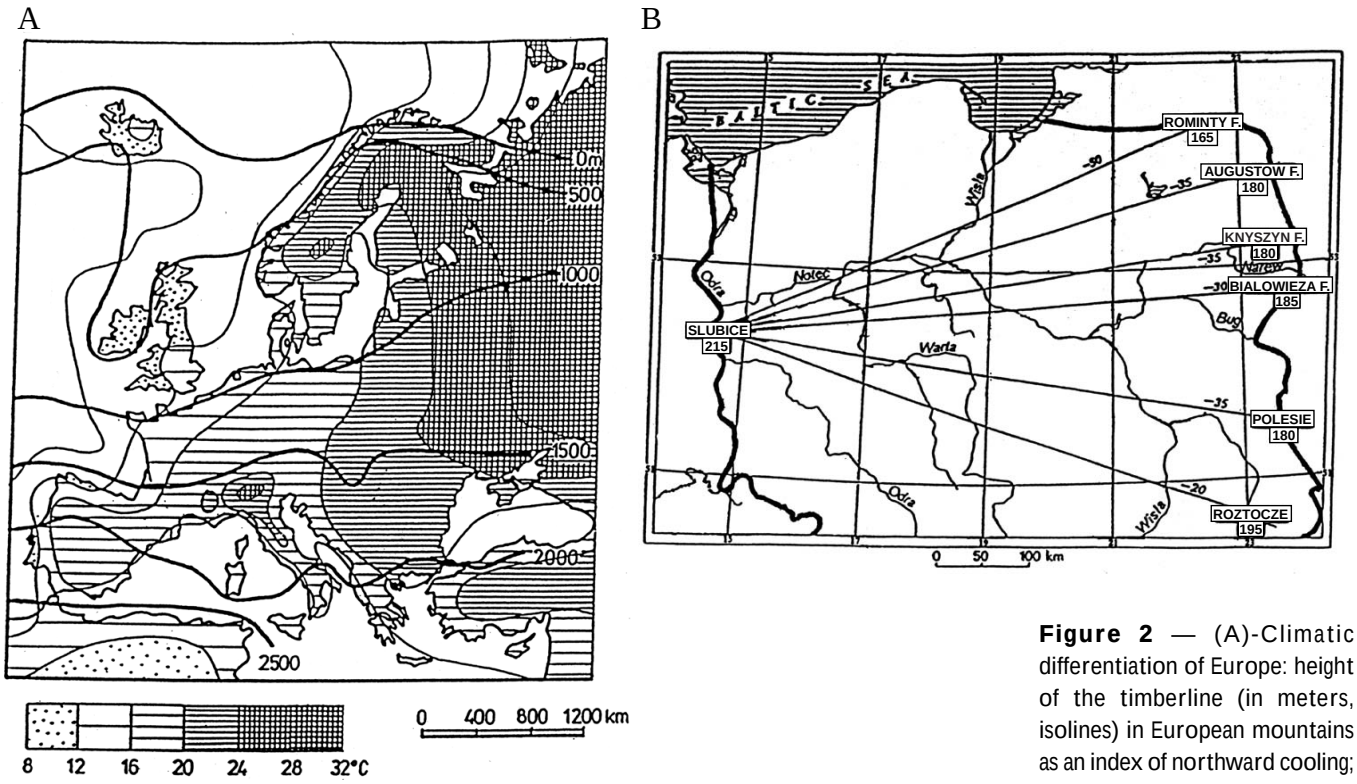

Figure 2 — (A)-Climatic differentiation of Europe: height of the timberline (in meters, isolines) in European mountains as an index of northward cooling; eastwards continentality expressed as increasing difference between temperatures of coldest and warmest month (in °C, shadowed surfaces) (Mayer 1984). (B)-Duration of the vegetation growing period (days) in various forests. Numbers placed on the lines show the difference between the warmest stand "Slubice" and individual stands (Falinski 1986).

Description of the Transects

Scots pine (*Pinus sylvestris* L.) dominates 70-80 percent of Poland's forests (in area covered or tree biomass) (Trampler and others 1987). The forests are very sensitive to pollution, and as a zonal community, must react in a distinct way to warming of the climate. It was accepted initially (Breymeyer 1994) that two complementary transects should be designated within Poland: a climatic (west-east) transect along parallel 52° N and a southwest-northeast transect running from polluted Upper Silesia to Bialowieza (*fig. 1*). Information on the state of Poland's forest across this large area has been collected by the forestry services. Monitoring of the state of Poland's forests is performed by various institutions. We used the network directed

by the Forest Research Institute. Basic data are collected on permanent research plots (SPOs) covering the entire country and established in 1989 in stands of Scots pine, Norway spruce, fir, oak, beech and birch greater than 40 years old. The selection and siting of SPOs reflects the cover, age and species structure, as well as the condition of Poland's forests. Over 1,000 SPOs were established, each with 20 marked trees of the dominant species observed annually for the following characteristics:

- Morphological characteristics of the crown;
- Defoliation and the coloration/discoloration of foliage and assignment of trees to the appropriate damage classes;
- A phytopathological evaluation and an assessment relating to insect pests;
- Tree diameter measurements every 5 years.

The results of all measurements are recorded in a uniform manner in a databank.

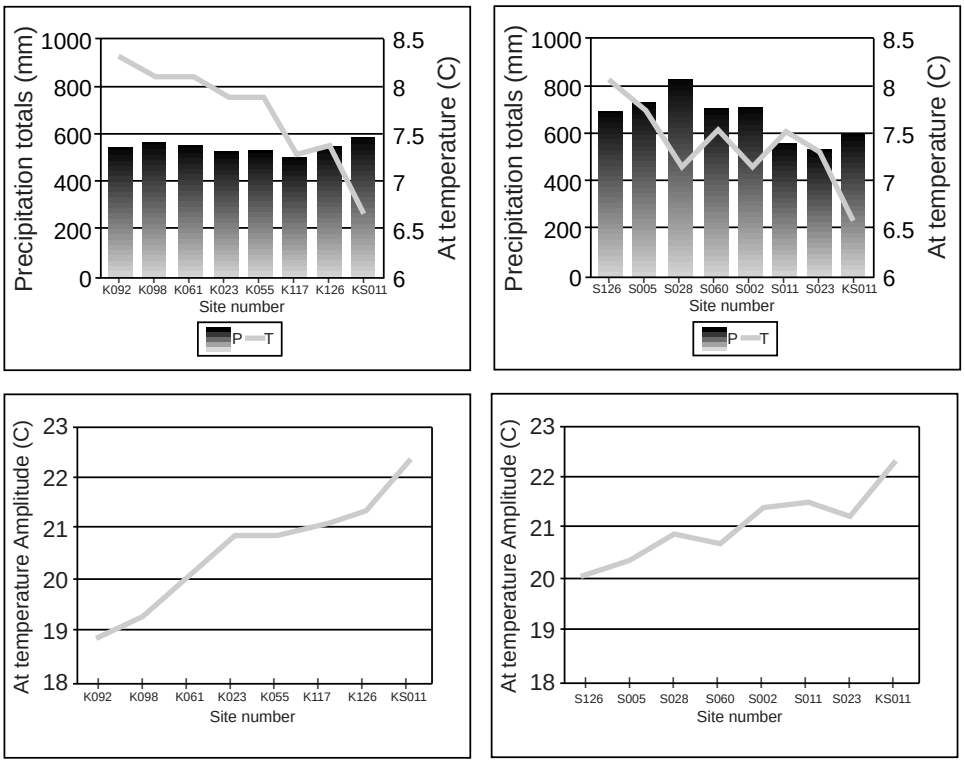

Figure 3 — Climatic characteristics of transects: mean annual air temperatures (lines), precipitation totals (bars) and annual amplitudes of temper-atures. Long-term data collected from meteorological stations in the vicinity of 15 studied sites (Smialkowski 1994).

Selection and Characterization of Sites

This research was based on a two-axis transect reflecting to some extent the influences of climate and pollution on the forests studied. The "Silesian" axis traced the gradient of air pollution from Silesia to the Bialowieza Forest, while the climatic axis traced the gradient of continentality along parallel 52° N from Poland's western to eastern borders, again terminating near Bialowieza on the same KS011 stand (*fig. 1*). Research sites were selected from a total of approximately 1100 SPOs based on the criteria of identical type of forest habitat, soil type, age of pine, indices of forest cover and stand quality, and similar aspect, slope, and elevations. The last stage of the selection process was implemented by site visits leading to the final choice of 15 permanent research plots.

Forests are complicated ecosystems; their functioning is conditioned by a long history (usually several generations of trees), by current and past management, and by a whole mosaic of habitat factors that vary considerably across Poland. Thus, the choice of sites along the transect was difficult, and despite considerable care and effort, we could not completely avoid the influence of confounding factors.

Ecosystems were mainly treated as functional units. According to Breymeyer (1981, 1984) and Breymeyer and Uba (1987), we assumed that the responses of ecosystems to environmental stresses can be measured via responses in the main processes occurring within them, including the cycling of matter (i.e., its production and decomposition). Responses to change by these processes may be intensified or reduced in strength under different conditions, and their proportions may change. However, the processes must continue and maintain the cycling of matter in the ecosystem. If these processes were to discontinue, the ecosystem would similarly cease to exist. Because changes in the intensity of ecosystem processes are measurable, the responses of ecosystems to stress are also measurable. Along the transects studied, the functioning of ecosystems was evaluated by measuring litter fall and the rate of decomposition of organic matter in the litter layer and in underlying soil layers.

The structural features of forest ecosystems are composed of soil, vegetation and different groups of organisms. The composition and structure of vegetation were used to group plant communities according to phytosociological classification. The forests of Poland and central Europe have been described and classified by using the Braun-Blanquet system, and maps of the potential vegetation of Poland and Europe have been established. The forests studied in our program have been described in accordance with this classification. All of the sites studied belong the *Dicrano-Pinion* alliance and include two associations of pine forest: *Peucedano-Pinetum* (sub-continental pine forest) and *Leucobryo-Pinetum* (sub-oceanic pine forest), as well as mixed forest of *Querco-Pinetum* type. Pine forests occupy the same place along the scale of humidity, but differ in the degree of continentality (Matuszkiewicz and Matuszkiewicz 1973) (*fig. 4*). The communities of the *Leucobryo-Pinetum* association cover a large area and are the most common types of forest in Poland, showing little regional differentiation. Further east, the sub-oceanic type is replaced by the sub-continental *Peucedano-Pinetum*, which occurs to the east of the line formed by the Bug and Lower Vistula Rivers. Sub-continental forest is much richer floristically and characterized by the presence of continental species, mainly perennials of a somewhat xerothermic nature, as well as by limited regional variation. Some of the sites had a deciduous component, so they were classified ultimately as mixed pine forest of the *Querco- Pinetum* type (*fig. 4*).

In the series of stands described above, the changes in behavior of pine ecosystems were measured by groups of specialists. The following indexes of ecosystem

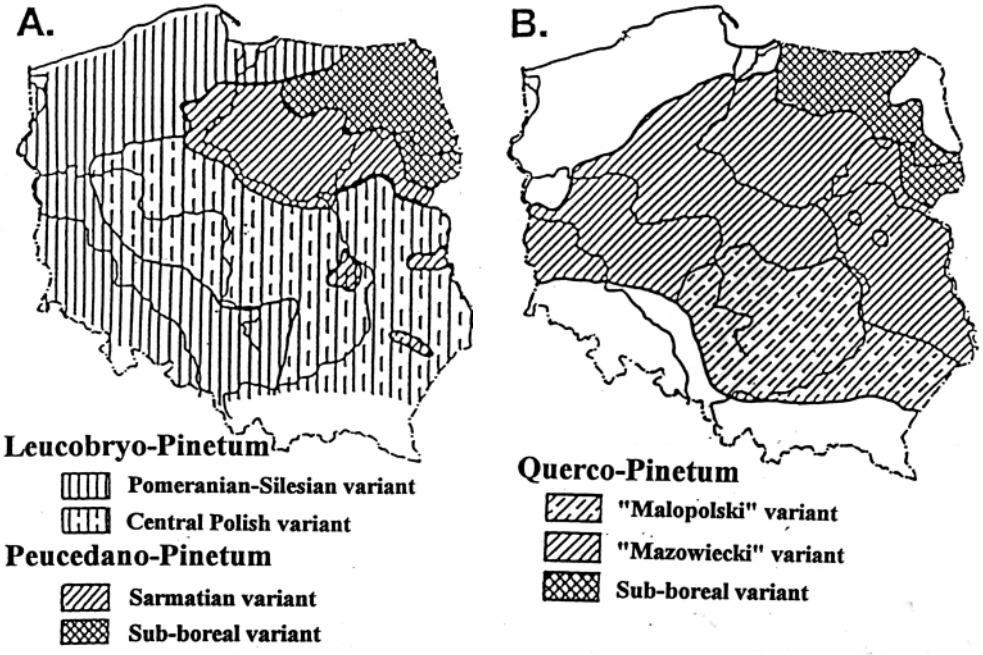

Figure 4 — Regional diversity of pure coniferous (A) and mixed (B) pine forests in Poland (Matuszkiewicz 1987).

function and structure were selected for study:

- Soil characteristics such as vertical structure and physio-chemical properties;
- Cotton decay in various soil layers as an index of the stratification of biological activity;
- Litter fall and decomposition as an index of cycling rate of organic matter (OM budget);
- Tree rings increment as an index of stand history influencing its present condition;
- Potential vegetation (sensu Tuxen 1956) of the stand as an index of stands similarity and position in the vegetation classification system;
- Selected characteristics of present plant cover: diversity of species and forms, biomass, and chemistry (nutrients and pollution) of dominant species of herbs and shrubs.

All these indexes are related to simultaneously evaluated climatic factors and inputs of pollutants.

The fundamental idea of the double transect, with gradients focused mainly on two groups of factors (climate, pollution), has since been affected by certain changes. At the time the research began it was obvious that the pollution transect must run from the south-west (Upper Silesia region) to the north-east (Bialowieza forests). In the literature from the previous 20 years, very high levels of pollution were reported for the industrial Silesia region. Furthermore, mathematical modeling of the origin and distribution of pollution throughout the country demonstrated that Silesia was the dominant pollution source region (Juda-Rezler and Abert 1994; *fig. 5*). However, recent years show a clear improvement in air quality, with a progressive decline in air pollution in the province of Katowice, Silesia's largest city (*table 2*). Recent years have also seen a distinct decrease in air pollution at "background" monitoring sites in regions far from emission sources (Przbylska 1995; *fig. 6*). Some time will doubtless pass before the first signs of improvement in the condition of forests are noted, but such changes will certainly occur. Thus, the research reported here is expected to document changes in forest condition which will certainly occur; and this research will coincide with a certain blurring in the gradient of pollution gradient along the axis between Silesia and northeastern Poland. In such a situation, local pollution is inevitably of greater significance than had been anticipated.

We decided to run the climatic gradient latitudinally along the 52nd parallel for several reasons. This avoided the inclusion of sites conditioned by higher altitude characteristics for the southern half of Poland. In addition, the selected transect is the longest one possible for the country's coniferous forests, and it allows for extension to the east or the west without changing the above conditions.

This extension was of value in the second year of the research, when a first expedition to Belarus led to the designation of six new sites between the Poland-Belarus and Belarus-Russia borders (*fig. 6*). The third year of the study, two more

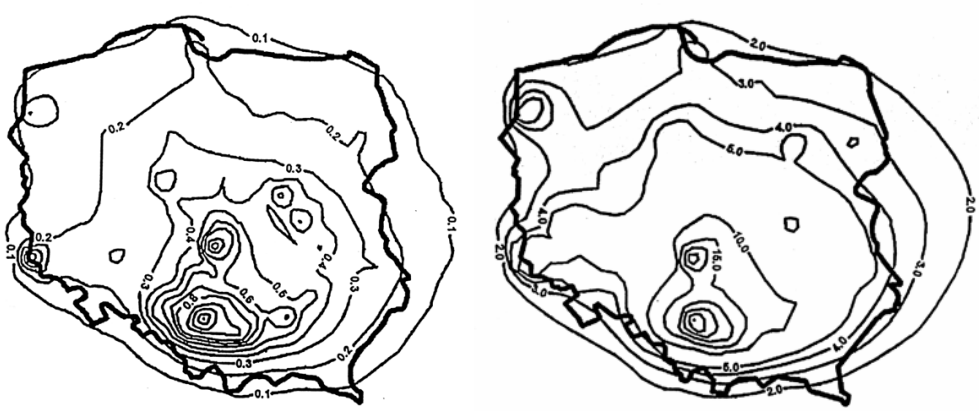

Figure 5 — Mean annual concentrations of S and N compounds over Poland in 1990 (Jura-Rezler and Abert 1994). (A)-Concentration of NO, isolines: 0.1-1.5 µg N per m^3. (B)-Concentration of SO_2 (µg S per m^3), isolines: 2, 3, 4, 5, 10, 15, 20, 25, 30, 35.

sites were added in Germany (at the western end of the transect). The extended transect currently runs across 20° of longitude (from 12°25' E to 32°60' E), has a range of annual temperatures of 3.5 °C (from 5 to 8.5°), and has annual amplitudes of temperature ranging from 18.2 (west) to 26 °C (east). However, this study has focused on results from the original Polish portion of the transect.

Table 2 *— Emission of air pollutants in the Katowice District (1985-1992) (Katowice Voivodship Statistical Manual 1993).*

	Emission of Pollutants (thousand tons / year)	
Year	**Dust**	**Gases**
1985	453.9	1540.9
1986	418.7	1509.9
1987	389.3	1505.2
1988	327.9	1306.3
1989	305.3	1309.2
1990	227.1	1003.3
1991	186.7	850.2
1992	127.1	738.1

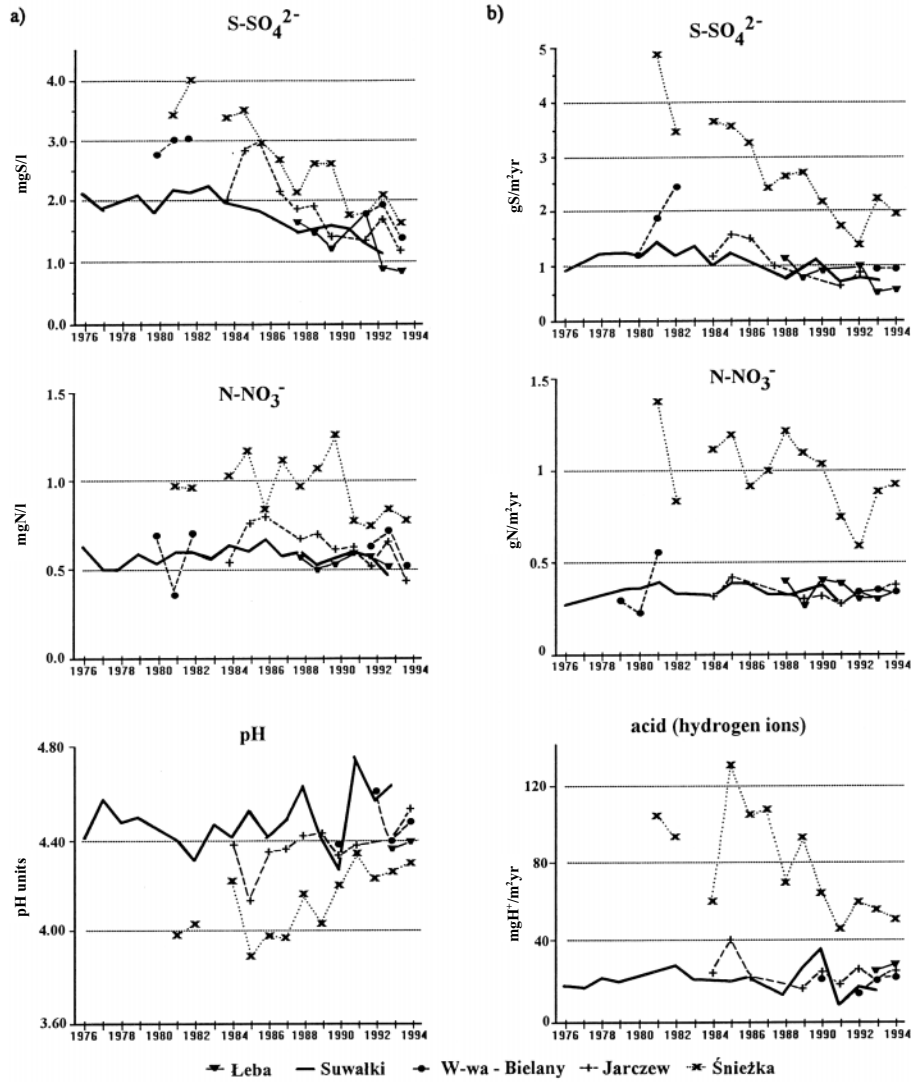

Figure 6 — Concentration in atmospheric precipitation (A) wet depositions (B) of some elements, hydrogen ions and pH as measured in five meteorological stations located across Poland. The program of observations is comparable to the international systems WMO, EMEP, BMP, as well as to the Polish national monitoring system (Przybylska 1995).

Acknowledgments

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Geographical Differentiation of the Floristic Composition and Structure of the Herb Layer of Forest Permanent Plots in East Germany, Poland, and Belarus[1]

Ewa Roo-Zielinska, Jerzy Solon[2]

Abstract

The influences of a geographical location on floristic composition, horizontal structure, and biomass of the herb layer in pine and mixed pine forest communities along climatic and pollution gradients in East Germany, Poland, and Belarus were determined. Phytosociological records were collected in permanent plots in May 1995. Each record covered an area of 400 m^2. The floristic composition of the study area provided the basis for the determination of the phytosociological differentiation of the plots along the transect. Similarity index between plots was calculated by using the Czekanowski index; a similarity dendrogram was constructed according to Ward's method. Biomass was determined in six 0.1 m^2 replications sampled in each permanent plot. All of the identified study areas were representative of the Dicrano-Pinion *alliance and included two communities of pine forest (*Peucedano-Pinetum *and* Leucobryo-Pinetum*) as well as the community of mixed pine forest (*Querco roboris-Pinetum*). On the basis of similarity indices, two geographical groups of plots were distinguished: suboceanic and subcontinental. Geographical differentiation of the stands is related to the presence of the groups of species of different geographical distribution type. For the western part of the transect, characteristic species are* Deschampsia flexuosa *and* Leucobryum glaucum. *Central and eastern parts are characterized by the presence of the following species:* Melampyrum pratense, Luzula pilosa, Veronica officinalis, Lycopodium clavatum, Peucedanum oreoselinum *and* Scorzonera humilis. *On the stands located on the easternmost part of the transect (Belarus) frequent species are:* Pirola chlorantha, Chimaphila umbellata, Koeleria pyramidata *and* Hypochoeris maculata. *The ground flora of the identified areas was not uniform; rather it was characterized by a mosaic of sparse populations of plants. In general, three types of these systems can be distinguished: one is dominated by a cover of mosses, the second is dominated by dwarfshrubs, and the third by species of grass. The ground cover biomass on the west-east transect ranged from 49 to 544 g/sq. m, but for pine forests the correlation between geographical longitude of the stand and biomass of mosses (corr. coeff. 0.701) and total biomass of the ground cover (corr. coeff. 0.779) was statistically significant.*

Introduction

This article presents a geobotanical analysis of the floristic composition of the herb layer and its structure and biomass in East Germany, Poland, and Belarus. It constitutes part of long-term, in-depth community research aimed at defining the influence of geographical location on pine forest ecosystems, in particular pine forests and mixed / pine forests (Breymeyer 1996, Degórski 1996, Smialkowski 1996).

The subjects of geobotanical analyses consisted of 17 permanent plots lying along a west-east transect following the gradient of continentality. These are mature or almost mature pine stands, basically single-species and single-aged (60-95 years). All the analyzed forest communities represent (in accordance with phytosociological typology) the *Dicrano-Pinion* alliance, and within it two communities of pine forest, *Peucedano-Pinetum* and *Leucobryo-Pinetum*, as well as the community of mixed pine forest, *Querco roboris-Pinetum*. In accordance with the principles of forest typology these are different forms of pine forest occurring on sandy substrates. Their economic use does not differ significantly.

The level of differentiation of the herb layer is one of the more important structural characteristics of forest phytocoenoses. On the one hand it influences the spatial differentiation for various processes like the production and decomposition of organic matter, and on the other it may be an indicator of habitat microdifferentiation (in particular topography, fertility, and humidity). A way of describing the level of diversity of the herb layer is through the identification,

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analysis, and typology of synusiae. Synusia is a grouping, within one layer of a community, of species characterized by similar life forms and similar ecological requirements. The synusia is sometimes used as a basic spatial unit, as well as a functional-temporal unit within a phytocoenosis (Gillet 1986, Julve and Gillet 1994, Mavriscev 1980).

Because the herb layer reacts most actively to all environmental variables, we attempted to define the influence exerted by climatic differentiation and habitat (also microhabitat) conditions on the species composition, structure, and biomass of the herb layer. The floristic composition and structural characteristics of the studied communities depend on many factors, such as:

- Geographical location, which determines both the richness of the local flora (influencing the occurrence of species of a defined type of range), and is an indicator of macroclimatic differentiation. Climatic characteristics (in particular mean annual temperature, the temperature of the warm and cold periods of the year and total precipitation) have a considerable influence on the phenology and competitive abilities of different populations;
- The type and properties of the habitat, mainly its fertility and humidity, which influence the presence or absence of the species building up a given phytocoenosis;
- Anthropogenic influences linked with the pollution of the environment, the history of its use, and the maturity of phytocoenoses.

This paper describes a study to define the mutual similarities and homogeneity of the study sites on the basis of the floristic composition of the herb layer in East Germany, Poland and Belarus; presents the synusial differentiation of the herb layer; draws up a typology of the pre-analyzed synusias, and defines the mean biomass within the different types of synusia and within the study areas.

Methods

Floristic Composition

The basis for the description of the 17 study sites used in this study was a floristic characterization of the patch of vegetation representing a given type of community and described in the form of a phytosociological record. This contained the following basic information on a given patch of vegetation: the species composition; cover (i.e., the proportion of the area occupied by each species); and layer structure (vertical structure of the community), identified most often as A–trees, B–shrubs, C–herbs, D–ground cover of mosses and lichens. The majority of phytosociological records were taken in May 1995. Each record covered about 400 m^2. The quantitative representation (cover) of species was estimated by using a modified 12-point scale on which r, +, and 1, refer to 0.1 percent, 0.5 percent, and as high as 10 percent, respectively, while the remaining values from 2-10 relate to the successive 10 percent divisions. Vascular plant species were defined by using the key of Szafer and others (1969) and the nomenclature of Rothmaler (1976), while nomenclature of mosses followed Szafran (1957, 1961), liverworts followed Rejment-Grochowska (1950) and lichens followed Faltynowicz (1993).

The phytosociological records obtained were presented in an unordered table, which was interpreted and described in accordance with the system of classification and list of diagnostic species of Matuszkiewicz (1981).

By using Czekanowski's measure of distance, we calculated the mutual similarity of the identified forest communities on the basis of the floristic composition of the herb layer. This was presented with the aid of a dendrogram produced by using Ward's method (Solon 1994).

Synusial Structure

The basis for the description of the synusial structure of an area's ground cover was field mapping of the most typical 5 by 20 m rectangular areas in 9 out of 17 study sites (N1, K092, K098, K061, K023, K055, K117, K126 and K011) lying on the west-east transect in Germany and Poland. In addition, floristic inventories were carried out within the different synusiae. These were the bases for the identification of types of synusia. Inclusions in the typological table were confined to those species whose cover was greater than 10 percent. The mapping and inventory of plant species was done in May 1995.

The similarity of the floristic compositions of different synusiae was calculated by using Czekanowski's measure of distance and presented with the aid of a dendrogram produced by using Ward's method (Solon 1994).

Biomass

Biomass was determined once during September and October 1995 at all 17 sites. Six circular biomass samples were taken from a 0.1 m^2 area of each permanent plot. The choice of the area for replicates was not random; rather, we attempted to take replicates of the dominant types of synusiae at a given study site. The entire biomass was collected in the field and then sorted into living moss biomass and higher plants. The dead parts of plants were removed. The samples were dried to constant weight at a temperature of 90 °C and weighed to an accuracy of 0.01g. The results obtained were averaged for the type of synusia and for the study area and recalculated per m^2.

Results

Floristic Composition

Areas K092 and K023 had *Leucobryo-Pinetum* pine forest in the typical form, while area K061 had a more humid variant with *Molinia caerulea*. In contrast, the two areas N1 and K098 were occupied by communities occurring in an acid beech forest habitat (*Luzulo-Fagion* alliance) or by mixed-pine forest, although they are almost identical to the communities representing *Leucobryo-Pinetum*. The *Peucedano-Pinetum* community is represented by areas K011, B2, B3 and B5 (type or sub-boreal forms), as well as by a more humid variant with *Molinia caerulea* on areas S023 and B7. The remaining areas had mixed pine forest (*Querco roboris-Pinetum*), with the typical form on areas K055, K126, B1, B4 and B6, and area K117 consisted of a degraded habitat that was a poor form of oak-linden-hornbeam forest *Tilio-Carpinetum*.

Analysis of the similarity of the species composition of vascular plants between the different sites clearly points to the occurrence of two large groups of communities: the subcontinental and the suboceanic. But the compositions of these groups differ, depending on the manner in which similarity is defined, e.g., on the basis of cover (quantitative representation) of species or on the basis of the presence or absence of species. The core of the suboceanic group is comprised of areas N1, K092, K098, K061 and S023. As cover (quantitative representation) of species, areas K117 and K126 also belong to this group, but if defined as the similarity of the species composition, these are part of the subcontinental group.

The division into groups of lower rank is also first and foremost geographical in nature. Distinguished within the sub-Atlantic group is a western subgroup, although on the basis of cover the habitat division is more important because it clearly separates out the subgroup of sites of greater humidity (K061, S023) characterized by a greater share of purple moor grass (*Molinia caerulea*). Two geographical subgroups may also be clearly distinguished within the subcontinental group. On the basis of the presence or absence of species, all the Belarus sites (B1-B7) create one subgroup, while on the basis of quantitative representation sites B1 and B3 are more similar to the Polish sites.

One of the factors in the spatial variation of the floristic composition of pine forests dependent on geographical location is the differentiation in the ranges of particular species of plant, which, at least theoretically, consist of vegetation conditions that are appropriate for their existence. A clear suboceanic character is, for example, displayed by *Deschampsia flexuos* and *Leucobryum glaucum*, while the areas of a subcontinental character are most strongly associated with such species as *Chimaphila umbellata, Hypochaeris maculata* and *Koeleria pyramidata* (*fig. 1*).

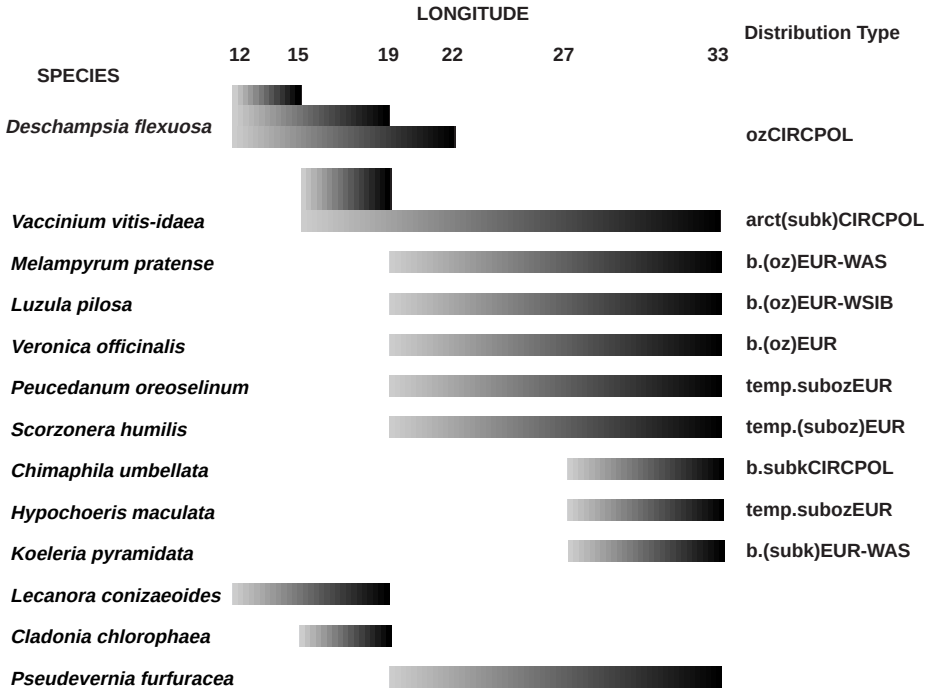


Figure 1 — Presence of chosen species along the the west-east transect (Rothmaler 1976).

Synusial Structure

On the basis of dominant species, life forms, and the degree of cover of the area with vegetation, it is possible to distinguish six main types of synusiae (*table 1*). The first type includes synusiae with an absolute dominance of *Vaccinium myrtillus*, a limited role of mosses, and the sporadic occurrence of various grassy species. The second type is characterized by the dominance of *Vaccinium vitis-idaea* and compared to the first type, it occurs in places that are drier locally. The third type includes mossy synusiae with a predominance of *Pleurozium schreberi*. This is the type occurring most frequently and at the same time most differentiated into subtypes in relation to the presence and character of codominants and subdominants. In particular, it is possible to distinguish pure mossy, mossy-cowberry, mossy-bilberry, mossy-sedge, mossy-fescue, mossy-hairgrass, and mossy-hairgrass-fescue subtypes. The fourth includes synusiae characterized by the codominance of three or four species and the lack of a clear dominant. Within this type, the first subtype includes a mossy-grassy synusia and the second a mossy-heather one. A fifth type includes a synusia with sheep's fescue (*Festuca ovina*) as dominant, while a sixth type is characterized by a very thin vegetation cover not exceeding 20 percent.

Different study areas differ clearly on the basis of the set of synusial types as well as in their spatial distribution (*fig. 2*). Mapped fragments of the areas have 2 to 4 types of synusia creating between 3 and 32 individual patches. However, mixed pine forests have a much more mosaic-like herb layer because these sites are observed to have greater numbers of patches of synusiae of different types (*fig. 2*; for example, compare sites K092 and K126).

The geographical differentiation of synusiae is poorly marked and results from differences in the ranges of the dominant species (Roo-Zielinska and Solon 1996). Western areas are associated with mossy-hairgrass, mossy-hairgrass-fescue, and mixed mossy-grassy synusiae. The majority of types of synusiae occur in both fresh pine forests and mixed/pine forests. The only synusiae to be closely associated with fresh pine forests are mixed mossy-grassy and mossy-heather, as well as mossy-bilberry and pure mossy synusiae. In contrast, mixed/pine forests more often have mossy-fescue, mossy-sedge, or fescue synusiae.

Table 1 — *Floristic composition of synusiae on the west-east transect.*

Type	I.1b	I.1a	II.1	III.2		III.1d			III.1a			III.1c	III.3		III.4	III.5		III.1e	III.6	III.7a	III.7.b	IV.1		V.1b	V.1a		III.1b		IV.2b	IV.2a	VI		
Description number	055 b	N1 a	098 c	061 a	011 d	011 b	023 b	061 d	055 c	023 a	126 a	011 c	126 d	011 a	055 a	055 d	126 b	117 d	092 b	092 a	098 a	098 b	N1 b	023 c	117 e	117 g	117 c	117 f	117 a	061 c	061 b	126 c	117 c	061 e
Vaccinium myrtillus	2	5									1	1	1	4	3	3						2	1											
Vaccinium vistis-idaea			3	4	4	2	3	2		1	1	1	1								1	1								2			1
Entodon schreberi	1	2	1	3	4	4	5	5	5	4	5	5	5	5	4	5	5	5	5	5	5	5	3	2	2	1	1	3	3	2	2	1	1	1
Deschampsia flexuosa		1																	1	3	3	3	3	3										
Festuca ovina		1								1	1	1	1				3	4	2	1	3	3	3	2	3	4	4	1	1			1	1	
Calluna vulgaris					1							1	2													1		1		2	3			
Molinia caerulea																														2	1			
Carex ericetorum																									2									
Calamagrostis arundinacea									1						1	4																		
Lichenes			1					1																									1	
Bare ground	4	1	3	1	1																				1		1	2	2	1	1	5	4	4

I. *Vaccinium myrtillus* **type**
I.1. - (a - typical; b - lose)
II. *Vaccinium vitis* **- idaea type**
II.1 *Vaccinium vitis* **- idaea**
III. *Pleurozium schreberi* **type**
III.1. (a - typical; b - lose; c - with *Calluna vulgaris;*
d- with *Vaccinium vitis - idaea;*
e - with *Festuca ovina*)
III.2. *Pleurozium schreberi - Vaccinium vitis - idaea* **subtype**
III.3. *Pleurozium schreberi - Vaccinium myrtillus* **subtype**
III.4. *Pleurozium schreberi - Calsmagrostis arundinacea* **subtype**
III.5. *Pleurozium schreberi - Festuca ovina* **subtype**
III.6. *Pleurozium schreberi - Deschampsia flexuosa* **subtype**
III.7. *Pleurozium schreberi - Festuca ovina - Deschampais flexuosa* **subtype**

IV. mixed type
IV.1. mixed moss - grass subtype
IV.2. *Pleurozium schreberi - Celluna vulgaris* **subtype**
(a - typical; b - with *Molinia caerulea*
V. *Festuca ovina* **type**
VI. degraded
VI.1. strongly degraded

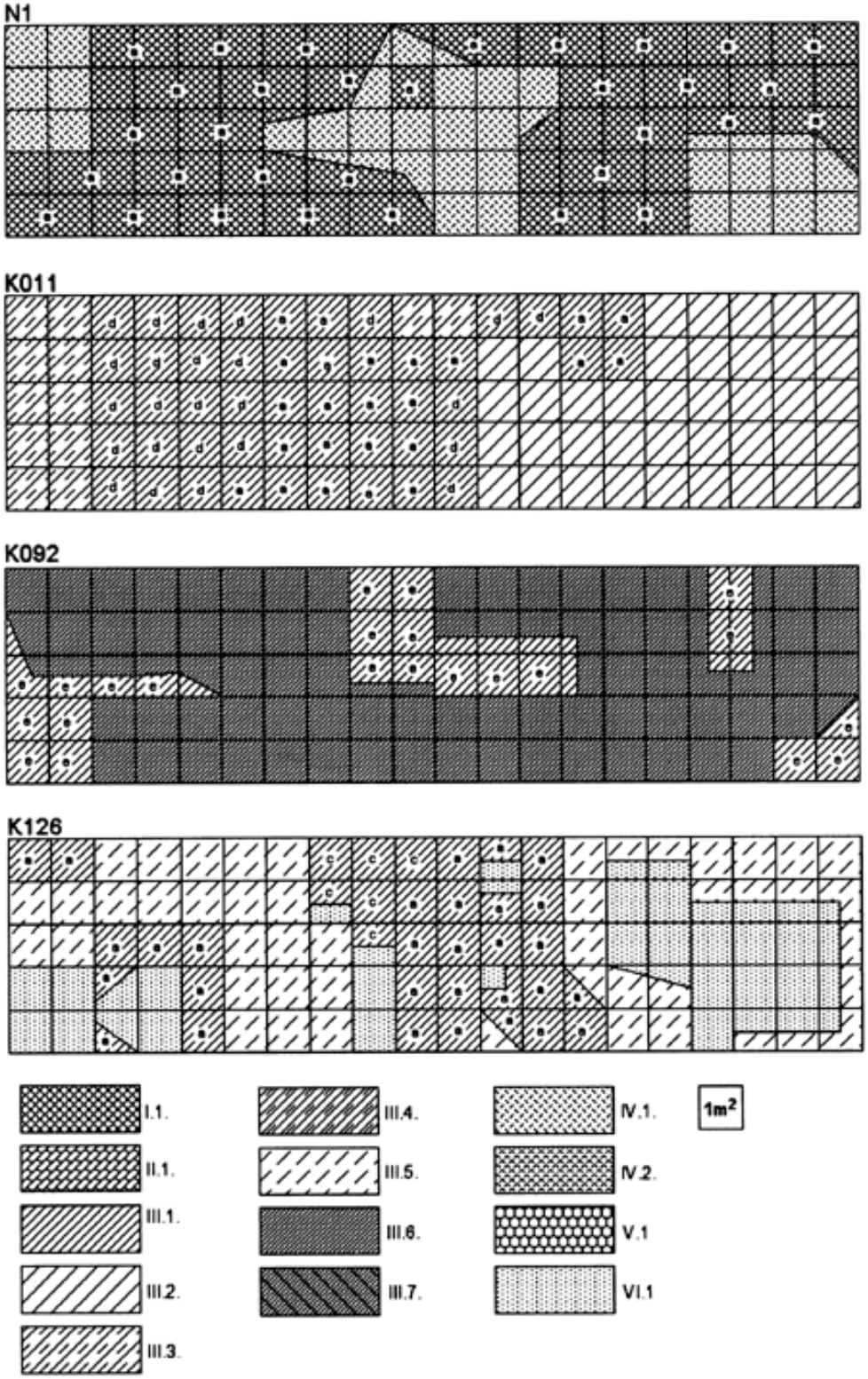
Biomass Differentiation

Biomass in Synusiae

The different synusiae and their types are highly diversified in terms of biomass of mosses and vascular plants. Degraded synusiae (type VI) have total biomass of about 15 g/m^2, while the mossy-cowberry subtype attains more than 600 g/m^2. The variability in the biomass of the different types of synusiae is also relatively great, and the differences between the most luxuriant and the poorest may be as large as 300 percent, as is the case for the mossy-bilberry subtype. This variability may only be partly explained by reference to geographical variability (*fig. 3*). The only statistically significant ($p < 0.05$) correlation ($r=0.67$) with longitude is noted for the biomass of mosses within the mossy-bilberry subtype of synusia. In the case of this subtype the correlations between longitude on the one hand, and the biomass of vascular plants and total biomass on the other are relatively great (about 0.5) but of very low significance. A similar situation is also noted in the case of pure mossy synusiae. The low level of significance of the correlation is linked with the limited abundance of the analyzed sample, and the results may attest to the limited variability of biomass in the different types of synusiae in the west-east direction.

Several types of synusia have ratios of vascular plant to moss biomass that are relatively constant and clearly different from those of other types. However, this is not a consistent ratio. In the case of the bilberry and cowberry synusiae (I and II), the ratio attains values between 4 and 5. For the mossy type (III), it is almost always below 1 (only in a few cases is it greater than 1, reaching about 3). For the mixed type (IV) it varies from about 1 to 3 and in the case of the fescue type (V) from about 3.5 to 5.5. The degraded type has values oscillating around 1, while the purple moor grass type (VII) always has a value more than 2 and sometimes much more than 10 (*table 2*).

Figure 2 — Synusial differentiation of study plots (see *table 2* for legend).



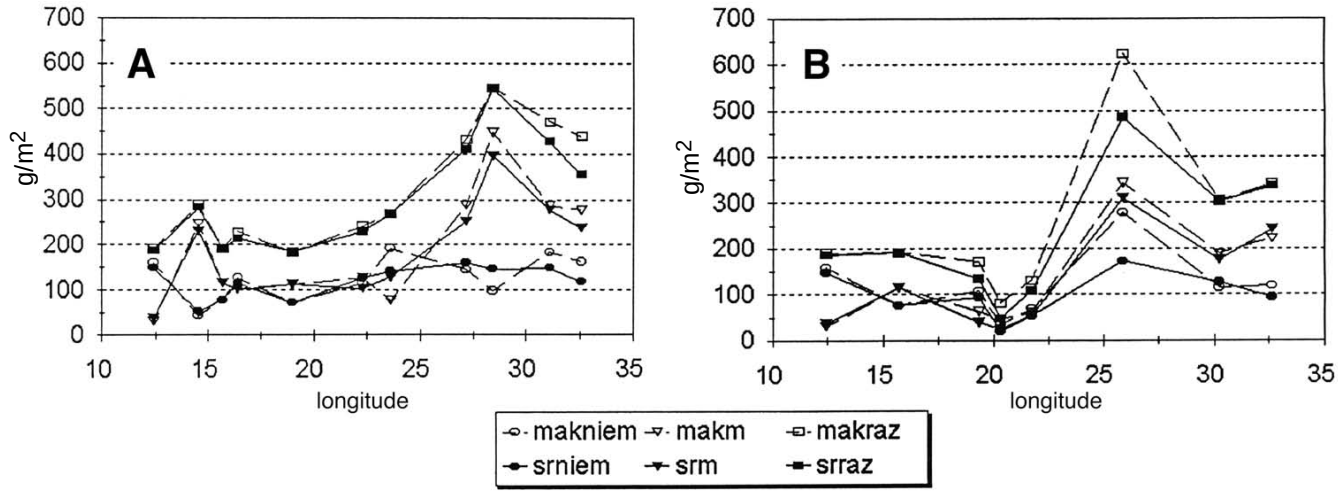


Figure 3 — Changes in biomass of study plots along the west-east transect: pine forests (A); mixed forests (B). Variables: makniem = mean biomass of vascular plants in the richest synusium of the plot; makm = mean biomass of mosses in the richest synusium of the plot; makraz = mean biomass of vascular plants and mosses in the richest synusium of the plot; srniem = mean biomass of vascular plants on the plot; srm = mean biomass of mosses on the plot; srraz = mean biomass of vascular plants and mosses on the plot.

Mean Biomass in the Community

The mean biomass of the herb layers and mossy layers at the study sites vary with geographical location (*fig. 3*). The correlation coefficient for moss biomass against longitude for all 17 sites on the west-east transect amounted to 0.701***, while the values for fresh pine and mixed/pine forests were 0.701* and 0.712*, respectively (in which ***, **, and * respectively indicate significance at the $p < 0.001$, 0.01, and 0.05 levels). Longitude is also correlated with the combined biomass of the herb layer, mosses in pine forests (0.779**), and in all sites (0.779***). If the mean biomass of the richest type of synusia is considered a site characteristic, then correlations have similar values: the correlation coefficient for moss biomass and longitude for all 17 sites on the west-east transect is 0.687***, while that for fresh pine forests is also 0.687*. Moreover, the coefficient defining the relationship between longitude and total biomass of mosses and vascular plants attains respective values of 0.847*** and 0.848***. Both mean biomass of the community and mean biomass of the richest synusia showed no statistically significant correlation between the biomass of vascular plants and longitude.

Discussion and Conclusions

Changes in floristic composition were detected along the west-east transect both within pine forests and mixed-pine forests. Differences in floristic composition correspond to differences in the habitat (connected with the division into pine forests and mixed-pine forests, and the occurrence of typical forms and wetter forms with *Molinia caerulea*) and also to the division into geographically varying communities within pine forests (the division into *Peucedano-Pinetum* and *Leucobryo-Pinetum*). The main spatial geographic factor in the variability of the floristic composition of pine forests is differentiation in the ranges of understory plant species, which find appropriate conditions for their existence in this type of vegetation. Sites in central and southern Poland may be intermediates between the well-defined western and eastern pine forests. The geographical and edaphic differentiation of the study areas is reflected most strongly in the herb layer, although it is not very visible among lichens. Differences in this group are influenced to a greater degree by local factors, including the land use history of the area, than by microhabitat factors.

Pine forests and mixed pine forests did not have homogeneous herb layers; they were differentiated into fragments creating a system of separate micro-areas varying in the dominance of particular species and life forms. A detailed analysis of the spatial variability of the biomass of the herb layer and of bryophytes was

Table 2 – Biomass differentiation (g/sq. m) of synusiae and permanent plots.

Site	Date 1995	Type	Number of replications	Herbs		Mosses		Herbs & Mosses	
				Mean	Standard deviation	Mean	Standard deviation	Mean	Standard deviation
N1	14.09.	IV.1	3	137.83	45.69	45.10	14.67	182.93	59.93
		I.1	3	159.77	24.07	31.57	33.20	191.33	40.53
		mean	6	148.80	38.13	38.33	26.54	187.13	51.33
K092	15.09.	III.1	3	42.70	5.42	247.17	19.19	289.87	13.85
		III.6	3	62.00	8.86	214.33	55.52	276.33	51.73
		mean	6	52.35	12.13	230.75	44.66	283.10	38.47
K098	13.09.	III.7	6	77.30	25.52	114.87	45.99	192.17	69.35
		mean	6	77.30	25.52	114.87	45.99	192.17	69.35
K061	13.09.	III.2	3	101.80	31.25	99.03	24.24	200.83	29.07
		IV.2	3	126.73	52.14	102.53	62.24	229.27	88.69
		mean	6	114.27	44.76	100.78	47.26	215.05	67.51
K023	12.09.	III.1	6	72.00	31.22	112.08	66.32	184.08	59.78
		mean	6	72.00	31.22	112.08	66.32	184.08	59.78
K055	12.09.	III.3	3	107.87	33.10	64.10	17.56	171.97	39.39
		I.1	3	79.97	19.38	16.33	11.80	96.30	26.10
		mean	6	93.92	30.50	40.22	28.18	134.13	50.48
K117	12.09.	III.5	3	34.93	18.66	46.63	3.23	81.57	18.51
		VI.1	3	8.60	4.52	7.03	4.45	15.63	8.20
		mean	6	21.77	18.91	26.83	20.18	48.60	35.94
K126	11.09.	III.3	3	69.13	26.76	61.93	11.77	131.07	17.68
		III.5	3	38.97	9.19	48.10	10.30	87.07	8.56
		mean	6	54.05	25.05	55.02	13.04	109.07	26.02
K011	11.09.	III.3	3	192.57	28.66	76.60	13.49	269.17	16.36
		III.2	3	90.13	5.44	178.73	36.85	268.87	31.47
		mean	6	141.35	55.22	127.67	58.12	269.02	25.08
S023	11.09.	VII.1	3	138.90	7.07	78.77	52.91	217.67	49.15
		III.3	3	114.77	34.71	127.20	38.66	241.97	71.85
		mean	6	126.83	27.80	102.98	52.28	229.82	62.75
B1	24.09.	III.3	3	278.83	39.85	344.33	86.88	623.17	49.99
		III.2	3	68.00	10.37	279.00	68.86	347.60	68.15
		mean	6	173.72	109.07	311.67	84.92	485.38	150.19
B2	23.09.	III.3	3	175.90	38.77	215.00	11.22	390.90	45.67
		IV.2	3	144.67	71.34	288.00	17.47	432.67	54.17
		mean	6	160.28	59.50	251.50	39.34	411.78	54.28
B3	22.09.	III.1	3	97.83	23.39	448.17	58.25	546.00	35.31
		III.3	3	184.33	70.21	347.00	58.8	541.33	126.63
		mean	6	146.08	71.18	397.58	77.38	543.67	92.99
B4	22.09.	III.3	3	115.67	45.84	190.33	21.45	306.00	62.12
		IV.1	3	139.83	27.48	163.67	50.33	303.50	76.46
		mean	6	127.75	39.68	177.00	40.92	304.75	69.67
B5	21.09.	III.3	3	113.00	12.68	270.10	18.61	283.10	28.39
		IV.1	3	183.83	62.28	287.33	16.01	471.17	68.33
		mean	6	148.42	57.22	278.72	19.38	427.13	68.38
B6	20.09.	III.3	3	119.00	57.71	223.00	19.80	342.00	66.71
		III.1	3	69.33	29.10	264.33	41.07	333.67	52.45
		mean	6	94.17	52.01	243.67	38.30	337.83	60.15
B7	20.09.	III.1	3	72.17	20.68	198.33	36.28	270.50	31.31
		III.5	3	162.17	31.56	277.67	50.31	439.83	51.79
		mean	6	117.17	52.32	238.00	59.14	355.17	94.87

done for various forest habitats in Puszcza Kampinoska, Poland (Kwiatkowska and Dudziec 1974). This showed unambiguously that significant differences existed both within and between phytocoenoses. This variability suggests that biomass should be measured in smaller, more uniform structural units of the herb layer. However, the synusial method is suitable for the description and characterization of such spatial systems. Because of the high patchiness of the herb layer the synusial approach to the typology of forest communities is inapplicable in the manner once used in Scandinavia (Du Rietz 1930, Lippmaa 1935) and also still used occasionally in forest communities of other regions (Gillet 1986).

The majority of the types of synusiae described in this work occur in both pine forests and mixed/pine forests. Among the 14 subtypes distinguished only 4 are clearly associated with pine forests and three with mixed/pine forests.

Variability in biomass from east to west occurred in both the different types of herb layer synusiae and in entire study areas. This variability is strongly expressed in the case of the biomass of mosses, and much more weakly in the case of vascular plants. Similarly, stronger and more significant correlations were found for pine forests than mixed/pine forests.

The data presented here on differences in biomass within the synusiae and different study areas along the west-east transect are a supplement to data from the literature. The values obtained are lower than those given for biomass in areas further to the east. Gabeev (1990) showed that pine forests in Western Siberia had mossy-cowberry herb layers with biomass ranging from 110 to 200 g/m^2 (of this the biomass of mosses was between 50 and 140 g/m^2). The pine forests with bilberry had herb layer biomass in the range of 90 to 130 g/m^2, while the biomass of mosses was lower – in the range of 15-20 g/m^2. In contrast, pine forests had grassy herb layers of highly variable biomass, with values as high as 304 g/m^2 but with minimal representation of mosses (about 1 g/m^2). It should be emphasized that mean biomass of the herb layer is influenced not only by geographical location, but by the edaphic conditions of the particular sites. For example, site S023 is characterized by a relatively high mean biomass (230 g/m^2) and at the same time by a higher (46.3 percent) proportion of species requiring moderately moist soils. These sites are also characterized by a higher percentage (17.8 percent) of nitrophilous species.

On the basis of vegetation features analyzed in these studies, we did not detect any changes at any site that can be explained by air pollution. Future studies of vegetation should concentrate on population level and such features of plant species as life form, phenology, and type of hemeroby.

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Herb Layer Diversity and Phytoindicative Evaluation of Habitat Conditions of Forest Permanent Plots in Poland[1]

Ewa Roo-Zielinska and Jerzy Solon[2]

Abstract

The influence of climatic and pollution gradients was determined on species richness, species diversity, and values of phytoindicators of the herb layer in pine and mixed pine forest communities in Poland. Geographical longitude was used as a synthetic measure of geographical position. Ellenberg's indices were used as synthetic measures of habitat differentiation, calculated as weighed averages: L (light index), T (temperature index), K (continentality index), F (moisture index), R (reaction index), N (nitrogen index). Richness and diversity variables were used: razem (total number of species), gatB (number of shrub species), gatC (number of herb layer species), diver (Shannon's diversity index $H = -\sum p\log p$), simps (Simpson's index $S = \sum p^2$), beta (beta =[1- simps]/[1-1/gatC]). Additional dependent variables included the proportion of number and cover of different Raunkiaer's life forms as well as shares of number and cover of species of different leaf anatomy and different syntaxonomic character. By using Ellenberg's indices the vegetation on consecutive sample plots was not strongly differentiated. All sites on the climatic transect occurred on dry, strongly acidic or acidic soil, with low nitrogen content. Values for the index of continentality "K" reflected clearly the climatic differentiation from west to east. The number of herb layer species depended on the geographical longitude (correlation coefficient 0.851) for pine and mixed forests together, 0.840 for pine forests only, and 0.926 for mixed forests only. All diversity indices were also strongly correlated with the longitude. Generally, the share of the number of chamaephytes and therophytes was positively correlated with the longitude. In pine forests there was also a general tendency for the number of scleromorphic species to decrease (and mesomorphic species to increase) in the west-east direction. Meadow species also generally tended to increase in the west-east direction. For all the plots taken together the cover of meadow species is correlated with the longitude (correlation coefficient 0.881), 0.905 for pine forests only, and 0.910 for mixed forests.

Introduction

Species richness and species diversity are some of the most important structural features of forest communities. They are influenced by a range of factors such as large-scale geography and habitat (Cowling 1990, Day and others 1988, Grime 1979, Jurko 1985, Prusinkiewicz 1970). Other important factors are the history of vegetation development in the Holocene (Huntley 1993) and the history of use (Law and Morton 1993) to interpopulational and successional interactions (Dollar and others 1992, Trevin and others 1993). The role of the different factors changes in relation to the scale of the phenomenon considered.

In analyses on a scale larger than the region, overall geographical factors, including macroclimatic differentiation, are most important. Macroclimatic differentiation consists of many elements, including mean annual temperature, total precipitation, moisture deficit, the amplitude of annual temperatures, the degree of continentality of the climate and the length of the growing season. These elements are mutually interlinked and are dependent on geographical location. It is for this reason also that many studies consider the link between structural features of the flora and vegetation and geographical location, treating the latter as a synthetic indicator of macroclimatic differentiation.

This paper defines the relationships between geographical location (which determines macroclimatic differentiation and reflects the history of development of the vegetation), habitat conditions (Ellenberg 1974), and selected characteristics of the species richness and diversity of the ground cover in two narrowly-defined types of forest community, namely, within typical pine forests and within mixed

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forests. The subjects of analysis are 17 study areas lying along a west-east gradient of continentality (Breymeyer, 1996).

Methods

The species diversity of the herb layer and the phytoindicators of habitat conditions were obtained from phytosociological records of all species occurring on 400 m^2 in each of the 17 plots. The majority of the records were taken in May 1995. A detailed description of the methodology and of the characterization of the vegetation cover is given in a separate study (Roo-Zielinska and Solon 1996).

Defined on the basis of information contained in each of the 17 phytosociological records were values for basic indices of diversity, namely: (a) the number of species of vascular plants; (b) the Shannon Index (Shannon and Weaver 1949), $H= -\sum plogp$; (c) the Simpson Index (Simpson 1949), $S=\sum(p^2)$; (d) the beta index (Simpson 1949) beta=[1- simps]/[1-1/gatC]; and (e) the index of evenness HHmax in the form HHmax=H/log(n) (where pi is the share of a species in the total cover of vascular plants in the ground cover layer; n = total number of species in a record). The number and proportion in cover of species representing different Raunkiaer life forms, types of leaf permanence, and types of anatomical structure also were calculated. Detailed characteristics of species were taken from Ellenberg (1974). The phytosociological structure of the ground cover layer was also described by using the list of characteristic species from Matuszkiewicz (1981).

The analytical method of indicative numbers from Ellenberg (1974) was used to assess habitat-climatic conditions (insolation "L" and the degree of continentality "K"), as well as habitat-edaphic conditions (the moisture "F" and acidity "R" of soils, as well as their nitrogen content "N"). All the phytoindicative analyses were done on the floristic composition of the ground layer, which reacts most vigorously to changes in abiotic conditions, and in which anthropogenic influences become visible most quickly. The index of insolation "L" indicates the relative strength of insolation (from 1: species requiring full shade; to 10: plant species requiring full light). The index of continentality "K" describes the resistance of a species to the frequency and length of occurrence of dry periods in the growing season, as well as to the length of the pre-frost period (from 1: the most marked features of an Atlantic climate; to 10: the most marked continental features). The index of humidity "F" expresses the ecological reaction of species in relation to the humidity of the substrate in the growing season (from 1: those requiring extremely dry soils; to 12: aquatic plants that are mostly completely submerged in water). The index of soil reaction "R" expresses biologically the acidity of the substrate experienced by plants (from 1: species requiring very highly acid soil [pH < 3.5]; to 10: those requiring neutral or basic soils [pH > 6.5]). The nitrogen index "N" expresses the ecological reaction of species to the content of nitrogen in the soil (from 1: species only occurring in soils poor in nitrogen; to 10: species only in soils very rich in nitrogen, such as those intensively fertilized). The mean values of phytoindicators were calculated twice for each of the phytosociological records — the first value was obtained only on the basis of the number of species of defined divisions of habitat requirements while the second (weighted mean) also included species cover. The method of Ellenberg and its various applications have often been discussed in the literature. It is useful in both the presentation of the spatial differentiation of current habitat conditions (Kostrowicki and Wojcik 1971, Roo-Zielinska 1982, 1994) and very well-suited to the dynamic expression of phenomena (Roo-Zielinska 1993, Roo-Zielinska and Solon 1990, 1994).

The relationships between geographical location and the values of phytoindicators (independent variables) and indices of diversity, as well as the proportions of species representative of different ecological groups (dependent variables) were defined on the basis of correlation and regression analyses for three groups of sites: for all 17 sites on the west-east transect; only for the 9 pine forest sites on the west-east transect; and only for the 8 mixed forest sites on the west-east

transect. The Stepwise Variable Selection procedure was used in the choice of the independent variables of strongest influence on changes in species diversity.

Results

The number of species and species diversity of the vascular plants in the ground layer increased in the west to east direction within both pine forests and mixed forests, with the value of the Shannon Index changing from 1.87 to 5.20, the value of the Beta Index within the range 0.82 to 0.99 and the value of the Simpson Index varying between 0.35 and 0.03. A similar range of variability was observed in the index of uniformity (HHmax), which varied from 0.71 to 0.96. The relationships between indices of richness and diversity and longitude were regularly statistically-significant. For all the indices analyzed the absolute correlation coefficients were above 0.74 *(table 1)*. The correlations between all the indices of richness and diversity and the Ellenberg indices were considerably weaker when all areas were considered together. All of them (except the number of lichens) were correlated with the coefficient of continentality *(table 2)*. In all three groups the relationships between the value for the index of species richness of the ground layer and the independent variables (longitude and phytoindicators) are best described by a straight line relationship gatC=(a*dlugeo)+(b*fito), in which gatC is the number of species of the ground layer; dlugeo the geographical longitude; fito the phytoindicative indices denoting: the mean index for light requirements (L) for all species (in the case of all 17 areas considered together or for pine forest only) or the mean index for moisture requirements (F) also calculated for all species only in the mixed forests *(fig. 1)*. Another situation arises in the case of the variability of values for the Shannon Index. In all three groups of sites, the relationships between the value of the Index and the independent variables is best described by a straight line with longitude as the dependent variable. However, the modifying influence of local habitat factors is minimal and not statistically significant *(fig. 2)*.

Variability was low in the proportion of different life forms across the gradient. Within mixed forests a strong positive correlation was found between the relative cover of chamaephytes and geophytes in the overall cover and the longitude. A similar, but weaker, relationship was also observed in the case of joint analysis of all transect sites *(table 1)*, while differences in habitat conditions exerted a weaker influence on the number of different life forms *(table 2)*.

Geographical variability in the proportion of species of different leaf longevity was not apparent. Only in the case of the number and cover proportion of species with green wintering leaves showed a clear correlation with longitude *(table 1)*. This relationship is best described by an exponential equation or regression the form Y=exp(a+bX), albeit with the parameters of the line not differing significantly between all the analyzed blocks (*fig. 3*).

Clear geographic variability was found in the proportion of species of different anatomical types. Positive correlations were found between the quantitative shares of meso- and hygromorphic species and longitude, while inverse correlations characterized the relationship between the quantitative shares of scleromorphic species and longitude *(table 1)*. This latter relationship was best expressed by an exponential regression *(fig. 4)*.

Species characteristics of the class *Vaccinio-Piceetea* were the main components of the ground cover layer of the areas studied. Their shares in the total number of species varied from 20 to 50 percent, but they always constituted the dominant group. The cover of this group varied from 21 to 74 percent of total cover.

The composition of the ground cover of the studied areas also included other groups of species of which the most important were those characteristic (Matuszkiewicz 1981) for the classes *Molinio-Arrhenatheretea*, *Sedo-Scleranthetea*, *Nardo-Callunetea*, *Querco-Fagetea*, *Trifolio-Geranietea*, *Festuco-Brometea* and *Epilobietea*.

The geographical variation in the numbers of species of differing syntaxonomic affiliation was different for different groups. Statistically significant correlation

relationships between longitude and the amount of a particular group were more numerous and stronger in the case of areal representation than in the case of quantitative representation *(table 1)*. The clearest relationship was observed in relation to the cover proportion of species characteristic of the classes *Molinio-Arrhenatheretea* and *Vaccinio- Piceetea*. This relationship was best expressed by the straight line relationship of the form Y=p1+p2X *(fig. 5)*.

Table 1 *— Statistically significant correlations between geographical longitude (independent variable) and diversity indices and shares of different species groups.*[1]

	All plots on transect		Pine forests on transect		Mixed forest on transect	
Number of plots	**17**		**9**		**8**	
all vascular species	0.8422***		0.8506***		0.9246***	
shrub species	0.7903***		0.8528***		0.9267***	
herb layer vascular species	0.8509***		0.8396***		0.9260***	
beta index	0.8978***		0.9241***		0.8823***	
Simpson's index	-0.8812***		-0.9156***		-0.8977***	
Shannon's index	0.9058***		0.9186***		0.9502***	
HHmax	0.8733***		0.9004***		0.8686***	
	by species cover	by species number	by species cover	by species number	by species cover	by species number
Herbaceous Chamaephytes	0.6047*	0.5013*			0.7919*	
Geophytes	0.5209*				0.8212*	
Terophytes		0.6911**	0.6953*	0.8363**		
evergreen species	-0.7409***		-0.9099***	-0.7489*		
overwintering green species	0.7261***	0.8363***	0.7834*	0.8484**	0.8208*	0.8439**
summergreen species		-0.6691**				-0.7871*
helo- and higromorphic		0.7528***		0.8571**		
mesomorphic species		0.5902*		0.7621*		
scleromorphic species		-0.7856***		-0.8918**		
EP	0.5165*				0.8619**	
MA	0.8813***	-0.6079**	0.9046***		0.9096**	
NC		0.7567***		0.7525*		0.8430**
QF				0.7297*	0.7328*	-0.7466*
TG	0.6815**			-0.7324*	0.8246*	
VP	-0.6542**		-0.7900*			

[1]* = $p < 0.05$
** = $p < 0.01$
*** = $p < 0.001$

Table 2 *— Statistically significant correlations between phytoindicative characteristics (independent variables) and species richness of C layer.*[1]

	All plots on transect		Pine forests on transect		Mixed forest on transect	
Number of plots	**17**		**9**		**8**	
	by species cover	by species number	by species cover	by species number	by species cover	by species number
L	-0.5277*	-0.5277*				
K	0.5702*	0.5563*				
F					0.7770*	
R	0.5682*	0.5682*	0.7073*	0.7073*		

[1]* = p < 0.05
** = p < 0.01
*** = p < 0.001

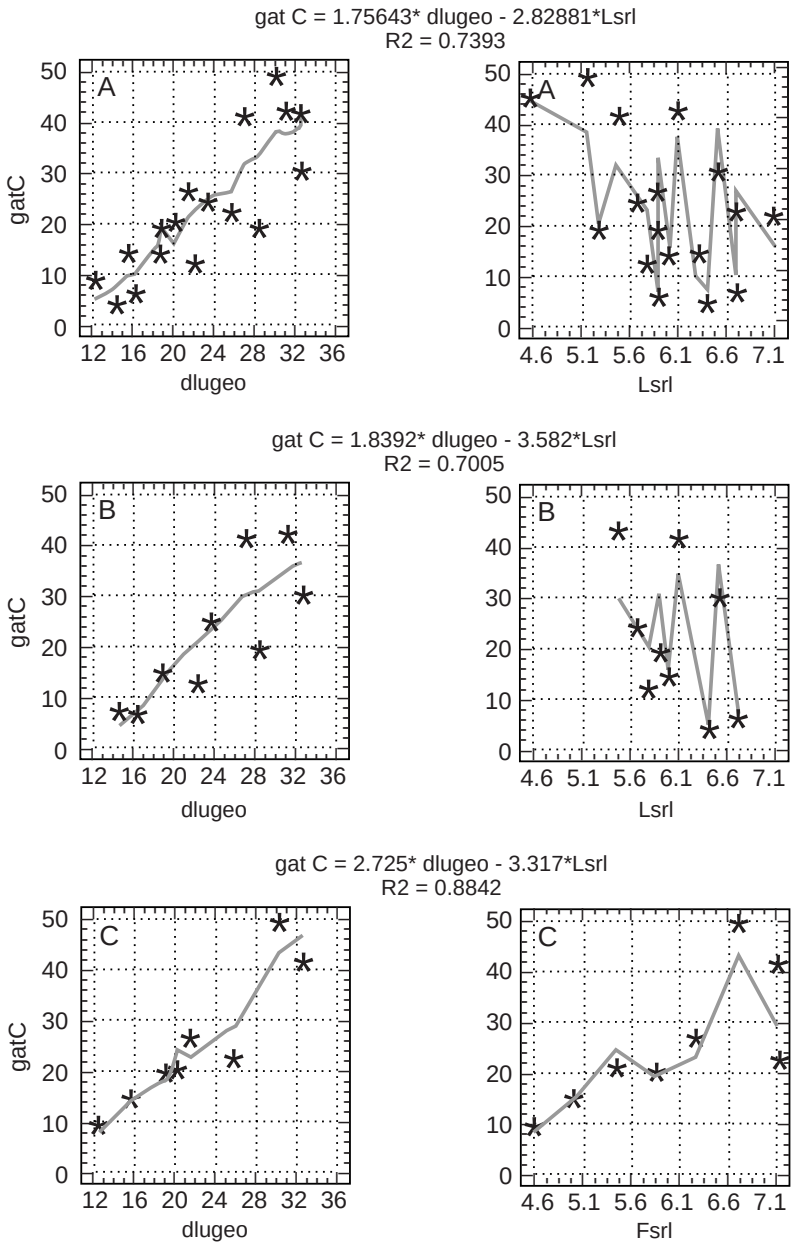

Figure 1 — Regression lines for the relationship between the number of species of the herb layer (y axis) and geographical longitude and phytoindicative indices (x axis). (A) All plots on the west-east transect; (B) pine forests on the transect; (C) mixed forests on the transect. Key: dluggeo = longitude; gat C = number of species; Lsrl = mean index for light requirements; and Fsrl = mean index for moisture requirement.

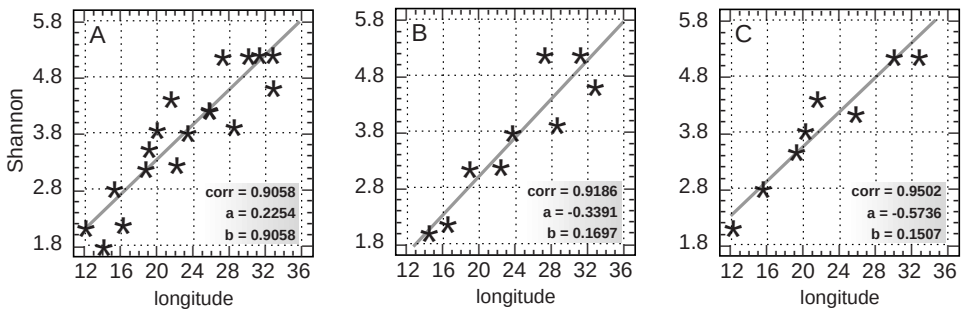

Figure 2 — Regression lines for Shannon index (y axis) on geographical longitude (x axis) according to the linear model Y=p1+p2X. (A) All plots on the west-east transect; (B) pine forests on the transect; (C) mixed forests on the transect. Dluggeo = longitude; gat C = number of species; LsrI = mean index for light requirements; and FsrI = mean index for moisture requirement.

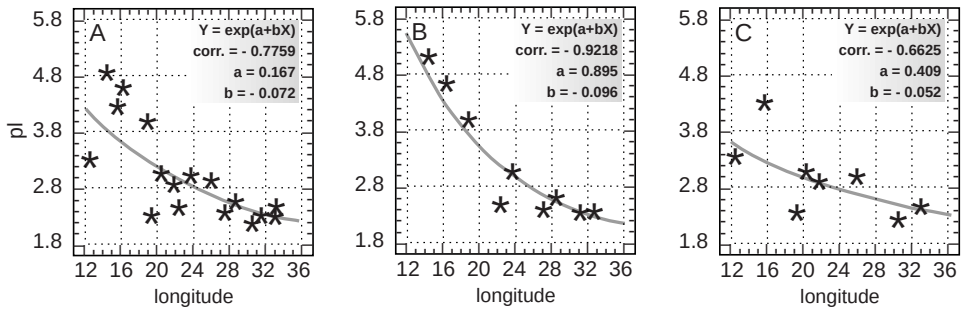

Figure 3 — Regression lines for the share in cover of overwintering green species (y axis) on geographical longitude (x axis) according to the model Y=exp(p1+p2X). (A) All plots on the west-east transect; (B) pine forests on the transect; (C) mixed forests on the transect.

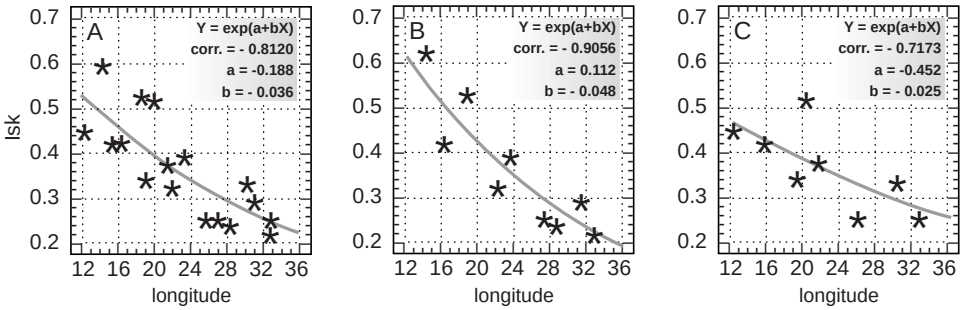

Figure 4 — Regression lines for the share of scleromorphic species (y axis) on geographical longitude (x axis) according to the model Y=exp(p1+p2X). (A) All plots on the west-east transect; (B) pine forests on the transect; (C) mixed forests on the transect.

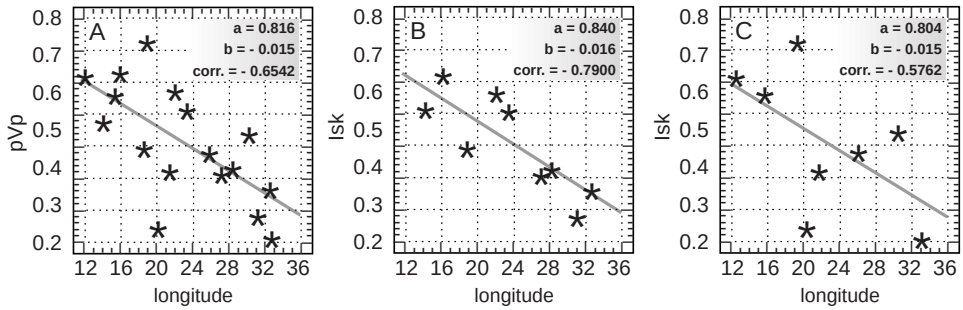

Figure 5 — Regression lines for the proportion of cover of characteristic species of the Vaccinio-Piceetea class (y axis) on geographical longitude (x axis) according to the model Y=p1+p2X. (A) All plots on the west-east transect; (B) pine forests on the transect; (C) mixed forests on the transect.

Results and Conclusions

The transect on the 52° north latitude between 12 and 33° longitude east in Poland had varying ground cover in sites with pine forest and mixed forest. From west to east we observed an increase in the total number of vascular plant species per unit area as well as in the species richness of vascular plants in both the shrub and the herb layer. However, the number of species of lichens is not correlated with longitude but is entirely determined by local habitat conditions and the degree of maturity of the phytocoenosis. An increase was also calculated in values of the Shannon diversity index and the beta index; and a simultaneous fall was found in the value of the Simpson Index. In addition, increases were determined for the value of the index of evenness HHmax, which attests generally to changes in the dominance structure from species poor to species rich; the number cover of species with green wintering leaves (with a simultaneous fall in the cover of evergreen species); the number of mesomorphic species (and a fall in the number of scleromorphic species); and the proportion of meadow species in the overall cover.

The results clearly confirm the assumption that within a single narrowly-defined type of forest habitat, longitude is the main factor influencing the species richness and diversity of the herb layer, as well as its ecological and phytosociological structure. In spite of this finding, the relationships obtained should be treated as first approximations of a general model linking the structural characteristics of the community with environmental conditions. According to Baker (1990), differences in species richness may only be explained on the basis of a multi-factor model; thus, with any hypothesis linking species diversity with only one factor, this has limited significance.

In the case of forest communities, the number of species in the ground layer is shaped not only under the influence of edaphic and geographic conditions, but is also dependent on the age, character, and structure of the tree stand. In particular, the character of the herb layer is often determined by the amount of solar energy reaching the forest floor (Specht and others 1990, Specht and others 1991, Specht and Specht 1993). Specht and Specht (1989b) showed that the influence of the light infiltration through the crown is different for different ecological groups of the herb layer. In particular, increased shading is associated with a linear decrease in the number of sclerophilous species. Also, the amount, time, and spatial variability of the litter fall may affect the heterogeneity and richness of the herb layer (Smith 1987). Other factors having a great influence on the richness and ecological character of the ground cover layer is the dynamic stage and level of anthropogenic impact on the forest community. It has been shown many times that mature forest communities and/or those little modified from the natural state are characterized by considerably lower proportions of annual and biennial plants when compared with early developmental stages or phytocoenoses subject to strong anthropogenic impacts (Dodge and others 1983, Dollar and others 1992, Kirby 1988, 1990, Trabaud and Lepart 1980, Trevin and others 1993, Vankat and Carson 1991). On the other hand, under moderate anthropogenic pressure overall species richness may increase (Grime 1979, Reader and others 1991).

Acknowledgments

The work was done within the framework of a project entitled "U.S.-Poland Cooperative Project on the Status and Long-Term Trends in Forest Ecosystems: Climate, Pollution and Forest Health" financed by the U.S. Environmental Protection Agency, USDA Forest Service, USDA Foreign Agriculture Service (ICD), Polish Academy of Sciences, Polish Bureau of GEF, and Forest Research Institute. Also participating in the financing of the work were: Michigan Technological University, Houghton, Michigan; Bowling Green State University, Bowling Green, Ohio, as well as the Institute of Geography and Spatial Organization of the Polish Academy of Sciences, Warsaw. The present work is not reviewed by the financing organizations and may not represent their views.

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Spatial and Vertical Distribution of Soil Physico-Chemical Properties and the Content of Heavy Metals in the Pedosphere in Poland[1]

Marek Degorski[2]

Abstract

The lithological and petrographical characteristics of soil pedogenesis was determined, and the spatial and vertical distribution of some soil physico-chemical properties (including heavy metal content) were studied along two transects in Poland. The genetic horizon for 22 soil profiles were described for particle size and petrographic composition, quartz grain abrasion, transformation of organic substance, carbon and nitrogen characteristics, base saturation, exchangeable cations, and acidity. The results showed that the morphogenesis of parent rock in all studied profiles is characterized by the glaciofluvial sedimentation of fine sands. All research plots were situated on valley terraces or outwash planes connected with glaciofluvial accumulation of redeposited, polygenetic sand formation. Soils belonged to three subtypes of Spodosols (podsolic class). All soil profiles were characterized by a very acid reaction (pH 3.0-4.5 in horizon A), and the degree of acidity corresponded to soil type. All other defined physico-chemical properties indicated that soils had low nutrient status. Mineral-humus layers of most of the soils were characterized by a high carbon:nitrogen ratio (from 10.6 to 53.7). This suggests limited biological activity of the soils and slow humification and mineralization rates. Overlying humus has three distinct subtypes (mor, moder/mor, mor/moder). In all soils the ratio of humic acids to fulvic acids was less than one, with the highest content of fulvic forms in humus compounds. Total exchangeable cations averaged 2.5 meq per 100 g of soil from the mineral-humus horizon and 0.5 meq per 100 g of soil in the parent rock. The degree that the sorption complex was saturated with base cations was also very low (less than 10 percent). Heavy metal spatial distribution in studied soils was not distinguished by region, but by local pollution sources or by natural concentration in the soil.

Introduction

Pine forest ecosystems with associated podzolic soils are very common in central and eastern Europe. For example, in Poland the potential natural plant association of coniferous communities (*Vaccinio-Piceetea*) occupies about 25.3 percent of the total area of the country (Matuszkiewicz 1991). Similarly, the same area of Poland is characterized by podzolic soils (Spodosols). The highest concentration of podzolic soils (39.7 percent) is in the central Polish lowland (Prusinkiewicz and others 1980). All the study plots of both the climatic and Silesian transects in Poland (Breymeyer, this volume) are located on podzolic soils. Research was done in 22 soil profiles (Breymeyer and others 1995, Breymeyer 1996).

This study determined the geographical differentiation of lithological and petrographical conditions of the pedogenesis of soils and analyzed the spatial and vertical distribution of some soil physico-chemical properties, including the content of heavy metals.

Methods

Base soil profiles and diagnostic excavations were done at study areas selected in accordance with the concept of the research. The basis for pedological research was the diagnosis of the different genetic horizons of 22 soil profiles on two research transects. For each profile, soil material was collected for analysis from different genetic horizons. A series of determinations were carried out concerning the morpholithogenic properties of the soil substratum (Degorski 1994a, b; Degorski [In press a]), the overlying humus and the organic matter in the mineral-humus layers (Degorski and Jefriemow [In press]), the physico-chemical properties of soils

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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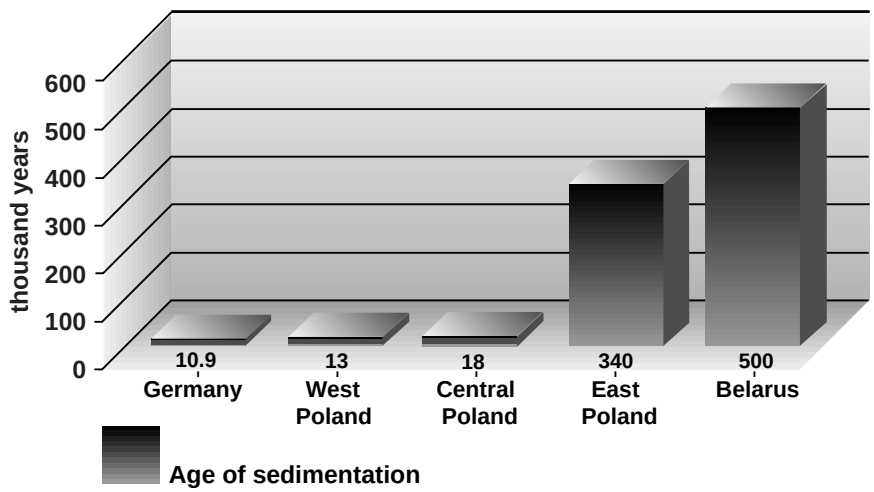
(Degorski 1995a, b; Degorski [In press b]), and the heavy metal content of the soils (Gworek and Degorski [In press]) by using standard soil science methods. Differences and similarities between the morpholithological features of the study sites were analyzed, as well as the physico-chemical features causing the breakdown of heavy metals and trace elements. Euclidean distance and Ward's method (Batko and Moraczewski, 1987) were used. Trace elements and heavy metals (Zn, Mn, Fe, Pb, Ni, Cr, and Cu) were determined using extraction with 20 percent HCl, after the combustion of organic matter at a temperature of 480 °C. Concentrations of these elements were determined by atomic adsorption spectrophotometry (AAS). An accumulation index was calculated from the data for all mineral-humus (A), eluvial (E) and illuvial (I) horizons. The accumulation index was calculated as the ratio of the content of a given element in the genetic horizon and in the parent rock, giving an indication of exogenic origin in the surface layers of the soil (Gworek 1985).

Results

Physico-Chemical Characteristics of the Soils

All of the soils studied were developed in redeposited, polygenic sandy formations that accumulated in the Pleistocene and Holocene. These differed only in the age of sedimentation, which is associated with the late Pleistocene (*fig. 1*).

Figure 1 — Age of last sedimentation for soil substratum.



The studied habitats of pine forests and mixed pine forests developed on formations of river accumulation terraces or glaciofluvial forms (Degorski [In press a]). All of the studied soil samples were loose and poor-clay sands of moderate or varied grain size and with clear features of presorting, as well as limited contents of skeletal fractions (over 1 mm). The surface layers of soils located on terrace areas (e.g., K092, K098, KB001, and S126) were characterized by increased proportions in their mechanical composition of silty and dusty fractions. This was caused by an increase in deflational processes that took place in these areas in periglacial periods. The fluvial character of the sediments and their limited surface aeolization was confirmed by granulometric indices calculated on the basis of their mechanical composition (Degorski [In press a]). Cumulative curves characterizing the fluvial material and constructed on Phi probability plots (Program Analiza Uziarnienia 2.0 - Mechanical Composition 2.0) depart clearly from the log-normal distribution. This characterizes a leptokurtic breakdown (GSP 1.5-1.9) because curves for sands with features scattering over a considerable interval resemble straight lines and have a distribution close to the mesokurtic (GSP 1.3-1.4) (*fig. 2*). All of the studied soils are Spodosols. Morphological features and chemical criteria allow for their inclusion within three subtypes: podsols, rusty podsols, and podsolic rusty.

All of the profiles studied are characterized by very acid reactions. For the mineral-humus layers, pH values ranged from 3.0 to 4.5, with the degree of differentiation corresponding to the type of soil (*fig. 3*). In addition to acidic reactions, other analyzed physico-chemical properties of the soils also indicate low nutrient status. Total exchangeable cations (V) amounted on average to 0.5 meq per 100 g of soil in the mineral-humus layer. The degree of saturation of sorption complexes with base cations is very low in the mineral-humus horizons and around 20 percent in the parent rock (*fig. 4*). All of the soils studied on both transects are also characterized by very broad ratios of organic carbon (C) to total nitrogen (N). The C:N ratio in mineral-humus horizons amounts to values of about 10.6 to 53.7, indicating very limited biological activity of the soil and slow processes of humification and mineralization (*fig. 5*).

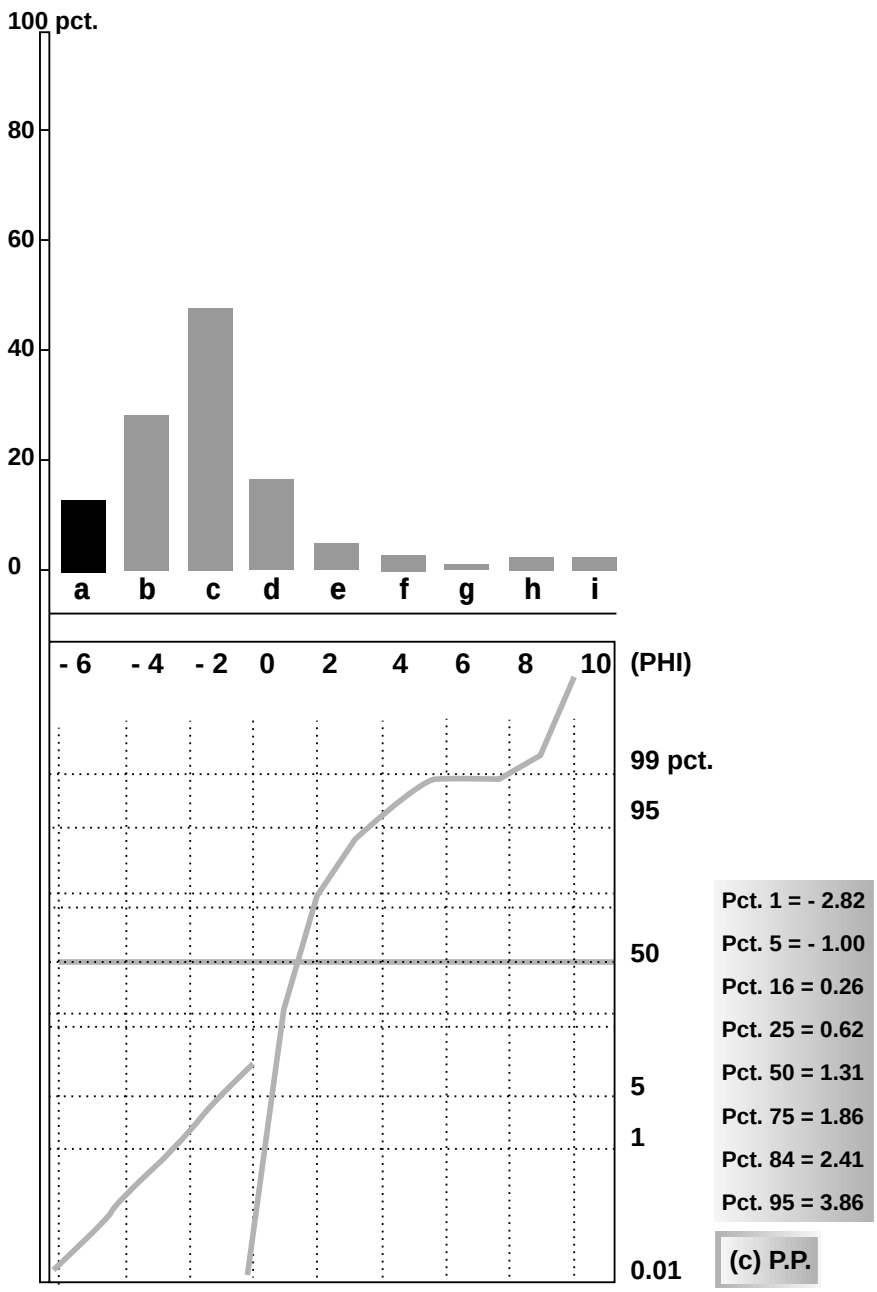


Figure 2 — Soil texture characteristics defined for research site KS011, horizon-B. a - i = particle size fractions:
a = stones and gravel (over 1 mm);
b = coarse sand (0.5 - 1 mm);
c = medium sand (0.25 - 0.5 mm);
d = fine sand (0.1 - 0.25 mm);
e = very fine sand (0.05 - 0.1 mm);
f = coarser silt (0.02 - 0.05 mm);
g = silt (0.005 - 0.02 mm);
h = fine silt (0.002 - 0.005 mm);
i = colloidal clay (below 0.002);
perc. = percentile.

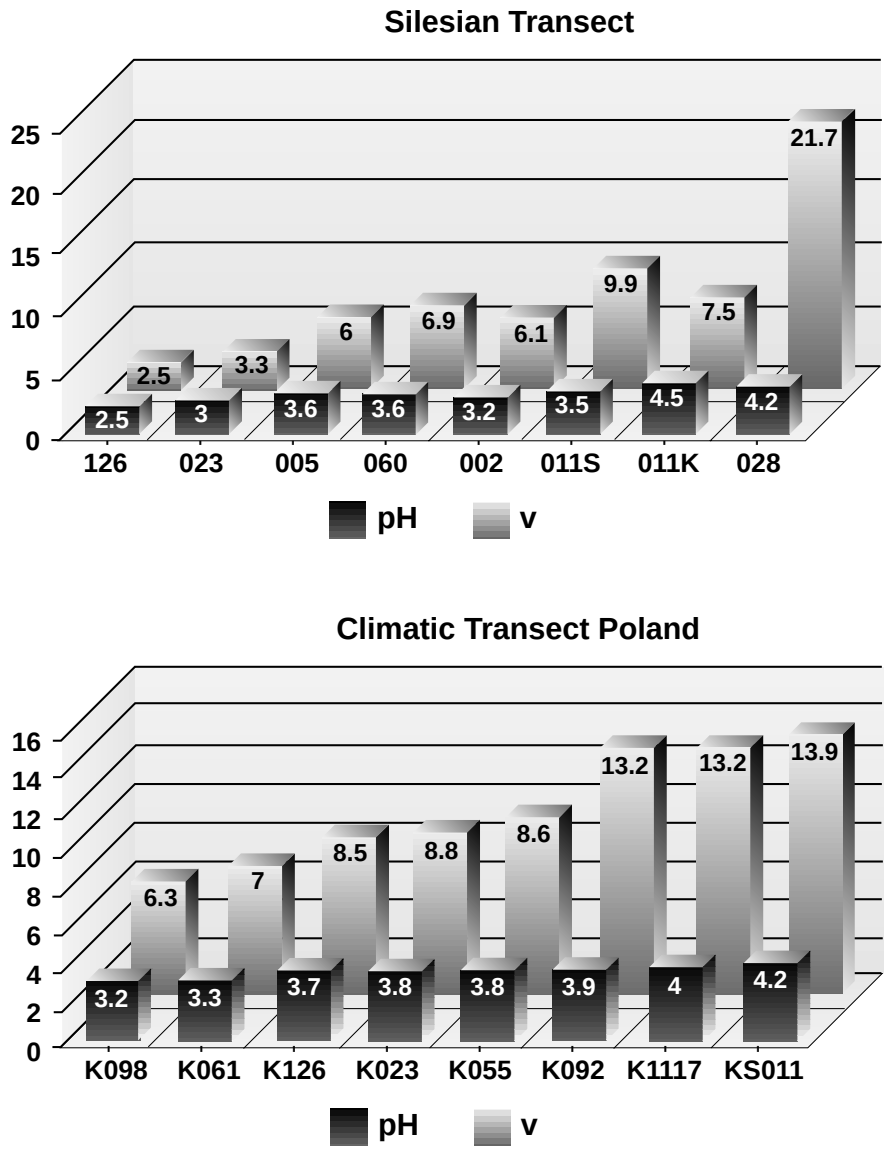


Figure 3 — Relationship between pH and base saturation (V) on two transects.

Analysis of organic matter also pointed to the poverty of the studied soils and slow processes of humification. The ratio of humic to fulvic acids is less than 1 in all the studied soils, indicating humus of the first type in the Kononowa classification (1968), i.e., that with a low degree of condensation of aromatic ring compounds (Degorski 1994a, b; Degorski 1995a, b). On the basis of statistical analysis of the results for physico-chemical properties, properties of organic matter, humus type and properties of the substrate, the studied soils may be divided into three subgroups differing from one another in a statistically significant way (*figs. 6, 7*). These groups are linked with the genetic type of the soil (podsols, rusty podsols, or podsolic rusty), with the type of the overlying humus (mor, moder/mor, mor/moder), and with the mean characteristics of physico-chemical properties.

The first group includes podsol soils with mor-type humus and the lowest pH values (3.0-3.5). These are characterized by the lowest rates of mineralization and humification (the greatest thickness of overlying humus and the widest ratio for humic to fulvic acids). The second group are rusty podsol soils, with moder/mor type humus of lesser thickness and with better sorption properties. The third group are rusty with poorly-marked processes of leaching (a lack of a clear eluvial horizon). The overlying humus is of the mor/moder or moder types and has the least thickness and the highest rate of mineralization and humification of organic compounds (the narrowest C:N ratio). The low carbon content together with the high

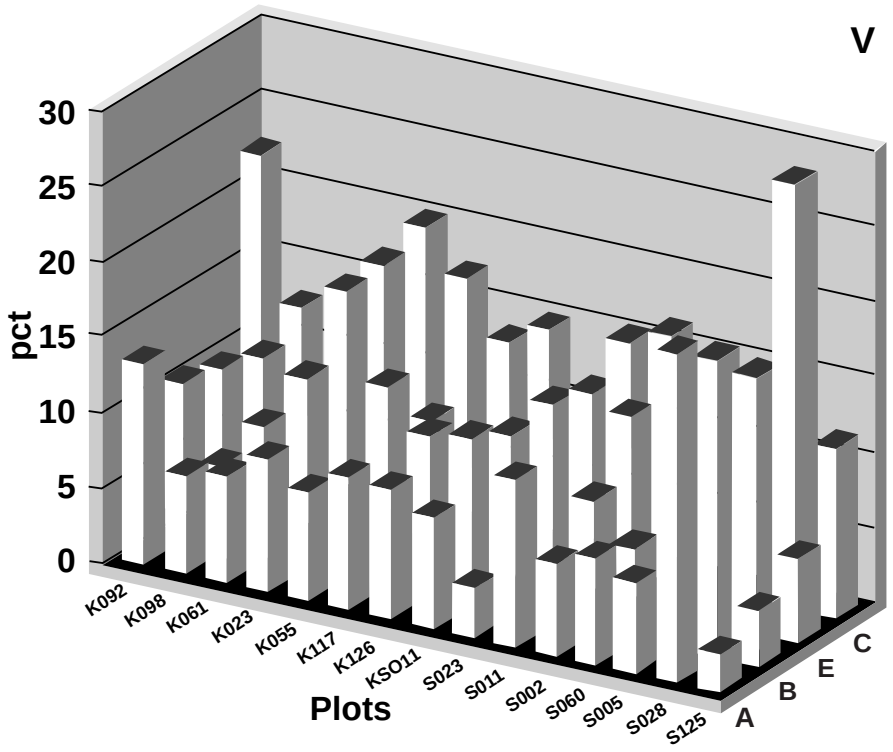


Figure 4 — Percent base saturation for whole soil horizons (A, E, B, C) and research plots.

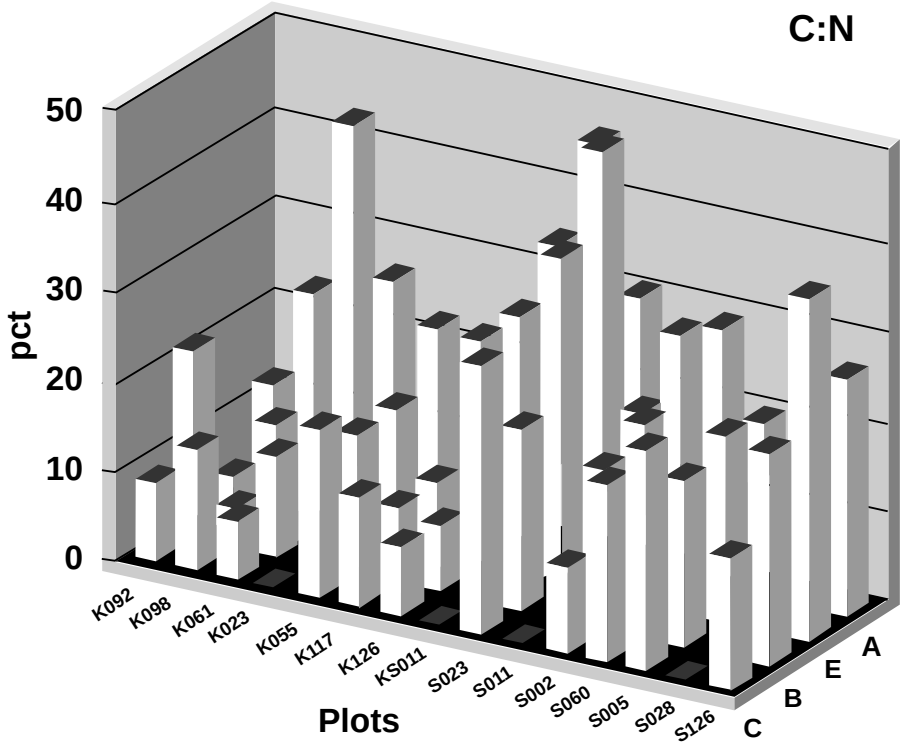


Figure 5 — C:N ratio in soils for each soil horizon.

degree of humification and the greater content of humates and humic acids (the narrowest Ch:Cf ratio) all indicate soils with significantly more advanced evolution of organic matter. The sorption properties of this group of soils are also the best among those studied. The degree of saturation of the sorption complex with base cations increases from 10-20 percent in the mineral-humus layers to about 30 percent in the parent rock, while the reaction (pH) varies from about 4 to 4.5 in the mineral-humus horizons (*fig. 7*).

The Breakdown of Heavy Metals and Trace Elements

There is no statistically-significant basis for claims regarding geographical-scale variation in contents of heavy metals and trace elements along the two transects. The only differences observed were in relation to regional distributions. The increase in the contamination of a soil with some element is most often caused by some local source of pollutant emissions. A good example of this situation is the spatial distribution of the lead content in soils along the two transects (*fig. 8*). The highest level of lead was recorded at point S028 at Klucze near Olkusz. This is traditionally

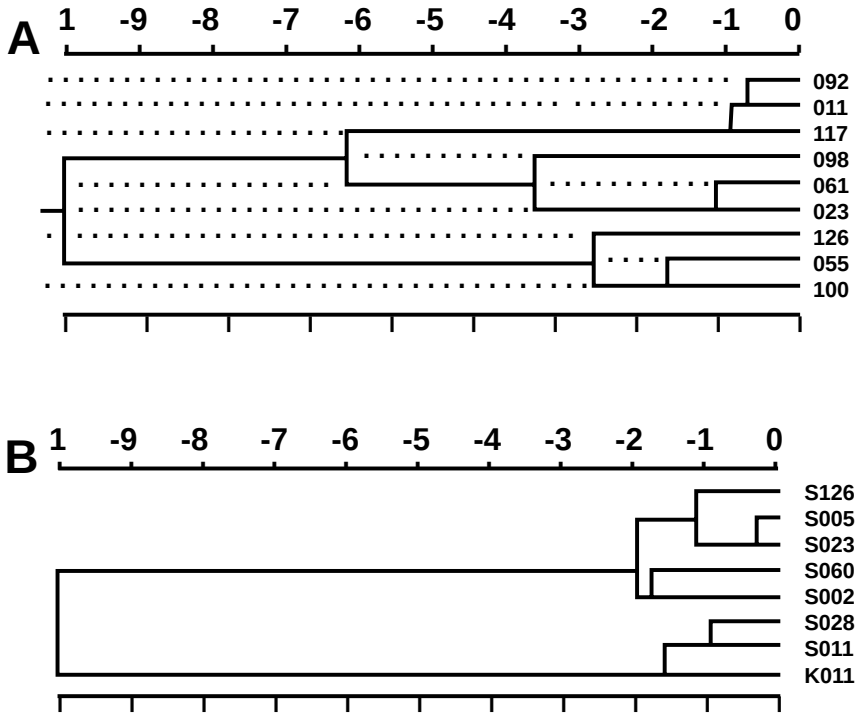


Figure 6 — Similarity of the soil properties of two transects (Euclidean distance/Ward's Method) (Batko and Moraczewski 1990). a = climatic transect, b = Silesian transect.

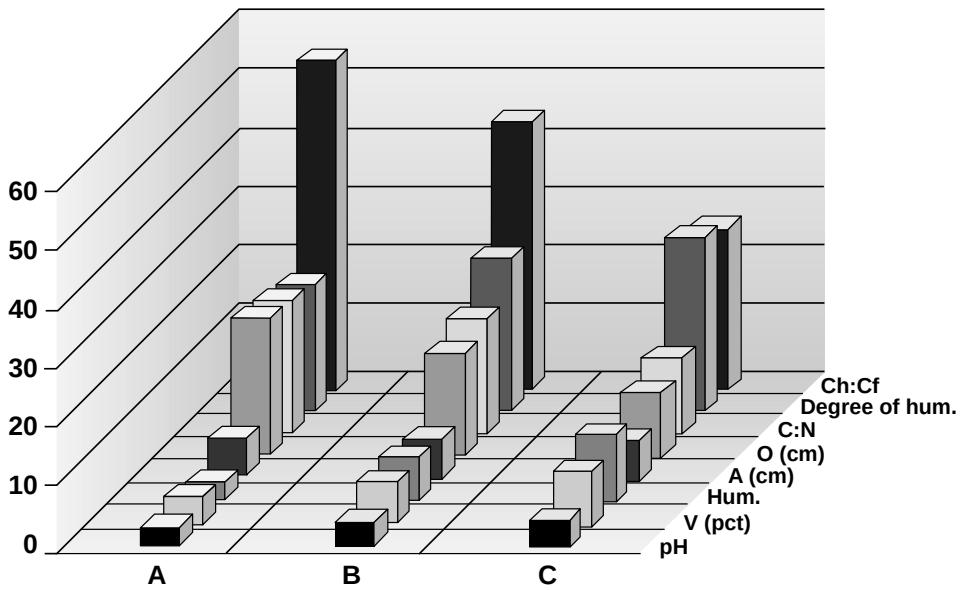


Figure 7 — Selected physical and chemical properties of the mineral-humic horizon defined for selected soil groups, pH = acidity determined in H_2O; V = saturation degree of the sorption complex with cations of basic character in percent; Hum = contents of humines; A = thickness of the A horizon in cm; O = thickness of the O horizon in cm; SH = humification degree in percent; Ch:Cf = the ratio of humine to fulvic acids x 100 percent. A = podzolic soil, B = rusty-podzolic soil, C = rusty soil

and remains a region in which zinc and lead ores are extracted and processed. The contents of lead in the remaining points analyzed were linked to both the location of industries and transportation routes.

The indices of accumulation defined for all of the studied soils confirmed ideas concerning the spatial distribution of the contents of heavy metals and trace elements in the profiles. The greatest concentration occurs in the levels of overlying humus (O) and of the accumulation-humus horizon (A). The highest values for the index were attained for lead (to about 15 times the content of this element in the parent rock) and manganese (to about 8 times the content) (*fig. 9*). The distribution of concentrations of heavy metals in the upper layers of the soil differed from those of the chemical background in the parent rock, indicating specific geographical relationships. The content of zinc is clearly higher in Upper Silesia in connection with the concentration of industry in this region. In contrast, the contents of nickel, iron, and chromium were clearly greater in the mineral-humus horizons of western Poland. The higher content of these three elements in the soils of the western part of the country may point to the transfer of pollution from western Europe, which evidently has an impact on the heavy metal contamination of the soils in this part of Poland. Such a finding is in accordance with the dominant wind direction in Poland (westerly and north-westerly).

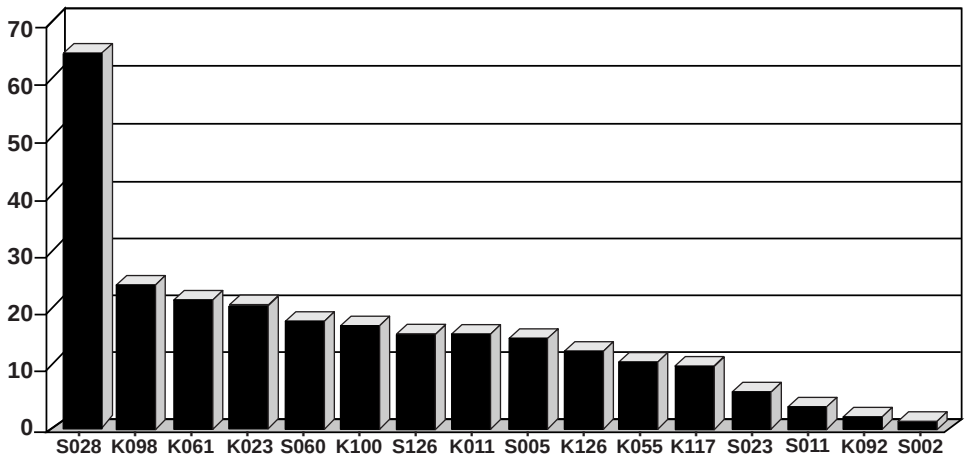
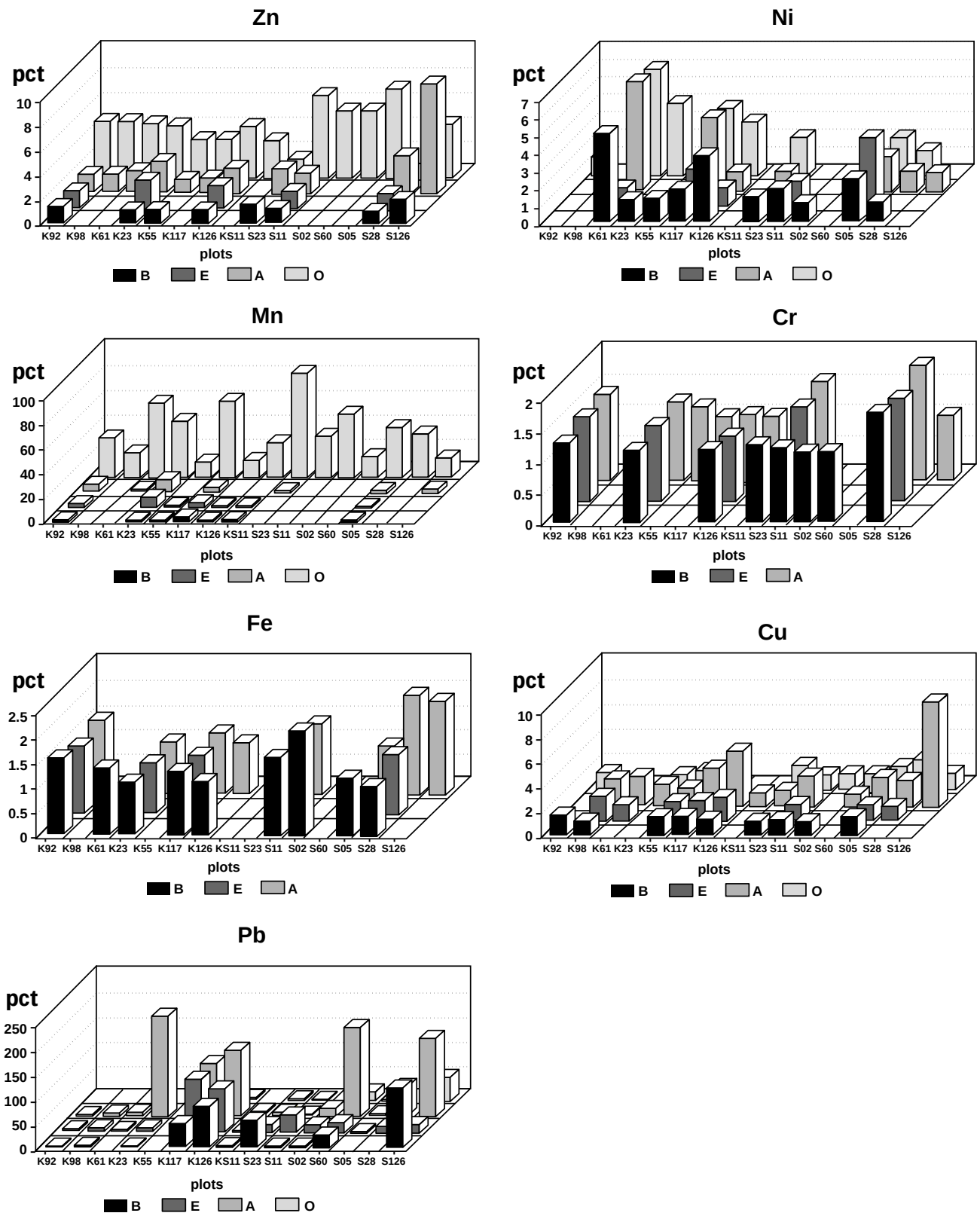


Figure 8 — Spatial variability of lead distribution on two research transects. Bars are showing Pb mg/1000 g of soil.

Conclusion

All of the research sites were linked morphogenetically with the accumulation of redeposited, polygenetic sand formations that occurred in the Pleistocene. Soils on both transects belong to three subtypes of Spodosols: podsolic, rusty podsolic, and rusty. All of the studied profiles were characterized by high acidity (pH of 2.5 to 4.5) and a degree of variation in this feature that corresponds to soil type. All other defined physico-chemical properties were also indicative of the fact that soils were of low nutrient status. Many of the soils were characterized by mineral-humus layers with a high C:N ratio (ranging from 10.6 to 53.7). This attests to the low nutrient cycling rates of the soils and the slowness of the processes of humification and mineralization. The lowest values of humic acids and humines were determined in podsolic soil, suggesting that mineralization rates are lowest in these soils.

Figure 9 — Accumulation index of heavy metals and trace elements.



Acknowledgments

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Transect Studies on Pine Litter Organic Matter: Decomposition and Chemical Properties of Upper Soil Layers in Polish Forests[1]

Alicja Breymeyer,[2] Marek Degorski,[2] and David Reed[3]

Abstract

The relationship between litter decomposition rate, some chemical properties of upper soil layers (iron, manganese, zinc, copper, lead, mercury, nickel, chrome in humus-mineral horizon-A), and litter (the same eight elements in needle litter fraction) in pine forests of Poland was studied. Heavy metal content in organic-mineral horizon of soils was highly correlated with needle decomposition rate for copper (correlation coefficient 0.901), zinc (corr. coeff. 0.901), nickel (corr. coeff. 0.834), and iron (corr. coeff. 0.850). Heavy metal content in needle litter was highly correlated with needle decomposition rate (iron—corr. coeff. 0.83; zinc—0.80; lead—0.82; nickel—0.89), and with mixed litter decomposition rate (iron—corr. coeff. 0.71; zinc—0.66; lead—0.68; nickel—0.72). Significant correlations were not found between the rate of decomposition of wood or cones and their content of metals, except for chromium. When comparing decomposition rates with degree of litter pollution, it was shown that in the case of needle decomposition for six significant correlations, two were positive; in the case of mixed litter decomposition for five significant correlations, two were positive. In both cases decomposition reacted positively to the presence of iron and lead. When comparing decomposition rates with the degree of soil pollution, it was shown that in the case of needle decomposition, five significant correlations were positive; in the case of mixed litter decomposition, only one significant correlation was positive.

Introduction

Although the level of pollution in Poland has declined steadily in recent years (Breymeyer 1996, Przybylska 1995, Statistical Yearbook of Katowice Voivodeship 1992), pine stands have continued to be damaged by it (Wawrzoniak and Malachowska 1996, 1997). The saturation of litter layers and soils with heavy metals is still at a level harmful to the proper functioning of the forest ecosystem (Laskowski 1994).

Research on soils and the rate of decomposition of forest litter was carried out at 15 sites with pine forest (*Leucobryo-Pinetum* and *Peucedano-Pinetum*) or mixed pine forest *(Quercoroboris-Pinetum)* growing on the podsolic soils developed on the glaciofluvial and fluvial deposits (Degórski 1996a). The sites were distributed along two transects: one west-east across Poland at parallel 52° N (climatic or continental transect), and the other from Upper Silesia to the Bialowieza area (Silesian transect) (Breymeyer 1996).

The sites were slightly different in annual precipitation, while mean annual temperature was distinctly lower to the east (Smialkowski 1996).

The chemical properties of overlying humus layers were studied, as well as the mineral-organic horizons of soils, where organic matter accumulates and is mineralized and humificated. The sorption complexes are most vulnerable to the accumulation of heavy metals, and their buffering properties determine the geochemical balance of the whole ecosystem. Permanent circulation of soil organisms between upper soil horizons, humus, and litter creates one functional unit of layers.

This study analyzed the relationship between the rate of decomposition of forest litter and the chemical characteristics of the mineral-humus horizon-A of soils by using deposition measurements of pollutants from local emissions in both Upper Silesia and the continental transect sites in Poland.

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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Methods

Rates of decomposition were measured using the common litter bags method (i.e., Pan-European Forest Ecosystem Research Program; Berg and Co 1993). Weighed portions of litter dried to constant weight were enclosed in bags (10 by 10 cm) made from 1 mm nylon netting. Bags were then located at the sites so that decomposition could take place. The surfaces on which the bags were placed were cleaned of moss and small shrubs to ensure that the exposed litter made contact with the upper layer of the soil. Sixty litter bags were placed simultaneously at each site: 20 with 1g of pine needles each, 20 with 1g of mixed litter, 10 with a single pine cone, and 10 with 1g of small twigs. The exposure of litter bags always began in autumn (October), with samples collected after two time periods: half after winter (in March or April) and the remaining half during the following autumn (October), when samples for the next year were placed on the soil. In every case only litter that had fallen to the forest floor during the previous year was used (litter plots were cleaned before the counted fall). Litter was collected from special demarcated areas, ensuring that the litter collected was always "of the same age." After field incubation, litter bags were transferred to the laboratory and cleaned of soil and plants overgrowing the netting. Litter was then dried to constant weight and weighed. The mass loss during incubation provided a measure of decomposition expressed in percentage terms or as the so-called "k" coefficient, which was calculated by applying the formula:

$$\ln\ (\text{final mass} / \text{initial mass}) = -k$$

Sinusoidal transformation of the results obtained for decomposition was also applied:

$$(\sin^{-1}(x^{1/2}).$$

The pollution of the organic-mineral horizon-A of the soil was estimated on the basis of analyses of soil samples collected on a single occasion from 10 points on each mineral-organic horizon of a given profile, with the mean value presented. Contents of heavy metals, such as zinc (Zn), manganese (Mn), iron (Fe), lead (Pb), nickel (Ni), chromium (Cr), and copper (Cu), in soil organic matter were determined using an atomic adsorption spectrophotometer (AAS) after digestion of samples in 20 percent HCl. An index of accumulation was calculated on the basis of the results obtained for all mineral-humus horizon-A as a measure of their enrichment by the heavy metals. The index of accumulation was calculated as the ratio of the content of a given element in the organic-mineral horizon-A and in the parent rock, giving an indication of an exogenic origin n the surface layer of the soil (Gworek 1985; Degorski, [In press]).

Litter (needles, twigs, and cones) collected annually from all sites was chemically analyzed for the concentration presence of eight metals as described above for soil organic matter. Statistical calculations involved analysis of variance for the 10 elements and the 3 fractions.

Results

The Pollution of Litter

The concentrations of Pb and K were not statistically different among the litter fraction ($p > 0.05$). Concentrations of the remaining elements (Fe, Mn, Zn, Cu, Cd, Ca, Mg, and Na) did vary significantly from one litter fraction to another ($p < 0.05$). Some trends in the composition of litter along both transects were visible. Two clear groups of elements were present in pine needles (the dominant fraction in litter): Zn, K, and Na were at significantly lower levels along the transect of continentality; and Mg, Mn, Cu and Cd occurred at lower levels along the Silesian transect. The remaining three elements (Pb, Fe, and Ca) did not show clear trends.

The three fractions of litter (needles, twigs, and cones) did not show significant differences in the contents of the elements Pb and K ($p > 0.05$). Contents of all the remaining elements were variable.

It is not possible to determine whether the different concentrations of metals in the different fractions of litter resulted from the accumulation of elements within tissues or on the surfaces of dead fragments of the plants forming it; litter was not washed prior to analysis, so both possibilities remain. Therefore, we focused on only the existence of different accumulations of metals by the three litter fractions.

The Pollution of Soils

Indices for the accumulation of eight metals in the organic-mineral horizon-A of soils were determined from all 15 study sites (*fig. 1*). In western Poland and Silesia, manganese, nickel, and copper occurred in concentrations exceeding those in the parent rock (about six times), while Fe, Zn, and Cr did not. The lowest concentrations of the six metals were observed in eastern Poland. Lead accumulated more intensively in soil than the other metals (*fig.1*). Lead concentrations may be 100 to 200 times higher in soil than in the substratum. The pollution of the environment by lead is primarily caused by vehicles using leaded gasoline. However, none of the sites in this study were in the vicinity of busy stretches of road, only within forested areas. The source of the lead in these soils must be from another local source.

The indices of accumulation defined for all of the studied soils confirmed the spatial distribution of heavy metals and trace in organic-mineral horizon; the highest values for the index were attained for Pb, Mn, and Cu (*fig. 1*).

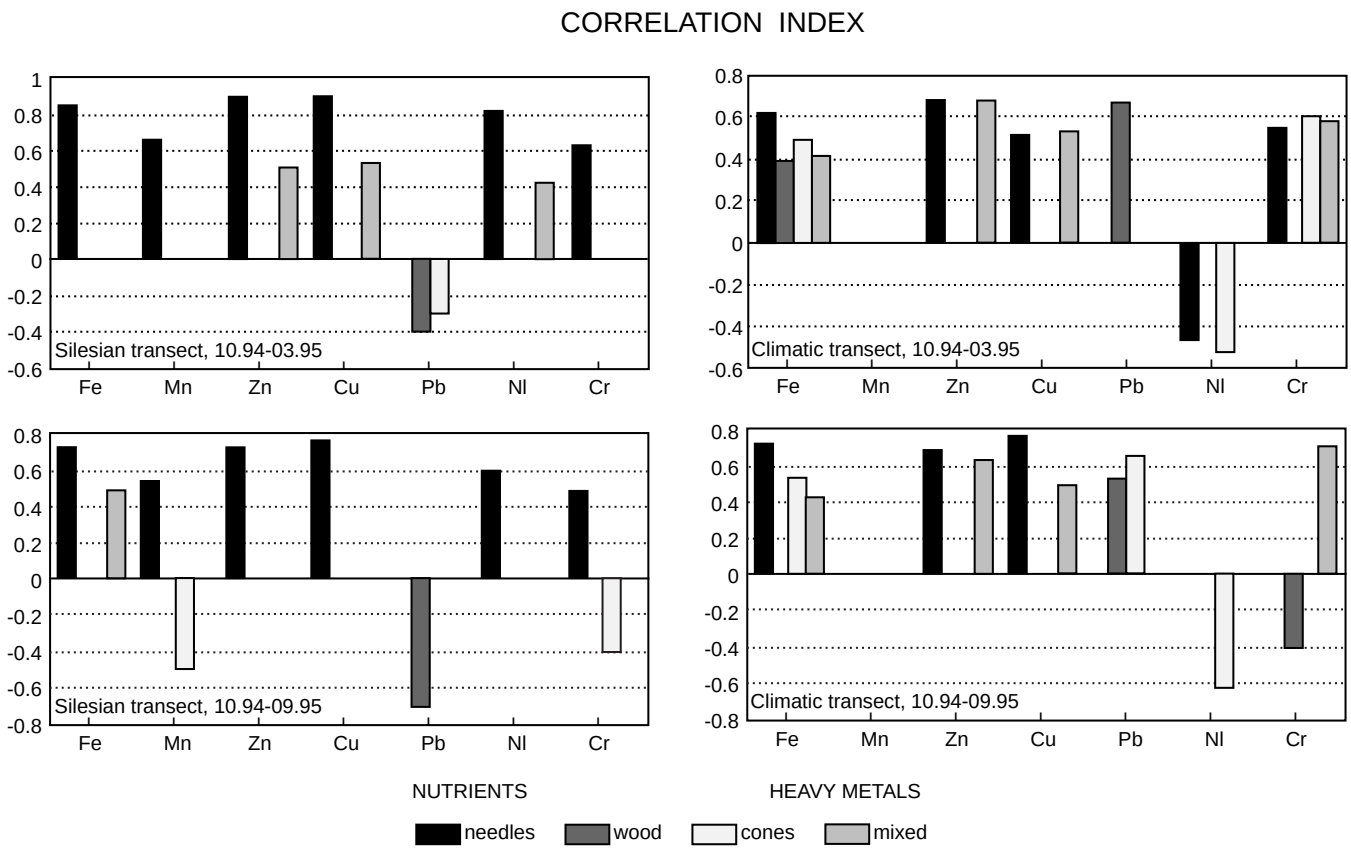

Figure 1 — Accumulation index for organic-mineral horizon-A of some chemical elements (*A*) with volume between 0-10 percent, and (*B*) with value over 10 percent (Breymeyer 1996). K= continental transect S= Silesian transect.

The Decomposition of Litter

The annual rate of decomposition of litter amounts to just over 30 percent and is higher in the first half of the exposition year (*fig. 2*). The mean annual rates of decomposition for the two transects were not statistically different. The course of these means over time showed a 4 percent difference between the transects (with more rapid decomposition on the climatic transect) after half a year. This difference had, however, disappeared after 12 months. The sites along the Silesian transect showed greater differences in the course of decomposition.

Analysis of the rates of decomposition of the different fractions of the litter revealed that the rate of breakdown of the woody fractions (cones and twigs) was clearly lower than the mixed litter, especially needles (*fig. 2*). After 12 months needle litter mass decreased more than 30 percent, compared to about 20 percent of mixed litter and 10 percent of the mass loss of cones and twigs.

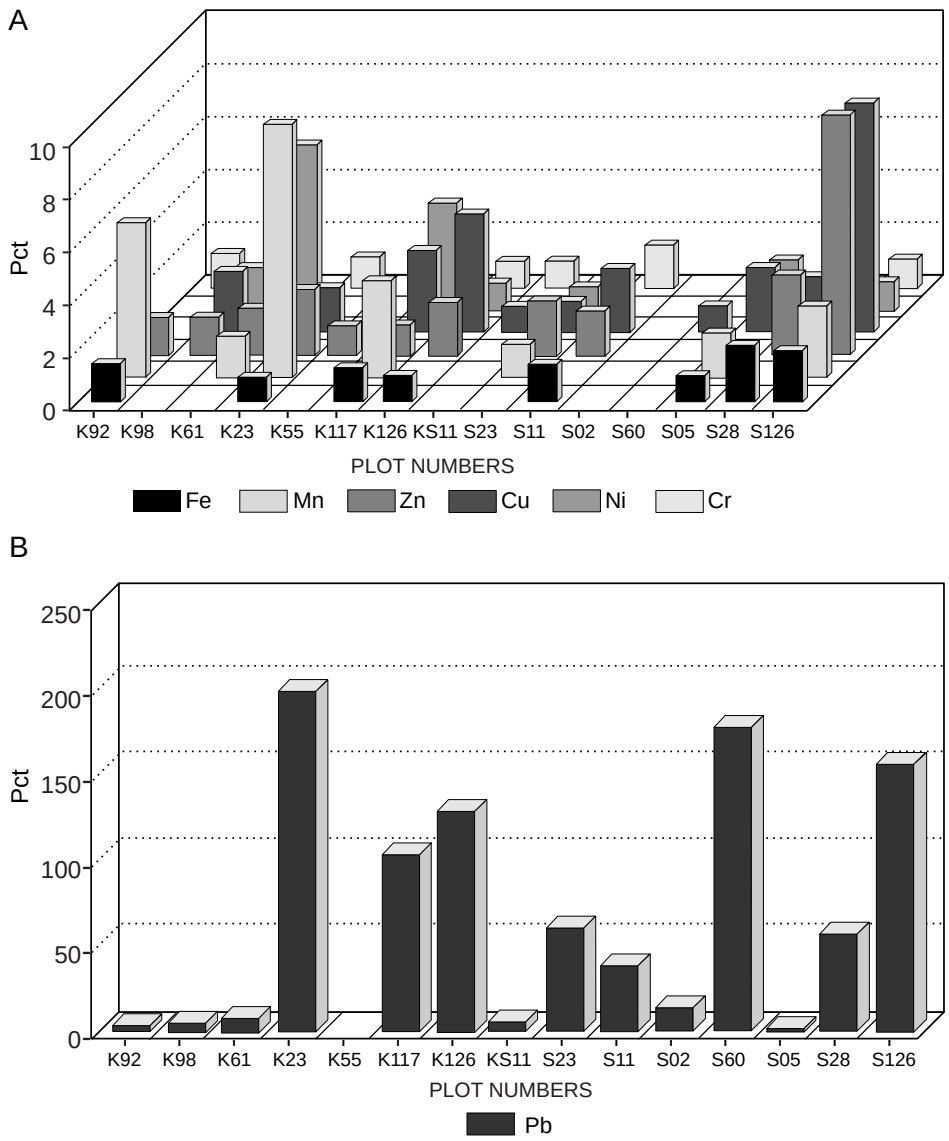

Figure 2 — Decomposition rates during the incubation year. The incubation year always begins in October when previous- year litter-fall packed into nylon-net bags is exposed in the stands; the bags are collected after 6 months and 12 months. (A) Average differences among litter fractions across all sites on continental transect. Solid line represents the mean values. (B) Differences among sites on Silesian transect.

Concentration of Metals in the Mineral-Organic Horizons of Soils and in Litter

Correlation index values between accumulated heavy metals in soils and the rates of litter decomposition were calculated (*figs. 3, 4*). The first consistent and notable result was the high prevalence of positive correlations between the breakdown of litter and the concentration of a metal in the soil. Among the calculations for material from the two transects for decomposition in winter and the whole year, a negative correlation was only noted for lead (*fig. 3*). Among the calculations for each transect separately, negative values for the coefficient were again noted for lead, as well as for nickel and chromium (*fig. 4*). In consequence, the rate of litter decomposition and the accumulation of a metal in the upper layer of the soil are most often correlated positively.

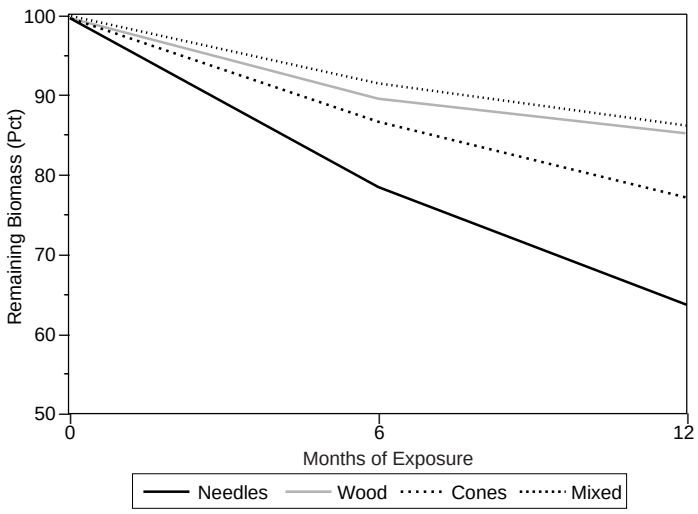

Figure 3 — Correlation index between heavy metals in organic-mineral soil horizon-A and litter decomposition rate (litter bags). Mean values for two transects.

Correlation index values between rates of litter decomposition and concentrations of the 8 elements were determined (*table 1*). A prevalence of positive correlations was again noted, but greater negative values were found than for soil correlations. Thus, pollution of the litter may hold back the process of decomposition much more often than the pollution of soils.

Table 1 *— Correlations between needle litter element concentrations and decomposition rate of different forms of organic deposition for the annual (1994-1995) exposure period on both the climatic and Silesian transects.*[1]

Element in litter	Mixed litter	Needles	Cones	Wood
Fe	[2] 0.71	[2] 0.83	0.06	- 0.38
Mn	[3] - 0.42	[3] 0.44	0.01	0.13
Zn	[2] - 0.66	[2] - 0.80	-0.08	0.4
Cu	- 0.25	- 0.34	-0.08	0.24
Pb	[2] 0.68	[2] 0.82	0.08	- 0.4
Cd	0.06	0.13	0.07	- 0.14
Ni	[2] - 0.72	[2] - 0.89	-0.08	0.36
Cr	[2] - 0.51x	[2] - 0.42x	[2] - 0.56x	[2] - 0.63

[1] A $\sin^{-1}(X^{1/2})$ transformation was performed before analyses; x = only winter (1995) exposure period.

[2] -p<0.001.

[3] -p<0.05.

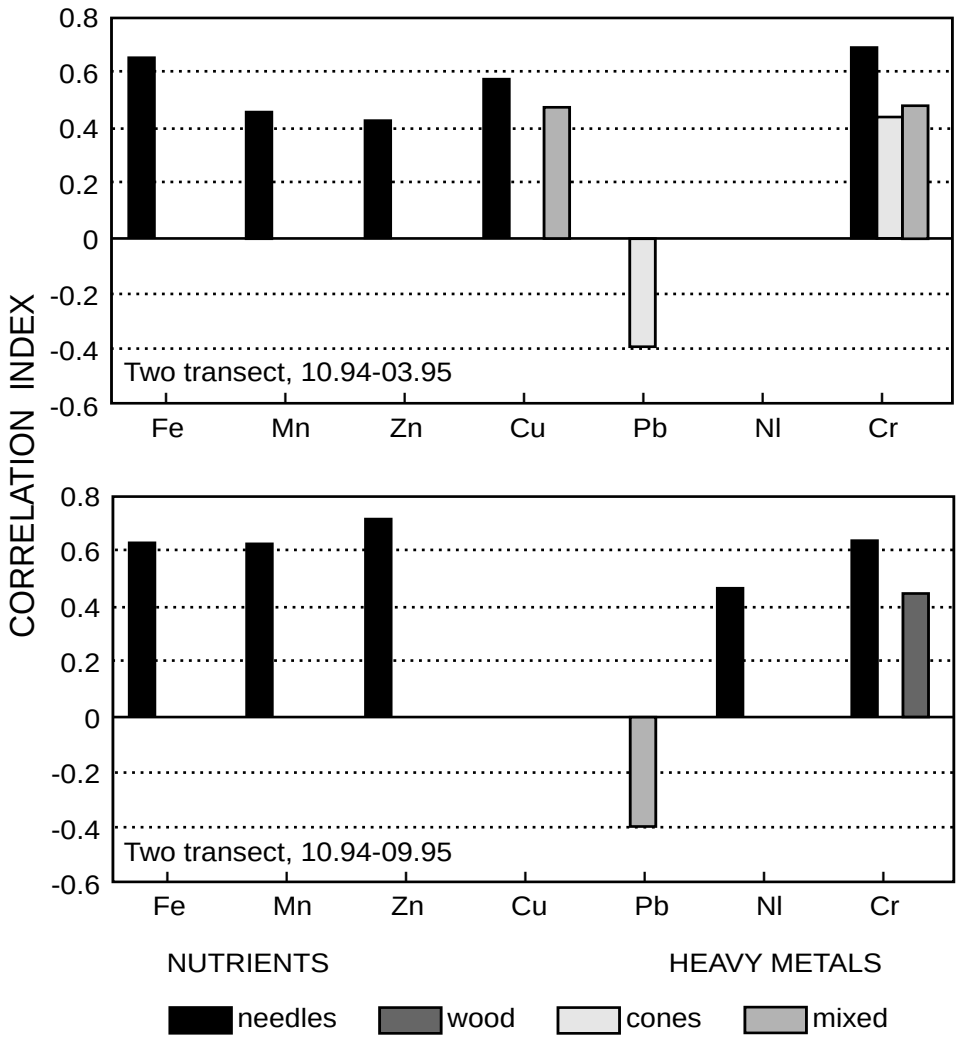

Figure 4 — Correlation index between nutrients and heavy metals in organic-mineral soil horizon-A and litter decomposition rate (litter bags) in the Silesian and climatic transect for 1994-1995 study.

These results indicate that a higher concentration of some of the heavy metals in the studied organic and mineral-organic soil horizons of poor coniferous forest habitats positively influence the transformation of organic matter.

Conclusions

The study found that decomposition of pine needles is similar along the climatic and Silesian transects. The mean rate is higher along the climatic transect in the first 6 months of the year, but mean rates for the transects do not differ significantly after 12 months. The different fractions of litter decompose at different rates: the slowest breakdown characterizes lignified parts (i.e., cones and twigs) resulting in statistically significant differences (*fig. 2*).

The accumulation of heavy metals in the upper layer of the soil showed that at the majority of sites, only lead occurs in amounts several tens or even several hundreds of times greater than in the parent rock (*fig. 1*). High concentrations of heavy metals in litter are most frequent in Silesia, and in the western sites along the climatic transect (i.e., in the western-border area of the country). The highest correlations between heavy metal contents in litter were found for needle litter decomposition and Fe (corr. coeff. 0.83), Zn (corr. coeff. 0.80), Pb (corr. coeff. 0.82), Ni (corr. coeff. 0.89), as well as for mixed litter decomposition and Fe (corr. coeff. 0.71), Zn (0.66), Pb (0.68), and Ni (corr. coeff. 0.72). There were not significant correlations

between the rate of decomposition of wood or cones and the content of metals in litter, except for Cr (*table 1*). The highest correlation between heavy metals content in organic-mineral horizon of soils and needle decomposition was determined for Cu (corr. coeff. 0.901), Zn (corr. coeff. 0.901), Ni (corr. coeff. 0.834), Fe (corr. coeff. 0.850) (*fig. 4*).

In some cases the correlations between decomposition rate and metal contents were positive. Comparing decomposition rates with the degree of litter pollution showed that in the case of needle decomposition for six significant correlations, two were positive; in the case of mixed litter decomposition for five significant correlations, two were positive. In both cases decomposition reacted positively to the presence of Fe and Pb. Decomposition rates compared with degree of soil pollution showed that in the case of needle decomposition for five significant correlations, all were positive; in the case of mixed litter decomposition, only one significant correlation was positive.

Thus, these results suggest that the pine forest ecosystems can react positively to the addition of pollution elements to the substratum and litter by enhancing the cycling of matter in the ecosystem as measured by the rate of litter decomposition.

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Geographic Information System (GIS) Analysis of Ecosystem Response to Industrial Pollution in the Niepolomice Forest in Southern Poland[1]

January Weiner,[2] Stefan Fredro-Boniecki,[2] David Reed,[2] Ann Maclean,[3] Marshall Strong,[3] and Michael Hyslop[3]

Abstract

The Niepolomice Forest is located near the city of Krakow in southern Poland. Since the erection of large iron works in the 1950's, the forest has suffered from heavy pollution with SO_2 and industrial dusts containing heavy metals. During the past 10 years, the ecology of the Niepolomice Forest has been intensively studied and the impact of industrial pollution has been significantly reduced. With the advent of modern computer intensive techniques, data gathered in the past have been reanalyzed with respect to the spatial and temporal variations of the forest ecosystem response to industrial pollution. In addition, the effects of natural conditions (soil, vegetation) and industrial pollution (heavy metals, sulfur dioxide) upon the pine stands (tree volume increment, crown injuries) in the Niepolomice Forest were studied by using a geographic information system. Procedures of statistical analysis involving bootstrapping were developed. Results suggest the apparent fertilization of forest stands as a result of industrial pollution—an effect not revealed in former studies.

Introduction

The Niepolomice Forest, an ancient forest complex situated close to the urban and industrial area of Kraków in southern Poland, may be regarded as an object of a large-scale ecological experiment. In the 1950's, the forest was suddenly exposed to enormous amounts of industrial pollution that increased steadily until the end of the 1980's and then decreased (*fig. 1*). This pattern was caused by the extensive development of the metallurgic industry and subsequent production limitation and technological improvements—a pattern clearly reflecting the political changes in the country. These changes provide a unique opportunity to study the response of forest ecosystems to such manipulations.

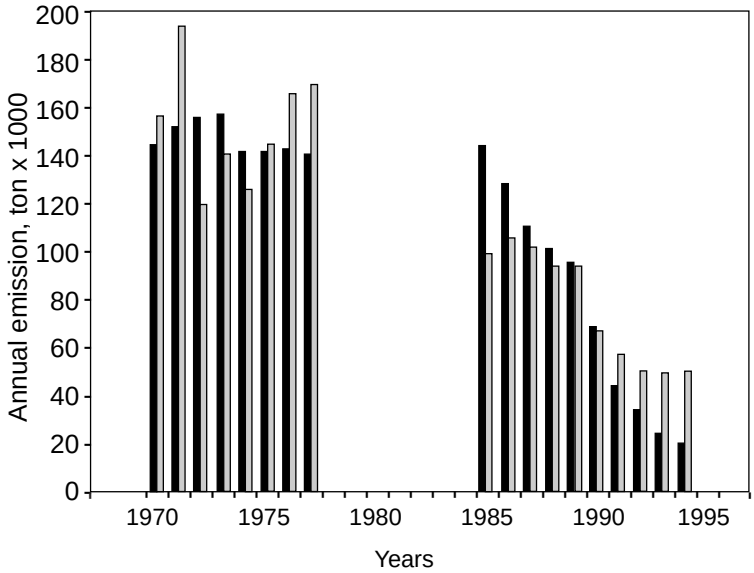

Figure 1 — Changes in the total emissions of industrial dusts (filled bars) and sulfur dioxide (open bars) in the Krakow agglomeration (Turzanski and Wetrz 1995). Note the distinct decrease of pollution during the last 10 years and an increased proportion of dust and SO_2 emissions in the last 5 years which may lead to increased acidification.

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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The Niepolomice Forest is a relatively large forest complex (about 110 km^2) which includes a variety of soils and vegetation types. For many years the forest ecosystems of Niepolomice were the subjects of research for many scientific institutions in Kraków. This activity resulted in hundreds of publications (Banasik 1978), including two synthesizing volumes (Grodzinski and others 1984, Kleczkowski 1981). However, the vast body of data could not be fully exploited. With the advent of efficient computer-intensive methods, such as geographic information systems (GIS), a more advanced analysis of the spatial and temporal variation can be undertaken and an explanation of the emerging patterns can be attempted. However, the data have not been collected expressly for the purpose of such analysis and therefore a number of methodological problems must be solved.

This paper reports a study that reanalyzes the available data on Niepolomice Forest ecosystems by using GIS to reveal possible spatial correlations between environmental conditions, pollution levels, and indices of tree stand quality.

Methods and Materials

As a basis for the GIS database, published topographic and thematic maps of Niepolomice Forest area were used: 1:25,000 topographic map; 1:20,000 forestry compartment map; 1:37,500 potential vegetation map (Gruszczyk 1981a); 1:37,500 present vegetation community (Cwikowa and Lesinski 1981); 1:37,500 forest soils map (Gruszczyk 1981b); 1:37,500 site type map (Gruszczyk 1981a); and 1:37,500 overstory stand type map (Maczynski 1981). The GIS (ARC Info) methods were developed at Michigan Technological University. After scanning and editing, the maps were registered to Universal Transverse Mercator coordinates using numerous global positioning system (GPS) readings made in the field (Weiner and others 1996). The database was converted to IDRISI 4.0 format (Eastman 1992)[4] and subsequent data input and analysis was performed at the Jagiellonian University in Krakow.

The basic information on the Niepolomice Forest is contained in the GIS layers such as soils (*fig. 2*) or forest stands (*fig. 3*), which are the electronic representations of previously published maps. New maps were produced based on published quantitative data. The sources available usually contain precise numerical information about the variable of interest, but the spatial coordinates for each sampling location are quite general. In most cases only the number of a forest compartment or subcompartment is provided and no better accuracy can be derived after the passage of 20 or more years since the field research was done. The forest subcompartments in Niepolomice can be of various sizes and shapes, sometimes not larger than 100 m^2, but a standard forest compartment is 400 by 800 m in size. Moreover, sampling for different variables was performed in different locations, although the sampling intensities were quite dense. Thus, a direct use of the data for a multivariate analysis is impossible because one cannot order the field data in sets of exactly corresponding spatial variates. An approximate analysis can be made by comparison of maps produced by the spatial interpolation of point data. This can be done only by presuming that the variables studied are continuous (forming a spatial gradient), at least locally. It was assumed that this was the condition in the case of the variables used.

Most of the data came from the larger, southern part of the Niepolomice Forest (mostly pine stands; *fig. 3*). Data for industrial pollution and ecosystem response were obtained from the index of tree crown injuries (Grabowski 1981). This index (DAM) combines various estimates of damage to pine needles and malformations of twigs and branches, most probably caused by SO_2 pollution (Grabowski 1981). Spatial interpolation of 86 data points resulted in a map (*fig. 4*).

Data were also obtained from the index of heavy metal accumulation. The content of six heavy metals (Cd, Pb, Zn, Cu, Fe, Mn) was measured in mosses at 15 locations in 1975 and in 1992 (Godzik and Szarek 1993). From these point data

[4] Unpublished data on file at Department of Forest Mensuration, Agricultural Academy, Krakow, Poland.

Figure 2 — The soils of the Niepolomice Forest (an IDRISI map): 1--proper brown (mesic) soils (5.1 percent); 2--podsolic and cryptopodzolic soils (13.0 percent); 3--brown podsolized soils (8.1 percent); 4--moist soils, black earths (23.7 percent); and 5--seasonal gleyish soils (50.1 percent) (Gruszczyk 1981).

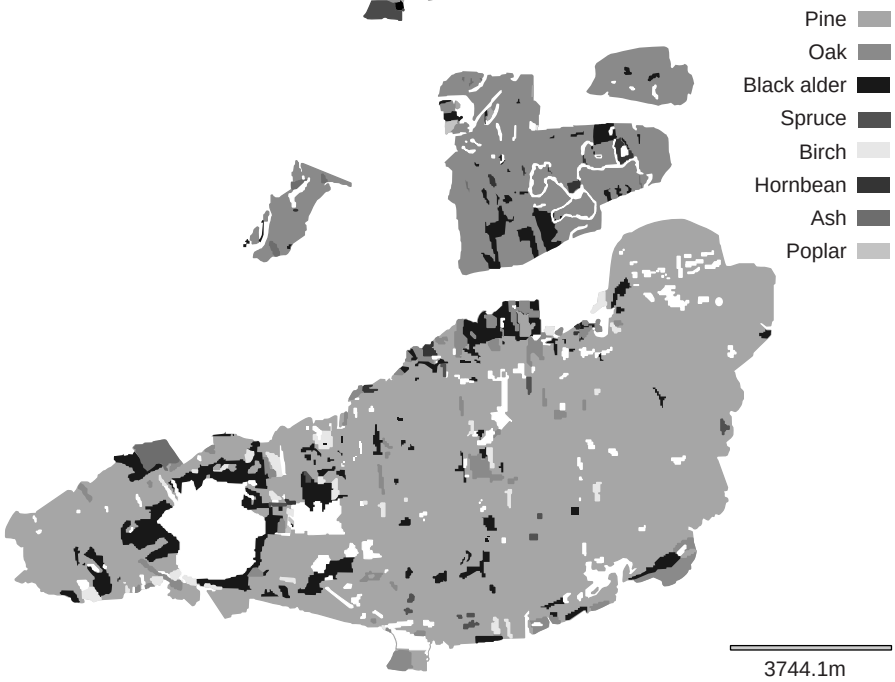

Figure 3 — Forest stands classified by dominant species (IDRISI map; Maczynski 1981).

interpolated maps have been produced. The contents of all metals are strongly intercorrelated; the first principal component (variable PC1) may constitute an index of contamination with industrial dusts (*fig. 5*).

Indices of tree volume increment were also used (Rieger and others 1987, 1989).[5] A detailed dendrometric survey of pine stands was carried out at 114 locations of the southern forest complex in 1981. The plots were located in younger (A: 20-45 years) and older (B: 60-130 years) stands. The data set, combined from published and unpublished information, includes: tree age, height, diameter at breast height (dbh.), and a numerical index of diameter increment dynamics during the last 5 years preceding the study, as related to the previous 5-year period (variables WZDA and WZDB for younger and older age classes, respectively; *fig. 6*). But, the original data on tree ring measurements were not accessible.

Another index was calculated from the data as a residual of the regression line of tree volume on age (*fig. 7*). The tree volume was calculated from height and dbh. according to allometric formulas and empirical parameters given for Scots pine

[5] Unpublished data on file at Department of Forest Mensuration, Agricultural Academy, Krakow, Poland.

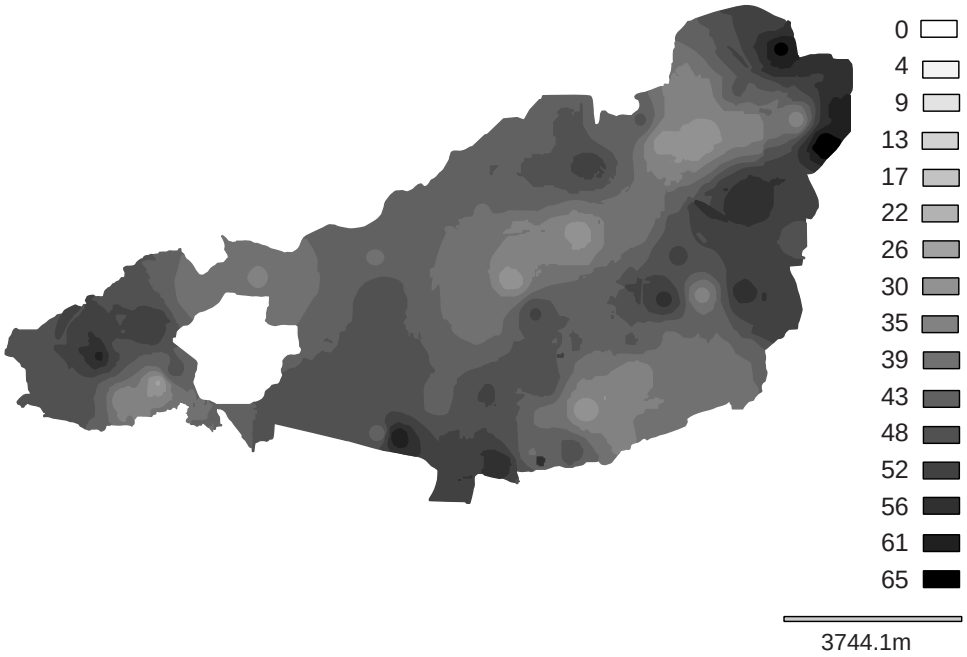

Figure 4 — Index of tree crown injuries in arbitrary units (Grabowski 1981).

Figure 5 — First principal component (PC1) of the contents of six heavy metals in mosses in 1975 (Godzik and Szarek 1993).

(Sulinski 1993). The residuals make up an index of cumulative volume increment during the whole life of a tree as related to the average tendency in the whole forest (variables TVRA and TVRB).

The first index should represent the recent changes in tree condition (e.g., related to changing pollution), while the second one should depend more on the natural stand quality. The map of both indices was obtained by a spatial interpolation of point data for younger (*A*) and older (*B*) age categories separately (*figs. 8, 9*).

Analysis and Discussion

Characteristic patterns of the spatial distribution of environmental and forest response variables are apparent according to the GIS maps. The areas most exposed

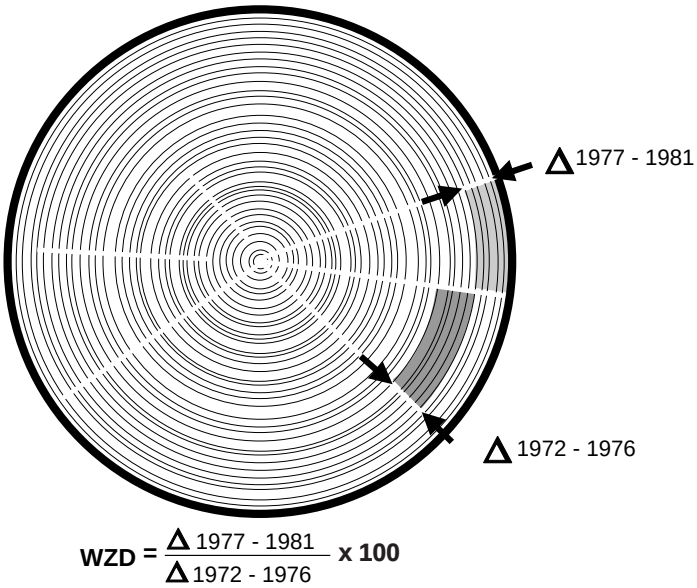

Figure 6 — Index of tree ring diameter dynamics, WZD (Rieger and others 1987, 1989).

Figure 7 — Regression of pine tree volume on age in the southern part of the Niepolomice Forest (double logarithmic plot).[6] Residuals for a given age represent a cumulative tree volume increment index (TVR).

to heavy metal pollution demonstrated an accelerated tree volume increment (*figs. 5, 6*). We did not find visible association between the index of heavy metal contamination and the injuries of tree crowns (*figs. 5, 6*).

The variables represented on maps by values obtained from spatial interpolation cannot be directly subject to a statistical analysis because they have extremely high autocorrelations (Moran's I in the range of 0.95 to 0.99 for different variables studied; Eastman 1992). The occurrence of autocorrelation is inherent in empirical distributions of variables of this kind (Legendre 1993) and an additional autocorrelation component is introduced by the procedure of spatial interpolation. This may cause an overestimation of significance of test statistics (Legendre 1993). To minimize this effect a randomization ("bootstrap") procedure was used (Noreen 1989). The variables of interest were sampled from a limited set of randomly selected point locations. Simulations revealed that with a decrease of the number of sampling points (from the original total number of pixels of about 870,000), the coefficient of autocorrelation (Moran's I) steadily decreases to a value

[6] Rieger and others 1987, Sulinski 1993).

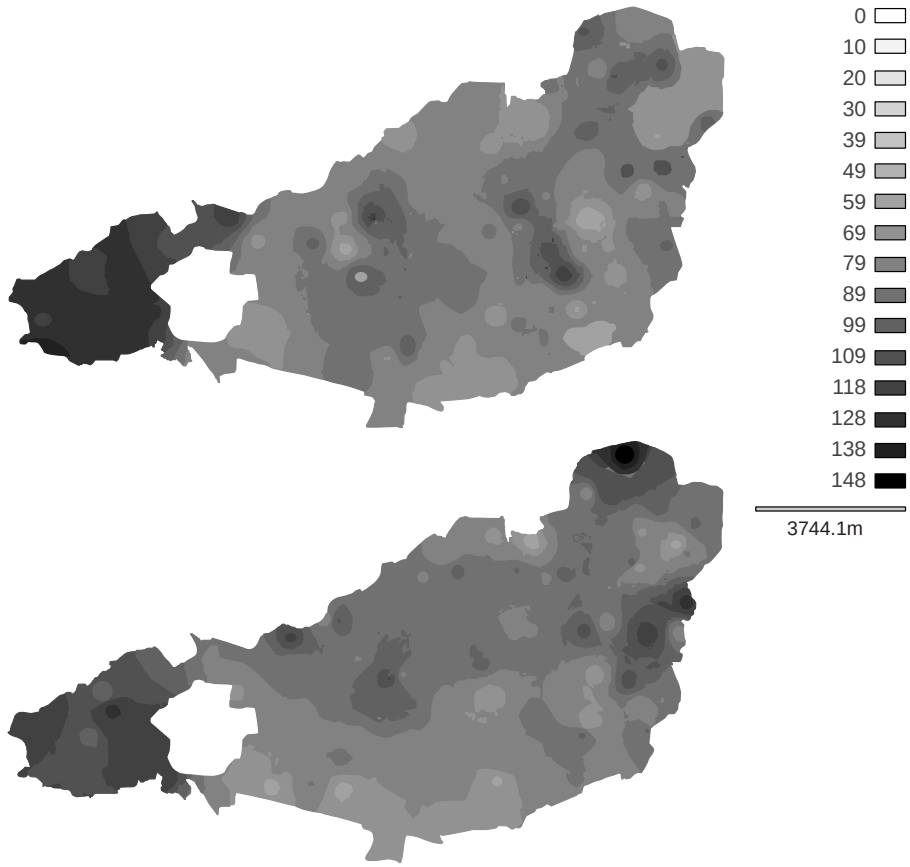

Figure 8 — Cumulative index of tree volume increment (TVR) in the southern part of the Niepolomice Forest (arbitrary units): (*A*) - pine stands 20-45 years old; and (*B*) - pine stands 60-130 years old.

of about 0.7 at 1,000 points and then its variation increases (*fig. 10*). Accordingly, the sample size was set at about 1,500 points. The statistical analysis of association between variables or analysis of variance by categories was performed on these samples. The procedure was repeated (with different sets of random point locations) and average statistics were calculated.

The associations between single variables were examined by using Spearman rank correlation coefficient r_s. Correlation coefficients were averaged (± standard deviation, range) for 12 randomly sampled sets of data (*table 1*). A statistically significant positive correlation ($r_s > 0$, $p < 0.05$) was revealed between the tree growth

Table 1 *— Average (± standard deviation, range) Spearman rank correlation coefficients for dendrometric variables (WZDA, WZDB, TVRA, TVRB), dust pollution index (PC1), and pine crown injury index (DAM).*[1]

Item	TVRA	TVRB	WZDA	WZDB	DAM
PC1					
Average	-0.098	[2]0.320	[2]0.492	[2]0.330	[2]0.071
Standard deviation	0.013	0.020	0.034	0.034	0.021
Minimum	0.115	0.294	0.444	0.297	0.035
Maximum	-0.073	0.348	0.544	0.427	0.113
DAM					
Average	[2]- 0.175	[2]- 0.247	[2]0.243	[2]0.331	
Standard deviation	0.257	0.023	0.069	0.067	
Minimum	-0.987	-0.278	0.088	0.165	
Maximum	-0.065	-0.208	0.311	0.401	

[1] TVRA = tree volume index for young stands (20-45 yrs.). TVRB = tree volume index for older stands (60-130 yrs.). WZDA = diameter growth index for young stands (20-45 yrs.). WXDB = diameter growth index for older stands (60-130 yrs.). PC1 = first principal component of the heavy metal concentrations. DAM = crown injury index.

[2] Average correlations that are different from zero ($p < 0.05$).

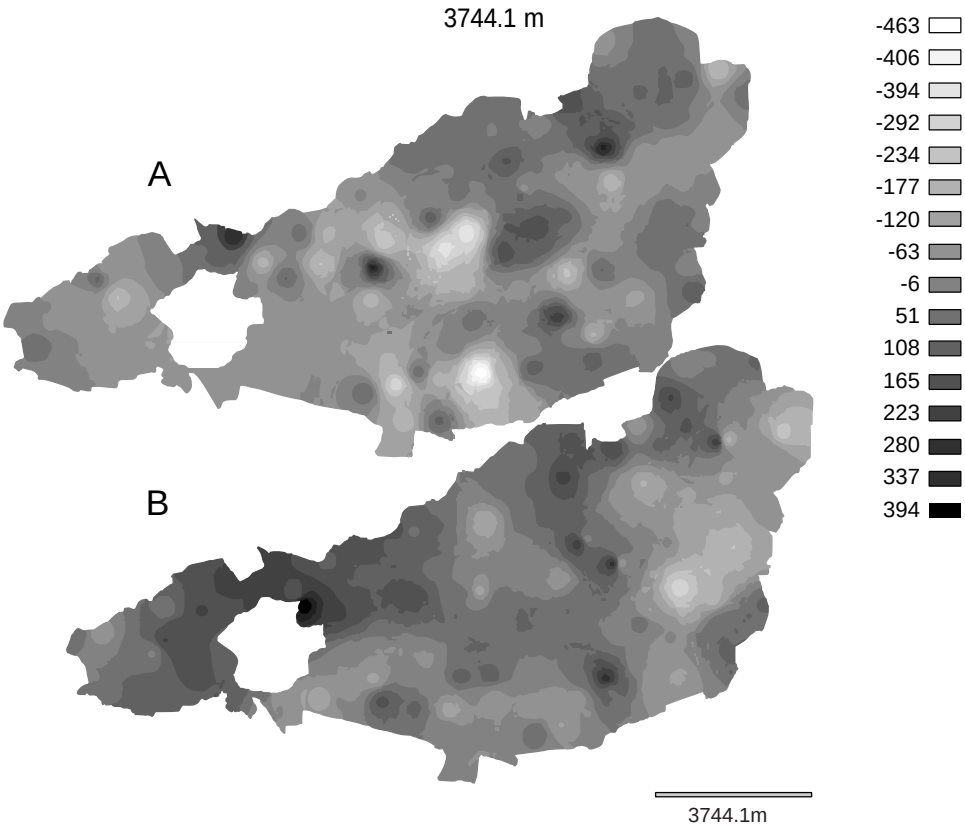

Figure 9 — Index of tree ring diameter dynamics (WZD, percent) in the southern part of Niepolomice Forest: WZD = 100 percent means an equal rate of tree ring diameter increment during 5-year periods 1977-1981 and 1972-1976; WZD > 100 percent denote acceleration of growth, and WZD < 100 percent is growth retardation. (*A*) - pine stands 20-45 years old; (*B*) - pine stands 60-130 years old.

volume increment indices (WZDA, WZDB) and the first principal component (PC1) of the heavy metal concentrations (i.e., the tree volume increments) (independent of age class) were increased in the most polluted areas. The total volume of older trees (TVRB) was also positively correlated with dust fall (*table 1*).

The tree crown injuries (DAM) were positively correlated with the indices of tree growth dynamics in both age classes (WZDA, WZDB), but negatively correlated with the total tree volume (TVRA, TVRB; *table 1*). Thus, a malformation of tree crowns was associated with an increased tree volume increment in the years immediately preceding the field studies, but the trees with most damaged crowns were also the smallest in the sample.

Five general types of soils have been distinguished in the southern part of the forest covering different proportions of the area (*fig. 2*). Soil type exerts a significant effect upon the characteristics of tree stands. The index of increase in both age classes (WZDA, WZDB) and the index of tree volume in older class (TVRB) is significantly higher in brown (mesic) soils. The total volume (TVRA, TVRB) tends to be the smallest on dry (podzolic) soils ($p < 0.01$; Kruskal-Wallis). Tree crown damage is highest on brown mesic and in dry podsolic soils.

In the southern part of the forest the brown (mesic) soils are concentrated at the western edge in the most polluted area. To discriminate the effects of soil type and dust pollution upon the dendrometric variables, the same statistical analysis was performed within the soil classes. Two distinct patterns persist after the effect of soil type is removed: the positive correlation of the index of dust pollution (PC1) with the indices of growth acceleration (WZDA, WZDB) and a negative correlation of tree crown injuries (DAM) with tree volume, particularly in older age classes (TVRB). In other words, the smaller the tree volume, the greater the damage. This last phenomenon may indicate a synergistic effect of poor habitat and gaseous pollution. In the older age class the trees were exposed to pollution during the recent one-third to one-fifth of their lifetime only. Thus, the relatively small volume must

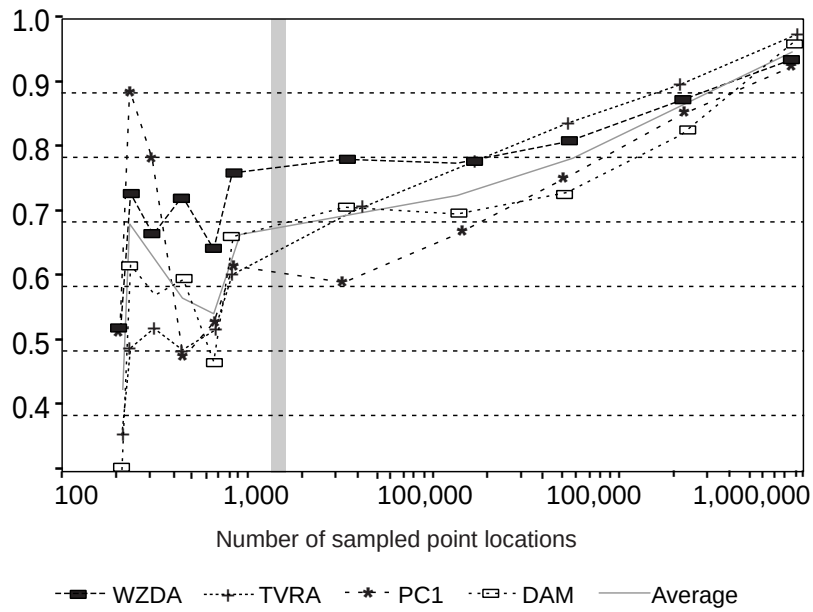

Figure 10 — Changes in the coefficient of autocorrelation (Moran's I) with decreasing number of points sampled from spatially interpolated data sets. Grey bar indicates the number of point locations used in analysis.

have been the effect of habitat conditions while the relatively strong malformation of crowns in the weakest individuals is a secondary phenomenon. No correlation was found between the tree crown injuries (DAM) and heavy metal contamination (PC1; *table 1*).

A simultaneous effect of two variables (PC1 and DAM) upon the dendrometric characteristics of pine trees was analyzed by using coefficient of determination (R^2) for a multiple regression:

$$Y = a_0 + a_1 \text{ PC1} + a_2 \text{ DAM}$$

in which PC1 is the dust contamination index, DAM is the pine crown injury index, and a_0, a_1, a_2 are parameters. The R^2 values were calculated and showed an indication of a positive (+) or a negative (-) effect of the independent variables (the numerical values of parameters are omitted) (*table 2*). The two independent

Table 2 — *Average (± standard deviation, range) coefficients of determination (R^2) for a multiple regression for dendrometric variables (WZDA, WZDB, TVRA, TVRB), dust pollution index (PC1), and pine crown injury index (DAM).*[1]

Item	TVRA	TVRB	WZDA	WZDB
R^2				
Average	0.018	[2]0.176	[2]0.290	[2]0.208
Standard deviation	0.0065	0.0203	0.0241	0.0241
Minimum	0.009	0.137	0.257	0.178
Maximum	0.028	0.211	0.346	0.250
Sign of a_1 (effect of PC1)	-	+	+	+
Sign of a_2 (effect of DAM)	-	-	+	+

[1] TVRA = tree volume index for young stands (20-45 yrs.). TVRB = tree volume index for older stands (60-130 yrs.). WZDA = diameter growth index for young stands (20-45 yrs.). WZDB = diameter growth index for older stands (60-130 yrs.). PC1 = first principal component of the heavy metal concentrations. DAM = tree crown damage index.

[2] Average R^2 values differing from ($p < 0.05$).

variables account for 18 to 29 percent of variation in dendrometric indices (except for TVRA, (i.e., total volume of trees in younger stands), for which the effect was not significant). These results agree with the Spearman rank correlation patterns for single variables.

The most probable explanation of this apparent paradox of growth acceleration of trees exposed to highest pollution lays in the high nutritional value of industrial dusts for plants. These pollutants contain a great load of Ca, Mg, N, P, and other nutrients (Grodzinski and others 1984) as well as heavy metals. Moreover, the dust is basic and therefore it can compensate for low pH caused by SO^2 pollution.

These patterns describe the situation in the Niepolomice Forest in the late 1970's. Since, then, the level of pollution has changed dramatically (*fig. 1*). A new field of study, GIS, repeated 20 years later, could help to confirm or to reject the hypothesized explanation of these effects.

Acknowledgments

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Chemical Composition of Needles and Cambial Activity of Stems of Scots Pine Trees Affected by Air Pollutants in Polish Forests[1]

Wojciech Dmuchowski,[2] Ewa U. Kurczynska,[2] Wieslaw Wloch[3]

Abstract

*The impact of environmental pollution is defined for the chemical composition of Scots pine (*Pinus sylvestris *L.) needles and cambial activity in the tree stems in Polish forests. The research investigated 20-year-old trees growing in two areas in significantly different levels of pollution. The highly polluted area was located near the Warsaw steelworks, while the area with relatively low level of pollution (control area) was in the Kampinos National Park, 10.5 km away from the steelworks. The research revealed significant differences in chemical composition of needles between the two areas. The increased content of almost all of the studied elements was observed in all of the typical pollutants such as sulfur, chlorine, and heavy metals; and in the nutrients such as nitrogen, potassium, magnesium, calcium, and sodium. Differences were also observed in relations between particular elements and accumulation of nutrient elements (ANE). The changes in chemical composition of needles affected by environmental pollution were accompanied by a reduction of cambial activity. This reduction was not expressed as a decrease in the number of cells in cambial zone, but as a decreased number of cells on the phloem side, both at the tree-top level and at breast height. These differences were particularly visible in the second half of the vegetative season.*

Introduction

Air pollution is considered one of the main causes of forest decline in central Europe (ECE 1984, Molski and Dmuchowski 1986, Nihlgard 1985, Schutt and Cowling 1985). According to the results of national forest health monitoring, more than 50 percent of trees in Poland represent the highest defoliation classes (Malachowska and Wawrzoniak 1995). Among the European countries, only the forest condition in the Czech Republic is worse than in Poland (United Nations Economic Commission for Europe 1997).

The negative influence of air pollutants on tree growth has been well established (Smith 1990). Air pollution affects various physiological and biochemical processes (Biggs and Davis 1981, Burton and Morgan 1983, Bytnerowicz and Grulke 1993), as well as morphology and anatomy of shoot and root system of trees (Heale and Ormrod 1982, Kozlowski and others 1991, Kurczynska and others 1996, Smith and Davies 1978). Air pollutants reduce shoot growth of seedlings, and height growth of older trees may be reduced as well (Kozlowski and others 1991). The leaf area of trees exposed to air pollutants may be reduced because of inhibition of leaf formation, arrested leaf expansion, and accelerated leaf abscission (Kozlowski and others 1991). Air pollutants also reduce diameter growth of seedlings and mature trees (Thompson 1981). However, influence of pollution on cambial activity and differentiation of its derivatives, processes which regulate tree growth and development (Zajaczkowski and Wodzicki 1978), have not yet been adequately investigated.

Recently published results showed that in the polluted environment physiological polarity of pine stems was affected. This resulted in changed polar transport of auxin at the cellular level of the cambial zone (Wodzicki and others 1990a). Polar transport of auxin, in turn, is responsible for regulation of cambial activity and differentiation of its derivatives (Larson 1994).

This study investigated the effects of industrial/urban air pollution on chemical composition of needles as well as cambial activity and differentiation of its derivatives, especially on the phloem side, of Scots pine (*Pinus sylvestris* L.) trees in the polluted and control forest sites near Warsaw, Poland.

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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Materials and Methods

Investigations were carried out at two sampling areas along an air pollution gradient (Molski and Dmuchowski 1990) during one growing season in 1992. These sampling areas were the Kampinos National Park (a control area located upwind, 10 km away from the Warsaw Steelworks) and a site in the vicinity of the Warsaw Steelworks (a polluted area located upwind, 0.2 km away from the Warsaw Steelworks). Sampling areas were established in relatively uniform and representative portions of the stands.

Three trees were selected from each sampling area for anatomical investigation, and six trees were selected for chemical analysis of needles (18-20-year-old; 8-9 m height) of Scots pine (*Pinus sylvestris* L.) with approximately the same diameter at breast height (30-32).

Foliar mineral composition of Scots pine (N, P, K, Mg, Ca, Mn, S, Cl, and heavy metals) was determined as an indicator of health condition and contamination of the experimental trees. The samples of needles were taken from the same six trees and at the same time as in the anatomical study. Needles from the previous years' growth from the second and third whorls were collected (Dmuchowski and Bytnerowicz 1995).

Concentration of chemical elements were determined by using the following analytical procedures: metals and phosphorus were assessed by spectrophotometry atomic absorption method (Perkin-Elmer 1990, Morton and Roberts 1991) on a Perkin-Elmer 1100 fitted with a Flows Analysis System (FIAS-2000) and Graphite Furnace (HGA-700).[4] Sulfur (total) was assessed by infrared absorption method by using LECO SC-132 Sulfur System (LECO Corporation 1987) and chloride by potentiometric titration using selective chloride ion electrode (LaCroix and others 1970).

Content of elements in needles was presented as the sum of the accumulation of nutrient elements (ANE). To calculate ANE, all parameters of the contents of elements in needles were converted into gram-equivalent values. The basis of the conversion was the atomic mass of elements and the valency of their ions (Ostrowska and others 1988).

For anatomical analysis, samples included current-year wood, cambium, and current-year phloem (once in the winter season of dormancy, and April 30, May 8, June 5, July 3 and 31, August 28, and September 25) from tree-top level and from breast height of each investigated tree. Samples were fixed, dehydrated, and embedded in Epon (Kurczynska and others 1996). Samples were cut into a series of cross sections (3 m thick) with an ultramicrotome. The sections were attached to microscope slides and stained with leucofuchsin (PAS-reaction) and toluidine blue, mounted in Euparal, and investigated under light microscope. To determine the number of wood, cambium, and phloem cells, 20 radial rows were counted on the cross section from each tree. Data were analyzed by Student's t-test ($p<0.05$) and compared between sampling areas.

The soil at the two sites is of the same origin and represents the same type podsolic (Dmuchowski and others [In press]). The differences between soil conditions are caused by over 30 years of pollutant emission from the Warsaw area. On both sites the percentage of exchangeable cations in the sorption complex in the organic level of soils was the same but differences were identified in the mineral levels of soils. In the soil from the Kampinos area hydrogen was the dominant cation, which is typical for coniferous forests, while in the soil from the Warsaw area, calcium was dominant. In the organic layer of the soil from the Warsaw area, 2 times more iron was measured, 30 times more zinc, 6 times more copper, 20 times more cadmium, and 3 times more chromium than in the Kampinos area.

[4] Mention of trade names or products is for information only and does not imply endorsement by the U.S. Department of Agriculture.

Results and Discussion

For both areas the chemical analysis of pine needles did not reveal a deficiency of any of 15 analyzed elements (*table 1*). Certain macroelements like magnesium, potassium, and phosphorus showed deficiencies in damaged trees (Huttl 1985, Rehfuess 1983, Tomlison 1987, Zotll and Huttl 1989).

The chemical composition of pine needles at the two locations was much different, particularly in regards to elements recognized as typical pollutants (*fig. 1*). For instance, 2-year-old needles from the Warsaw area contained eight times more lead than the needles from the Kampinos National Park. The levels of lead, zinc, and chromium defined as toxic for the Scots pine were exceeded. The sulfur content in plants in Central Europe reflects the impact of air pollution with sulfur dioxide. Both investigated areas differed considerably in this point. The high sulfur levels in the needles from the steelworks are evidence of a detrimental impact of air pollution with sulfur dioxide on the Scots pine (*fig. 2*). In the Kampinos National Park the impact of pollution was not so strong. At present there are no areas in Poland excluded from the impact of air pollution by sulfur dioxide.

Differences were also observed also in the contents of elements performing basic physiological functions (*fig. 1*). The needles from the Warsaw area contained more magnesium, calcium, potassium, and sodium, but less nitrogen and manganese than the needles from the Kampinos National Park. The level of phosphorus was the same.

Table 1 *— Concentration of macronutrients in Scots pine needles and normal and deficiency levels of these elements in plants.*

Element	Kampinos	Warsaw	Level normal	Level deficiency	Reference
N (percent)	1.32-1.66	1.83-1.44	1.3-1.6	<1.2-1.3	Wehrmann 1963; Gussone 1964, Fiedler and others 1969, cit. Smidt 1988.
P (percent)	0.11-0.18	0.12-0.18	0.11-0.17	<0.11	Gussone 1964, Fiedler and others 1969, cit. Smidt 1988.
K (percent)	0.31-0.50	0.44-0.66	0.35-0.66	<0.30	Zottl 1964; Heinzdorf 1967, cit. Smidt 1988.
Ca (percent)	0.15-0.33	0.32-0.58	0.23-0.50	<0.05-0.10	Gussone 1964, cit. Smidt 1988: Kraus 1965.
Mg (mg/kg)	753-910	1207-1518	500-1300	<500	Wehrmann 1963; Fiedler and others 1969, cit. Smidt 1988.
Mn (mg/kg)	538-916	76-139	175-810 [1](40-2000)	[1](20)	Wehrmann 1963; Fiedler and others 1969, Materna 1962; Bergman1982; Knabe 1985; Isermann 1986.

[1] *Picea abies*

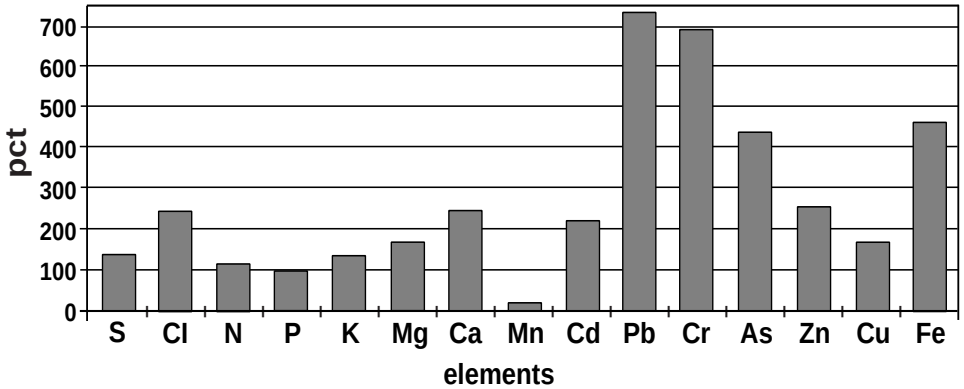

Figure 1 — Chemical composition of needles in the polluted area shown as percent of the concentrations in the control areas.

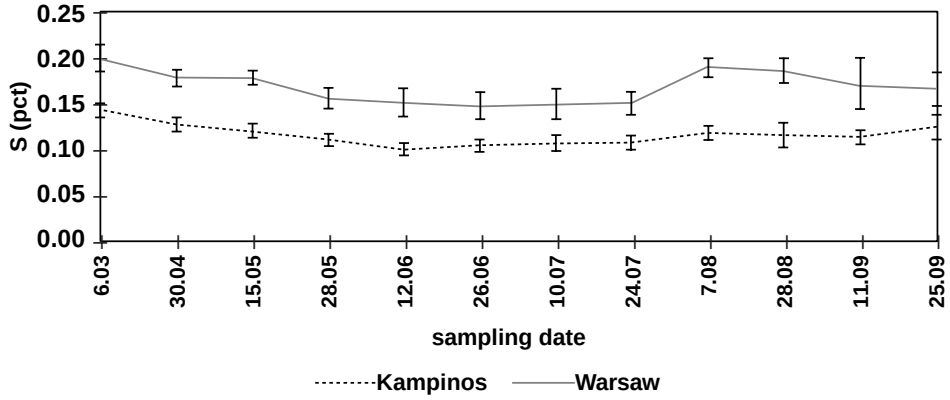

Figure 2 — Total sulfur content in previous-year needles.

In order to identify the presence of possible ionic equilibrium disturbances in the trees affected by environmental pollution, the ionic equilibrium indicators were calculated: ratio of calcium to magnesium (*fig. 3*) and ANE total of all sample elements (*fig. 4*). The calculation revealed certain differences, but these were not statistically significant. It seems that anatomical changes in stems of Scots pine in the Warsaw area are not caused by mineral deficiencies, but perhaps by toxic effects of environmental pollutants. Concentration of sulfur in needles was at toxic levels. Concentrations of heavy metals were always higher than normal levels in plants, but below toxic levels. Elevated concentrations of sulfur and heavy metals can cause negative effects on trees, especially under synergistic influences of other pollutants (*table 2*).

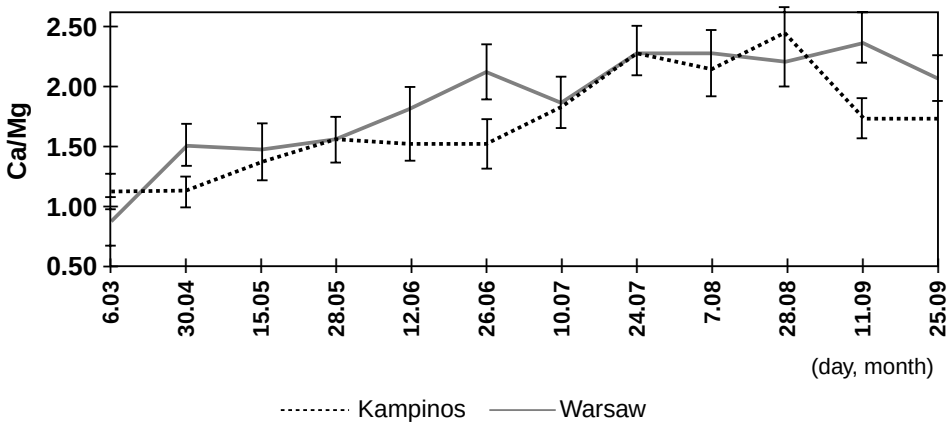

Figure 3 — Ratio Ca/Mg in previous-year needles.

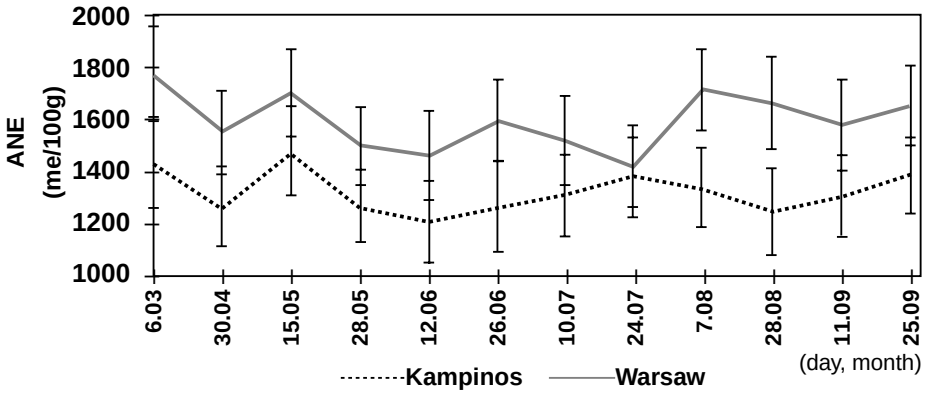

Figure 4 — Accumulation nutrients elements (ANE) in previous-year needles.

Table 2 *— Concentration of macronutrients in Scots pine needles and normal and toxic levels of these elements in plants.*

Element	Scots pine Kampinos	Scots pine Warsaw	Level normal	Level toxic	Species	Reference
S (percent)	0.10-0.14	0.15-0.20	0.06-0.09	>.12	*Pinus sylvestris*	Huttunen and others 1985; Dmuchowski and Bytnerowicz 1995.
Cd (mg/kg)	0.18-0.36	0.39-0.60	0.01-0.3		plants	Allen and others 1974;
			0.05-0.2	>5	plants	Kabata-Pendias and Pendias 1993;
				4.8	*Picea* sp.	Burton and Morgan 1983.
Pb (mg/kg)	1.9-2.4	11.9-17.6	<10	>30	plants	Kabata-Pendias and Pendias 1993;
				19	*Picea* sitch.	Burton and Morgan 1983.
As (mg/kg)	0.05-0.12	0.29-0.54		>1	plants	Macnicol and Becket 1985;
				>5	plants	Kabata-Pendias and Pendias 1993.
Cu (mg/kg)	3.3-5.1	5.8-8.9	2-20	>20	plants	Kabata-Pendias and Pendias 1993;
			2.5-25		plants	Hewitt and Smith 1974;
			25		leaved trees	Linzon and others 1979.
Zn (mg/kg)	31-42	68-104	10-100	>100	plants	Kabata-Pendias and Pendias 1993; Karweta 1976;
			226		*Picea* sitch.	Burton and Morgan 1983;
			250		leaved trees	Linzon and others 1979.
Cr (percent)	0.30-0.98	2.47-4.72	0.1-1.0	>1	plants	Kabata-Pendias and Pendias 1993
Cl	0.02-0.04	0.04-0.08	0.02-0.10		*Picea abies*	Keller 1972

During the growing season, cambial activity and the number of cells on the xylem and phloem side were higher at the tree-top level than at breast height, independent on the sampling area. We found no statistically significant differences in the number of cells in the cambial zone between trees grown in control and polluted areas (*fig. 5*), and in the number of xylem cells between trees grown in these two areas (*fig. 6*). The number of phloem cells was significantly different between trees grown in control and polluted area (*fig. 7*). The decrease in the number of phloem cells was especially visible in the second part of the growing season. It is possible that changes in the phloem are very important cause of tree decline in the polluted environment. For instance, in normally growing trees the maximal mass of functioning phloem appears in the second part of the growing season, when the most intensive transport of sugars to the stem and roots occurs. Decreasing the number of cells participating in that transport could lead to the disturbance in translocation of assimilates to different parts of trees and cause limited growth.

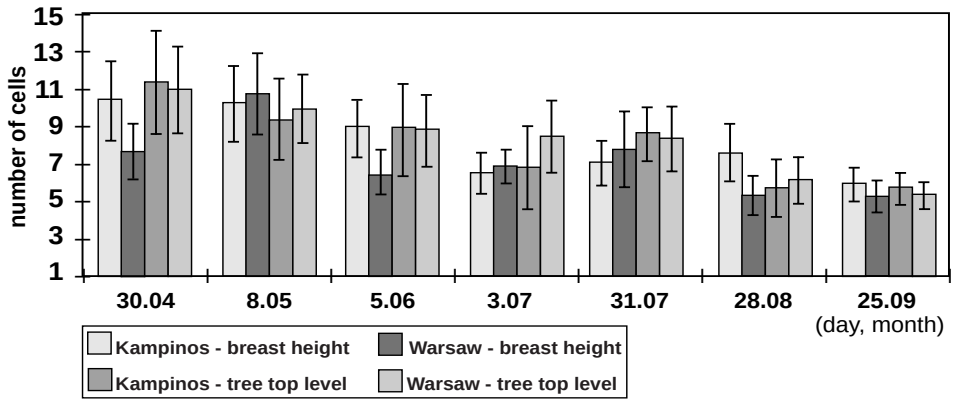

Figure 5 — Number of cells in the cambial zone during growing season. Each bar represents the average value (± SD) from three trees.

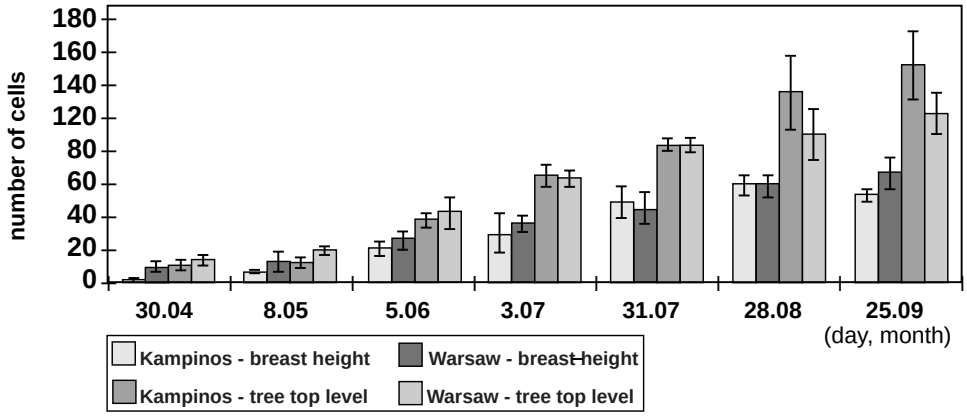

Figure 6 — Number of wood cells during growing season. Each bar represents the average value (± SD) from three trees.

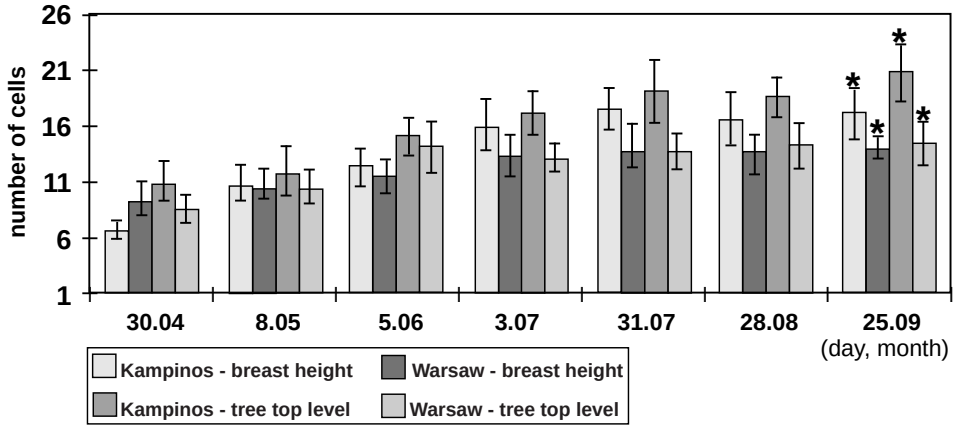

Figure 7 — Number of phloem cells during growing season. Each bar represents the average value (± SD) from three trees. Bars marked by * are significantly different in area category.

Conclusions

The Scots pine trees growing in the more polluted Warsaw Steelworks environment differed considerably from the trees growing in the relatively unpolluted Kampinos National Park environment. In the polluted environment, the needles contained much higher concentrations of elements recognized as typical pollutants: sulfur and heavy metals. And the levels of sulfur, lead, zinc, and chromium in the needles of trees from the Warsaw area can be defined as toxic for the Scots pine.

The differences in the content of elements performing basic physiological functions were of minor importance. No deficiency of any element was identified. The differences in chemical composition of needles were accompanied by differences in ionic equilibrium. However, these differences were not statistically significant.

Tree stem tissues such as cambium, phloem, and wood at the tree-top level were more sensitive to pollutants than tissues in older parts of trees (breast height). The limited production of phloem (which was statistically significant) and xylem cells of the trees growing in the polluted environment may have an essential impact on the disturbances in transport of water to tree-crowns and transport of assimilates from crowns to roots. This fact may cause the limitation of growth and development of the trees growing in the polluted environment and may lead to their mortality

The growth limitation of the trees from the Warsaw area seems to be caused to a larger extent by environmental pollution from sulfur, heavy metals, and other toxic agents (which were not examined, such as nitric oxide) rather than by the disturbances in ionic equilibrium in trees.

Acknowledgments

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Ectomycorrhizae of Young and Mature Scots Pine Trees in Industrial Regions in Poland[1]

Barbara Kieliszewska-Rokicka, Maria Rudawska, Tomasz Leski[2]

Abstract

*Ectomycorrhizae of Scots pine (*Pinus sylvestris *L.) trees grown in forests influenced by different levels of air pollutants were investigated. Total numbers of mycorrhizal root tips in the soil horizons and the frequency of mycorrhizal morphotypes were compared as indicators of ectomycorrhizal status. The studies were conducted in two comparable young pine plantations in western Poland and in two mature forest ecosystems in southern Poland differing in pollution level. Soil at the polluted young pine plantation (Lubon) had lower pH and increased availability of aluminum ions than the control site (Kornik). During 3 years of observations a lower number (50 percent) of mycorrhizal root tips per soil volume were found at the polluted site compared to the control site. The percentage of particular mycorrhizal morphotypes in the two young pine plantations was similar. The mature pine stands represented heavily-polluted (Niepolomice Forest) and moderately-polluted (Ratanica forest catchment) forest ecosystems. The total number of mycorrhizae in the Niepolomice Forest was lower in sites located closer to the urban-industrial area than at more distant plots. In the Ratanica Forest catchment the total number of mycorrhizal root tips varied between the sampling sites; however, on an average it was considerably higher than in the Niepolomice Forest. The mycorrhizal diversity at the heavily-polluted stand was reduced (four mycorrhizal morphotypes) as compared to the moderately-polluted site (eight mycorrhizal morphotypes). The results indicated that ectomycorrhizae of mature Scots pine trees were more affected by environmental pollutants than mycorrhizae at young pine plantations.*

Introduction

Acidification of forest soils caused by acid deposition of sulfur and nitrogen compounds is considered the main factor of forest decline in Northern Europe (Ulrich 1983). At low pH (< 4.5) the rates of dissolution of minerals and solid complexes increase and the chemistry of soil changes (Ritchie 1995). Increased amounts of soluble aluminum, manganese, and other metals can reach concentrations that may be toxic to plants. At the same time, base cations (Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}) are also solubilized and easily leached from the upper rooting horizon of the soil. It is believed that aluminum toxicity is related to increased Ca/Al ratios in the forest soil rather than pure Al^{3+} concentration (Ulrich 1983).

Ectomycorrhizae are an integral part of the root system of Scots pine *(Pinus sylvestris* L.*)*, the dominant forest species in Poland. Mycorrhizae improve growth of plants by enhancing uptake of water and nutrients (particularly phosphorus) and are totally dependent on plant derived carbohydrates transported from shoot to root. Mycorrhizal Scots pine seedlings have been shown to be more tolerant to aluminum ions than non-mycorrhizal plants (Goransson and Eldhuset 1991). It is considered that ectomycorrhizae may protect tree roots from the effects of environmental pollution, acting as the physical and/or chemical barrier. On the other hand, ectomycorrhizae and ectomycorrhizal fungi may be negatively influenced by indirect factors, such as low photosynthetic rates after yellowing of needles or needle loss, or direct factors, such as low pH and high soluble aluminum concentrations, low availability of nutrients, elevated soil nitrogen, or drought. Soil texture, aeration, and humus compounds are also important factors affecting mycorrhizae (Kottke and others 1993).

Changes in the chemical composition of soil affect the production of fine roots, which potentially could develop mycorrhizae. A significant decrease of fine root biomass in soils degraded by environmental pollutants was documented for Scots pine (Jozefaciukowa and others 1995, Munzenberger and others 1995). Some authors indicated that environmental pollutants reduce the number of mycorrhizal root tips and the diversity of mycorrhizal morphotypes (Dighton and Skeffington 1987, Kowalski 1987, Kowalski and others 1989).

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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Forest ecosystems in Poland have been under pressure of heavy industrial pollution, mainly SO_2, NO_x, and heavy metals, for about half a century (Godzik 1990). Although in recent years the levels of toxic pollutants emitted to the atmosphere have decreased, the southwestern part of Poland remains one of the most polluted areas in Europe (Tickle and others 1995). Toxic pollutants, such as heavy metals, accumulated in the contaminated soils and have affected the terrestrial and aquatic ecosystems.

This paper describes a study that compares the mycorrhizal status of Scots pine trees in forest ecosystems influenced by varying intensities of environmental pollution, using total number of ectomycorrhizal root tips in a soil volume and the diversity of mycorrhizal morphotypes as indicators.

Materials and Methods

Study Areas

The investigations were conducted at four Scots pine stands in Poland (*fig. 1*) influenced by different deposition rates of air pollution. At Lubon and Kornik young Scots pine trees planted in 1984 as 2-year-old seedlings were studied. The plot at Lubon is located about 2 km from a phosphate fertilizer plant established in 1917. The plant has emitted continual increasing levels of toxic air pollutants (Hernik 1987). In 1980 the emission of toxic gases were estimated: sulfur dioxide at 2,806 t yr^{-1}, fluorides at 162 t yr^{-1}, and nitric oxides at 850 t yr^{-1}. In recent years (since 1987) the amount of toxic pollutants emitted to the atmosphere has decreased considerably compared to previous years. In 1994 mean soil pH in the soil horizon 0-5 cm was: $pH_{water} = 4.2$, $pH_{salt} = 3.7$, and the Ca/Al ratio was 0.14 (Rudawska and others 1995). The Kornik stand is situated 12 km to the southeast of Lubon in the experimental forest of the Institute of Dendrology. This area, free of direct industrial pollution, was a control site. Soil at the control site was more than 10-fold less acidic than at the polluted site (mean $pH_{water} = 5.8$, $pH_{salt} = 4.9$). The Ca/Al ratio in the soil horizon 0-5 cm was 17 (Rudawska and others 1995). At the two young Scots pine plantations mycorrhizas were investigated, from autumn 1992 to autumn 1995, in the top 5 cm of the soil horizon. In 1995 additional samples from deeper soil layers were taken.

At the Niepolomice Forest and at Ratanica forest mature Scots pine trees (50-80 years old) were investigated. The Niepolomice Forest ecosystem has been affected for more than 40 years by gaseous and dust pollutants emitted by the adjacent urban-industrial area of Krakow (15-30 km west of the margin of the Forest) and by

Figure 1 — Experimental Scots pine plots in Poland.

the distant industrial area (Upper Silesian Industrial District — one of the most polluted areas in Central Europe, 70 km west of the Forest). The forest complex has been influenced by SO_2, NO_x, CO, CO_2, and heavy metals, including Fe, Zn, Pb, Cu, Cd, Ni (Manecki 1984). The average pH of soil in the upper 5 cm in 1994 was: pH_{water} = 4.3, pH_{salt} = 3.3. Scots pine is the major tree species of the forest stand, comprising 71 percent of the total area (Grodzinska 1984). Four sites (94, 124, 161, and 232) located on a west-east transect from Krakow were chosen as the experimental plots. Mycorrhizae were assayed in 1994, three times per year, at 0-5 cm and 5-10 cm soil depth.

The Ratanica experimental plot is located 40 km south of the Krakow urban area. It is a small forest catchment (88 ha) in the Carpatian foothills in the region of a drinking water reservoir, surrounded by intensively managed agricultural land. The catchment is influenced by gaseous and dust pollutants (containing SO_2, NO_2, Cl, Mn, Zn, Cu, Pb, Cd) transported there by winds from several distant industrial areas and by local agricultural emissions from surrounding villages (Niklinska and others 1995, Szarek 1995). Average pH in the organic horizon was reported as pH_{water} = 3.9 and in the mineral layer as pH_{water} = 4.3 (Grodzinska and Szarek 1995). The level of air pollution in this area was qualified as moderate but chronic (Grodzinska and Szarek 1995). The catchment is covered by a mixed forest with a predominance of Scots pine *(Pinus sylvestris* L.*)* and beech *(Fagus sylvatica* L.*).*

Mycorrhizal Frequency

Fine-root sampling was carried out with a 16 cm^3 soil corer from three places around a pine tree, at about 60-80 cm distance from the trunk. Samples were taken three times per year (May, August, October). At Lubon and Kornik 24 young trees at each plot were tested. At Niepolomice Forest six mature trees in each of the forest sites were investigated. At the Ratanica forest catchment three to seven mature Scots pine trees were studied in each of five areas of the entire catchment. Samples were taken to the laboratory and stored at 3 °C until analyzed. Roots were washed over a mesh, sorted, and analyzed under a stereomicroscope. The number of mycorrhizal root tips was counted in the root samples.

Characterization of Ectomycorrhizal Morphotypes

Each root tip was characterized according to morphological traits (shape, color, surface texture). The ectomycorrhizal morphotypes were distinguished and named according to the main types of ectomycorrhizal systems presented by Agerer (1987-1995) (*fig. 2*).

- **Type A:** Simple (unramified), elongate my corrhizal root tips, with a thin fungal mantle, their diameters were similar to non-mycorrhizal roots, yellow-brown in color, the surface was more or less smooth.
- **Type A_1:** Simple, elongate, thick mycorrhizae, their diameters were mostly three times the diameter of the supporting roots, light yellow in color, abundant fungal mantle with extramatrical hyphae on the surface.
- **Type B:** Dichotomous, elongate, thin mycorrhizae, orange-brown in color. The fungal mantle thin with a very smooth surface, no evidence of extramatrical hyphae.
- **Type C:** Shortened, abundantly dichotomizing mycorrhizae, light yellow, with extramatrical mycelium.
- **Type D:** Stumpy coralloid mycorrhizae with thickened yellow-brown branches. The surface was more or less smooth, without extramatrical hyphae. Older forms were darker in color and had a wrinkled appearance.

- **Type E:** *Cenococcum sp.* characteristically jet-black mycorrhizae with abundant emanating black hyphae stiffly projecting around the root tip. Infection by the fungus was also overgrowing other types of mycorrhizae, as A and B. Type
- **Type F:** Tuberculate mycorrhizae with a thick mantle and a smooth surface.
- **Type G:** Monopodial-pinnate mycorrhizae with thick fungal mantle, yellow or yellow- brown, with a smooth surface.
- **Type H:** Monopodial-pyramidal mycorrhizae with thick mantle, white-yellow or yellow- brown, smooth or grainy surface, old forms brown, with wrinkled surface.

Statistics

Data were processed using Duncan's multiple range test (ANOVA) or Scheffe test (Statistix 4.1). Statistical significance was accepted at $P < 0.01$.

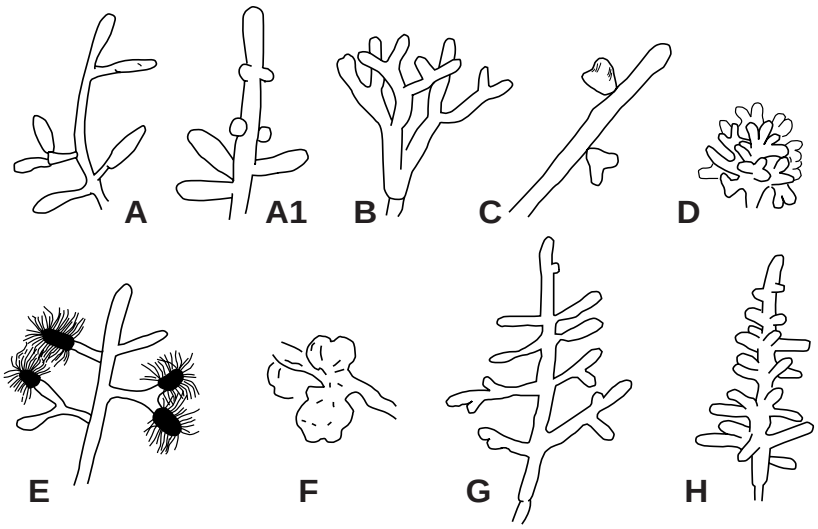

Figure 2 — Ectomycorrhizal morphotypes distinguished on Scots pine roots at the experimental plots:
A, A_1 - simple mycorrhizae;
B, C - dichotomous mycorrhizae;
D - coralloid mycorrhizae;
E - mycorrhizae with *Cenococcum* sp.;
F - tuberculate;
G - monopodial-pinnate;
H - monopodial- pyramidal.

Results

Young Plantations of Scots Pine (Lubon and Kornik)

At the polluted site (Lubon) the total number of mycorrhizal root tips in the upper soil horizon (0-5 cm), expressed per volume of soil sample, was significantly lower than at the control site in Kornik (*fig. 3*), although at both sites non-mycorrhizal roots were hardly found. The dry mass of fine roots (< 2 mm diameter) was slightly lower in soil samples from the polluted than from the control site; however, the differences were not significant (data not shown). In the 15-30 cm soil depth the average number of mycorrhizal root tips per soil volume was a little higher (not significantly) at the polluted than at the control plantation (*fig. 4*).

Mycorrhizal morphotypes distinguished on pine roots at the experimental plots (*fig. 2*), corresponded to the main types of ectomycorrhizal systems described by Agerer (1987-1995). At the control plantation at Kornik and the polluted site at Lubon, a maximum of seven morphotypes (A, A_1, B, C, D, E, F) was found (*table 1*). At both sites, the most frequent were the simple (unramified) mycorrhizae A and A_1. The frequency of the thin A type, considered as the less efficient, was similar at both plots. The dichotomous mycorrhizae with abundant fungal mantle (type C) and coralloid mycorrhizas (type D) were present at both polluted and unpolluted plantations at low frequencies.

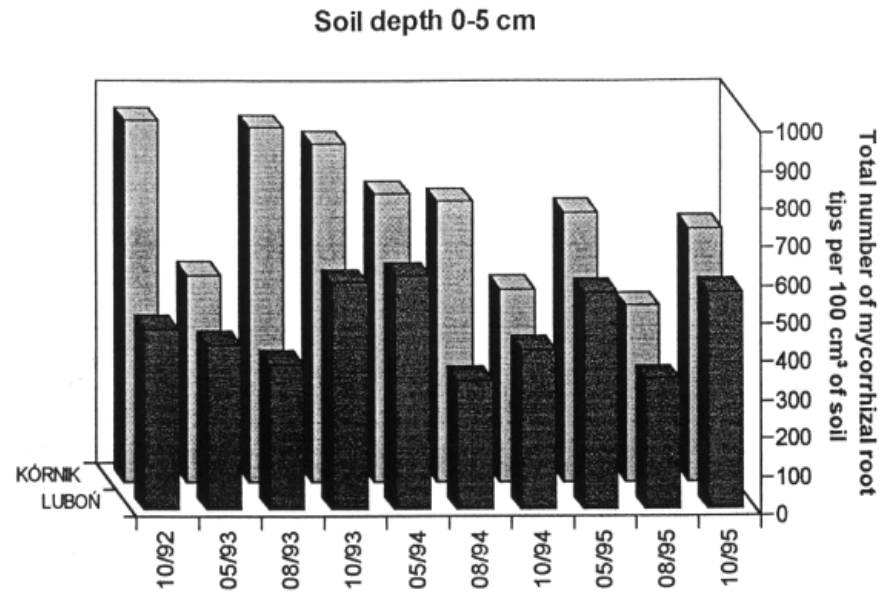

Figure 3 — Total number of ectomycorrhizal root tips in soil samples (100 cm³) taken from a 0-5 cm depth at the control (Kornik) and the polluted (Lubon) Scots pine plantations, from 1992 to 1995, three times per year. Data are means of 24 replicates. The differences between the sites are statistically significant ($P < 0.01$, Scheffe test).

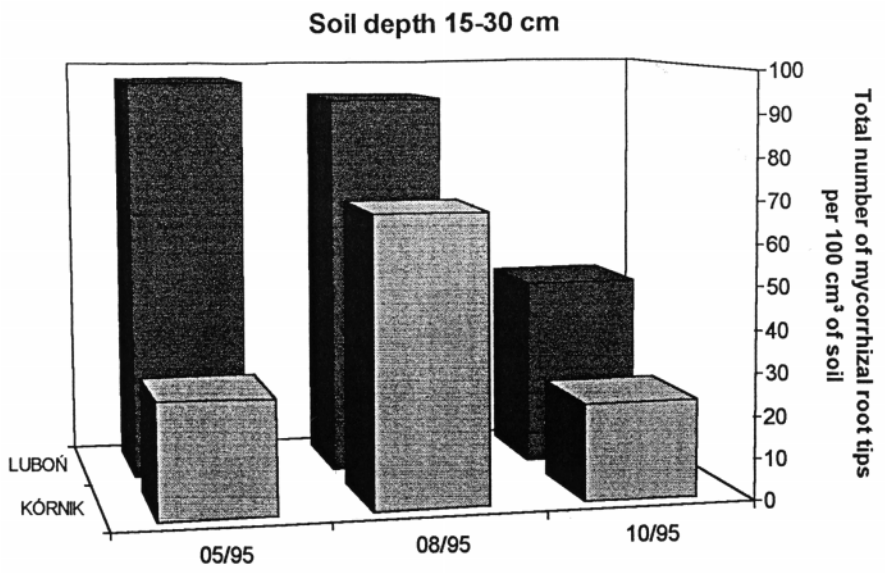

Figure 4 — Total number of mycorrhizal root tips in soil samples (100 cm³) taken from a 15-30 cm depth at the control (Kornik) and polluted (Lubon) Scots pine plantations in 1995. Data are means of 24 replicates. The differences between the sites are not significant statistically, according to the Scheffe test.

Mature Pine Trees (The Niepolomice Forest and Ratanica Forest Catchment)

In the Niepolomice Forest the total number of ectomycorrhizae was lower at the sites located closer to the urban-industrial centers than at the more distant sites, however, in August and October, in the soil depth 0-5 cm, the differences were not significant statistically (*fig. 5*). Four mycorrhizal morphotypes (A, A[1], B, D) were found on pine roots in the Niepolomice Forest (*table 2*). Generally, Scots pine roots developed at these forest sites mostly simple, thin mycorrhizae A (up to 84 percent). The dichotomous mycorrhizae (B) and coralloid morphotypes (D) were typically a low percentage of the mycorrhizae in the plots, with a few exceptions.

Considerable variations in the total number of mycorrhizae per soil volume (from 360 to 2,030) were found among 22 pine trees tested in the Ratanica forest catchment, however, the mean values count for various sites of the forest were similar or differed slightly (*fig. 6*). The average number of mycorrhizal root tips in the Ratanica forest catchment, in soil depth 0-5 cm, was more than 100 percent higher in relation to the Niepolomice Forest.

Table 1 *— Frequency of mycorrhizal morphotypes of Scots pine at two soil depths: 0-5 cm and 15-30 cm, at control (Kornik) and polluted (Lubon) plantations. Root samples were observed in May, August, and October 1995.*[1]

Time	Site	Percentage of mycorrhizal morphotypes						
		A	A_1	B	C	D	E	F
		Soil depth 0-5 cm						
05/95	Kornik	61 ± 16	21 ±11	7 ± 3	1 ± 1	5 ± 5	0	5 ± 5
	Lubon	53 ± 17	20 +14	13 + 8	1 + 1	10 + 3	0	3 + 3
08/95	Kornik	74 ± 7	5 ± 3	14 ± 6	2 ± 2	3 ± 2	2 ± 2	0
	Lubon	60 ± 13	8 ± 7	23 ± 10	1 ± 1	7 ± 7	1 ± 1	0
10/95	Kornik	73 ± 7	10 ± 5	6 ± 3	3 ± 2	5 ± 5	3 ± 3	0
	Lubon	59 ± 15	14 ± 7	9 ± 3	7 ± 7	8 ± 3	3 ± 3	0
		Soil depth 15-30 cm						
05/95	Kornik	67 ± 22	10 ±10	9 ± 7	1 ± 1	9 ± 9	0	3 ± 3
	Lubon	60 ± 24	11 ± 11	12 ± 9	0	17 ± 12	1 ± 1	0
08/95	Kornik	79 ± 9	1 ± 1	13 ± 7	0	6 ± 4	1 ± 1	0
	Lubon	60 ± 13	1 ± 1	17 ± 4	0	20 ± 10	0	2 ± 2
10/95	Kornik	71 ± 17	14 ± 14	9 ± 9	1 ± 1	4 ± 4	1 ± 1	0
	Lubon	47 ± 27	12 ±10	10 ± 6	4 ± 4	15 ± 12	0	12 ± 12

[1] Data are means ± SD (n = 24).

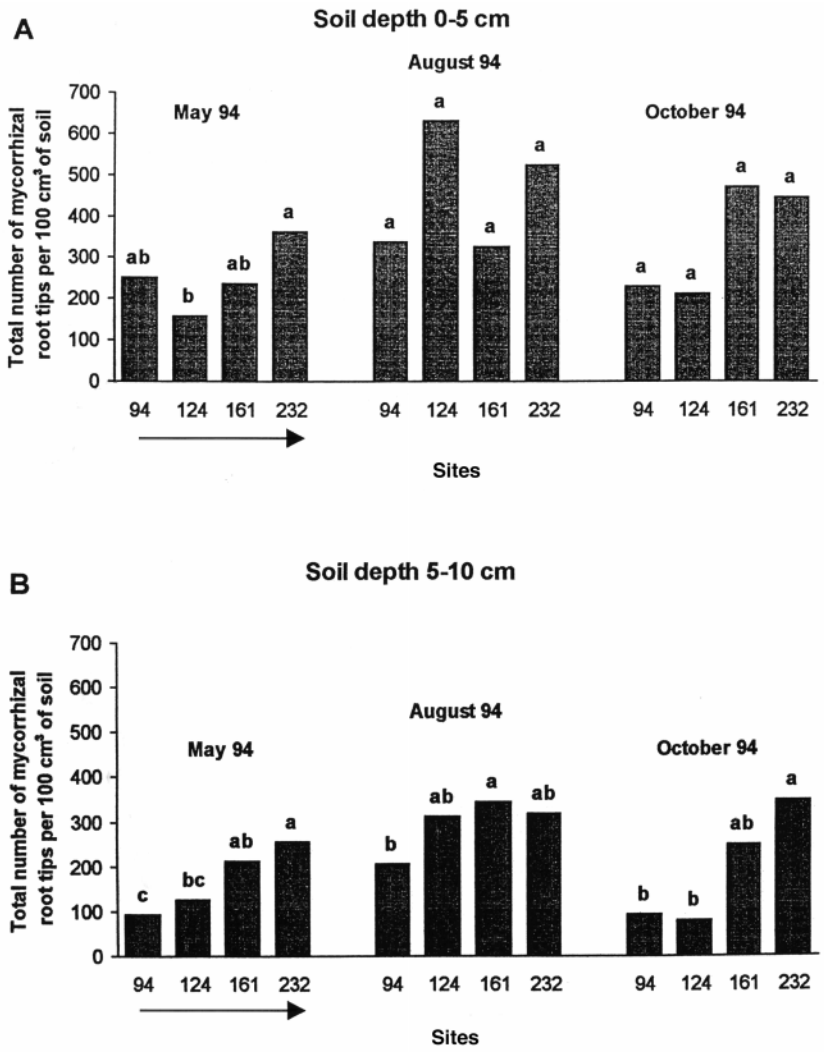

Figure 5 — Total number of mycorrhizal root tips in soil samples (100 cm³) taken from 0-5 cm and 5-10 cm depths at four sites (94, 124, 161, 232) in the Niepolomice Forest, located along a transect extending from the Krakow area, the source of air pollutants. Data are means of six replicates. Means followed by the same letter do not differ significantly (P < 0.01, ANOVA, Duncan's test).

Eight morphotypes of pine ectomycorrhizae were distinguished in the Ratanica forest catchment (*table 3*). The average frequency of the simple, thin mycorrhizas (type A), considered as the less beneficial for a tree, was lower than 50 percent. Occurrence of new morphotypes, compared to the Niepolomice Forest was observed: abundantly dichotomizing (C), monopodial-pinnate (G), monopodial-pyramidal (H), and Cenococcum sp. mycorrhizae (E).

Discussion

Mycorrhizal status of forest trees provides accurate information about changes in the forest soil affected by environmental pollution. In this study reduced numbers of Scots pine mycorrhizae were found at the severely polluted forest stands in Poland (Lubon, the Niepolomice Forest) compared to the slightly or moderately polluted sites (Kornik, Ratanica forest catchment) (*figs. 3, 5, 6*). Significant reduction in mycorrhizae was found at 11-18 year-old Scots pine plantations (Kowalski and others 1989) and in mature Scots pine stands (Ohtonen and others 1990) influenced by industrial pollutants. A decrease in numbers of mycorrhizae was also caused by low pH and available aluminum in perlite/peat cultures (Metzler and Oberwinkler 1987). The influence of other important factors on mycorrhizae, like soil type and soil moisture must also be considered (Kottke and others 1993).

Frequencies of ectomycorrhizal morphotypes are considered accurate indicators of the mycorrhizal status of forest trees. Dighton and Skeffington (1987) observed that artificial acidic precipitation and an increase of Al^{3+} concentration in

Table 2 *— Frequency of mycorrhizal morphotypes of Scots pine at soil depths: 0-5 cm (I) and 5-10 cm (II) at four forest sites in order of decreasing pollution exposure (94, 124, 161, 232) in the Niepolomice Forest.* [1]

Time	Site	Percentage of mycorrhizal morphotypes							
		A		A_1		B		D	
		I	II	I	II	I	II	I	II
05/94	94	82 ± 11	75 ±19	10 ± 7	7 ± 7	4 ± 3	16 ±14	3 ± 3	1 ± 1
	124	74 ± 24	59 ±25	17 ± 23	8 ± 7	5 ± 3	6 ± 5	4 + 4	27 ±25
	161	50 ± 6	35 ± 25	45 ± 19	64 ±32	3 ± 3	1 ± 1	2 ± 2	0
	232	65 ± 16	68 ±17	26 ± 13	24 ± 12	2 ± 2	4 ± 4	6 ± 5	4 ± 4
08/94	94	84 ±13	77 ±17	6 ± 6	14 ± 14	6 ± 6	5 ± 5	4 ± 2	4 ± 4
	124	58 ± 13	76 ± 7	37 ± 17	18 + 9	2 ± 2	2 ± 2	3 ± 2	4 ± 2
	161	70 ± 18	70 ± 22	23 ± 16	25 ± 24	2 + 2	2 ± 2	6 ± 5	3 ± 3
	232	59 ±30	80 ± 21	32 ± 12	17 ± 17	4 ± 3	1 ± 1	5 ± 4	2 ± 2
10/94	94	60 ± 10	73 ± 9	25 ±12	19 ± 13	6 ± 3	3 ± 3	8 ± 4	4 ± 4
	124	80 ± 19	77 ±127	7 ±6	8 ±8	5 ± 5	8 ± 7	9 ± 8	6 ± 5
	161	70 ±19	56 ±27	28 ±17	44 ± 31	2 ± 2	0	1 ± 1	0
	232	62 ±19	81±13	25 ±16	15 ±10	8 ± 5	1 ± 1	5 ± 4	2 ± 2

[1] Data are means ± SD (n = 6)

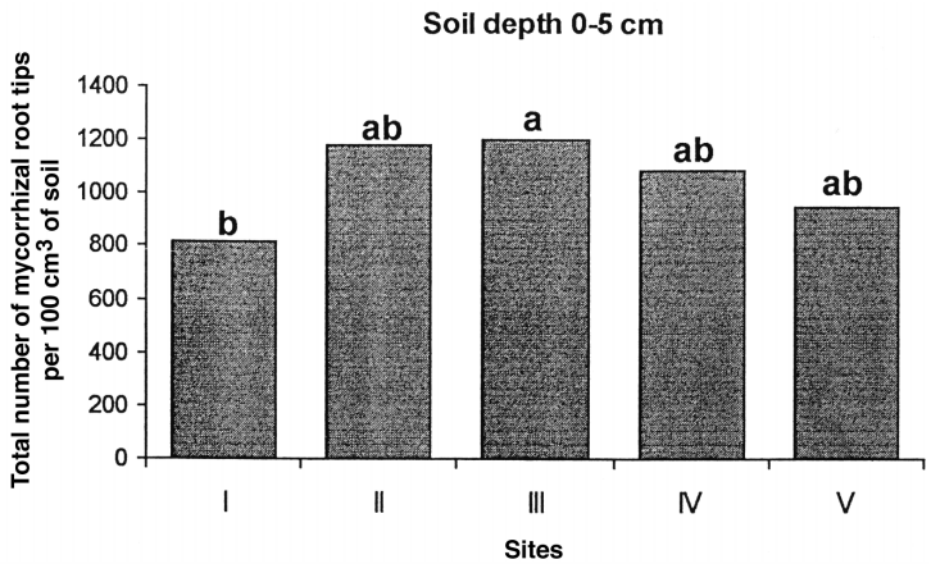

Figure 6 — Total number of mycorrhizal root tips in soil samples (100 cm^3) taken from a 0-5 cm depth at five sites in the Ratanica forest catchment. Data are means of three to seven replicates. Means followed by the same letter do not differ statistically ($P < 0.01$, ANOVA, Duncan's test).

soil solution significantly reduced the branching of fine roots of Scots pine and decreased the occurrence of two coralloid mycorrhizal morphotypes. Mycorrhizal roots of Scots pine at highly contaminated areas in Poland occur mainly as simple mycorrhizae with a scant fungal mantle, whereas in forests with low-level industrial pollution, mycorrhizae are more commonly of the dichotomous morphotypes (Kowalski 1987, Kowalski and others 1989).

In the present studies, no evident differences in the proportion of ectomycorrhizal morphotypes were found between the polluted and the control young Scots pine plantations (*table 1*); however, considerable differences in the occurrence of mycorrhizal morphotypes were observed between the severe-polluted and moderately-polluted mature pine trees (*tables 2, 3*). Simultaneously, at the polluted young pine plantation we recorded a more abundant occurrence of species and carpophores of ectomycorrhizal fungi (Rudawska and others [In press]). Termorshuizen and Schaffers (1987) observed that mycorrhizal fruitbodies were not depressed by air pollution in young stands of Scots pine (5-13 years), but they were reduced in mature stands (50 years). This phenomenon could be explained by relatively low susceptibility of mycorrhizal fungi species living in symbiosis with young Scots pine trees (Shaw and others 1992). Ectomycorrhizal fungi show differential responses to Al in pure cultures (Kieliszewska-Rokicka and others [In press], Thompson and Medve 1984). Since Scots pine tolerates a wide range of ecological conditions and it is also classified as relatively tolerant to soluble Al, (Schaedle and others 1989) the mycorrhizae formation could be limited by the sensitivity of fungi rather than of the plant (Metzler and Oberwinkler 1987).

Reduced mycorrhizal diversity in the Niepolomice Forest compared to the Ratanica forest catchment suggests a lower biodiversity of mycorrhizal species at the severely-polluted mature pine stand than in the moderately-polluted forest. Kowalski and others (1989) reported that the reduction in diversity of mycorrhizal morphotypes at a contaminated Scots pine plot was accompanied by significant changes in the composition of mycorrhizal fungi species and a decrease in the production of fruitbodies. Termorshuizen and Schaffers (1991) have found a negative correlation between SO_2 concentration in the air and the number of mycorrhizal species and carpophores in mature Scots pine forests in The Netherlands. Carpophores of mycorrhizal fungi and fungal species have declined during this century in Europe (Arnolds 1991). The disappearance of various species of mycorrhizal fungi has been observed also in many areas in Poland (Wojewoda and Lawrynowicz 1986).

Environmental pollutants undoubtedly affect mycorrhizae directly, inhibiting the uptake of nutrients and causing nutrient imbalance in root and shoot (Janhunen and others 1994) or limiting growth of the mycorrhizal root tips (Metzler and

Table 3 *— Frequency of mycorrhizal morphotypes of Scots pine in the top 5 cm of soil at five sites in the Ratanica forest catchment. The fine-root samples were collected in August 1995.*[1]

Site	Percentage of mycorrhizal morphotypes							
	A	A_1	B	C	D	E	G	H
I	54 ± 12	8 ± 6	10 ± 5	7 ± 1	5 ± 3	10 ± 1	4 ± 4	2 ± 2
II	37 ± 12	33 ± 7	8 ± 4	10 ± 5	4 ± 3	5 ± 5	1 ± 1	2 ± 2
III	52 ± 8	31 ± 9	4 ± 2	7 ± 1	3 ± 1	1 ± 1	1 ± 1	1 ± 1
IV	50 ± 9	26 ± 11	12 ± 5	4 ± 3	3 ± 1	3 ± 2	1 ± 1	1 ± 1
V	42 ± 17	26 ± 16	15 ± 8	7 ± 2	5 ± 1	2 ± 1	1 ± 1	2 ± 2
MEAN	47 ± 7	25 ± 10	10 ± 4	7 ± 2	4 ± 1	4 ± 3	2 ± 1	2 ± 1

[1] Data are means ± SD (n= 3-7).

Oberwinkler 1987). At young Scots pine plantation contaminated by soluble Al, an increased Al/Ca ratio in needles was correlated with a decrease in the net photosynthesis, biomass production, and allocation to roots (Reich and others 1994). Simultaneously, a high, negative correlation between aluminum content in pine needles and the number of mycorrhizal root tips in the upper 5 cm of soil has been recorded for the young pine trees (Rudawska and others 1995). Other external factors, like drought, frost, and pathogens could intensify the effects of environmental pollutants.

Conclusions

The decrease in the number and diversity of mycorrhizae at the heavily polluted sites is presumably related to a direct negative effect of aluminum and other metals on fungal hyphae as well as the influence of pollutants on photosynthesis and transport of carbohydrates to roots. Mycorrhizae of mature Scots pine trees seem to be more sensitive to the stress factors than mycorrhizae of young trees. As Scots pine changes ectomycorrhizal partners during its ontogeny, a modification of mycorrhizal status with the age of the plantation could be possible.

Acknowledgments

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Impact of Urban Environmental Pollution on Growth, Leaf Damage, and Chemical Constituents of Warsaw Urban Trees[1]

Waldemar Chmielewski, Wojciech Dmuchowski, Stanislaw Suplat[2]

Abstract

*In the last 10 years, 3.5 percent of the tree population died annually in Poland's largest and most polluted cities, which is a problem of economic importance. Dieback of streetside trees in Warsaw is a long term process. It is an effect of biological reactions of trees to unfavorable conditions in the urban environment, particularly air and soil pollution and water deficiency. The process is being intensified by microclimatic changes leading to xerism of the environment (lowering of air humidity and rising of air temperature). To study this problem Crimean linden (*Tilia euchlora *K. Koch) trees were examined. Crimean linden, constituting over 40 percent of Tilia in urban areas, and used in the reconstruction of urban plantings after World War II at the beginning of 1950. Observation of phenologic development of trees indicated a shortening of their vegetation period (up to 30 days) depending on the proximity to the city center. The dendrometric examinations revealed a reduction of trunk diameter of the affected trees. These changes were accompanied by morphological dieback of leaves observed in the beginning of July. Significant correlations were found between the leaf damage and chemical concentration of some nutrients and pollutant elements.*

Introduction

During the last 15 years, urban forestry research in Warsaw, Poland, has focused on applied problems. The results may be of interest for specialists evaluating specific ecological conditions of urban areas and health of forest trees in central and eastern Europe.

Change of climatic conditions in urban areas is a part of the recently observed climatic changes on a global scale. Unfortunately, climatic changes caused by urbanization, have led to extensive die-back of urban trees. In the beginning of the 1980's, in western Europe, the estimated number of trees dying annually was over 830,000 (Fluckinger and Braun 1981). Poland also faces the same problem: in the last 10 years up to 3.5 percent of the tree population has died annually in the country's biggest and the most polluted cities (Suplat 1989).

During periods of rapid urbanization, local scale monitoring of changes in environmental pollution and their impact on trees can be achieved on a shorter time scale than at global scales. The rate of changes in soil and air pollution is faster in urban ecosystems than in forest ecosystems. Our research suggests that climatic and environmental changes adversely impact urban trees at a time scale of 1 to 2 decades.

This paper discusses our long term research to define the biological reaction of trees to specific environmental pollutants and climatic conditions in urban areas in Poland.

Objectives and Methods

The first phase of research was to determine biological responses to urban temperature anomalies. We applied a 15-year phenological monitoring program from 1973 to 1988 in the Warsaw area. This program was also part of a study to determine the chronic effects of pollution on urban trees.

More than 100 Crimean linden (*Tilia euchlora* K.Koch) trees, 30-40 years of age and growing on 12 examined areas, were monitored from 1973-1988. The areas represented different habitat conditions and included parks, city squares, and streets (*fig. 1*). Observations were made of phenological development, chemical composition of leaves, and trunk growth by diameter at breast height (DBH) (Chmielewski 1990).

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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In the second research phase, 143 Crimean linden trees were analyzed in 1995 for health (e.g., leaf damage) and their nutritional status. The trees were located in an ecologically homogenous habitat and were exposed to extensive traffic and soil pollution (such as de-icing salt).

Measurements included evaluation of tree condition by determining the index of leaf damage in relation to tree-crown condition (United Nations Economic Commission for Europe 1994): the seasonal dynamics of the foliar nutrients, status of trees showing various damage levels, and dendrometric measurements (such as DBH). Every 2 weeks during the vegetative season (May-September), leaf damage, their chemical constituents, and tree-stem diameter were measured.

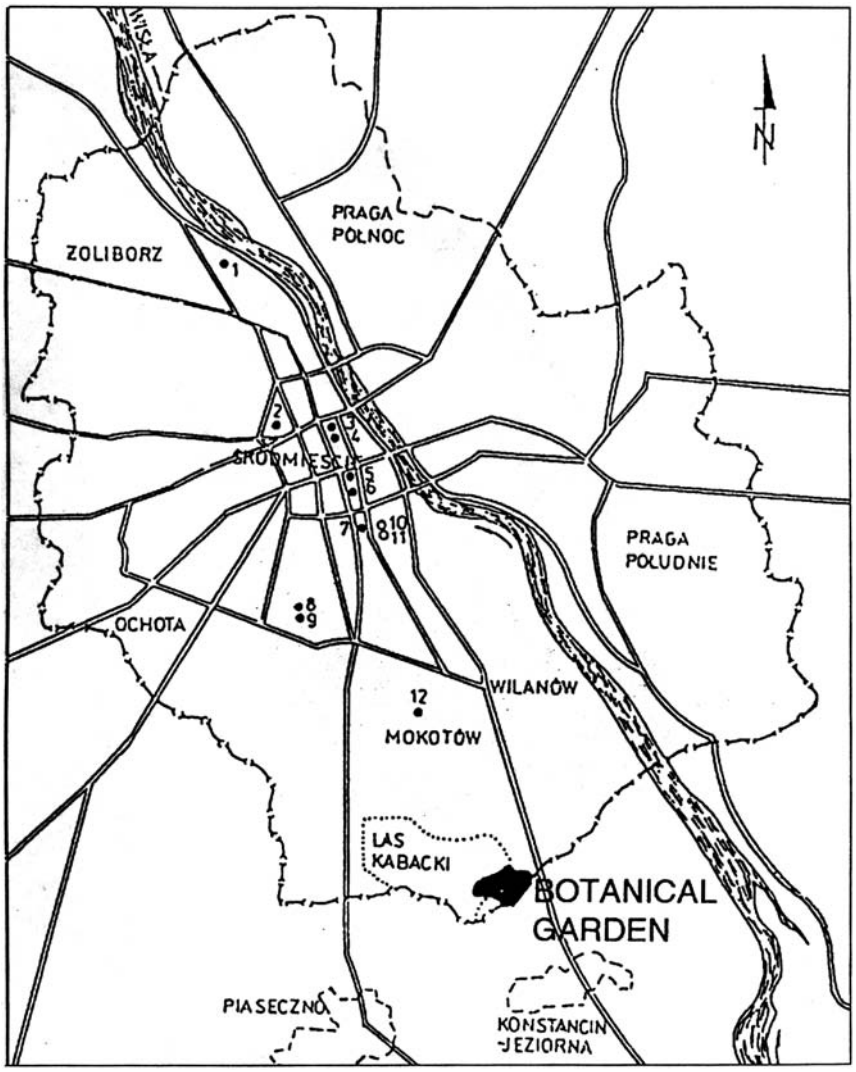

Figure 1 — Distribution of observation plots (habitats) in Warsaw along the north-south gradient. Outskirts and large city parks: 1—Bielanski Forest; 2—Towarowa street; 8—Solders Cemetery; 9—Zwirki, Wigury street; 10—Ujazdowskie street; 11—Lazienki Park; 12—Ursynow. Squares, streets, and parks in the city center: 3—Saski Garden; 4—Krolewska street; 5—Marszalkowska street; 6—Konstytucji Sq.; 7—Unii Lubelskiej. Sq.

Results and Discussion

The vegetation period is defined as the beginning of leaf-buds bursting to the date of leaves falling. Results suggest that over the period 1973-1988, environmental pollution affected the length of in-leaf season for urban trees (*fig. 2*). Twelve study areas were arranged along an urban-suburban gradient. The in-leaf season for trees located in the city center was shorter than for trees in suburban areas. The average difference between habitats located in the city center (Konstytucji Square) and at the outskirts (Ursynów) was 25 days. Similar results for other tree species were reported by Feige and others (1980), Supuka (1988), and Zacharias (1972), who also attribute the shortening of the vegetation season to urbanization and degradation.

For trees in streets and city squares, there was also a gradual shortening of the vegetative period throughout 1973-1982 period. Average reduction in the length of the vegetative period for 12 habitats was about 30 days, with considerable fluctuations between the individual years.

In the population of examined trees, the rate of leaf damage was increasing throughout the vegetative season. By the end of July, only 30 of 143 examined trees showed leaf damage while the index of damage ranged from 6 to 75 percent. Analyses of the chemical composition of tree leaves revealed differences between the trees of various leaf damage indices.

A seasonal variation was found in the chlorine accumulation in the Crimean linden (*Tilia euchlora* K.Koch) leaves (*fig. 3*). The trees with strongly damaged leaves showed high chlorine concentrations immediately after the leaves emerged. Leaf chlorine concentrations also increased during the vegetative season. In the group of trees not showing leaf damage, leaf chlorine concentrations at the beginning of the vegetative season were relatively low, but increased with the time. Leaf chlorine concentrations in healthy trees located at the city outskirts (e.g., Bielanski Forest) were more than three times lower than in unhealthy urban tree leaves that were damaged (Zwirki, Wigury Street). At the suburban site, the rise in chlorine concentrations during the vegetative season was slight. According to Pauleit (1988), healthy Crimean linden leaves contain 0.27-0.30 percent chlorine. For damaged leaves the level is 1.0 to 6.0 percent (Ruge and Stach 1968).

The results of the chemical analysis of leaves did not indicate any deficiency of the nutrient elements (Chmielewski and others 1994). However, higher than normal physiological levels of chlorine, sodium, sulfur, lead, and copper concentrations were measured, suggesting that the ionic equilibrium of unhealthy trees was disturbed. The measured chlorine concentrations could be characterized as toxic. However, nutrient ratios are also significant indicators of crown condition (Brogowski and others 1988, Packham and others 1992).

To characterize the ionic equilibrium of tree tissues, the gram-equivalent values were used. In the interpretation of research results of this study, the gram-equivalent values we used included relation of bivalent cations to monovalent cat-

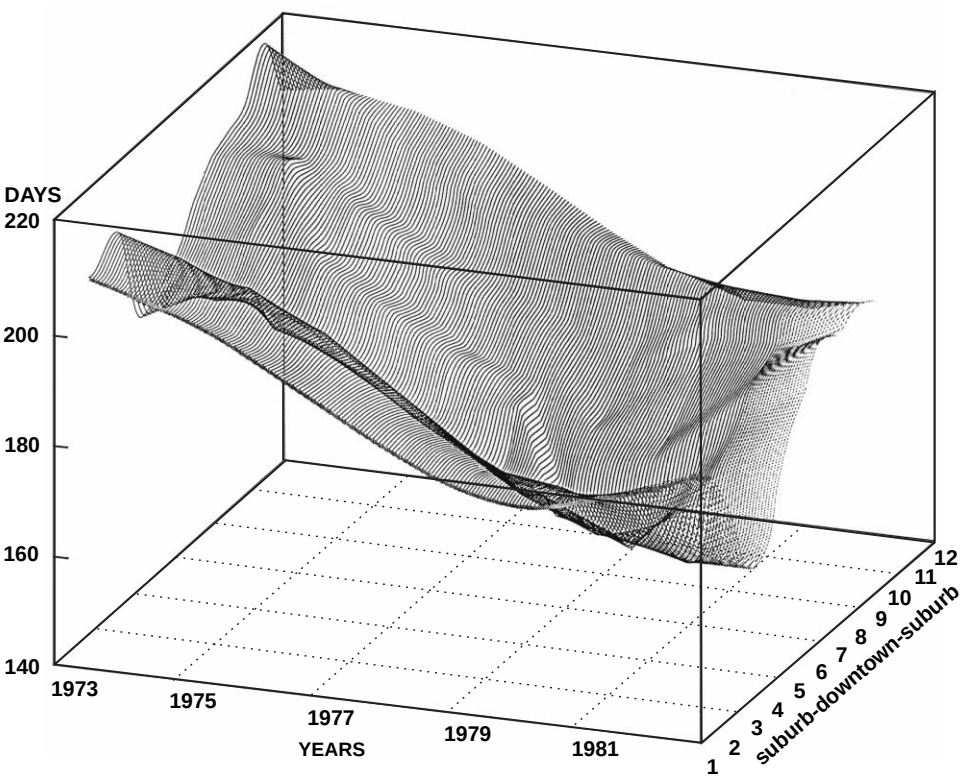

Figure 2 — Two tendencies (1973-1982) in the vegetation period of Crimean linden trees (*Tilia euchlora* K. Koch) in the 12 examined habitats in Warsaw.

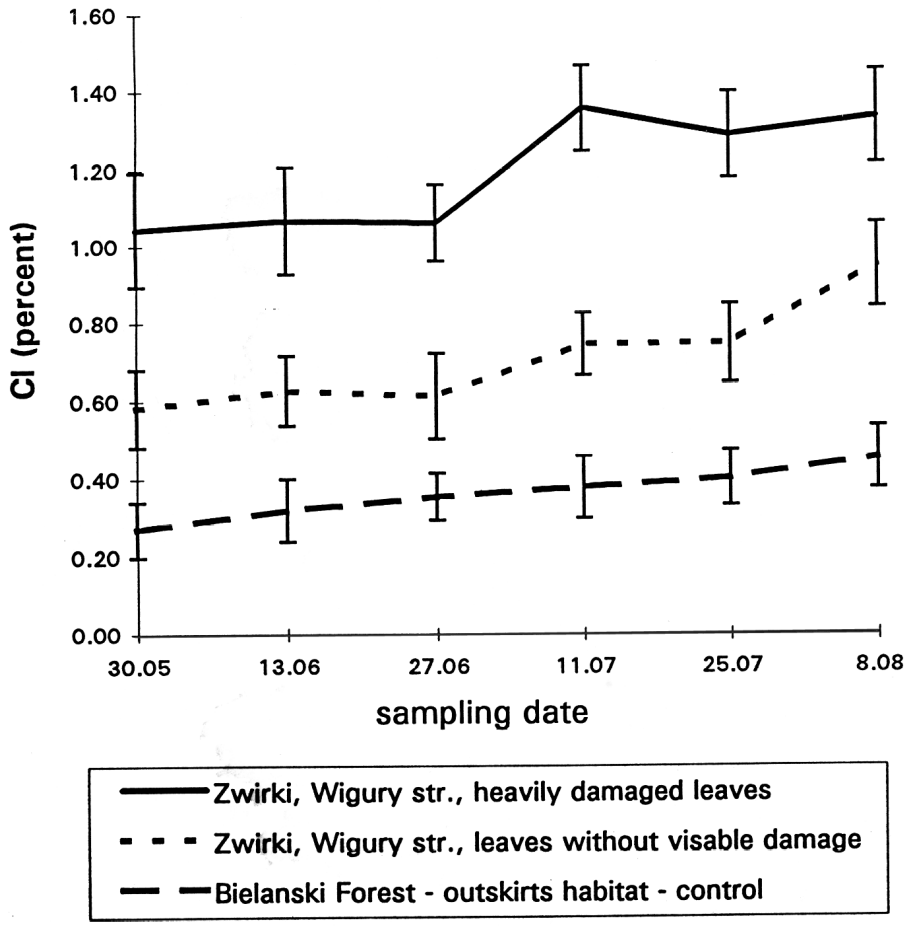

Figure 3 — Seasonal changes of chlorine accumulation in the Crimean linden tree leaves in three classes of leaf damage.

ions, total cations, relation of calcium to magnesium, and ANE (accumulated nutrient elements—total of all examined elements) (Ostrowska and others 1988).

Measurements were made of tree stem diameter at the breast height (DBH) and the index of leaf damage from July and September. The DBH ranged from 71 to 172 centimeters and was positively correlated with the leaf damage index for September. The index of leaf damage was higher in the trees containing more leaf chlorine and calcium. The leaf damage index was inversely proportional to magnesium content (*table 1*).

The September leaf damage index, the DBH, and the coefficient of ionic equilibrium (independent variables) were correlated (*table 2*). There was no correlation between DBH and ionic equilibrium in trees. The leaf damage index was positively correlated with the ANE indicator and with the high calcium to magnesium ratio given in gram-equivalent values.

These are preliminary results. Detailed statistical analysis of current data, including anatomic studies of cambium, phloem, and xylem should be done in future studies to further determine the effects of urban environments on trees in Poland.

Table 1 *— Correlations between the index of damaged Crimean linden tree (*Tilia euchlora *K.Koch) leaves, the tree-stem DBH (dependent variables), and the concentration of selected elements in leaves.*[1]

Dependent Variables	Chlorine	Sodium	Magnesium	Calcium
Index of leaf damage	+	0	-	+
Decrease in tree-stem DBH	+	-	+	-

[1] 0 = nonsignificant correlation

+ = positive significant correlation

- = negative significant correlation

Table 2 *— Correlation between the index of damaged Crimean linden tree (*Tilia euchlora *K.Koch) leaves, the tree-stem DBH (dependent variables), and the coefficient of ionic equilibrium (independent variables) in gram-equivalent values.*

Dependent Variables	A[1]	B	C	D	E
Index of leaf damage	[2]0	0	+	+	+
Decrease in tree-stem DBH	0	0	0	0	

[1]A = ratio of bivalent cations to monovalent cations ($Ca^{+2} + Mg^{+2}/K^{+} + Na^{+}$)

B = sum of cations ($Ca^{+2} + Mg^{+2} + K^{+} + Na^{+}$)

C = ratio of calcium to magnesium (Ca^{+2} / Mg^{+2})

D = ANE accumulation nutrient elements

E = tree-stem DBH.

[2]0 = nonsignificant correlation

+ = positive significant correlation ($p \leq 0.05$)

- = negative significantcorrelation ($p \leq 0.05$)

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Seasonal Influences on Ozone Uptake and Foliar Injury to Ponderosa and Jeffrey Pines at a Southern California Site[1]

Patrick J. Temple and Paul R. Miller[2]

Abstract

*Ambient ozone was monitored from 1992 to 1994 near a forested site dominated by mature Jeffrey and ponderosa pines (*Pinus jeffrey *Grev. & Balf. and* Pinus ponderosa *Dougl. ex Laws.) at 2,000 m in the San Bernardino Mountains of southern California. Ozone injury symptoms, including percent chlorotic mottle and foliage retention, were evaluated on 130 mature pines in September of 1991 to 1994 at a site 2 km from the ozone monitor. Symptom measurements were combined in an ozone injury index (OII) that represented the average crown condition of each plot. Leaf stomatal conductance and available soil water were measured biweekly from March to November at one of the plots from 1992 to 1994. Mean annual ozone concentrations were relatively constant for the 3 years. However, the distribution of individual trees in the 0 to 100 range of the OII showed a shift from 1991 to 1994 to higher frequencies of trees in classes representing greater amounts of injury. Compared with 1991, in 1993 and 1994 more trees remained in the higher injury classes. Maximum rates of stomatal conductance and maximum ozone uptake occurred in June of each year, but the total annual ozone flux varied as the amount of available soil water varied with seasonal precipitation. The period of April to June or May to July, depending upon year, accounted for 70 to 75 percent of annual ozone flux, confirming the high correlation observed between ambient ozone concentrations from April to June and development of chlorotic mottle on the needles. Foliar injury to current-year needles, which served as a marker of ozone response for a single exposure season, was greatest in 1992, coincident with higher flux of ozone to foliage in 1992, compared with 1993 and 1994. These data confirm that the period of maximum ozone impact on these pines is early in the growing season, and that ambient ozone after July had less effect on the development of foliar ozone injury on older pine needles. Year-to-year variation in ozone injury to pines was influenced both by the status of soil moisture availability in summer at this location and the internal foliar dynamics of the trees as they recovered from drought.*

Introduction

Annual precipitation throughout the ranges of ponderosa and Jeffrey pines (*Pinus ponderosa* Dougl. ex Laws. and *Pinus jeffreyi* Grev. & Balf.) in the mountains of California is subjected to strong seasonal influences, with more than 90 percent of total precipitation falling in the months of October to April. Annual precipitation totals are also highly variable, and the long-term annual mean is composed of many years with low total precipitation and a few years of much greater than average precipitation (NOAA 1995). Summer drought is the norm, interrupted by occasional brief, local thunderstorms. The growth cycles of the trees have become adapted to these highly variable conditions, and gas exchange in the pines closely followed patterns of soil water availability and vapor pressure deficit (Helms 1970, 1971).

Because the uptake of ozone by plants is controlled by stomatal conductance (Runeckles 1992), annual and seasonal patterns of ozone uptake in ponderosa and Jeffrey pines may also be mediated by environmental factors, such as soil water availability and vapor pressure deficit, that control gas exchange (Helms 1971). Intrinsic factors, such as leaf age (Helms 1970) and amount of foliar ozone injury (Coyne and Bingham 1982), also influence the rate of gas exchange in ponderosa pines. Those factors that influence stomatal conductance could also affect the response of the tree to ozone and could be used to model ozone uptake and foliar injury responses of trees in the field. Thus, measurements of stomatal conductance and calculations of ozone flux to leaves may be used to bridge the gap between measurements of ambient ozone concentrations in the forest and measurements of foliar injury symptoms on pines.

The objectives of this study were to measure seasonal ozone flux to the foliage of mature ponderosa and Jeffrey pines as influenced by environmental and physiological factors, and to correlate ozone flux with observed changes in ozone injury measured on mature pines at the same location in the San Bernardino Mountains of southern California.

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

[2] Project Consultant and Plant Pathologist, respectively, Pacific Southwest Research Station, USDA Forest Service, 4955 Canyon Crest Drive, Riverside, CA 92507, U.S.A.

Materials and Methods

Ambient Ozone Concentrations

Continuous measurements of ozone concentrations from January 1992 to December 1994 were obtained from a meteorological station located at Barton Flats at 2,000 m in the mixed conifer forest zone of the San Bernardino Mountains in southern California. Ozone air quality was measured with a certified Environmental Protection Agency (EPA)-equivalent ozone analyzer, calibrated and maintained according to standard air quality protocols (Miller and others 1996a). Precipitation data were obtained from a meteorological station located at Big Bear Lake, 10 km north of the site.

Pine Plot Evaluations

Ozone injury symptoms were evaluated each September from 1991 to 1994 on 5 branches of 130 mature ponderosa or Jeffrey pines located at an elevation of 2,000 to 2,200 m, 2 km from the meteorological station. The number of annual needle whorls was counted and each whorl was evaluated for percent of needle area with chlorotic mottle, percent of needles remaining in each whorl, and average needle length of each whorl. The percent live crown was determined as the proportion of the total crown with any live branches relative to the total tree height. These parameters were combined into an ozone injury index (OII) by weighting chlorotic mottle by 0.40, abscission by 0.40, needle length by 0.10, and percent live crown by 0.10 (Miller and others 1996b). Injury scores were averaged to obtain the average crown condition for each plot for each year. The distribution of trees in 10 unit increments of the 0 to 100 of the OII were also determined annually. Higher numerical values of the OII mean greater chronic injury.

Measurements of Stomatal Conductance

Leaf stomatal conductance was measured twice monthly from March to November and monthly from December to February of 1992 to 1994 at 25 m height in the canopy of a stand of mature ponderosa and Jeffrey pines on one of the pine evaluation plots. Access to the top of the trees was provided by a tower that penetrated through the canopy to a height of 30 m. Branches from three trees were accessible from the tower, and stomatal conductance measurements were taken on all available age classes of needles on three branches from each of three mature pines. Stomatal conductance was measured with a steady-state porometer (Model LI-1600, Licor Inc., Lincoln, NE)[3] equipped with a cuvette specifically designed for conifer foliage. Needle surface area was calculated by using the diameter length of the cuvette opening (5 cm) multiplied by the diameter of the needle fascicle and the number of needles in the fascicle (usually three). Fascicle diameter was measured with an electronic digital caliper (Mitutoyo Corp., Japan) to the nearest 0.01 mm. Measurements were taken usually between 0900 and 1300, when rates of stomatal conductance were at maximum. However, stomatal conductance measured hourly throughout three complete diurnal cycles showed that conductance rates in mature pines varied little during most of the daylight hours.[4] Photosynthetically active radiation, relative humidity, leaf and ambient temperature were measured simultaneously with conductance measurements by sensors on the porometer cuvette. Volumetric soil water was measured bi-weekly under the tree canopy, except during months with snow cover, using a time domain reflectometry instrument (TRASE Model 1010X, SoilMoisture Corp., Santa Barbara, CA) equipped with 15 cm soil probes.

Ozone Flux

Flux of ozone to pine foliage was calculated as:

$$\text{Flux } (O_3)(\mu\text{mol m}^{-2}\text{ d}^{-1}) = [O_3]_a\,(\mu\text{mol mol}^{-1}) \times g_s(O_3)\ (\text{mol m}^{-2}\text{ d}^{-1})$$

in which $[O_3]_a$ = ambient ozone concentration per day, averaged over the number

[3] Mention of trade names or products is for information only and does not imply endorsement by the U.S. Department of Agriculture.

[4] Unpublished data on file at the Pacific Southwest Research Station, Riverside, Calif.

of daylight hours from 30 min after sunrise to 30 min before sunset. Stomatal conductance to ozone ($g_s(O_3)$) was calculated from $g_s(H_2O)$ by multiplying by the ratio of the molecular weights of O_3 to H_2O. Boundary layer resistance in the well-mixed conifer cuvette was factory-set in the instrument at 0.07 s m^{-1}. The actual boundary layer resistance of the measured pine needles would depend upon wind speed, but generally at the upper layers of the canopy wind speed > 5 m s^{-1}. Other aspects of ozone transport from the atmosphere to internal leaf surfaces (Runeckles 1992) were ignored for the purposes of these calculations. Mean daily ozone flux (µg m^{-2} d^{-1}), averaged over each month, was calculated from mean monthly daylight ozone concentrations and mean daily rates of stomatal conductance for each month of the 3-year study.

Results and Discussion

Average ozone concentrations were relatively uniform at Barton Flats during the 3 years of the study. Mean monthly daylight ozone concentrations ranged from 38 parts per billion (ppb) in January to 83 ppb in June (*fig. 1*). Highest mean ozone concentrations occurred in June of 1992 and 1994, but ozone levels were lower in July and August of 1992 because of unusually cloudy and rainy conditions during the late summer of 1992. Ambient ozone averaged 18 percent lower in July 1992 relative to July 1994; but for the 5-month period from May to September, ozone concentrations averaged 73 ppb, 74 ppb, and 74 ppb for 1992, 1993, and 1994, respectively.

Annual precipitation totals were highly variable at the Barton Flats site (*fig. 2*). For the 4-year period preceding the study, precipitation totals averaged 50 percent below the long-term mean of 83 cm. This prolonged period of drought was reflected in the foliage of ponderosa and Jeffrey pines at the site, which were significantly shorter and lighter in weight than needles produced after the drought had been relieved in 1991 (Miller and others 1996a). More than 90 percent of annual total precipitation fell during December to March of each of the three years of the study (Miller and others 1996a).

This pattern of several years of drought before the study, followed by rapid recovery from drought stress beginning in 1991 was reflected in the amount of

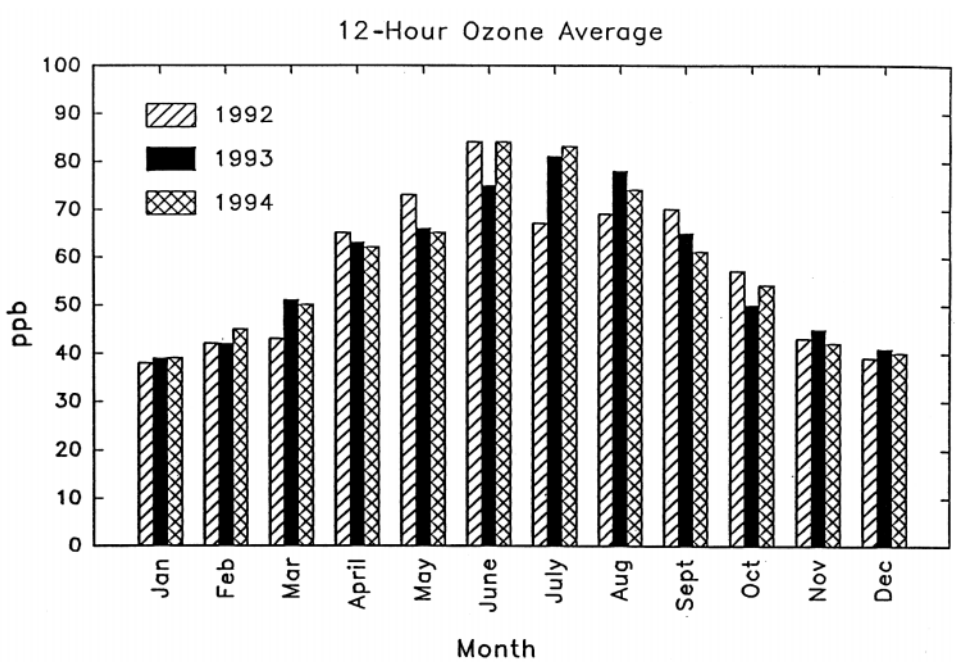

Figure 1 — Monthly 12-hour (0600-1800) ozone averages for Barton Flats, San Bernardino Mountains, California, 1992 to 1994.

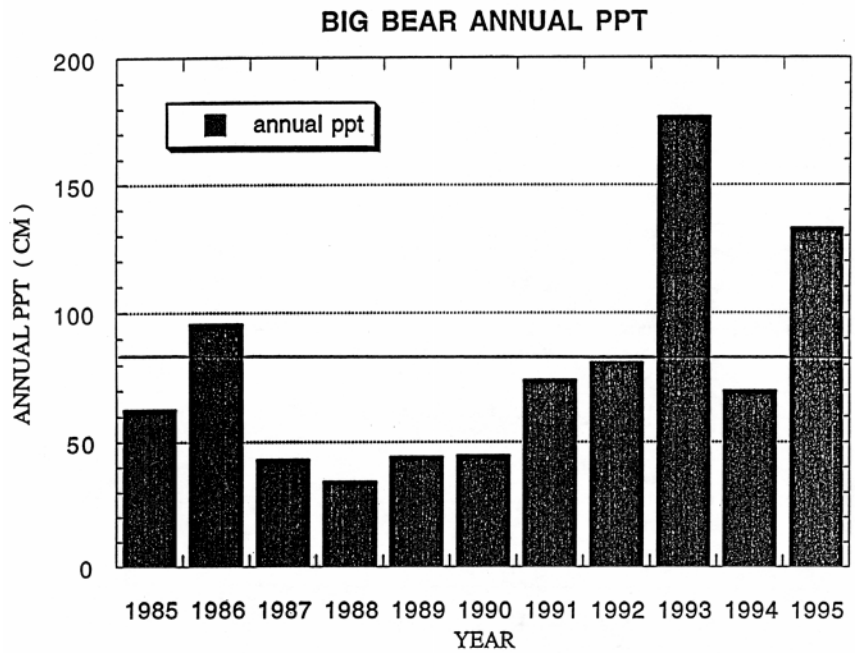

Figure 2 — Annual total precipitation at Big Bear Lake, San Bernardino Mountains, 10 km north of Barton Flats, 1985 to 1995.

ozone injury on the ponderosa and Jeffrey pines at Barton Flats (*table 1*). In 1991, the mature pines averaged more than 4 years of needle whorl retention and the ozone injury index (OII) averaged 38.3. By 1992, the number of age classes of needles had decreased to an average of 3.6 and the OII had increased significantly ($p < 0.001$) to 47.5, an increase of 24 percent. The OII stabilized over the next 2 years, but the number of needle whorls retained continued to decline (*table 1*), as the fascicles retained during the drought were replaced by larger, more physiologically-active needles.

Table 1 — *Ozone injury index (OII), percent chlorotic mottle (CM) on current-year needles, and number of age classes of needles retained on mature ponderosa and Jeffrey pines at Barton Flats, San Bernardino Mountains, recorded in September 1991 to1994.*[1]

Year	OII	CM (pct)	Age classes retained
1991	38.3	0.076	4.2
1992	47.5	0.798	3.6
1993	47.5	0.258	3.4
1994	49.2	0.005	3.3

[1] Data are means of 5 branches from 130 trees in 3 plots at the site.

Distribution curves of the frequency of trees in 10-unit classes of the OII (*fig. 3*) showed a shift from a relatively flat distribution in 1991 to clustering in the higher injury classes beginning in 1992 and continuing in 1993 and 1994. Injury on current-year needles in 1992 was 10 times greater than in 1991 and three times greater than in 1993. Injury on current-year needles was very low in the drought year of 1994 (*table 1*). The effect of drought in reducing the impact of ozone on these mature pines in the San Bernardino Mountains was comparable to the responses of drought-stressed ponderosa pine seedlings exposed to ozone under experimental conditions (Temple and others 1992).

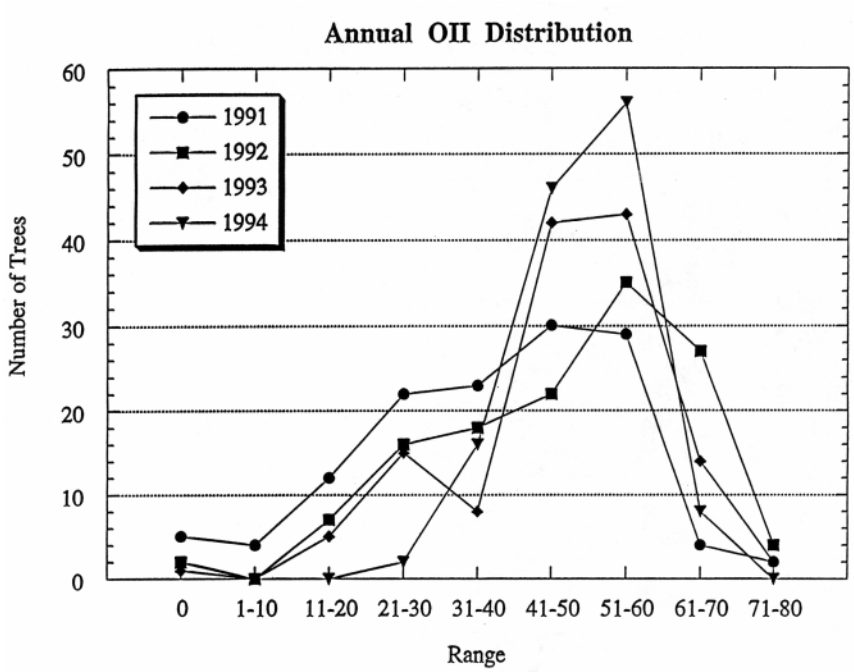

Figure 3 — Distribution of 130 ponderosa and Jeffrey pines in 10 unit classes of the ozone injury index (OII) from 1991 to 1994 (higher OII scores mean more injury).

Seasonal trends in rates of stomatal conductance of pine needles generally followed the same patterns in 1992 to 1994, although variations in total precipitation and summer rainfall patterns that influenced soil water retention altered conductance rates, particularly in mid- to late summer. In 1992, conductance in 1-year-old needles peaked in June and gradually declined (*fig. 4*), although conductance increased again in late August in response to summer rains that had increased soil water content 2 weeks earlier. Current needles showed measurable rates of gas exchange in July, but conductance rates in these needles remained low, relative to maximum rates of 1-year-old needles in June. In 1993, maximum conductance rates of 1-year-old needles peaked in early June, then gradually declined, except for small increases in July and September in response to increased soil water content 2 to 3

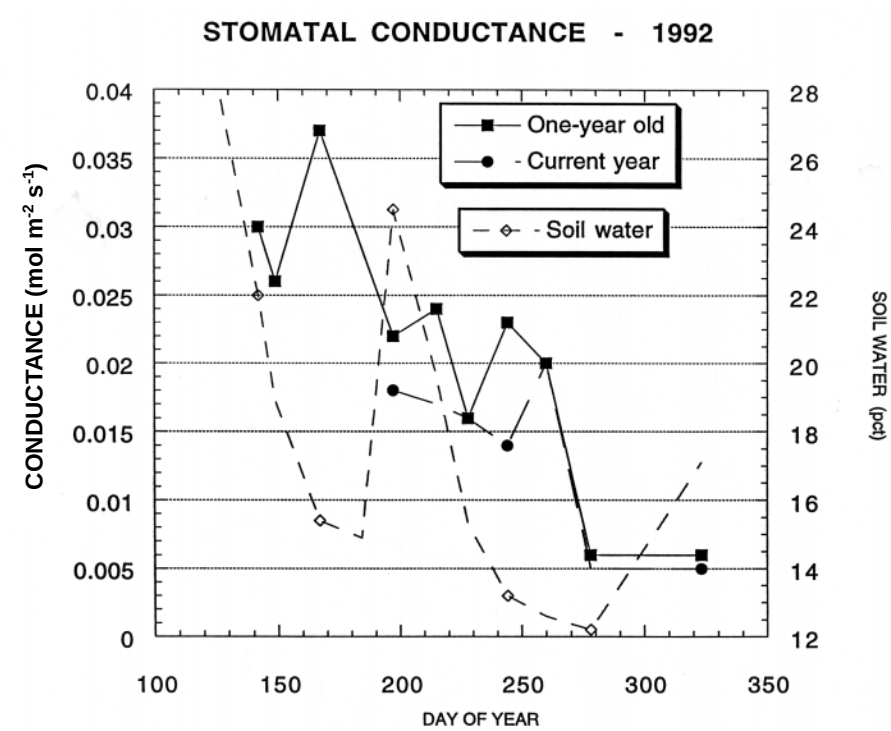

Figure 4 — Seasonal trends in maximum rates of stomatal conductance in 1-year-old needles of mature ponderosa and Jeffrey pines (n=3), and volumetric soil water content under the canopy at Barton Flats, San Bernardino Mountains, 1992.

weeks earlier (*fig. 5*). Current-year needles developed measurable conductance in mid-August, and conductance rates in current needles peaked in mid-September, but maximum conductance rates in current needles were below peak conductance rates in 1-year-old needles. In 1994, winter precipitation (1993-1994) was 30 percent below average, and soil water content depleted rapidly during the summer. Rates of stomatal conductance closely followed soil water content, with peak conductance in April and May, followed by rapid reductions in conductance as the soil dried during the summer (*fig. 6*). Current needles became active in early August, but conductance in current needles remained low for the rest of the year.

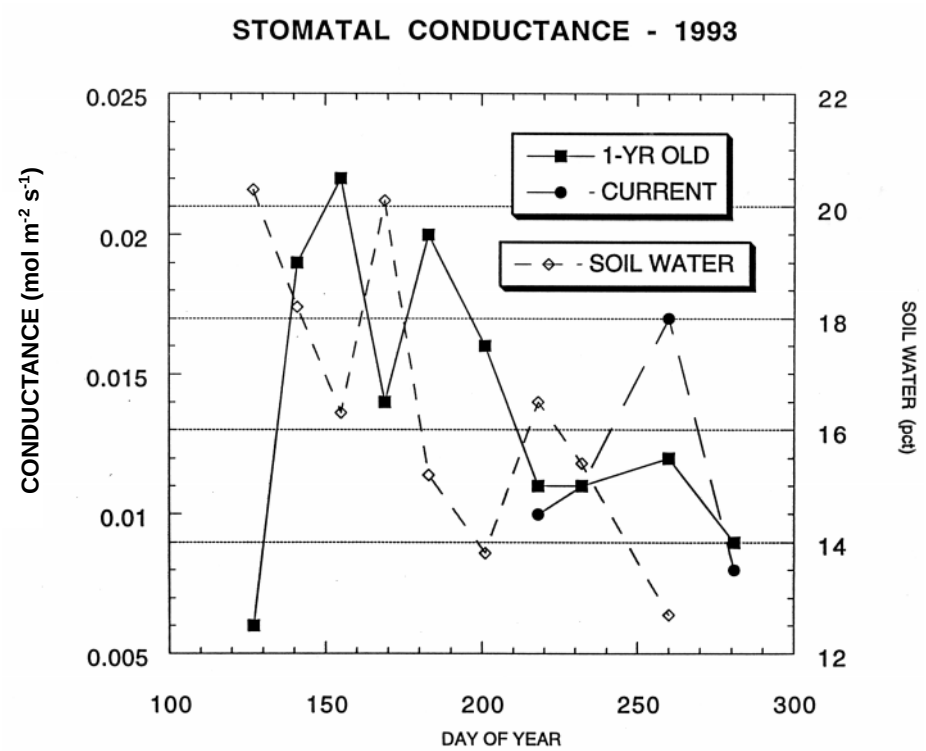

Figure 5 — Seasonal trends in maximum rates of stomatal conductance in 1-year-old needles of mature ponderosa and Jeffrey pines (n=3), and volumetric soil water content under the canopy at Barton Flats, San Bernardino Mountains, 1993.

These results are similar to those reported for ponderosa pine in the Sierra Nevada, where gas exchange in photosynthesis was influenced primarily by temperature and light intensity under favorable soil water conditions. In mid- to late summer, under high temperature and low soil water, photosynthetic rates declined in ponderosa pine in response to increased vapor pressure deficits (Helms 1970, 1971). However, reports of higher rates of gas exchange in Jeffrey pine during winter months compared with rates in September (Patterson and Rundel 1989) were *not* confirmed by these results

Highest mean daily ozone flux to 1-year-old needles of mature pines at Barton Flats for the study period was in 1992 (*fig. 7*), reflecting the high stomatal conductance rates measured in 1-year-old needles in 1992. The high rates of conductance in these 1991 needles in 1992 may have been due to the high rate of physiological activity in these needles, as previous years' needles produced during the drought years were significantly smaller than 1991 needles, and had already begun to senesce by 1992 (Miller and others 1996a). Ozone flux peaked in June, and this month accounted for 28 percent of the total yearly ozone flux in 1992. In 1993, ozone flux was more evenly distributed throughout the growing season, but total ozone flux averaged 40 percent less than in 1992. Flux rates were higher in 1994, although precipitation was lower than in 1993 (*fig. 2*). In response to rapid decreases in rates of conductance (*fig. 6*), ozone flux was higher in April 1994 but lower in July, August, and September than in 1992 and 1993. For the 3 years of the study, the months of April to July accounted for more than 60 percent of total annual ozone flux.

These results are consistent with statistical analyses of ambient ozone concentrations and development of ozone injury symptoms on pines, in which highest correlations were found between ozone injury and ozone concentrations between April 15 and July 15 (Miller and others 1996a).

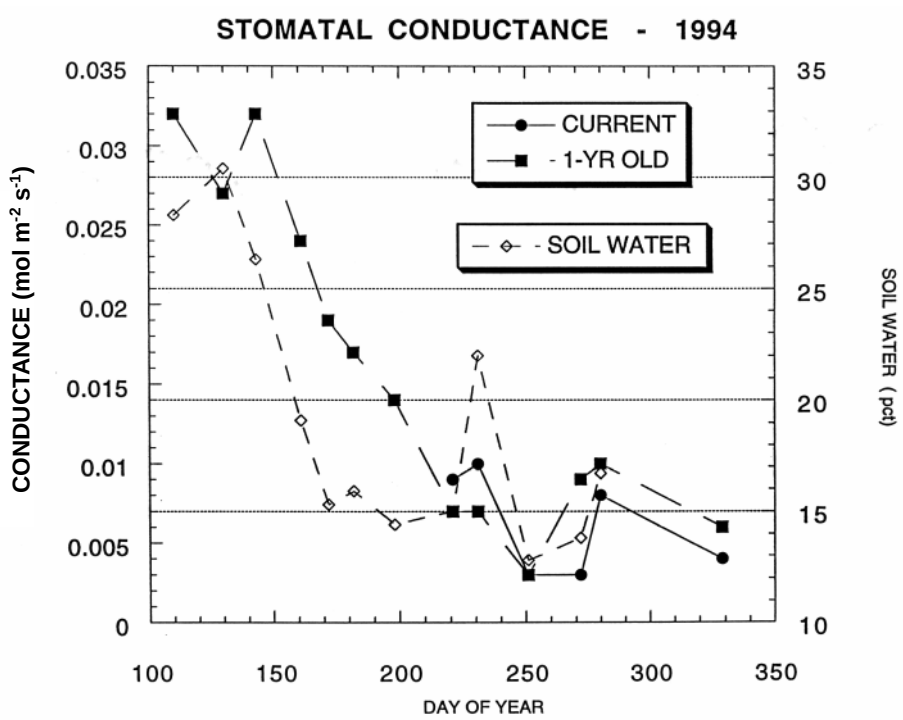

Figure 6 — Seasonal trends in maximum rates of stomatal conductance in 1-year-old needles of mature ponderosa and Jeffrey pines (n=3), and volumetric soil water content under the canopy at Barton Flats, San Bernardino Mountains, 1994.

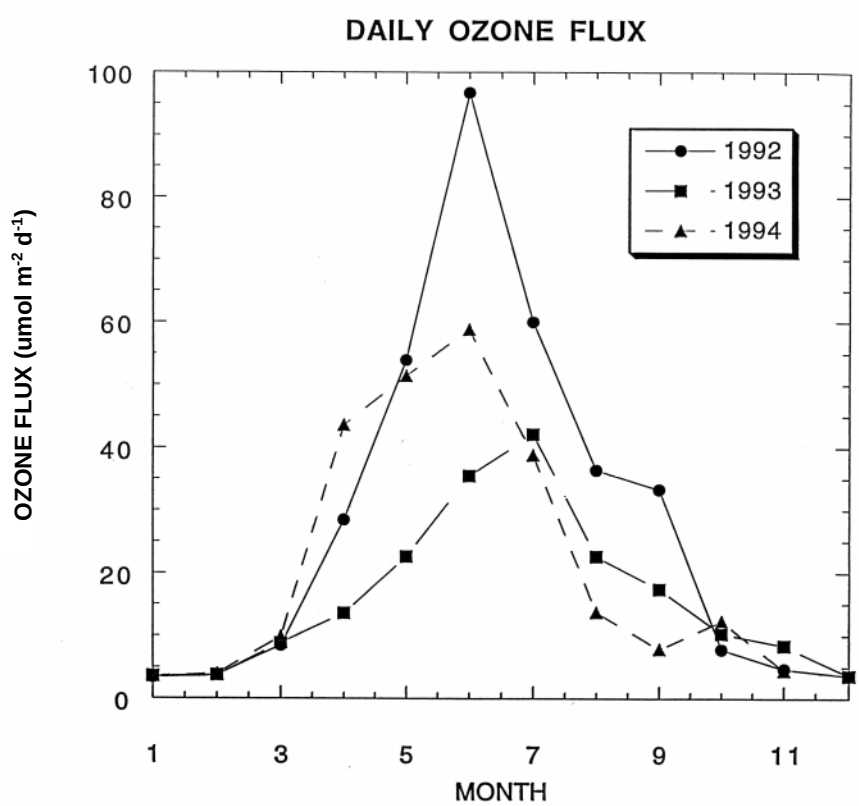

Figure 7 — Mean daily ozone flux to 1-year-old needles of mature ponderosa and Jeffrey pines at Barton Flats, San Bernardino Mountains, 1992 to 1994.

Conclusions

Results of this study of ozone injury development relative to stomatal conductance and ozone flux to foliage of mature ponderosa and Jeffrey pines in the San Bernardino Mountains showed that ozone injury increased substantially in the year following relief from a prolonged period of drought. The distribution of individual trees in OII classes showed a shift from 1991 to 1992 to greater numbers of trees in classes representing higher injury levels. Seasonal patterns of conductance followed trends of diminishing soil water as the summer progressed, and highest rates of ozone flux occured early in the growing season, with relatively little ozone flux after July. However, the high level of variability in annual precipitation influenced the size and persistence of annual needle whorls, particularly during and after recovery from a prolonged drought. These internal foliar dynamics interacted with external environmental parameters, such as winter precipitation and soil water depletion, to influence rates of stomatal conductance and ozone flux to pine foliage, which influenced development of ozone injury on these trees.

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Effect of Ozone Exposure on Seasonal Gas Exchange of Five Western Conifers[1]

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Abstract

*Five species of western conifers (*Pinus ponderosa, Abies concolor, Pseudotsuga menziesii, Abies lasiocarpa, *and* Picea engelmannii*) were exposed, in two standard open-top exposure chambers per treatment, to charcoal-filtered air and a simulated diurnal ozone exposure profile (120 d sum of 136 ppm-h) to test their relative sensitivity.* CO_2 *exchange rate (CER), stomatal conductance (gs), and total leaf nitrogen of current year foliage were measured at 1-month intervals.* P. ponderosa *was the most adversely affected by ozone, based on reductions in mid- and late season CER under saturating light (1.18 vs. 2.17* μ*mol* CO_2 m^{-2} s^{-1}*), lower light compensation point (43 vs. 33* μ*mol quanta* m^{-2} s^{-1}*), and lower CER at a given gs value.* P. menziesii *showed reduced CER and higher gs by the end of the experimental ozone exposure.* A. concolor *and* P. engelmannii *showed no reduction in* CO_2 *uptake and no clear effects of ozone exposure on gs. Adverse ozone effects on biomass or growth in the year of exposure were seen for* A. lasiocarpa *and* P. ponderosa*, whereas* A. concolor, P. engelmanii *and* P. menziesii *showed adverse effects only in the year after exposure. Few consistent patterns in which plant component was affected were found, either in the year of exposure or the year after exposure. An analysis of the number of chambers required or the difference between treatment means necessary to obtain statistical significance with the population variance measure was an important result.*

Introduction

Considerable variations in the response of trees to air pollutants may be caused by differences in seed source, nutritional status of plants and other stressors, plant age, and plant phenological stage (Bytnerowicz and Grulke 1992). Many experimental exposure studies have analyzed the effects of pollutants on gas exchange at one phenological period: near the end of the growing season when cumulative experimental exposure to the pollutant is at a maximum. However, because gas exchange rates may be at a minimum, treatment effects may be difficult to distinguish statistically.

Plant response to ozone may not necessarily be cumulative, but relatively few studies have studied this by examining seasonal responses during ozone exposures (Byres and others 1992, Grulke and others 1989, Hanson and others 1994, Samuelson and others 1996, Weber and others 1993). As long as concentrations are sufficiently low, ozone may have little effect on gas exchange. However, high ozone uptake rates have been found to reduce stomatal conductance (gs) (Weber and others 1993).

This study determined the relative susceptibility of five species of western conifers — *Abies concolor* [Gord. & Glend.] Lindl., *Abies lasiocarpa* [Hook.] Nutt., *Picea engelmannii* Parry, *Pinus ponderosa* Dougl. ex. Laws, and *Pseuotsuga menziessii* Franco — to ozone exposure by analyzing gas exchange characteristics (CO_2 exchange rate, stomatal conductance, and dark respiration), growth, and biomass allocation.

Materials and Methods

Plant Culture

Two-year-old, bare-root *A. concolor* seedlings were obtained from the California Department of Forestry nursery in Ben Lomond (seed zone 531.50). One-year-old, container-grown *P. ponderosa* seedlings were obtained from the same nursery (seed zone 522.20, lot 3245). Two-year-old *A. lasiocarpa* seedlings in containers were obtained from Plants of the Wild, Tekoa, Washington (seed from south of Lewiston, Idaho, 1,372 m). Two-year-old, bare root stock of *P. engelmannii* (seed source Warm Lake, Idaho) was obtained from Lucky Peak Nursery. One-year-old container-grown seedlings of *P. menziesii* were obtained from Rex Timber, Inc., Cottage Grove, Oregon (seed zone 471, 620 m).

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside California.

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Seedlings (6.4 cm dia., 25.4 cm tall) were potted in a mix consisting of equal parts Promix B[1] and perlite. Before fumigation, plants were kept in an activated charcoal-filtered, evaporative-cooled greenhouse adjacent to the fumigation site. The greenhouse and fumigation site were installed at the Kenworthy Ranger Station (33° 37′ N, 116° 37′ W), in the San Jacinto Mountains, 22 km SE of Idyllwild, and 80 km W of Palm Springs, California. The elevation of the site was 1,387 m. A drip irrigation system was used to water and fertilize the plants (North Carolina State University fertilizer mix). Plants were watered two times weekly to pot capacity. Nutrients were applied every 1-2 weeks through the irrigation system from March through October 11, 1988.

Ozone Exposure

Seedlings were exposed to either charcoal-filtered air (CF), or charcoal-filtered air augmented with ozone ($+O_3$) under ambient conditions in open-top exposure chambers (Heagle and others 1979). Chambers operated from June 4 (Julian day 155) through October 11, 1988. For the elevated ozone treatment, a 30 d diurnal profile was developed from average air quality characteristics during the summer months in the midwest (1.8 times ozone exposure level; Hogsett and others 1989; Lefohn and others 1986a, 1986b). The 30 d base profile was repeated about four times in the 124 d exposure period *(fig. 1)*. The ozone exposure in each treatment was estimated from the following equations:

$$\text{CF:} \quad y = 0.278x - 1.44 \quad (1)$$

$$O_3: \quad y = 1.118x - 1.66 \quad (2)$$

in which y was ozone exposure in ppm-h and x was number of days of exposure.

Ozone was generated, distributed, and sampled within each chamber by using a computer-controlled program with a data acquisition system (Hewlett Packard).[4] Oxygen (O_2) was supplied to an ozone generator (Griffin Technics, Model GTC-0.5b), and distributed to chambers with mass flow controllers (Datametrics, Type 825). Ozone was sampled from the center of each chamber by using a sequential sampler (Scanivalve, custom-made electronics), Blower boxes provided a chamber exchange rate of 2 min^{-1} and ozone concentrations were determined by ultraviolet photometry (Dasibi, Model 1003AH, calibrated bi-weekly on site). Fiberglass dust filters and activated charcoal-filters were in place for all chambers. Concentrations of ozone in chambers were recorded hourly as were ambient ozone, chamber air temperature, soil temperature at 5 cm, and total global radiation (Licor 200SZ).

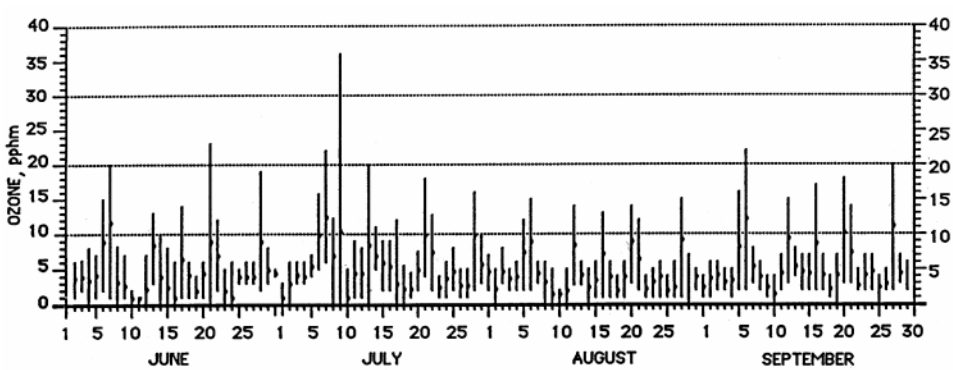

Figure 1 — Seasonal course of daily mean, maxima, and minima ozone concentration in the elevated ozone chambers. The lowest ozone concentrations occurred in the morning from 0700 to 0900, and the highest ozone concentrations occurred between 1500 and 1700 hr.

Gas Exchange

Before experimentation, a typical subset of plants for physiological measurement was placed in each chamber: all had one terminal bud, appeared to be in the same phenological stage, and were visually similar in height, numbers of side branches, and foliage color. Bud flush had occurred before placement in chambers. Three seedlings of each species per chamber and two chambers each for charcoal filtered air and elevated ozone treatments were available for physiological

[4] Mention of trade names or products is for information only and does not imply endorsement by the U.S. Department of Agriculture.

measurements. Means of each plant were obtained from two to four sequential runs (three to four observations averaged per run). Gas exchange from the three seedlings of each species in one chamber were averaged together to obtain the chamber mean and standard error. The analysis of variance (ANOVA) calculations were strictly applied to the chamber means (one degree of freedom within a treatment).

The conifers were exposed to charcoal-filtered air and elevated ozone in open-top chambers for one growing season. The conifers included *Abies concolor* [Gord. & Glend.] Lindl., *Abies lasiocarpa* [Hook.] Nutt., *Picea engelmannii* Parry, *Pinus ponderosa* Dougl. Ex Laws, and *Pseudotsuga menziesii* Franco. Gas exchange of current year foliage was measured four times during the exposure period from early June through late September. Dark respiration rate and light response of *A. concolor, P. ponderosa,* and *P. menziesii* were measured once at the end of the experimental exposure and the growing season. Biomass was measured twice on a separate set of seedlings: at the end of the experiment in the fall, and in the fall after 1 year of "recovery." The seasonal pattern of gas exchange and their relative susceptibility to ozone exposure of the five conifers was measured. For each species, gas exchange of current year foliage from the same three seedlings from each chamber was measured at the beginning of the exposure period in early June and subsequently at 1-month intervals through late September. CO_2 exchange rate (CER) and stomatal conductance (gs) were measured with a Licor 6200 photosynthetic system (Lincoln, Nebraska). A 0.25 L cuvette was used for all species except *P. ponderosa*. For this species, a custom cuvette (12 x 15 x 15 cm) was constructed from Acrylite lined with Teflon film so that shoot architecture was not disturbed during measurement. Leak rates of the custom cuvette were low and comparable to the purchased cuvette.

Gas exchange measurements were made outside fumigation chambers between 0930 and 1530 h. Plants of the same species were out of chambers approximately 1 h before measurement (Hinckley and others 1990) and were measured at the same time of day throughout the growing season. During all measurement sessions, plants from CF chambers were alternated with ozone chambers to nullify possible diurnal effects. In addition to biweekly watering, all plants were watered to capacity 0.5 d before measurements.

Gas exchange was measured within 1 minute. Gas exchange measurements for the ozone experiment were taken at ambient light within a photosynthetic photon flux density (PPFD) range of 750-1525 μmol m^{-2} s^{-1}, leaf temperature of 29-37 °C, and vapor pressure deficits (VPD) of 2.75-4.50 kPa. CO_2 exchange rate and stomatal conductance appeared to have a flat response across this range of physical factors, other than light, for which lower levels could have limited gas exchange up to 10 percent. Dark respiration rate of *A. concolor, P. ponderosa,* and *P. menziesii* was measured near the end of the season-long experiment, approximately 0.5 h after sunset. Measurements were taken within an air temperature range of 16-20 °C.

CER was calculated on the basis of needle surface area. Surface area was calculated from simple geometric models of leaf surfaces with consideration for surfaces with stomata. Measures of needle width (in the middle) or fascicle diameter (just above the sheath) and length were made with a digital micrometer.

Light response was determined near the end of the summer ozone exposure period. Four frames consisting of a light source, heat/light separation mirror, and neutral density filters were used to control light intensity (Grulke and others 1989). The same plants measured for seasonal response were also measured in this analysis. Plants were allowed to equilibrate for 0.5 h at each light level before gas exchange was measured. Gas exchange measurements were made between cuvette temperatures of 22-26 ˚C. For light response determinations of *P. ponderosa,* three fascicles were enclosed in a 0.25 L cuvette (spread to reduce self- shading). A simple leaf model was used to define the relationship: ($CER = P_{max} (1-e^{-¢I/Pmax})$, in which ¢ is apparent quantum efficiency and I is PPFD; Thornley 1976). The CO_2 light compensation point and CER with saturating light were calculated from the fitted regression equations for each plant.

Leaf Nitrogen

Leaf tissue was sampled for nitrogen content just after the monthly gas exchange measure by using a standard micro-Kjeldahl digest technique on dried tissue (65 °C, 24 h), ground to pass a 40 mesh. Digests were run on an auto-analyzer (Technicon) with standards, blanks, replicates, and duplicate runs every 20 samples. Samples with $\geq$ 2.5 percent error ([S.D./x] x 100 percent) were re-analyzed on a Carlo-Erba NA1500 Series 2 analyzer.

Biomass and Plant Size

Plant height (mm) and stem diameter (0.01 mm) was measured before experimental exposure for six additional seedlings per chamber. At the end of the experiment, plant height, stem diameter, and biomass allocation to roots, stem, older needles, and new (current year) needles were determined for the same six seedlings. The change in seedling height and stem diameter was relativized by using the initial measures taken before the experiment (Δ ht/ht_ix 100 percent). For yet another set of six seedlings per chamber, plant size and biomass allocation was measured after 1 full year of growth in a charcoal-filtered air glass house after experimentation.

Results

CO_2 Exchange Rate

Seasonal trends in CER were different for each species: maximum CER occurred at the first sampling date for *P. ponderosa,* at the second sampling date for *Abies lasiocarpa,* and at the last sampling date for *Abies concolor, Picea engelmannii,* and *Pseudotsuga menziesii* (*fig. 2*). The maximum observed CER varied nearly twofold for current year foliage between the species (*A. concolor,* 1.68$\pm$0.42; *A. lasiocarpa,* 1.42$\pm$0.08; *P. engelmannii,* 2.40$\pm$0.11; *P. ponderosa,* 1.52$\pm$0.11; *P. menziesii,* 2.50$\pm$0.22 [µmol CO_2 m^{-2} s^{-1}). Particularly low values of CER were obtained on the second sampling date for *P. menziesii* because of high vapor pressure deficits.

Only *P. ponderosa* showed significant effects of elevated ozone, which were measured at the second sampling date (*fig. 2*). Despite the few chambers used, several trends are apparent. Although initial CER in the elevated ozone treatment was sometimes greater than that of the charcoal filtered air treatment (e.g., *A. concolor, P. engelmannii*), the end of season mean CER was lower in the elevated ozone treatment for all species. After the exposure regime began, the largest differences in CER between the elevated ozone and charcoal filtered air treatments occurred early season for *A. concolor* and *A. lasiocarpa,* and midseason for *P. ponderosa*. We found no differences attributable to treatment for both *P. engelmannii* and *P. menziesii*.

Because of high within-chamber variability due to an uneven number of sensitive and tolerant individuals in each chamber, and because the seasonal gas exchange patterns were different for the plants within each chamber, significant differences were not found between treatments (except for *P. ponderosa*). For example, one *A. concolor* seedling had a positive parabolic pattern, one had a constantly declining seasonal gas exchange pattern, while the remaining four seedlings within a treatment had a negative parabolic pattern of seasonal gas exchange. The net effect of these differing seasonal patterns of gas exchange is a relatively flat seasonal response with high standard errors (see *A. concolor, fig. 2*). Neither condition (sensitivity or asynchronous seasonal patterns of gas exchange) could have been known before the experimental exposure.

Measured at the end of the experiment, dark respiration was higher in seedlings exposed to elevated ozone than charcoal filtered air treatments for all species, but differences were not statistically significant (*table 1*).

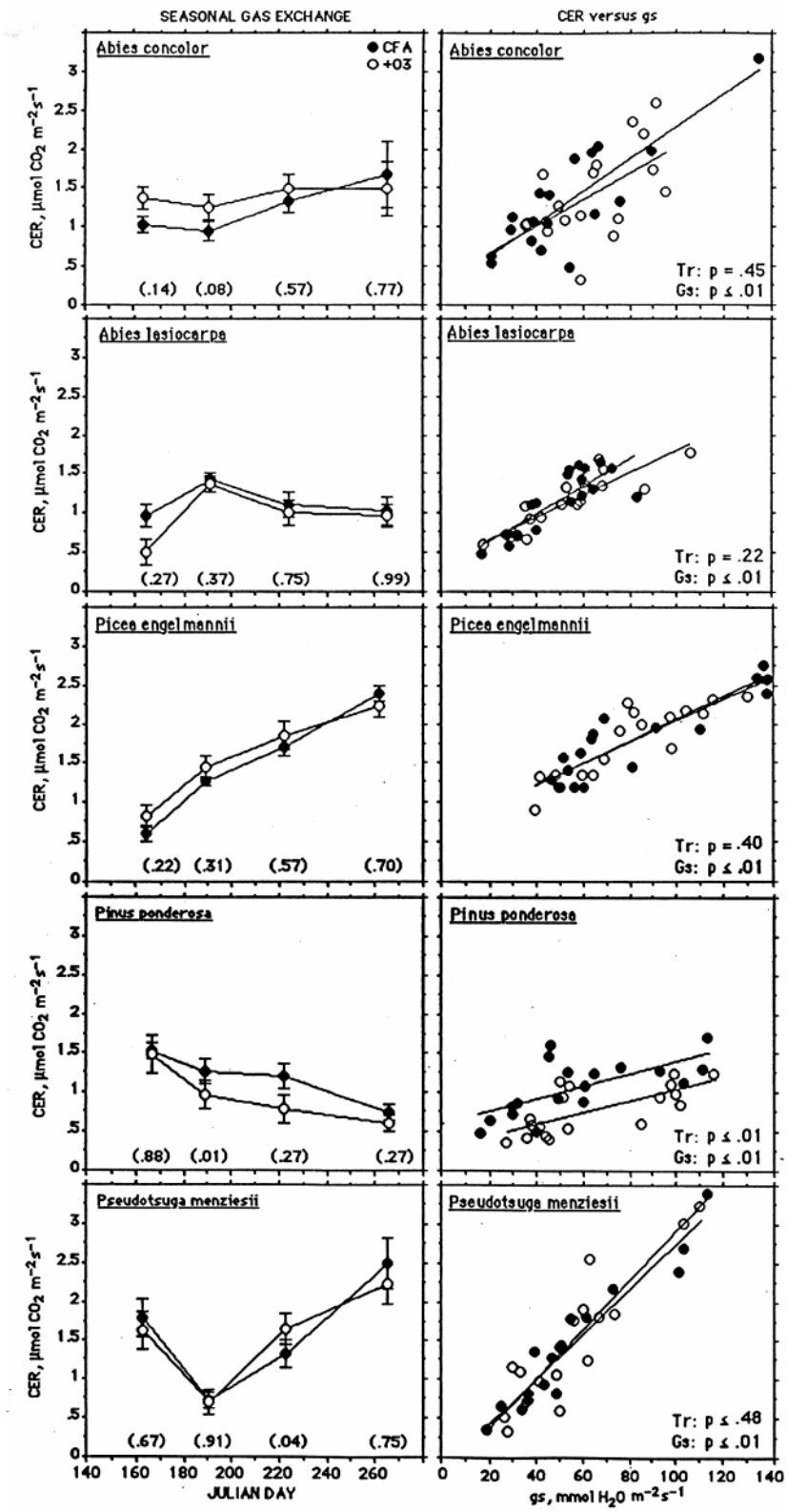

Figure 2 — Seasonal trend in CO_2 exchange rate (CER) for charcoal-filtered (CF) air (●) and elevated ozone treatments (O) for five species of western conifers. Symbols and error bars represent averaged means for the two chambers ± 1 S.E. Values in parentheses are probabilities that the two means are significantly different in a one-way ANOVA. The relationship between CO_2 exchange rate (CER) and stomatal conductance (gs) for the five species for month 2-4 of treatment is also given. Symbols represent average values for 2-4 runs per plant; in all cases, standard error was contained within the symbol. Values in lower right-hand corner are probabilities that the treatment (Tr) and stomatal conductance (gs) were significantly different. Slopes of the lines were not significantly different between treatments for any species.

Leaf Nitrogen

No seasonal trends in needle nitrogen content were apparent for any of the species. Needle nitrogen differed significantly between the charcoal filtered air and elevated ozone treatments averaged over the duration of the experiment for *A. concolor,* but not for the other species (*table 2*). CER based on leaf nitrogen content did not improve statistical separation of response to experimental treatment (*fig. 1*).

Stomatal Conductance

Seasonal patterns in stomatal conductance (gs) largely followed CER (*fig. 1*). However, after a month of treatment exposure, gs was generally greater in elevated ozone treatment for *A. concolor, A. lasiocarpa,* and *P. ponderosa* than in the charcoal-filtered air treatment. CER was plotted as a function of gs to further test for the effects of elevated ozone exposure *(fig. 1)* for months 2 to 4 of treatment exposure. The slopes of the regressions were not significantly different for four of the species (Draper and Smith 1981). For *P. ponderosa*, CER was significantly reduced in the elevated ozone treatment relative to the charcoal filtered air treatment at the same gs, suggesting reduced water use efficiency. The differences in CER were greater than differences in dark respiration (*table 1*).

Light Response

Light compensation points were not significantly different between treatments for *A. concolor, P. ponderosa,* or *P. menziesii* (*table 3*) using a one-way ANOVA. CER at light saturation (1,600 µmol $m^{-2}s^{-1}$) was significantly different between treatments for *P. ponderosa,* but not for *A. concolor* or *P. menziesii* (*table 3*). The apparent quantum efficiency (the slope of the regression line between 0 and 100 µmol quanta m^{-2} s^{-1}) for *P. ponderosa* seedlings growth in charcoal-filtered air and elevated ozone was statistically significant ($p \leq 0.01$) (Draper and Smith 1981). The difference in light response between treatments in *P. ponderosa* was much greater than the difference in dark respiration observed (*table 1*).

Biomass Allocation

Biomass allocation to roots, stems, old needles (produced before this experiment), and new needles (current year) was analyzed for seedlings immediately after the growing season, and 1 full year after experimental exposure (*fig. 3*). Some species

Table 1 *— Dark respiration (µmol CO_2 m^{-2} s^{-1}) at the end of the experiment for charcoal-filtered (CF air) and elevated ozone (+ O_3) treatment for the five conifer species studied.*

Species	CF air [1]	+ O_3 [1]	p [2]
Abies concolor	0.21±00	0.22±0.02	(0.52)
Abies lasiocarpa	0.16±0.01	0.17±0.02	(0.72)
Picea engelmannii	0.15±0.01	0.16±0.01	(0.14)
Pinus ponderosa	0.20±0.04	0.24±0.03	(0.50)
Pseudotsuga menziesii	0.30±0.03	0.34±0.00	(0.24)

[1] Mean and error represent averaged means for the two chambers and within-chamber standard error in each treatment.

[2] Probability for significant differences between values (one-way ANOVA).

Table 2 *— Leaf nitrogen (percent) averaged for the four sampling points throughout the duration of the experiment for charcoal-filtered (CF air) and elevated ozone (+ O_3) treatment.*

Species	CF air [1]	+ O_3 [1]	p [2]
Abies concolor	1.38±0.08	0.98±0.01	(±.01)
Abies lasiocarpa	1.27±0.06	1.14±0.04	(0.11)
Picea engelmannii	1.84±0.10	1.90±0.11	(0.72)
Pinus ponderosa	1.20±0.06	1.15±0.05	(0.51)
Pseudotsuga menziesii	1.11±0.08	1.13±0.12	(0.89)

[1] Mean and standard error.

[2] Probability for significant differences between values (one-way ANOVA).

showed adverse ozone effects on biomass or growth in the year of exposure (*A. lasiocarpa, P. ponderosa*), some showed only adverse effects in the year after exposure (*A. concolor, P. engelmanii, P. menziesii)*. There were few consistent patterns in which plant component was affected, either in the year of exposure or the year after exposure. For all species, the percent biomass allocation to older needles was lower for seedlings grown in elevated ozone chambers relative to charcoal-filtered air chambers, but the differences were not significant (*table 4*). Total biomass was lower for seedlings in elevated ozone than charcoal-filtered chambers immediately after exposure for all species except *P. ponderosa,* where seedlings were larger (+10.6 percent in total biomass, and 10.1 percent in dia.) at the end of the experimental exposure. Plants in the elevated ozone treatments were also larger in May before experimental exposure. Two species also had lower biomass allocation to new needles (*A. lasiocarpa* (p=0.01), *P. menziesii,* p =0.57), but *A. concolor, P. engelmanii,* and *P. ponderosa* had greater allocation to new needles.

Discussion

In this study testing the sensitivity of five western conifers to ozone, *P. ponderosa* appeared to be the most sensitive to ozone on the basis of reductions in CER (in midseason, CER versus gs responses, and in light saturation experiments at the end of the growing season). Among all of the species, *P. ponderosa* had the greatest gs early in the season when young leaf tissues were most likely to be damaged by oxidants. In general, species of pine appear to be more sensitive to ozone exposure than any other conifer (Bytnerowicz and Grulke 1992).

Although *A. lasiocarpa* had a seasonal pattern of gs similar to that of *P. ponderosa,* few symptoms of sensitivity to ozone developed. In this study, as well as in their natural habitat, CER and gs of *A. concolor* (Conard and Radosevich 1981) and *P. engelmannii* (Carter and others 1988, DeLucia and Smith 1987) increased gradually through the growing season. Comparisons of instantaneous gas exchange measures showed that both species appear to be relatively resistant to ozone exposure (*fig. 1*).

Instantaneous measures of gas exchange as well as biomass of *P. menziesii* did not statistically demonstrate evidence of damage by exposure to ozone. However,

Table 3 *— Light compensation point and CO_2 exchange rate at the end of the exposure for charcoal-filtered air (CF air) and elevated ozone (+ O_3) treatments.*

Species	CF air [1]	+ O_3 [1]	p [2]
Light compensation point [3]			
Abies concolor	28 ± 5	28 ± 4	(0.98)
Pinus ponderosa	47 ± 5	49 ± 6	(0.86)
Pseudotsuga menziesii	20 ± 4	19 ± 4	(0.89)
CO_2 exchange rate at light saturation [4]			
Abies concolor	3.17 ± 0.34	3.21 ± 0.36	(0.93)
Pinus ponderosa	2.67 ± 0.19	1.21 ± 0.24	(0.001)
Pseudotsuga menziesii	4.84 ± 0.51	5.45 ± 0.39	(0.37)

[1] Mean and error represent averaged means for the two chambers and within-chamber standard error in each treatment.

[2] Probability for significant differences between values (one-way ANOVA).

[3] μmol quanta m^{-2} s^{-1}.

[4] μmol CO^2 m^{-2} s^{-1}.

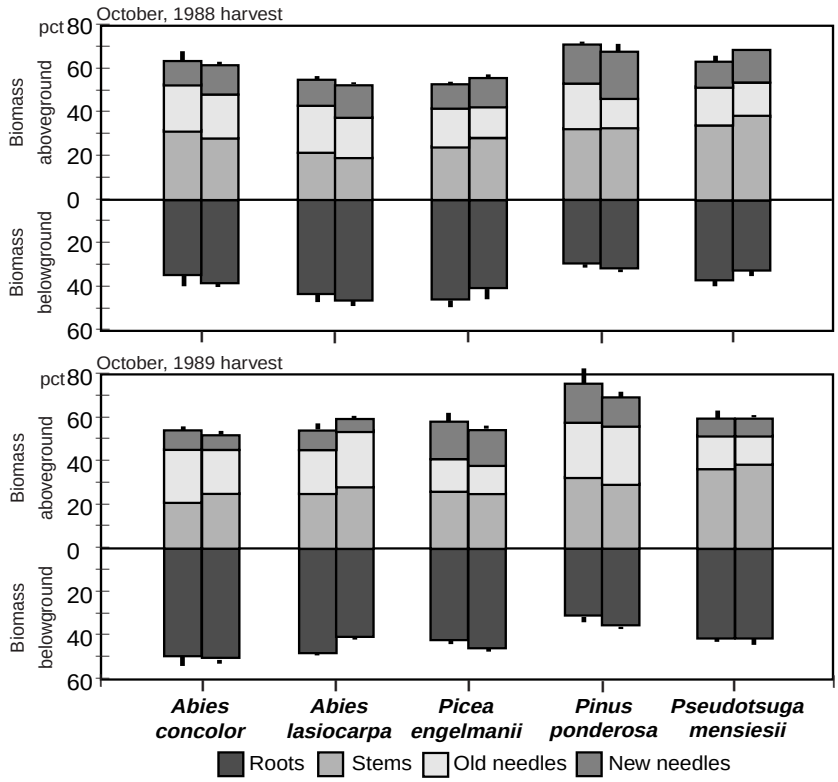

Figure 3 — Change of biomass allocation by percent for seedling growth in charcoal-filtered air (first column) compared to those in elevated ozone chambers (second column). Two sets of seedlings were harvested: (a) immediately after the growing season of experimental exposure and (b) after 1 full year of growth in a glass house supplied with charcoal-filtered air. Bar lengths represent the means of the two chambers and ± 1 S.E.

all species showed a trend of increased instantaneous measures of dark respiration, and subsequent decreased biomass of older needles (increased needle turnover rates) immediately after the experiment. Most species (except *P. menziesii*) showed a trend of decreased biomass of new needles 1 full year after experimental exposure. In another study of *P. menziesii* exposed to elevated ozone in Washington, measures of instantaneous gas exchange did not reflect significantly reduced stem diameter and root to shoot biomass at the end of a 2-year study (Hinckley and others 1990).

Instantaneous values are appropriate to understand what plant processes may be sensitive to oxidants, but measuring long-term biomass changes may be a better measure to detect damage. Modeling efforts that incorporate the small differences in gas exchange and subsequent carbon allocation within the plant year after year may demonstrate the consequences of subtle changes in carbon balance more clearly.

Few of the differences observed between the charcoal-filtered air and the elevated ozone treatments were significant. The lack of statistical significance was primarily because of high within-chamber variance. Even if chamber effects were assumed to be null and each plant were treated as a true replicate, there were few additional statistically significant differences between treatment means. In many pollutant exposure studies, plants are tested after the maximum cumulative exposure period, often at the end of a growing season, when gas exchange rates are declining and the ability to statistically distinguish between means is the least powerful. In this study, the statistically significant differences between the charcoal filtered air and elevated ozone treatments occurred in the second month of exposure for *P. ponderosa* (*fig.1*). Cumulative exposure to ozone was not necessarily correlated to peak deleterious effects on gas exchange in other studies (Byres and others 1992, Weber and others 1993). In this study, high variability in plant response within a chamber and too few chambers significantly hindered the evaluation of the effects of exposure to elevated ozone.

To better clarify the lack of statistical significance, and assuming the same within- chamber standard error, the difference between treatment means required for significance at the $p = 0.05$ level, or the number of chambers of plant gas exchange that would have been required for the means to have significance was calculated (*table 5*). Untenably large numbers of chambers would

have been required to demonstrate statistical significance between treatments for all species other than *P. ponderosa.*

One source of high population variance may be naturally occurring genotypes that are "sensitive" or "resistant" to pollutants (Ernst and others 1985, Taylor 1978). Evidence for differences in population sensitivity to ozone was found in different individuals of *P. jeffreyii* (Patterson and Rundel 1989), different seed sources of

Table 4 *— Magnitude and direction of change of biomass allocation by percent for seedlings grown in charcoal filtered air compared to those in elevated ozone chambers. Two sets of seedlings were harvested: (A) immediately after the growing season of experimental exposure and (B) after 1 full year of growth in a glasshouse supplied with charcoal-filtered air.*

A. 1988 October Harvest

	Abies concolor	*Abies lasiocarpa*	*Picea Engelmannii*	*Pinus ponderosa*	*Pseudotsuga menziesii*
biomass	[1] - 1.80 (0.91)	- 15.6 (0.12)	- 25.0 (0.35)	+ 10.1 (0.51)	- 14.0 (0.48)
height	+ 3.3 (0.85)	+ 23.4 (0.78)	+ 23.8 (0.71)	+ 8.9 (0.76)	- 18.6 (0.31)
diameter	+ 21.6 (0.25)	+ 8.3 (0.79)	- 31.7 (0.72)	[2] + **10.1** (0.01)	- 18.2 (0.27)
root	+ 5.3 (0.63)	+ 5.7 (0.47)	- 13.9 (0.32)	+ 7.1 (0.27)	- 17.3 (0.39)
stem	- 9.6 (0.17)	- 8.7 (0.38)	+ 14.1 (0.30)	+ 1.3 (0.89)	- 14.2 (0.41)
old needles	- 7.1 (0.51)	- 17.3 (0.35)	- 11.1 (0.67)	- 50.2 (0.23)	- 17.8 (0.46)
new needles	+ 17.0 (0.62)	[2] **- 15.5** (0.01)	+ 27.0 (0.25)	+ 19.5 (0.28)	- 8.8 (0.57)

B. 1989 October Harvest

	Abies concolor	*Abies lasiocarpa*	*Picea Engelmannii*	*Pinus ponderosa*	*Pseudotsuga menziesii*
biomass	[2] + **29.5** (0.01)	0.0 (0.99)	- 10.3 (0.61)	+ 50.7 (0.10)	+ 10.0 (0.30)
height	+ 11.6 (0.10)	+ 12.0 (0.36)	+ 8.2 (0.56)	+ 26.9 (0.16)	+ 11.1 (0.23)
diameter	[2] + **14.9** (0.03)	+ 0.8 (0.91)	- 3.2 (0.78)	+ 9.8 (0.09)	+ 19.3 (0.23)
root	- 3.5 (0.77)	- 13.8 (0.10)	[2] + **8.6** (0.23)	+ 7.2 (0.63)	+ 0.5 (0.96)
stem	+ 14.4 (0.28)	+ 11.4 (0.15)	- 4.6 (0.37)	- 7.7 (0.32)	+ 7.0 (0.52)
old needles	- 21.9 (0.27)	+ 18.9 (0.29)	- 14.3 (0.44)	+ 3.3 (0.89)	[2] **- 22.8** (0.02)
new needles	[2] + 23.4 (0.40)	- 59.8 (0.38)	- 5.3 (0.71)	- 26.3 (0.86)	+ 10.8 (0.81)

[1] Parenthetical values are the probability that the change is significant between treatments (chamber means of six seedlings were the replicates, df=1). Bold indicates values significant at $p<0.05$.

[2] Greater allocation to a tissue type in elevated ozone versus charcoal filtered chambers.

Table 5 *— Evaluation of statistical significance for CO_2 exchange from* figure 1. *The differences between treatment means or the number of chambers that would have been required for the means to have been significantly different at the p=0.05 level have been calculated, assuming the same population variance.*

Species	p[1]	Date of comparison	Difference in mean required	No. of chambers required
CO_2 exchange				
Abies concolor	(0.77)	9/22	1.8x	500
Abies lasiocarpa	(0.99)	9/22	5.2x	330
Picea engelmannii	(0.70)	9/19	4.0x	190
Pinus ponderosa	(0.27)	8/10	1.1x	2
Pseudotsuga menziesii	(0.75)	9/21	3.5x	140

[1] Parenthetical values are probability of significant differences between gas exchange values from *figure 1.*

P. strobus (Barnes 1972), and in different families of *P. ponderosa* (Beyers and others 1992). In addition to genetic differences, plant water status, leaf tissue nutrient content, and different seasonal patterns of gas exchange or phenology may alter individual plant sensitivity to ozone and contribute to the lack of statistical power to detect treatment effects or to resolve discrepancies between published studies.

Acknowledgments

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Mature Ponderosa Pine Nutrient Use and Allocation Responses to Air Pollution[1]

Mark A. Poth and Mark E. Fenn[2]

Abstract

*Current-year needles from mature ponderosa pine (*Pinus ponderosa *Dougl. ex. Laws.) were sampled at four sites across the air pollution gradient in the San Bernardino Mountains in southern California. The sites, in order of decreasing air pollution exposure, included: Sky Forest (SF), Conference Center (CC), Camp Angelus (CA) and Heart Bar (HB). Needle nutrients measured were: nitrogen (N), phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), manganese (Mn), zinc (Zn), iron (Fe) and aluminum (Al). Concentrations of N in foliage (12.5 g/kg) were significantly higher at SF, the site most exposed to air pollution. This level is similar to concentrations for container-grown seedlings fertilized with nutrient solution. Mature tree needle concentrations of K, Mg , Fe and Al were higher at sites more exposed to air pollution. Mature trees with two annual whorls of needles (more severely injured) had significantly higher current year needle concentrations of P, K, Zn, and Fe at SF than trees with three annual whorls of needles (less severely injured). However, significant differences were not found in foliar nutrient concentrations between trees with different levels of needle retention (injury) at CC, which is also a high pollution site. At Barton Flats (BF), a low to moderate air pollution site in the San Bernardino Mountains, N, K and P were measured in current-year needles and litter fall. The resorption efficiencies (the percent reduction in nutrient concentration upon senescence of leaves or needles) for mature trees at BF were: 45 percent for N, 60 percent for P and 80 percent for K. N resorption proficiency (the concentration a nutrient is reduced to before needle abscission) was high with N reduced to 4.4 g-N/kg (considerably less than the defined threshold of 7g-N/kg). However, P resorption proficiency was very poor with P reduced to 0.59 g-P/kg (higher than the defined threshold of 0.4 g-P/kg). A complete understanding of the changes in tree nutrition associated with air pollution will require more research.*

Introduction

Long-term chronic exposure to ozone and other associated air pollutants has a pronounced effect on growth and mortality of ponderosa pine (*Pinus ponderosa* Dougl. ex. Laws.). This has been clearly documented (Miller and others 1963, Miller and others 1989) over the last 25 years in the San Bernardino Mountains of southern California. The San Bernardino and San Gabriel Mountains form the transverse ranges, which run from west to east and form the northern boundary of the South Coast Air Basin. The Basin includes Los Angeles and the surrounding communities. Vegetation in the San Bernardino Mountains varies with elevation and aspect. Scrub and chaparral species dominate the lower elevations (1,200-1,600 m) on the western and southern ends of the range, while the pinyon-juniper type is found on the desert-facing eastern and northern extremes of the San Bernardino Mountains. The mixed conifer forest occurs at elevations of about 1,500 to 2,400 m. Ponderosa pine is the dominant pine species of the mixed conifer forest, except on the eastern end of the San Bernardino Mountains where ponderosa pine is replaced by the closely-related Jeffrey pine (*Pinus jeffreyi* Grev. & Balf.). Visible indications of ozone stress in ponderosa pine in the mixed conifer zone include a combination of reduced needle retention, thinning of the lower crown, chlorotic mottle of foliage, the production of shorter needles, and mortality—typically as the result of an insect attack (Miller 1983).

A major research effort in Germany has shown that forest decline in a spruce forest was caused by nutrient imbalances from atmospheric deposition of sulfur (S), and nitrogen (N). (Huettl and others 1990, Schulze 1989). Magnesium (Mg) was retranslocated from old needles to new foliage until luxury levels were reached in the current-year foliage, and Mg deficiency levels occurred in the older and subsequently-chlorotic foliage. Although Mg levels were in excess

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, Riverside, California, February 5-9, 1996.

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in current-year foliage, N levels in new foliage became deficient, notwithstanding the excessive deposition of N (Schulze 1989). Similarly, given the dramatic effects of ozone and associated air pollutants in the San Bernardino Mountains on foliar injury, growth, and abscission in ponderosa pine (Miller 1983), it is likely that nutrient relations of these ponderosa pine trees are also significantly affected.

Despite the importance of ponderosa pine in western North America, information on ponderosa pine nutrition is scant. Available data are restricted to studies of N fertilization or the responses of seedlings or unique site situations, some of which are thought to define the limits of species range. Rarely is any nutrient other than N and P examined. This is a reflection of the pattern of use for the ponderosa pine resource. Historically, ponderosa pine has been harvested primarily from old-growth stands. Second-growth from natural regeneration can take 80 to 100 years for trees to reach marketable size. Although silviculturalists have successfully developed methods for ponderosa pine regeneration, specific knowledge of nutrient use and requirements remains poor (Powers 1981), particularly compared to the available information for more intensively managed and shorter rotation species like Douglas fir. A search of the National Agricultural Library database yields 1,086 references with ponderosa pine in the title or as a key word and 2,344 for Douglas fir. Further examination of the ponderosa pine citations reveals the paucity of nutritional data.

In this study we measured and compared foliar nutrients at four sites along the ozone gradient in the San Bernardino Mountains. We present data on foliar nutrient concentrations, and relate the concentrations to visible ozone symptoms (needle retention) in ponderosa/Jeffrey pine. Our objectives were to determine if foliar nutrient levels might be related to the severity of ozone stress incurred by mature trees, and to help define the range of nutrient concentrations seen in mature ponderosa/Jeffrey pine. In addition, we have examined the influence of air pollution on nutrient use by mature and container-grown ponderosa pine by comparing nutrient concentrations in needles and litter. The nutrient resorption efficiencies (the percent reduction in nutrient concentration upon senescence of leaves or needles) and resorption proficiencies (the concentration a nutrient is reduced to before needle abscission) determined for ponderosa pine are also reported.

Materials and Methods

Needle samples were collected from four plots located along a west to east ozone concentration gradient spanning 55 km in the San Bernardino Mountains. The plots were originally established in 1972-73 as part of a large interdisciplinary study of the effects of air pollution on the mixed conifer forest in the San Bernardino Mountains (Miller 1983). We sampled trees in the plots, in order of decreasing exposure, at Sky Forest (SF), Conference Center (CC), Camp Angelus (CA), and Heart Bar (HB). Ozone concentrations in the permanent plots were measured during July and August from 1974-78. The respective estimated 24-hour average ozone concentrations ($\mu g/m^3$) at the sites were 196 at SF and CC, 157 at CA, and 59 at HB (Miller and others 1989). Ozone concentrations have decreased slightly during the 1980s, but concentrations within the greater Los Angeles basin remain among the highest in the nation. Visible symptoms of ozone injury were still prominent at SF and CC, and to a lesser degree at CA.

We selected dominant and codominant trees for needle sampling. A minimum of 24 mature ponderosa pine trees with a dbh of at least 30 cm were sampled at each of the four sites, except at HB where Jeffrey pine was sampled. Trees with mistletoe, pest problems or obvious abnormalities were avoided. Current-year foliage was collected from exposed branches (one branch per tree) in the middle crown in late August of 1986. For each tree sampled we also recorded the number of annual whorls of needles and the presence of needle chlorotic mottle characteristic of ozone damage.

Sampled needles were dried and ground for chemical analysis. N and P were determined by standard colorimetric methods following Kjeldahl digestion. Calcium (Ca), magnesium (Mg), potassium (K), manganese (Mn), zinc (Zn), iron (Fe), and aluminum (Al) were analyzed by atomic absorption spectrometry after perchloric acid digestion of tissues. All analyses were done in the laboratories of the Institute of Soil Science and Forest Nutrition, Albert-Ludwigs University, in Freiburg, Germany.

In a separate study at Barton Flats in the San Bernardino Mountains, needles were collected in August of 1993 from mature ponderosa pines. This was done in late summer/early fall when new needles were fully expanded. More than 100 trees were sampled. Needles were divided into age classes, oven dried at 70 °C and ground to pass through a 40-mesh screen. Needles from each tree were analyzed for Ca, Mg, K, Mn, and Zn by atomic adsorption spectroscopy after perchloric - nitric acid digestion. Phosphorus (P) was determined from the same digest (Glaubig and Poth 1993). Coupled combustion - gas chromatographic analysis was used to measure total N and carbon (C). Older age classes of needles were only analyzed for total C and N.

Pine litter was collected during the fall from litter traps at Barton Flats. After collection, combined samples (all fall collection dates) were oven dried at 70 °C and ground to pass through a 40 mesh screen. Pine litter samples were also analyzed for C, N, P, Ca, Mg, K, Mn, and Zn. For mobile nutrients, nutrient efficiency (the percent reduction in nutrient concentration upon senescence of leaves or needles) was calculated.

Results and Discussion

Foliar concentrations of K and Mg were highest at the high pollution sites SF and CC (*fig. 1*). Foliar N was significantly higher at SF than at any other site (*fig. 2*). Nutrients did not appear to be severely limiting for mature ponderosa pine, as was reported in a previous study (Huettl and others 1990, Powers 1991).

Critical levels of N and P in ponderosa pine foliage (Powers 1981) are reportedly 9 and 0.8 g/kg. Below these levels the N and P supply becomes deficient for adequate growth. Foliar concentrations of K, Mg, and Ca in this study appear to be within the range of values for ponderosa pine reported in the literature (Clayton and Kennedy 1980). Levels of Fe and Al were highest in the two high-pollution sites (SF and CC), and Mn levels were very high at CC compared to the other sites (*fig. 3*). Zn was highest at HB, but showed no pattern of variation across the ozone gradient. Micronutrient levels are within the normal ranges reported for ponderosa pine (Powers 1981).

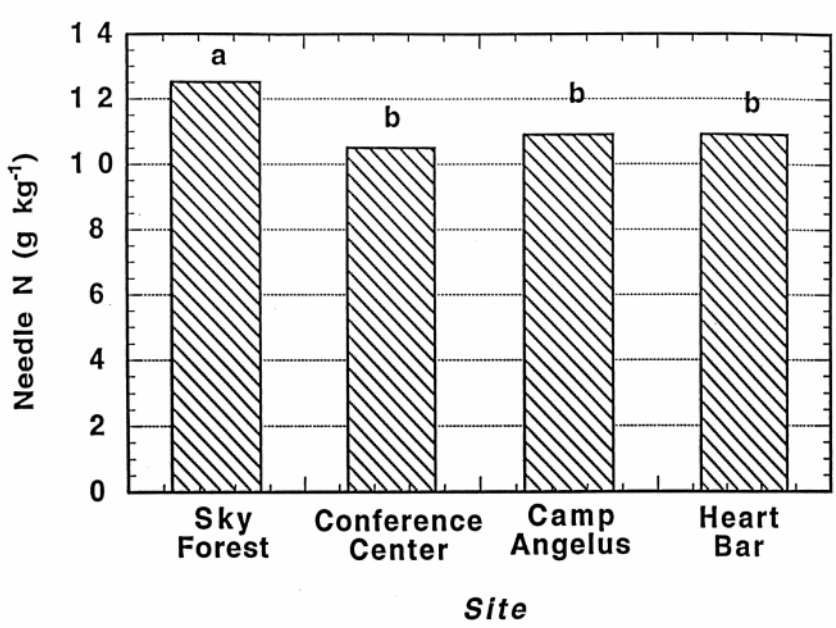

Figure 1 — Macronutrient concentrations for current-year foliage of mature ponderosa pine trees in the San Bernardino Mountains. Letters associated with bars indicate significant differences in foliar concentrations between the plots according to Tukey's multiple comparison test (p=0.05).

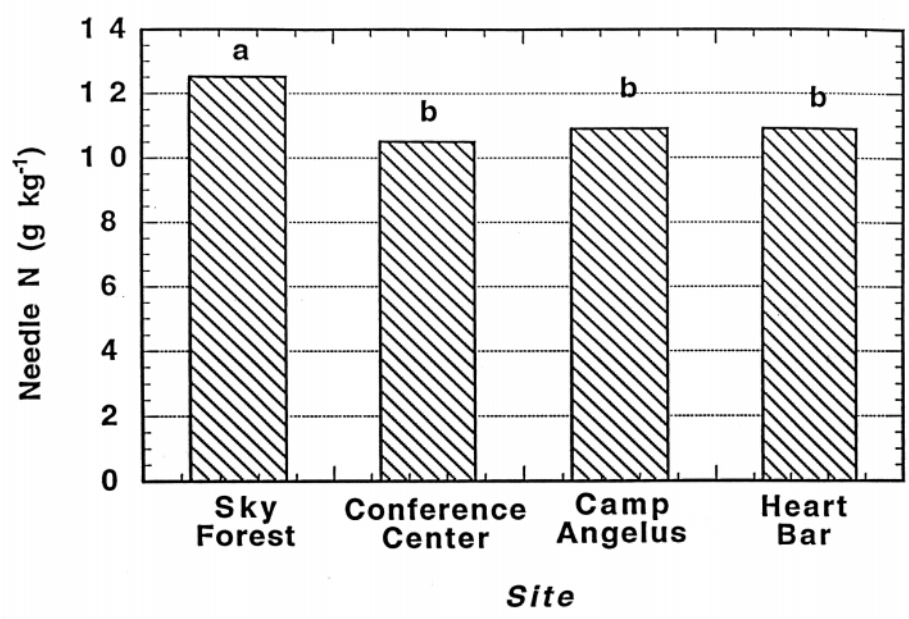

Figure 2 — Nitrogen concentrations for current-year foliage of mature ponderosa pine trees in the San Bernardino Mountains. Letters associated with bars indicate significant differences in foliar concen-trations between the plots according to Tukey's multiple comparison test (p=0.05).

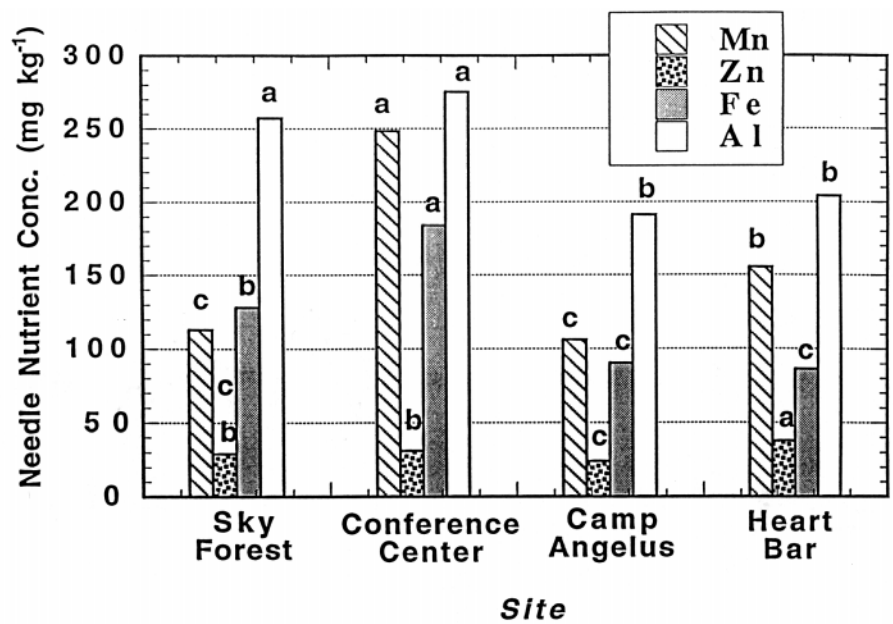

Figure 3 — Micronutrient concentrations for current-year foliage of mature ponderosa pine trees in the San Bernardino Mountains. Letters associated with bars indicate significant differences in foliar concentrations between the plots according to Tukey's multiple comparison test (p=0.05).

Ozone stress is the main inciting factor leading to premature needle abscission of ponderosa pine in the high-pollution sites in the San Bernardino Mountains. Other factors, such as drought stress, could also reduce needle retention. However, the high-pollution sites receive higher levels of precipitation (Miller and others 1989), yet needle retention is much greater in the drier, less-polluted sites. Thus, the number of annual whorls of needles retained, especially when accompanied with symptoms of chlorotic mottle, is considered a reliable measure of the level of ozone stress affecting ponderosa pine. In some cases there are statistically significant differences in nutrient concentrations between trees retaining 2 or 3 years of foliage at the two high pollution sites. Trees with only two annual whorls (the current-year and the previous year) at SF have higher needle concentrations of P, K, Zn, and Fe than trees with three annual whorls (*fig. 4, 5,* and *6*); presumably the higher concentrations in the current-year whorls are the result of ozone-induced senescence and nutrient retranslocation.

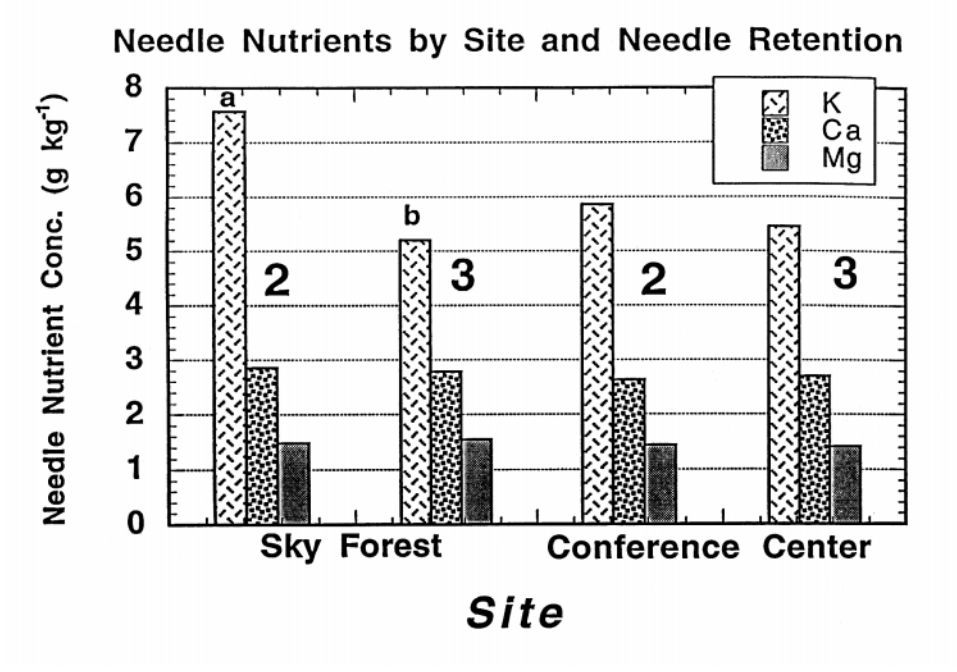

Figure 4 — Macronutrient (K, Ca, and Mg) concentrations for current-year foliage of mature ponderosa pine trees in the San Bernardino Mountains retaining two or three years of foliage. Letters associated with bars indicate significant differences in foliar concentrations between trees at the same plot according to Tukey's multiple comparison test (p=0.05).

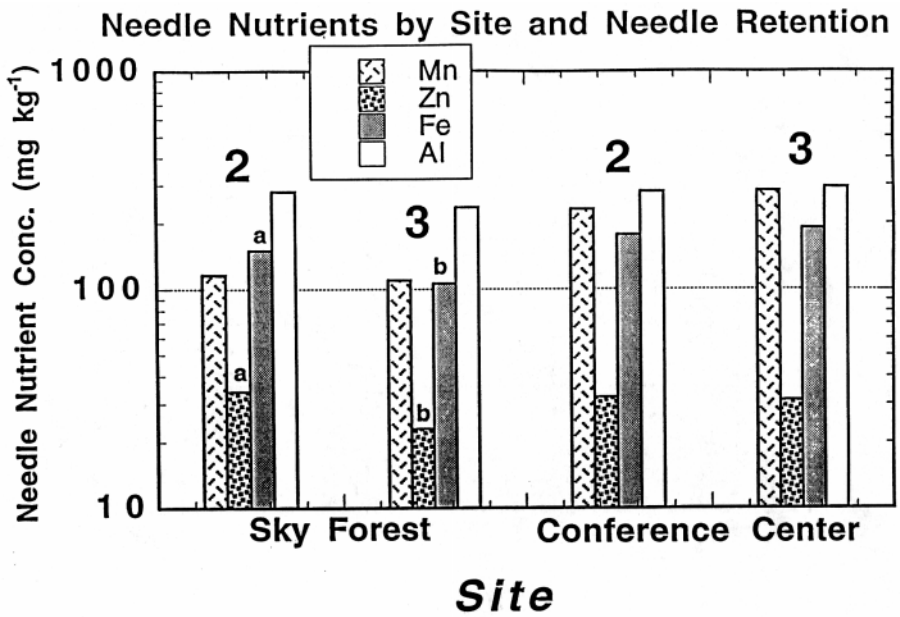

Figure 5 — Micronutrient concentrations for current-year foliage of mature ponderosa pine trees in the San Bernardino Mountains retaining two or three years of foliage. Letters associated with bars indicate significant differences in foliar concentrations between trees at the same plot according to Tukey's multiple comparison test (p=0.05).

In a previous study at Strawberry Peak, a site 4 km west of SF, current-year foliage of ponderosa pine trees retaining only one annual whorl contained significantly higher levels of P than foliage of trees retaining two annual whorls (Fenn 1991). Phosphorus and potassium are highly mobile and are readily remobilized in plants. Zinc is considered to be intermediate in terms of plant mobility (Epstein 1972, Salisbury and Ross 1985). Zinc remobilization is closely related to leaf senescence (Marschner 1986). Iron generally shows low to intermediate plant mobility (Epstein 1972, Salisbury and Ross 1985). The retranslocation of micronutrients is markedly increased during foliar senescence (Marschner 1986).

We did not detect significantly higher nutrient concentrations in current-year foliage of trees with either two or three annual whorls at CC. Although estimated ozone concentrations at SF and CC were reported to be equivalent based on limited data (Fenn 1991), the severity of ozone damage to ponderosa pine was greater at SF than at CC (Miller and others 1989). This difference may explain why higher

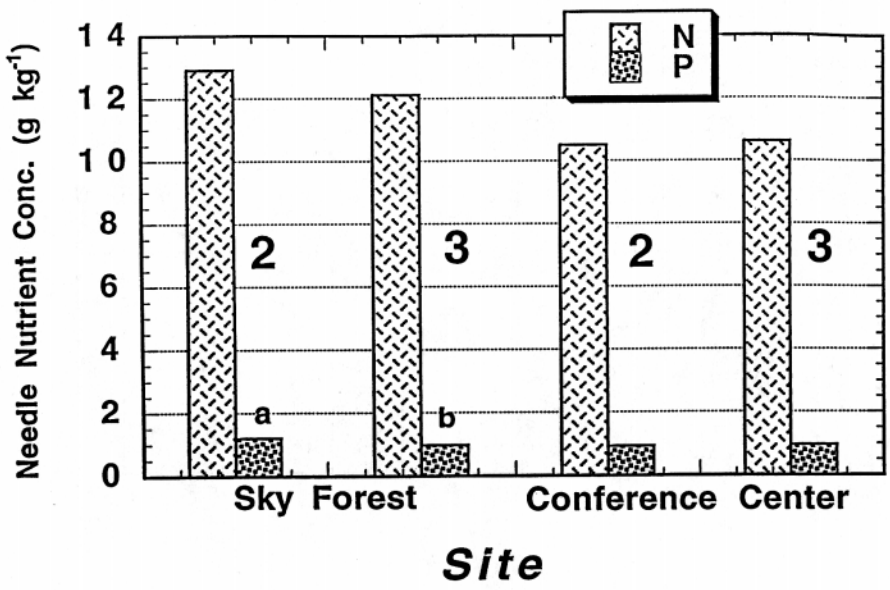

Figure 6 — Nitrogen and phosphorus concentrations for current-year foliage of mature ponderosa pine trees in the San Bernardino Mountains retaining two or three years of foliage. Letters associated with bars indicate significant differences in foliar concentrations between trees at the same plot according to Tukey's multiple comparison test (p=0.05).

levels of nutrients were not found in foliage of trees with fewer annual whorls at CC, contrary to the results at SF in this study, and in an earlier study at Strawberry Peak in the San Bernardino Mountains (Fenn 1991). More severe ozone damage at SF may be a result of greater water availability, resulting in greater stomatal uptake of ozone. Available soil water in the top 90 cm (19 and 13 percent) and annual precipitation from 1973 to 1977 (109 and 90 cm) were higher at SF than at CC (Miller and others 1989). Ambient temperature is not expected to vary greatly between CC and SF based on their proximity and similar elevations (1,705 m at SF and 1,610 m at CC; Miller and others 1989).

In a previous study in the San Bernardino Mountains, N and P retranslocation in ponderosa pine occurred just before or during needle senescence and death (Fenn 1991). In the present study we sampled foliage in August, during the summer growing season. A considerable amount of nutrient retranslocation probably occurred later in fall. However, where the ozone stress is sufficient to result in premature needle loss, nutrient retranslocation may occur earlier in the season. The timing of nutrient retranslocation may also differ for the various mobile elements. Had we collected foliage later in fall, concentrations of other mobile elements may have also increased in current-year foliage, especially in trees severely damaged by ozone. N and Mg were not higher in new foliage of trees with only two annual whorls. Because of the mobility of N and Mg in plants (Salisbury and Ross 1985), they would also be expected to retranslocate to younger foliage before abscission of older needles. Retranslocation of N at the high-pollution sites may be reduced because of a fertilizer effect of high N deposition (Fenn and Bytnerowicz 1993). Reduced retranslocation of N from senescing foliage in response to fertilization has been observed in several tree species (Flanangan and Van Cleve 1983).

Increased nutrient uptake by ozone-impacted trees could also account for the increase in nutrient concentrations in trees retaining only two annual whorls. This seems unlikely, however, given that ozone has been shown to reduce root growth and root carbohydrate content in ponderosa pine (Andersen and others 1991). The greatly reduced foliar biomass of ozone-stressed ponderosa pine is likely to result in reduced total photosynthetic rates in the long term; although younger foliage has been shown to compensate, at least in the short term, for the loss of older foliage in ozone-stressed ponderosa pine seedlings (Beyers and others 1992). Assuming that total photosynthate production is reduced in mature ozone-stressed ponderosa pine trees, it is doubtful that increased nutrient uptake occurs in ozone-stressed trees.

Which nutrients have been conserved, presumably by retranslocation, could be influenced by the supply of nutrients in the soil. P, K, Zn, and Fe were all found in higher concentrations in current-year foliage of trees at SF that retained only two annual whorls of needles. No data are available for soil concentrations of Zn or Fe in the four plots. P levels in soil are similar in the high- and moderate-ozone sites in the San Bernardino Mountains, but are significantly lower in the eastern low-ozone sites (Fenn 1991). In the original studies in the 18 permanent plots established along the ozone gradient, the soils at SF, CC and HB contained 0.49, 0.57, and 0.52 milliequivalents of K per 100 g soil (Arkley and others 1977). Soil at HB contains only 0.88 milliequivalents of Mg per 100 g, which is also reflected in the lower Mg concentrations in foliage at HB. It is unlikely that the apparently slight differences in soil P and K concentrations that occur at SF and CC account for why at SF, but not at CC, new foliage of trees with two annual whorls had higher nutrient levels than trees with three annual whorls. Apparently ozone stress (as indicated by needle retention), and not soil deficiency, is the primary factor that causes enhanced nutrient retranslocation in the more severely ozone-impacted trees at SF.

It may be a general phenomenon in perennial plant species sensitive to ozone that ozone exposure results in enhanced nutrient retranslocation to younger foliage, especially under nutrient-limiting conditions. The concentration of N and S in current-year foliage of loblolly pine (*Pinus taeda* L.) was higher in seedlings exposed to ozone than in plants exposed to sub-ambient ozone levels (Tjoelker and Luxmoore 1991). Phosphorus concentrations in foliage also increased ($p=0.10$) in seedlings exposed to ozone; and K tended to increase, but differences were not significant (Tjoelker and Luxmoore 1991). Leaves from cottonwood (*Populus deltoides* J. Bartram ex Marsh.) saplings exposed to ozone abscised prematurely and had higher concentrations of N than control plants (Findlay and Jones 1990).

Higher nutrient concentrations in current-year foliage of ponderosa pine with prematurely-abscised needles is indicative of how ponderosa pine trees respond to ozone stress. Before total senescence and abscission of older foliage, mobile nutrients are retranslocated to younger foliage, which then assume a larger physiological role. After two seasons of ozone exposure, ponderosa pine seedlings compensated for loss of older foliage by increasing net assimilation rates and N concentration of current-year foliage (Hom and Reichers 1991). Nutrient retranslocation to current-year foliage, and enhanced productivity of new foliage appear to be key adaptive mechanisms in ponderosa pine for overcoming the ozone-induced loss of productivity from older foliage.

Another important consideration of plant nutrient relations and ozone exposure is the influence of plant nutrient status on sensitivity to ozone. The co-occurrence of chronic N deposition in many areas with elevated ozone concentrations further emphasizes the importance of investigating the interactions and relationships between atmospheric deposition, soil-plant nutrient relations, site fertility, and plant sensitivity to ozone. For instance, Tjoelker and Luxmoore (1991) found that growth of current-year foliage of loblolly pine was more sensitive to elevated ozone concentrations when seedlings were grown in soil with higher levels of N.

The upper limit for needle N content can be defined by growing trees in defined nutrient-rich conditions. Results from such a study with seedlings (Bytnerowicz and others 1990) yielded needle N concentrations of 12.8 g/kg. The lowest needle N concentrations that can support growth are those in ponderosa pine stands that have developed on extremely poor soils. Isolated stands growing on nutrient depleted soils formed on hydrothermally altered parent materials in the Sierra Nevada of California had needle N concentrations of 8.1 g/kg (Schlesinger and others 1989). Mature needles on the trees at the Barton Flats sites had N concentrations ranging form 6.25 to 9.92 g/kg depending on the needle age. This comparison seems to demonstrate little difference between the Barton Flats site and remote sites in California, Nevada and Arizona (Klemmedson and others 1990, Schlesinger and others 1989, Powers 1981). However, N deposition at Barton Flats was above historic background levels.

With initial N deposition we would expect greater N availability to trees (Aber and others 1989, Schulze and others 1989, van Dijk and Roelofs 1988). This difference would parallel the situation at more fertile sites or sites that are fertilized. As a result trees could conserve energy by translocating less N before needles are abscised. At sites with poor soil fertility 60 percent of needle N is typically translocated before needles are abscised (Schlesinger and others 1989). Fertile volcanic soils with associated greater N availability result in less N translocation and readsorption (48 percent) before needle abscission (Klemmedson and others 1990). At Barton Flats only 45 percent of N is readsorped before needle abscission. This is the lowest resorption efficiency reported in the literature for ponderosa pine. Resorption efficiency (the percent reduction in nutrient concentration upon senescence of leaves or needles) has recently been contrasted by Killingbeck (1996) with resorption proficiency (the nutrient concentration achieved in senesced leaves or needles). Plants that can reduce N concentrations in senesced leaves below 7 g/kg are highly proficient. Ponderosa and Jeffrey pines at Barton Flats could reduce the concentration of N in senesced needles to 4.4 g/kg and so were highly proficient in N resorption.

P and K resorption efficiencies at Barton Flats by ponderosa pine were 60 percent and 70 percent effective, respectively. This level of P readsorption matches that found for ponderosa pine growing on very infertile soils (Schlesinger and others 1989). The P resorption proficiency for these two sites differ considerably: 0.59 g/kg at Barton Flats compared to 0.42 g/kg at the infertile soil sites in the Great Basin (Schlesinger and others 1989). Killingbeck (1996) has defined complete resorption proficiency in evergreen species as reducing the P concentration below 0.4 g/kg and incomplete resorption as a P final concentration greater than 0.5 g/kg. Clearly, P resorption proficiency at Barton Flats is incomplete.

The poor correlation of needle nutrient content with indicators of air pollution damage may be the result of the plants' ability to compensate physiologically for some air pollution damage and the variability of response between genotypes (Wellburn 1994) The principal air pollutant causing visible injury and thought to be responsible for growth changes is ozone (Peterson and others 1991, Temple and Miller 1994). The influence of ozone on nutrition is certainly indirect and chronic, whereas the response of a needle may be integrated over the much shorter time span (the lifetime of the needle). Consequently the possibility of developing a simple diagnostic needle nutrient analysis to assess air pollution influences seems remote.

Ozone exposure can lead to physiological disruptions that cause changes in the nutrient concentrations of current-year ponderosa pine needles. At SF, the site receiving the greatest ozone exposure, concentrations of P, K, Zn, and Fe in current-year foliage were highest in ponderosa pine trees with the most severe ozone injury, presumably due to retranslocation from older, ozone-damaged needles to younger foliage. Further studies are needed to elucidate mechanisms whereby ozone-stressed trees respond to, and in some cases possibly compensate for, acute and chronic ozone exposures and the interaction of these responses with associated N deposition.

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SESSION IV

Forest Health Monitoring: National Perspectives

Paul R. Miller and Oleg Blum, Chairs

Assessment of Forest Quality in Southwestern Poland with the Use of Remotely Sensed Data[1]

Zbigniew Bochenek, Andrzej Ciolkosz, Maria Iracka[2]

Abstract

A three-stage approach was applied to assess the quality of forests in southwestern Poland, which are heavily affected with air pollution and insect infestations. In the first stage a ground evaluation of spruce stands was done within the selected test areas. Three main characteristics of forest quality were determined as a result of these works: defoliation, discoloration, and vigor of trees. Next, aerial color infrared photographs were taken over the study area. They covered stands at different stages of forest decline. Detailed analysis of these photographs enabled discrimination of four levels of damage to spruce stands, as well as various phenomena, which accompany forest decline and transformation (loose canopy closure, windbreaks and windfalls, afforestations and regenerations). Results of analysis of aerial photographs and ground-truth information supported the main stage of the work: digital classification of Landsat Thematic Mapper (TM) satellite images. Digital classification of Landsat TM images can be useful for evaluation of forest quality and for rapid large-area assessment of forest changes, which accompany degradation processes. Different methods of analyzing satellite data were studied to obtain maximum information content. Supervised classification based on the TM3, TM4 and TM5 spectral bands was then applied. As a result of classifications for the test site, which is seriously affected with air pollution, three levels of forest damage were distinguished, as well as clear-cuts and vegetated openings accompanying forest decline. Comparison of two classifications performed on images collected in 1984 and 1990 revealed great changes in forest structure.

Introduction

Air pollution has been considered the main cause of damage and deforestation to forests in Europe. Air pollution increases along the direction of the prevailing wind from southwest to northeast so that pollution originating in southern and western Europe accumulates in Poland. This country itself contributes significantly to total air pollution, adding great amounts of pollutants from fuel and power industries, means of transport, and fossil fuel-heated homes.

Poland belongs to the group of most polluted nations because of the high concentration of SO_2. Limitation of the emission of sulphur compounds has not improved the situation considerably because about 50 percent of the total amount of SO_2 comes from neighboring countries.

The technology applied by foresters to determine forest damage has been based on analysis of the morphological changes in needles and tree growth. In the 1980's the Remote Sensing and Spatial Information Centre, OPOLIS, began to investigate the application of remote sensing to the detection and determination of damage to pine and spruce stands. At the beginning of the work, false-color aerial photographs were used to detect air pollution damage to pine stands. It was possible to distinguish between dead and dying trees based on changes in crown color as seen on aerial photographs. Five categories of damage to pine forest have been recognized by calculating the percentage of dead and dying trees in a given stand. A number of maps have been elaborated and published presenting the state of health of forests in different regions of Poland.

Damage to some Polish forests has been so extensive that it was decided to use satellite images in the investigation of the problem. The Landsat Multispectral Scanner (MSS), Thematic Mapper (TM), SPOT images, and Cosmos space photographs have been applied in these works. After several attempts we found that the Landsat TM images have been the best source of information for this type of investigation.

This paper discusses a three-stage approach ground evaluation, aerial photograph analysis, and digital classification of satellite images — to assess the extent of air pollution to Polish forests.

[1] An abbreviated version of this manuscript was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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Study Area

The Sudety Mountains, where the damage to coniferous forests was one of the most serious in Poland, were chosen as a test site. This region, located at border areas of Poland, Germany, and the Czech Republic, has been highly affected by industrial activity and insect infestations. Concentration of industrial plants and coal mines within the region caused serious degradation of large forest areas. Extent and intensity of the damages are so significant that this area is currently treated as a region of ecological disaster and it has been called "Black Triangle."

The test area covers the western part of the Sudety Mountains, i.e., part of the Karkonosze and Izerskie Ranges, with dominating spruce stands forming a large forest complex. The most serious damage of coniferous stands, as well as different levels of forest transformation, occur in this region.

This differentiation of stands within the study area can support optimally the main aim of the studies; it enables one to determine, which stand characteristics, useful for large-area forest evaluation, can be derived from satellite data.

Materials and Methods

At the first stage of the work we decided to examine the usefulness of high-resolution satellite images for classifying health conditions of the forest, its species composition, and other forest phenomena. To fulfill this task, Landsat TM data collected in July 1984 and in August 1990, covering both test areas, were selected. The analysis was done with the use of ERDAS image processing system. This system enables interactive analysis of satellite images: radiometric and geometric correction, location of training areas on satellite images, determination of spectral signatures, and data classification.

Source Materials — Field Work and Aerial Photographs

To find relationships between field characteristics of stands and their signatures on satellite images, field evaluation of spruce stands was done at a preliminary stage of the works by using more than 500 test fields located at the Karkonosze, Izerskie, Sowie, and Stolowe Ranges. Field appraisal was performed on the basis of instruction prepared by the Forestry Research Institute. Each test field covered a homogeneous appraisal unit (1-10 ha), which was characterized through direct measurements and observations. The following information was collected during field work:

- Description of terrain conditions (type, relief, slope, aspect, type of cover and degree of coverage by rocks, stones, debris, etc.).
- Description of stand (structure, layer, species composition, age, form of mix ing, closure, average breast-diameter, mean height, height of crown base, number of trees per unit area, including dying and dead trees).
- Evaluation of damage of stand (defoliation — Df, discoloration — Dc, quality of trees — Dm).

Defoliation and discoloration was evaluated in accordance with International Cooperation Program on Assessment and Monitoring Pollution Effects on Forests (ICP) recommendations, while quality of trees was assessed on the basis of the state of assimilatory apparatus, height increments, and vigor of trees.

At first stage of the work numerous relations were studied between stand/terrain characteristics and spatial signatures of stands, derived from Landsat TM satellite images, including elevation/aspect information. The following conclusions were drawn from these studies (Zawila-Niedzwiecki 1994):

- The higher elevation above sea level, the lower height of trees and height of crown base; lower quality trees found more on a high elevation than less damaged trees.

- High correlations (R^2>0.90) were found between spectral response in Landsat TM bands and some indices of quality of stands (Df, Dc, Dm). The strongest relations exist for defoliation and quality index, while discoloration is less correlated. Usually the relations are defined through multiple regression, which involves use of 2 to 4 spectral TM bands (most commonly visible near infrared and middle infrared bands).
- Loose canopy closure disturbs relationships between ground parameters and spectral signatures of stands, so if it exists on a larger area, this information should be taken into account in the process of supervised classification.

Analysis of the Karkonosze — Izerskie Ranges Test Area

Aerial false-color photographs taken in July 1984 were the supplementary source of information on the state of forests within the study area. The map of the health status of forests, prepared on the basis of these photographs, was the reference material for making appropriate selection of training and test areas, which was used further in the process of supervised classification of satellite data.

At the first stage of analysis the rectification of a satellite image was performed, through its transformation to the Gauss-Kruger projection system, which is commonly used for medium-scale topographic maps in Poland (1942 system). Next, correlation analysis was done for all seven bands of Landsat TM, in order to choose the three least-correlated channels. As a result of this analysis the following TM spectral bands were selected: TM3, TM4, and TM5. This choice is consistent with the other studies conducted at the European and American remote sensing centres. These studies revealed high usefulness of the mentioned spectral bands for classifying forest areas (Horler and others 1986, Kadro 1988, Rock and others 1986). According to these studies TM4 and TM5 give the best information for vegetation analyses, while TM3 combined with the latter ones forms a color composite that emphasizes spectral differences within vegetation classes. TM5 is especially useful for examining damage caused by loss of water in plant tissue and characterizing vigor of vegetation. In the case of forests at the Sudety Mountains, we found as a result of preliminary analysis that coniferous forests can be delineated with TM3, while TM4 differentiates grasslands, deciduous and mixed forests. TM5 allows different levels of defoliation to be discriminated, as well as clear-cut areas.

After preliminary analysis of color composite images, eight forest categories were selected:

1. Slightly impaired spruce stands
2. Heavily impaired spruce stands
3. Dying and dead stands
4. Young spruce stands
5. Openings with vegetation cover
6. Clear-cuts
7. Deciduous and mixed stands
8. Dwarf mountain pine.

Training areas representing these categories were transferred to the satellite image; next, spectral separability analysis was done to determine those categories that could be classified with the acceptable accuracy (exceeding 80 percent). We found that all selected classes, except dwarf mountain pine, were spectrally separable. So, seven forest categories were finally selected for classification, while areas of dwarf mountain pine were transferred from topographic map.

After selecting the classes, supervised classification of the Karkonosze-Izerskie Ranges test area was performed by using the 1984 image. Next, the same process was repeated for the 1990 image by using the analogue set of training areas. Results of both classifications were printed in the form of color maps. Finally post-classification accuracy assessment was done for both classification images by using the set of test fields distributed randomly throughout the study area.

Discussion

Comprehensive analysis of Landsat TM satellite images, conducted on two test areas for two dates of image acquisition, proved that spectral differentiation of spruce stands in this region enables the discrimination of a maximum of three classes of forest quality. Class No. 1 includes healthy and slightly impaired stands, with up to 30 percent loss of assimilatory apparatus. Initial phases of spruce damages cannot be separated from healthy stands with the use satellite data. So training fields located at the stands with 0-30 percent loss of assimilatory apparatus should be treated as representatives of one forest category.

Accuracy of discriminating on satellite images the other classes of spruce damage (more than 30 percent) depends on local conditions: canopy closure, soil cover, understory vegetation, etc. For instance, if stands with loose canopy closure contain openings with vegetation cover, spectral signature for these stands changes towards a class of better quality. Generally, there is the possibility to distinguish heavily impaired stands with 30-60 percent loss of assimilatory apparatus, as well as dying and dead stands that have lost more than 60 percent of needles.

Young coniferous stands were delineated well on the satellite image from 1984. This class includes mainly spruce stands (and partly pine stands) at ages of 7-25 years, covering 9 percent of the Karkonosze and Izerskie Ranges test area. The other classes, which are usually hardly discernible on satellite images (i.e. grasslands and afforestations) were quite well discriminated. As afforestations cover about 7 percent of the total classified area, this class significantly characterizes structure of the studied forests.

Main forest categories — three levels of damage of spruce forests and deciduous/mixed forests — were classified with accuracy exceeding 80 percent. By considering complex natural phenomena that existed at the studied region, the obtained results should be treated as fully satisfactory. They characterize both structure and health conditions of forest existing at the Sudety Mountains with acceptable accuracy.

Comparative analysis of two classified images from 1984 and 1990 enabled evaluation of qualitative changes that occurred during 6 years within the study area. The main observed phenomena were as follows (Zawila-Niedzwiecki 1994):

- An increase of the area comprising impaired spruce stands by 30 percent.
- A decrease of the area of dying and dead trees from 15 percent to 5 percent.
- An increase of the area of openings and clear-cuts from 8 percent to 15 percent with domination of vegetated openings in 1990.

Comparison of the classified image with forest maps revealed satisfactory consistency between classes determined from satellite data and stand characteristics contained at forest maps. Thus, characteristics of stands, obtained as a result of classification of Landsat TM images for the Karkonosze and Izerskie Ranges region, are suitable for large-area description of health conditions and structure of stands in this region. Landsat TM satellite images, then, enable evaluation of general characteristics of health conditions and structure of stands and thereby permit rapid evaluation of degradation processes.

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Response of Scots Pine Stand Vitality to Changes in Environmental Factors in Poland, 1991 - 1995[1]

Jerzy Wawrzoniak[2]

Abstract

Vitality inventories of Scots pine stands, the most common species in Poland, have been done since 1991 by using the ICP-Forest methodology. In Scots pine stands older than 40 years, 1,040 observation plots were established. Defoliation was used as the primary indicator of stand vitality. During 1991 to 1995, SO_2 and NO_x were measured at 1,417 forest locations using passive samplers. The interrelation between changes of precipitation amounts during the growing season and vitality of Scots zones with different air pollution levels was studied. The analysis was done in three zones of Poland: north Poland is considered a relatively clean area; the central part of Poland is characterized as having an average level of pollution; and south Poland has the highest pollution. In each of these zones, decreasing concentrations of SO_2 and NO_x were observed during the past 5 years. Changes of vitality in Scots pine stands caused by the amount of precipitation and air pollution level in the growing season were studied for each of the three zones.

Introduction

Forest health status in Poland has been strongly influenced for many years by high levels of air pollution. Emissions of SO_2 (5,000,000 tons) and NO_x (2,500,000 tons) in the 1980's were among the highest in Europe (Ochrona Srodowiska i Gospodarka Wodna 1987). Vegetation condition of forest trees in Poland have been unfavorable. The forests in Poland grow on mainly sandy soil with poor nutrient availability and shortage of water because of low precipitation during the growing season. In these circumstances impacts of air pollution on forest health are particularly profound and cause increases of damage reflected in high defoliation levels (Malachowska and Wawrzoniak 1995). In the beginning of the 1990's, rapidly declining air pollution levels were observed. Emission of SO_2 and NO_x was reduced by 30 to 50 percent (Ochrona Srodowiska 1993). In spite of this, forest health condition did not indicate signs of improvement. A second essential factor that impacted forest health condition was the amount of precipitation during the growing season. The combination of air pollution and water deficiency determined the health status of foliage and its photosynthetic ability. Generally, the long-term health condition of Scots pine stands on a regional scale has been determined by the level of air pollution, but yearly changes of health condition has depended on the amount of precipitation during the growing season (Malachowska and Wawrzoniak 1995). If water deficiency occurs during the growing season, response of stand vitality differs between relatively healthy stands situated in low polluted areas and more damaged stands in highly polluted areas.

The purpose of this study was to identify the reaction of Scots pine stand vitality in various levels of air pollution in three zones of Poland impacted by a shortage of water during the growing season in 1991-1995.

Methods

Mean defoliation of Scots pine stands in specific zones was used as an indicator of forest vitality. This parameter was calculated on the basis of defoliation assessments on 5,000 trees in each zone *(fig. 1)*. According to the ICP-Forest Manual (United Nation Economic Commission for Europe 1994), assessments were done by trained staff every year between July 1 and August 30, 1991-1995.

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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Figure 1 — The three district air pollution zones and locations of observation plots in Poland. I –north Poland (low level of air pollution), II – central Poland (medium level of air pollution), III – south Poland (high level of air pollution).

SO_2 and NO_x deposition were determined by passive method with 1-month exposure of samplers (Krochmal and Kalina 1997). Measurements were made at 600 points in each zone. Mean values of SO_2 and NO_x were calculated separately for winter and summer periods in each zone. Yearly precipitation during 1991-1995 was calculated as the mean of values from about 60 meteorological stations.

Results and Discussion

Comparison of air pollution levels during the summer period expressed in SO_2 and NO_x indicator values shows differences between the three zones (*fig. 2*). The highest values were above 8 mg/m²/24h in southern Poland and below 4.5 mg/m²/24h in north Poland (*fig. 2*).

Considering mean defoliation of Scots pine stands in specific zones, a distinct difference was found between south Poland and north Poland or central Poland, but no difference between north and central Poland (*fig. 3*). The low level of air pollution in north Poland seems to be an important factor influencing the condition of pine stands that appear more healthy than in south Poland where air pollution is much higher. The lack of differences between central and north Poland is difficult to explain, despite differences in air pollution level.

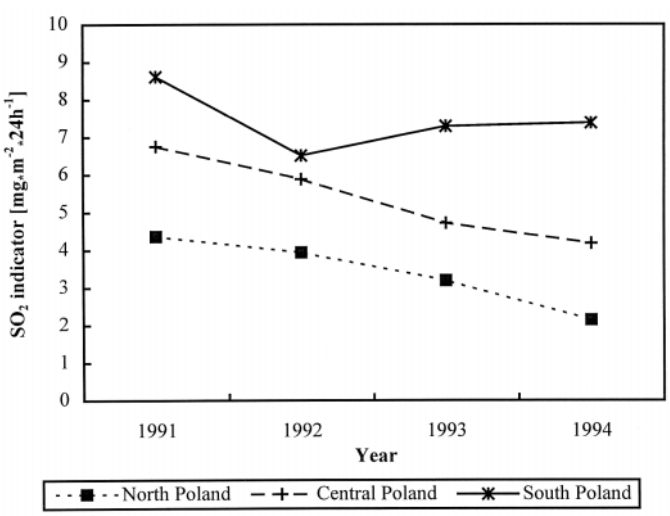

Figure 2 — Changes in the SO_2 indicator during the summer period for 1991 to 1995.

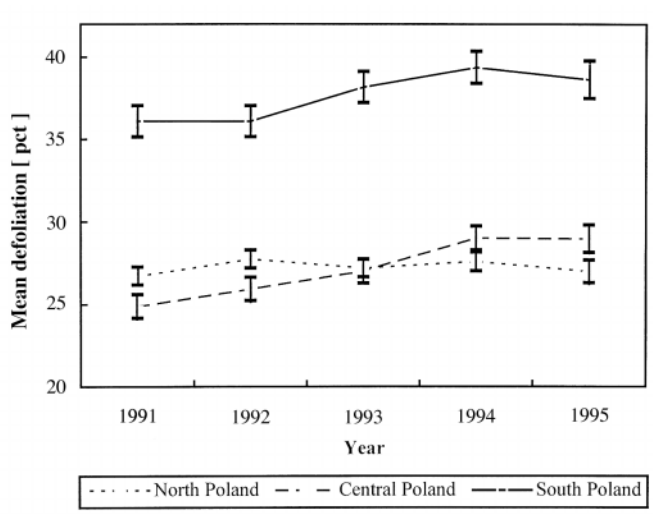

Figure 3 — Changes in Scots pine defoliation (two-tailed Tukey's Test) from 1991 to 1995.

The most interesting question that should be considered is the response of Scots pine stands of various health status to changes of precipitation during the growing season. By comparing changes between mean defoliation in a specific year and changes of precipitation in the same year, a pattern was found in the healthy stands of the north Poland zone. In 1992 and again in 1994, decreased precipitation (Miesieczny Preglad Agrometeorologiczny 1991-1995) was followed by increased mean defoliation in the same year. In 1993 and in 1995, increased precipitation was followed by decreased mean defoliation levels in the same year (*fig. 4*).

The response of Scots pine stand vitality, indicated by mean defoliation to changing amounts of precipitation, was different in the more polluted and more damaged stands in the south Poland zone. Decreasing precipitation in 1992 was not followed by increasing mean defoliation in the same year, as happened in north Poland. However, mean defoliation increased in 1993 despite increased precipitation. Improvement in stand health condition, indicated by decreased mean defoliation, was found in 1995 after 2 years of increased precipitation (*fig. 5*).

Similar reactions of Scots pine stands to precipitation amount was also found in central Poland (*fig. 6*) where the air pollution level is lower than in south Poland (*fig. 2*), and the health condition of Scots pine stands is better (*fig. 3*).

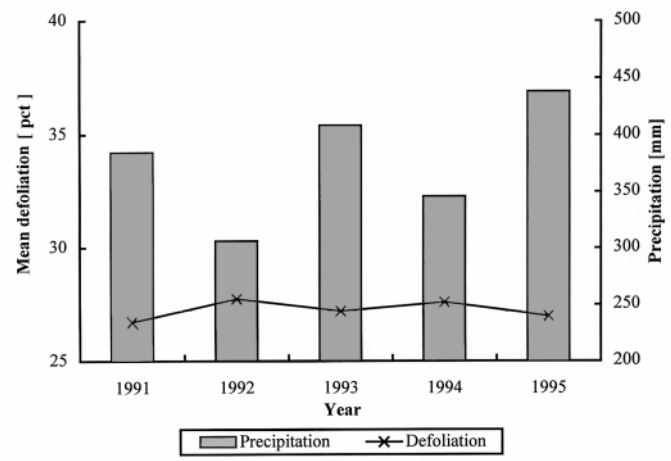

Figure 4 — Changes in Scots pine defoliation compared to the sum of growing season precipitation in north Poland from 1991 to 1995 .

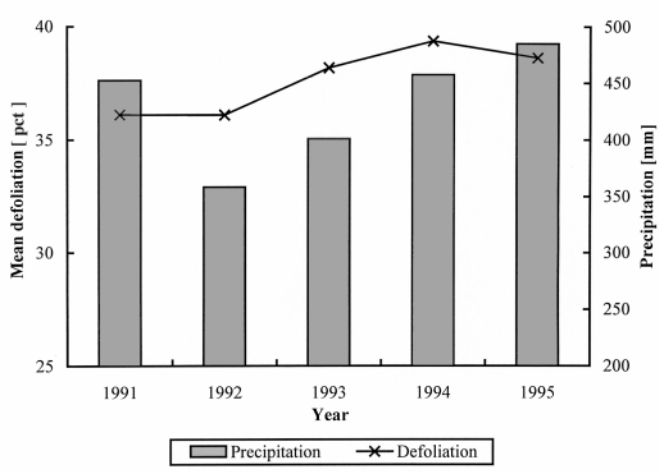

Figure 5 — Changes in Scots pine defoliation compared to the sum of growing season precipitation in south Poland from 1991 to 1995.

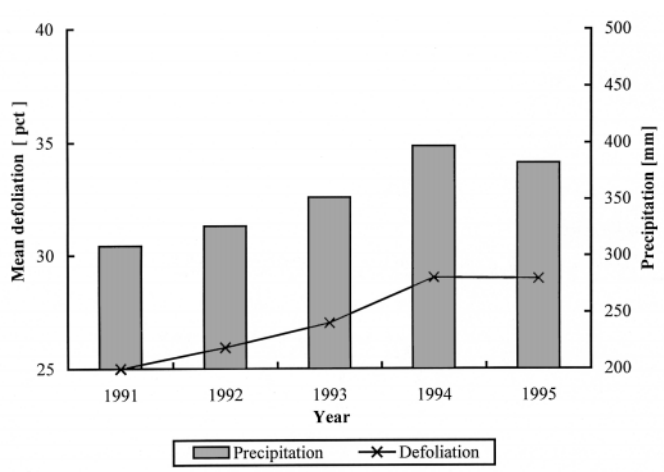

Figure 6—Changes in Scots pine defoliation compared to the sum of growing season precipitation in central Poland from 1991 to 1995.

The different responses of the vitality indicator (mean defoliation) to changes in precipitation amount during the growing season seems to be connected to the level of health condition at the beginning of the period studied.

In healthy pine stands in north Poland, the majority of trees had three or four years of needles. A small amount of precipitation during the growing season resulted in a shortage of water and trees immediately dropped one or two of the oldest years of needles, regardless of the consequences to buds for the next year. Trees in this region easily regenerate foliage the following year, when precipitation in the growing season is increased.

Scots pine stands in south Poland are in poor health condition and usually trees have 2 years of needles. If precipitation is low during the growing season, trees cannot drop needles without weakening the assimilation processes and endangering their survival for the next year. The only possible response in this condition is limiting bud formation in the same year and maintaining needle retention at the same level in the following year. If precipitation is adequate during the growing season, Scots pine trees show more effective bud formation in the current year and the improved vitality of stands is reflected by increased needle retention in the next year.

Conclusions

The response of Scots pine stand vitality to a shortage of water during the growing season is different in high polluted areas and low polluted areas. The health condition of Scots pine stands at the beginning of drought periods is crucial to the response of the stands. Scots pine stands in good health condition respond to drought during the growing season in the same year by dropping needles. Scots pine stands in poor health condition respond to drought in the growing season by changes in the bud formation process and increased defoliation in the following years.

Acknowledgments

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Air Pollution and Forest Health Studies along a South-North Transect in Poland[1]

Stefan Godzik, Jerzy Szdzuj, Tomasz Staszewski, Wlodzimierz Lukasik[2]

Abstract

*Air pollution, bulk deposition and throughfall, soil characteristics, needle chemistry, and forest injury were studied on six permanent plots from the south (Brenna and Salmopol in the Beskidy Mountains) to the north (Gac, the Baltic Sea coastal area) in Poland. The concentrations of sulfur dioxide and nitrogen dioxide were the highest at the Katowice location and the lowest at Gac. Annual mean concentrations of sulfur dioxide at the Brenna, Salmopol, Katowice, and Puszczykowo sites, have exceeded the critical level of 20 $\mu g\ m^{-3}$ proposed by United Nations- Economical Committee for Europe (UN-ECE) bodies, and the Polish national standard (32 $\mu g\ m^{-3}$ for the whole country, and 11 $\mu g\ m^{-3}$ for special protected areas, e.g. National Parks). Nitrogen dioxide concentration has neither exceeded the UN-ECE nor the Polish national standards for the whole country (30 and 50 $\mu g\ m^{-3}$, respectively). Bulk deposition from all locations was acidic. The pH of throughfall in spruce stands was lower than the bulk deposition by about 0.5 of a unit. A larger decrease of pH has been found occasionally. Concentration of all elements measured was higher in throughfall as compared to bulk deposition. However, the differences in concentration of elements between bulk deposition and throughfall varied. The differences were from three to five times for sulfates, but for nitrogen compounds, such as nitrates and ammonium, differences were smaller or did not exist at all. The soil pH of all plots ranged from 2.9 to 3.6 (H_2O) in the organic layer. The highest concentrations of lead and zinc were found in the organic layer of the sites in southern Poland. The greatest needle loss (crown transparency) was found at the highest elevation site (Salmopol) for Norway spruce (*Picea abies*) and in the most polluted site (Katowice) for Scots pine (*Pinus sylvestris*). The smallest crown transparency of spruce was found for the Gac location. Discoloration of needles was the highest at Gac, followed by Katowice and Salmopol. Insects contributed to the observed higher crown transparency on plots at Brenna and needle discoloration at Gac.*

Introduction

Sulfur dioxide (SO_2) air pollution acting directly or indirectly has been one of the major causes for poor forest health status and forest decline in Poland (Ashmore and others 1990, Forschungsbeirat Waldschaeden/Luftverunreinigungen 1989, Godzik 1984, Godzik and Sienkiewicz 1990, Narodowy Program 1988, Koth-Jahr and Kollner 1993, Paluch 1968). According to model calculations, a gradient of both concentrations and depositions of sulfur compounds was found from the south to the north of Poland (Narodowy Program 1988) *(fig.1)*. The reason of such uneven distribution was mostly emission from sources located in the south of Poland, the Czech Republic, and Germany (former German Democratic Republic). Results of forest injury inventories in Poland have followed a similar pattern. The most severe injuries (class three) were noted in southern Poland along the border with the Czech Republic. Severe injury was also observed on a local scale in other parts of the country (Godzik and others 1996, GUS 1995, Malachowska and Wawrzoniak 1995). Although emission of air pollutants in recent years has decreased, the pattern of its distribution has remained similar.

The total area of forest in Poland is 873,200 hectares, comprising 27.9 percent of the total area of the country (GUS 1995). Scots pine (*Pinus sylvestris*) cover about 70 percent of the total forest area. Norway spruce (*Picea abies*) cover about 6 percent of the forest (GUS 1995).

On the basis of data concerning the gradient of sulfur dioxide concentration in Poland, six permanent plots in coniferous forests have been established. Climatic

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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differences are represented by plots at different locations above the sea level (asl) in the Brenna region and at the Baltic Sea coast. Differentiation of air pollution level is represented along the line from the south to the north of Poland *(fig. 1)*.

This paper reports a study of the concentrations of sulfur dioxide and nitrogen dioxide in the air, deposition of air pollutants and mineral nutrients, and their relationship to forest stand parameters, such as needles and soil characteristics, on six permanent plots in Poland.

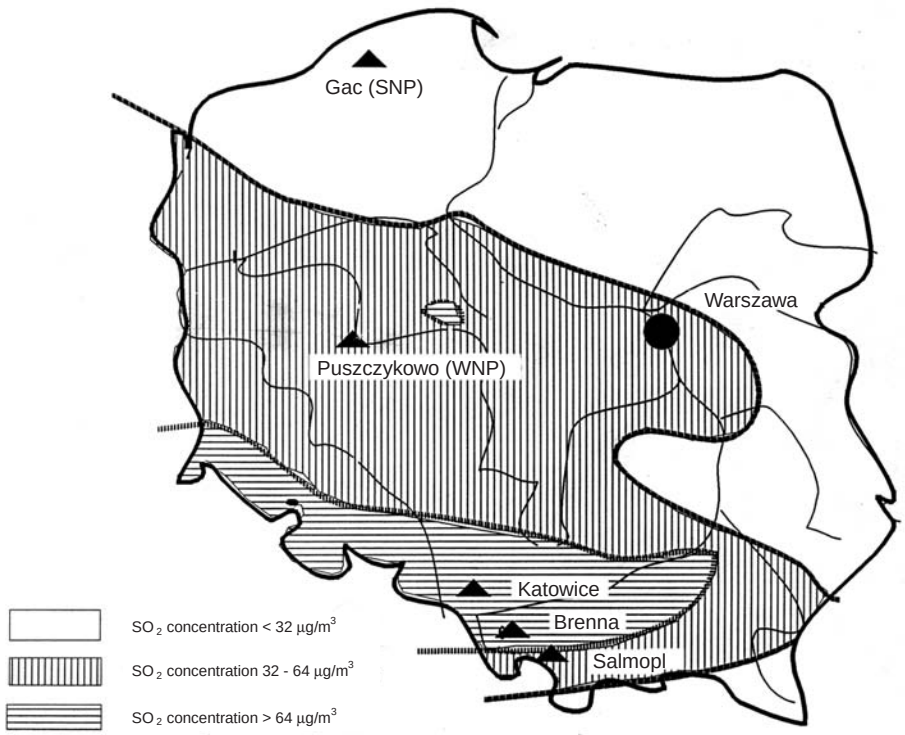

Figure 1 — Locations of six forest permanent plots in Poland with isolines showing SO_2 annual mean concentration in the air. SNP = Slowinski National Park; WNP = Wielkopolski National Park.

Site Description, Material, and Methods

Permanent forest plots were established in southern Poland (two plots in the Brenna area and one plot in the Katowice region), central Poland (Puszczykowo — Wielkopolski National Park (WNP)), and two plots in northern Poland (Gac — Slowinski National Park (SNP) — at the Baltic Sea coast) *(fig.1)*. The size of the plots was 50 by 50 m. All plots met standards required by the measurement program for sites for Integrated Monitoring for Forest Ecosystems within the Long Range Transboundary Air Pollution (LTRAP) UN-ECE protocol (Environmental Data Centre 1989).

The two plots in the Brenna area, the Beskidy Mountains, were located at 1,000 m asl (Salmopol) and 650 m asl (Brenna). The main tree species in those sites were Norway spruce with additions of white fir *(Abies alba)* or beech trees *(Fagus sylvatica)*. The plot in the Katowice region was basically a Scots pine stand with admixture of broad leaf species in the lower layer. This region was the most polluted and has been influenced by air pollution for more than a century. The only tree species at the Puszczykowo plot was Scots pine. In Gac, at the Baltic Sea coast, two plots were established in spruce and pine stands. On every plot 15 throughfall collectors were randomly distributed. In an open area, collectors for bulk precipitation were placed. Ceramic vacuum cup lysimeters were installed at 25 and 50 cm depth on each of plots in a regular net. All samples were collected and analyzed monthly.

Ions were determined with ion chromatograph DX-100 with column AS4A.[3] Metallic cations were determined with atomic absorption spectroscope Varian Spectra AA 300. Concentration of NH_4 was determined by using the Nessler method (Hermanowicz and others 1976). Soil samples were collected from each horizon and analyzed separately. For the chemical soil analysis, standard methods recommended by the Soil Service Laboratories were applied (Litynski and others 1976).

Needles for chemical analysis were taken in late October. Forest class injury was assessed based on degree of crown transparency and discoloration from methods recommended by the UN-ECE ICP Forest working group. Concentrations of sulfur dioxide and nitrogen dioxide in Brenna, Salmopol, Katowice, and Puszczykowo were determined as 24-hour means. Sulfur dioxide was absorbed in H_2O_2, producing sulfates that were determined with ion chromatography. Nitrogen dioxide was determined by methods recommended by EMEP: absorbed on a sintered glass filter impregnated with KI-$NaAsO_2$ mixture, and colorimetrically determined by using sulfonamide and N-(1-Naphtyl)ethylenediamine dichloride (NEDA) reagents (Environmental Data Centre 1989). Data for the Baltic coast area is from the national base network for air pollution measurements (PIOS 1995).

Results and Discussion

Air Pollution

Differences in concentration of sulfur dioxide and nitrogen dioxide were not found between two elevations in the Beskidy Mountains *(fig. 2)*. The data show that monthly and yearly mean concentrations of sulfur dioxide were higher than for the Rhine-Westphalen area in 1987 (Koeth-Jahr and Koellner 1993); and the values for nitrogen dioxide were lower than in the same area of Germany (Koeth-Jahr and Koellner 1993). For the Schwarzwald area, the values did not exceed 29 µg m^{-3}, and for the growing season were lower by 50 percent (Forschungsbeirat Waldschaeden/ Luftverunreinigungen 1989). The concentration pattern of sulfur dioxide on the plots showed similar differences between winter and summer. The mean values were rather high, given that sulfur dioxide is not the only air pollutant hazardous for trees. Concentrations of this air pollutant are lower than proposed by the International Union of Forest Research Organizations (IUFRO) for not extreme environmental conditions (50 µg m^{-3}). However, except for the Gac area, values of sulfur dioxide were in the range or higher than proposed as critical levels by the UN-ECE convention on Long Range Transboundary Air Pollution (20 µg m_{-3} and 30 µg m^{-3} for SO_2 and NO_2, respectively; Ashmore and Wilson 1994). Concentration of sulfur

[3] Mention of trade names or products is for information only and does not imply endorsement by the U.S. Department of Agriculture.

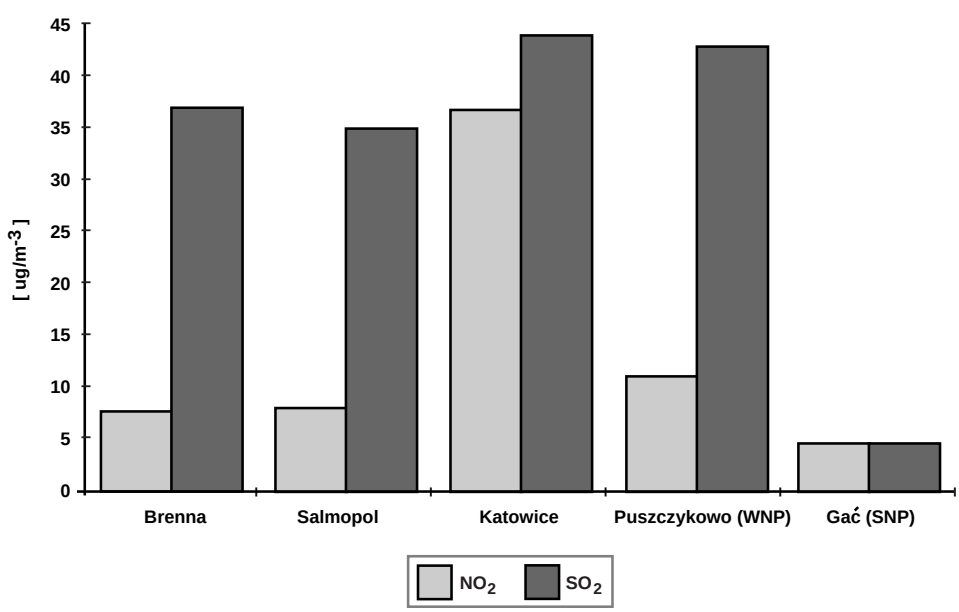

Figure 2 — Annual mean concentration of SO_2 and NO_2 (µg m^{-3}) in the air over permanent plots.

dioxide has exceeded or was very close to the Polish national standard (32 µg m^{-3} as yearly mean value) for the whole country. For the WNP, as a "special protected area," the standard is 11 µg m^{-3}. Concentration of sulfur dioxide was found to be several times higher.

The most probable direct effects on the forests caused by air pollutants (only SO_2) were the sites in the Katowice area, and to a lesser extent in the WNP. Direct effects of nitrogen dioxide seems to be less probable. For the plots located in the Beskidy Mountains, ozone should be considered as a possible factor contributing to the forest health status (Godzik, this volume).

Deposition

As expected from the model calculations (Narodowy Program 1988), the bulk deposition concentrations of most elements were higher in the polluted locations: Katowice and Puszczykowo (WNP) (see Staszewski 1996 for results of all six locations). The most interesting data for loads of sulfur and nitrogen are the differences between pine and spruce stands at the Gac site *(table 1)*. This difference may be the result of species-specific characteristics and/or number of trees for the same area. No similar comparison could be made for the southern Poland stands. Higher loads of sulfur and nitrogen for the spruce stands in Brenna and Salmopol compared to the Katowice area can be explained by rainfall that is almost two times higher in the Brenna region. If spruce stands were compared, loads for spruce in the Katowice region would likely be even higher. We may also consider that the extent of the earlier and more severe deterioration of spruce stands compared to pine stands in the Silesia Industrial Region could be attributed to differences in loads (Godzik and Harabin 1968). Amount (load) of sulfur and nitrogen deposited to all forest ecosystems under study exceeded, except SNP sites, the critical loads recommended by the UN-ECE.

Table 1 *— Annual loads of sulfur and nitrogen calculated for bulk deposition and throughfall (kg ha^{-1} a^{-1}).*[1]

Location and species	Sulfur	Nitrogen
Salmopol: spruce		
Bulk deposition	15.1	26.6
Throughfall	36.3	30.3
Brenna: spruce		
Bulk deposition	13.6	17.2
Throughfall	41.0	28.0
Katowice: pine		
Bulk deposition	12.7	10.4
Throughfall	27.3	13.0
WNP: pine		
Bulk deposition	13.1	13.7
Throughfall	27.8	15.5
SNP		
Bulk deposition	8.4	8.8
Throughfall(pine)	10.0	8.0
Throughfall(spruce)	17.0	12.5

[1] The values are a sum of data from separate sampling periods (sampling intervals). During the whole sampling period, concentration of elements in throughfall was always higher if compared to the same sampling time for the bulk deposition.

Soil Characteristics

Soils of all but one location (spruce site in Gac) were of the podzolic type and of low pH (Godzik and others 1994). Soil profiles represented typical handbook descriptions of distributions of genetic horizons. Low amounts of calcium and magnesium that are mostly responsible for the soil pH were found. In all soils potassium content was higher than calcium. The highest concentrations of lead and cadmium were found in the organic soil layer. The concentration of lead in samples from southern Poland exceeded the concentration recognized as "normal" for soils of this region.

Soil investigation data showed that soils on all but one plot were comparable. Thus, this data shows that climate and air pollution were the major variables in our studies. Differences in forest injury class at plots under investigation could not be explained by differences in soil properties. This conclusion is based on the comparison of soil data from two plots in the Brenna area for spruce, and Katowice-WNP for pine. Investigations performed earlier for the Izerskie-Beskidy Mountains showed similar conclusions (Godzik and Szdzuj 1994).

Needle Chemistry

The highest concentration of sulfur in needles of spruce has been found for the coastal area (Gac), followed by that found in the Salmopol and Brenna needles *(fig. 3)*. The highest injury class has been assessed at the Salmopol location, and the least at the coastal area (SNP). Further investigations are needed to explain the highest concentration of sulfur in spruce needles from the Gac location, where the air pollution was the lowest. One possible explanation is the difference in soil properties at this site, as compared to the locations at Brenna and Salmopol. The influence of the nearby Baltic Sea seems to be less likely when looking for the results for sulfur concentration in pine needles. For the pine stands, the sulfur content of needles is higher in forests of higher injury class: Katowice had the highest pine needle injury, followed by WNP and SNP.

Forest Injury

The highest mean defoliation (crown transparency) in spruce stands was found in Salmopol, followed by Brenna and the Gac (SNP) *(fig. 4)*. At the Salmopol location, 100 percent of trees were in injury classes 2 and 3. At the Brenna and Gac locations 60 and 30 percent, respectively, were in these classes. Only trees at the Brenna location were found with no injury. At the Brenna and Gac locations, insect injuries were noted.

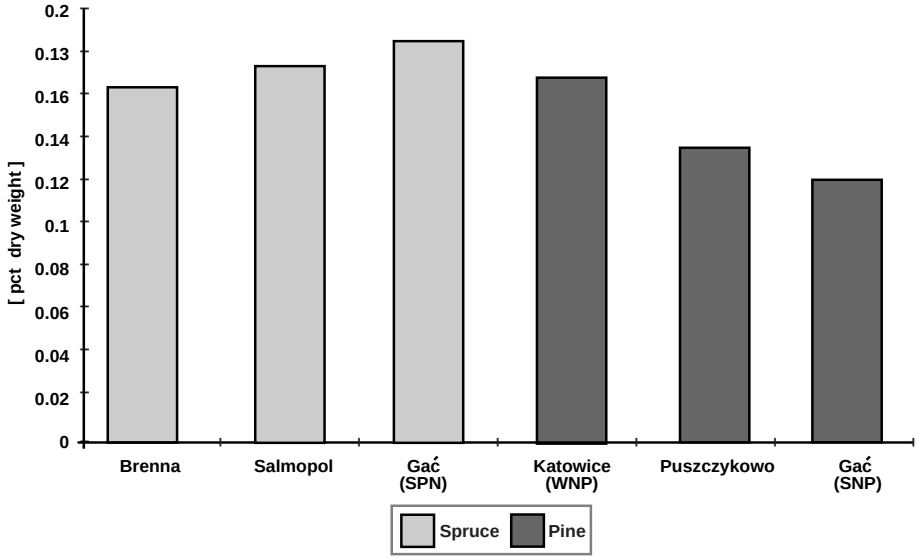

Figure 3 — Content of sulfur (percent dry weight) in 1-year-old spruce and pine needles.

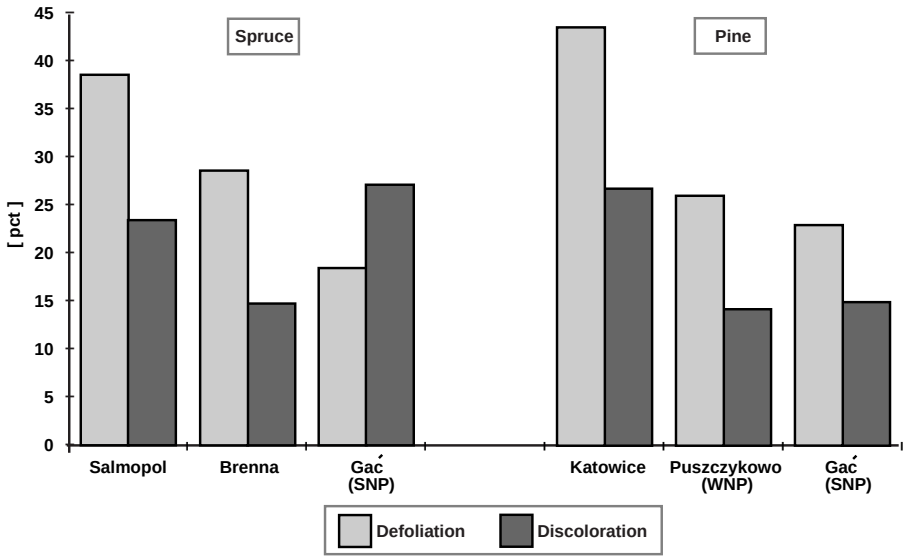

Figure 4 — Mean values (percentage) of defoliation and discoloration of pine and spruce stands on permanent plots.

Most severe defoliation of pine trees has been found at the Katowice plot, where 80 percent of trees were in the injury class 3. The numbers for the WNP and SNP plots are 20 and 25 percent, respectively.

Mean values for discoloration of spruce trees are the highest for the Gac, followed by Salmopol and Brenna *(fig. 4)*. However, at the Salmopol plot 35 percent of trees are in the discoloration classes 2 and 3. For the two remaining plots the numbers are 10 percent for Gac and 5 percent for Brenna. No discoloration was found on 35 percent of trees at the Brenna plot, followed by Gac with 25 percent, and Salmopol with 5 percent.

The highest mean discoloration of pine trees was found at the Katowice plot. No difference in discoloration was found between the WNP and SNP locations (*fig. 4)*. All trees at the Katowice plot are in the discoloration class 1 (60 percent) and 2 (40 percent). For the WNP plots they are 30 percent and 10 percent; and 55 percent and 0 percent for the SNP plots — class 1 and 2, respectively. Similar patterns of sulfur concentration in pine needles have been found. Because soil properties at all plots did not differ distinctly, the discoloration differences may be more related to concentration of sulfur dioxide and load of sulfur deposited to these ecosystems.

Conclusions

Concentrations of sulfur dioxide were the highest in the Katowice region, followed by the Wielkopolski National Park (WNP), the Brenna (Beskidy Mountains) and Gac (Slowinski National Park) locations. Concentrations of nitrogen dioxide followed a similar pattern. Load of sulfur and nitrogen to the forest floor differs depending on localization and tree species. Higher loads were measured on spruce stands as compared to the pine (Gac). Higher loads have been found for spruce stands in Salmopol and Brenna when compared to pine stands in the Katowice region (where concentration of air pollutants is higher). The result is caused by rainfall that is almost two times higher in the Beskidy location. However, other factors may be involved as well.

Forest soils at all sites were acidic. Except for the soil at the spruce stand at Gac, they were also of similar physico-chemical characteristics. The highest levels of lead and cadmium were found in an organic layer and in sites in the southern part of the country.

No clear relationships were found between air pollution — concentration and load — and forest health on plots under investigation. However, injury of spruce and pine trees were higher at locations where air pollution levels were also higher. The comparison of results from Brenna and Salmopol suggests that climatic parameters are an important factor in forest injury. However, a lack of measurements resulted in the inability to assess the possible contribution of ozone to the more severe forest injury at the higher elevations.

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Forest Health Status in Russia[1]

Vladislav A. Alexeyev[2]

Abstract

About 886.5 Mha in Russia is occupied by forests, including 763.5 Mha of tree stands and 123 Mha of non-stocked lands. The Russian forests comprise about 22 percent of the earth's forest area or 43 percent of the earth's temperate and boreal forests. Main forest- forming species are Larix *sp. (32 percent of the growing stock),* Pinus sylvestris *(20 percent),* Picea *sp. (15 percent),* Betula *sp. (13 percent),* Pinus sibirica *(10 percent),* Populus tremula *(5 percent),* Abies *sp. (3 percent),* Quercus *sp. (1 percent) and other species (1 percent). About one-third of tree stands and two-thirds of forest ecosystems in Russia are disturbed by natural and anthropogenic stresses. Most prevalent are natural causes of disturbance (climate, aging of stands, fires, pests) that affect 200-250 Mha of forest stands. About 45 Mha of forest stands (6 percent of stocked area) are under the impact of anthropogenic influence. Atmospheric pollution is the most dangerous form of anthropogenic stress in the Russian forests.*

Introduction

According to national inventory data from 1993, lands classified from the Forest Fund of Russia (1995) occupy 1,180.9 Mha. These lands include 294.4 Mha of non-forest land (mires, water, rocks, wastelands, etc.) and 886.5 Mha of forest land, including 763.5 Mha of tree stands and 123.0 Mha of non-stocked lands (woodlands, clear-cuttings, burned stands, etc.). These forest lands comprise about 22 percent of the Earth's forest area or 43 percent of the earth's temperate and boreal forests (United Nations 1992).

Forests are distributed in plain forest-tundra and subarctic mountains (14.1 percent), boreal plain and mountain regions (66.0 percent), zone of mixed forests (1.7 percent), zones of forest-steppes, steppes, and subboreal mountains (17.9 percent), and zone of semi-deserts and subarid mountains (0.3 percent). Natural forests comprise 98 percent and plantations 2 percent of the total forested area. About 350 Mha of forests are in permafrost territory, and about 340 Mha in mountains (mainly in subarctic and boreal regions). The average yield of the Russian forests is not high: the growing stock of mature and overmature forests is 136.7 m^3/ha *(table 1)*. Main forest-forming species are *Larix* sp. (32 percent of the total growing stock), *Pinus sylvestris* (20 percent), *Picea* sp. (15 percent), *Betula* sp. (13 percent), *Pinus sibirica* (10 percent), *Populus tremula* (5 percent), *Abies* sp. (3 percent), *Quercus* sp. (1 percent), and other species (1 percent). Forests are considerably damaged by natural and anthropogenic stresses.

This paper presents the current health status of the Russian forests and predicts possible trends caused by global climate change.

Forest Health Status and Factors of Disturbance

Climate

One of the primary disturbance factors of trees in the frontier forests of the forest-tundra and subarctical mountains on permafrost territories of Siberia and eastern Russian is severe climatic conditions (*table 2*). Trees in open, sparse unproductive forests and woodlands of these regions have crowns heavily damaged by frost and wind. Leaf area index (LAI) of trees is three to five times less than LAI of trees in more southern territories. Healthy stands and healthy trees in stands and

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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woodlands of permafrost territory are a rare phenomenon. The area of such naturally disturbed forests is estimated to exceed 150-200 Mha.

Long-term monitoring of several Siberian stands showed no change for the worse of the health status of such forests. Moreover, data indicate improvement of health condition of these forests. However, the improved condition of forests has not yet been statistically proven.

As a result of global warming in the next 100 years, an improvement of forest health on the present permafrost territory and decrease of damaged area of stands can be expected.

Table 1 *— Age structure of area and growing stock in Russian forests (Forest Fund of Russia 1995).*

Statistic data	Age classes of stands: Young	Middle-aged	Premature	Mature and overmature	Total
Area, million ha	134.4	201.6	74.8	352.7	763.5
percent	17.6	26.4	9.8	46.2	100
Growing stock, billion m^3	3.7	20.2	10.5	46.3	80.7
percent	4.6	25.0	13.0	57.4	100

Table 2 *— Present and forecasted forest area of Russia disturbed by natural and anthropogenic factors.* [1]

The primary reason of damage	Present status		Forecasted status in year 2100			
	Stands Mha	pct [2]	Ecosystems Mha	pct [2]	Stands	Ecosystems
Natural factors						
Climate	150-200	20-26	150-200	20-26	decrease	decrease
Fires	10	1	50	7	increase	increase
Pest	10	1	20	3	increase	increase
Aging of trees	100-150	13-20	100-150	13-20	decrease	decrease
Subtotal pct[3]	200-250	26-33	250-300	33-40	decrease	decrease
Anthropogenic factors						
Harvesting	20	3	100	13	the same	the same
Pollution	5	1	15	2	decrease	decrease
Fires	20	3	100	13	the same	the same
Subtotal [3]	45	6	215	28	the same	the same
Total[3]	240-290	31-38	460-510	60-67	decrease	decrease

[1] The data presented in this table are the author's expert estimates of disturbed forest areas (except the total area of burnt stands and stands damaged by pests) and may not agree with point of view of Federal Forest Service of Russia.

[2] Percent of total stocked area.

[3] Sum subtotal and total do not equal to mathematical sum of items because different factors can act simultaneously in the same territory.

Aging and Mortality of Forests Stands

Aging of trees is a genetically determined process of weakening and disturbing plants and their communities. As a rule, deterioration of old trees' condition reveals itself through action of various factors such as climatic stresses, insect pests, rots, etc. The age structure of stands in the ecoregions and administrative territories of Russia is extremely non-uniform (*table 2*). The forests of the European part of the country are strongly depleted by logging, and the areas of the old-growth forests in some regions are very small. The territories of Siberia and eastern Russian, on the contrary, have a large portion (> 50 percent) of mature and overmature forests. The forests of the overmatured and weakened stands categories amount not less than 150 Mha.

As a result of succession processes, during the next century these forests will be a strong source of carbon dioxide.

Fires

According to the national inventory data from 1988, the total area of burnt stands in Russia is 27 Mha (Forest Fund of the USSR 1990). About 17 to 20 Mha of these stands have burnt as a result of negligence of people. Statistical data do not include the area of forests burnt by surface fires. This type of forest fire damages tree stands; but more importantly, other compartments of ecosystems are also damaged, such as understory, litter, and part of organic substances of soils. Natural reforestation of young forest stands requires about 10 to 20 years. However, for recovery of other components of ecosystems, for example lichen cover, 50 to 100 years or more are necessary (Gorshkov 1994). This difference explains why the area of disturbed forest ecosystems is larger than the area of damaged tree stands. Main territory of wild fires occurs in the Asian part of Russia, especially Yakutia and eastern Russian. We can expect that global climate warming will increase the number and areas of fires, burnt stands, and disturbed forest ecosystems (*table 2*).

Pests

Russia has 10 Mha of forests damaged by different pests (*table 2;* Isaev, pers. comm.). In 1995 in Middle Siberia about 1 to 2 Mha of conifers stands were damaged by the Asian gypsy moth alone.

Based on unfavorable age structure of Russian forests, namely, large areas of overmature forests (*table 1*) we can predict increasing outbreaks of insects, fungi, and other pathogens in the future.

Harvesting

As a result of harvest activity, Russia has about 25 Mha of disturbed tree stands and about 100 Mha of recovering forest ecosystems (*table 2*). It is very difficult to predict the rate of clearcutting, other types of fellings, and future damage of forests. This is mostly because Russia is presently undergoing various, often unpredictable, political and economic changes.

Air Pollution

Area of stands damaged by air pollutants equals about 5 Mha (0.6 percent of total stocked area) (*table 2*). About 15 Mha of forest ecosystems (2 percent of total stocked area) have injured lichen cover. This relative damage is much better than in other industrialized countries.

After Russia became a new independent country, atmospheric deposition of air contaminants in Russia decreased during the last 4 years. This decrease is not the result of improved industrial technology, but an economic depression and a decrease in industrial production. *Table 3* shows the dynamics of sulfur dioxide emission from two industrial giant copper-nickel smelters in the European and Asian parts of Russia.

Table 3 — *Emissions of sulfur dioxide from Norilsk and Monchegorsk smelters (1,000 t/year) (Barkan, pers. comm.; Idimichev and others 1995; Pozniakov 1993; Vlasova and others 1991).*

Smelters	Years										
	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994
Norilsk	2648	2724	2325	2244	2242	2216	2180	ND [1]	ND	1863	1860
Monchegorsk	257	254	243	224	212	200	287	196	182	137	135

[1] No data available.

It is well known that the weakening of plants and their communities by pollutants changes the consortive interrelations in ecosystems. In particular, it may stimulate outbreaks of forest insects. On the other hand, the combined damaging effects of pollutants on the ecosystems already weakened by other anthropogenic or natural stresses are much stronger and faster than air pollution effects alone (Alexeyev 1995). This consideration is extremely important for the Russian forests because a significant part of them are damaged by multiple stresses.

Hopefully, the impact of atmospheric pollution in the next century will be less pronounced because of changing social awareness (Green Peace, and other ecological movements).

Conclusion

About one-third of tree stands and two-thirds of forest ecosystems of Russia are disturbed by natural and anthropogenic factors. Most prevalent are natural causes of disturbance; their impact affects 200 to 250 Mha of stands and 250 to 300 Mha of forest ecosystems. Natural disturbance processes are an integral part of natural forest life. Nevertheless, determinations of the size of damaged areas and trends of the observed changes must take into account management strategies of the Federal Forest Service of Russia. Not less than 45 Mha stands (6 percent of stocked area) are under the impact of anthropogenic influence. Atmospheric pollution is the most dangerous form of the anthropogenic stresses in forests in Russia.

Acknowledgments

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Air Pollution and Climate Change Effects on Health of the Ukrainian Forests: Monitoring and Evaluation[1]

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Abstract

Forests in the Ukraine are affected by environmental pollution, intensive forestry practice, and recreational uses. These factors make them sensitive to impacts of climate change. Since 1989 Ukraine has participated in the International Cooperative Program on Assessment and Monitoring of Air Pollution Effects on Forests (ICP-Forests). A network of monitoring plots has been established in 14 administrative regions and in the Crimea. From 1989 to 1995 the primary estimation criteria by the ICP Level 1 Program — defoliation and discoloration of forest stands — have been observed on 215 plots. Data obtained by the ICP-Forests protocol show that the recent decrease in air pollution was not followed by reduced forest defoliation. Thus, the estimation parameters established for that monitoring program are inadequate to determine accurate cause-effect relationships between forest health and external impacts. Defoliation and discoloration extended and expanded with the support of the USDA Forest Service's Forest Health Monitoring Program (FHM). More intensive and more informative monitoring has been implemented, complementing activities on the primary extensive monitoring network. Climate change impact may indicate pest damage, air pollution, fungi, or weather anomalies. These causes are difficult to identify at the extensive monitoring level. In summer 1995, climate change impact on the Ukrainian forests was estimated using the FHM protocols for forest vulnerability and adaptations assessment. Potential forests, main tree species distribution, and forest communities biodiversity was made estimated by using a geographical information system (GIS).

Introduction

Ukrainian forests cover 10 million hectares and about 14.3 percent of the total country. Forest distribution pattern, growth, and health are connected with temperate, continental climate conditions, except for a narrow strip of subtropical coastal forest in the southern Crimea. Ukrainian forests are composed of about 54 percent coniferous trees and 46 percent deciduous trees. The average annual growth increment is about 4 cubic meters per hectar. Managed plantations are about half of the forested lands. *Pinus sylvestris* L. (about 35 percent of forest stands), *Picea abies* Dietr. (16 percent), and *Abies alba* Mill. (3 percent) are the main forest- forming coniferous species. *Quercus* spp. (22 percent), *Fagus sylvatica* L. (13 percent), and *Carpinus betularius* L. (2 percent) are dominant hardwood species. Softwood genuses are *Betula* spp., *Populus* spp., *Alnus* spp., and *Tilia* spp. (9 percent).

Ukrainian forests are considered valuable environment-forming factors. Among the countries of the former Soviet Union, Ukrainian forests have been strongly affected by anthropogenic sources such as the high levels of air pollution. Overall industrial emission of pollutants was about 17 million tons per year. The main sources are ferrous metallurgical plants and the coal industry. Industrial emissions consist of 21.4 percent particulate pollution, 30.2 percent SO_2, 8.0 percent NO and NO_2, 33.5 percent CO_2, and 6.9 percent hydrocarbons. It is estimated that motor vehicles add about 40 percent to the overall air pollution: more than 51 percent CO_2, 46 percent hydrocarbons, and 22 percent NO_2 of the total volume of emissions are caused by motor vehicles.

This paper discusses the spatial distribution and degree of anthropogenic damage to Ukrainian forests as determined by using the monitoring protocol of the ICP-Forests and the FHM programs.

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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Air Pollution

To evaluate the intensity of air pollution, we used generalized data on industrial pollutant emissions from a network of plots (grid size 16x16 kilometers) in 18 administrative regions and big industrial cities in the Crimea Republic from 1985 to 1993 *(fig. 1)*. Since 1985 until 1993, the total amount of pollutant emissions from industrial sources decreased by 52.8 percent. Air pollution levels are expected to continue decreasing with the industry recession.

To determine geographical zones that are homogenous in relation to air pollution, we used the cluster analysis method (Duran and Odell 1974). By using Chebyshev's metrics, data on the total amount of pollutant emissions, their sources, and dynamics were plotted on a dendrogram reflecting the degree of similarity between the territorial units *(fig. 2)*. The dendrogram shows that all the administrative regions can be divided into two classes that differ by total intensity and composition of emission. Further K-means clustering (Duran and Odell 1974) was based on the preliminary hypothesis of only two classes in the sample. The analysis shows that the first class includes the Donetskaya region and parts of Luganskaya, Zaporozhskaya, and Dnepropetrovskaya region (territorial units 3, 5, 7, 8, and 11). The second class includes the remainder units. The first class shows a higher density level of pollution as indicated by emissions within that area and the impact of stationary sources *(figs. 3, 4)*. This resulting classification scheme is used to study relationships between changes in forest health and atmospheric pollution.

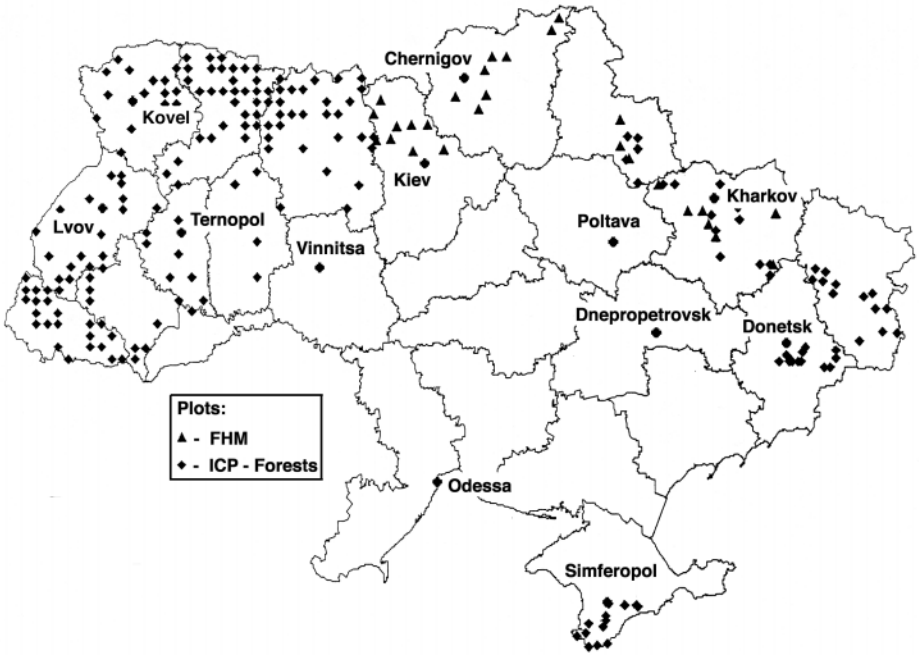

Figure 1 — International Cooperative Program on Assessment and Monitoring of Air Pollution Effects on Forests (ICP-Forests) and Forest Health Monitoring (FHM) plots.

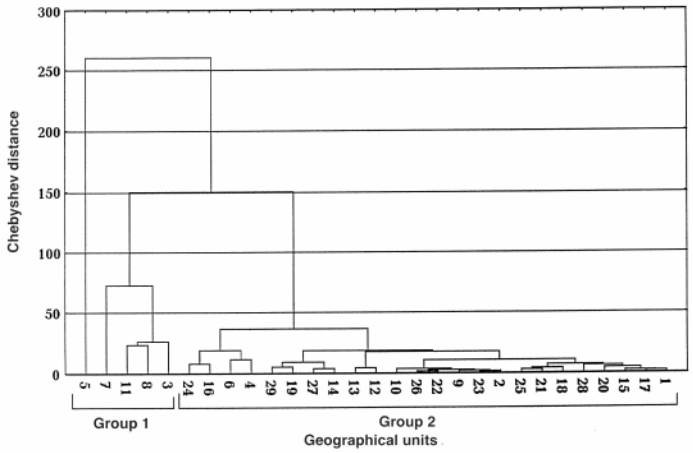

Figure 2 — Differences between geographical units by pollution.

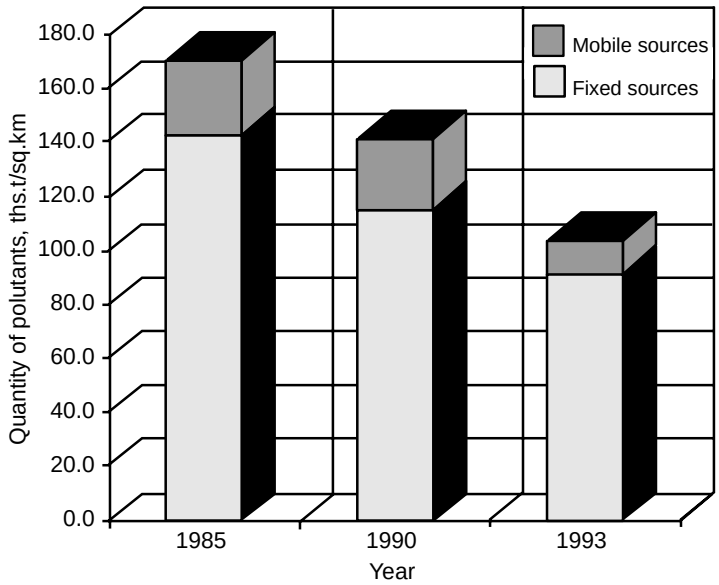

Figure 3 — Emissions by regions with intensive air pollution (Group 1).

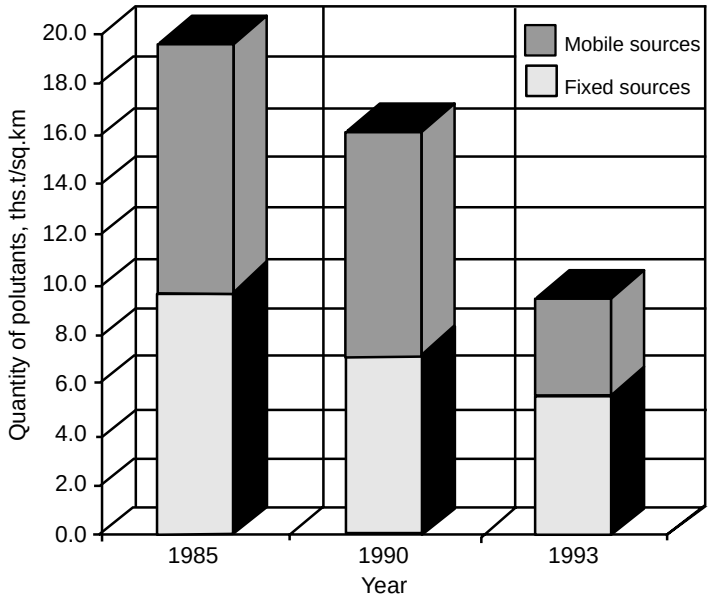

Figure 4 — Emissions by regions with low air pollution (Group 2).

Assessment of Dynamics and Distribution of Forest Damage in the Ukraine

In the Ukraine network, 215 forest monitoring plots have been established *(fig. 1),* but some plots were not observed annually. There were 53 plots in 1989 (552 coniferous and 736 deciduous trees) and 134 in 1995 (1,467 coniferous and 1,743 deciduous trees). The monitoring is designed to identify zones of forest decline and to study cause-effect relationships between changes in forest health and atmospheric pollution.

Degree of crown defoliation in the stands is a common indicator of monitoring that characterizes the forest condition. For a comparative analysis all data of observations are divisible by the following classes according to degree of crown defoliation: not defoliated, 0-10 percent; slightly defoliated, 11-25 percent; moderately defoliated, 26-60 percent; severely defoliated, more than 61 percent. In accordance with ICP Forests assessment criteria, defoliation less than 25 percent is not considered an indicator of worsening forest condition, since such values may be within the range of natural fluctuation of tree phytomass *(tables 1-4).*

Table 1 *— Defoliation of all species by classes (1995).*

Total plots	Total trees	Percent of defoliation Class 0 0-10	Class 1 11-25	Class 2 26-60	Class 3 61-99	Class 4 100	Classes 2-4	Classes 1-4
134	3,210	23.6	46.8	28.1	1.5	0.0	29.6	59.3

Table 2 *— Defoliation of all species by 10 percent classes (1995).*

Species	Total trees	Tree quantity (pct) in 10 percent defoliation classes 0-10	11-20	21-30	31-40	41-50	51-60	61-70	71-80	81-90	91-100
All	3,210	23.6	1.9	3.6	13.0	3.7	1.8	1.0	0.6	0.5	0.3
Conifers	1,467	28.1	0.3	5.6	9.3	3.4	1.4	0.9	0.5	0.5	0.0
Broadleaves	1,743	19.8	3.3	1.9	16.1	3.9	2.2	1.0	0.7	0.5	0.6

Table 3 *— Defoliation of conifers by age group and species.*

Defolia-tion class	Defolia-tion (pct)	Conifers age to 59 years *Pinus sylvestris*	*Juniperus communis*	*Abies alba*	*Picea abies*	Total	age equal or more than 60 years *Pinus sylvestris*	*Juniperus comm-unisabies*	*Picea abies*	*Abies alba*	Other	Total	Overall
0	0-10	14.3	100	33.3	12.2	50.5	21.4	100	95.5	28.8	100	24.2	28.1
1	11-25	71.4	0.0	50.0	61.1	36.1	51.6	0.0	4.5	27.3	0.0	48.1	46.2
2	26-60	10.7	0.0	16.7	25.6	12.5	25.3	0.0	0.0	43.9	0.0	25.8	23.9
3	61-99	3.6	0.0	0.0	1.1	0.9	1.7	0.0	0.0	0.0	0.0	1.9	1.8
4	100	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Total		100	100	100	100	100	100	100	100	100	100	100	100

The results showed an increase in tree defoliation during the observational period from 1990 to 1995 *(figs. 5-7)*. Fraction of trees that defoliated to 25 percent increased in 1990 to 1995 from 4 to 27 percent. Average yearly defoliation of the forest stands increased from 9.5 to 22.6 percent. Maximum defoliation as high as 44 percent was found in the Luganskaya region in 1995; the minimum was about 5 percent in the Ternopol'skaya region in 1990. Average defoliation for the whole monitoring period was highest in the Donetskaya (36 percent) and the Luganskaya (28 percent) regions probably because of air pollutants emitted in the Kharkovskaya and Rovenskaya industrial areas *(fig. 8)*. The lowest defoliation levels for the whole period were in the Ternopol'skaya (8 percent) and in the L'vovskaya (9 percent) regions. The greatest defoliation was seen in 1994.

Comparative analysis of tree species was accomplished by assessing differences in defoliation levels by years. Neuman-Keuls post-hoc test (Murphy and others 1988) was used to obtain the matrix of the differences reliability. Further analysis related only species with reliable differences from other ones (P-level less than 0.05). Then species' sets were divided into classes so that within a class there is no species with reliable differences in defoliation levels.

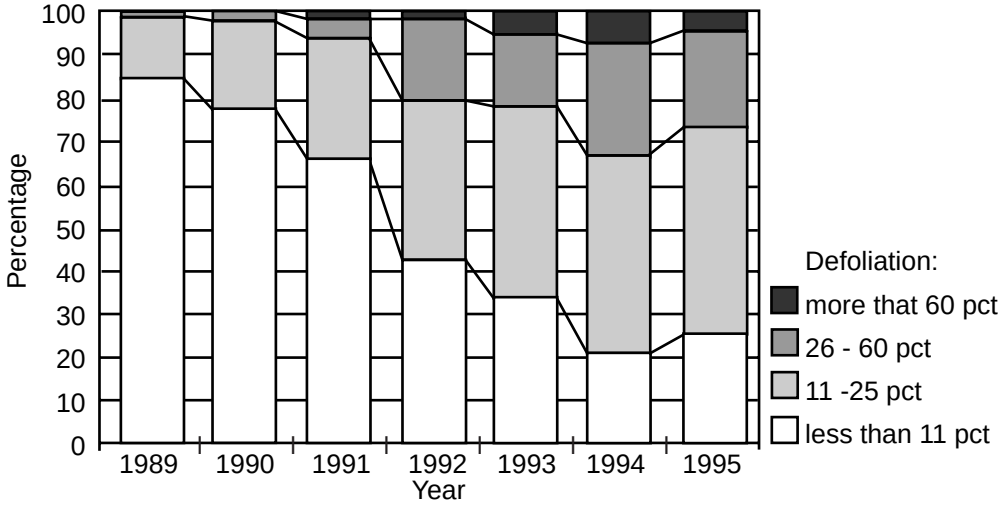

Figure 5 — Defoliation of conifers from 1989-1995.

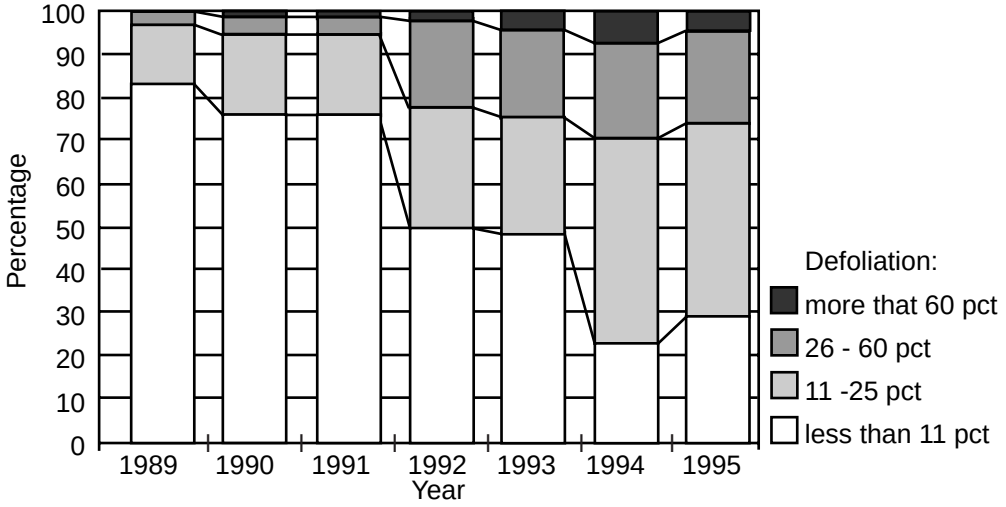

Figure 6 — Defoliation of broadleaves from 1989-1995.

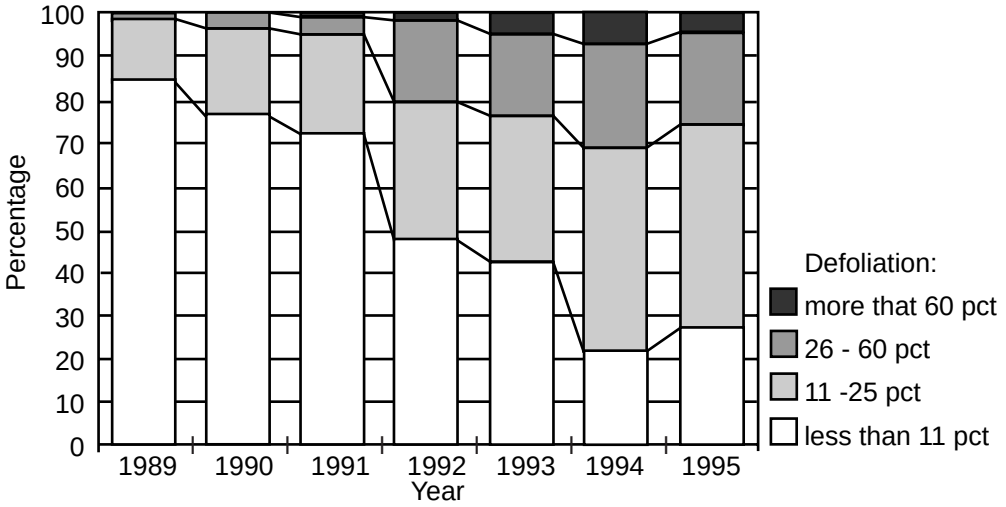

Figure 7 — Defoliation of all species from 1989-1995.

Table 4 *— Defoliation of broadleaves by age group and species.*

Defoliation class	Defoliation percent	Broadleaves												
		Age to 59 years					Age equal or more then 60 years							Overall
		Quercus robur	*Fagus sylvatica*	*Caprinus betularius*	Other	Total	*Quercus robur*	*Quercus petrea*	*Caprinus betularius*	*Fagussylv atica*	*Caprinus betularius*	Other	Total	
0	0-10	20.3	4.2	24.4	44.4	18.1	9.1	12.6	20.9	34.7	20.0	30.2	20.0	19.8
1	11-25	29.0	50.0	24.4	44.4	34.5	49.5	50.3	59.7	33.8	64.0	46.2	48.6	47.2
2	26-60	40.6	43.7	51.2	11.2	42.7	39.5	35.2	19.4	31.5	16.0	23.6	30.5	31.7
3	61-99	10.1	2.1	0.0	0.0	4.7	1.9	1.9	0.0	0.0	0.0	0.0	0.9	1.3
4	100	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Total		**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**	**100**

The data collected were used to group species by defoliation change rate by using cluster analysis of average defoliation values:

- Group 1—*Betula verrucosa, Ulmus spp., Quercus robur, Alnus incana, Pinus sylvestris;*
- Group 2—*Fagus sylvatica, Quercus petrea, Acer tataricus, Acer pseudoplatanus, Tilia cordata, Juniperus communis, Abies alba, Fraxinus exelsior;*
- Group 3—*Betula pubescens, Fagus orientalis, Carpinus orientalis, Carpinus betularius, Acer platanoides, Alnus glutinosa, Populus tremula, Pinus pallasiana, Padus racemosa;*
- Group 4—*Quercus pubescens, Picea abies, Juniperus oxycedrus.*

Group 1 had a relatively uniform defoliation growth rate *(fig. 9)*. Its average defoliation rate was greater than for any of the other groups (except for group 4 in 1993). Group 3 had the same pattern. However, the average defoliation rates for group 3 was reliably lower than the rest. The defoliation rate of group 2 steadily increased from 1989 to 1994, then noticeably decreased in 1995. Group 3 was different from others due to the high defoliation peak in 1993. We expect the cause of the peak was the particular climate conditions in 1993 and decreased defoliation in 1994 and 1995. The measured pattern of defoliation changes in the Ukraine forests showed that their photosynthetic function continues to decline. Defoliation of group 1—consisting of main forest forming species, *Quercus robur* and *Pinus sylvestris*—tended to steadily increase. Its correlation coefficient for linear trend D=3.7279xT+2.4141 was 0.8989. To isolate causes of defoliation change patterns will require longer time-series observations and more adequate and complete data sets of measurements.

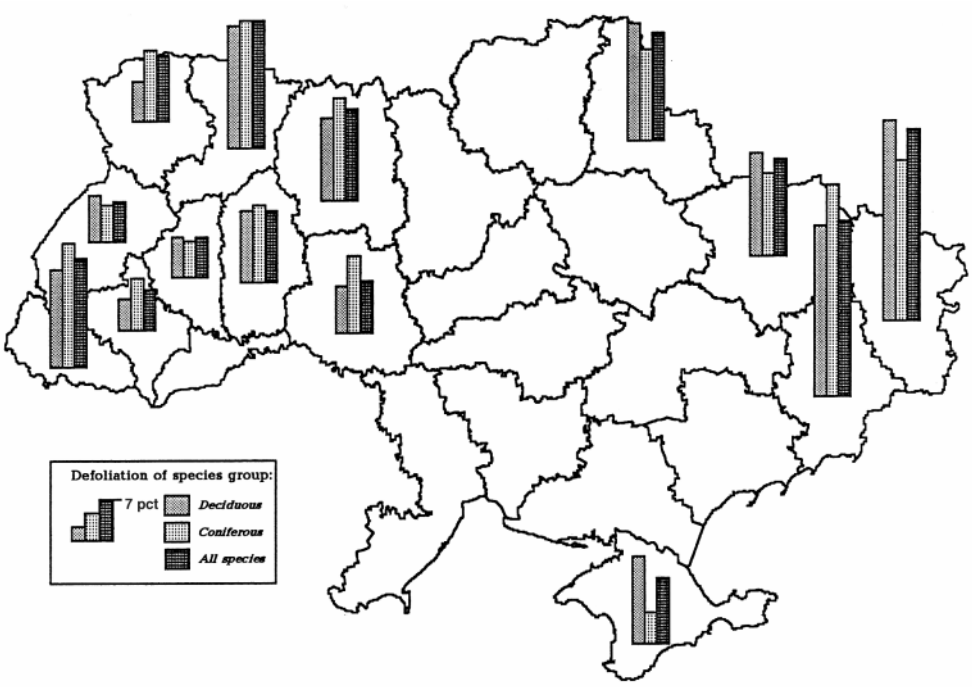

Figure 8 — The mean defoliation by regions in the Ukraine from 1989-1995.

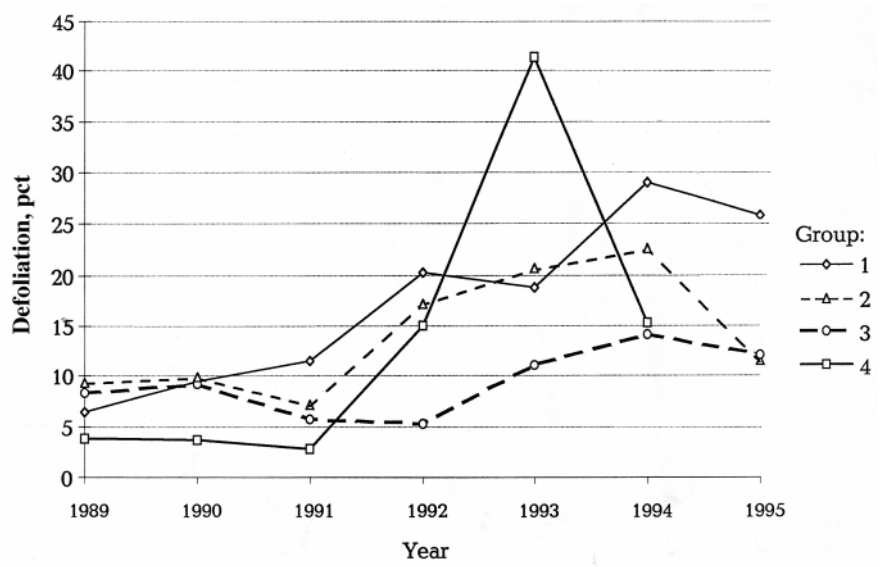

Figure 9 — Defoliation of species groups from 1989-1995.

The expected decrease of general defoliation has not occurred in spite of the considerable decrease of air pollutant emissions since 1992 in the Ukraine. The possible causes of this phenomenon include:

- Affect of accumulation of pollutants in soil from previous years;
- Global climate changes;
- General increase of anthropogenic factors on forest ecosystems (lumbercamp, pesticides and herbicides, recreation, drainage, and irrigation, etc.);
- Advection of pollutants to Ukrainian forests from upwind industrial countries.

In 1989, 82.9 percent of coniferous trees were not defoliated, 15.9 percent were slightly defoliated, and 1.4 percent were moderately defoliated. In 1995 the percent of non-defoliated trees fell (24.7 percent), and the percent of slightly and moderately defoliated trees increased (48.7 and 24.6 percent, respectively). In 1989, 84 percent of deciduous trees were not defoliated, 12.9 percent slightly, 3 percent moderately, and 0.1 percent were severely defoliated. In 1995 the portion of non-defoliated trees was the least (19.8 percent), and the portion of slightly, moderately, and severely defoliated trees increased (49.3, 30, and 0.9 percent, respectively).

Among the coniferous trees, *Pinus* and *Picea* (*Juniperus* in Crimea) had the highest defoliation, while *Quercus, Fagus, Caprinus,* and *Betula* were the most defoliated among the deciduous trees. However, the observed defoliation and discoloration may be induced by pests, diseases, air pollution, or weather anomalies. In order to isolate a real cause-effect relationship between forest conditions and external impacts, we used complex methods of analysis.

GIS Mapping

Sustainable forest management requires the development of methods for forest health and productivity forecasting. Assessment of the forests' vulnerability to global climate change is one of the generally recognized problems that needs such forecasting. The research team from the Ukrainian Forestry Research Institute has gained experience related to forests' vulnerability assessment to global climate change as a participant in the USDA Forest Service's Forest Health Monitoring Program (FHM). The computer geographic information system (GIS) mapping of potential vegetation land cover was the primary approach to assessing the impact of climate change on forests. Vegetation cover prognosis was based on Holdrige (1967)

life zones and on Ukrainian forest land classifications (Holdrige 1967, Vorobjov 1960). Climate change scenarios GFDL, GISS, UKMO, and CCCM (IPCC 1990) from general circulation models were used for projection with the classification models, and maps of potential forest distribution were used to assess the general pattern of climate change impacts. Large scale changes in biological productivity of forest stands were estimated with climate-related productivity indexes (Anuchin 1977). As an initial approach to describe future forest composition, the ranges of climatic species were modelled to determine dominant tree species. Incorporation of forest ecosystem models (gap-type models) provided more detailed forecasts for the most vulnerable forest sites. However, gap-models are inadequate to assess conditions in the Ukraine, and serious problems within the use of gap-models developed in other regions. Ukrainian silviculture uses particular assessment indices for plant response to climate change that are different from the gap-models requirements.

The methodological void in the assessment approach is between the large-scale level of climate conditions and the "locally specific" gap-models that are difficult to spatially extrapolate. Thus, methods for mid-scale impacts assessment were proposed by using the Ukrainian school of forest typology (Alekseyev 1929, Lavrinenko 1954, Pogrebniak 1968, Vorobjov 1960), which synthesizes forest-typological maps from mid-scale maps of soils, hydrology, and relief (1:100,000 - 1:1,000,000).

There are numerous forest-typological classification schemes developed by the Ukrainian school (Ostapenko 1978). They mainly differ by forest land classification criteria used to determine soil humidity and trophic conditions. The latest research involves the improvement of quantitative indices for the criteria and extension of climatic and edaphic grid models onto representation of trophically poor, more warm, and more dry sites. To make mid-scale maps of forest growth conditions (e.g.,forest land maps, forest edaphic maps), forest typology schemes require synthesis of two maps: forest trophic conditions and soil moisture availability. Forest trophic conditions may be derived from available soil maps: tables of correspondence can be established between soil types and trophic indexes compiled from empirical data. However, a preferable for trophic index evaluation method is to estimate the content of phosphorus and potassium in the root inhabited layer of soil. The territorial units of the resulting mid-scale edaphic maps describe edaphic conditions with some degree of generalization, depending on the scale and accuracy of source maps. However, the forest-typological meaning of such an edatope is subject to study. Actually, adjustments of forest estimation methods to the "mid-scale" edatope are required. This methodology is applicable to the broad scale assessment of global climate change and damage caused by air pollution.

Discussion

This assessment of the relative changes of tree foliage suggests a progressive deterioration of crown condition from 1989 to 1994 followed by a slight improvement in 1995. The highest level of defoliation was found among the main conifers, *Pinus* spp. and *Picea* spp., and among the main broadleaves *Quercus* spp., *Fagus* spp., *Carpinus* spp., and *Betula* spp. Forests of the southeast region were the most affected, and the most damaged forests are located in the regions with higher pollution density. However, air pollution is not the only influence on forest health because climate conditions of this region are also not favorable to forest growth.

The set of estimation parameters used for the initial forest monitoring program are not adequate for the determination of real cause-effect relationships between forest conditions and external impacts in the Ukraine. These impacts are air pollution, pests, fungi diseases, unfavorable weather conditions, and intensification of anthropogenic pressure on forest ecosystems. Study of the global climate change influence on forests shows the attributive data composition that is necessary to resolve this problem.

To obtain adequate forest ecosystem data, we must develop a multilevel monitoring system that includes extensive and intensive items. Since 1995 an extensive monitoring system designed according to the ICP-Forests approach was extended with the support of USDA Forest Service's Forest Health Monitoring Program (FHM). This observation program can provide data collection methods that may be used for gap-model development and forestry statistics interpretation. In summer 1995 the Ukraine implemented these methods with the support of the FHM program. A data base has been established to gather data on 32 plots. In addition, more intensive programs to observe Ukrainian forest ecosystems have also been prepared.

To effectively use these methods, they must include forest typology classification of forest growth conditions. This will ensure proper interpretation of the monitoring results and adequate assessment of forest condition dynamics affected by various external factors, including global climate change.

Acknowledgments

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Forest Condition in Latvia[1]

Madis Sipols[2]

Abstract

*Systematic assessment and observation (survey, inventory) of forests in Latvia has been underway since the 1700's. Latvia's forests are in the boreal/temperate forest zone and cover 44 percent of the country. Forest growing conditions are subdivided into five site class types: forests on dry mineral, wet mineral, wet peat, drained mineral, drained peat soils. The dominant tree species in Latvia are Scots pine (*Pinus sylvestris *L.), 39.7 percent; Norway spruce (*Picea abies *Karst.), 20.6 percent; and common birch (*Betula pendula *Roth.), 28.4 percent. Systematic assessment of air pollution impacts on forest health in Latvia was started in 1990 by the International Cooperative Program on Assessment and Monitoring of Air Pollution Effects on Forests - ICP Forests. Assessment methods were changed to the USDA Forest Service's Forest Health Monitoring Program in 1994. Assessments of forest ecosystems in Latvia show the alteration and disappearance of stand structure of typical, common forest ecosystems (especially Scots pine stands) , and the invasion of plant species non-typical to these forests, which has been caused by human activities and air pollution (sinantropization). It has also been found that Norway spruce stands will be endangered in the near future because of their sensitivity to environmental factors, and that currently forest stand health is mainly affected by localized entomological damage (insect and fungal diseases). In addition, environmental pollution has caused biological destabilization in forest ecosystems.*

Introduction

Latvia is one of three small Baltic countries transitioning to a free market economy. Latvia occupies 64,000 km^2, and 44 percent is covered by forests, which are of great importance for the stability of the economy, landscape, environment, and flora and fauna of Latvia. The first forest inventory in Latvia was carried out in 1788 for the Lencu estate. Inventories of all the biggest estate forests were done until the beginning of the 1900's. Then the process was periodically repeated after intervals of 10 to 15 years, including measurement and assessment of a forest stand and its health condition. On this basis long-term forest changes, even for 100 years, were assessed.

During the late 1900's, when environmental pollution significantly increased, there was a necessity for more comprehensive forest assessment methods. In 1990 the European program, "International Cooperative Program on Assessment and Monitoring of Air Pollution Effects on Forests (ICP Forests)," was implemented all over the territory of Latvia. In 1994 establishment of sample plots for the Forest Health Monitoring (FHM) program was started. This program is a component of the United States' Environmental Monitoring and Assessment Program (EMAP).

This paper discusses the current status of forests in Latvia as determined by the newly implemented Forest Health Monitoring Program.

Forest Composition

The forests in Latvia are in the boreal / temperate forest zone. The total area covered by forests has changed substantially over time. Agriculture lands freed from forests have been reforested as a result of various events such as wars or plague. The productive forest area has changed from 25 percent in 1923 to 27 percent in 1949, 38 percent in 1973, and 44 percent 1995 *(fig. 1)*.

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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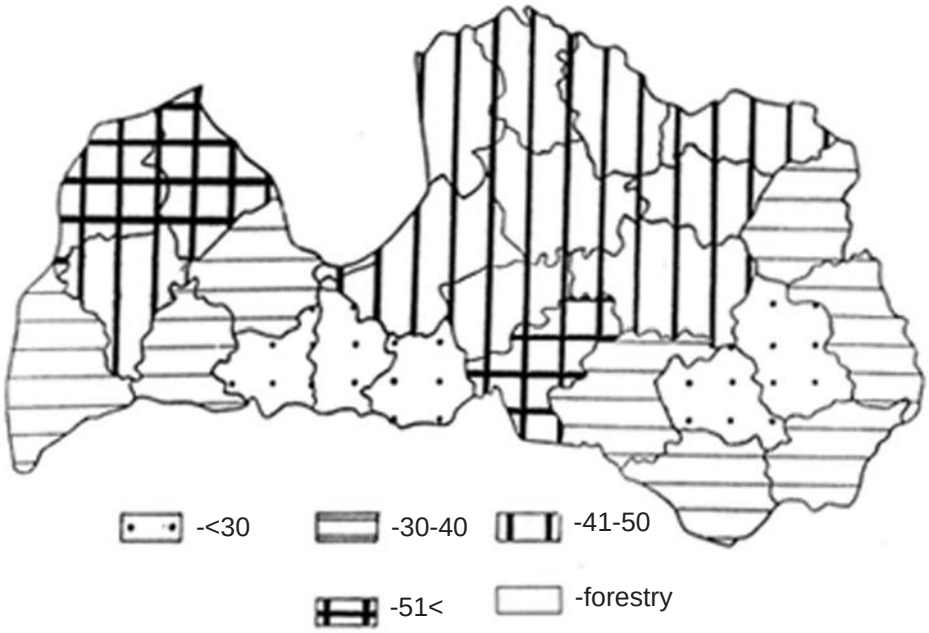


Figure 1 — Forest cover in Latvia by percent.

Forest growing conditions in Latvia are varied and include dry mineral soils (58 percent), wet mineral (10.4 percent), wet peat (12.0 percent), drained mineral (9.6 percent), and drained peat (10 percent). The main tree species in Latvia are Scots pine (*Pinus sylvestris*) (39.7 percent), Norway spruce (*Picea abies*) (20.6 percent), common birch (28.4 percent), common alder (2.4 percent), aspen (2.5 percent), and grey alder (5.5 percent) (*fig. 2*). All forests are divided into three categories according to their functions and importance (ecological, economical, etc.):

- **Protected forests** _____ **(9.6 percent)**
  - Nature reserves _____ 38,700 ha
  - National park forests _____ 51,600 ha
  - Nature park forests _____ 15,000 ha
  - Nature protected forests _____ 87,600 ha
  - Soil protected forests _____ 72,200 ha
- **Restricted management forests** __ **(16.3 percent)**
  - Landscape forests _____ 55,100 ha
  - Green zone forests _____ 244,000 ha
  - Environment protection forests __ 225,700 ha
- **Commercial forests**_____ **(74.1 percent).**

Scots pine occupies the greatest part of the forests in all three categories. Scots pine is also widely represented through all soil types — from sandy to fertile soils, as well as in peat soils. Scots pine is a biologically stable tree species, stable in windstorms, and relatively less damaged by insects and fungi diseases. Compared with Norway spruce, Scots pine is more resistant against various stresses and often reaches 150 to 300 years. However, for the last 20 years the condition of pine stands have been deteriorating. When the European system of forest monitoring was carried out from 1990 to 1995, Scots pine had a high level of defoliation for all age classes — 30.0 percent in 1990, 33.2 percent in 1992, and 24.1 percent in 1995 (*table 1*).

Forest Damage

The methodology of the FHM system is different from the previous systems and uses more detailed condition indicators. About 40 percent of the planned sample plots are established. The first results of this research show that the crown density

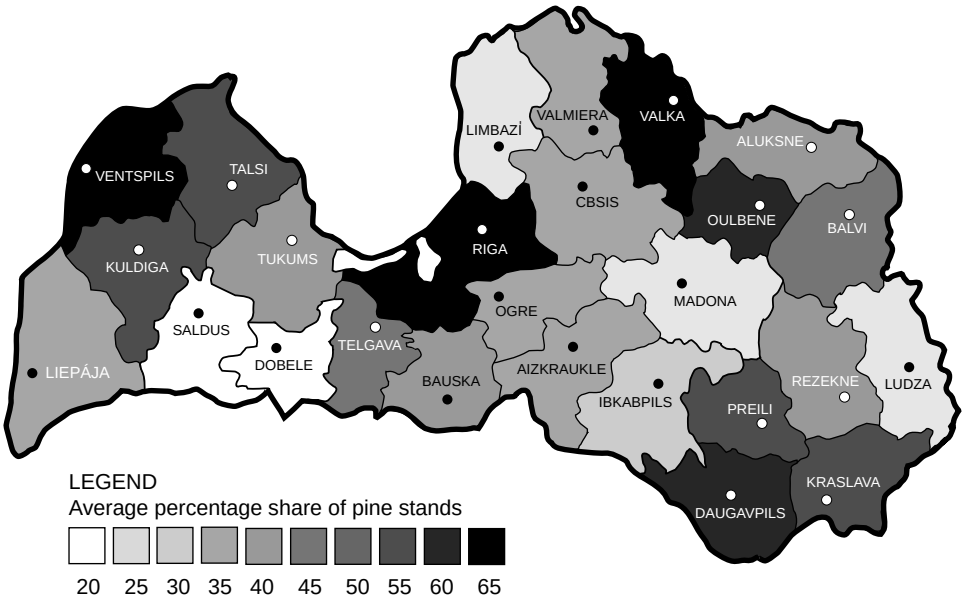


Figure 2 — Distribution of Scots pine (*Pinus sylvestris* L.) in Latvia.

Table 1— *Changes in defoliation by most represented tree species (1990-1995).*

	Percentages of defoliation by mean age classes (years)											
Tree species	**1990**		**1991**		**1992**		**1993**		**1994**		**1995**	
	<60	**>60**	**<60**	**>60**	**<60**	**>60**	**<60**	**>60**	**<60**	**>60**	**<60**	**>60**
Scots pine	27.0	32.2	28.9	33.0	31.6	34.1	29.1	30.4	27.1	27.2	24.1	24.1
Norway spruce	12.9	23.8	14.8	20.5	15.6	21.1	15.3	20.3	14.3	19.4	13.6	18.3
Common birch	16.3	17.9	18.7	18.8	19.5	21.1	18.1	19.5	14.5	18.3	13.1	16.3
Common alder	10.6	7.7	14.1	9.0	14.3	11.5	14.1	12.2	11.6	10.3	13.4	9.6
Aspen	13.2	15.1	15.0	18.2	18.3	20.3	18.3	18.4	17.5	16.5	14.5	15.0

of conifers on average is less than the broadleaved species (*figs. 3,4*). Usually crown damages are associated with different insects. For example, in 1995 high levels of defoliation of Scots pine in the vicinity of Riga was associated with *Gilpinia pallida* Kl. and *Lymantria monacha* L. The total damaged area there was about 10,000 ha. In 1992/93 *Pissodes piniphilus* Herbst caused serious crown damage on Scots pine.

At the same time invasion of new uncommon plant species in the forests on dry sandy soils have been observed during the last 10 years. At several levels these new species have become dominant. Scots pine stands undergo the process of sinantropization, and the natural characteristics of these forests have changed and disappeared.

The second most important tree species is Norway spruce, which is biologically unstable and very sensitive towards secondary factors such as insect damage and fungus diseases. Currently, Norway spruce is seriously endangered. ICP Forests monitoring data from 1995 shows that 8 percent of all observed spruce are damaged, and 70 percent are damaged by *Ips typographus* L.

Forest health is closely related to environmental changes (air pollution, climate, urbanization, etc.). An important environmental quality indicator is the national emission of SO_x, NO_x, CO, particulate pollution, and other substances. The total emission amount of gases and particulate matter from 1991 to 1995 has decreased. In 1991 the total emission of gases and solid substances was 144,100 tons/year; in 1992 — 92,310 tons/year; in 1993 — 87,130 tons/year; in 1994 — 103,840 tons/year; and in 1995 — 86,440 tons/year (*fig. 5*).

Figure 3 — Crown density of Scots pine in Latvian forests (FHM plots established in 1994, 1995).

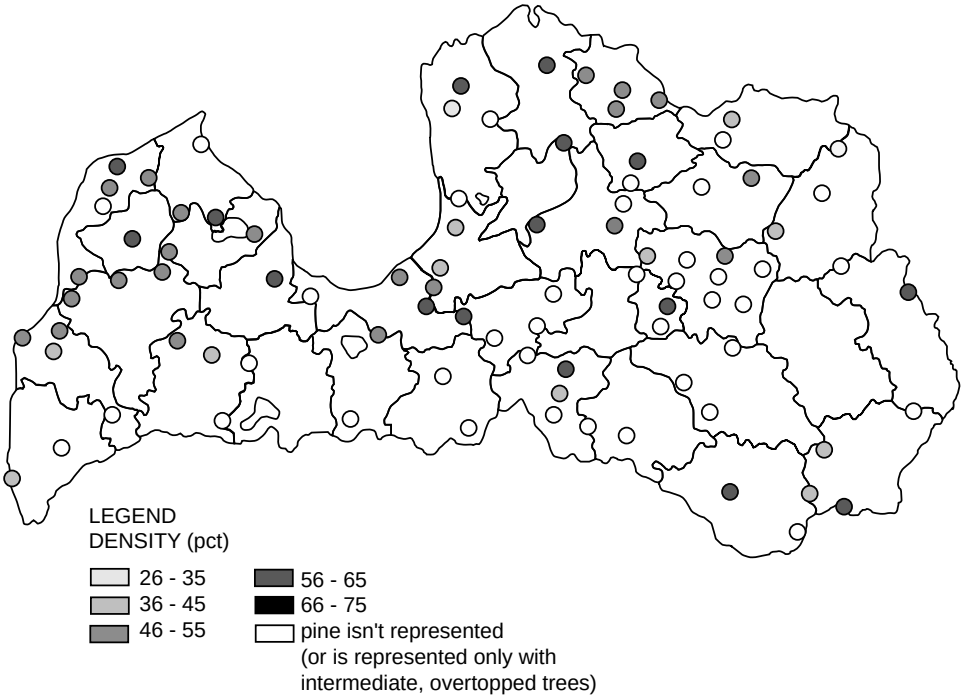
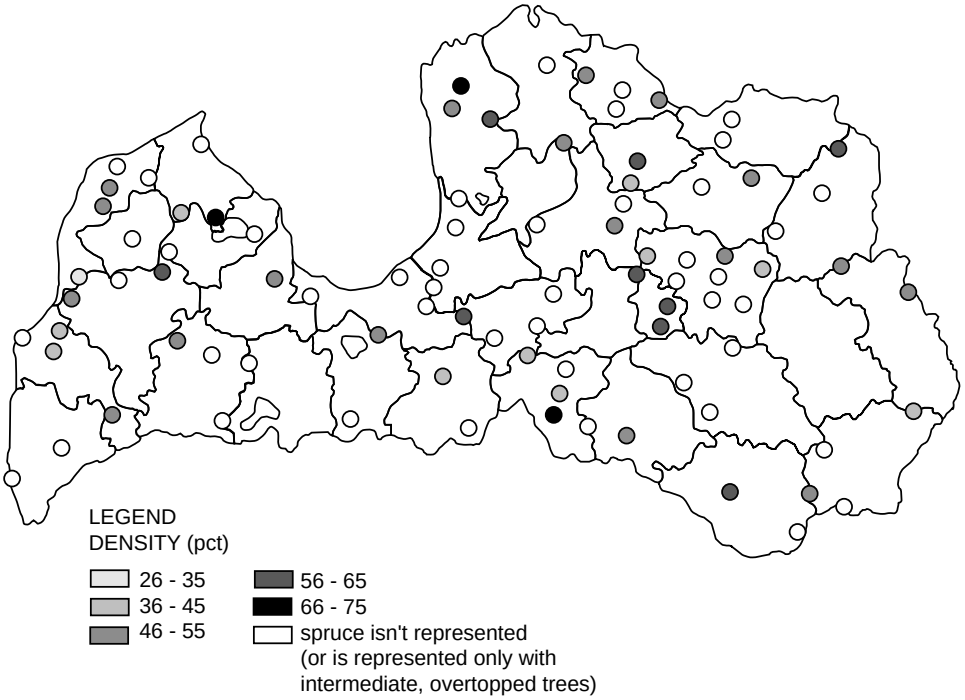


Figure 4 — Crown density of Norway spruce in Latvian forests (FHM plots established in 1994, 1995).



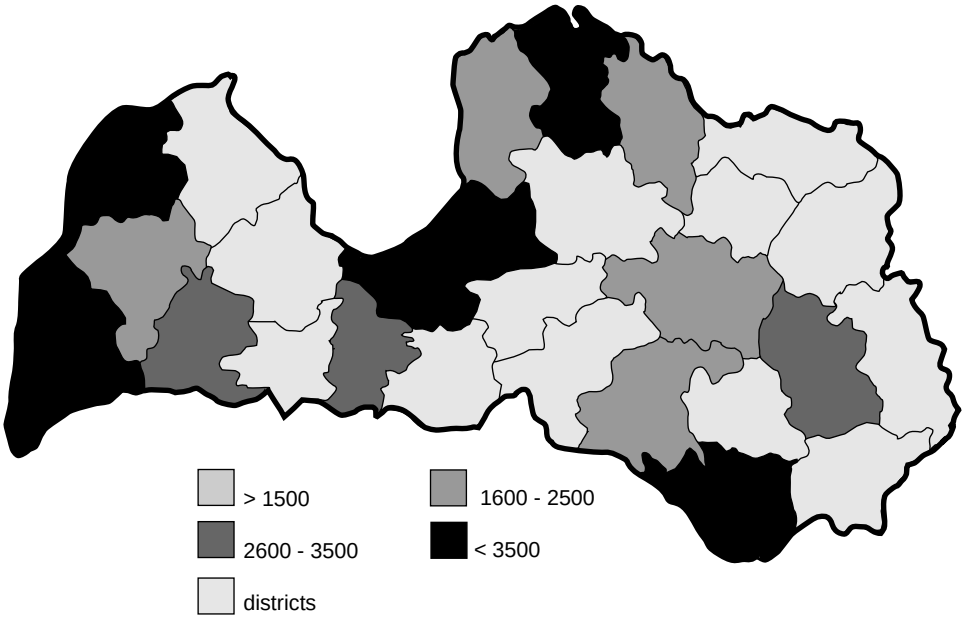


Figure 5 — Total emissions of gases, solids, and other substances (tons/year) in Latvia in 1995.

Summary

The area covered by forests in Latvia has increased in recent years. Health of trees is mainly affected by local entomological damages including cyclical repetition of insect damages. During recent years intensity of insect damage has not reached the catastrophic levels observed in earlier years. Environmental pollution impact on ecosystems has also changed forest stand structure (especially in Scots pine stands) and a process of biological destabilization has been observed

Acknowledgments

I thank Laurie Dunn for technical editing of this manuscript.

Forest Health Monitoring and Forestry Implications in the Czech Republic[1]

Martin Cerny, Pavel Moravcik[2]

Abstract

In recent years, a forest monitoring program in the Czech Republic was extended into more detailed monitoring that aimed to describe the extent of changes in forest vitality and identify the nature and the main causes of these changes on local and regional scales. Studies were undertaken in six mountain areas in the Czech Republic. The program of regional forest monitoring is divided into three levels according to the extent of evaluation of the parameters of forest stand health and other components of the forest ecosystem. Level 1 is large scale monitoring in a 1 by 1 km grid of permanent plots. The total number of plots in a single regional study varies from 60 to more than 500. The monitoring at level 1 plots includes a visual assessment of a broad set of features of the health state of individual trees, repeated yearly. Assessment of health includes measurement of tree diameter and height and a basic description of growing conditions. At monitoring level 2 the research assessment is extended to other parameters that characterize the forest stand and environment. The number of plots is usually 5-10 percent of level 1 plots. Monitoring level 3 includes analysis of the processes of nutrient cycling. Detailed analysis of stand structure is done at the plots, including biomass measurements. Results of field measurement are recorded into a database which allows a logical organization of a large amount of data and effective processing of them. Results of monitoring are analyzed using statistical methods and modeling. A geographical information system (GIS) is used for further analyses and for a final interpretation of results. From some studies, 4-5 years of results are now available. The studied regions cover a broad range of conditions, making it possible to assess global trends in the health of Czech forests.

Introduction

Forests in the Czech Republic cover an area of 2,642,064 ha. This corresponds to 33.4 percent of the total area of the Republic. Mean timber volume of forests is 225 m^3/ha and mean annual increment is 6.91 m^3/ha. Conifer species occupy 79 percent of forest area, and the main tree species is Norway spruce, occupying 54 percent of forest area. The primary forest regions of the Czech Republic are mountains located mainly along the mountainous border *(fig. 1)* areas.

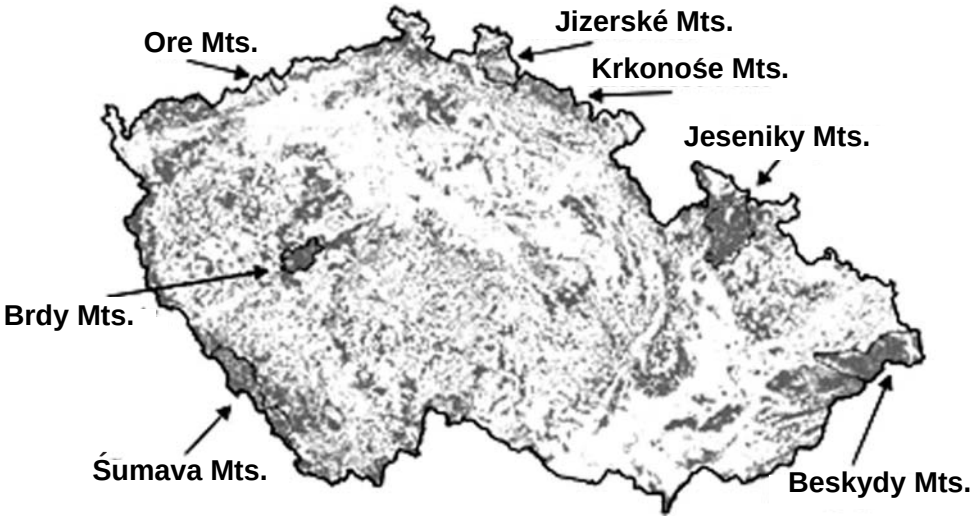


Figure 1 — Forest coverage of the Czech Republic, and location of main mountain ranges (interpretation of LANDSAT TM image done by Stoklasa TECH, Prague).

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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The Czech Republic, particularly the north-western part (Ore Mountains) has been strongly influenced by air pollution for the past several decades. A well-known "ecological disaster" began in the Ore Mountains in the 1950's and 1960's. Ten years later severe problems started in the Jizerske Mountains and Krkonose Mountains. The significant decrease of land covered by forests is visible in these previously forested areas (*fig. 1)*; however, almost all deforested areas are now covered by young stands.

At the beginning of the 1980's the symptoms of forest decline were also observed in northern Moravia (Beskydy and Jeseniky Mountains), and by the mid-1980's symptoms were observed in western and southern Bohemia (Brdy Mountains, Sumava Mountains, etc.).

At the same time the forests have been impacted by air pollution in some areas, increased forest increment has been recorded in the Czech Republic based on the data from regular forest inventories and data from permanent research plots. Thus, it has been proved that during the past 100 years, the current height increment of Norway spruce increased from 25 to 35 percent. On the other hand, the clear differentiation of tree growth within individual forest stands related to the symptoms of forest decline has been documented by data of tree ring analyses *(fig. 2):* the diameter increment decreases with increase of health damage symptoms (e.g.,tree defoliation). This example of the contradiction between forest decline and the continuous increase of height increment of Norway Spruce in the mountains of the Czech Republic clearly indicates the need for additional research.

This paper discusses the various forest health monitoring programs implemented in the Czech Republic that provide current data to forest policy makers for the improvement of forest health.

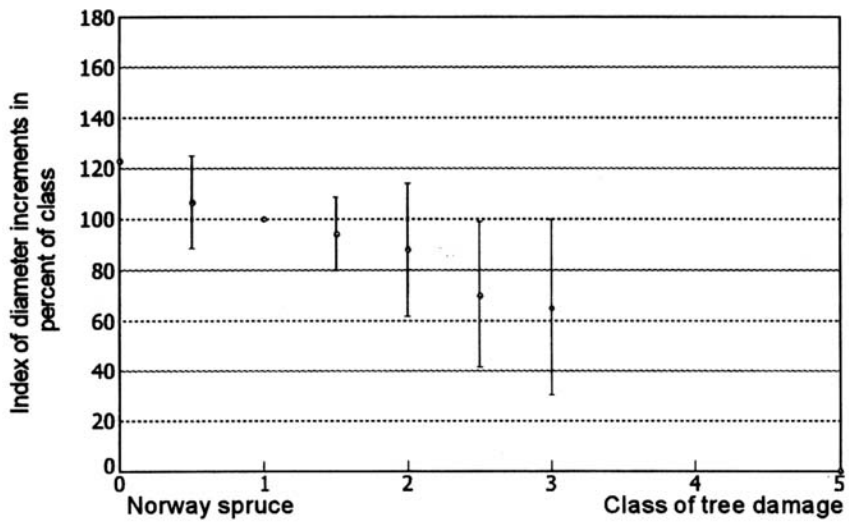

Figure 2 — Relationship between tree damage (classes of tree defoliation) and diameter increment. Index of diameter increment compares the current increment of the past 5 years to the previous 5-year period.

Initial Conditions and Forest Health Monitoring Programs

Forests of the Czech Republic are endangered by the continuous influence of a range of stress factors. These factors disrupt forest ecosystems and cause the forest decline. The significance of potential stress factors is time and area specific. Among the stress factors, sulfur dioxide is still important, although it has gradually decreased over the past years. Ozone, nitrogen oxides, and the influence of potential climatic changes should also be considered. The impacts of complex stress factors have not been sufficiently described.

The pan-European forest monitoring program (ICP Forests) has been implemented in most of the European countries to provide policy makers with basic information on forest health. Within this program a standardized methodology has been developed, and data exchange is supported. The Czech Republic has participated in the ICP Forest program since its establishment in 1986.

However, in the Czech Republic several systems of forest monitoring or research monitoring have been established over the past few decades. These systems have similar scientific aims as the ICP Forest programs, but were designed independently with more or less different methodologies.

In addition to the forest monitoring program, the regular forest inventory is used to estimate forest health status for practical forest management. Forest inventory data describing the whole area of the Czech Republic has been collected every 10 years. A planned national forest inventory program using a regular grid of permanent inventory plots will contribute additional information on forest health. Several other monitoring programs addressing different environmental components (air pollution, agricultural land, etc.) have also been implemented in the Czech Republic under the direction of the Czech Ministry of Agriculture and the Czech Ministry of Environment.

During the past two decades, remote sensing technology has improved significantly. With the support of terrestrial monitoring, remote sensing can serve as a fast and efficient source of information on forests health status. However, because the level of available information is limited to descriptive information on several stand characteristics, it cannot be used to determine causal relationships.

Forest management practice lacks information that is necessary for ecosystem oriented management on local and regional levels. Forest monitoring should serve as a permanent source of continuous information enabling forest managers to evaluate results of management procedures on a regional level and evaluate and reorient forestry policy on the national level.

A system of forest monitoring should be developed to both follow changes in forests, and to identify main potential stress factors. On the basis of the correlative analysis of forest state parameters and characteristics of the growing environment, hypotheses can be derived and then tested by the detailed research on ecosystem mechanisms.

Objectives of Forest Monitoring

Forest monitoring is defined as the long-term investigation of the state of the forest and growing environment by using a set of selected parameters that allow evaluation of forest changes caused by environmental changes and forest management practices.

The primary aims of forest monitoring programs are to clarify the extent of changes in forests in the Czech Republic, describe the character of these changes, and estimate the main causes of changes. The forest monitoring program should also be a unified and internationally compatible system.

Structure of Forest Monitoring

The forest monitoring program in the Czech Republic is differentiated into separate subprograms according to the topics addressed within the program *(table 1)*.

- **Permanent Sample Plots:** The set of permanent sample plots was established in the 1960's to collect data on the growth of stands of the main tree species found in the Czech Republic. These data were collected to develop regional growth and yield tables. Most of the plots are still active and produce valuable information on the development of the forest ecosystem.

- **Regional Forest Monitoring:** Regional forest monitoring is derived from the European program of forest monitoring. For the purposes of local forest managers, and regional authorities in forestry and the environment, the information from the sparse European grid of monitoring plots (16 x 16 km) is not satisfactory. For that reason a dense grid, 1 x 1 km, has been established in selected regions of the Czech Republic. Currently, there are five such regions, and several additional smaller areas are also covered by using this approach.

- **Large Scale Forest Inventory:** Large scale forest inventory by using a terrestrial survey in the grid of permanent inventory plots is one of the main tools for collecting data about forests and forest production. Recent developments in large scale forest inventory methodology allow the evaluation of forest ecosystem status including biodiversity.
- **ICP Forests:** The Czech Republic has participated in the program of European monitoring (ICP Forests) since its beginning in 1986. The data collected in the 16 x 16 km grid are used for the purposes of European forest policy.
- **Remote Sensing:** Satellite imagery from LANDSAT TM data is widely used for the estimation of forest health status.

Forest Monitoring Projects in the Czech Republic

The forest monitoring program in the Czech Republic is coordinated by the Czech Ministry of Agriculture and Ministry of Environment *(table 2)*. Except for ICP Forests, all monitoring projects are funded by grants.

Methodology of Terrestrial Forest Monitoring

The methodology of terrestrial forest monitoring has been derived from European unified methodology for ICP Forests *(table 3)*. The methodology is split into three levels. Level one is applied to all monitoring plots. Level two is used in about 10 percent of the monitoring plots, and separation into level two is defined by the additional cost of methodological procedures.

Table 1 *— Different programs of forest monitoring in the Czech Republic.*

Program	Subject	Characteristics	Size of area unit	Intensity of research	Interval (years)	Application of results
Permanent sample plots.	forest ecosystem.	forest health, environment, production, forest structure.	-	medium to high	1-5	cause-effects studies, modeling, support of decisions on conceptual level.
Regional monitoring.	site, region.	forest health, environment, production.	small	high	1	detailed characteristics of selected region, planning on the regional level, forest improvement measures.
Large scale inventory.	region, country.	structure of forests, production, ownership, quality of management, forest health.	medium to small	medium	5-10	large scale forest inventory, forestry planning, management politics.
ICP Forests.	country.	forest health, environment, production.	large	medium	1	current state of forest health on country level.
Remote sensing.	country, region, site.	forest health, changes in forest coverage.	small	low	1 and more	current information on state of forest.

Table 2 — Terrestrial forest monitoring network in the Czech Republic.

Project	Forest area covered, ha	Grid km x km	Number of plots	Level of monitoring	Time of project execution	Run by
National monitoring network						
Permanent sample plots	2,650,000	non regular	900	I, II	1980+	IFER [1]
ICP Forests	2,650,000	16 x 16	126	I, (II)	1986+	FGMRI [2]
Large scale forest inventory	2,650,000	1.41 x 1.41	13,000	I, (II)	([3])	(IFER) [4]
Regional monitoring network						
Brdy Mts.	23,000	1 x 1	220	I, II, III	1989+	IFER
Sumava Mts.	60,000	1 x 1	528	I, II, III	1990+	IFER
Krkonose Mts.	30,000	1 x 1	262	I, II, III	1991+	IFER
Beskydy Mts.	45,000	1 x 1, 4 x 4	176	I, II, III	1992+	IFER
Jizerske Mts.	20,000	1 x 1	120	I	1992-93	IFER
Other local projects	25,000	1 x 1	55	I	1992+	IFER

[1] Institute of Forest Ecosystem Research, Jilove u Prahy

[2] Forestry and Game Management Research Institute, Jiloviste-Strnady

[3] Planned for 1999.

[4] Development of methodology phase

Level three includes procedures that evaluate the ecosystem nutrient cycle and is performed to characterize growing conditions of the forests. The data of field assessments are stored in the MONitoring dataBASE. Version 5 of MONBASE allows efficient computer storage of data as well as implementation of the set of basic processing routines. Pre-processed data are further analyzed and evaluated by using GIS techniques. Point layers of terrestrial monitoring data are combined with interpretation layers (e.g., interpreted satellite images, forest soil maps, forest types maps, and climate maps). By using GIS techniques like co-kriging and multicriteria analysis, the monitoring data are processed and interpreted for further use in forest management practice.

Conclusions

The forest monitoring program in the Czech Republic has significantly developed during the past 5 years. The primary features of the program include a general monitoring approach that is performed in cooperation with other research programs on permanent research plots in the Czech Republic (forest production research plots, forest inventory). Forest monitoring data are interpreted for use by forest managers, and the emphasis is on developing cause-effect relationships.

The unified methodology of the program uses three levels of research monitoring (up to ecosystem level) and extends to cause-effect relationships. The regional forest monitoring network uses permanent research plots (including sites that are less common but important ecologically) and has been extended to several new regions.

The database used for forest monitoring is MONBASE version 5, which is a flexible tool for handling monitoring data. Progressive methods of data processing and evaluation include statistics, GIS, and modeling. Forest damage and production data are interpreted by decision makers and forest policy makers at both national and regional levels. Forests are then zoned according to the level of risk (as a result of multicriteria analysis), and they are managed on the basis of nutrition and species composition. By understanding these forest ecosystem processes, policy makers can apply the monitoring results in forest production research for better management of the Czech forests.

Table 3 *— Methodological elements for different terrestrial forest monitoring levels.*

Level of monitoring	Element of methodology	Extent	Interval
I	**Level one:**		
	basic descriptive data	plot	once
	humus forms	plot	6-12 yrs
	soil chemistry	plot	12 yrs
	needle/leaf chemistry	plot	6 yrs
	phytocenological description	plot	10 yrs
	natural afforestation	plot	3 yrs
	occurrence of lichens	plot	1 yr
	tree species	trees (all)	1 yr
	tree class	trees (all)	1 yr
	mortality of trees	trees (all)	1 yr
	occurrence of crown/stem break	trees (all)	1 yr
	defoliation of whole crown	trees (all)	1 yr
	defoliation of upper third of crown	trees (all)	1 yr
	vitality of the crown tip	trees (all)	1 yr
	occurrence of dead branches	trees (all)	1 yr
	mechanical injury to the stem	trees (all)	1 yr
	type of color changes (discoloration)	trees (all)	1 yr
	intensity of color changes	trees (all)	1 yr
	diameter at breast height	trees (all)	3 yrs
	tree height	trees (10-15)	3 yrs
	occurrence of secondary shoots	trees (5-10)	1 yr
	needle retention	trees (5-10)	1 yr
	occurrence of combs	trees (5-10)	1 yr
	crown ratio	trees (5-10)	1 yr
	angle of branching	trees (5-10)	1 yr
	type of branching	trees (5-10)	1 yr
	injury caused by insects	trees (all)	1 yr
II	**Level one plus:**		
	humus forms	plot	3 yrs
	soil chemistry	plot	6 yrs
	soil profile description & chemistry	plot	12 yrs
	needle/leaf chemistry	plot	3 yrs
	small root vitality	plot	6 yrs
	core analyses	trees (10)	once
	LAI measurement	plot	1 yr
III	**Level two plus:**		
	soil solution chemistry	plot	2 weeks
	wet/dry deposition chemistry	plot	2 weeks
	amount & chemistry of litter	plot	1 month
	air pollution/passive samplers	plot	2 weeks
	cont. measure. of tree diameter incr.	trees (10)	1 month
	detail stand structure description	plot	10 yrs

Dynamics of Forest Health Status in Slovakia from 1987 to 1994[1]

Julius Oszlanyi[2]

Abstract

*Slovakia is a mountainous and forested country (40.6 percent forest cover) in central Europe and has a large variety of vegetation zones, forest types, and a rich diversity of forest tree species. The most important tree species are beech (*Fagus sylvatica *L.), Norway spruce (*Picea abies *Karst.), oak species (*Quercus *sp.), Scots pine (*Pinus sylvestris *L.), silver fir (*Abies alba *Mill.), European hornbeam (*Carpinus betulus *L.), European larch (*Larix decidua *Mill.), poplars and willows (*Populus *L. sp.,* Salix *L. sp.), and other hardwood broadleaves. On the basis of results of the Forest Health Monitoring System from 1987 to 1994, the scientific information is presented for the following parameters: defoliation, discoloration, percentage of tree number in classes of damage, and percentage of salvage cut in the total annual cut. The percentage of trees in the defoliation classes 0, 1, 2, 3, and 4 were 14.7, 43.5, 36.2, 4.3, and 1.3, respectively (class 0 means healthy trees; class 4 dead trees). The discoloration classes (0, 1, 2, 3, and 4) were represented by 98.0, 1.7, 0.3, 0, and 0 percent for all tree species together. The average defoliation percentage from 1987 to 1994 decreased with time and is expected to further decrease in the following years. However, the percentage of annual salvage cut in the total annual cut increased between 1987 and 1994. Various biotic and abiotic factors influence forest health in Slovakia, such as air pollution, wind and snow, damages by beetles, sucking insects, and game. Global climate change seems to be the most important among them.*

Introduction

Forest health status is a phenomenon studied and monitored in central and eastern Europe, where the natural vegetation cover consists of forests and where forestry is an important part of the economy. Numerous scientific and scientific-technical projects have been involved in the process to describe the main factors influencing the health status of forest trees and to propose measures that could partially solve the problem of forest decline.

Because Slovakia is a mountainous and forested country (about 1.94 million ha. of forest, i.e., 40.6 percent of the total area), it is important to study changes in forest health status. Because of the mountainous character of the country, forested habitats in Slovakia encompass a wide range of environmental conditions and elevations (*table 1*). Trees are subjected to many natural stresses, including those induced by changes in water and nutrient relations, light, temperature, and biotic factors. Because of their longevity, trees usually react to the changes abruptly, and very often they are not able to adapt themselves to the changed growing conditions.

The dominant forest types in Slovakia are the beech (*Fagetum pauer*, *Fagetum typicum*), beech-oak (*Fageto-Quercetum*), and oak-beech (*Querceto-Fagetu*m, *Fagetum dealpinum*) types, which together cover about 52 percent of the total forest area. Beech-fir (*Fageto-Abietinum*), fir-beech (*Abieto-Fagetum*), and fir-beech with spruce (*Fagetum abietum - piceosum*) forests cover an additional *25* percent. Other locally important forest types include pine-oak and oak-pine *(Pineto-Quercetum* and *Querceto-Pinetum*, 1.96 percent), Cornel oak forests (*Corneto-Quercetum*, 0.91 percent), oak-ash forests (*Querceto-Fraxinetum*, 0.41 percent), elm-ash forest (*Ulmeto-Fraxinetum populeum* and *Ulmeto-Fraxinetum carpineum*, 1.07 percent), hornbeam-oak forest (*Carpineto-Quercetum*, 6.99 percent), acid oak-beech forest (*Fagetum quercinum*, 2.69 percent), linden-maple forest (*Tilieto-Aceretum*, 0.63 percent), ash-maple forest (*Fraxineto-Aceretum*, 2.28 percent), dealpine pine forest (*Pinetum dealpinum*, 0.96 percent), fir-spruce forest (*Abieto Piceetum*, 2.84 percent), mountain ash-spruce forest

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(1.02 percent), maple-spruce forest (*Acereto-Piceetum*, 0.53 percent), and dwarf pine forest (*Mughetum*, 0.60 percent).

According to the newest forest inventory results (Novotny and others 1994), the tree species percentage in the Slovakian forests include:

Species	*Percent*
Norway spruce (*Picea abies* Karst.)	27.2
European larch (*Larix decidua* Mill.)	2.0
Silver fir (*Abies alba* Mill.)	5.0
Scots pine (*Pinus sylvestris* L.)	7.0
Oaks (*Quercus* L. sp.)	14.3
Beech (*Fagus sylvatica* L.)	29.8
European hornbeam (*Carpinus betulus* L.)	5.6
Other hardwood broadleaves	4.5
Poplars and willows (*Populus* L. sp., *Salix* L. sp.)	3.5

Slovakia is situated in the central European region where prevailing northwest winds bring pollutants from the industrial areas in the Czech Republic, Poland, and Germany. Air pollution effects on forests have become critical in the past decades in Slovakia. Other causal factors affecting forest health in Slovakia include soil pollution, wind damage, snow damage, damage by bark-beetles, damage by wood-destroying beetles, damage by phytophagous and sucking insects, browsing damage, and bark peeling (game) damage.

Like most other European countries, Slovakia began a forest tree health monitoring program, which started in 1987 and continued through summer 1995. The ninth monitoring cycle concentrated on the permanent network of forest health monitoring plots (16 by 16 km grid). According to the methods used in the European forest health monitoring system, the permanent monitoring plots, which have no specific size, contain 15 trees close to the center of the plot and are used for measurement and observations.

This paper discusses the results of the forest health monitoring system in Slovakia that studied these parameters: tree stem circumference at the height of 1.3 m above the soil surface (the diameter at breast height calculated—dbh); biosociological position of trees (Kraft tree classes scale); loss of assimilation apparatus (defoliation); heights of three trees (by Blume-Leiss hypsometer); fertility of trees; mechanical damage on trees; fungi; and insects.

Table 1— *General survey of forested habitats divided into vegetation zones.*

Vegetation zone	Altitude (m a.s.l.)	Mean annual temp. (°C)	Annual precip. (mm)	Vegetation period (days)
Oak	<350	>8.0	<600	>165
Oak beech	350-400	7.5-8.0	600-650	160-165
Beech-oak	400-550	6.5-7.5	650-700	150-160
Beech	550-600	6.0-6.5	700-800	140-150
Fir-beech	600-700	5.5-6.0	800-900	130-140
Spruce-beech	700-800	4.5-5.5	900-1,500	115-130
Beech-spruce	900-1,050	4.0-4.5	1,050-1,200	100-115
Spruce	1,050-1,350	2.5-4.0	1,200-1,500	60-100
Dwarf pine	>1,350	<2.5	>1,500	<60

Methods

The numerous data and information obtained from the network of forest health monitoring plots (monitoring was performed August 11-20) provide excellent material for studies. Results that concentrated on the most visible changes on the forest trees document a more complicated system of causal factors responsible for the health of the forests. The variety of anthropogenic causal factor complexes have interacted with natural events and processes that induce stresses in forests and can be observed as ecological response.

Numerous papers (Lapin and others 1994, Lapin and others 1995, Lapin 1995) give information on trends of means of temperature and precipitation in different parts of Slovakia, starting in the year 1870. Results from localities situated in the lowland of south Slovakia and in the mountains in the north document a positive tendency in mean annual temperature and negative tendency in precipitation totals and in relative air humidity.

Results

Defoliation and discoloration of foliage are considered the best and most significant indicators of the health status of forest trees (*tables 2-6*). Both scientists and forestry managers are convinced that defoliation and discoloration are visible responses of the internal and external influences on the life of the individual tree (Novotny and others 1995).

For the evaluation we used the 5 degree scale of defoliation (Novotny and others 1995) to maintain consistency with data from other European countries. But in the Slovak Republic, narrower defoliation classes (each class is 10 percent) were used because this provides greater precision for the ecological evaluation of forest health and its dynamics in a longer period.

The internationally accepted scale denotes degree of damage (0 to 4) in terms of percent of foliage loss (0 to 100 percent):

0 - healthy trees (without symptoms), 0-10 percent
1 - partially ill trees, 11-25 percent
2 - ill trees, 26-60 percent
3 - very ill trees, 61-95 percent
4 - dead trees, 100 percent.

Table 2 *— Tree species percentage in defoliation degrees (1994) (Novotny and others 1995).*

	Degree of Defoliation						
	0	1	2	3	4	1+2+3+4	2+3+4
	Defoliation (percent)						
Tree species	**0-10**	**11-25**	**26-60**	**61-95**	**100**		
Fagus sylvatica L.	27.5	47.8	23.6	1.0	1.0	72.5	24.7
Quercus sp. L.	1.5	53.9	40.0	2.3	2.3	98.5	44.6
Carpinus betulus L.	30.8	45.7	22.9	0.3	0.3	69.2	23.5
Other broadleaf	3.2	23.0	52.9	16.9	4.0	96.8	73.8
Broadleaf total	19.3	45.1	31.0	3.5	1.1	80.7	35.6
Picea abies Karst.	8.4	41.9	43.6	4.9	1.2	91.6	49.7
Abies alba Mill.	15.6	36.0	37.6	8.6	2.2	84.4	48.4
Pinus sylvestris L.	4.6	40.5	48.1	4.8	2.0	95.4	54.9
Larix decidua Mill.	5.1	57.6	28.8	6.8	1.7	94.9	37.3
Coniferous total	8.2	41.5	43.5	5.3	1.5	91.8	50.3
Total	**14.7**	**43.5**	**36.2**	**4.3**	**1.3**	**85.3**	**41.8**

When the data on defoliation are put into a regression, the negative relationship is obvious. Decrease of defoliation percentage from 1987 to 1994 is related to the changes in the economy of the central European countries: the decrease in industrial activity has resulted in decreased pollution. Air and soil pollution has been responsible for the defoliation and discoloration of trees.

Defoliation is not the only and not the most significant indicator of health state of the forest trees; the salvage cut percentage is also a dynamic indicator. Despite of the fact, that the defoliation has the negative tendency in the studied years, the salvage cut percentage has a positive tendency (*table 7*). There are different causal factors responsible for the salvage cut amount in different years. Among them, the most important are the climate changes that seriously influence the growing conditions at each forest site and subsequently forest health and the stability of forest ecosystems.

The data concerning the degree of defoliation show a trend that is controversial to the trends of forest health status in Slovakia. The salvage cut and its percentage of the total cut is the most appropriate and useful indicator of the forest health status in Slovakia. The salvage cut quantifies the amount of the trees (wood) that had to be cut because of biological and abiological damage. Salvage cut quantifies the ecological instability of part of the forest and the bad health status of the forests. It also signifies the decline of forest health status because the salvage cut percentage is increasing permanently.

The annual cut in Slovakia has been relatively stable (average annual cut from 1991 to 1992 was 4.24 million m^3), but the increasing salvage cut documents the necessity of cutting dead, weakened , and damaged trees in increasing volume (*table 7*).

Discussion and Conclusion

As one of the most forested European countries, Slovakia is characterized by forests containing richness of tree species, ecosystems, habitats, and site diversity. However, the data from the monitoring plots network give us very interesting scientific-technical information that documents the worsening state of forest health, with negative prognosis for the future. Although the importance of the causal factors varies in the respective years, the tendency is clear, and it is negative.

Table 3 *— Tree species percentage in discoloration degrees (1994) (Novotny and others 1995).*

	Degree of discoloration						
	0	1	2	3	4	1+2+3+4	2+3+4
	Discoloration (percent)						
Tree species	**0-10**	**11-25**	**26-60**	**61-95**	**100**		
Fagus sylvatica L.	99.6	0.3	0.0	0.1	0	0.4	0.1
Quercus sp.L.	97.1	1.9	0.6	0.2	0.2	2.9	1.0
Carpinus betulus L.	96.5	3.2	0.3	0	0	3.5	0.3
Other broadleaf	97.1	2.4	0.5	0	0	1.9	0.5
Broadleaf total	98.4	1.3	0.2	0	0.10	1.6	0.3
Picea abies Karst.	99.3	0.3	0.4	0	0	0.7	0.4
Abies alba Mill.	100.0	0	0	0	0	0.0	0.0
Pinus sylvestris L.	90.6	8.9	0.5	0	0	9.4	0.5
Larix decidua Mill.	100.0	0	0	0	0	0.0	0.0
Coniferous total	97.5	2.1	0.4	0	0	2.5	0.4
Total	**98.0**	**1.7**	**0.3**	**0**	**0**	**2.0**	**0.3**

Table 4 *— Tree species percentage in defoliation classes (1994) (Novotny and others 1995).*

Tree species	Defoliation (percent) 0-10	11-20	21-30	31-40	41-50	51-60	61-70	71-80	81-90	91-100
Fagus sylvatica L.	27.5	28.6	31.9	6.3	3.4	1.3	0.7	0.2	0.0	0.1
Quercus sp. L.	1.5	24.3	52.7	12.8	3.1	1.0	1.3	0.2	0.2	2.9
Carpinus betulus L.	30.8	25.5	37.1	5.7	0.3	0.0	0.3	0.0	0.0	0.3
Other broadleaf	3.2	10.7	26.9	20.1	12.7	5.5	6.9	6.1	3.7	4.2
Broadleaf total	19.3	24.6	35.8	9.5	4.4	1.7	1.7	1.1	0.6	1.3
Picea abies Karst.	8.4	24.8	30.7	16.1	9.2	4.7	2.8	1.3	0.7	1.3
Abies alba Mill.	15.6	25.8	18.8	12.9	10.2	5.9	4.8	2.2	1.1	2.7
Pinus sylvestris L.	4.6	23.7	37.7	18.3	6.1	2.8	1.8	2.0	1.0	2.0
Larix decidua Mill.	5.1	28.7	39.0	11.9	6.8	0.0	3.4	3.4	0.0	1.7
Coniferous total	8.2	24.7	31.3	16.1	8.6	4.3	2.8	1.6	0.8	1.6
Total	**14.7**	**24.6**	**33.9**	**12.3**	**6.1**	**2.8**	**2.2**	**1.3**	**0.7**	**1.4**

Table 5 *— Average defoliation percentage of tree species (1987-1994) (Novotny and others 1995).*

Tree species	Average defoliation (percent) 1987	1988	1989	1990	1991	1992	1993	1994
Fagus sylvatica L.	23	19	23	17	13	17	17	21
Quercus sp. L.	24	30	35	31	25	27	27	30
Carpinus betulus L.	18	14	20	18	13	18	25	20
Fraxinus excelsior L.	29	23	29	38	40	38	30	40
Acer sp. L.	39	35	46	39	33	30	29	32
Robinia pseudoacacia L.	32	37	38	74	46	61	51	57
Populus sp. L.	26	40	37	38	45	50	32	36
Broadleaf total	24	23	27	25	19	23	23	26
Picea abies Karst.	34	28	31	29	25	27	29	32
Abies alba Mill.	52	31	39	37	31	33	32	33
Pinus sylvestris L.	40	45	44	44	33	42	29	32
Larix decidua Mill.	24	20	33	33	17	26	27	30
Coniferous total	35	32	35	33	27	29	29	32
Total	**30**	**27**	**30**	**28**	**23**	**26**	**26**	**28**

Defoliation and discoloration are not the only and the most significant indicators of forest health. The tendency of defoliation is negative because the trees lose fewer and fewer leaves by August 15 (their "health state" is getting better), but the other important indicators show the opposite tendencies. For instance, the percentage of the annual salvage cut in the total annual cut is increasing (i.e., the stability of the forest is decreasing).

Among other causal factors, partially influencing the health state are pollution, wind and snow damage, damage by beetles, sucking insects, and game. The most important factor is climate change observed in Slovakia.

To evaluate forest health status, defoliation and discoloration of foliage must be always combined with other parameters, especially the salvage cut percentage, which is the most complete indicator of health of forest trees and of ecological stability of forests.

Table 6 *— Percentage of all tree species in respective damage degrees from 1987 to 1994 in the Slovak Republic (Novotny and others 1995).*

Year and Tree Species		Percentage of tree number in damage degrees							
		0	1	2	3	4	1-4	2-4	3-4
1987	Coniferous	11	36	41	11	1	89	53	12
	Broadleaf	26	47	22	5	0	74	27	5
	Total	**19**	**42**	**32**	**7**	**0**	**81**	**39**	**7**
1988	Coniferous	14	33	43	9	1	86	53	10
	Broadleaf	33	39	23	5	0	67	28	5
	Total	**25**	**36**	**32**	**6**	**1**	**75**	**39**	**7**
1989	Coniferous	9	32	49	9	1	91	59	10
	Broadleaf	20	38	37	4	1	80	42	5
	Total	**15**	**36**	**42**	**6**	**1**	**85**	**49**	**7**
1990	Coniferous	14	30	47	8	1	86	56	9
	Broadleaf	23	45	25	5	2	77	32	7
	Total	**20**	**29**	**34**	**6**	**1**	**80**	**41**	**7**
1991	Coniferous	14	47	34	4	1	86	39	5
	Broadleaf	41	38	17	3	1	59	21	4
	Total	**30**	**42**	**24**	**3**	**1**	**70**	**28**	**4**
1992	Coniferous	15	44	33	7	1	85	41	8
	Broadleaf	31	40	23	5	1	69	29	6
	Total	**24**	**42**	**27**	**6**	**1**	**76**	**34**	**7**
1993	Coniferous	8	42	46	3	1	92	50	4
	Broadleaf	28	43	25	3	1	72	28	4
	Total	**20**	**43**	**33**	**3**	**1**	**80**	**37**	**4**
1994	Coniferous	8	41	44	5	2	92	51	7
	Broadleaf	20	45	31	4	1	80	36	5
	Total	**15**	**43**	**36**	**5**	**1**	**85**	**42**	**6**

Table 7 *— Percentage of the annual salvage cut in the total annual cut in 1985 (Novotny and others 1995).*

Year	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994
Percent of salvage cut	29.7	31.1	31.9	35.9	41.5	49.2	42.8	43.2	51.7	59.6

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Forest Health Status in Hungary[1]

András Szepesi[2]

Abstract

Because Hungary has about 18 percent forest area, it is not as densely forested as most of the countries in Europe. Forests are dominated by native species such as oaks, beech, hornbeam, and other broadleaves. As a result of an intensive afforestation in the last 50 years, introduced species, such as black locust, Scots pine, and improved poplars were widely planted and forest area increased by 50 percent — to 18,000 km^2. Several observations in Hungary from the early 1980's detected an unusual amount and type of forest damage affecting native and introduced species. A shocking observation was the oak decline sweeping through the country with a dieback rate of 10 to 70 percent within a decade. New methods had to be developed to detect and explain changes of forest health status and forest ecosystems. In close cooperation with the International Cooperative Program on the Assessment and Monitoring of Air Pollution Effects on Forests (ICP Forests), a national multi-level monitoring and research program was launched in 1986. The first level of the program — large scale monitoring — accepted defoliation and discoloration as key parameters, but a detailed survey of each sample tree was added. On the basis of the annual survey of about 22,000 trees, a significant increase of defoliation was observed. Although 79 percent of the trees were healthy in 1988, only 44 percent were healthy in 1995. Proportion of damaged trees (defoliation greater than 25 percent) seems to have stabilized at about 20 percent in the last 4 years, close to the average of the European survey results ranging from 6 to 60 percent. Oaks, black locust and Scots pine were particularly heavily defoliated, while beech and hornbeam were only lightly affected. The disturbed balance of forest ecosystems is closely related to the lack of water — a result of river regulation since the 18th century and especially the 12-year period of dry and hot weather in the Carpathian Basin. Although SO_2 emission has been reduced by 50 percent since 1980 and NO_x by 30 percent since 1987, and direct damages have not been observed, air pollution can be a predisposing factor of forest damages. Additional intensive research is necessary to clarify the role of climatic factors and air pollution effects on forest ecosystems, including direct effects, carbon and nutrient cycling at different sites, and damaging biotic factors.

Introduction

Because Hungary has about 18 percent forest area, it is not as densely forested as most of the countries in Europe. Forests are dominated by native species, such as oaks (35 percent), beech *(Fagus sylvatica* L) (7 percent), and other broadleaves (13 percent). As a result of an intensive afforestation in the last 50 years, introduced species, such as black locust *(Robinia pseudo-acacia* L), Scots pine *(Pinus silvestris* L), and improved poplars were widely planted and forest area increased by 50 percent to 18,000 km^2. Although the total area of the country is only 93,000 km^2, about 150 tree species and subspecies occupy the landscape as a result of diverse site conditions with a variety of different forest types, ranging from the nearly treeless steppe to the mountainous beech forests.

Different Sources of Information about Forest Damage

Forest health has been a major concern of forest scientists and practitioners and has been continually observed. In the last 40 years different systems have been developed to detect or forecast forest damage.

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Foret Ecosystems, February 5-9, 1996, Riverside, California.

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Local Damage Report

The traditional and most widely used method was based on the damage reports of forest owners that were collected and evaluated by the Forest Research Institute (FRI). This method is still in use and can provide fairly good results for forestry practice, but it cannot be considered as representative. Thus, its role is limited.

Light Traps

Another program, a network of 25 light traps, has been run for 32 years. This monitoring provides reliable information on insect migrations and gradations. The results are collected and evaluated by the FRI and published annually with special focus on forecasts and recommendations.

Monitoring of Oak Decline

Because of the oak decline in the early 1980's, a specific monitoring program was established in 1987 to follow the development of the disease. Although the decline recently reached its peak, the network is still in use. Similar specific monitoring started at the end of the 1980's in beech forests.

Data are collected by the University of Forestry, the Forest Research Institute, and the crews of the Forest Management Planning Service (FMPS). Sample plots are located in selected stands. The data collection is completed by using visual assessments that apply specific codes describing the different stages of the decline. Reports are published annually.

Large Scale Surveys

Several observations in Hungary from the early 1980's detected unusual amount and type of forest damage affecting native and introduced species. In close cooperation with the International Cooperative Program on the Assessment and Monitoring of Air Pollution Effects on Forests (ICP Forests) — a European program initiated and run by the Economic Commission for Europe (ECE) of the United Nations under the Convention on Long-range Transboundary Air Pollution — a national multi-level monitoring and research program was launched in 1985. With special regard to the site and climatic conditions in Hungary, the priority of air pollution effects of the international program was accepted, but special emphasis was put on all kinds of different damaging factors.

The forest management plans cover the entire forest area, with a 10-year renewal period. The methodology was extended by the FMPS for the collection of more detailed data on health status at the stand level, which was a specific part of the national program in 1985. Results have been evaluated on the national and regional level, and were the basis of the national health condition reports until 1988, when the first survey of the systematic monitoring plots was done. The main goal was to inform forest owners on the health status of the forests and to adjust management practices according to these findings.

The national forest protection program followed the outline initiated by the ICP Forests Manual (1986) and distinguished three different intensity levels:

- **Level I:** Large scale survey of forest condition to follow the spatial and temporal development of damages.
- **Level II: I**ntensive monitoring of forest condition to recognize the key factors and processes.
- **Level III:** Special forest ecosystem analysis aimed at gaining a deeper insight into cause-effect relationships.

In the first level of the national program — large scale monitoring run by the FMPS — recommendations of the ECE Manual were accepted, especially defoliation and discoloration as key parameters. A detailed list of all kinds of visible abiotic or biotic symptoms was also added for a better and more detailed description of the

sample trees. With the establishment of 1,027 sample plots (24 trees/plot) in a 4 by 4 km grid (1987) and the first survey in 1988, the first forest health status data from a representative sampling became available for decision makers, which was an effective tool to identify the problems and concentrate the efforts to the critical areas.

Air Pollutants

The ICP Forests cooperation is based on a hypothesis that new types of forest damage are closely related to air pollution effects. In the recent ICP Forests report (U.N. Economic Commission for Europe, European Commission 1995) more than half of the participating countries mentioned air pollution in their national reports as possible predisposing, accompanying, or triggering factor of forest damage. A clear advantage of the transitional period in Hungary was the reduction of air pollutants as a result of the recession of heavy industry and the restructuring of the whole economy. In addition, the environment protection authorities became more effective during the last decade and implemented the European air pollution protocols. Therefore, an improvement of the air quality was a result of two substantially different factors.

However, numerous disadvantages of the transitional period, include the lack of an updated and comprehensive traditional statistical system. Consequently, the emission charts show 1992 as the last reliable publicly available data. (According to different sources, the levels have been stabilized or continued to sink since then.)

Sulfur Dioxide

Sulfur dioxide emission is still rather high in Hungary, mainly because of brown coal combustion. The 1.6 million tons/year emitted in 1980 was reduced to 830,000 tons/year in 1992 (*fig. 1*). This reduction corresponded with the Helsinki (Finland) protocol (1985) that targeted 30 percent sulfur dioxide reduction between 1980-1993 in Europe. (All the 20 signatories fulfilled the commitments.)

Nitrogen Oxides

The nitrogen oxides protocol signed in Sofia, Bulgaria in 1988 required that by 1994 the emissions of all signatories did not exceed the 1987 levels. The majority of the 26 signatories fulfilled the commitments, including Hungary with 31 percent reduction. Total emission of nitrogen oxides was reduced from 270,000 tons/year in 1980 to 180,000 tons/year in 1992 (*fig. 2*).

Concentration of Gaseous Compounds

Emission is only one factor; it is more important to follow the changes of concentrations and depositions of pollutants in the forests. Concentration of gaseous pollutants — sulfur dioxide (SO_2) and nitrogen oxides (NO_x) — were considerably lower in 1994 than at the beginning of the observations in 1988 (*fig. 3*). Charts are based on data from the three measuring stations continuously operating since 1988 in the western, northern, and central forested region of Hungary. Reduction of concentrations was unexpectedly fast, not clearly explained, but related to the Europe-wide efforts to reduce the emission of these air pollutants (for instance, neighboring Austria also reduced SO_2 emission to one-fifth within the last 10 years.)

Despite these reductions, there are some areas where clear evidence of local air pollution impact has been observed on forests. The general level of these frequently cited air pollutants seems to be rather low compared to central and western Europe. There is a real danger, however, that the reconstruction of the industry may cause an emission increase.

Figure 1 — Sulfur dioxide (SO_2) emission in Hungary, 1980-1992.

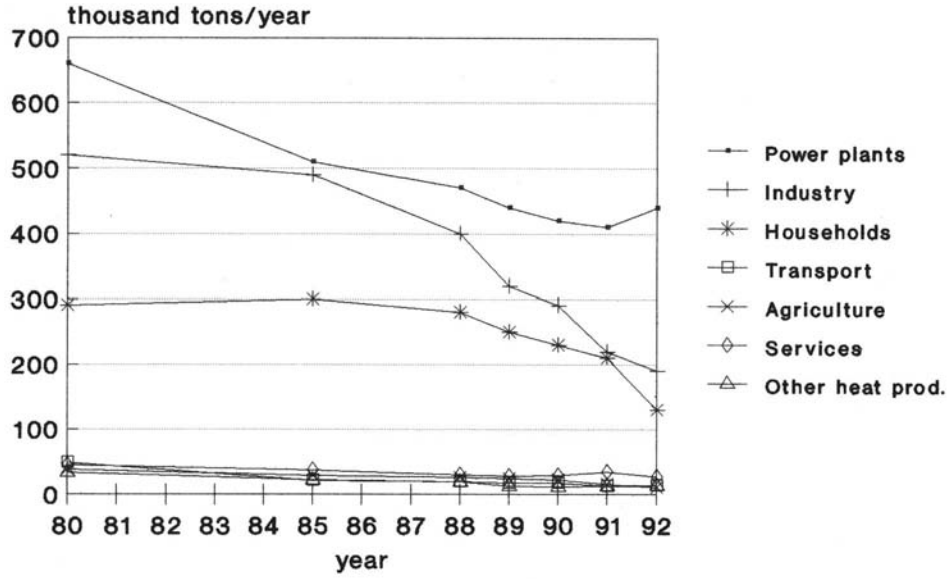


Figure 2 — Nitrogen oxide (NO_x) emission in Hungary, 1980-1992.

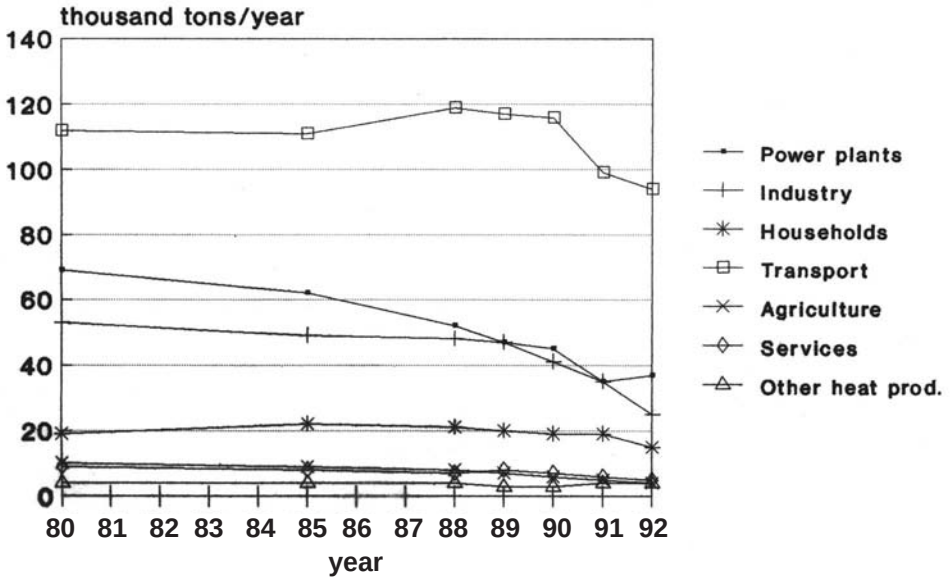
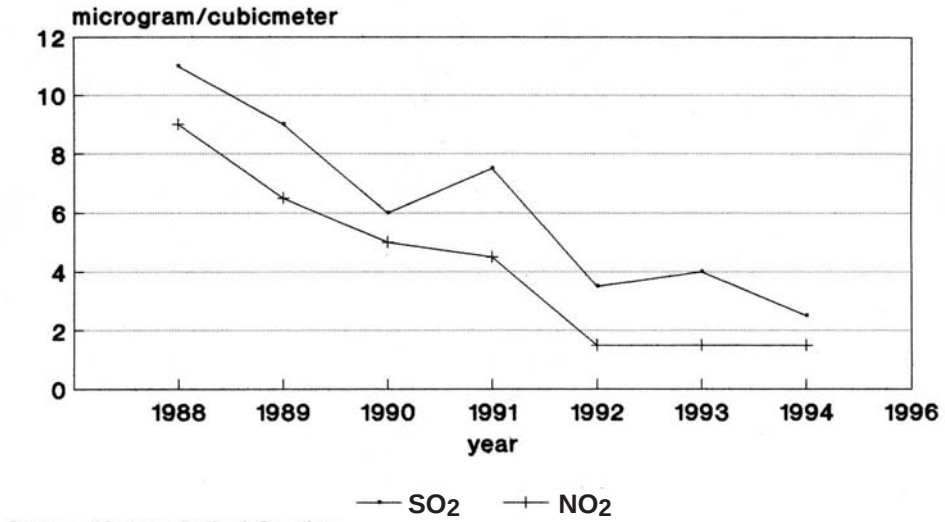


Figure 3 — Sulfur dioxide and nitrogen dioxide concentrations in Hungary's forest area, 1988-1994.



Source: Meteorological Service

Soil Acidification

Acidification of the soil is a real threat and a detected ongoing process in large forest areas of Europe (Germany, Czech and Slovak Republics, Poland). The reduction rate of hydrogen ion deposition is much less than that of the gaseous compounds in the forest area (*fig. 4*). It has also been measured since 1988; however, because the majority of the soils in Hungary are calcareous or well-buffered, the risk of acidification is moderate in general. To detect any kind of acidification may require more time, especially on soils with good buffer capacity. Measurements on research sites have shown that acidic precipitation is neutralized on the leaves in broadleaved forests, which diminishes the effects of acid deposition.

Clear evidence of acidification on agricultural lands was detected in the last two decades, but the main reason was the intensive fertilization and the selection of fertilizers. Role of air pollution was not measured or estimated.

In forested areas resampling of old soil pits and changes of the grass vegetation may indicate the acidification process, but there are not enough reliable data available. A special sampling method was applied with nine subsamples by using a 5- or 10- year resampling period to detect the changes of main soil characteristics quantitatively.

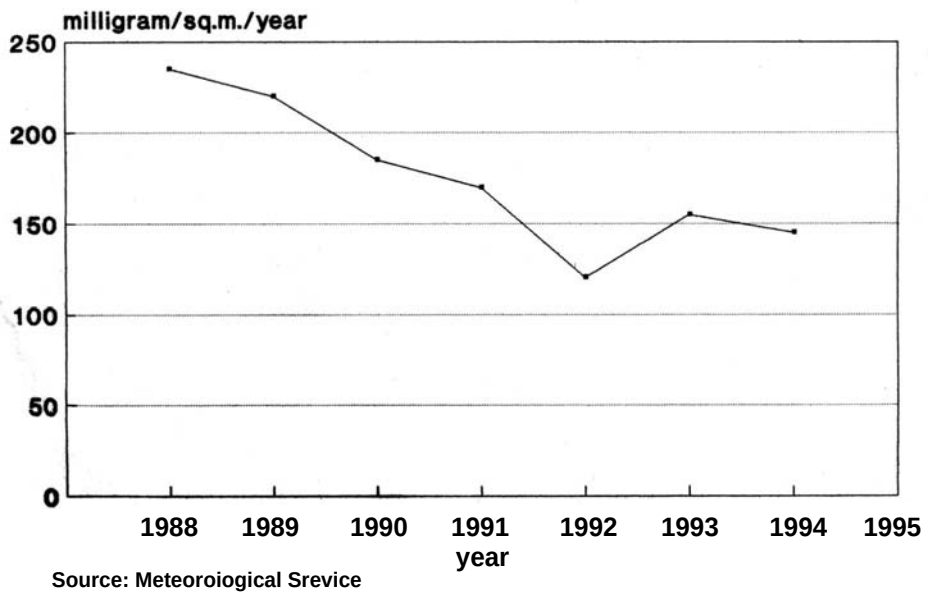


Figure 4 — Hydrogen ion deposition in Hungary's forest area, 1988-1994.

Climate

Climate change and its possible effects on forests are debated. Hungary lies in the Carpathian Basin and is a meeting point of three main climatic regions: the Atlantic from west, the Mediterranean from the south, and the continental from the east. Even a little shift of the main climatic regions may cause considerably different weather conditions in the country dominated by lowlands.

According to the recent assumptions on climate change, a drier and milder climate is predicted for the Carpathian Basin. Because one-third of the country is already characterized by a continental climate, productive closed forests are not expected without a surplus of ground water. Deterioration of large forest areas and decrease of productivity may occur as a result of drier climate. Foresters did not pay much attention to the climate change scenarios in Hungary because of the uncertainties, but the dry and mild climate of the last 12 years put the issues in the focus of the experts. Abatement strategies, such as the reduction of carbon dioxide were not effective enough, with about 25 percent reduction of the annual emission within the last 15 years. This left the emission on a high level of 65 million tons/

year (*fig. 5*). The increase of the growing stock, including afforestation and better use of the different assortments, has also been calculated as capturing about 10 percent of the carbon dioxide emitted recently.

Precipitation has an outstanding importance in the life of forests in Hungary. The stress of high temperature and lack of water is characterized by a drought index that depends on the precipitation in October to August and on the monthly average temperature in April to August (additional factors, like hot days, rainless periods, and groundwater level are included as well) (*fig. 6*). The index is well-known in agriculture and with some modifications, it can also be informative in forestry. Main categories of the index include:

- Moderate drought 5 - 6
- Medium drought 6 - 7
- Strong drought 7 - 8
- Very strong drought 8 +

Although the level of air pollutants decreased in the last 15 years, the weather conditions became more unfavourable for the forests in general. This statement does not refer necessarily to climate change effect, since the meteorological data are within the natural variation of the long-term observation results, rather, this phenomenon is an indication of the possible causes regarding the health status observed from the large scale damage survey since 1988.

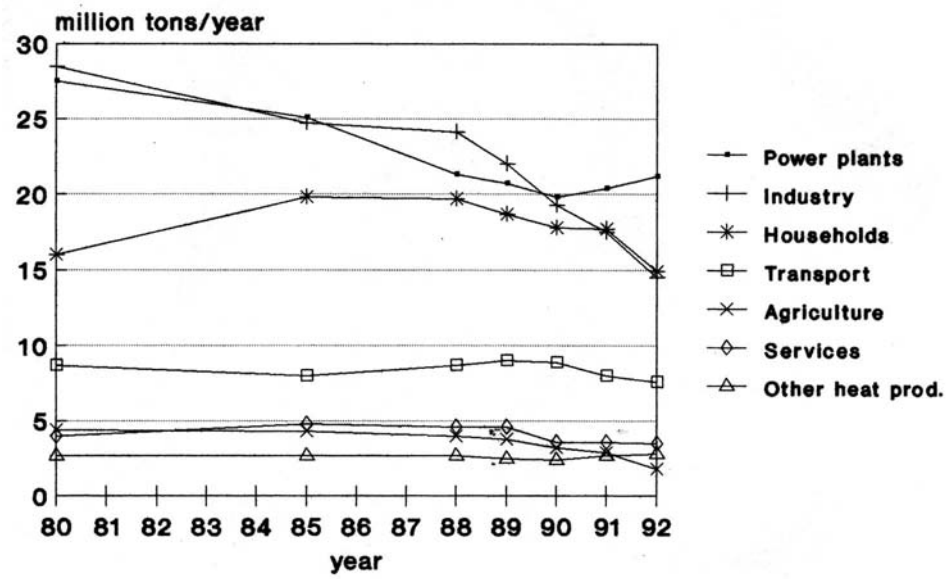


Figure 5 — Carbon dioxide (CO_2) emission in Hungary, 1980-1992.

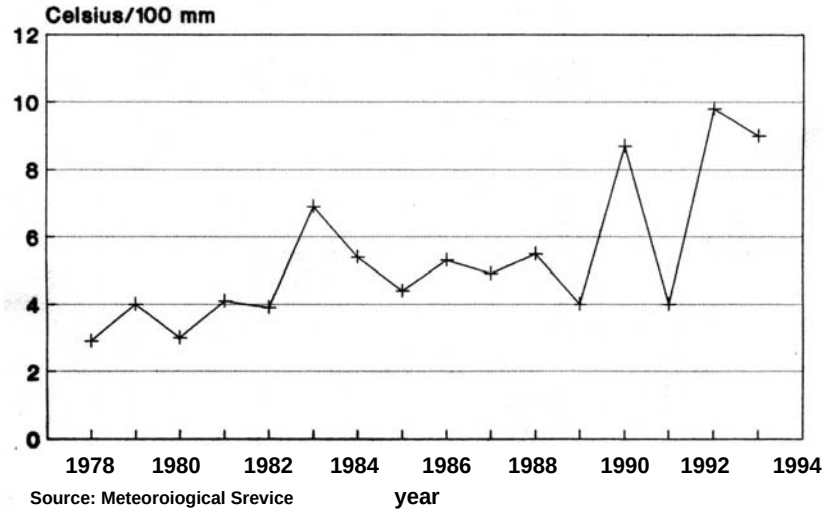


Figure 6 — Drought index in Hungary, 1978-1993.

Results of the Large Scale Forest Damage Survey

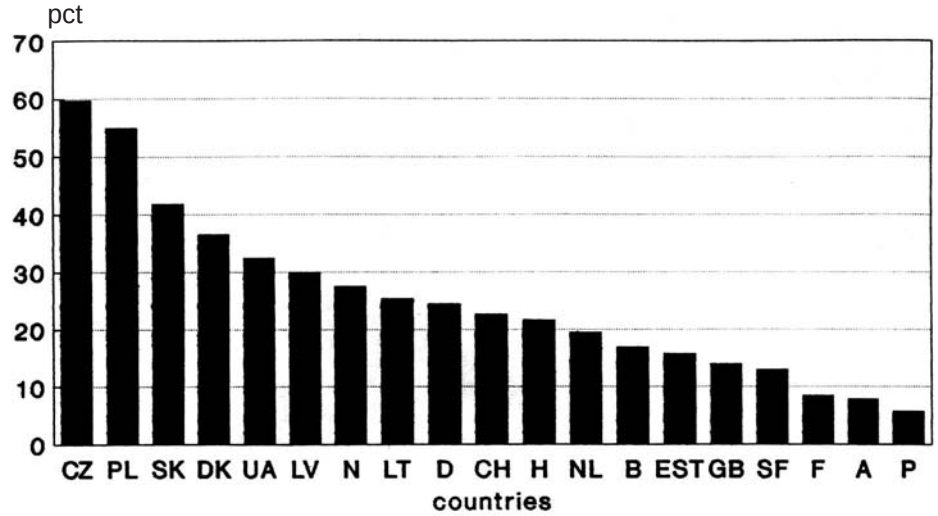
Comprehensive Observations

Defoliation, discoloration and about 44 different symptoms (classified according to the main parts of the tree where they can appear or cause damage) have been assessed by individual sample trees annually since 1988. In 1995 about 1,104 sample plots and 23,000 sample trees were assessed. In 1991-1993 a soil survey was performed based on the national methodology, soil pits, and sampling by genetic horizons. In 1994 additional soil samples were collected with a special sampling design that detected changes of the most important soil characteristics on 67 plots within the 10-year resampling period.

Increment measurements started in 1993, based on 2.83 by 2.83 km net and 200 by 200 m tracts, including all plots of the 4 by 4 km damage survey network. All data collected are important for a detailed and reliable inventory of forest health status and for a scientific interpretation of the observations.

Defoliation in Europe

Defoliation is considered a general indicator of the health status. Despite the common European Manual and the continuous efforts on better harmonization, international comparability is still biased by the differences among the national methods. However, we can compare the forest condition in some European countries to highlight the Hungarian situation. Because about 22 percent of the trees are considerably defoliated (defoliation above 25 percent), Hungary is close to the average defoliation observed in Europe in 1994 (*fig. 7*). The highest damage was observed in the Czech Republic, where 60 percent of the trees were considerably defoliated, while the rate of damaged trees was only 6 percent in Portugal.



Source: Forest Condition in Europe

Figure 7 — Defoliation (greater than 25 percent) in Europe in 1994. Key:
CZ=Czech Republic;
PL=Poland;
SK=Slovak Republic;
DK=Denmark;
UA=Ukraine;
LV=Latvia;
N=Norway;
LT=Lithuania;
D=Germany;
CH=Switzerland;
H=Hungary;
NL=Netherlands;
B=Belgium;
EST=Estonia;
GB=United Kingdom;
SF=Finland;
F=France;
A=Austria;
P=Portugal.

National Results

Defoliation

All types of damages are assessed in 10 percent classes, but interpretation is often combined into 5 classes to follow a classification that is accepted internationally. The following classes are used in the charts:

- Not damaged 0 - 10 percent
- Slightly damaged 11 - 25 percent
- Moderately damaged 26 - 60 percent
- Severely damaged, dead 61 -100 percent

Two different categories of defoilations were used in Hungary. The first is 'total defoliation' including all kinds of leaf losses regardless of the reasons of defoliation; and the second is simply 'defoliation', which is the international terminology that excludes clearly identified reasons of defoliation, such as leaf eaters, fungi, or breaks.

According to the total defoliation category (*fig. 8*), 33 percent of the trees were classified healthy, while 27 percent showed considerable defoliation (above 25 percent leaf loss). Noble oaks (European oak and sessile oak) were in the worst condition with about 50 percent of the trees defoliated considerably. Although clear evidence has not yet been found, oak stands may have been affected by an increment loss because of the intensive reduction of the leaf area. Primary damage factors include foliage insects (mainly caterpillars) and some fungi that attack the native species like oaks or hornbeam (*Carbines beetles* L), while conifers or black locust have been attacked only scarcely and regionally.

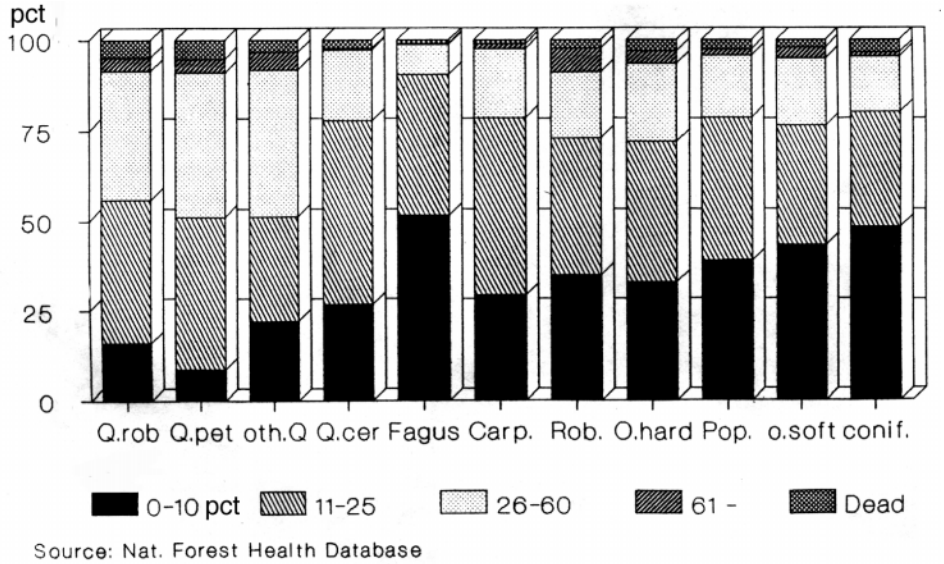


Figure 8 — Total defoliation (23,000 sample trees) in Hungary, 1995. Key:
Q. rob = *Quercus robur*;
Q. pet = *Quercus petraea*;
oth. Q = other quercus species;
Q.cer = *Quercus cerris*;
Fagus = *Fagus sylvatica*;
Carp. = *Carpinus betulus*;
Rob. = *Robinia pseudoacacia*;
o. hard = other hard broadleaves;
Pop. = poplars;
o. soft = other soft broadleaves;
conif. = conifers

The second category, defoliation (*fig. 9*), had a better relationship to air pollution or drought effects. Oaks and especially hornbeam were more healthy than on the chart of total defoliation, but noble oaks were still in the worst condition. With at least moderate defoliation (above 25 percent) European oak (*Quarks robber* L) (32 percent), sessile oak (*Quarks petard* L) (28 percent), and black locust (26 percent) had the highest defoliation, while beech and hornbeam were the least affected (7 and 9 percent, respectively).

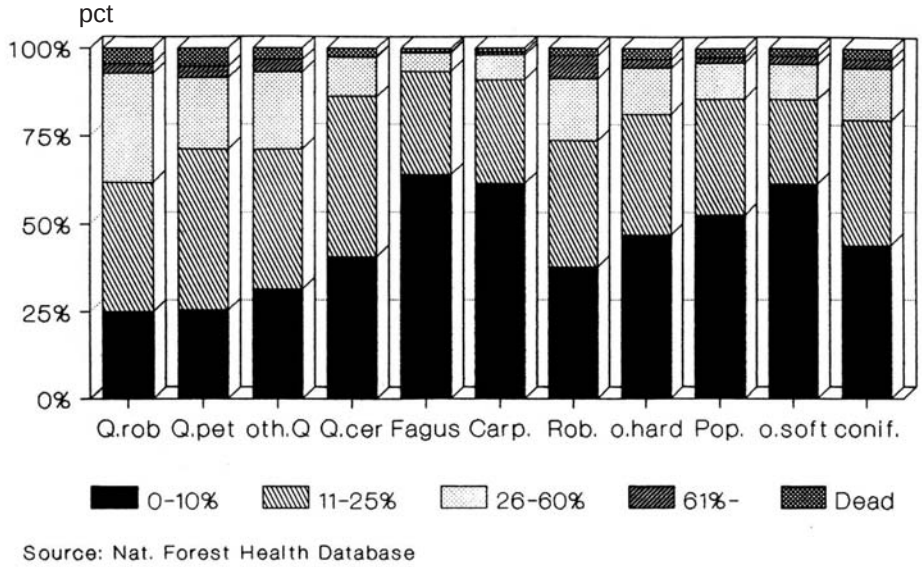


Figure 9 — Defoliation (23,000 sample trees) in 1995.

Defoliation change was alarming in the first 3 years of the survey, showing a rapid increase. But since 1991 the situation seems to have stabilized (*fig. 10*). The proportion of healthy trees decreased from 80 percent to 44 percent and trees with considerable defoliation (class 2-4) increased from 7 percent to 20 percent within the survey period of 1988-1995.

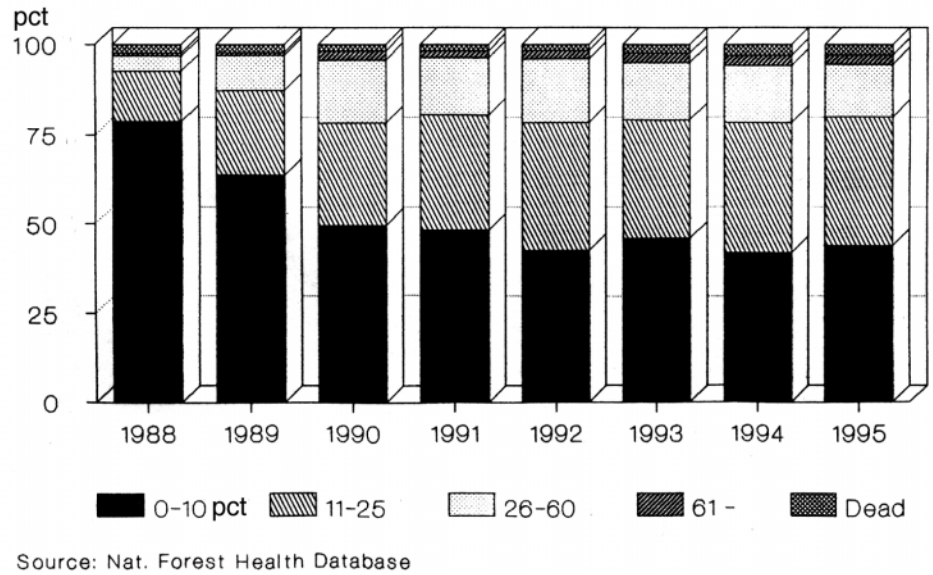


Figure 10 — Defoliation in 1988-95, total sample (17,100 to 23,300 trees).

Mortality is a complex problem in managed forests where the intensity of sanitary fellings and thinning may considerably change the results collected on the permanent plots. In 1995 about 2.9 percent of the sample trees were dead with great differences among the tree species (spruce —12 percent, beech — 1 percent). The mortality rate doubled during the survey period because of the health condition, and the privatization of 40 percent of the forests, in which proper management was not always performed by a responsible owner.

While the results of the total sample are slightly distorted by the replacements and the new sample plots, a comparison was also made for the common sample trees (common trees in 1989-1995) (*fig. 11*). To reduce the possible errors of the first survey, when the staff had only moderate experience, the 1988 data was excluded. Defoliation of the common sample trees was faster than the total sample and the defoliation in 1995 is higher — only 38 percent of the sample trees are classified as healthy, and 25 percent defoliated considerably. Defoliation of the most important

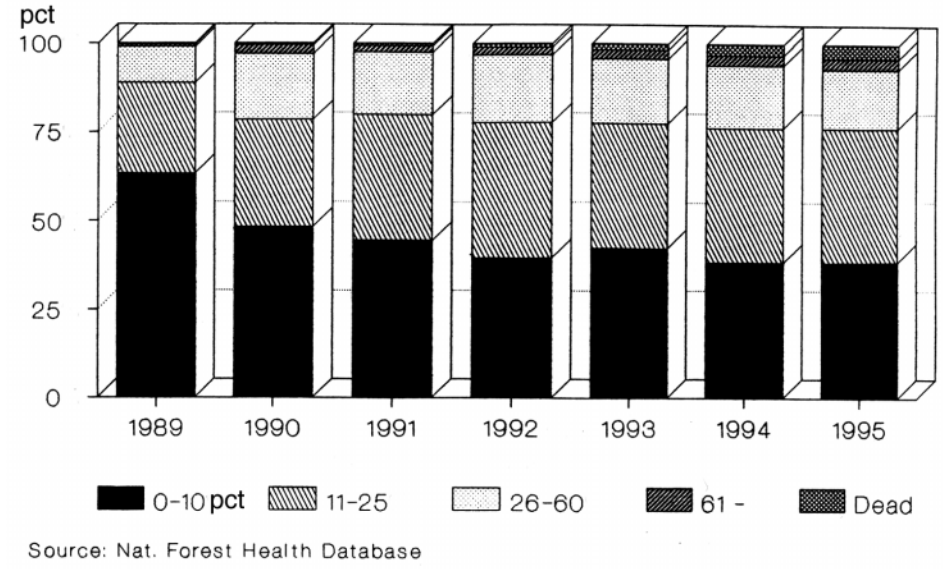


Figure 11 — Defoliation in 1989-95, common sample trees (13,100 trees).

tree species was predetermined by site conditions and annually modified by weather conditions as well as other damaging factors.

The oak decline, affecting sessile oak, peaked in 1988, and on a lower level still exists. Oak stands on shallow soils were severely affected by the long dry period of the last 12 years. In the northern mountainous region, third and fourth generation coppice stands were the most sensitive. High defoliation is not only related to the oak decline; the water reserves in the soil also became exhausted and resistance of the stands to various damages decreased (*fig. 12*).

Dieback of European oak started at the beginning of the survey (*fig. 12*) and was closely related to the traditional water management practice, which lead the water out of the Carpathian Basin, an area frequently threatened by floods, that diminished surface waters in springtime on agricultural areas. Roots of middle-aged and older stands were not able to follow the sinking water, while water deficiency and possible uptake disturbances reduced the resistance of the oak. Thus, the trees also became more sensitive to insect (*Tortrix* sp, *Lymantria dispar* L., *Melolontha melolontha* L.) and fungus *(Microsphaera quercina* Schw. Burr., *Armillaria* sp.) attacks.

Turkey oak (*Quercus cerris* L.) is less sensitive to drought (Mediterranean species, but native in Hungary), often growing on shallow soils or on the drier southern or western slopes. Even the increase of slight defoliation (*fig. 12*) to 40 percent (similar to noble oaks) reduced the canopy closure so that dense grass vegetation became an important obstacle of natural regeneration. Unexplained diebacks were reported in some localities of the northern mountainous region.

Natural and artificial regeneration was more difficult in this dry period and frequency of good seed crops decreased (oaks, beech). Moreover, the regeneration areas are to be protected by fences to reduce game damages. In several areas the grass vegetation became a dangerous competitor of the seedlings because in the spring only the upper 10-20 cm of the soil is saturated properly with water.

Beech was the healthiest species (*fig. 13*). Unlike other tree species in Hungary, beech seldom grew out of its ecological niche. In the humid and cool climate, where the 7 percent beech forests are located, effects of drought were less severe. However, local foresters and pathologists have focused on these species for the past 10 years and have recorded various types of damage related to climatic or biotic factors. Higher defoliation in 1994 was connected with high seed production throughout the country, a phenomenon that reduced the amount of leaves, and increased defoliation.

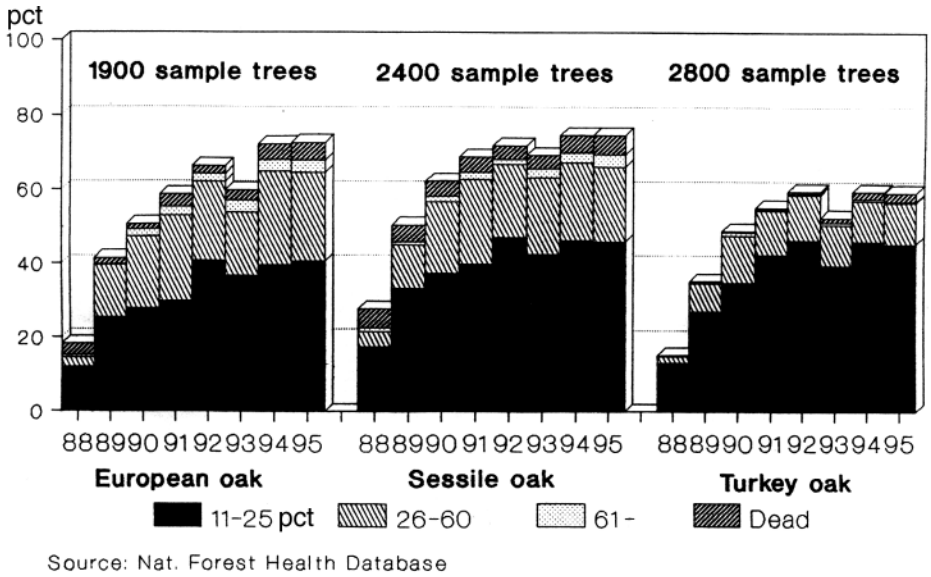


Figure 12 — Defoliation of European oak, sessile oak, and turkey oak in 1988-1995.

Black locust was introduced in the beginning of the 19th century and intensively planted in the country by small owners for fuelwood and later by the cooperatives. Because only a few fungi and insects attacked black locust in Hungary, the species appeared healthy until the mid 1980's, but large scale observations showed rapid deterioration afterwards (*fig. 13*), especially on the sandy and shallow soils. Although the oaks could not respond to the higher precipitation of 1994-1995, the black locust responded, and defoliation in these years decreased. Because black locust was mainly planted close to or on the edges of the villages, resulting in numerous scattered small forest lots, and dead trees have been continuously exploited by local people, the number of dead trees has remained at the same level since 1988 in spite of the increasing mortality.

The group of poplars is heterogeneous, dominated by the improved poplars in plantations (15-30 m^3 annual increment) regularly suffering from specific fungi (*Chryptodiaporthe populea* Sacc. Butin., *Marssonina brunnea* E. et E., Magn.), or the white and black poplar, native in the country. Despite the dominance of the sensitive improved poplars, the health condition of the poplar group has only slightly deteriorated in the last 4 years (*fig. 13*). It is characterized by a high variation of spatial distribution depending on soils and different clones. The stabilized health condition is also attributed to the focus on the selection of sites suitable for new plantations, as well as new more resistant clones.

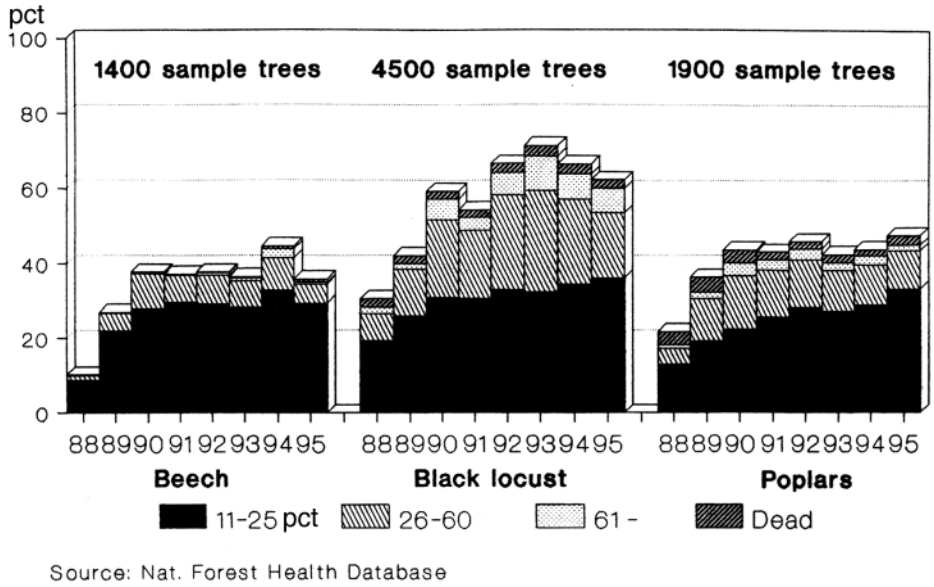


Figure 13 — Defoliation of beech, black locust, and poplars in 1988-1995.

Although 15 percent of the forest area is covered by conifers, they are not native in Hungary. Even spruce that was planted in the humid sites does not receive sufficiently high precipitation and permanently high air humidity. Consequently, rapid and alarming deterioration was observed within the survey period (*fig. 14*) attributed to low precipitation. Although pines dropped only the eldest needle curls in the dry years, spruces lost more and more needles annually. In 1995 about 12 percent of the sample trees were dead, partially because trunks were attacked by root decaying tinder (*Heterobasidion annosum* Fr. Bref.) in several regions where the replanting of conifers required special technologies or they would not be economical any longer. Intensive attack of bark beetles in the weakened stands with dead trees may result in the collapse of the forest within 2-3 years.

The problem is the same in Scots pine stands (about 8 percent of the forests) where defoliation has been rather stable in the last 5 years (*fig. 14*), but the level was constantly high. Trees were more defoliated in the western border region and in higher elevations — within more favourable climatic conditions — than that in the lowlands, where the existence of closed forest is uncertain (there are some different hypotheses, but no clear explanation about the phenomenon). In these dry sandy

regions, Scots pine has no deciduous alternative if attacked by root decaying tinder or killed by the lack of water so that replanting of conifers (species already introduced to the country) becomes impossible. The original vegetation — steppe and scattered small groups of trees and bushes — can only be restored within a long period, depending on the weather conditions.

Austrian black pine (*Pinus nigra* Arn.) is very tolerant to drought, so it has been planted in the poor sites where soil protection has primary importance. In addition to the lack of water, new pathogens, like *Diplodia pinea*, native in the Balkan peninsula, attacked the stands and increased the high defoliation attributed to drought — especially in 1992. Since then a definitely higher mortality was observed (*fig. 14*). Despite the unfavorable conditions, Austrian black pine has a relatively low defoliation, but a prolonged dry period or additional fungus attacks may rapidly increase diebacks and deforestation is also possible in some regions.

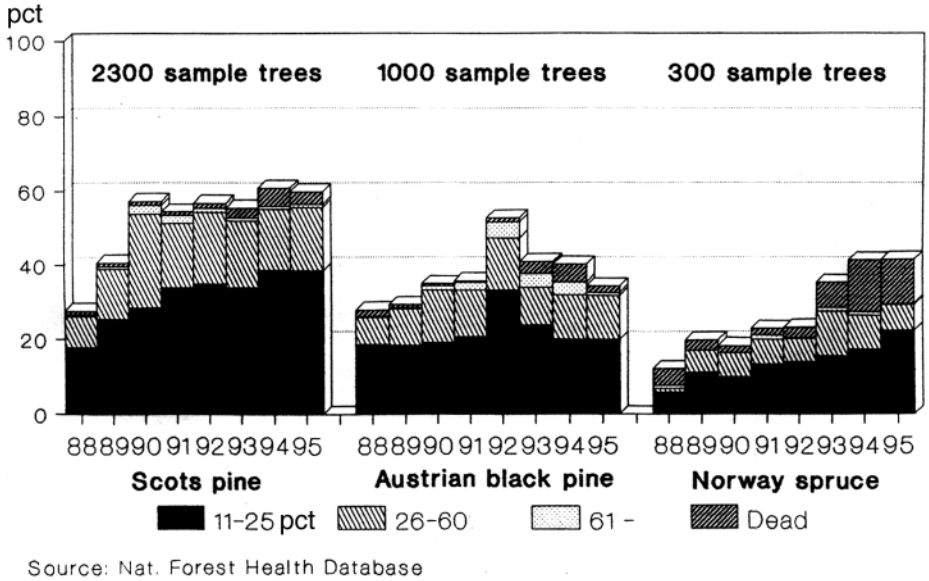


Figure 14 — Defoliation of Scots pine, Austrian black pine, and Norway spruce in 1988-1995.

Discoloration

Although discoloration caused by air pollution or disturbed nutrient balance, such as lack of calcium or magnesium, has been described in some European countries, it has not been observed in Hungary. In the dry years or after unusually long hot and dry summer periods early fall of leaves was reported, starting with yellowing in the second half of the survey period (in the first half of August). Discoloration of black locust, hornbeam, and Scots pine have always been high, with outstanding results in 1990 when about 20 percent of the respective tree species were considerably discolored (more than 25 percent discoloration); but only 3 percent was found in 1995 (*figs. 15,16*). While 39 percent of the sample trees were at least slightly discoloured in 1990, less than 12 percent were discolored in 1995. Presence of fungi or insects is another common reason of discoloration observed on oaks, poplars, and conifers. However, decrease of discoloration was not in correlation with the decrease of fungus and insect damage; on the contrary, the frequency of identified crown damage increased gradually within the survey period.

Other Damage

Methodology of the field survey was extended by a detailed list of 44 various biotic and abiotic damage factors or symptoms identified on the tree. The damage factors were grouped according to the main parts of the tree where they appear or cause damage (*fig. 17*). The intensity of the damage is also recorded in 10 percent classes. Some damage was typical to certain tree species during the whole survey period, like foliage insects on European oak and hornbeam, and frost ribs on Turkey

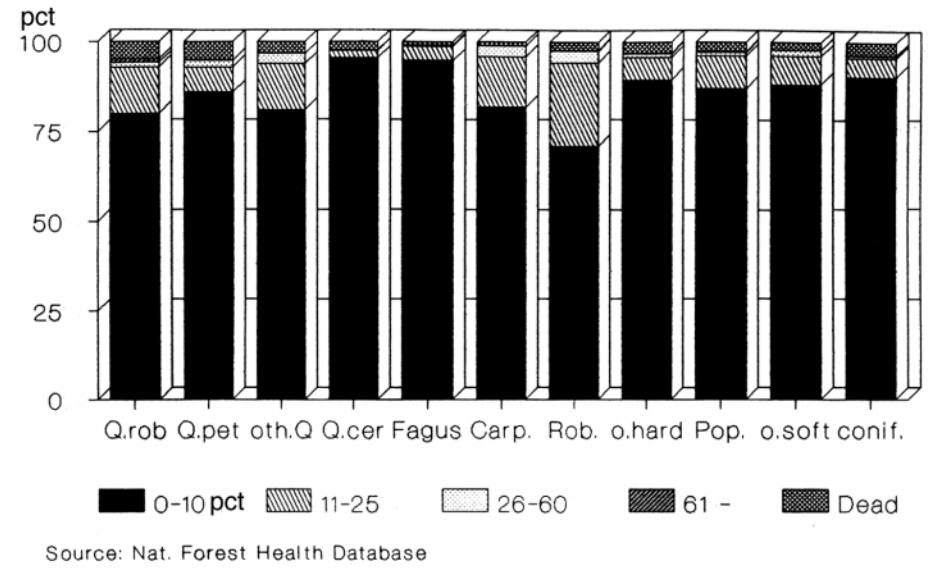


Figure 15 — Discoloration of 23,300 sample trees in 1995.

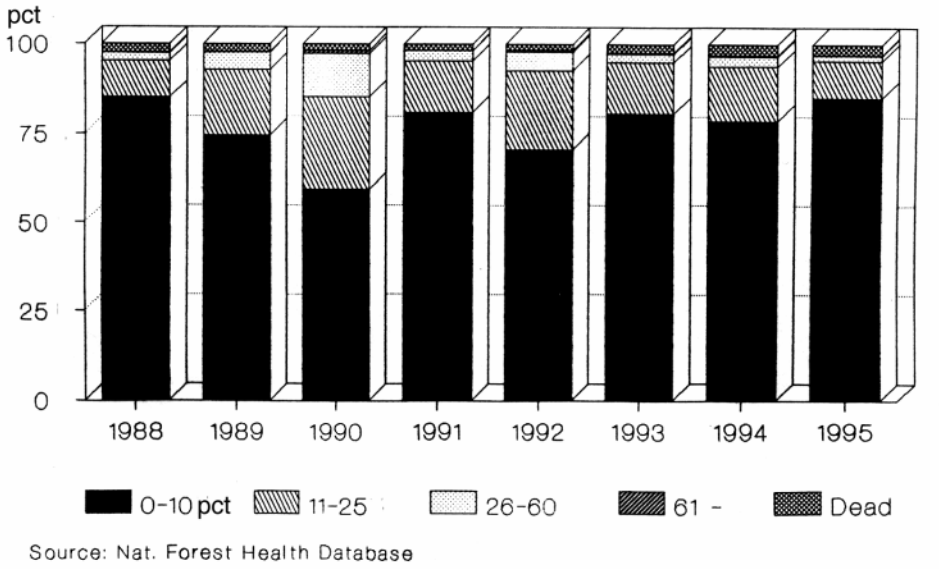
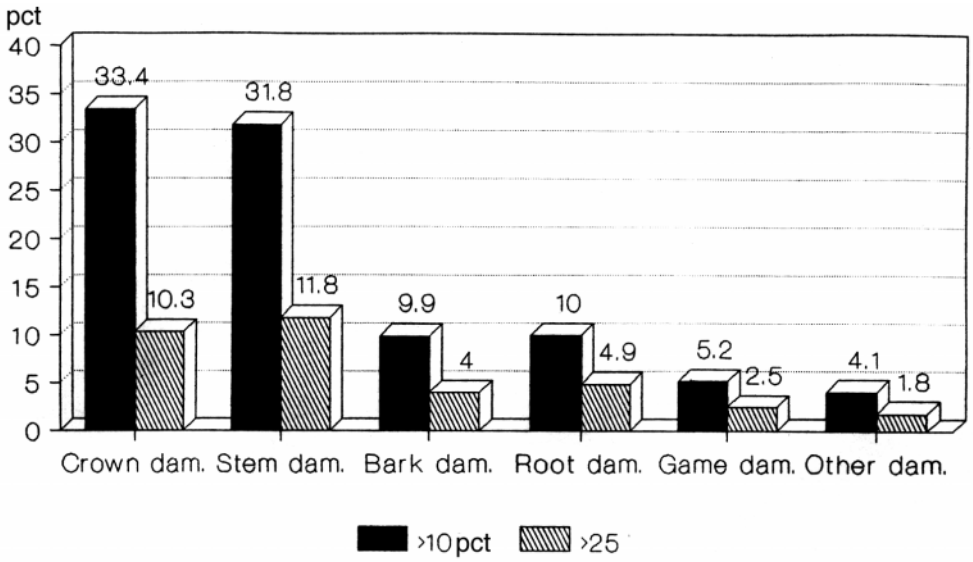


Figure 16 — Discoloration of 17,100 to 23,300 sample trees in 1988-1995.

oak and some improved poplar clones, and bark injuries on beech. Other damage from game and drought affected nearly all tree species and has been more frequently identified recently. It is difficult to clarify the role of damaging factors based on the defoliation and discoloration results that show a general worsening of the condition of forest health. Unfavourable weather conditions and some suspected effects of air pollution might have reduced the resistance of the trees or caused deterioration directly, and these weather conditions may have also resulted in an abundance of various insects and fungi on forest trees.

Future Outlook

Despite the monitoring and research efforts, it is difficult to predict the development of forest damage in Hungary. The large-scale survey highlighted the alarming development and extent of forest damage and encouraged the forest authorities as well as the public to pay more attention to the protection of our forests. The forest protection program is supported by the government, and as a second level of the national implications of the ICP Forests cooperation, 14 intensive monitoring plots were selected and equipped according to the standards of the ECE Manual. The government provided a special fund available for forest owners to diminish forest damages and encourage regeneration.



Source: Nat. Forest Health Database

Figure 17 — Primary damage symptoms of 23,300 sample trees in 1995.

Effective air pollution control may keep emissions on a constant level after the recession, or even further decrease can be expected due to the acceptance of the European Union's air pollution standards. Better cooperation is necessary with water management authorities to keep more water in the Carpathian Basin and ensure the permanent feedback to the ground water without creating new swampy areas.

Additional intensive research is necessary to clarify the role of climatic factors and air pollution effects on forest ecosystems, including direct effects, carbon and nutrient cycling at different sites, and damaging biotic factors.

Acknowledgments

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Forest Health Monitoring Indicators and Their Interpretability: A Lithuanian Case Study[1]

Romualdas Juknys,[2] Algirdas Augustaitis[3]

Abstract

Growth and productivity of forests are important indicators for understanding the general condition and health of forests. It is very important that indicators detected during monitoring procedures afford an opportunity for direct or indirect evaluation of forest productivity and its natural and anthropogenic changes. Analysis of the U.S. Forest Health Monitoring (FHM) crown indicators showed that primary crown indicators are not enough for biological interpretation of collected data and estimation of potential tree growth. The length of crown is necessary for calculating meaningful crown indicators; consequently, measurements of tree height should be included into a FHM sampling design. Statistically reliable tree height measurements as well as basal area estimation are necessary for evaluation of tree volume and biomass. Results of a pilot study on relationships between tree growth and crown indicators, and a multiple regression model for assessment of potential productivity of trees based on primary and composite crown indicators are presented in this paper. New composite crown indicators are developed to increase interpretability of FHM data. A modification of FHM tree height, basal area measurement and estimation procedures is proposed.

Introduction

Increased environmental pollution, especially from anthropogenic sources, emphasizes the importance of monitoring the environmental role of forests in Lithuania. Forest vegetation, especially trees, are a basic component of forest ecosystems and are characterized not only by their impressive dimensions and age, but also by their high sensitivity to environmental pollution. Main coniferous species (Scots pine and Norway spruce), which make up to about 60 percent of Lithuanian forests, are especially sensitive to air pollution. As the health and structure of forest ecosystems worsens, biodiversity and productivity decreases, and aging processes accelerate in this polluted environment.

Because of their longevity, forests have an important effect on various processes in the entire biosphere. Their sensitivity to anthropogenic influence integrates past and current environmental situations and is one of the most suitable objects for environmental monitoring. Long range transboundary pollution and environmental acidification affects are considered a main reason for the wide range of forest decline during the last decade. Lithuania has participated in the European Forest Monitoring program (International Cooperative Program on Monitoring and Assessment of Air Pollution Effects on Forests) since 1988. The main forest condition indicator in this program is the visually assessed crown defoliation. According to data of the Lithuanian National survey (Ozolincius 1996), defoliation of the main tree species has affected more than 80 percent of Lithuanian forests (*table 1*).

Defoliation of different tree species has increased rather substantially during the last several years. A significant increase in defoliation was registered for Norway spruce. Although there has been a significant decrease in air pollution in Lithuania and in Europe during the last 3-4 years (Juknys 1995), the increased defoliation can be explained as a result of the integrated effects of anthropogenic and natural stressors. Unfavorable climatic conditions of past years, especially extremely hot and dry summers in 1992 and 1994, and the invasion of insects have contributed to this negative process. Some stabilization and even decreased defoliation

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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for pine and birch recorded in 1994-1995 (*table 1*) can hopefully be considered a first sign of positive biological reaction to the improved environmental situation.

Forest Health Monitoring activities in Lithuania, as well as in other Baltic countries, were started in 1993 when field training with the participation of specialists from the U.S.A. and all three Baltic countries was begun in Birstonas (Lithuania). By following FHM protocol in the Forest Health Monitoring Field Method Guide (FHM 1994), field work started in 1994.

This paper discusses the beginning of the FHM methodology implementation in Lithuanian environmental monitoring procedures and the biological interpretation of the collected data; we also analyzed the necessity to change the field work methodology to enhance the interpretability of the data.

Table 1 *— Average defoliation of different tree species.*

	Average defoliation (percent)						
Tree species	**1989**	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**
Scots pine	22.8	23.3	25.5	23.6	25.2	24.2	24.0
Norway spruce	15.6	16.5	17.9	17.5	23.1	21.7	28.6
Birch	19.3	21.7	23.3	19.7	23.8	20.7	20.5
Total	**18.8**	**19.7**	**21.3**	**21.0**	**23.4**	**23.0**	**24.2**

Objectives and Methods

Growth and productivity of forests are one of the main indicators reflecting the general condition and health of monitored forests. Therefore, it is very important that indicators used during monitoring procedures allow for direct or indirect evaluation of forest productivity and its natural and anthropogenic changes.

Because trees are main component of forest ecosystems, growth, productivity, and mortality of trees are very important indicators for forest monitoring. Because direct measurements and estimation of tree growth and productivity is very time consuming and usually requires destructive sampling procedures, indirect methods based on easily detected external indicators are preferred.

Different crown indicators are usually used to evaluate potential growth of forest trees. Many investigations were performed during the last decade to evaluate relations between crown indicators (mainly defoliation of trees) and tree growth (Braker and Gaggen 1987, Juknys 1991, Kontic and Winkler-Seifert 1987, Kramer 1986, Schweingruber 1985). Different results were obtained, but the main conclusions were that correlation between crown defoliation and growth of trees (usually width of annual rings) is rather weak and the coefficient of negative correlation usually doesn't exceed 0.3-0.5.

Our investigation of relations between growth of trees and FHM crown indicators was performed during 1994-1995 using the European forest monitoring system. Crown defoliation was assessed at the same time. Primary proposals to improve interpretability of FHM data were presented during the Baltic countries-U.S.A. FHM meeting, which was held in Kaunas (Lithuania) in December 1994.

Investigations were performed in Scots pine forests surrounding the Jonava mineral fertilizer plant where nitrogen and sulfur compounds are the primary emissions.

The main objectives of this research were:

- To elaborate methodology and models for assessment of potential productivity of trees on the basis of primary and composite crown indicators.
- To define the most suitable tree growth indicator as a response function to crown quantity and quality indicators.

- To develop new indicators and proposals to improve field work methodology in order to enhance interpretability of FHM data.

During the investigation, all tree crown and stem indicators were separated a priori into **primary indicators** (measured or evaluated during field work) and **composite indicators** (secondary or tertiary, calculated on the basis of primary indicators). On other hand, for better understanding and interpretation, all (primary and composite) indicators were defined as **quantitative** or **qualitative** indicators.

Primary analysis of FHM crown indicators have already shown that the list of quantitative indicators measured during FHM field work is not sufficient for biological interpretation of collected data and estimation of potential growth of trees. Length of crown is absolutely necessary for calculation of meaningful composite crown indicators. For live crown ratio evaluated according to the FHM field protocol, height should be measured for some subsample of trees to have data on crown length. Statistically reliable tree height data are also necessary for determination of stand productivity (m^3/ha or m^3/ha/year).

Three semimature Scots pine stands were chosen for the investigation and 25 trees were systematically sampled in each of them. The following primary crown and stem indicators were measured for each sampled tree:

- **Quantitative indicators**: Stem diameter (DBH; average of maximal and perpendicular crown diameters); tree height (THG); height of crown bottom (HCB) and crown diameter (CRD; average of maximal and perpendicular crown diameters).
- **Qualitative indicators** (in percent): Crown density (CDN; included into FHM program); foliage transparency (FTR; included into FHM program); crown dieback (CDB; included into FHM program) and crown defoliation (DEF) according to the European forest monitoring system.

Additionally, increment cores were taken from each tree at breast height and two increment indicators were estimated — radial tree increment (RI) and basal area increment (BI) — from 1 to 10 years. Kramer (1986) showed that for assessment of potential tree growth both crown quantity and crown quality should be considered. The following composite tree crown indicators were used for further analysis:

- Crown surface area (CSA) as an integrated crown quantity indicator.
- Total crown losses (TCL) as an integrated crown quality indicator.

Crown surface area and crown volume usually are calculated on the basis of crown diameter and crown length data, and assume that the crown is cone or parabolic shaped. But more detailed investigations on the basis of extensive study of crown for felled trees (Kravchenka 1972) showed that such a strictly geometrical approach can give only a rough approximation to crown surface area or crown volume. Shape of Scots pine crown depends essentially on the age of tree and crown class. Crown shape coefficient (K) (ratio of crown surface area or crown volume to the surface area or volume of cylinder with the same diameter and length) varied from 0.530 for young and suppressed trees to 0.674 for mature dominant trees (Kravchenka 1972).

Crown surface area (CSA) was calculated as follows:

$$CSA = CRD * CRL * \pi * K \qquad (1)$$

in which CRD — crown diameter, CRL — crown length.

For the investigated forest stands (70 years), the following crown shape coefficients (K) were used for different crown classes: 1 (open growth) — 0.660; 2 (dominant) — 0.630; 3 (codominant) — 0.600; 4 (intermediate) — 0.570; 5 (overtopped) — 0.550.

Reduced crown density and increased foliage transparency and dieback were considered crown losses (losses of branches, shoots, and foliage) because of the influence of different internal (competition) and external (environmental pollution, insects, diseases, etc.) factors. **Total crown losses** (TCL) in percents were calculated by using the equation:

$$TCL = 100 - ((100 - DNL) * (100 - DBL) / 100 * (100 - FTL) / 100) \quad (2)$$

in which DNL — crown losses because of decreased tree crown density (losses of branches, shoots, and foliage); DBL — crown losses because of increased dieback (losses of shoots and foliage); FTL — crown losses because of increased foliage transparency (losses of foliage).

For determination of crown losses, these questions were examined: what can be considered normal density, normal dieback, and normal foliage transparency? On the basis of 2 years of FHM data, normal (maximal) crown density for Scots pine was assessed as 70 percent, normal (minimal) crown dieback as zero, and normal (minimal) foliage transparency as 5 percent.

Tree crown density losses (DNL) in percent were calculated:

$$DNL = (NDN - CRD) / NCD * 100 \quad (3)$$

in which: NCD — normal crown density; CRD — crown density of tree.

Tree crown losses because of increased foliage transparency (FTL) in percent were calculated:

$$FTL = ((FTR - NFT) / 100 - NFT) * 100 \quad (4)$$

in which: NFT — normal foliage transparency; FTR — foliage transparency of tree.

Crown losses because of increased crown dieback should be assessed according to the analogous formula (4), given that normal dieback for Scots pine equals zero, DBL = CDB.

The composite crown indicator — relative crown amount (RCA) — is considered a mostly integrated crown indicator, which includes practically all quantitative and qualitative crown indicators:

$$RCA = CSA * NCD / 100 * (100 - TCL) / 100 \quad (5)$$

in which: CSA — crown surface area; NCD — normal crown density; TCL — total crown losses.

Results

In order to define the most suitable tree growth indicator as a response function to crown indicators, we correlated radial tree increment (RI) and basal area increment (BI) for different durations (from 1 to 10 years) with the FHM crown indicators (*table* 2). We found that correlation of crown indicators with radial tree increment is essentially weaker than with basal area increment because basal area increment reflects biomass production much better than ring-width increments. The same ring width increment can result in very different biomass increment for trees with different diameters. In addition, foliage transparency is mostly related to radial tree increment between qualitative Scots pine indicators. Correlation of crown dieback with radial tree increment is weak and usually statistically insignificant ($\alpha = 0.05$) for Scots pine.

Table 2 *— Correlation of radial and basal area increments with primary FHM crown indicators for different periods.*

Crown indicator	Length of period (years) 1	2	3	4	5	6	7	8	9	10
	Correlation coefficient (significance level)									
	Radial increment									
Density	**0.296**	**0.260**	**0.302**	**0.363**	**[1]0.417**	**[1]0.439**	**[1]0.434**	**[1]0.426**	**[1]0.417**	**0.403**
	0.159	0.220	0.151	0.085	0.043	0.032	0.034	0.038	0.042	0.056
Transparency	**-0.368**	**-0.300**	**0.370**	**[1]-0.473**	**[1]-0.578**	**[1]-0.623**	**[1]-0.628**	**[1]-0.624**	**[1]-0.618**	**[1]-0.541**
	0.076	0.154	0.074	0.019	0.003	0.001	0.001	0.001	0.001	0.007
Dieback	**-0.036**	**-0.174**	**-0.128**	**-0.137**	**-0.148**	**-0.173**	**-0.223**	**-0.236**	**-0.247**	**-0.237**
	0.867	0.417	0.548	0.519	0.489	0.416	0.295	0.266	0.244	0.261
	Basal area increment									
Density	**[1]0.451**	**[1]0.446**	**[1]0.489**	**[1]0.541**	**[1]0.555**	**[1]0.554**	**[1]0.532**	**[1]0.517**	**[1]0.507**	**[1]0.491**
	0.027	0.029	0.015	0.006	0.005	0.005	0.007	0.010	0.011	0.012
Transparency	**[1]-0.533**	**[1]-0.495**	**[1]-0.569**	**[1]-0.665**	**[1]-0.710**	**[1]-0.726**	**[1]-0.706**	**[1]-0.616**	**[1]-0.688**	**[1]-0.623**
	0.007	0.013	0.004	0.000	0.000	0.000	0.000	0.000	0.000	0.001
Dieback	**-0.057**	**-0.240**	**-0.202**	**-0.242**	**-0.251**	**-0.269**	**-0.323**	**-0.374**	**-0.349**	**-0.320**
	0.791	0.245	0.315	0.243	0.273	0.210	0.123	0.072	0.095	0.126

[1] Statistically significant (α=0.05).

Coefficients of correlation were averaged between different durations of tree increment indicators and foliage transparency for all investigated tree stands (*fig. 1*). For 5 to 6 years the correlation of tree increment with crown indicators essentially increased, but further increase of duration usually caused stabilization or even decrease of correlation coefficient. Five-year basal area increment can be considered the most suitable tree increment indicator relating to FHM crown indicators.

FHM qualitative crown indicators and crown defoliation were correlated according to the European forest monitoring system (*table 3*). Closest correlation with crown defoliation (according to European system) was detected for foliage transparency. Correlation of crown defoliation with crown dieback is weak and statistically insignificant (α= 0.05).

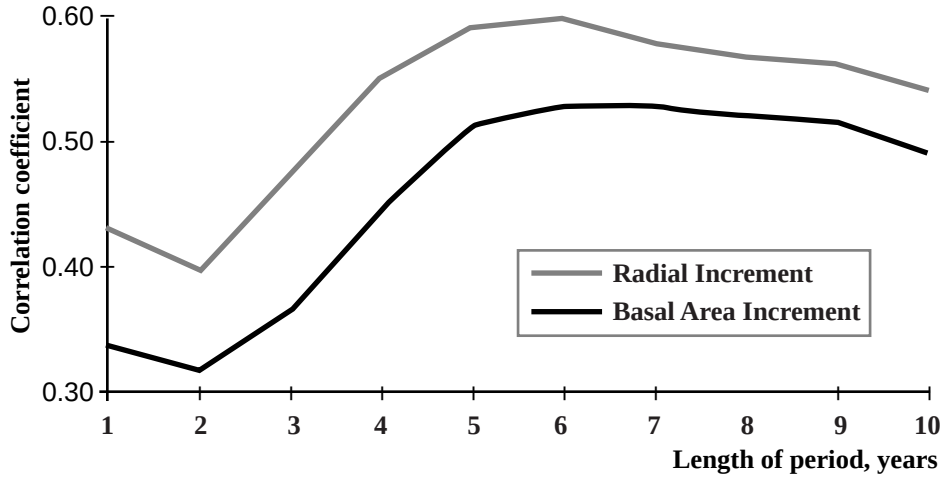

Figure 1 — Correlation of tree increment with foliage transparency for different periods.

Five-year basal area increment and different crown indicators were correlated (*fig. 2*). Crown length is more closely related to tree increment than crown diameter. Foliage transparency and crown density area are most strongly related to tree increment from primary crown qualitative indicators. Foliage transparency is also closely correlated with crown defoliation according to the European forest monitoring system (*table 3*).

Relative crown amount, which integrates quantitative and qualitative crown indicators, is most strongly correlated with tree increment from composite crown indicators. Correlation of total crown losses, as a composite qualitative crown indicator with tree increment is closer to that for crown surface area, which integrates quantitative crown indicators. Correlations of primary qualitative crown indicators (with the exception of crown dieback) with tree increment in general is larger than correlations of quantitative primary crown indicators (*fig. 2*).

Finally, the influence of crown quality on the growth of trees with a different crown quantity (crown surface area) was investigated. Results of multiple regression analysis showed (*fig. 3*) that for suppressed trees with small crowns the influence of crown quality (total crown losses) is much less than for trees with well developed crowns (big crown surface area). Multiple regression coefficient equal about 0.7; consequently, almost one-half of basal area increment variation can be explained on the basis of composite qualitative and quantitative crown indicators.

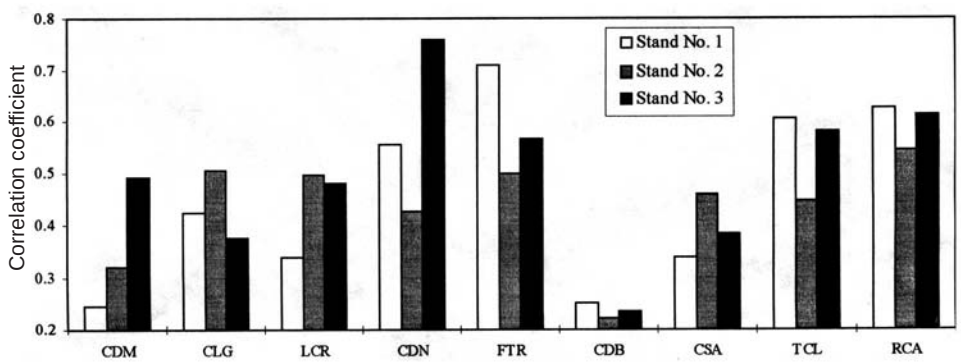

Figure 2 — Correlation between 5-year basal area increment and different crown indicators. Key: CDM - crown diameter, CLG - crown length, LCR - live crown ratio, CDN - crown density, FTR - foliage transparency, CDB - crown dieback, CSA - crown surface area, TCL - total crown losses, RCA - relative crown amount.

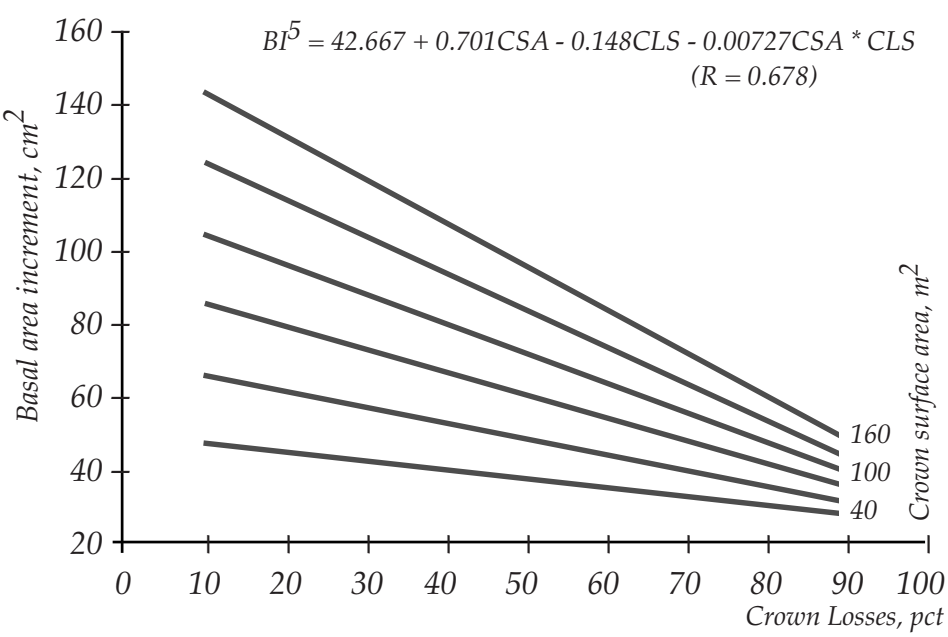

Figure 3 — Influence of qualitative and quantitative composite crown indicators on tree increment.

Table 3 *— Results of correlation of qualitative FHM crown indicators with crown defoliation (significance level).*

Stand	Crown density (pct)	Foliage transparency (pct)	Crown dieback (pct)	Total crown losses (pct)
1	[1]-0.813 (0.000)	[1]0.878 (0.000)	0.215 0.314)	[1]0.801 (0.000)
2	[1]-0.830 (0.000)	[1]0.931 (0.000)	[1]0.404 (0.049)	[1]0.818 (0.000)
3	[1]-0.781 (0.000)	[1]-0.823 (0.000)	0.206 (0.334)	[1]-0.789 (0.000)

[1]statistically significant ($\alpha = 0.05$)

Discussion

For meaningful interpretation of forest monitoring data, the possibility to assess general features of forests, such as productivity, sustainability, etc., on the basis of easily detected external indicators is very important. Analysis of FHM indicators showed that the list of primary indicators measured and evaluated during field work is not sufficient for biological interpretation of collected data and estimation of potential tree growth and its anthropogenical changes. For assessment of tree growth on the basis of crown indicators, both crown quality and crown quantity should be evaluated and measured. The FHM field work program includes only one quantitative tree crown indicator — crown diameter — and one semiquantative — live crown ratio. On the basis of these two indicators it is impossible to assess crown quantity (crown surface area, crown volume). Tree height should be measured for some subsample of trees in order to have data on crown length. Statistically reliable tree height data are also necessary for determination of stand productivity.

Five-year basal area increment can be considered the most suitable tree increment indicator relating to FHM crown indicators. Correlation of radial (linear) increment with crown indicators is weaker because the same linear increment can result in different biomass increases. The composite tree crown indicator of total crown losses integrates all tree FHM crown qualitative indicators (crown density, foliage transparency, and crown dieback) and reflects general crown quality. Total crown losses correlates more closely with tree growth than the composite crown quantity indicator — crown surface area. An elaborated multiregresional model showed that on the basis of qualitative and quantitative crown indicators, a reliable assessment of tree growth is possible.

Besides separate tree productivity (increment), assessment of productivity of entire monitored stands, such as biomass, volume, and basal area per unit of area, is absolutely necessary. According to the FHM field work program, basal area measurement is included in the list of tree site measurements, but really is not statistically based. Basal area of monitored stands calculated on the basis of measured tree diameters on subplots would be more appropriate. Tree height measurements should be adopted to supplement tree diameter measurements on subplots as special subsamples. Simplified methodology for minimally statistically reliable tree height measurements was developed by the Lithuanian FHM project and is presented in the report for 1995 field season.

In Lithuania, as well as in other Baltic countries, three different forest monitoring systems were introduced during the last decade:

- The European International Cooperative Program on Assessment and Monitoring of Air Pollution Effects of Forests (ICP-Forests) began in 1987, and information on the condition of Lithuanian forests is presented every year. This very simple and operative method was supplemented by the forest soil monitoring subsystem from 1993 and more sophisticated second-level forest monitoring methods from 1995.
- From 1993 Lithuania and other Baltic countries (Latvia and Estonia) joined the International Integrated Monitoring Program on Small Catchments. The pilot phase of this program (Integrated Monitoring in Terrestrial Reference Areas of Europe and North America) was started in 1988 within the UN/ECE Convention on Long-range Transboundary Air Pollution. On the basis of financial support from Nordic countries, two to three very well- equipped Integrated Monitoring Stations were established in afforested background areas in each Baltic country during 1993-1995.
- The Forest Health Monitoring Program, with sound ecologically based methodology and emphasis on biodiversity of forest ecosystems, may be a very good extension of the Forest Monitoring System. It may help in filling a gap between the very simple European Forest Monitoring System and the very sophisticated and expensive Integrated Monitoring System.

Better coordination and coadaptation between those three systems are necessary. Baltic countries could be a very good case study for such international and interprogram activities.

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Photosynthetically Active Radiation (PAR) Measurements and Their Use in Forest Health Monitoring

Craig Palmer, Chair

Photosynthetically Active Radiation Measurements in Pure Pine and Mixed Pine Forests in Poland[1]

Jaroslaw Smialkowski[2]

Abstract

Photosynthetically active radiation (PAR) has been measured in pure pine and mixed pine forests on 15 sites in two transects in Poland: the "climatic" (from the western to the eastern border), and the "Silesian" (from the most to the less polluted part of the country). PAR was measured by using the standard procedure developed by the USDA Forest Service's Forest Health Monitoring Program. At each forest site PAR was measured at the "central point" and at six points 7.3 m from the central point. PAR was also measured at three additional points (at the location of litterfall collection). Eight out of 15 forests were classified as pure pine, and 7 as mixed pine. Two series of measurements were completed in September/October 1994 and in July/August 1995. The first measurements showed that the mixed pine forests were more shaded than the pure pine forests. Average leaf area index (LAI) for mixed pine forests was 2.03 and 1.78 for pure pine. Mixed forests were characterized by greater seasonal changeability of LAI because of a higher amount of broadleaved trees and shrubs. LAI and transmitted PAR (TPAR) standard deviation on a particular forest site may be treated as an index of internal forest differentiation. The indexes differ between particular forests, but average values for pure pine and mixed pine forest are almost identical. The first attempt to correlate PAR indexes and other ecological indexes suggest that there is no correlation between PAR and litterfall and the rate of litter decomposition.

Introduction

Photosynthetically active radiation (PAR) was measured in pine forests of Poland under the Polish-American cooperative program entitled "The Effect of Pollution and Climate Change on Forest Ecosystems." This program includes climatic, ecological, geobotanical, and soil research along two transects. The first transect (eight sites) runs east-west along the 52nd parallel and is associated with the increasing continentality of climate. The second transect (also of eight sites) runs from the most polluted areas of Upper Silesia to the relatively unpolluted part of Poland around the Bialowieza Forest. The most north-easterly point (K011) is common to both transects; and out of a total of 15 sites, 8 were categorized as pure pine forest, and the other 7 as mixed pine forest *(fig.1)*. All the stands contain trees between 70 to 100 years old (Breymeyer and others 1995).

This study measured PAR along the climatic and Silesian transects in Poland by using an analysis of the different amounts of light reaching the forest floor and procedures of the USDA Forest Service's Forest Health Monitoring Program (FHM 1994), in order to explain the rate of some ecological processes.

Methods

PAR research on the transect sites began in 1994. The first series of measurements at the study sites were recorded in September and October 1994, and the subsequent series in July and August 1995. Unfavorable weather conditions forced the abandonment of measurements at three sites in October 1995. Thus, the series of measurements at three forest sites were incomplete in October 1995.

Organic matter decomposition was sampled at each site at a central point (Breymeyer and others 1995), and the PAR measurements were made at the central point and at six other points that were 7.3 m from the central point (*fig. 2*). In addition,

[1] An abbreviated version of this paper was presented at the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California.

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PAR was measured at three locations where litterfall was collected. Thus, PAR was measured at a total of 10 points. Analysis of results from the first series of measurements (autumn 1994) showed that data collected in conditions of variable cloud cover (especially with convection cloudiness) were difficult to interpret. Transmitted PAR (TPAR) and leaf area index (LAI) for the forest site were calculated by using incoming light below the canopy and at an ambient station in an open area. The under canopy measurement was a single ceptometer measurement lasting 10 to 15 seconds. The simultaneously collected data from the ambient station (located in an open area) were stored as 1-minute averages (FHM 1994; Decagon Devices, Inc. 1989; LI-COR, Inc. 1990)[3]. A false result is obtained (LAI may be below 0 and TPAR above 1) if the sun shines during the 15-second under canopy measurement, and is then covered by cloud for the rest of minute (Campbell 1991). Similar distortions of the results may be linked to the distance between a measurement point under the canopy and the ambient station (which is sometimes 300 - 500 m). Uneven convection cloudiness may easily allow the sun to shine over a forest stand, while the ambient station remains in shade. Because of the problems associated with variable cloudiness, the measurements in 1995 were only made in sunny, cloud-free weather, or at times with minimal cloud cover.

[3] Mention of trade names or products is for information only and does not imply endorsement by the U.S. Department of Agriculture.

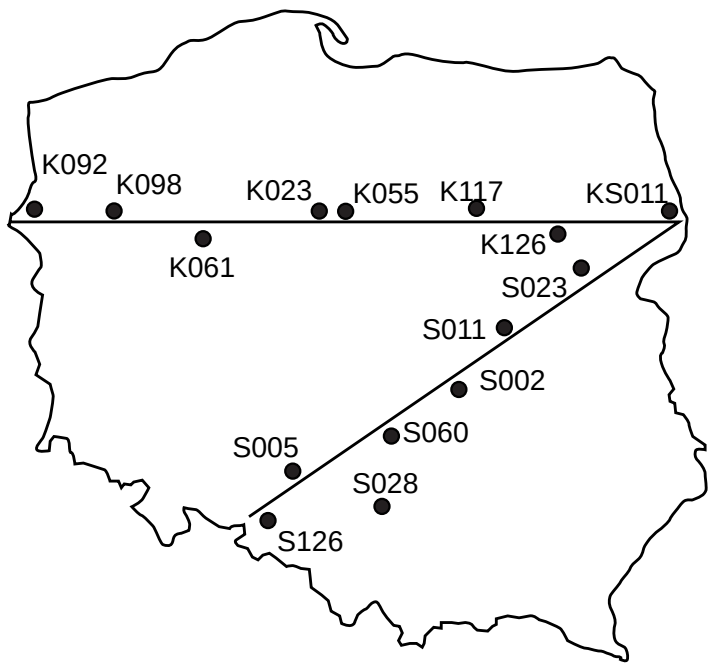

Figure 1 — Forest sites studied on the climatic and Silesian transects in Poland.

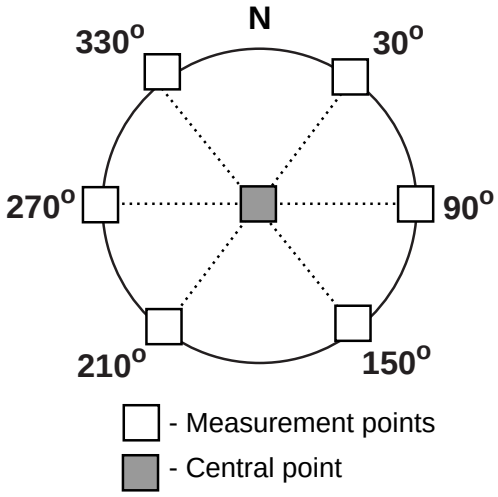

Figure 2 — Schematic diagram of PAR measurements on the forest sites.

Results

The results of the three series of measurements show that mixed pine forests are characterized by lower levels of PAR reaching the forest floor than pure pine forests. The mean for PAR in mixed pine forests was 0.25, while that for the pure pine forests was 0.21 (*fig. 3*). Still greater differences were noted in the case of LAI, which varied between 1.3 and 3 at the study sites. The mean LAI for pure pine forests was 1.78, and that for mixed pine 2.03 (*fig. 4*). These differences are associated with the greater proportion of broadleaved trees and shrubs in the mixed pine forests.

Comparison of summer and autumn measurements yielded these results: all sites for which both summer and autumn data are available showed LAI values to be higher in summer because of the presence of leaves. An exception was site S023, in which the autumn results may be erroneous because of cloudiness. When data for this point are deleted, the mean difference in summer and autumn LAI in mixed pine forest is 1.61 (i.e., a figure greater than the 1.36 recorded from pure pine forests) (*fig. 5*).

Standard deviations of the measured values of TPAR and LAI may be interpreted as a measure of heterogeneity of the forest sites under study. Standard deviation values for LAI were not related to the type of forest, but rather to its internal structure. In the case of LAI these ranged from 0.4 to 1.3. The greatest value was noted for site S005, which had a considerable number of young spruce trees growing in the forest. The mean LAI standard deviation in pure pine forests was 0.75, while that for mixed pine forests was 0.77 (*fig. 6*).

Preliminary attempts were made to correlate the PAR measurements and the other ecological indices (e.g., litterfall and litter decomposition rates). However, despite the fact that LAI is sometimes calculated based on litterfall, statistically significant linkage has not yet been found, except for the inverse correlation between LAI and TPAR.

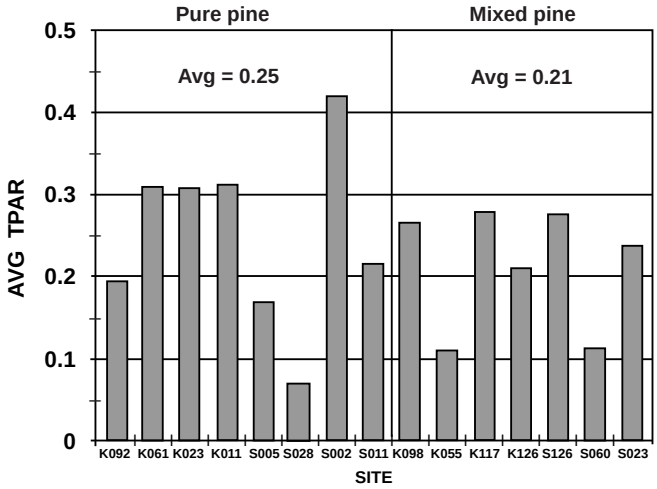

Figure 3 — Average TPAR on the forest sites.

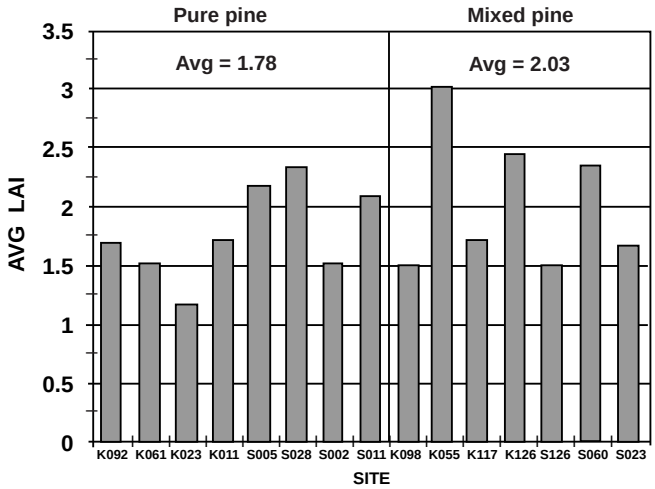

Figure 4 — Average LAI on the forest sites.

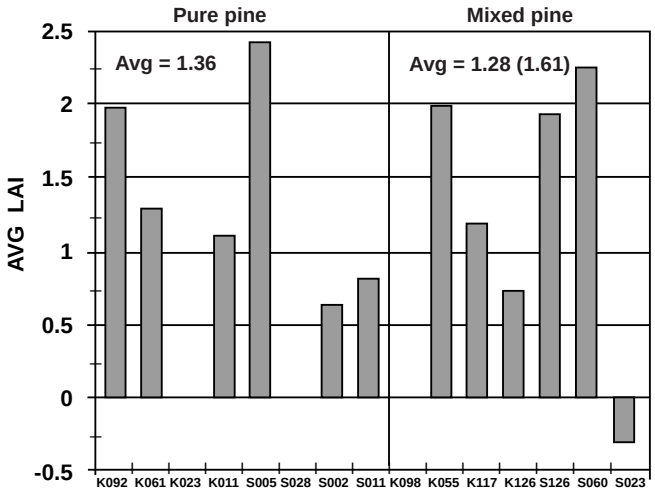

Figure 5 — Differences in LAI between autumn and summer on the forest sites. Average value for mixed pine in parenthesis is the value when data for site SO23 is deleted due to cloudiness at the site (see text).

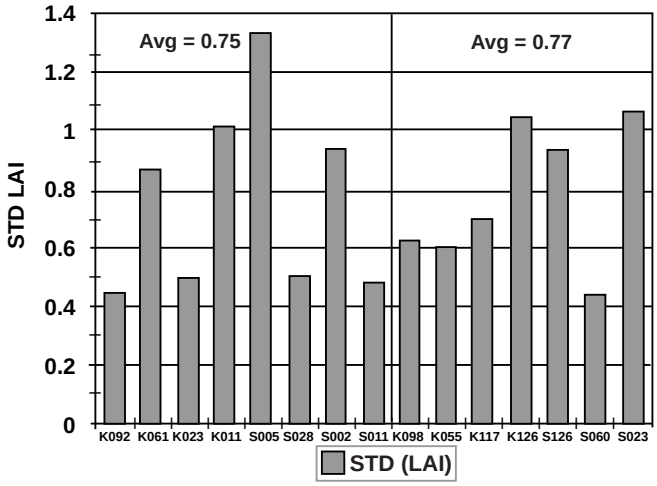

Figure 6 — Internal differentiation of the forest sites measured as standard deviation of LAI.

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Remote Sensing Applied to Resource Management[1]

Henry M. Lachowski[2]

Abstract

Effective management of forest resources requires access to current and consistent geospatial information that can be shared by resource managers and the public. Geospatial information describing our land and natural resources comes from many sources and is most effective when stored in a geospatial database and used in a geographic information system (GIS). The information on the location and condition of current vegetation is one of the key elements in resource management. Remote sensing data, such as aerial photographs, satellite imagery, aerial video, and data collected by other remote devices, are primary sources for mapping vegetation. Comparison of images acquired several days or several years apart can assist in determining changes that occurred in that time period. The Forest Service, U.S. Department of Agriculture, has been using various remote sensing and associated technologies to map vegetation and monitor changes. Current needs include an increased training and awareness in use of this technology, as well as development of new applications, standards, and guidelines.

Introduction

Geospatial information about land and natural resources is essential to effective management of natural resources. Data from various remote sensing instruments make a substantial contribution to the geospatial information needed in the USDA Forest Service (FS). Remote sensing is a source of current and repeatable information on the location, quantity, and quality of land cover and other resource information, particularly vegetation. It allows change detection and monitoring of land cover and vegetation over time. Data from various remote sensing tools form an important source for various geographic information system (GIS) databases and subsequent analysis. Many types of data are already in digital form, and others can be digitized. Satellite imagery covers large areas and has the locational precision and spatial resolution to satisfy many natural resources mapping requirements. For many applications satellite imagery needs to be used in conjunction with a closer view provided by aerial photographs and airborne video images. The global positioning system (GPS) used in conjunction with satellite imagery helps managers determine changes in land cover over time.

This paper examines the ability to integrate remote sensing imagery in a GIS combined with ancillary data, including cartographic feature files (CFFs), digital elevation models (DEMs), digital orthophotoquads (DOQs), and other resource data layers to provide a solid geospatial data foundation for support of Forest Service management of natural resources.

Data Collection Through Remote Sensing

The USDA Forest Service uses data from a variety of remote sensing instruments, such as the advanced very high resolution radiometer (AVHRR), which uses data obtained through the U.S. Geological Survey (USGS) EROS Data Center (EDC), from the U.S. Department of Commerce, and from the National Oceanic and Atmospheric Administration (NOAA) weather satellites. It has relatively course spatial resolution of 1.1 by 1.1 km pixel size with five spectral bands. Broad area

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coverage is obtained with a swath width of 2,700 kilometers. Temporal resolution is excellent with repeat coverage over a given geographic area every 12 hours. Major applications for this data include: vegetation discrimination, vegetation biomass, snow/ice discrimination, vegetation/crop stress, and geothermal mapping. This imagery is currently used by the Forest Service in association with the USGS on development of a global vegetation data set and with the United Nations on verification for a global vegetation characterization data base.

Another remote sensing instrument is the Landsat Multispectral Scanner (MSS). MSS imagery has a spatial resolution of 80 m, spectral resolution in 4 bands, revisits the same geographic location every 16 days, and covers an area of 185 by 170 km per scene. The Landsat program gathered digital MSS data from 1972 through 1992. The result is a 20-year time span of data that can support evaluations of change in landscapes or land cover over a longer time period than any available earth observation system. MSS data is available through the EDC working in cooperation with the National Aeronautics and Space Administration (NASA) Landsat Pathfinder Program. Pathfinder efforts are focused on evaluation of global change by using available remote sensing technologies. The North American Landscape Characterization (NALC) Project, a component of the Landsat Pathfinder Program, is developing an archive of MSS data and production of three-date georeferenced data sets, called triplicates, acquired in 1970's, 1980's and early 1990's. The NALC images are geometrically rectified, georeferenced, and placed into a UTM map projection. Pixels are resampled into a 60 by 60 m size format for compatibility with the 30 by 30 m Landsat Thematic Mapper (TM) data resolution.

TM imagery has 30 m spatial resolution with seven spectral bands. The satellite covers the same geographic location every 16 days, each scene covering 185 by 170 km. The Landsat Program is the longest running program in the collection of multispectral, digital data of the earth's land surface from space. The temporal extent of the collection, the characteristics and quality of Landsat data, and the ability to collect new data directly comparable to that in the archive, make Landsat data a unique resource to address a broad range of issues in ecosystem management. An interagency consortium called the Multi-Resolution Land Characteristics Monitoring System (MRLC) has been established to purchase TM imagery to cover the United States (not including Alaska). This group has selected the appropriate scenes and developed image processing, georeferencing, terrain correction, clustering and classification strategies. This Federal consortium includes U.S. Environmental Protection Agency, EMAP; U.S. Fish and Wildlife Service, GAP Program; USGS EDC and National Water Quality Assessment Program; and NOAA Coast Watch Change Analysis Program. High resolution airborne video and digital cameras provide a powerful tool for ecosystem management. A camera can be mounted in a small airplane and imagery can be viewed as soon as the flight is complete (Bobbe 1995). Video imagery linked with GPS provided location information for each frame, allowing users to quickly locate specific areas on maps or images. Video images can be digitized and used with other GIS layers.

Use and Access to Remote Sensing Data

Applications of the various remote sensing sources vary widely among resource management agencies, depending on their specific requirements. Users of digital imagery range from novices, who simply use imagery as a backdrop for overlaying GIS information, to expert users, who do classification of landscapes for input into large area assessments or forest planning efforts. Thus, a wide range of user needs must be met in the form of technology transfer and training to fullfil the Forest Service's mission. Access to remote sensing data should rapidly increase for several reasons, among them, new cooperative agreements among agencies to share data

acquisition costs, and improved information access on the information super highway. Currently, many potential users of remote sensing are either not aware of how to access the data or believe that the cost of acquiring the data is prohibitively high. Improved access to the data, both physically and financially, will likely increase the number of users.

New data procurements, such as the one recently made by the Forest Service, will provide AVHRR, Landsat Multispectral Scanner triplicates, and Landsat TM imagery, for all lands in the continental U.S. This low cost purchase will provide imagery except for Alaska at very low cost to the entire Forest Service at three different scales. Purchases such as these provide inexpensive and easy access to data and will surely spur interest in using digital imagery for applications such as Resources Planning Act (RPA) assessments and forest planning. Access to the internet now gives users the capability to preview satellite imagery acquired from several platforms including Landsat. This will allow users to assess the condition of the imagery (e.g., cloud cover) and will ensure that the user is confident in the quality of the imagery before purchasing. Because internet access will also provide an easy method for ordering and transferring digital imagery, the process of data acquisition that was recently time-consuming and inefficient at best will be streamlined. Several new satellite sensors are being developed and are scheduled to be launched within the next decade. Once these plans come to fruition, users of remote sensing data will have several new options. These sensors will allow for different perspectives of the landscape because of the differences in spectral, spatial, and temporal resolutions of the imagery they will provide.

Applications of Remote Sensing

With new GIS hardware and software available to the Forest Service, and existing GIS and image processing capabilities, the use of digital imagery for effective resource management will continue to increase. Virtually everyone in the Forest Service will have the capability to view and manipulate imagery in some way. The Forest Service's use of remote sensing has been shaped not only by technological advances, but by the changing needs for information (Lachowski 1995). Increasingly, the synoptic view that is offered by remote sensing has become more important as issues such as endangered species and cumulative effects have shaped management activities on Forest Service lands. The need for information on the type and condition of the vegetation is almost always critical to these issues and remote sensing plays an important role in collecting this information.

Mapping Vegetation and Land Cover Characteristics

The Forest Service has a wealth of experience with using digital imagery for projects ranging in scales from RPA assessments to forest and project plans. Perhaps the most common use of digital imagery in the agency is for mapping vegetation and land cover characteristics. For example, digital imagery from Landsat Thematic Mapper has been used to map vegetation and model old-growth forests in California (Warbington 1993), Washington, and Oregon (Congalton 1993, Tepley 1993). These successful, region-wide projects are examples of how satellite imagery can be useful at the forest level for mapping vegetation and land cover. In addition, numerous individual forests in every region have used digital satellite imagery to map vegetation characteristics (Evans 1992, Gonzales 1992). Applications of this information serve a variety of purposes ranging from multi-region and multi-agency planning (Columbia River Basin Assessment) to forest planning efforts (Warbington 1993), and project level applications such as wild and scenic river planning (Linderman 1995) and range allotment mapping and analysis (Martinez 1994). On a broad scale, the Forest Service has used digital satellite imagery, such as AVHRR

data from NOAA weather satellites, to map forest land for the entire U.S. and Mexico. These maps are used for RPA assessments and are useful as a baseline analysis for forests at the national level. Currently, other studies are planned to assess the feasibility of linking this information with forest inventory and analysis (FIA) inventories.

Change Detection

In addition to mapping vegetation and land cover, Forest Service managers have recognized the usefulness of multi-date digital imagery in performing change detection for mapping insect and disease outbreaks as well as for updating management activities and forest plan monitoring (Maus 1992). Because each National Forest is mandated by law to monitor the effects of their management over time, the ability to perform change detection is potentially the most effective use of digital imagery from remote sensing. Performing change detection may also be the most underutilized application of digital imagery. This is particularly true as the span of digital imagery available for an area becomes ever wider and the costs of older data continue to plummet. Using digital imagery for performing change detection offers many advantages over traditional methods. The traditional approach for mapping change was to compare two maps, each interpreted independently. Often, false changes would result because of differences in classification schemes or mapping techniques. With advances in geocoding of digital imagery, image-to-image comparisons can be made where spectral values between two or more dates of imagery can be compared. These techniques can help eliminate problems associated with traditional map comparison methods. As awareness of these simple techniques for performing change detection increases and more resource managers have access to the necessary computer hardware and software, the demand for such analysis will certainly increase.

Ecological Unit Mapping

Current direction of the Forest Service requires that an ecological approach to resource management be used. To ensure ecosystem management is applied over various scales of landscapes, the National Hierarchy of Ecological Units was developed (Avers 1993). Within the National Hierarchy, various scales of units are being mapped and used for planning efforts at large scale (multi-region), medium scale (multi-forest), and small scale (multi-district). The boundaries of these units will be based on ecological influences, social/political considerations, and multi-agency needs. Currently, remote sensing is playing a major role in the delineation of these boundaries at all levels. Digital imagery from satellites is being heavily used for the smaller scale units (e.g., ecoregion, sections, and subsections). The larger scale units (e.g., land type associations, and landtype phases) are being mapped using combinations of aerial photographs and satellite imagery (Fallon 1994). Once ecological units are mapped, their characteristics can be monitored over time by using remote sensing techniques. Additional information can be added to ecological units for project level analysis that can also be derived from remote sensing.

Disaster Assessment, Forest Pest Management, and Other Areas

Recent fires in large ecosystems such as the Greater Yellowstone Area and the Payette National Forest have been mapped by using digital imagery from satellites. Other landmark disasters, such as the Mount St. Helens eruption and the Mississippi River flooding, have also been mapped from satellite imagery. Although the use of satellite imagery for such events is somewhat rare, it is often the only type of current information available or accessible. Satellite imagery will continue to fulfill a role in these assessments and will likely increase as access to and interest in digital imagery increases.

Current Needs

Training and Awareness on How to Use the Data

New users of remote sensing imagery will need to know opportunities and pitfalls of using the new technology. They will need to know when it is appropriate to use a particular type of imagery and when it may not be. With personnel reductions, fewer employees are likely to have graduated from recent university remote sensing programs; thus, more will need to be trained. Using remote sensing imagery will likely be a skill they acquire to compliment a wide range of other job responsibilities they will have. New and experienced users will need technical instruction in how to best use digital imagery for different scales of projects ranging from simple applications, such as using imagery as a backdrop for updating GIS layers to more complex applications like automated image analysis for vegetation mapping and change detection. Users of digital imagery will need to share ideas and techniques for using new types of digital imagery. This transfer of information among users will need to take place between all levels of the Forest Service, including Ranger Districts, National Forests, regions, research stations, and State and Private Forestry.

New Sensor Applications Testing and Guidelines

As new types of imagery become available they will need to be assessed for cost effectiveness as well as application suitability. Some new sensors will have hyperspectral bands with very narrow wavelengths. Others will have higher spatial resolution than current satellite sensors. Still others will acquire imagery at different times of the day such as early morning. However, increased access to remote sensing information may also increase potential for abuse of the technology. As applications are developed for new types of imagery, guidelines for using the imagery will need to be developed to inform prospective users of pitfalls and potentials. Pilot projects will need to be carried out to evaluate imagery for meeting various user requirements.

Data Standards and Methods

The Forest Service will need to provide standards and guidelines in the collection, design, use, and storage of remote sensing related data. These types of standards will allow for information sharing between districts, forests, and regions. Information sharing across administrative boundaries will be critical to the success of ecosystem management practices. Standardized naming conventions, file structures, and methodologies will promote information sharing and development of corporate database structures in the FS. These processes will simplify the use of geospatial data, including data acquired through remote sensing and make the application of these data sources to resource management easier and more efficient.

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