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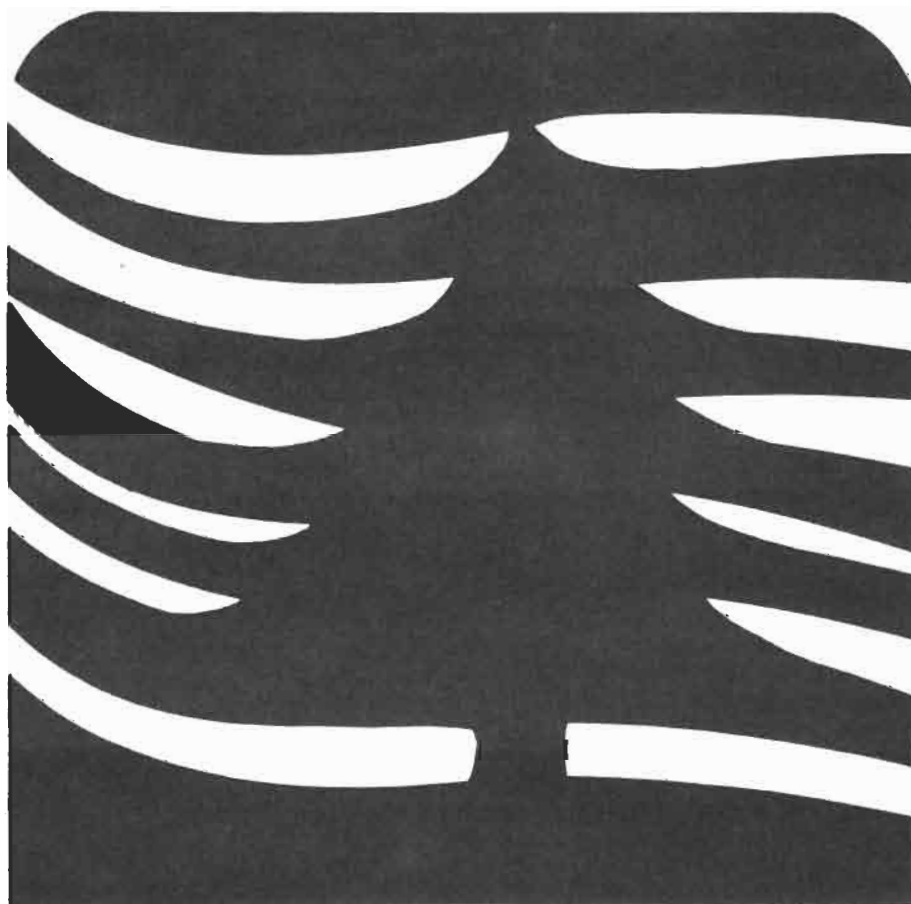
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Effects of Air Pollutants on Mediterranean and Temperate Forest Ecosystems

June 22-27, 1980, Riverside, California



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Paul R. Miller
Technical Coordinator

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PREFACE

Sulfur dioxide from fuel combustion and ore smelting operations has caused significant damage to forest communities throughout the industrialized world. In the temperate regions, notably in Europe, the United States, Canada, and Japan, examples of damage and losses resulting from this pollutant are well-documented. Hydrogen fluoride emissions from aluminum reduction plants, brick kilns, and phosphate fertilizer plants also have caused significant damage in many localities. In Mediterranean climates, the combination of abundant sunshine and poorly controlled emissions of nitrogen oxides and hydrocarbons has resulted in extensive forested regions being exposed to photochemical oxidant air pollution. Ozone is the most damaging pollutant in this mixture. Acidic precipitation, derived principally from sulfur oxide emissions, recently has been shown to have severe effects on aquatic ecosystems in northeastern United States, Canada, and northern Europe. The projected increase in the use of coal for energy generation and the continuing growth of urban centers, accompanied by automobile emissions, are two conditions that suggest a continuing and more pervasive influence of air pollution on terrestrial and related aquatic ecosystems.

A large body of knowledge has been assembled that describes pollutant effects on individual species as a result of both field observations and controlled experiments. Efforts are being made to use the tools of systems analysis (modeling) to interpret and predict pollutant effects on processes at both the individual species and plant community levels. The ultimate goal is to improve interpretation of pollutant effects on ecological systems so that optimal protective and management measures can be taken to assure a more healthy environment.

Experimenters and modelers can advance more rapidly if a better exchange of ideas and essential data can be stimulated. A symposium was planned to encourage closer communication between experimentalists carrying out specialized studies of the effects of major air pollutants on individual forest species and

researchers using computer simulation models to interpret and predict long-term pollutant effects at the plant community and ecosystem levels. This Symposium, held in Riverside, California, June 22-27, 1980, was designed to report and discuss the state of knowledge of single species-single pollutant relationships, the interactions of producers, consumers, and decomposers under pollutant stress, and the use of ecological systems models for interpretation and prediction of pollutant effects. In addition, the present state of knowledge was examined in relation to an overarching ecological concept: resilience of ecosystems. Another important question was the search for indicators of systems-level effects of air pollution on ecosystems. For example, is an effect on nutrient cycling a reliable indicator of system-level change induced by pollution?

Twenty-eight papers were presented in the formal sessions and 29 poster summaries were displayed concurrently. Registered participants numbered 128. Most participants attended a field trip to the San Bernardino mountains for one-half day. Fifteen nations were represented including Austria, Canada, Czechoslovakia, Denmark, Egypt, West Germany, Japan, Mexico, Norway, Poland, Saudi Arabia, Sweden, Switzerland, United States of America, and Yugoslavia.

To facilitate the publication of the Symposium Proceedings, we decided to have each author assume full responsibility for submitting manuscripts in photo-ready format by the time the conference convened. The views expressed in each paper are those of the author and not necessarily those of the sponsoring organizations. Trade names are used solely for necessary information and do not imply endorsement by the sponsoring organizations.

Paul R. Miller
Forest Service, U.S. Department of
Agriculture
Technical Coordinator

Introduction

Welcoming Remarks¹

James N. Pitts, Jr.²

Good morning. On behalf of Dr. David Saxon, President of the University of California, Dr. Tomás Rivera, Chancellor of UCR, and we of the Statewide Air Pollution Research Center, I would like to welcome you to this international symposium. We trust you will have a scientifically interesting and challenging experience during this week.

The subject of this meeting is timely and important. The accurate assessment of biological, economic, and aesthetic impacts of air pollutants on forest ecosystems is essential if we are to develop cost-effective control strategies of air pollution. Overcontrol can lead to economic penalties in the form of extra costs for expensive technologies for pollutant removal. On the other hand, undercontrol can lead to economically unacceptable plant damage which impacts not only our agricultural and forest industry but also our recreational activities. We trust this symposium will elucidate areas of future research that will provide a more extensive data base upon which to generate reliable models that can be used for such cost-effective control strategies.

We in California are particularly sensitive to the threat, indeed the actuality, of serious air pollution damage to crops, forests, etc. Agriculture remains our number one industry, with tourism and associated recreational activities also making a major contribution to the economic well being of our state.

The fact that the symposium is being held here at UCR seems appropriate since it was a group of plant scientists headed by John Middleton who, in the late 1940s, first showed that plant damage seen in Los Angeles County was in fact due to a new type of air pollution.

Through their efforts and the pioneering research of the late Arie Haagen-Smit and others, it became clear that we were dealing with an oxidizing atmospheric system formed by the action of sunlight on hydrocarbons and oxides of nitrogen.

During the last twenty years, much of it under the leadership of Clif Taylor, research has been conducted here in two major areas: Studies of pollutant effects on plants, vegetation, and forest ecosystems, and the chemistry of air pollution. This has been a particularly useful combination of interests because we have one group of scientists working on one axis of the classic dose-response curve, that is, the atmospheric chemists whose function is to describe the dose received by man, animals, or plants, and another group working on the response axis, the plant scientists investigating the interactions of air pollutants with vegetation.

Most of you are familiar with the work which has been done here in the plant sciences area. Let me just mention that one of the major roles of the atmospheric chemists at the Center has been the unequivocal spectroscopic identification and measurement of several new gaseous oxygenated and nitrogenous species that are formed in photochemical air pollution. These include formaldehyde, formic acid, nitric acid, nitrous acid, and the nitrate radical, NO_3 . Whether or not such gaseous species will prove to be significant phytotoxicants is a question that we leave to you "response" specialists.

In closing I want to thank Drs. Paul Miller and Clif Taylor and Mrs. Neva Friesen for their outstanding efforts in organizing this symposium. Many other people deserve a great deal of credit as well, but there simply is not time to acknowledge them individually. Let me just say that we are pleased to host a meeting of this importance and we look forward to learning of the significant results that will emerge from your gathering. I am certain these results will be of lasting importance to this critical area of the air pollution problem.

¹Presented at the Symposium on Effects of Air Pollutants on Mediterranean and Temperate Forest Ecosystems, June 22-27, 1980, Riverside, California, U.S.A.

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Opening Remarks¹

Robert Z. Callahan²

My reasons for being here are to welcome you on behalf of the Forest Service, U.S. Department of Agriculture, and to introduce you to this conference. I want to explain to you the objectives of the conference, why it was organized, and who made it possible. But I also want to introduce you to each other, so that you will know which countries you represent, and why you are here.

I will begin by stating the three objectives of this conference. The first is to review current information on specific gaseous and particulate pollutants and their effects on forest ecosystems. Forests as sources of pollutants and as sinks for pollutants are included. The second objective is to analyze primary, secondary, and interactive effects of chronic pollutants on ecosystems. Modeling will be explored as a tool to simulate observed and expected effects. Late in the program, strategies for assessing and managing environmental impacts of air pollutants will be discussed. The third objective is to stimulate international communication to assess the state of knowledge and to identify gaps in our knowledge.

Several years ago, the U.S. Environmental Protection Agency (EPA) gave a grant for research to Dr. Clif Taylor in the Statewide Air Pollution Center here at Riverside. The Forest Service has actively cooperated and participated in the resulting research up to the present time. Research under that grant is drawing to a close. This conference was planned, therefore, to sum up what has been accomplished. Scientists involved in this multimillion dollar research effort have this opportunity to tell others what they have learned

and to identify problems that remain to be solved. Another purpose in organizing this conference is to broaden local perspectives by importing experts from around the world to talk about our problems. The final purpose is to transfer technology generated by this research program to the managers of air, land, and forest resources.

This conference was made possible through the cooperative efforts of several agencies. The Forest Service, U.S. Department of Agriculture, and the Statewide Air Pollution Research Center, University of California, at Riverside, are sponsors. IUFRO, the International Union of Forestry Research Organizations, having about 380 member organizations in 86 countries around the world and a subject group concerned with air pollution, is a cosponsor. The U.S. Department of Energy and the U.S. Environmental Protection Agency have made their people and their resources available. EPA will help the Forest Service to publish the proceedings. The U.S. Department of State, particularly the Man and the Biosphere Program through its Project 2--Mediterranean and Temperate Forest Ecosystems--has provided financial support. UNESCO, the international home of the Man and the Biosphere Program, has paid to bring three international participants here. For all of this support and cooperation, the organizers are most grateful.

My final and, perhaps, unexpected reason for being here is to introduce you to each other. Although this is not often done at conferences, I have found it to be an effective means of stimulating communication. I am going to call the roll of countries, more or less in alphabetical order, and ask the individuals named to stand. (Introductions followed.) About 20 percent of the people here are from outside North America. Let us give these visitors a special welcome. Walk up to them. Introduce yourselves. Ask these visitors about programs and problems in their countries.

Now that we know what countries are represented, let me call for a show of hands to find out why

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130 of you are here. How many are primarily teachers or professors? About 7 percent. How many are managers of land or forest resources? About 9 percent. How many are managers of air resources? Only about 2 percent. That is surprising. How many of you are students, not yet into professional activities? About 2 percent. The remainder of you--about 80 percent--are scientists and investigators. That is about what I expected.

And now my role is fulfilled. I have introduced you to the conference and to each other. I expect you to benefit both professionally and personally from this conference. Lastly, I express my deep appreciation to all who have contributed to the organization of this conference and particularly to Dr. Paul Miller.

On behalf of the Forest Service, I thank you for coming. The ultimate success and meaning of this conference depend on you.

Air Pollution in Forests: Social Costs, Predictive Models, and Public Policy¹

Charles F. Cooper²

Abstract: Long time scales, spatial variation in ecosystems, and differing value judgments make models almost essential for societal consensus about air pollution. Three categories of policy-oriented models are described. Empirical time series models are good for immediate decisions but are inherently a short-term device. Detailed structural-functional models emphasize relationships among components and demonstrate the significance of interconnections. Small errors, however, can lead to erroneous quantitative results, limiting their value for direct policy decisions. Aggregated policy-oriented models provide better compliance between model output and validation data at the cost of loss of resolution. Good models should be clearly documented, results should be comprehensible, limits and probable error bands clearly stated, they should be flexible enough to deal with unanticipated problems without attempting total generality, and results should be clearly displayed. A model is an aid to decisionmaking, not a decision maker. For it to be effective in that role, there must be mutually supportive interaction among modelers, biological and social scientists, and decision makers. Perhaps the most significant role of models is in helping to avoid suboptimization and in facilitating communication among disciplines and practitioners.

Atmospheric pollution is affecting forest ecosystems in much of the world. A major purpose of this symposium is to establish a scientific consensus about the nature, magnitude, and time trend of these effects. A scientific resolution, however, is not enough. Analysis of "The Effects of Air Pollutants on Mediterranean and Temperate Forest Ecosystems" must also take societal objectives and limitations into account.

Ecosystems used and enjoyed by man are embedded in a larger social system; dealing with effects of air pollution on these ecosystems, locally, regionally, or globally, thus becomes a question of public policy. Essential to sound public policy formulation is knowledge of the social costs of air pollution and its control. A major problem, of course, is how to measure these costs. This symposium should lead us some way toward better assessment of the real costs of forest air pollution.

The ecological and social consequences of air pollution in forests are the result of complex interactions of processes with many temporal, spatial, and value scales. Long-time scales, variations among ecosystems, and differences of opinion about values make classical laboratory experimentation almost useless for defining these large scale consequences. Models are thus essential for helping to arrive at a societal consensus about how to treat air pollution.

COMPLEXITY OF THE FOREST AIR POLLUTION PROBLEM

It is a truism that the forest air pollution problem is complex, but the nature of that complexity must be understood if we are to visualize the role of models in dealing with it. The points mentioned here are elaborated by other authors in this volume.

Air pollution affects individual plants directly, in ways which change during the course of the plant's life history. It affects plants indirectly, through impacts on soils and on consumers and decomposers. Effects on plants, in turn, are reflected in other trophic levels. Critically important is the dynamic reaction of ecosystems, which usually cannot be predicted from a simple summation of the responses of individual organisms.

Social consequences stem from loss of productive resources and amenity values. Wood and

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forage growth may diminish, a loss that will be increasingly significant if forests are in greater demand for biomass as a source of energy or structured chemicals, or if increasing need for grain as food leads to more pressure on rangelands. Aesthetic and recreational values are lost. The degraded appearance of the smog-affected forests of the San Gabriel Mountains are apparent to all who have seen them. Wildlife, both game and non-game, may suffer. Lakes and streams in several parts of North America and Europe have lost much of their capacity for fish production (Loucks 1980). Of course, there may be beneficial impacts as well--alleviation of local sulfur deficiencies, for instance. Both beneficial and detrimental effects vary in time and space.

Air pollution operates at many time scales. Holling (1973) has distinguished between fast and slow variables. Fast variables are generally amenable to conventional laboratory experimentation, and are the kind that are usually studied in biological research. Slow variables, however, take long enough to manifest themselves that controlled experimentation is impractical in many real world situations where action cannot wait. Decision makers must also deal with a high degree of spatial heterogeneity in both pollutants and their target ecosystems. Theory and research in ecology has not until now dealt very well with spatial processes.

Finally, air pollution is only one of many stresses affecting forest ecosystems. Multiple stresses may interact synergistically or they may help to counteract one another. We need to know more, for instance, about the combined impact of air pollution and climate change, whether due to deliberate weather modification or inadvertent climatic change. Increased atmospheric carbon dioxide from burning of fossil fuel seems likely to warm the earth's climate and, perhaps, to stimulate plant growth directly. How will these processes interact with air pollutants? Forest harvest and regeneration is itself a stress on the ecosystem which will interact with increased pollutant loading. Air pollution in forests thus is part of a complex network of biological and social interactions whose integrated impacts are almost impossible to untangle through single-factor analysis.

A SCIENTIFIC APPROACH TO COMPLEXITY

There are some four possible societal responses to complex problems such as air pollution. We can attempt to treat the symptoms through such means as fertilization or irrigation, alleviate the cause through emission control, accept the degradation as gracefully as possible, or convert the affected ecosystem to one more resistant to stress. Actual policy solutions will probably include some combination of these.

How do we go about choosing the appropriate policy response? One way is simply to rely on the

judgment, hopefully good, of the people, hopefully experienced, in charge. This is the common procedure. There is, however, a more organized scientific approach to complex problems with many temporal and spatial scales. This process includes eight basic steps.

1. Make a model of the process, based on existing knowledge and understanding of the system. The kinds of model which might be undertaken in this step are discussed in more detail below.
2. Fit the parameters of the model to data, preferably obtained from laboratory or field experimentation; otherwise from observational studies.
3. Validate the model. This involves comparison of model results with real world outcomes in systems other than those used in fitting the parameters. This is a crucial but also a most difficult step, because the objective of the whole modeling exercise is often to predict responses of systems under stresses that exceed the range of existing validation data.
4. Test the sensitivity of the model to parameter changes. This can help to locate critical features where better understanding or more accurate data are needed. It can also help locate parts of the system where relatively small changes may have large effects. Sensitivity analysis is often said to be one of the great virtues of a modeling approach, in that it leads to directing limited resources to areas where they will do the most good, or conversely avoids the expenditure of effort on measures unlikely to have much effect. Points of special sensitivity are often hard to find, however. Both control theory and practical experience are increasingly demonstrating that many complex interlinked systems are relatively insensitive to small changes in one or two variables. This should come as no surprise to those who have observed the evident resilience of ecosystems under stress (Holling 1973). Sensitivity analysis remains, nevertheless, an important application of policy-oriented models.
5. Use the model outside the ranges of stresses previously experienced. One of the pitfalls that all of us have been warned to avoid in science is extrapolation, yet it is just because of the need for extrapolation that models are called for in predicting ecosystem consequences of air pollution. The response of the model system will usually need to be estimated under pollution loads greater than those already experienced by that system. Even more important is the time dimension. A primary goal often is assessment of the long-term consequences of chronic or episodic air pollution. It is just because of this

extended time dimension that models are needed, and yet this is perhaps the most difficult element in their construction.

KINDS OF POLICY-ORIENTED MODELS

Models may be merely conceptual and verbal--an intuitive, possibly quite accurate, visualization of how the world works. We are concerned here, however, with mathematical models able to deal with interactions among more variables than the unaided human mind can readily handle. These are of several basic kinds, which differ in both their underlying structure and their range of application. (This section owes much to discussions with W. R. Emanuel, Environmental Sciences Division, Oak Ridge National Laboratory.)

6. Array the output for public discussion. Since the purpose of the models we are considering here is to help in arriving at some sort of consensus about appropriate societal response, presentation must go beyond the immediate scientific community. Seldom, if ever, will the output of a realistic air pollution model lead to deterministic predictions. Rather, there will be a range of alternative outcomes, each with a probability level attached. It is notoriously difficult to interpret risk probabilities in terms of public attitudes. There is a growing literature on probabilistic risk assessment which is highly pertinent to the air pollution problem (e.g., Kates 1978, Starr and Whipple 1980).
7. Amalgamate with output of other relevant models of societal issues for final public evaluation. Air pollution is only one of many problems facing society. Measures taken to alleviate the consequences of air pollution are likely to ramify into many other aspects of society. In my view, the most important single use of a policy-oriented model is as an aid in avoiding suboptimization. By suboptimization, of course, is meant choosing what is clearly and logically the best solution to a small part of a problem without adequately considering the impact of that solution on the total system. An example of how suboptimization along a narrow path may turn out to be not just slightly wrong, but exactly wrong in a broader context is the disposal of chemical wastes at Love Canal, N.Y. Out-of-sight, out-of-mind burial was a good solution at the time for the potential hazard to workers and the public of this material; its consequences are now affecting all Americans, as taxpayers, if not as recipients of direct chemical insult. Less extreme cases of suboptimization may be more difficult to identify before action is taken. If a properly designed model, by exploring a wider range of alternatives than can the human mind alone, helps to avoid the long-term costs of suboptimization, the effort in its construction will be well rewarded.
8. Move toward a decision. There is no hope that a decision, even if based on the best conceivable model, will satisfy all interested parties in a controversial issue. One could expect, however, that the decision would be more rational than if based on emotion and maximization of each participating individual's personal objectives.

1. Empirical time series analysis. The emphasis here is on analysis of the secular trend of the variables of interest. It is assumed that the processes during the period of record will continue over the interval of extrapolation. A time series model need not include explicit casual relationships. It must, however, incorporate sufficient data to establish the statistical significance of the observed patterns. In the words of Dennis Meadows (1975), it is "data rich, theory poor."

Such a time series analysis is often ideal for decisions which must be made immediately but which can be revoked in the light of new information without lasting damage, biological, or political. Time series analysis has the advantage that it is easily understood by decision makers who are not analytically inclined. It is inherently a short-term tool, however. Lack of explicit casual relationships makes extrapolation even more risky than with other models.

2. Detailed structural-functional models. These incorporate the structure and function of the system to the extent that it is known. There is a wide variety of suggested procedures and approaches for constructing such models; several are discussed in this volume. The emphasis throughout is on understanding relationships and processes, not trends. In Meadows' (1975) words, they are "theory rich, data poor."

Their principal value is as an aid to understanding relationships among components and the significance of interconnections. They are often useful for pointing out to decision makers why certain relationships whose importance is not intuitively obvious are actually more significant than they seem.

Properly constructed functional models can lead to prediction of ecosystem responses to stresses, which are likely to differ markedly from those of individual organisms tested in isolation. West and others (1980) used a model of successional dynamics to test the long-term impact of air pollution on eastern deciduous forests. Their model predicted

enhanced growth of some species despite pollutant stress, since they may gain a competitive advantage because they are less sensitive than other species with which they interact in the successional process.

Neither ecological models, nor ecological theory in general, deal well with spatial dynamics in forests. Most emphasis has been on successional dynamics over time at a point or in a small area. There have been attempts, as by Shugart and others (1973) to model the "flow" of one form of land use or vegetation condition to another, but this approach is chiefly usable for very large units. Because air pollution is both spatially extended and spatially variable, there is a need to incorporate these features into ecological models of air pollution. A promising approach seems to be the linking of existing forest succession models with cartographic models developed by geographers for dynamic map analysis. Efforts to do this are now underway in several research units. The results should be valuable for air pollution studies.

Despite their great value for many purposes, however, detailed structural-functional models are usually unsuitable for deciding upon specific actions or policies. Small errors, either in theoretical underpinnings or in parameter estimation, can lead to quantitative predictions that turn out to be quite wrong when tested against the limited validation data usually available. We can hope, though, that improvement in both modeling technique and in biological and social knowledge will lead to functional models truly useful as decision tools.

3. Aggregated policy-oriented models. Here, there is an attempt to combine the many structural elements of the system into a relatively few well-understood components for which good cause and effect data exist. The major structural and functional relationships are preserved, but at a lower level of resolution with respect to their interconnections. Full understanding of the complex system is traded for greater computational tractability, and a greater possibility of showing time series data.

With this sort of model in hand, scientists can interact with decision makers in a quantitative way to prepare an array of alternative actions and their probable consequences. Various tools of optimization, an active area of current research, come into play here. Particularly valuable in a problem with as many conflicting value judgments as effects of air pollution may be the multiple objective optimization extensively used in evaluating water resource development alternatives (Cohon and Marks 1975). This is a planning concept which

provides a quantitative framework for the task faced by all decision makers, that of achieving an acceptable compromise among a set of competing objective functions. The various analytical techniques used for this purpose agree in using the model to offer tradeoff functions of some kind to the decision maker. But he, not the model or the modeler, establishes priorities among the planning criteria. The model is thus just one more tool for bringing precision into the planning process and for evaluating the consequences of alternative choices.

Ecological policy models would be substantially improved if they could directly incorporate human decision making in response to ecosystem change. Active research is in progress to make this possible. For instance, C. L. Smith, J. M. Stander, and A. V. Tyler (personal communication 1980), of Oregon State University, an anthropologist, an ecosystem modeler, and a fisheries biologist, have collaborated in developing an interactive model of a mythical human hunting and gathering society and an exploited fishery. Human participants are faced with alternatives which force them to make choices. The consequences of the choices are then evaluated by the models in accordance with previously developed decision rules. They found that decision making in simulations did alter model outcomes. Decisionmaking in the fishery simulation led to a stable equilibrium; without it, there was periodic cycling of fishing vessels and fish biomass. Wrong decisions, though, led to extinction--economic extinction of the fishery before biological extinction of the fish. Similar interactive modeling ideas are being developed by Holling (1978) and his associates and followers under the rubric of Adaptive Environment Assessment.

DESIRABLE CHARACTERISTICS OF MODELS

To be effective tools for assisting in policy decisions, models should have several characteristics (Cooper 1976). It is almost totally irrelevant in this context whether the model uses differential or difference equations or whether it is written in FORTRAN or BASIC. There are more fundamental features which determine whether a model is likely to be accepted and used in decisionmaking.

1. It should be extensively and clearly documented. How was the model developed and what are its assumptions? This should be evident, if not to the ultimate user, at least to those who might serve as technical consultants. Unfortunately, this is a serious weakness of most models. This arises partly from funding limitations--documentation is an obvious thing to skimp if the budget has to be cut. I suspect that not infrequently, though, there is a wish, maybe subconscious, to keep what the modeler did to himself. After all, knowledge is power.

2. Its results should be understandable--surprising, perhaps, but not incomprehensible. Forrester (1971), in a widely quoted article, discussed the counterintuitive nature of social systems, and prescribed computer modeling as an antidote. But Forrester was able to explain quite clearly how he obtained his counterintuitive results (others disagree with his analysis, but that is irrelevant here). He would have had no credence whatever if he had not been able to provide such explanation.
3. The limits and probable range of errors should be well explained. Few computer models yield deterministic results, and all are limited in their acceptable degree of extrapolation. This is often not well understood by those not analytically inclined; it needs to be made clear.
4. The model should be flexible enough to deal with problems that had not been fully anticipated, but a general all-purpose model is not a desirable goal. Senator S. I. Hayakawa's dictum, "The map is not the territory," although made in quite another context in his role as a semanticist, is wholly applicable to modeling. A model is a map that tells us how to get from one place to another, even to some places we had not originally intended to visit. It cannot reproduce every feature of the system and still retain its usefulness as a guide.
5. The results should be displayed in an effective and understandable manner. Too often model results are presented as arcane and incomprehensible printout. Territorial defensiveness again? Results need to be presented in a form comprehensible to those who react to graphs rather than to columns of figures, to pictures rather than to equations. Visual presentation almost never receives enough attention.
6. The model should be portable--usable on other computers with a minimum of reprogramming. I have the feeling that incompatibility has become worse in recent years. This will presumably correct itself eventually, but for now it is a serious problem.

CONCLUSIONS

The impact of air pollution on forests, especially when combined with other stresses, has biological, social, and political implications which operate at several time scales extended over space. The unaided human mind is not well adapted to explore the consequences of each of the large number of possible combinations of variables. Therefore, some sort of computer model is virtually essential if the most reasonable array of alternatives is to be presented for rational choice.

A policy-oriented model is not a decision maker. It is an aid to informed decisionmaking. If it is to function effectively in that role, there needs to be mutually supportive interactions among modelers, biological and social scientists, and decision makers.

A modeler dealing with ecological public policy questions related to air pollution in forests needs the knowledge of a biologist, to understand the essence of the mechanisms by which air pollution affects biological processes. He needs the skills of an applied mathematician, to understand the structure of the model and its implications, and usually to direct the programmers preparing the actual computer code. Most important, he needs the patience and interpersonal abilities of a diplomat, to persuade the biologist and the decision maker alike that he is helping them to do their job better, and not usurping their legitimate roles.

Biological and social scientists dealing with modelers need to recognize that modelers are there to help them, not vice versa. A good model, properly presented, will usually provide the decision maker with a fuller explanation of the consequences of alternative policies than will unaided scientific statements or position papers. Thus, the scientist may be better able to get his points across through the medium of a well-constructed and well-presented model.

Decision makers should realize that a good model is primarily a means for exploring the consequences of alternative policy choices of nearly equal rank. If the model shows one or two choices to be so superior to others that only they should be considered, this will surely be obvious to competent analysts in the absence of a model. The real utility of a policy-oriented model is to explore those situations where there are a large number of alternatives which are neither all good nor all bad. The broader the array of choices set up for exploration, the greater the final range of opportunities will be.

Finally, I suggest that the greatest value of a modeling approach to ecological problem solving may be its stimulus to exchange of information among disciplines and among practitioners. I mentioned, earlier, the relatively unsatisfactory incorporation of spatial dynamics in forest models. Until a few years ago, this would have been true of environmental biology generally. There have been recent significant advances, though, in dealing with spatial heterogeneity in insect dispersal and in dynamics of marine plankton. In taxonomically oriented science, such advances take a long time to come to the attention of workers in adjacent fields. The fact that modeling approaches tend to cross taxonomic boundaries seems likely to cut this unnecessary time lag. This alone would be adequate justification for wider use of modeling in a complex interdisciplinary field such as the effect of air pollutants in forests.

LITERATURE CITED

- Cohon, J. L., and D. H. Marks
1975. A review and evaluation of multiobjective programming techniques. *Water Resour. Res.* 11:208-220.
- Cooper, Charles F.
1976. Ecosystem models and environmental policy. *Simulation* 26:133-138.
- Forrester, J. W.
1971. Counterintuitive behavior of social systems. *Tech. Review* 73(3):53-68.
- Holling, C. S.
1973. Resilience and stability of ecological systems. *Ann. Rev. Ecol. Systematics* 4:1-23.
- Holling, C. S. [Ed.]
1978. Adaptive environmental assessment and management. Wiley Inter-Science, Chichester, U.K. 377 pp.
- Kates, Robert W.
1978. Risk assessment of environmental hazard (SCOPE 8). John Wiley and Sons, New York, 112 pp.
- Loucks, Orie
1980. Acid rain: living resource implications and management needs. *Trans. 45th North Amer. Wildlife Nat. Res. Conf.* [In press]
- Meadows, Dennis
1975. A critique of short-term perspectives implicit in most resource models. *In Mineral Materials Modeling: A State-of-the-art Review.* W. A. Vogely, ed., pp. 66-80.
- Shugart, H. H., Jr., T. R. Crow, and J. M. Hett
1973. Forest succession models: a rationale and methodology for modeling forest succession over large regions. *Forest Sci.* 19:203-212.
- Starr, Chauncey, and Chris Whipple
1980. Risks of risk decisions. *Science* 206:1114-1119.
- West, D. C., H. B. McLaughlin, and H. H. Shugart
1980. Simulated forest response to chronic air pollution stress. *J. Env. Quality* 9:43-49.

Historical Perspectives and International Concerns About Air Pollution Effects on Forests¹

Edwin Donaubaue²

Abstract: Air pollution from man's activities has a long history; real hazards for forest vegetation occurred more than a century ago which marked the starting point of forest research in the field. The objectives of research have been subject to a steady metamorphosis from simple causal relationships to the task of investigating the long-term influence of pollutant mixtures on trees directly and on entire ecosystems. The IUFRO Subject Group S2.09, Air Pollution, encourages interdisciplinary work in the field and offers an organizational home for close cooperation on an international basis. The consideration of air pollution effects on forest ecosystems goes far beyond limited forest interests, the findings can provide fundamental data on the deterioration of the human environment.

Some forest insect pests develop outbreaks characterized by high insect population densities that use up the food resource represented by the host species. The direct damage by insects and the predisposition of weakened trees to certain diseases both result in a serious degradation or even destruction of the local forest environment.

To some extent man-made air pollution problems show some similar aspects. In many regions of the world mankind cleared the forests for agriculture, for settlements, for roads, for mining, for outdoor recreation, etc. Wood

is still in high demand as an energy source, and in fact is still the most important source for over 90 percent of the world's population. Collection of fuel wood and the harvesting of timber for many uses have influenced the extension and quality of forests over long periods of prehistoric and historic time. All these activities had, at least in some major regions of the globe, serious consequences for the ecosystem and human environment. We are reminded of the historical fact, that 2000 years ago forests existed around the Mediterranean Sea and protected large and prospering agricultural areas in North Africa, where deserts are now present. These were direct influences, but among others an additional indirect threat developed by the quick progress in technology and by increasing quantities and numbers of toxic substances released into the air.

Certainly, man-made air pollution has been mentioned long ago in historic times, but the dimensions of the problem have changed in relation to regional economic and technical developments. The effects of air pollution to forest ecosystems were and seem still to be underestimated, maybe for these reasons:

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- symptoms are often unspecific, or develop late and slowly, or are even invisible (as depression of growth increment);
- serious changes in forest ecosystems may become obvious only after many years of accumulation of low concentrations or amounts of toxic substances (heavy metals, alkaline dusts, acid rain).
- trees are more sensitive to widely distributed pollution than humans and this difference is not fully known or realized.

REMARKS ON HISTORY

Perhaps Pliny (65 A.D.) was the first to observe and describe apparent SO₂-damage to vegetation surrounding a smelter. Later on we find references frequently in documents of the Middle Ages concerning air pollution by coal burning; in general people of higher social classes felt inconvenienced. Especially in such reports from England and Central Europe persecutions happened even at that time if air pollution was caused in a prohibited area or time.

Air pollution became more than a local and occasional affair when industrialization and the use of fossil energy resources increased rapidly in Western and Central Europe during the past century. By the time forest research institutes and forest faculties were established - more than a hundred years ago - they had several reports of prior experience and observations of botanists to draw upon. Therefore forest research was stimulated to pay attention to air pollution problems from the beginning. The present research projects provide numerous publications on air pollution effects.

CHANGING AIMS AND TASKS

The objectives of research projects concentrated first on sulphur dioxide including methods for detection and evaluation of air pollution damage. Among these early works we find very forward-looking ones, as Rusnov's (1919) study about the influence of 'acid rain' (this term was introduced much later) on forest soils. Several authors proved the use of chemical analysis of foliage for detection and evaluation of air pollution (SO₂, HF) and its effects on forest stands. These publications caused long lasting discussions and stimulated further investigations; however, many years passed until the chemical analysis of foliage (needles) became a useful tool for a great number of cases. Step by step the method was refined, so that today sufficient knowledge is available for proper sampling, and the thresholds of natural sulfur and fluorine contents and their variation are better defined.

Certainly, sulfur dioxide is still the pollutant occurring over largest areas, but the list of substances thus far identified as responsible for damage to forest trees becomes longer and longer. More attention is now given

to indirect and chronic effects of acidic or alkaline precipitation and heavy metals on forest ecosystems.

Formerly, the interest of forest research concentrated on a single-pollutant-single-tree species relationship. Today, increased efforts are invested in elucidating the complex influences on forest ecosystems and studying the interaction of pollutant mixtures which occur in many cases.

Regional investigations on air polluted forest zones use all assistance of modern technology, such as remote sensing combined with physiological, chemical and tree mensuration methods for verification.

Forest air pollution problems are linked with the economic standard in general and with certain industrial development.

But it is not necessarily so, that only point sources of air pollution or concentrations of industries cause serious problems. Automobile traffic or even a single source area can under specific orographic and meteorologic conditions lead to pollutant accumulation and perhaps to heavy damage. Further, we are learning more about long distance pollutant transport over hundreds and thousands of kilometers.

The experience in nations with highly developed industry makes it advisable to stimulate more interest in this field of research in many developing countries in the world. First of all the experiences of others should be transmitted and applied in all planning for economic development to avoid the same historical lessons received by many developing countries. There are many forest research workers and institutions in North America, Europe (including the Asiatic part of USSR), and Japan who are experienced with the various aspects of air pollution effects on forest ecosystems and could provide scientific information and assistance.

I U F R O AND AIR POLLUTION RESEARCH

The International Union of Forest Research Organizations (IUFRO) has six major organizational divisions. Division 2, 'Forest Plants and Forest Protection' is home for ten Subject Groups, one of which is S2.09 - 'Air Pollution.' The titles of the various Working Parties elucidate the interdisciplinary scope: Researchers are included from the fields of forest mensuration, soil science, pathology, remote sensing, tree physiology, etc. They work together, and exchange research results that stimulate their further research. This IUFRO Subject Group has a long history of activity. For more than a quarter of a century meetings have brought together an increasing number of participants. A few western and central European countries were represented at first; now more countries from eastern and southern Europe are represented too, as well as a slowly increasing number of participants from Canada and the U.S.A. The eleventh meeting of all Working Parties will

be held this year in Graz, Austria and previous conferences were hosted by nine other European countries. It is somewhat disappointing that the group met only once in another continent: Gainesville, Florida, U.S.A. at the occasion of the IUFRO Congress in 1971. We expect to have a meeting at the occasion of the 1981 IUFRO Congress in Japan.

The interdisciplinary scope of all Subject Group meetings makes them attractive to a certain group, seldom represented at such scientific conferences. We could characterize this group as 'consumers' of research results including: forest managers, representatives from industries, and sometimes politicians. There are of course some difficulties in accommodating the needs of each group, but I consider the interest in such meetings as an expression for a demand to receive the information from the scientists directly and not via more traditional channels, i.e., technical publications. I feel we should think of possibilities for such direct information flow in specific fields like air pollution where we find more and more vital interest from the public.

I feel that this international symposium sets the stage for new conceptualizations by placing the view of effects of air pollutants on forest ecosystems into the center of the deliberation. The discussions here should stimulate further research and interdisciplinary, international cooperation.

In addition I will emphasize that air pollution effects on forests are not only a question for researchers or foresters, but of fundamental interest to mankind. That may seem overstated but forest ecosystems demonstrate due to the long rotation periods the dynamics and the consequences of long-term influences on complex ecosystem-level processes. Various pollutant effects that are dangerous not only to the forests themselves, as an important resource, but also as an invaluable part of the human environment can be predicted by ecosystem-level research. In this sense one can consider the forest ecosystems as sensitive warning systems for the vital, fundamental interests of life.

Natural Influences of Forests on Local and Regional Air Quality

Emissions and Air Resource Management Within Forests¹

Michael A. Fosberg and Hollis Record²

Abstract: Substantial portions of the emissions inventory within forested lands are from dispersed intermittent sources. Chief sources are smoke from wildfire and prescribed fire, emissions associated with concentrated recreation and second-home developments, and fugitive dust from unpaved roads and eolian soils. Effects of smoke on flora range from reduced photosynthetic efficiency at low dosages to tissue necrosis at high dosages. Effects on fauna are not clearly defined. Effect of smoke on social values, primarily visibility, is recognized but not understood. Dispersion process in complex terrain, the physiographic setting for most forested lands, is complicated by topography and spatially varying wind fields, a higher degree of anisotropy of turbulence, and a wider range of turbulence intensities than found over level ground. Management of air resources within forested areas is limited to land management planning activities because of the complexity of emission characteristics, dispersion processes, and effects of pollutants from within forest sources.

Much of this symposium treats effects of pollutants from major stationary sources on ecosystems. Significant pollutants treated in other papers are nitrous oxides, oxides of sulfur, reactive hydrocarbons, and the photochemical derived pollutant, ozone. Effects of NO₂, SO₂, and O₃ are well documented in the companion papers in this volume.

Pollutants from within forest sources also include emission from wildfire, prescribed fire, unpaved roads, eolian soils, concentrated recreation, and second-home developments. Chief emis-

sions from fire are CO₂, CO, particulates, and hydrocarbons. Recreational and second-home developments emit CO₂, CO, particulates, and hydrocarbons from fireplaces and campfires; SO₂ and NO from dispersed transportation systems^x—namely, private automobiles. Fugitive dust from unpaved roads and eolian soils also contribute to the particulate loading.

Effects of the above pollutants, particularly smoke, flora, fauna, and social values are poorly understood. Documented effects range from reduced photosynthetic activity through blockage of solar radiation to tissue necrosis. Effects on microorganisms range from inhibition of some spores and fungi to increased germination of one fungus. Effects of smoke on fauna are documented, but without explanation.

Visibility is both a physical and social value. Visibility can be quantified in terms of visual range and ability to define details at specified distances. Visibility is also a personal value based on past and expected experiences. A recent popular country and western song (McCall and others 1976) goes "..... One of the guys from New York said 'Hey, look at the smog in the sky, smog clear out here in the sticks'. Someone said, 'Hey Joe,

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that's not smog, that's the Milky Way', Joe had never seen the Milky Way" Contrast Joe's reaction to that of a resident of the Four Corners area of Utah visiting any urban area in the world on a clear air day. Local residents would comment on the clarity of the air, but the visitor would notice the impairment of visibility.

The relationship between a source of pollution and the effect of that pollutant on Mediterranean and temperate forest ecosystems is through dispersion of the pollutant between the source and the receptor point. Nearly all dispersion calculations are based on the Gaussian model in which transport is treated through definition of a mean windspeed and direction and turbulent diffusion is based on a transformation of the turbulence structure to a Gaussian statistical distribution (Turner 1969). Validity of the coefficients used in the Gaussian model are uncertain in situations where terrain features are complex. In particular, winds are known to contain a high degree of spatial and temporal variability (Fosberg and others 1980) and the turbulence intensities are highly anisotropic.³ Because much of the Mediterranean and temperate forest ecosystems are found in complex terrain throughout the world, the dispersion or delivery system of pollutants from the source to the receptor must account for complex terrain atmospheric processes.

An understanding of each physical and biological process is necessary but not sufficient to develop management plans for air resources within forests and brushlands. Acts, laws, regulations, and codes established by Congress down through local county regulatory agencies specify goals and objectives for air quality and frequently specify the methods in which air quality objectives will be met. As example, organic acts of most Federal agencies in the United States require that the agency protect or preserve, or meet air quality objectives. The Clean Air Act of 1977 (U.S. Congress 1977) specifically requires that air quality objectives be met through emissions control. Because it is not feasible to install scrubbers on prescribed fire, emission control is achieved through emissions density planning.

Each of the following sections addresses the specific topics of emission characterization, dispersion in complex terrain, effects of smoke on flora, fauna, social values, and air resource management.

EMISSIONS FROM WITHIN FOREST SOURCES

Major emissions from wildfire and prescribed fire are CO₂, CO, hydrocarbons, and particulates.

³ Lanham, Lucy M. 1980. Wintertime dispersion processes in the Lake Tahoe Basins. Proc. 2nd Conf. on Application of Air Pollution Meteorology. Amer. Meteorol. Soc. [in press]

Although CO₂ is not a pollutant as such, CO₂ is of considerable interest in analysis of the global heat balance. Carbon dioxide emissions range from 1000 kg per metric ton of fuel to 1750 kg per metric ton of fuel (Ryan and McMahon 1976), with extreme values near 1830 kg per metric ton (Vines and others 1971).

Carbon monoxide emissions from fire are highly dependent on combustion efficiency. Values range from 17 to 98 kg per metric ton (Sandberg and Martin 1975, Darley and others 1966, Gerstle and Kemnitz 1967). The U.S. Environmental Protection Agency (1978) recommends a value of 45 kg per metric ton from hemlock, Douglas-fir, and cedar and 98 kg per metric ton from ponderosa pine. Inefficient combustion for smoldering damp fuels have resulted in emissions as high as 250 kg per metric ton of fuel (Ryan and McMahon 1976). Emissions as high as 250 to 400 kg per metric ton of fuel have been reported when energy release from fire is less than 750 watts per square meter (Sandberg and Martin 1975).

Hydrocarbon emissions range from 2 to 7 kg per metric ton of fuel (U.S. Environmental Protection Agency 1978) although emissions as high as 20 kg per metric ton have been reported (Ryan and McMahon 1976, Darley and others 1966). Speciation of hydrocarbons (fig. 1) shows saturated hydrocarbons (mostly methane) comprise about 30 percent at peak fire intensity and about 15 percent at low fire intensities (Sandberg and others 1979). Low molecular weight olofines make up about 17 percent of the emission from flaming fire and 3 percent from smoldering fire (Sandberg and others 1979).

Emissions of SO_x and NO_x are negligible. Most fuel contains less than 0.2 percent sulfur and combustion temperatures are low, preventing formation of NO_x.

Particulate emissions from fire are given as 2 to 6 kg per metric ton of fuel (U.S. Environmental Protection Agency 1978). A range of emissions are given in table 1. Particulate sizes are mainly in the submicron diameter classes with only a few particles larger than a micron (fig. 2). Particulate sizes are dependent on combustion efficiency (Schaefer 1976) with the most efficient fire-producing sizes in the respirable range.

National emissions of particulates from prescribed fires and wildfires are shown in figure 3. Prescribed fire consists of less than 20 percent of all fire emissions nationally. Emissions of the major pollutants, particulates, CO, and hydrocarbons by State and regions are given in table 2 for prescribed burning.

In addition to wildfire and prescribed fire, concentrated recreation and second-home developments contribute smoke through fireplace burning and more increasingly through use of wood for home heating.

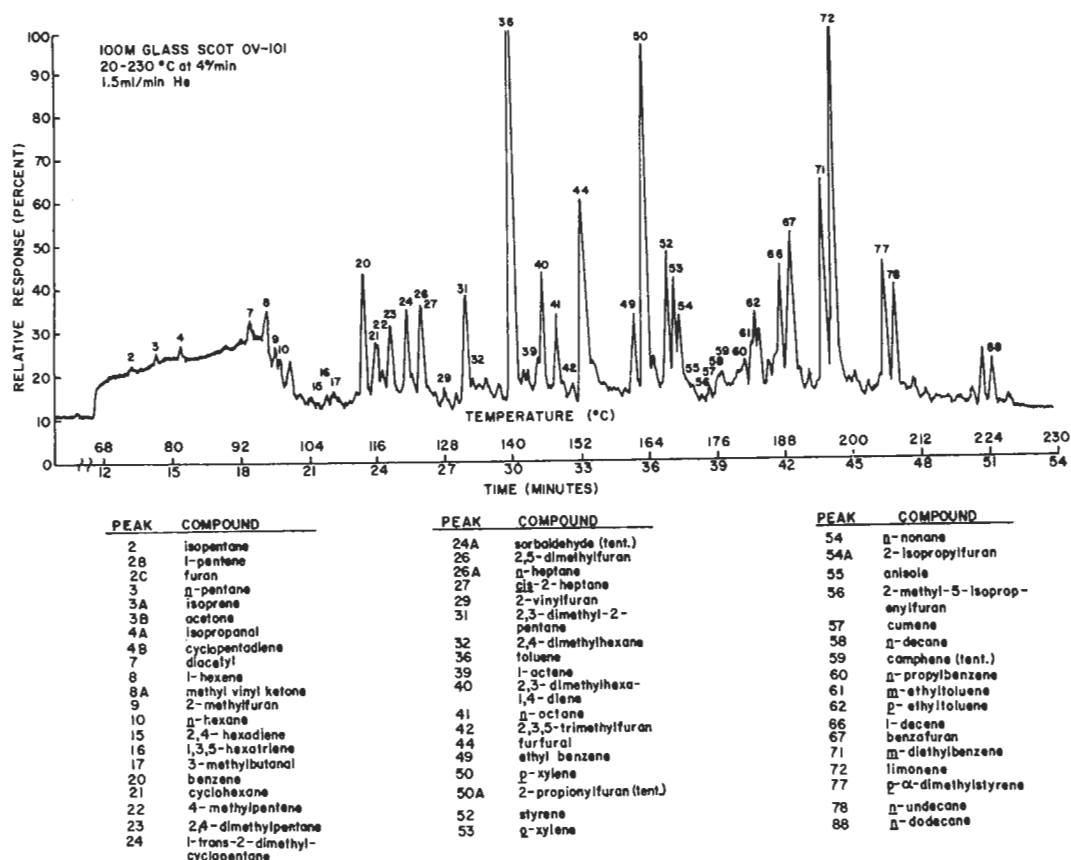


Figure 1--Chromatogram of organic vapors in loblolly pine smoke (from Ryan and McMahon 1976).

Table 1--Summary of particulate emission yields reported from wildland fuels (from Sandberg and others 1979)

Fuel type	Lab/field experiment	Particulates (kg per metric ton of fuel burned)		Reference
		Type of fire		
		Heading	Backing	
Logging residues (Western)	Field	14-53		Sandberg (1974)
	Laboratory		3-12	Sandberg (1974)
	Field	~40		Radke and others (1978)
	Laboratory		2	Fritschen and others (1970)
Landscape refuse	Laboratory		12	Feldstein and others (1963)
Grass burning	Field		8	Boubel and others (1969)
Live understory (Australia) (Southern)	Field	7-20		Vines and others (1971)
	Laboratory	14-20		Vines and others (1971)
	Field		7-15	Ward and others (1976)
	Laboratory		12-49	Ryan (1974)
Pine litter (Southern)	Field		22-27	Ward and others (1976)
	Laboratory		3-14	Ryan and McMahon (1976)
	Laboratory	11-63		Ryan and McMahon (1976)

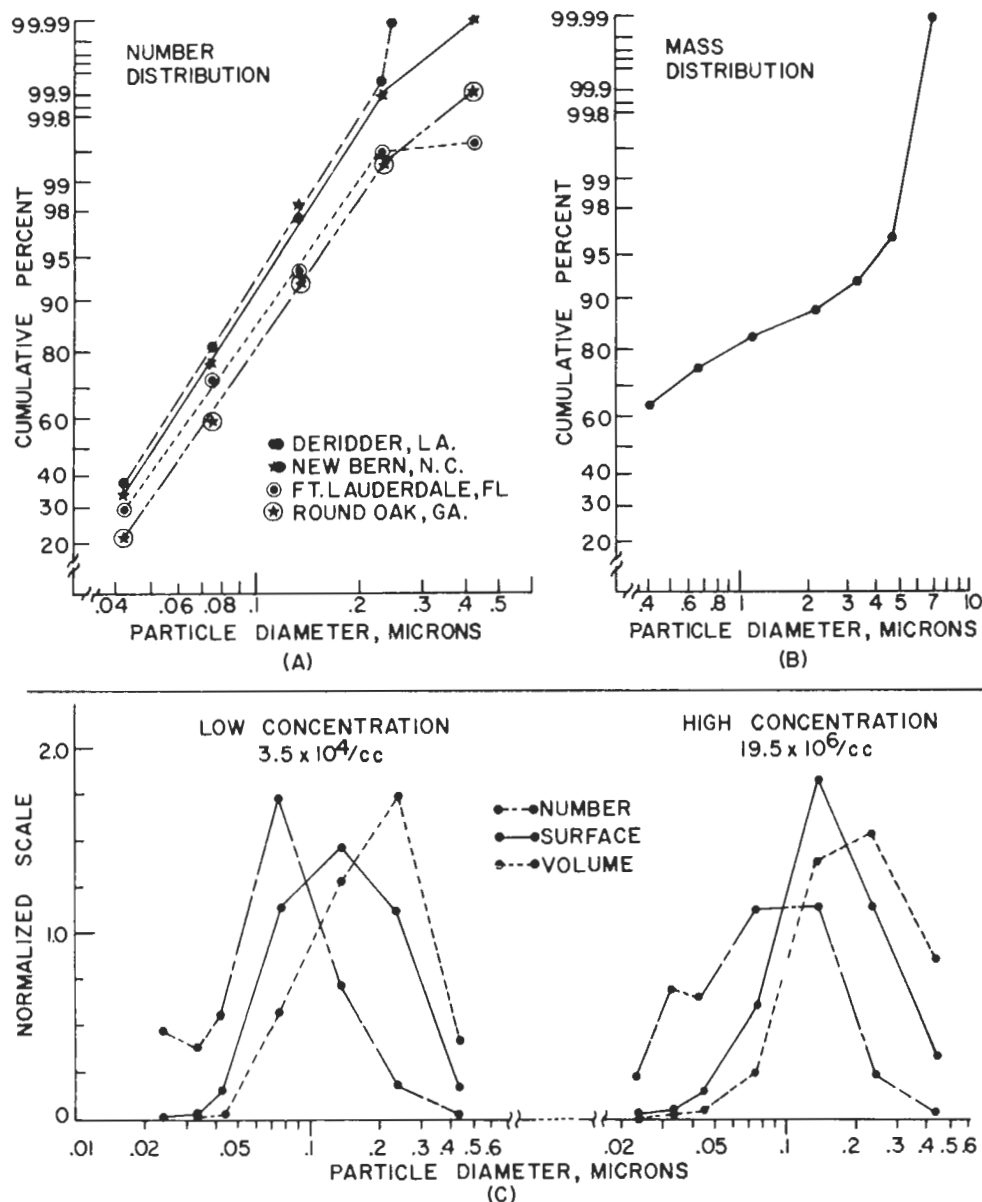


Figure 2--Particle size distribution: (A) Single fire in four fuel types; (B) Grand Average, all fuel types; and (C) Normalized distribution, number, surface area, and volume for a high and low concentration (from Ryan and McMahon 1976).

Some hydrocarbons and NO_x are also emitted from transportation systems and private vehicles in recreational and second-home developments.

Fugitive dust from unpaved roads and eolian soils occasionally contribute substantially to the particulate location of pollutants, but this has not been completely quantified (Singer 1980).

EFFECTS OF SMOKE ON FORESTS

Few studies exist that clearly define the effects of smoke and forest biota. Effects of smoke on micro-organisms suggest that smoke reduced growth of spore germination of several fungal pathogens, but increased spore germination on one

fungi (Parmeter and Uhrenholt 1975a, b). Effects of smoke on photosynthesis at low dosages reduces the photosynthetic rate by direct blockage of solar radiation. Increased CO_2 concentrations, however, could increase carbon fixation and photosynthetic activity (Green and Wright 1977).

Effects of smoke on social values, primarily visibility, are not clearly defined. Although physical aspects of visibility; that is, visual range, maximum distance an object can be seen, and discrimination of details on a distant object can be defined quantitatively (Malm 1979), perceived psychological benefits (Driver and others 1979) of visibility are interrelated with other demands on the sensory system. Paraphrasing Driver and others (1979) a little noise pollution, a little light

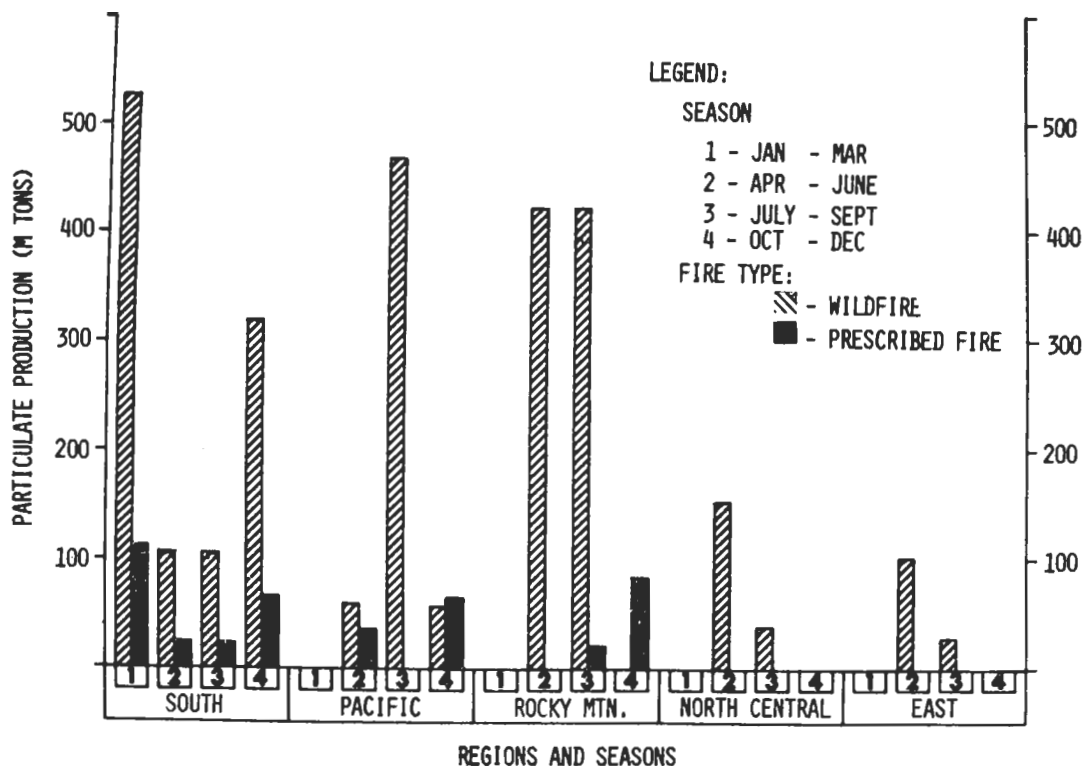


Figure 3--Forest fire particulate production by region and season (from Ward and others 1976).

pollution, a little loss of open space, awareness of water pollution, nonbiodegradable substances, aerosol cans and cancer, change expected values of visibility. Referring to the quote from a popular song used in the introduction of this paper, we can ask, did Joe's attitudes toward visibility change on his visit to Colorado, and how would those changes influence his view of the New York skyline? The point Driver and others (1979) are emphasizing is that the perceived values are the real values and that the physically measurable values of visibility are indices of the values. In particular, scattering and attenuation of light is not a social value.

DISPERSION PROCESSES IN COMPLEX TERRAIN

Wind patterns in complex terrain are highly variable in time and space. Local mountain and valley circulations frequently mask the large-scale patterns such that a mean transport wind for pollutant movement is difficult to define (Fosberg and others 1976a, b; Fosberg and Fox 1978). Spatial variability of winds is clearly illustrated in figure 4 over the Oregon Coast Range and over the Cascade Mountains. Contrast this variability with the uniformity of winds over the Pacific Ocean and within the Willamette Valley.

In addition to mean transport of pollutants, turbulent diffusion is important in dispersing air

pollutants. The most frequently used method of quantifying the dispersion process is through the so-called Gaussian dispersion model (Turner 1969). Downwind concentrations, χ , are related to emission Q by

$$\chi = \frac{Q G}{u} \quad (1)$$

where u is the mean windspeed and

$$G = \frac{1}{2\pi\sigma_y\sigma_z} \exp\left[-\frac{1}{2}\left(\frac{y}{\sigma_y}\right)^2 - \frac{1}{2}\left(\frac{z}{\sigma_z}\right)^2\right] \quad (2)$$

Here, σ_y and σ_z are the variances in the Gaussian statistical distribution; y and z are the distance of the pollutant element from the plume centerline. The variance σ is related to the turbulence structure through

$$\sigma^2 = \frac{2Kx}{u} \quad (3)$$

in which K is the eddy turbulence coefficient and x is the distance downwind from the source. Traditional interpretations of atmospheric processes that were developed for level ground suggest that the K 's be defined through analysis of the inertial subrange of turbulence and that this mean wind is constant over substantial distances, this is, 10's of km. Such assumptions are extremely difficult to satisfy in complex terrain.

Table 2--Summary of prescribed fire acres burned and tons of criteria pollutant emitted (by geographic region, annual basis) (from Sandberg and others 1979)

States by Region	Area hectares (10 ³)	Fuel consumed		Particulates 8 kg/metric tons	Carbon monoxide 10 kg/metric tons	Hydrocarbons 5 kg/metric tons
		Metric tons/ha	Metric tons (10 ³)			
California	5	157	768	6,529	7,682	3,841
Oregon	37	74	2,740	23,290	27,400	13,700
Washington	45	76	3,457	29,380	34,566	17,283
Total	87		6,966	59,209	69,658	34,829
ROCKY MTN.						
Arizona	11	18	189	1,604	1,887	943
Colorado	1	18	19	161	190	95
Idaho	11	146	1,645	13,985	16,453	8,226
Montana	19	101	1,918	16,305	19,183	9,592
New Mexico	3	18	52	439	517	258
North Dakota	1	4	4	31	36	18
Total	46		3,826	32,519	38,257	19,129
N. CENTRAL						
Michigan	1	7	8	69	82	41
Minnesota	2	7	16	139	163	82
Wisconsin	3	7	18	154	181	91
Total	6		44	365	435	218
EASTERN						
Delaware	0	4	2	15	18	9
New Jersey	8	7	54	463	544	272
Total	8		56	478	562	281
SOUTHERN						
Alabama	85	7	568	4,826	5,678	2,839
Arkansas	22	7	149	1,264	1,487	744
Florida	291	9	2,609	22,173	26,085	13,043
Georgia	293	7	1,972	16,760	19,718	9,859
Louisiana	89	7	600	5,104	6,004	3,002
Mississippi	68	7	460	3,908	4,598	2,299
N. Carolina	47	7	319	2,714	3,192	1,596
S. Carolina	157	7	1,058	8,989	10,576	5,288
Texas	34	7	227	1,927	2,268	1,134
Virginia	13	11	144	1,225	1,442	721
Total	1,100		8,100	68,887	80,995	40,498
USA Total	1,197		18,991	161,422	189,908	94,954

An alternative method of defining the variances for complex terrain is mathematically identical but does not require that the turbulence lie in a particular portion of the energy spectrum (Fosberg and others 1976b, Fosberg and Fox 1978). In particular, a mean wind is defined statistically over the dispersion distance of interest. Deviation of wind about this mean, whether in the turbulent inertial subrange or produced by organized flows of scale smaller than the averaging distances, are treated mathematically as components of K . The deviations about the statistically defined mean wind are u' . The K 's are then defined by

$$K = \overline{u'^2 \tau} \quad (4)$$

in which the line over the square of the deviations is the averaging operator. The time constant τ is related to the averaging time and space. These K 's do not represent turbulence. Instead, the K 's

define the wind variability at scales smaller than those used to define the mean wind.

MANAGEMENT OF AIR RESOURCES

Emission control is required to meet air quality objectives (U.S. Congress 1977). Because direct limits on emission from open burning can be achieved only by limiting the mass of fuel burned at any given time, a model for air resource allocation was developed. The Air Resource Allocation Model (ARAM) is based on the Gaussian dispersion model defined in equation 1. Because the intent is to limit emissions, equation 1 is rearranged to

$$Q = \frac{X u}{G} \quad (5)$$

The concentration X is interpreted here as the increment of air quality available for prescribed

Table 3--Change in annual burn (in hectares by pollutant)

Basin	C.O.	T.S.P.	H.C.
<u>Monterey Ranger District</u>			
1. Little Sur	5,260	9	757
2. Big Sur	5,260	9	757
3. Carmel	6,290	10	909
4. Arroyo Seco	12,700	21	1,840
5. Ocean Front	13,500	23	1,960
6. San Antonio	11,250	19	1,640
7. Nacimiento	8,590	13	1,210
<u>Santa Lucia Ranger District</u>			
8. Salinas (A)	5,110	85	3,170
9. Salinas (B)	1,080	10	365
10. Lopez Canyon	1,080	10	363
11. Cuyama (A)	43,000	297	5,660
12. Sisquoc	193,000	298	25,400
<u>Mount Pinos Ranger District</u>			
13. Cuyama (B)	37,900	297	4,980
14. San Joaquin Valley	22,800	189	3,010
15. Piru	24,300	201	3,190
<u>Ojai Ranger District</u>			
16. Sespe	20,500	170	2,700
17. Santa Paula	6,200	-17	no data
18. Ventura	25,500	-70	no data
<u>Santa Barbara Ranger District</u>			
19. Santa Ynez	27,500	190	3,620
20. Santa Barbara Front	17,300	-29	-7,760

burning. ARAM is based on earlier development on emission limits for single sources, single receptor relations defined in the TAPAS model (Fosberg and Fox 1976, Fox and Fosberg 1976). ARAM differs from TAPAS in that ARAM considers multiple sources and multiple receptor sites and contains improvements in characterization of the dispersion processes. Relations between multiple sources and multiple receptor sites are defined through matrix algebra as

$$x_1 = \frac{Q_1 G_{11}}{u} + \frac{Q_2 G_{21}}{u} + \dots \quad (6a)$$

$$x_2 = \frac{Q_1 G_{12}}{u} + \frac{Q_2 G_{22}}{u} + \dots \quad (6b)$$

Here x_1 , x_2 , and so on are the increments of pollution allowed at receptor sites 1, 2, and so on; Q_1 , Q_2 ,... are the allowable emissions at source sites 1, 2,... and G_{11} is the dispersion relation between source 1 and receptor 1; G_{21} is the dispersion relative between source 2 and receptor 1, and so on. Expressing equation 6 in matrix format

$$|x_i| = \frac{|Q_j| |G_{ji}|}{u} \quad (7)$$

The generalized form of ARAM expressed in equation 5 is this

$$|Q_j| = |x_i| |G_{ji}|^{-1} u$$

ARAM has been applied on one National Forest in California, the Los Padres, in support of use of prescribed fire in vegetation management. Current air quality regulations in California recognize that prescribed fire is an alternative to wildfire for vegetation management. In particular, the regulatory agencies accept the concept that a ton of fuel burned in prescribed fire can be used to offset a ton of fuel burned in wildfire.

In the following example, the increment for prescribed burning is defined as the incremental departure from the existing emission from prescribed fire and wildfires. This approach circumvents the difficulties associated with development of a complete regionwide emission inventory base. The assumption here is that the existing emission from prescribed fire and wildfire are defined within the State Implementation Plan. The following calculations then represent an analysis of

where prescribed burning can be increased and where burning must be reduced. The air quality database is the California Air Resources Board (1977) Three-Year Summary of Air Quality. Nearly all the air quality monitoring stations are in urban areas and, therefore, do not necessarily reflect conditions in the wildlands. The following calculations are conservative estimates because of the bias in the database. The meteorological database is from the National Fire Weather Library (Furman and Brink 1975). This database is the only readily accessible database for wildlands. Plume rise calculations were made through the equations developed by Craig and Wolf (1980) for prescribed burning. Three criteria pollutants were evaluated. These pollutants were particulates, hydrocarbons, and carbon monoxide.

The physical setting of the Los Padres National Forest is along the California Coast extending from near Monterey in the north, around Point Conception, to near Santa Barbara in the California

Bight. The Los Padres National Forest lies within the California Coast Range and, therefore, can be broken up into a series of small airsheds. In particular, 20 airsheds were defined. Several Class I wilderness areas are included. Emissions in each of the airsheds were converted to hectares through a fuels inventory of tons of fuel per hectare and the emission characteristics defined in the second section of this paper. Calculated changes in combined emissions of prescribed and wildfires are defined by this least acres for increase or largest negative numbers for decrease from current emissions. As an example, all airsheds on the Monterey District are limited by particulate pollutants (table 3). Most airsheds on the Los Padres National Forest could sustain minor increases in prescribed burning. Only three airsheds show a need to decrease the combined prescribed fire, wildfire emissions. All three airsheds are heavily populated, and the Forest must compete with numerous other pollutant sources for the air resource.

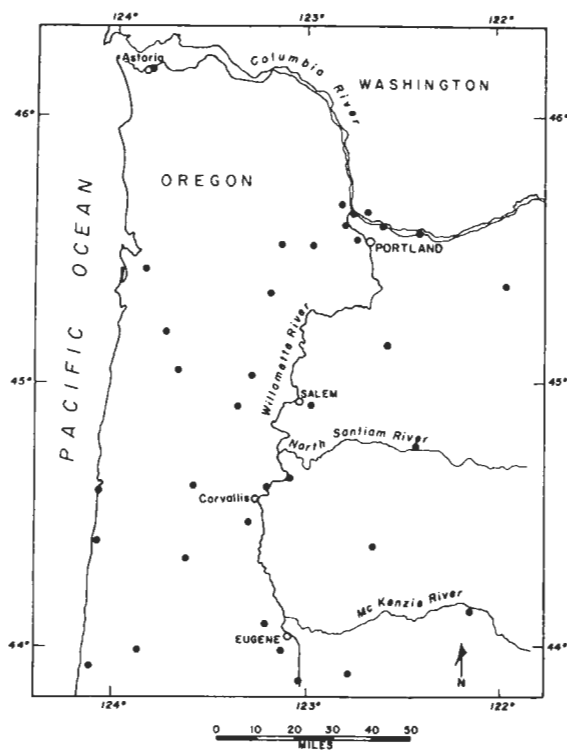


Figure 4a--Location of weather stations in northwest Oregon used to calculate wind patterns shown in figure 4b.

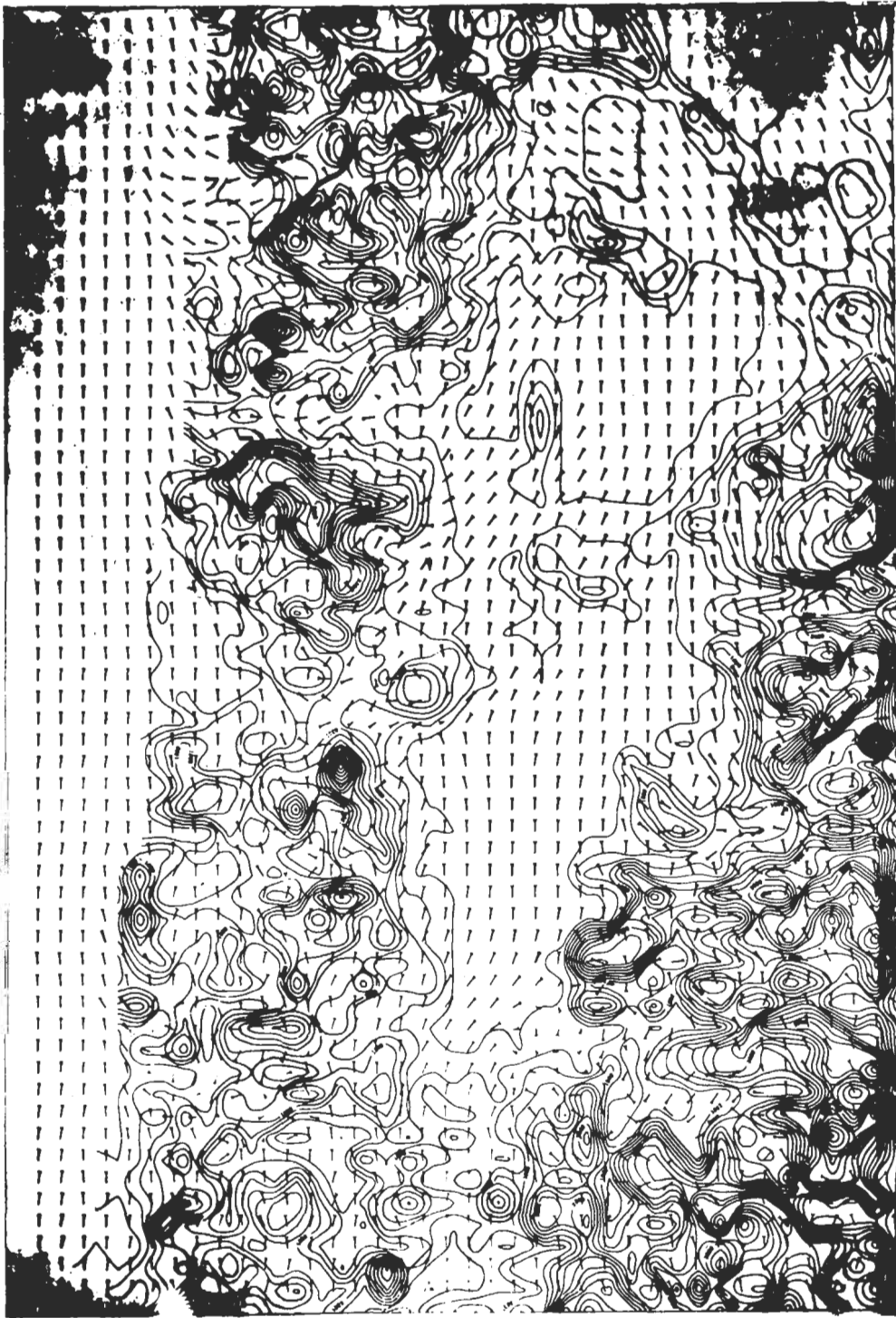


Figure 4b--Calculated wind patterns. Mesh length is 4 km by 4 km. Wind-speed is proportional to length of arrows. Note the uniformity of wind direction and speed over the Pacific Ocean and within the Willamette Valley. Winds in complex terrain, the Coast Range, and the Cascade Range show a high degree of speed and direction variability.

LITERATURE CITED

- Boubel, R. W., E. F. Darley, and E. A. Schuck.
1969. Emissions from burning grass stubble and straw. J. Air Pollut. Control Assoc. 19:497-520.
- California Air Resources Board.
1977. Three year summary of California air quality data. 346 p.
- Craig, C. D., and M. A. Wolf.
1980. Factors influencing particulate concentrations resulting from open field burning. Atmos. Environ. 14:433-443.
- Darley, E. F., F. R. Burleson, E. H. Mateer, J. T. Middleton, and V. Posterli.
1966. Contributions of burning agricultural waste to photochemical air pollution. J. Air Pollut. Control Assoc. 16:685-690.
- Driver, B. L., Donald Rosenthal, and Lynn Johnson.
1979. A suggested research approach for quantifying the psychological benefits of air visibility. Proc. Workshop in Visibility Values. USDA Forest Serv. Gen. Tech. Rep. 18, p. 100-105.
- Feldstein, M., S. Duckworth, H. C. Wohlers, and B. Lusky.
1963. The contributions of the open burning of land clearing debris to air pollution. J. Air Pollut. Control Assoc. 13:542-545, 564.
- Fosberg, Michael A., and Douglas G. Fox.
1976. An air quality index to aid in determining mountain land use planning. In Proc. 4th National Conf. on Fire and Forest Meteorology. USDA Forest Serv. Gen. Tech. Rep. RM-32, p. 167-170.
- Fosberg, Michael A., W. E. Marlatt, and Lawrence Krupnak.
1976a. Estimating airflow patterns over complex terrain. Res. Paper RM-162, 16 p. Rocky Mountain Forest and Range Exp. Stn., Forest Serv., U.S. Dep. Agric., Fort Collins, Colo.
- Fosberg, Michael A., Douglas G. Fox, E. A. Howard, and Jack Cohen.
1976b. Non-turbulent dispersion processes in complex terrain. Atmos. Environ. 10:1053-1055.
- Fosberg, Michael A., and Douglas G. Fox.
1978. Reply: non-turbulent dispersion processes in complex terrain. Atmos. Environ. 12:976.
- Fosberg, Michael A., Charles D. Craig, and Marshall P. Waters, III.
1980. Application of infrared data from a geosynchronous meteorological satellite in surface wind modeling. Proc. 6th Conf. on Fire and Forest Meteorology. Soc. Amer. For. p. 265-275.
- Fox, Douglas G., and Michael A. Fosberg.
1976. Estimating regional air pollution impact. In Proc. 4th Int. Clean Air Congress. Int. Union of Air Pollut. Control Assoc. p. 229-301.
- Fritschen, Leo, Harley Bovee, Konrad Buettner, Robert Charlson, Lee Monteith, Stewart Pickford, James Murphy, and Ellis Darley.
1970. Slash fire atmospheric pollution. Res. Paper PNW-97, 42 p. Pacific Northwest Forest and Range Exp. Stn., U.S. Dep. Agric., Portland, Oreg.
- Furman, R. William, and Glen F. Brink.
1975. The fire weather library: what it is and how to use it. Gen. Tech. Rep. RM-19, 8 p. Rocky Mountain Forest and Range Exp. Stn., Forest Serv., U.S. Dep. Agric., Fort Collins, Colo.
- Gerstle, R. W., and D. A. Kemnitz.
1967. Atmospheric emissions from open burning. J. Air Pollut. Control Assoc. 17:324-327.
- Green, Kathleen, and Robert Wright.
1977. Field response of photosynthesis to CO₂ enhancement in ponderosa pine. Ecology 58: 687-692.
- Malm, William.
1979. Visibility: a physical perspective. Proc. Workshop in Visibility Values. USDA Forest Serv. Gen. Tech. Rep. 18, p. 56-68.
- McCall, C. W., Bill Fries, and Chip Davis.
1976. Aurora Borealis on record album Wilderness, PD-1-6069. Polydor Inc., New York.
- Parmeter, J. R., and B. Uhrenholt.
1975a. Effect of smoke on pathogens and other fungi. Proc. Tall Timbers Fire Ecology Conf. No. 14:299-304.
- Parmeter, J. R., and B. Uhrenholt.
1975b. Some effects of pine needles or grass smoke on fungi. Phytopathol. 65:28-31.
- Radke, L. F., J. L. Smith, D. A. Hess, and P. V. Hobbs.
1978. Airborne studies of particulate and gases from forest fires. J. Air Pollut. Control Assoc. 28:30-34.
- Ryan, P. W.
1974. The quantity and quality of smoke produced by southern fuels in prescribed burning operations. Bull. Amer. Meteorol. Soc. 55:70.
- Ryan, P. W., and Charles K. McMahon.
1976. Some chemical and physical characteristics of emissions from forest fires. Paper No. 76-2.3 presented at the 69th Annual Meeting of the Air Pollution Control Association [June 22-July 1, 1976, Portland, Oregon], 15 p.

Hydrocarbon Emissions from Vegetation¹

David T. Tingey
Walter F. Burns²

Abstract: A wide range of volatile organic compounds may be emitted by vegetation. The identified emittants, however, are mainly terpenoid in nature. Their emission rates are controlled primarily by the physical/chemical processes that regulate hydrocarbon vapor pressure. Emission rates vary between species and are influenced by environmental factors such as light and temperature. Regional emission estimates indicate that vegetation may emit as much as 30 kg of hydrocarbons km⁻² day⁻¹. The measured atmospheric concentrations are in reasonable agreement with the estimated emission rates. Within the atmosphere, these hydrocarbons may participate in photochemical reactions leading to aerosol production and the consumption or formation of ozone.

High levels of ozone have been measured in rural and remote locations far from significant anthropogenic sources of oxidant precursors. These elevated concentrations may have resulted from long distance transport and/or the photo-oxidation of locally-produced biogenic hydrocarbons. Robinson (1978) proposed that ambient hydrocarbon concentrations were governed by both long distance transport and local production. Volatile organics, including monoterpenes and isoprene, have been detected in the atmosphere (Rasmussen and Went 1965; Schjoldager and Watine 1978; Whitby and Coffey 1977; Arnts and Meeks 1980; Lonneman and others 1977) and in laboratory studies shown to produce ozone (Arnts and Gay 1979), suggesting that they may contribute to ambient ozone concentrations.

Plants contain a number of potentially volatile organic compounds including monoterpenes, isoprene, aldehydes, alcohols, and ketones (Meigh

1955; Rasmussen 1972; Zimmerman 1979a). Individual species have relatively distinctive emission profiles. For some species, only one or a few compounds dominate the emission profile; however, other species have a diffuse emission profile with no dominant compounds (Rasmussen 1972; Zimmerman 1979a). Despite the wide range of potentially volatile compounds, only isoprene, monoterpenes, and a few aromatics have been conclusively identified as emission products from vegetation (Rasmussen 1972; Zimmerman 1979a), hence they form the basis for further discussion.

METHODS FOR ESTIMATING EMISSION RATES

A variety of experimental methods have been used to estimate emission rates. A tree branch or a few small plants were enclosed in a large Teflon bag to estimate biogenic hydrocarbon emission rates in the field (Zimmerman 1979a, 1979b). The bag was sealed, evacuated and refilled with hydrocarbon-free air. A small gas-exchange rate was maintained through the bag. After an accumulation period, the head space was sampled to determine the gas phase concentration. Vertical gradients of temperature, water, and α -pinene, both within and above the canopy of a loblolly pine (*Pinus taeda* L.) plantation were measured and used to calculate emission rates (Arnts and others 1978).

¹ Presented at the Symposium on Effects of Air Pollutants on Mediterranean and Temperate Forest Ecosystems, June 22-27, 1980, Riverside, California, U.S.A.

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- Sandberg, D. V.
1974. Measurement of particulate emissions from forest residues in open burning experiments. Ph. D. thesis, Univ. of Washington. 165 p.
- Sandberg, D. V., and R. E. Martin.
1975. Particle sizes in slash fire smoke. Res. Paper PNW-199, 7 p. Pacific Northwest Forest and Range Exp. Stn., Forest Serv., U.S. Dep. Agric., Portland, Oreg.
- Sandberg, D. V., J. M. Pierovich, D. G. Fox, and E. W. Ross.
1979. Effects of fire on air. Gen. Tech. Rep. WO-9, 40 p. Forest Serv., U.S. Dep. Agric., Washington, D.C.
- Schaefer, Vincent J.
1976. The production of optimum particle smokes in forest fires. Proc. Air Quality and Smoke from Urban and Forest Fires. National Academy of Science-National Research Council. p. 27-29.
- Singer, Michael J.
1980. Climate variables in soil erosion processes. Proc. of the National Weather Service Agricultural Meteorology Training Conf., Univ. of Calif., Davis. Atmos. Sci. Paper 19, p. 75-87.
- Turner, D. Bruce.
1969. Workbook of atmospheric dispersion estimates. U.S. Dep. Health, Education, and Welfare Public Health Service Publ. N. 999-AP-26. 84 p.
- U.S. Congress.
1977. Clean Air Act Amendment of 1977. 91 Stat. 685, 42 U.S.C. 7401 of seq.
- U.S. Environmental Protection Agency.
1978. Compilation of air pollutant emission factors, 3d ed., Supplemental No. 8. AP 42.
- Vines, Robert G., L. Gibson, A. B. Hatch, M. K. King, D. A. MacArthur, D. R. Packhan, and R. J. Taylor.
1971. On the nature, properties, and behavior of brush-fire smoke. CSIRO, Div. of Appl. Chem. Tech. Paper 1. 32 p.
- Ward, D. E., C. K. McMahon and R. W. Johansen.
1976. An update on particulate emissions from forest fires. Paper No. 62-2.2. 6th Annual Meeting of Air Pollut. Control Assoc. 14 p.

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Semi-quantitative estimates of emission rates were made using static gas-exchange chambers containing either detached leaves, twigs or whole plants (Rasmussen 1970; Rasmussen 1972; Sanadze and Kalandadze 1966a). Within these chambers carbon dioxide may be elevated or depleted depending on light intensity. This may modify plant metabolism and the stomatal aperture; humidity will increase; and high concentrations of hydrocarbon gases will build up within the chamber reducing diffusion gradients. These factors can lead to an underestimation of emission rates. Dynamic mass-balance gas-exchange chambers and leaf cuvettes which simulate the gaseous environments of plants in the field have also been used to estimate hydrocarbon emissions (Kamiyama and others 1978; Tingey and others 1979 and 1980; Tyson and others 1974). These chambers may be used to determine the influence of environmental factors on emission rates.

MECHANISM OF HYDROCARBON VOLATILIZATION

Terpenoid Biosynthesis

Knowledge of the mechanism and sites of terpenoid biosynthesis aids in understanding the factors controlling emission rates. Terpenoid biosynthesis starts with the conversion of carbon dioxide to sucrose with its subsequent metabolism to acetyl-CoA and mevalonic acid to form isopentenyl pyrophosphate. The hemiterpene, isoprene, (C_5) is formed from isopentenyl pyrophosphate; monoterpenes (C_{10}) are formed from a condensation of dimethylallyl pyrophosphate and isopentenyl pyrophosphate. Subsequent additions of isopentenyl pyrophosphate units form higher homologs in the terpenoid series (Loomis and Croteau 1980).

The conditions that promote isoprene biosynthesis indicate that it is likely synthesized within the chloroplast. Isoprene biosynthesis is affected by metabolic inhibitors that regulate photorespiration (Loomis and Croteau 1980).

Monoterpenes appear to be ubiquitous in higher plants (Loomis and Croteau 1980). The accumulation or secretion of significant quantities of monoterpenes is associated with the presence of secretory structures such as glandular hairs or trichomes, oil cells, resin ducts or glandular epidermises, and lenticles. It is generally assumed that monoterpenes are synthesized within the secretory cells, although this point has not yet been conclusively demonstrated (Loomis and Croteau 1980).

Hydrocarbon Diffusion from Plants

Gaseous diffusion between the plant and its environment is controlled by the chemical potential gradient between the inside and the outside of the leaf and the resistance to mass transfer along the diffusion pathway. The chemical potential gradient can be approximated by a concentra-

tion gradient. The larger the concentration gradient, the larger the hydrocarbon flux; conversely, the larger the resistance to mass transfer, the smaller the flux (Nobel 1974).

Only hydrocarbons with appreciable vapor pressures at ambient temperatures will be emitted at significant rates. The vapor phase concentration of hydrocarbons within the leaf is controlled by the liquid phase concentration, vapor pressure, and solubility. The vapor pressure of terpenoid compounds increases exponentially with the temperature (Jordan 1954). Monoterpene emission rates from black sage (*Salvia mellifera* Greene) and slash pine (*Pinus elliottii* Engelm.) also exhibit an exponential increase with temperature (Dement and others 1975; Tingey and others 1980), indicating that vapor pressure is a significant factor in controlling emissions. Emission rates from dead slash pine needles and black sage leaves are similar to emission rates from live tissue (Tingey and others 1980; Dement and others 1975), supporting the concept that the volatilization is primarily a physical process. Hydrocarbons with chain lengths greater than C_{10} generally have low vapor pressures and will not have a large emission rate.

When monoterpenes occur in high concentrations in resin ducts, oil cells or glandular trichomes, their emission rates are essentially independent of concentration. Large pools of isoprene, however, have not been detected. Below 35°C (the boiling point for isoprene), the emission rate is closely linked to its synthesis rate. Above 35°C, the emission rate is diffusion-limited (Tingey and others 1979).

Monoterpenes have a low aqueous solubility (hydrophobic) and higher vapor pressures than similar, more hydrophilic compounds. Therefore, monoterpenes would be emitted at a higher rate than similar oxygenated compounds at equal concentrations within the tissue. Similarly, if the concentration exceeds its aqueous solubility limit, then vapor pressure and emissions are independent of tissue concentrations.

Resistance to mass transfer can occur along either a stomatal or cuticular pathway (Nobel 1974). Either one or both pathways may be significant, depending on the species. Stomata are apparently the main pathway for diffusion of monoterpenes (Hanover 1972), isoprene and other compounds synthesized within the leaves. However, for plants with glandular trichomes or glandular cells in the epidermis, such as in the Labiatae and Solanaceae, the cuticular pathway is the main one for diffusion.

BIOGENIC EMISSIONS

Emission Rates

Emission rates for several plant species are shown in table 1. Total non-methane hydrocarbon and monoterpene emission rates are similar among

Table 1--Biogenic hydrocarbon emission rates estimated at 30°C.

Species	TNMHC ¹	Isoprene	Monoterpenes	References
	$\mu\text{g (g dry weight)}^{-1} \text{ hr}^{-1}$			
Slash Pine	4.1		2.6	Zimmerman 1979a
Longleaf Pine	7.3		5.6	Zimmerman 1979a
Sand Pine	13.6		11.0	Zimmerman 1979a
Cypress	14.2		8.1	Zimmerman 1979a
Slash Pine			6.4	Tingey and others 1980
Loblolly Pine			3.7	Arnts and others 1978
Cryptomeria			3.0	Kamiyama and others 1978
Laurel Oak	12.6	10.0		Zimmerman 1979a
Turkey Oak	26.5	23.4		Zimmerman 1979a
Bluejack Oak	56.4	43.9		Zimmerman 1979a
Live Oak	10.8	9.1		Zimmerman 1979a
Live Oak			41.2	Tingey and others 1980
Willow	22.1	12.4		Zimmerman 1979a
Saw Palmetto	11.5	8.6		Zimmerman 1979a
Mean 7 Hardwood Trees--Isoprene	20.0	15.7		Flyckt and others 1980
Wax Myrtle	7.5			Zimmerman 1979a
Persimmon	2.9			Zimmerman 1979a
Orange	9.4			Zimmerman 1979a
Grapefruit	4.3			Zimmerman 1979a
Red Maple	6.5			Zimmerman, 1979a
Hickory	3.2			Zimmerman, 1979a
Mean 10 Hardwood Trees--Non-Isoprene	7.3			Flyckt and others, 1980

¹ Total non-methane hydrocarbons.

the conifers and as much as 50 percent less than emission rates from hardwoods that emit isoprene. Monoterpenes account for 50 to 75 percent of the total non-methane hydrocarbon emissions in conifers. Similarly, isoprene accounts for 60 to 90 percent of total non-methane hydrocarbon emissions from isoprene emitters. Plants whose emissions were not dominated by either isoprene or a few monoterpenes had emission rates roughly similar to the conifers.

Total non-methane emission rates were estimated at several locations in the United States on similar vegetation types:

Vegetation Type and Location:	Emission Rate ¹ ($\mu\text{g (g dry weight)}^{-1} \text{ hr}^{-1}$)
Conifers	
Washington	5.0
Florida	8.8
Texas	5.0
Oaks	
California	28.7
Florida	21.9
Texas	26.0
Non-Conifers, Non-Isoprene Emitter	
Washington	7.8
California	4.1
Florida	4.7
Texas	0.2

¹ Data from Zimmerman 1979b and c.

Emission rates for the conifers, oaks and non-conifer, non-isoprene emitting vegetation are similar within each vegetation type. This indicates an apparent high uniformity in emission rates among locations using the same estimation technique.

Hydrocarbon emission rates/unit tissue multiplied by biomass density yield emission factors. Emission factors for the Tampa-St. Petersburg, Florida, area (Zimmerman 1979a), indicate that 92 percent of the total non-methane hydrocarbon emissions occur in the following four land use types: evergreen forests (35 percent); citrus groves (22 percent); pasture and rangeland (19 percent); and residential areas (16 percent). The remaining 8 percent was distributed among crop lands, deciduous forests, mangroves, freshwater, marine, and barren lands. Emission factors for trees were approximately $6 \text{ mg m}^{-2} \text{ hr}^{-1}$; shrubs, $2.0 \text{ mg m}^{-2} \text{ hr}^{-1}$; pastures, mud flats and other land use types were less than $0.1 \text{ mg m}^{-2} \text{ hr}^{-1}$.

Emission factors were developed to characterize the various biomes in the United States:

Emission Rate ($\text{mg m}^{-2} \text{ hr}^{-1}$)

Biome:	Day	Night
Grassland	0.3	0.2
Sclerophyll Scrub	2.4	1.4
Temperate Rain Forest	10.7	9.4
Deciduous Forest	6.6	2.7
Coniferous Forest	6.5	2.7
Desert	1.3	0.7
Tundra, Alpine Fields	0.5	0.4

¹ Data from Zimmerman 1979b.

Daytime biome emission factors ranged from a low of $0.3 \text{ mg m}^{-2} \text{ hr}^{-1}$ for grasslands to a high of 10.7 for temperate rain forests. Nighttime emissions were 10 to 60 percent lower reflecting, in part, the absence of isoprene emissions.

Environmental Influences on Emission Rates

Isoprene production is light dependent, and persists for only a few minutes when plants are darkened (Rasmussen and Jones 1973; Sanadze and Kalanadze 1966b). Emissions increase with increasing light intensity until a maximum is reached and then remain constant (Sanadze and Kalanadze 1966a; Tingey and others 1979); similar to a light saturation curve for photosynthesis. Isoprene emissions are light saturated at moderate light intensities (Sanadze and Kalanadze 1966a; Tingey and others 1979). In contrast, monoterpene emissions from slash pine, black sage, and several other plant species, are similar in the dark and light (Tingey and others 1980; Dement and others 1975; Rasmussen 1972).

Isoprene emissions increase sigmoidally with temperature; low emissions occur at 18-20°C and increase exponentially between approximately 20 and 35°C, then plateau. At higher temperatures (between 43 and 47°C), depending upon the species, there is the large, precipitous decline in isoprene emissions (Sanadze and Kalanadze 1966a; Rasmussen and Jones 1973; Tingey and others 1980).

The increase in isoprene emissions with temperature is greater at high light intensities than low (Tingey and others 1979). Isoprene emissions from several hardwood trees and live oak (*Quercus virginiana* Mill.) increased at approximately 20 and 16 percent/°C (20-35°C), respectively (Flyckt and others 1980; Tingey and others 1979).

Monoterpene emissions from conifers, black sage and hardwood trees increase exponentially with the temperature (Arnts and others 1978; Kamiyama and others 1978; Rasmussen 1972; Flyckt and others 1980; Dement and others 1975; Tingey and others 1980). The relative percent increase per degree temperature varies between species and ranges from approximately 6 to 20 percent/°C. In conifers, extensive genetic variations in monoterpene pools (Hanover 1972) may explain the lack of an exponential relationship between temperature and emission rates in some field studies (Flyckt and others 1980).

Typical diurnal emission patterns for isoprene and monoterpenes and environmental conditions for an average of summer days in Tampa, Florida, were used to illustrate the interaction of light and temperature on terpenoid emissions (Tingey and others 1979, 1980). During early morning and late afternoon, when the leaves are not light-saturated and the temperature is moderate, light would be the main factor controlling isoprene emissions from live oak. However, during most of the day, the leaves of the canopy are light-saturated; thus varying air temperature would control emission rates. More than 80 percent of the isoprene emissions were expected to occur after mid-morning, ceasing in the evening. Monoterpene emission rates from slash pine increase after sunrise, peaking during early afternoon, and declining to a minimum shortly before sunrise. Approximately 55 percent of the total daily monoterpene emissions occurred during daylight hours (0600-1800) with an additional 25 percent emitted between sunset (1800) and midnight (2400).

Seasonal emission patterns were estimated for individual ponderosa pine and red oak trees (Flyckt 1979). Monoterpene emissions from ponderosa pine were sinusoidal, at a maximum during late spring (May and June), declining to a minimum around November, and then gradually increasing. In contrast, isoprene emissions from red oak were maximum during July and August and decreased during the fall. No isoprene emissions were detected during the winter; emissions reappeared in the spring with the initiation of new leaves. It is not clear whether seasonal emission changes were due solely to changes in environmental conditions or were, in part, due to changes in terpenoid pools. In addition to changes in the emission rates, there were also qualitative changes in the monoterpene emissions throughout the year (Flyckt and others 1980).

Table 2--Estimated emissions for biogenic hydrocarbons.

Location	Emissions	Emission Factors	References
	metric tons	kg km ⁻² day ⁻¹	
World	1.75 x 10 ⁸ /year		Went 1960
World	4.38 x 10 ⁸ /year		Rasmussen and Went 1965
World	8.30 x 10 ⁸ /year		Zimmerman 1979b
United States	0.23-4.64 x 10 ⁷ /year		Rasmussen 1972
United States	6.5 x 10 ⁷ /year		Zimmerman 1979b
Florida (81 x 60 km)	157.0/day	32.3	Zimmerman 1979a
Texas (38 x 31 km)	32.4/day	27.5	Zimmerman 1979c
Pennsylvania	3,580.0/day	30.7	Flyckt and others 1980

Regional Emissions

Biogenic hydrocarbon emission rates for a variety of plant species and biomass estimates, were used to estimate emissions for various areas (table 2). The emission rate estimates for various estimation scales (world, United States, or regional) were approximately similar. The close agreement between the emission estimates from the three regional studies may have occurred because the same experimental approach was used. Emission estimates for Pennsylvania and the Tampa-St. Petersburg, Florida, area indicate that biogenic emissions range from 12 percent greater to 20 percent less than anthropogenic emissions (Flyckt and others 1980; Zimmerman 1979a; Wayne and Kochis 1978).

Relationship Between Primary Productivity and Emission Rates

A relationship between biogenic hydrocarbon emissions and primary productivity should exist because they are ultimately derived from photosynthetically fixed carbon dioxide. Measurements of the ratio of carbon lost as volatile terpenoids to primary productivity for several tree species indicated loss rates of 0.2 to 2 percent for isoprene and 0.06 to 0.4 percent for monoterpenes (Sanadze 1969; Tingey and others 1979, 1980; Tyson and others 1974). The relationship between primary productivity and biogenic hydrocarbon emissions could be used to delineate geographic areas where emissions would tend to be high. Based on the work of Leith (1975), primary productivity is highest in the Southeast, followed by the Mississippi Valley area and the central part of the United States and lowest in the Great Basin and the Southwest. Zimmerman (1979b) estimated that 45 percent of the total national biogenic hydrocarbon emissions occurred in the South, an area with the highest primary productivity in the United States (Leith 1975).

AMBIENT CONCENTRATIONS OF TERPENOID

Biogenic hydrocarbons were measured in the atmosphere over several vegetation types. The average isoprene concentrations varied from 10 ppb carbon for an oak forest to 0.1 ppb carbon for a pine forest. The average monoterpene concentrations ranged from 24 ppb carbon in the coniferous

forest in Norway to 2.7 ppb carbon in the coniferous forest in Idaho (Schjoldager and Watine 1978; Arnts and Meeks 1980; Coffey and Westberg 1978). Measured ambient concentrations of biogenic hydrocarbons and ambient concentrations predicted from biogenic emission rates are in reasonable agreement (Zimmerman 1979c; Flyckt and others 1980; Scully, 1979; Coffey and Westberg, 1978).

Atmospheric hydrocarbon concentrations are dependent on emission rates, mixing height, and the reactivities (photolysis and ozonolysis) of the individual components. Peterson and Tingey (1980) used a box model to estimate ambient air concentrations of isoprene and monoterpenes. The predicted isoprene concentrations increased during the daylight hours, reaching a maximum at mid-afternoon and then disappearing during the early evening when isoprene emissions ceased. Predicted ambient monoterpene concentrations were the lowest during mid-day when atmospheric dilution, photo-oxidation, and ozonolysis, were the highest, despite the fact that monoterpene emissions were a maximum. In contrast, monoterpene concentrations were the largest during the evening and early morning hours because monoterpenes are emitted at night when atmospheric mixing is low and hydrocarbon decay reactions are slow, permitting an accumulation of monoterpenes. This predicted monoterpene profile was verified by field measurements (Arnts and Gay 1979).

ATMOSPHERIC FATES OF TERPENOIDS

There are several atmospheric fates for the biogenic hydrocarbons, including conversion to aerosols, carbon monoxide formation, and photochemical reactions, both forming and consuming oxidants. Went (1960) attributed the blue haze found over many coniferous forests to the conversion of terpenes to aerosols. When limonene, a monoterpene, was reacted with NO_x and O_3 , greater than 50 percent of the limonene_x was converted to aerosols within 2.5 hours (Schuetzle and Rasmussen 1978). The aerosols contained both mono- and dimeric alcohols, aldehydes and acid-substituted products.

Zimmerman and others (1978) suggested that atmospheric isoprene and monoterpenes could be oxidized in the atmosphere to CO with a yield of 60-80 percent. Based on biogenic emission estimates, they concluded that natural hydrocarbons may be the largest contributor to the United States carbon monoxide budget.

Nitrogen oxides in the presence of solar radiation form OH radicals (Cleveland and Graedel 1979). Hydroxyl (OH) radicals react with hydrocarbons, forming peroxy radicals, which in turn convert NO to NO_2 , perturbing the photostationary state releasing a free oxygen, forming ozone and other oxygenated products. The exact mechanisms are not clearly delineated because of competing secondary reactions with hydrocarbons, O_3 , NO_x , and OH radical. Arnts and Gay (1979) reported

ozone, PAN, formic acid, acetone, aldehydes, CO, and CO_2 formation when terpenes were irradiated in the presence of nitrogen oxides. The amount of ozone formed depended on the C/ NO_x ratio. At a low C/ NO_x ratio, 1 ppb C from terpenoids produced 2-4 ppb O_3 . However, when the ratio was large, 1 ppb C produced 0.3 to 0.1 ppb ozone, suggesting that terpenoids also consume ozone. Eschenroeder (1974) and Coffey and Westberg (1978) concluded that emissions of biogenic hydrocarbons did not significantly alter ambient ozone concentrations through scavenging reactions.

Zimmerman (1979c) suggested that photooxidation of isoprene from forests could contribute 22 ppb ozone to the ambient concentration. Similarly, Coffey and Westberg (1978) suggested that emissions from coniferous forests could react to add 1 to 5 ppb ozone to the ambient air.

LITERATURE CITED

- Arnts, R.R., R.L. Seila, R.L. Kuntz, F.L. Mowry, K.R. Knoerr, and A.C. Dudgeon.
1978. Measurements of α -pinene fluxes from a loblolly pine forest. In Proc. 4th Joint Conf. on Sensing of Environmental Pollutants. Amer. Chem. Soc. Washington, D.C., p. 829-833.
- Arnts, Robert R., and Bruce W. Gay, Jr.
1979. Photochemistry of some naturally emitted hydrocarbons. EPA-600/3-79-081, U.S. Environmental Protection Agency, Research Triangle Park, North Carolina.
- Arnts, Robert R., and Sarah A. Meeks.
1980. Biogenic hydrocarbon contribution to the ambient air of selected areas: Tulsa, Great Smoky Mountain, Rio Blanco County, Colorado. EPA-600/3-80-023, U.S. Environmental Protection Agency, Research Triangle Park, North Carolina.
- Cleveland, William S., and T.E. Graedel.
1979. Photochemical air pollution in the northeast United States. Science 204(4399):1273-1278.
- Coffey, P. E. and H. Westberg.
1978. The issue of organic emission. International Conference on Oxidants, 1976. Analysis of Evidence and View Points. Part IV. EPA-600/3-77-116, U.S. Environmental Protection Agency, Research Triangle Park, North Carolina.
- Dement, William A., Bennett J. Tyson, and Harold A. Money.
1975. Mechanism of monoterpene volatilization in *Salvia mellifera*. Phytochem. 14(12):2555-2557.

- Eschenroeder, Alan.
1974. Reaction and diffusion of natural hydrocarbons in the atmosphere over forests. General Research Corporation, Santa Barbara, California.
- Flyckt, Donald L.
1979. Seasonal variation in the volatile hydrocarbon emissions from ponderosa pine and red oak. Masters Thesis. Washington State University, Pullman.
- Flyckt, D.L., H.H. Westberg, and M.W. Holdren.
1980. Natural organic emissions and their impact on air quality. Preprint 80-69.2 Annual Meeting Air Pollution Control Association. [June 22-27, 1980, Montreal, Canada].
- Hanover, James W.
1972. Factors affecting the release of volatile chemicals by forest trees. Mitteilungen der forstlichen Bundes-Versuchsanstalt Wien. 97:625-644.
- Jordan, T. Earl.
1954. Vapor pressure of organic compounds. 266 p. Interscience Publishers, Inc., New York.
- Kamiyama, K., T. Takai, and Y. Yamanaka.
1978. Correlation between volatile substances released from plants and meteorological conditions. In Proc. Inter. Clean Air Conf. E.T. White, P. Hetherington, B.R. Thiele, eds. Brisbane, Australia. p. 365-372.
- Lieth, Helmut.
1975. Historical survey of primary productivity research. In Primary productivity of the biosphere. Helmut Lieth and Robert H. Whittaker, eds. p. 7-16. Springer-Verlag, Berlin.
- Lonneman, William A., Robert L. Seila, and Sarah A. Meeks.
1977. Preliminary results of hydrocarbon and other pollutant measurements taken during the 1975 Northeast Oxidant Transport Study. In Proc. Symp. 1975 Northeast Oxidant Transport Study. EPA-600/3-77-017, U.S. Environmental Protection Agency. p. 40-53.
- Loomis, W. David, and Rodney Croteau.
1980. Biochemistry of terpenoids. In The biochemistry of plants. P.K. Stumpf and E.E. Conn, eds. 4:363-415. Academic Press, New York.
- Meigh, D.F.
1955. Volatile alcohols, aldehydes, ketones and ester. In Modern Methods of Plant Analysis. K. Paech and M.V. Tracey, eds. 2:403-443. Springer-Verlag, Berlin.
- Nobel, Park S.
1974. Introduction to Biophysical Plant Physiology. 448 p. W.H. Freeman and Co., San Francisco.
- Peterson, Ernest W., and David T. Tingey.
1980. Contributions of biogenic sources to airborne hydrocarbon concentrations. Atmos. Environ. 14(1):79-81.
- Rasmussen, Reinhold A.
1970. Isoprene: Identified as a forest type emission to the atmosphere. Environ. Sci. Tech. 4(8):667-671.
- Rasmussen, Reinhold A.
1972. What do the hydrocarbons from trees contribute to air pollution? J. Air Poll. Con. Assoc. 22(7):537-543.
- Rasmussen, R.A., and C.A. Jones.
1973. Emission isoprene from leaf discs of *Hamamelis*. Phytochem. 12(1):15-19.
- Rasmussen, Reinhold A., and F.W. Went.
1965. Volatile organic material of plant origin in the atmosphere. Proc. Nat. Acad. Sci. 53(1):215-220.
- Robinson, Elmer.
1978. Hydrocarbons in the atmosphere. Pure Appl. Geophys. 116(2/3):372-382.
- Sanadze, G.A.
1969. Light dependent excretion of molecular isoprene. Progress in Photosynthesis Res. 2:701-706.
- Sanadze, G.A., and A.N. Kalandadze.
1966a. Light and temperature curves of the evolution of C₅H₈. Sov. Plant Physiol. 13(3):411-413.
- Sanadze, G.A., and A.N. Kalandadze.
1966b. Evolution of the diene C₅H₈ by poplar leaves under various conditions of illumination. Dokl. Bot. Sci. 168:95-97.
- Schjoldager, J., and B.M. Watine.
1978. Preliminary study of hydrocarbons in forests. Norwegian Institute for Air Quality Studies, Lillestrom, Norway.
- Schuetzle, Dennis, and Reinhold A. Rasmussen.
1978. The molecular composition of secondary aerosol particles formed from terpenes. J. Air Poll. Con. Assoc. 28(3):236-240.
- Sculley, Robert D.
1979. Correspondence to the editor. Environ. Sci. Tech. 13(3):234-235.
- Tingey, David T., Marybeth Manning, Louis C. Grothaus, and Walter F. Burns.
1979. The influence of light and temperature on isoprene emission rates from live oak. Physiol. Plant 47(2):112-118.

Tingey, David T., Marybeth Manning, Louis C. Grothaus, and Walter F. Burns.

1980. The influence of light and temperature on monoterpene emission rates from slash pine. *Plant Physiology* 65 (in press).

Tyson, Bennett, William A. Dement, and Harold A. Mooney.

1974. Volatilization of terpenes from Salvia mellifera. *Nature* 252(5479):119-120.

Wayne, L.G., and P.C. Kochis.

1978. Assessment of the anthropogenic hydrocarbon and nitrogen oxide emissions in the Tampa Bay area. EPA 904/9-77-016, U.S. Environmental Protection Agency, Region IV. Atlanta, Georgia.

Went, F.W.

1960. Organic matter in the atmosphere, and its possible relation to petroleum formation. *Proc. Nat. Acad. Sci.* 46(2):212-221.

Whitby, Robert A., and Peter E. Coffey.

1977. Measurement of terpenes and other organics in an Adirondack Mountain pine forest. *J. Geophys. Res.* 82(37):5928-5934.

Zimmerman, P.R.

1979a. Determination of emission rates of hydrocarbons from indigenous species of vegetation in the Tampa/St. Petersburg, Florida, area. Appendix C. Tampa Bay Area Photochemical Oxidant Study. EPA-904/9-77-028. U.S. Environmental Protection Agency, Region IV. Atlanta, Georgia.

Zimmerman, Patrick R.

1979b. Testing of hydrocarbon emissions from vegetation, leaf litter and aquatic surfaces, and development of a methodology for compiling biogenic emission inventories. EPA-450/4-79-004. U.S. Environmental Protection Agency, Research Triangle Park, North Carolina.

Zimmerman, Patrick.

1979c. Natural sources of ozone in Houston: natural organics. In *Proc. APCA Houston Specialty Conference on Ozone Oxidants*. Houston, Texas. p. 299.

Zimmerman, Patrick R., Robert B. Chatfield, Jack Fishman, Paul Crutzen, and Phillip L. Hanst.

1978. Estimates of the production of CO and H₂ from the oxidation of hydrocarbon emissions from vegetation. *Geophys. Res. Lett.* 5(8):679-682.

Background Levels of Trace Elements in Forest Ecosystems¹

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Abstract: This study was conducted as part of a project to develop a pollutant monitoring system for biosphere reserves. Sampling was carried out in the Great Smoky Mountains National Park and Olympic National Park. Results are reported for copper, lead, manganese, aluminum, calcium and phosphorus. Olympic National Park had much lower levels of lead and copper than Great Smoky Mountains National Park. Moss appeared to be a good collector for lead and copper. Results indicate that reference levels for trace elements can be established for remote areas, although they cannot be considered true background levels.

Introduction

Biosphere reserves are remote, pristine areas set aside in perpetuity. A pollutant monitoring system is being developed for implementation on the reserves (Wiersma and others 1978a; Wiersma and others 1979). Purpose of monitoring pollutants on these areas are:

1. to serve as locales for background reference levels of certain pollutants
2. to provide a frame of reference against which changes in impacted areas can be measured
3. to reflect changes of a global nature before such changes are obvious in more impacted areas.

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The reserves are areas that can be used to monitor the behavior of pollutants that have long range transport characteristics such as trace elements (Zoller and others 1974; Duce and others 1975; Thrane 1978; Weiss and others 1971; Scheslinger and others 1974; and Chow and Earl 1970).

Since many trace elements have the potential for long-term transport the question becomes, in a monitoring program for background areas, what elements should be of prime interest. Two parameters can be used to estimate potential for long-term transport. First, elements that have a high vapor pressure, and second, elements that would have a significant small particle (1.0μ or less) association. There is evidence that these two phenomena may work in conjunction. Ondov and others (1977a) analyzed the relationships existing between particle size and elemental composition in power plant emissions. They stated that elements with low vapor pressures tended to be associated with larger particle sizes. In a subsequent paper Ondov and others (1977b) listed several elements as having significant small particle association including manganese, lead and copper.

Kyser and others (1978), using a variety of microprobe analytical techniques, found that arsenic, cadmium, cobalt, chromium, manganese, nickel, lead, sulphur, antimony, selenium, thallium, vanadium and zinc were present on particles primarily as surface material. This lends further support to the hypothesis that

elements with high vapor pressure tend to condense on smaller particles.

Methods

Analytical

This paper will present data for trace element levels in vegetation and forest litter. The analytical procedure used was spark source emission spectroscopy (SSES) which determines 26 elements per sample. The analytical procedure has been previously described by Alexander and others (1975).

Every tenth sample submitted was a quality assurance sample, alternating between known value samples and replicated samples. Samples were submitted in a set order. The analytical laboratory was required to analyze the samples in the order submitted.

Field Sampling

Two biosphere reserves, the Great Smoky Mountains and Olympic National Park, have been sampled as part of a pilot research project to develop a cost effective pollutant monitoring system. Great Smoky Mountains National Park was originally sampled in the fall of 1977 and again in 1978. The results of this effort have been reported by Wiersma and others (1979). Olympic National Park was sampled in the summer of 1979.

Figures 1 and 2 show sample site locations for the Great Smoky and Olympic National Parks respectively. Details of the Olympic study design are presented by Brown and Wiersma (1979).

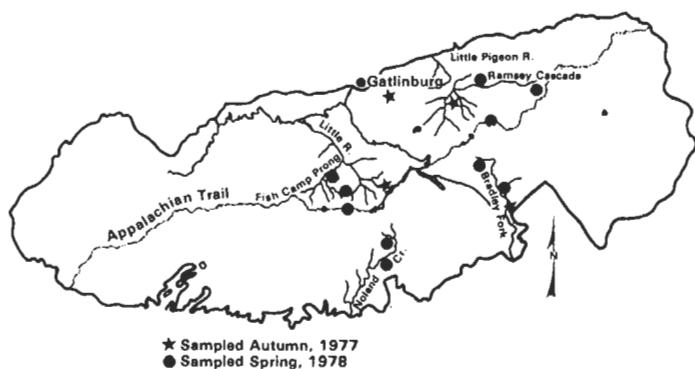


Figure 1--Sampling locations in the Great Smoky Mountains Biosphere Reserve.

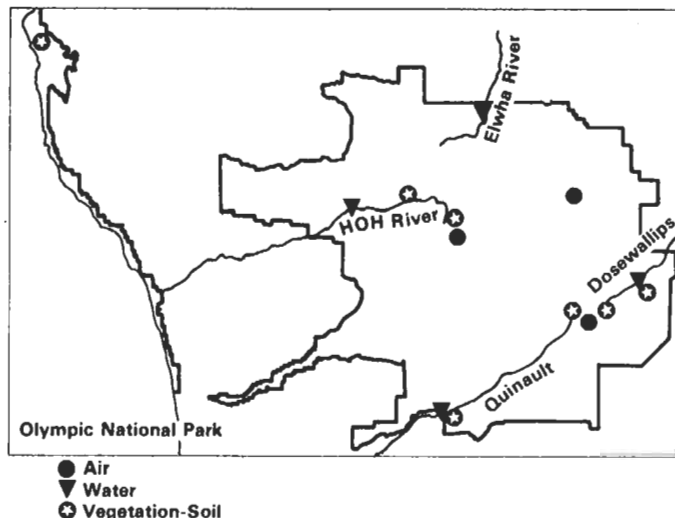


Figure 2--Sampling locations in the Olympic National Park Biosphere Reserve.

Results and Discussion

Trace Element Selection

The 26 trace elements analyzed by SSES are given in table 1.

Table 1--elements and detection limits for spark source emission spectroscopy.

Element	ppm	Element	ppm
P	50.0	V	1.0
Na	1.0	Co	1.5
K	150.0	Ni	0.5
Ca	1.0	Mo	0.2
Mg	50.0	Cr	0.2
Zn	5.0	Sr	0.2
Cu	0.2	Ba	0.2
Fe	0.6	Li	0.3
Mn	0.1	Ag	0.1
B	0.2	Sn	0.3
Al	0.1	Pb	1.0
Si	1.0	Be	0.2
Ti	0.5	Cd	3.0

Using this listing and selection criteria previously described, we are limiting our presentation in this paper to copper, lead and manganese. Also included are two biologically essential elements, calcium and phosphorus and one non-essential element, aluminum, which also is not one of the elements which has the potential for long-term transport.

Quality assurance results are shown in table 2. Good agreement exists between the analytical results and certified levels. In addition, the replicated samples, which were analyzed sequentially, showed no drift and had acceptable reproducibility.

Table 2--Quality assurance results for elements from Great Smoky Mountains.

	Analytical results for QA samples $\mu\text{g/g}$	NBS reported values $\mu\text{g/g}$	Drift of replicated samples ¹
Phosphorus	1,950	2,000 to 2,200	Not significant
Lead	44.7	44	Not significant
Copper	12.0	11.0 to 13.0	Not significant
Manganese	92.4	87 to 95	Not significant
Calcium	2.1 ³	2.1 ³	Not significant
Aluminum ²	-	-	Not significant

¹As determined by 95 pct. correlation coefficient of replicated samples and site numbers analyzed in order with site number

²No standard

³pct. by weight

Table 3 shows elemental levels for seven species of vegetation collected in the Great Smoky Mountains in the spring of 1978. Lead levels are highest in moss samples. Copper levels tend to be higher in wood fern and witch hobble, while manganese appears to be higher than previously reported for agricultural crops (Hemphill 1972). However, Romney and others (1977) report manganese levels for desert vegetation ranging from approximately 20 ppm to 220 ppm. Van Hook and others (undated) report manganese values in chestnut oak and hickory on the Walker Branch water shed ranging up to 1,000 ppm. Grodzinska (1978) reported manganese levels in moss from Poland ranged from 79 to 880 ppm. Therefore, the levels of manganese reported appears reasonable when compared to other studies, particularly from forested areas. Calcium and phosphorus levels appear to be equal to or slightly below calcium levels reported for agricultural crops (Hemphill 1972).

Table 4 presents the results for the second sampling that occurred in the fall of 1978. There was a large increase in lead levels in moss

samples, but the rest of the vegetation samples reflected a slight decline in lead levels. A similar relationship was shown for copper.

Studies using vegetation, particularly for those elements where root uptake is small, as indicators of airborne pollution have ranged from interception phenomenon of vegetative surfaces for modeling purposes (Shreffler, 1978; Davidson and others 1976) to the use of individual species as collectors of airborne pollutants. Smith (1977) has reviewed the probability of urban vegetation filtering out airborne particulates. He reported that fine hairs on vegetative surfaces increase particle trapping phenomenon. Carlson and others (1976) and Wedding and others (1975) found in controlled studies that rough, pubescent leaves entrap seven times more particles than smooth nonpubescent leaves and the particle load increases with leaf area sometimes by a factor of 10. Removal of particle from vegetation surface appears to be through solubilization in rain and not physical impaction from droplets (Carlson and others 1976). Compounding this phenomenon is data reported by Harris and others (1976) which states "...all elements are in a far more soluble phase in the ambient aerosol (and are associated with particles retained by biological and inert surfaces) than in fly ash collected from in stack deflector plates...".

From the above discussion, plants with large leaf surface areas, or those with very rough pubescent surfaces (ferns, witch hobble) should collect larger particulate loads, but because of wash off, they probably cannot be expected to increase the particle load throughout a growing season unless another phenomena were at work. This is shown in table 4 by a decrease for lead and copper concentrations for ferns and witch hobble when compared to the results in table 3. Manganese, however, does not follow this pattern.

Tyler (1972) states that mosses, via passive ion exchange, can accumulate a variety of airborne elements. If this is the case then solubilization of surface material will not be an important removal process and fall moss samples should have higher levels than spring, particularly for copper and lead. Data in tables 3 and 4 tends to support this hypothesis.

The forest floor of the Great Smoky Mountains was sampled in the spring of 1978. Two types of samples were collected, the first was the unincorporated organic material and the second was the partially decomposed material of the fermentation layer. The results are shown in table 5. Except for manganese, significant differences existed between the litter and fermentation layer for all trace elements. The fermentation layer showed an increase in lead, copper, phosphorus, aluminum, and a significant decrease in calcium. Site to site correlations for each element, excluding aluminum, were significant between unincorporated organic matter and the fermentation layer.

Table 3--Average concentration of selected elements for samples collected in the Great Smoky Mountains biosphere reserve, spring 1978.

	Lead μg/g	Copper μg/g	Manganese μg/g	Calcium pct.	Phosphorus μg/g	Aluminum μg/g
Moss	42.3	13.4	368	0.32	1,430	1,410
Yellow birch <u>Betula</u> <u>allegheniensis</u>	12.2	13.2	2,090	1.38	2,540	165
Red maple <u>Acer rubrum</u>	3.8	9.0	597	0.51	2,320	94
New York fern <u>Thelypteris</u> <u>novaborecensis</u>	12.3	9.9	279	0.15	2,250	555
Wood fern <u>Dryopteris</u> <u>spinulosa</u>	16.0	29.0	382	0.09	4,940	210
Witch hobble <u>Viburnum</u> <u>alnifolium</u>	14.5	28.2	1,730	0.68	3,660	252
Fraser fir <u>Abies fraseri</u>	3.1	3.8	590	0.17	2,090	278

Table 4--Average concentration of selected elements for samples collected in the Great Smoky Mountains biosphere reserve, fall 1978.

	Lead μg/g	Copper μg/g	Manganese μg/g	Calcium pct.	Phosphorus μg/g	Aluminum μg/g
Moss	107.6	15.3	459	0.30	145	1,850
Yellow birch	5.1	10.5	2,370	1.50	2,360	154
Red maple	0.9	6.6	1,070	1.14	1,920	66
New York fern	9.9	3.9	1,000	0.68	854	2,040
Wood fern	5.3	11.5	1,350	0.50	2,200	424
Witch hobble	13.3	9.2	2,800	1.50	1,700	426
Fraser fir	0.2	3.8	915	0.30	2,680	375

Reiners and others (1975) found lead levels in New Hampshire in litter increase with altitude. At extreme elevations, a slight decrease was noted. The levels of lead in litter ranged from 35 to 336 ppm. The fir forest sites had the highest lead concentrations. Figure 3 shows a similar relationship from our data for lead in the Smoky Mountains. This relationship did not exist for any of the other elements. Previous work by

Wiersma and others (1978b, 1980) indicate that the probable source of this lead is from anthropogenic activities.

Some data are available from samples collected in the summer of 1979 in Olympic National Park. The forest floor was not sampled by the method previously used in the 1978 Smoky Mountains study. With the modified technique, unincorporated

Table 5--Comparison of elemental levels in unincorporated organic material and the fermentation layer, Great Smoky Mountains biosphere reserve, spring 1978.

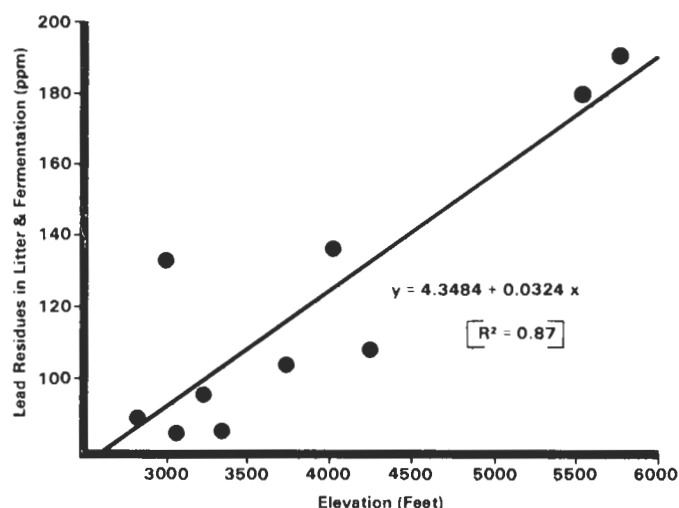
	Unincorporated organic material	Fermentation layer	T value ¹	Coefficient correlation (8 df) ²
Lead $\mu\text{g/g}$	37.8	83.7	9.6 ²	0.87 ⁴
Copper $\mu\text{g/g}$	18.2	28.8	9.4 ²	0.80 ⁴
Manganese $\mu\text{g/g}$	1,790	2,110	1.0 N.S.	0.77 ⁴
Calcium pct.	0.89	0.58	-5.2 ²	0.92 ⁴
Phosphorus pct.	0.28	1.18	5.0 ²	0.73 ³
Aluminum $\mu\text{g/g}$	1,980	5,420	Not Calculated	

¹Paired "T" test

²Element in unincorporated organic matter versus element in fermentation layer.

³95 pct. confidence

⁴99 pct. confidence



Copper levels in moss in the Great Smoky Mountains average about 3 times greater than in the Olympic National Park. Lead levels in the Smoky Mountains are 7 to 19 times greater than the Olympic National Park.

A similar relationship exists for the forest floor samples:

Comparison of forest floor samples from Olympic National Park and Great Smoky Mountains National Park. ($\mu\text{g/g}$)

	Copper	Lead
Olympic litter	10.0	16.7
Smokys unincorporated organic matter spring 1978	18.2	37.8
Smokys fermentation spring 1978	26.2	83.6

Figure 3--Relationship between lead residues and altitude in the Great Smoky Mountains Biosphere Reserve.

organic materials were sampled along with the fermentation layer.

Copper and lead levels in moss and litter are compared for the two biosphere reserves:

Comparison of moss samples from Olympic National Park and Great Smoky Mountains National Park. ($\mu\text{g/g}$)

	Copper	Lead
Moss-Olympic	4.4	5.7
Moss-Smoky Spring	13.5	42.4
Moss-Smoky Fall	15.3	108.0

True "background" levels for many trace elements are probably impossible to determine. Lead levels for moss in the Smoky Mountains appear high but in line with values reported in the literature. For example Ruhling and Tyler (1968) analyzed museum samples of moss. They found samples collected after 1950 had lead concentrations of 80 to 90 ppm. Samples collected around 1860 contained average lead levels of 20 ppm. These researchers believed that the 20 ppm level did not represent "natural" lead levels. They suspected "natural" lead levels might be considerably lower. The moss samples from Olympic

National Park averaged 5.7 ppm lead with some remote sites having average lead levels of 0.4 ppm and 2.2 ppm. Hirao and Patterson (1974) estimated for a remote site on the high sierra crest that 97 percent of the lead detected was from anthropogenic sources. Therefore, even for sites as remote as the high Dosewallips/High Quinault (over 13 miles from the nearest road) in a park that primarily receives wind off the Pacific Ocean, it may not be appropriate to consider the lead levels natural background levels.

It is our opinion that reference levels can be established in vegetation in remote areas for a variety of trace elements. Sites should be regionally representative and sampling should be repetitive through time, at least once a year, preferably twice a year. Biosphere reserves are ideal places to use because of their protected nature and the fact that they are selected to be representative of various biological systems.

Monitoring systems can and should be established in these reserves for the purposes listed.

Literature Cited

Alexander, G. V., D. R. Young, D. J. McDermott, M. J. Sherwood, A. J. Mearns, and O. R. Lunt.
1975. Marine organisms in the Southern California bight as indicators of pollution. In International Conference on Heavy Metals in the Environment. [1975. Toronto, Canada], p. 955-972.

Brown, K. W. and G. B. Wiersma.
1979. Pollutant monitoring in Olympic National Park biosphere reserve. In Proc. Second Conference on Scientific Research in the National Parks. [Nov. 28-30, 1979, San Francisco, Calif.] 12 p.

Carlson, R. W., F. A. Bazzaz and J. J. Stukel and J. B. Wedding.
1976. Physiological effects; wind reentrainment and rainwash of Pb aerosol particulate deposited on plant leaves. *Environmental Science and Technology* 10(12):1139-1142.

Chow, T. J., and J. L. Earl.
1970. Lead aerosols in the atmosphere: increasing concentrations. *Science* 169:577-580.

Davidson, C. I., S. V. Herring and S. K. Friedlander.
1976. The deposition of Pb-containing particles from the Los Angeles atmosphere. In Proc. International Conference on Environmental Sensing and Assessment. [Sept. 14-19, Las Vegas, Nevada] pp. 6-3 to 6-4.

Duce, R. A., G. L. Hoffman, and W. H. Zoller.
1975. Atmospheric trace metals at remote northern and southern hemisphere sites - Pollution or natural? *Science* 187:59-61.

Grodzinska, Krystyna.
1978. Mosses as bioindicators of heavy metal pollution in Polish national parks. *Water, Air and Soil Pollution* 9:83-97.

Harris, W. F., B. S. Ausmus, G. J. Dodson, Sidney Draggan, G. K. Eddlemon, Cyrus Feldman, J. M. Giddings, J. W. Huckabee, D. R. Jackson, S. A. Janzen, M. J. Levin, S. E. Lindberg, L. K. Mann, E. G. O'Neill, R. V. O'Neill, Cheryl B. Phillips, B. M. Ross, W. J. Selvidge, D. S. Shriner, R. R. Turner, P. Van Voris, Martin Witkamp.
1976. Environmental behavior of trace contaminants. The role of vegetation aerosol scavenging. ORNL-5257. Environmental Sciences Div. Annual Progress Report. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Hemphill, D. D.
1972. Availability of trace elements to plants with respect to soil-plant interaction. *Annals New York Academy of Sciences* 199:46-61.

Hirao, Y., and C. C. Patterson.
1974. Lead aerosol pollution in the high sierra over-rides natural mechanisms which exclude lead from a food chain. *Science* 184:989-992.

Keyser, T. R., D. F. S. Natusch, C. A. Evans, Jr. and R. W. Linton.
1978. Characterizing the surfaces of environmental particles. *Environmental Science and Technology* 12(7):768-773.

Ondov, J. M., R. C. Ragaini, R. E. Hett, G. L. Fisher, D. Silberman and B. A. Prentice.
1977a. Interlaboratory comparison of neutron activation and atomic absorption analyses of size-classified stack fly ash. Preprint UCRL-78194. Lawrence Livermore Laboratory, Livermore, Calif.

Ondov, J. M., R. C. Ragaini, A. H. Bierman, C. E. Choquette, G. E. Gordon and W. H. Zoller.
1977b. Elemental emissions from a western coal fired power plant: preliminary report on concurrent plume and in-stack sampling. Preprint UCRL-78825. Lawrence Livermore Laboratory, Livermore, Calif. 7 p.

Reiners, W. A., R. H., Marks and P. M. Vitousec.
1975. Heavy metals in subalpine and alpine soils of New Hampshire. *OIKOS* 26(3):264-274.

Romney, E. M., A. Wallace and G. V. Alexander.
1977. Boron in relationship to a coal-burning power plant. *Communications in Soil Science and Plant Analysis* 8(9):803-807.

- Ruhling, Ake and Germund Tyler.
1968. An ecological approach to the lead problem. Bot. Notiser 121:338-341.
- Schlesinger, W. H., W. A. Reiners and D. S. Knapman.
1974. Heavy metal concentrations and deposition in bulk precipitation in montane ecosystems of New Hampshire, U.S.A. Environmental Pollution 6:39-47.
- Shreffler, J. H.
1978. Factors affecting dry deposition of SO_2 on forests and grasslands. Atmospheric Environment 12:149-153.
- Smith, W. H.
1977. Removal of atmospheric particulates by urban vegetation. Implications for human and vegetative health. The Yale J. of Biology and Medicine 50:185-197.
- Thrane, K. E.
1978. Background levels in air of lead, cadmium, mercury and some chlorinated hydrocarbons measured in south Norway. Atmospheric Environment 12:1155-1161.
- Tyler, G.
1972. Heavy metals pollute nature, may reduce productivity. AMBIO 1(2):52-59
- Van Hook, R. I., W. F. Harris, G. S. Henderson and D. E. Reichle.
Undated. Patterns of trace element distribution in a forested watershed. Environmental Sciences Division, Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Wedding, J. B., R. W. Carlson, J. J. Stukel and F. A. Bazazz.
1975. Aerosol deposition on plant leaves. Environmental Science and Technology 9(2):151-153.
- Weiss, H. V., M. K. Koide, and E. D. Goldberg.
1971. Mercury in the greenland icesheet: evidence of recent input by man. Science 74:692-694.
- Wiersma, G. B., C. W. Frank, K. W. Brown and C. I. Davidson.
1980. Lead particles in the Great Smoky Mountains Biosphere Reserve. EPA-600/4-80-002. Environmental Monitoring Systems Laboratory, U.S. Environmental Protection Agency, Las Vegas, Nevada.
- Wiersma, G. B.
1979. Kinetic and exposure commitment analyses of lead behavior in a biosphere reserve. MARC Report 15. 41 p. Monitoring and Assessment Research Center, Chelsea College, Univ. of London, London, U.K.
- Wiersma, G. B., and K. W. Brown.
1979. Recommended pollutant monitoring system for biosphere reserves. In Proc. Second Conf. on Scientific Research in the National Parks. [Nov. 26-30, 1979, San Francisco, Calif.] 19 p.
- Wiersma, G. B., K. W. Brown, R. Herrmann, C. Taylor and J. Pope.
1979. Great Smoky Mountain preliminary study for biosphere reserve pollutant monitoring. EPA-600/4-79-072. Environmental Monitoring Systems Laboratory, U. S. Environmental Protection Agency, Las Vegas, Nevada.
- Wiersma, G. Bruce, Kenneth W. Brown and Alan B. Crockett.
1978a. Development of a pollutant monitoring system for biosphere reserves. EPA-600/4-78-052. Environmental Monitoring and Support Laboratory, U.S. Environmental Protection Agency, Las Vegas, Nevada. 114 p.
- Wiersma, G. B., K. W. Brown and A. B. Crockett.
1978b. Development of a pollutant monitoring system for biosphere reserves and results of the Great Smoky Mountains pilot study. In Proc. 4th Joint Conference on Sensing of Environmental Pollutants [1978, New Orleans, Louisiana], p. 451-456.
- Zoller, W. H., E. S. Gladney and R. A. Duce.
1974. Atmospheric concentration and sources of trace metals at south pole. Science 183:198-200.

Effects of Chronic Exposures to Gaseous Pollutants on Primary Production Processes

Photochemical Oxidant Impact on Mediterranean and Temperate Forest Ecosystems: Real and Potential Effects¹

John M. Skelly²

Abstract: Photochemical oxidants (primarily ozone) as air pollutants pose a more serious problem to forests of the United States than any other single air pollutant. Temperate and Mediterranean forests elsewhere have most likely been similarly impacted and current investigations of such effects are being pursued. Ozone (and its photochemically reactive precursors) has been demonstrated to occur at considerable distances downwind of major urban sources. Ozone has induced perturbations to vegetation over large areas and has therefore impacted innumerable and diverse forest ecosystems. Direct injury due to ozone has been documented to occur on numerous individual forest vegetation species but direct alterations of forest ecosystems as related to ozone induced effects have only been extensively documented in the San Bernardino Mountains of California; to a lesser degree similar studies have been done in the Blue Ridge Mountains of Virginia. Due to the current sulfur dioxide (SO₂) problems confronting European forests and due to the planned increased utilization of fossil fuels in much of the North American Continent, the influence of ozone in combination with SO₂ must be fully considered. The influence of other photochemical oxidants such as nitrogen oxides and peroxyacetyl nitrate on forest vegetation has remained relatively unknown.

The difficulty of understanding the real and potential impact of air pollutants such as ozone (O₃) on individual species within any given plant community as compared to relating direct or indirect effects of O₃ on entire and complex

forest ecosystems must be several orders of magnitude in difference. Several studies have related dose effects to full-sib crosses of various forest tree species but such studies have listed a series of caveats for the reader to consider while interpreting even the most basic results obtained. Such caveats have taken into account possible unknown effects due to exposure chamber systems, physical and biotic factor interactions, and/or related monitoring and data analysis methodologies. As research advances from studies of full-sib crossed plant materials through half-sib (open pollinated, maternal lines) through species and forest types, to forest communities and complex forest ecosystems, the number of influencing variables to be considered while interpreting the results becomes disproportional and difficult to comprehend.

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Most studies that have dealt directly with O_3 effects to any given plant species have not considered the effects of pollutant combinations even at higher doses. Under natural conditions, forest ecosystems are impacted by a multiplicity of atmospheric depositions including anthropogenic pollutants in gaseous, dry, and wet formulations. In addition such depositions do not occur singly at doses similar to those reported for various exposure chamber studies, e.g., 1-8 hours per day for 5 days/week for 10 weeks, but rather they occur at low concentrations, in various and constantly changing combinations and over extended exposure periods. Thus, ecosystem studies that have not considered even these few factors must be considered of limited value for any intrinsic or extrinsic interpretation.

It is a well known concept that air pollutant incited perturbations lead to simplification of forest ecosystems. Continued exposures to SO_2 complexes have resulted in simplification and reversal of once forested land to grassland communities or in some severe cases even to sterility of the soil and accompanying erosion. However, such severe impacts due to O_3 pollution have not been reported to occur with the possible exception of major forest vegetation species decline in the San Bernardino Mountains of Southern California. The complexity of such simplification scenarios involving minor changes in receptor plant physiology and related long-term effects on competitive abilities such as reproduction, nutrient cycling, and related food chain events has not been well documented.

The main purpose of this paper is not to present a review of literature pertaining to ozone impact to forest species and related ecosystems. Several excellent reviews of the subject have been previously published and due to these comprehensive up-to-date analysis of the topic any further review would result in a redundancy in the literature (Miller 1973; Miller and McBride 1975; Brown and others 1979 and Kozlowski 1980). Rather this symposium paper will attempt to relate these review articles and accompanying new literature into an analysis of the real and potential long-term effects of ozone on forest ecosystems. Additionally, an effort will be made to relate existing literature into suggestions for future consideration of ecosystem impacts into the National Ambient Air Quality Standards.

BACKGROUND AND ANTHROPOGENIC CONCENTRATIONS OF OXIDANTS IN FOREST SITUATIONS

A comprehensive review of ozone and other photochemical oxidants has been published by the National Academy of Sciences (NAS 1977); a recent compilation of papers has also been published (USEPA 1977b). These collective sets of papers have provided an excellent review of the origin, chemistry, transport, and atmospheric modeling of these pollutants.

Background Concentrations

Ozone is a naturally occurring component of the Earth's atmosphere and concentrations of 0.03-0.05 ppm O_3 are generally considered to be normal due to mixing via the stratospheric transport effect (Corn and others 1975) (fig. 1).

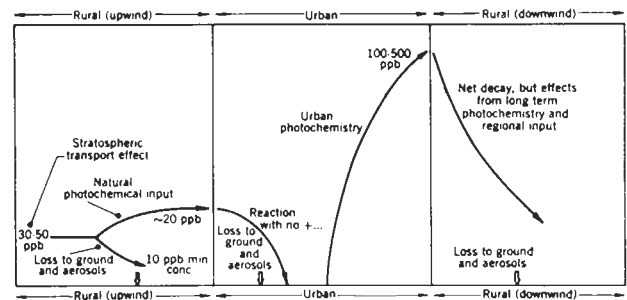


Figure 1--The tropospheric ozone cycle. (from Corn and others 1975).

The emission of oxidant forming precursors such as oxides of nitrogen and hydrocarbons into the lower atmosphere leads to the buildup of photochemically produced ozone and related peroxyacyl-nitrates (fig. 2) (NAS 1977).

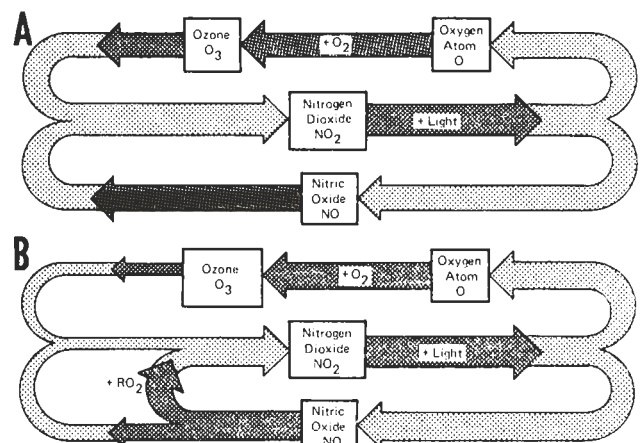


Figure 2--The normal nitrogen oxide photolytic cycle (A) and the cycle as altered by the addition of hydrocarbons leading to increased ozone concentrations (B) (from NAS 1977).

PAN has not been detected in non-anthropogenically polluted atmospheres but oxides of nitrogen do occur naturally in the atmosphere and concentrations of 0.02-0.10 ppm NO_x have been reported (NAS 1977). Brennan (Personal communication) has reported a high PAN concentration in the Eastern United States of 10 ppb but suggested overall concentrations of PAN are well below West Coast observations.

Anthropogenic Concentrations in Forested Areas

The long distance transport of oxidant precursors and ozone into remote forested areas has

been well documented (Miller and others 1972; Husar and others 1977; Hayes and Skelly 1977; and Cleveland and Kleiner 1975). Concentrations of ozone above the current National Ambient Air Quality Standard (NAAQS) of 0.12 ppm O_3 one hour average per 24-hour period; twice (2 days) per year have recently been reported within the forested areas of Eastern (Skelly and others 1979) and mid-Western United States (Wolff and others 1977). Numerous reports of high ozone concentrations have been issued from the extensive San Bernardino Mountain Studies (SBM) in southern California (USEPA 1977a). Major episodes of ozone have developed sporadically over the summer months of May through October in eastern forests whereas a better defined and stable oxidant season exists in the forests of southern California. Galloway and Skelly (1978) defined a major air pollution episode in July 1977 that involved high ozone concentrations and the highest ever recorded large and fine aerosol SO_4 concentrations (fig. 3, 4, and 5).

By comparing monitoring data from the SBM studies with those of the Blue Ridge Mountain Studies (BRM) a 3-4 fold greater concentration of ozone is apparent in the former over the latter. Peaks of 0.20 to 0.40 ppm O_3 daily one-hour maximums have occurred in the SBM forest (fig. 6) but the peak one-hour average ever recorded in the BRM area has been 0.166 ppm O_3 . Table 1 presents the monthly and peak one-hour concentrations of ozone as monitored at various sites in the Blue Ridge Mountains of Virginia. PAN and NO_x have not been monitored in eastern forests of the United States and although detected in western forests little has been done to distinguish differences in their respective effects over those induced by ozone alone (fig. 6).

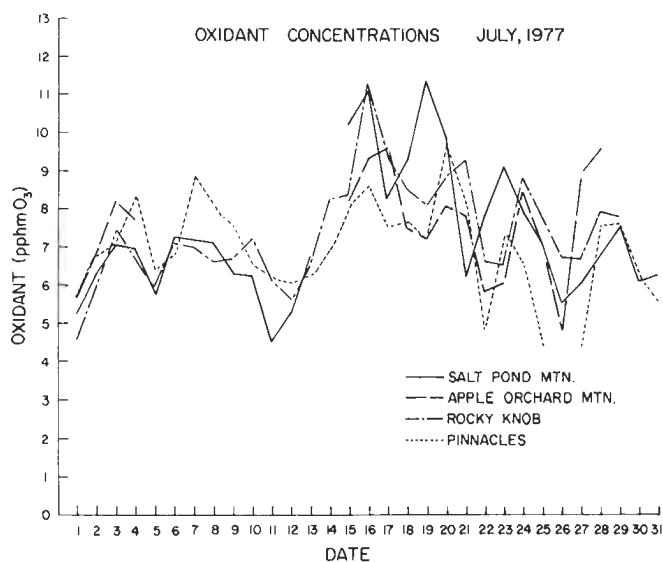


Figure 3--Oxidant concentrations as monitored at several locations in the Blue Ridge and Southern Appalachian Mountains of Virginia during July, 1977. Daily averages (24 hours) are indicated. Note peak period of July 15-20.

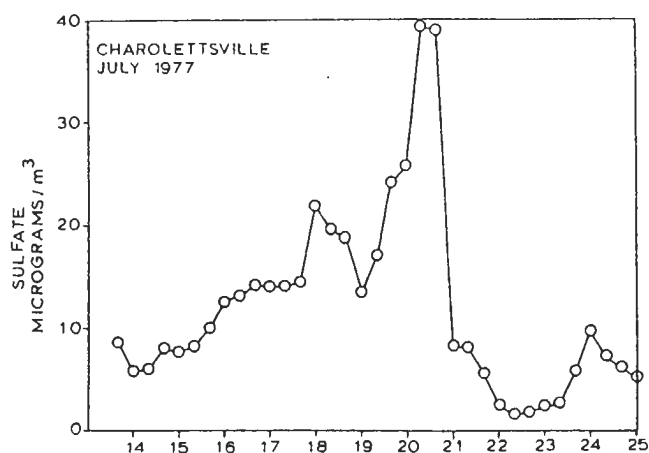


Figure 4--Total sulfate (mg/m^3) as monitored at Charlottesville, VA during air pollution episode of July, 1977. Note buildup of July 15-20 and similar sharp drop as in figure 3 on July. (From Galloway and Skelly 1978).



Figure 5(A)--A review of the Peaks of Otter as taken on a clear day in the Blue Ridge Mountains. Photograph taken from Pine Tree Overlook at a distance of 11.6 KM from the Peaks. (B) Photo taken on July 20, 1977 during the worst air pollution stagnation experienced in Virginia. Same view as (A).

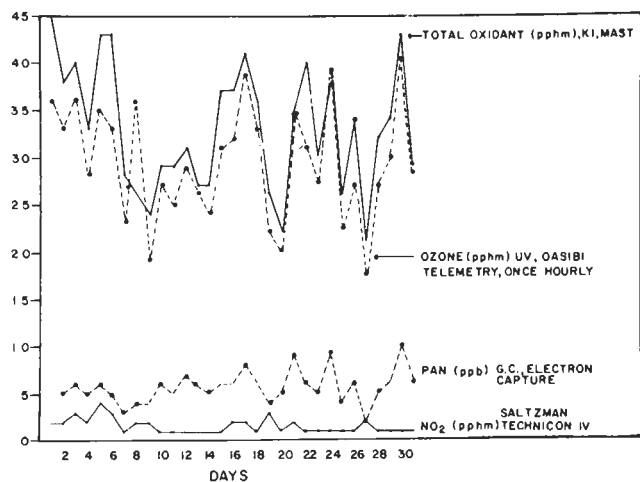


Figure 6--Comparative daily maximum hourly averages for ozone, total oxidant, PAN, and NO₂ at Sky Forest August, 1974. (From USEPA 1977).

Ozone concentrations in European forests have not been extensively investigated but initial studies indicate that ozone appears to be episodal in its occurrence in a manner similar to that experienced in Eastern United States. Grennfelt (1979) reported on ozone concentrations as monitored at Rörvik about 40 KM south of Gothenburg,

Norway; an area situated close to the coast and considered to be normally unaffected by local oxidant producing sources. The highest O₃ concentration observed at Rörvik was 0.20 ppm (August 1975) and in Gothenburg the highest O₃ concentration was 0.13 ppm (table 2). They suggested clockwise air movement as associated with high pressure systems and long distance transport from Europe to be related to the ozone episodes. Skärby (1979) reported that ozone values recorded in the summertime have been too high to be considered as normal background concentrations for Swedish conditions. During the summer of 1977 high O₃ concentrations (0.20 ppm) were recorded on 21 out of 92 days. She also suggested long-range transport to be involved.

From these few reports it is quite obvious that ozone concentrations in excess of the NAAQS occur frequently in the temperate and Mediterranean forests of the world. Therefore, it is also obvious that the injury thresholds for numerous plant species have also been frequently exceeded and the effects of these exposure doses are discussed below.

DIRECT EFFECTS TO FOREST SPECIES

The responses of any given forest ecosystem to

Table 1--Ozone concentrations (ppm) monitored at Rocky Knob, Floyd Co., Va. (Blue Ridge Parkway) and at Pinnacles and Big Meadows, Madison Co., Va. (Shenandoah National Park).

Month	1977				1978				1979			
	Rocky Knob		Pinnacles		Rocky Knob		Pinnacles		Rocky Knob		Big Meadows ¹	
	Month Aver.	Peak Hour Aver.	Month Aver.	Peak Hour Aver.	Month Aver.	Peak Hour Aver.	Month Aver.	Peak Hour Aver.	Month Aver.	Peak Hour Aver.	Month Aver.	Peak Hour Aver.
January	.035	.057	-	-	.024	.069	.032	.061	.033	.049	.042	.061
February	.041	.088	-	-	.033	.063	.047	.074	.042	.084	-	-
March	.046	.084	-	-	.048	.088	.052	.084	.057	.106	.059	.099
April	.055	.102	-	-	.052	.093	.051	.085	.072	.128	.054	.100
May	.054	.095	.051	.104	.060	.104	.059	.115	.065	.116	.048	.082
June	.051	.095	.057	.105	.068	.116	.073	.109	.071	.112	.051	.082
July	.073	.166	.069	.120	.064	.099	.070	.116	.057	.114	.044	.094
August	.056	.092	.059	.104	.052	.093	.057	.103	.052	.085	.050	.072
September	.051	.100	.051	.097	.054	.092	.054	.100	.038	.072	.040	.095
October	.041	.072	.038	.072	.051	.105	.051	.114	.037	.074	.038	.082
November	.032	.061	.033	.067	.044	.103	.066	.091	.032	.062	.036	.072
December	.024	.044	.032	.052	.030	.050	.039	.049	.018	.046	.024	.045
Average for monitored mo.	.047		.049		.048		.054		.048		.045	

¹ Pinnacles site moved 14 KM to Big Meadow (SNP) May 1979.

² Only 8 days out of the month were used for this data.

Table 2--The number of days with high ozone concentrations in Gothenburg and Rörvik, Norway 1972-1978 expressed as days with one hour mean of ozone exceeding indicated values (Grennfelt 1979).

Year	≥ 0.08 ppm	≥ 0.10 ppm	≥ 0.12 ppm	≥ 0.15 ppm	Max hourly mean (ppm)
<u>Gothenburg</u>					
1972	18	3	0	0	0.11
1973	12	1	0	0	0.10
1974	4	3	0	0	0.11
1975	13	1	0	0	0.10
1976	11	1	0	0	0.11
1977	9	1	1	0	0.13
¹ 1978	8	6	2	0	0.12
<u>Rörvik</u>					
1975	-	18	11	5	0.20
1976	-	17	5	0	0.13
1977	-	6	0	0	0.11
¹ 1978	-	12	8	1	0.15

¹Includes data only until June 30, 1978.

photochemical oxidants must be as a result of the expression of direct effects to the individual component species within that ecosystem. The ability of scientific research to define those direct effects to individuals and to relate such defined effects to the whole has been limited. Three terms have been used somewhat interchangeably to define these effects and for purposes of this paper they shall be defined as:

Injury - the result of one or more deleterious alterations of normal physiological processes as manifested by the presence of chronic or acute visible symptoms and/or growth reductions (growth reduction may be the only manifestation).

Damage - injury that results in measurable economic loss to specific crops e.g. reduced height or radial increment growth of forest trees resulting in reduced value of the commercial forest.

Impact - the total influence (detrimental or beneficial) of air pollutants on all aspects of the forest ecosystem including even minor shifts towards reduced diversity of species, indirect effects to watersheds and water quality, or direct effects to recreational values due to reduced visibility at vistas or overlooks in National Forests and Parks.

Extensive research has been justifiably done on specific forest species to better define the injury phase of this scenario of increasingly inclusive terminology. However, even within this type of problem definition research a larger endeavor has been made to define visible plant responses over the less easily measured physiological responses. The latter responses may actually be of more importance to understanding the damage and impact phases of subsequent ecosystem deteriorations. Thus, three additional terms emerge:

acute injury - involves expression of clinical symptoms leading to death of cells, tissues, organs, or entire plants and/or plant communities. Such injury is usually initiated by exposure to high doses of pollutants ($>\text{concentration} \times <\text{time}$) but may result from exposure to lower concentrations of pollutant over extended periods of time.

chronic injury - involves non-lethal types of clinical symptom expressions such as reduced chlorophyll production and related pigmentation changes, and reduced growth rates. Such injury usually results from still lower dose exposures.

functional injury - involves injury to the functional efficiency of the plant as expressed by reduced growth or other expression of loss

without the development of any other clinical symptoms, i.e. injury is only of a physiological and pre-clinical nature. Further visible symptoms do not develop. This form of injury is most difficult to define and measure and is the result of lowest dose exposures at predominantly near background concentrations of pollutants over extended periods of time.

An attempt to define the current status of knowledge concerning these various forms of direct effects due to ozone has been presented in table 3. A more specific assessment of recent documented studies of chronic pollution effects as induced by all species of air pollutants at the ecosystem level has been presented by Kickert and Miller (1978).

It is evident from table 3 that our knowledge of ozone effects following high dose exposures using artificial exposure systems for species level investigations is considered somewhat superior to that of the other levels of activity. However, when the abundance of plant species

indigenous to a mixed hardwood-conifer forest ecosystem of the Northeastern United States is taken into account our knowledge is very limited even for the clinical expression of symptoms by most species. Interpretation of available knowledge in the forest community and forest ecosystem columns may be somewhat harsh but most probably realistic. A similar table as constructed for sulfur dioxide related investigations would be much more optimistic thus attesting to the relative ease of working on predominant point sources of pollutants and their related effects. However, our knowledge of SO₂ induced perturbations to forest ecosystems located at distances from various sources is relatively poor, e.g. knowledge of the subtle influences of S-related compounds to the productivity of Northeastern United States forests is virtually non-existent.

Direct Effects to Forest Trees

Davis and Wilhour (1976) have provided the most complete listing of woody plant sensitivities to sulfur dioxide and photochemical oxidants as

Table 3--The current status of knowledge concerning ozone induced effects to Temperate and Mediterranean forest tree species, forest communities, and forest ecosystems. Comparisons of the San Bernardino Mountain Studies (SBM) versus all other investigations (OI) have been noted.¹

Effect	Study	Forest Species ²	Forest Communities	Forest Ecosystems
Injury				
overall	OI SBM	moderate moderate	poor moderate	poor moderate
acute	OI SBM	abundant abundant	moderate moderate	poor moderate
chronic	OI SBM	moderate moderate	poor moderate	non-existent poor
functional	OI SBM	poor moderate	non-existent moderate	non-existent poor
Damage	OI SBM	moderate abundant	poor moderate	non-existent poor
Impact	OI	<u>3/</u>	non-existent moderate	non-existent poor

¹In consultation by author with P. R. Miller (personal communication).

²Estimates in this column include responses obtained in fumigation chamber studies. If such information was to be detected poor to non-existent descriptions would be appropriate for each category.

³By definition not applicable.

derived from their review of United States, Canadian, and European literature.

Eastern White Pine

The predominant forest tree species studied in Eastern United States has been eastern white Pinus strobus L.; extensive literature reviews by Gerhold (1977) and Nicholson (1977) are available. Interest in this species has remained high since the first discovery of its somewhat unique sensitivity to ozone by Berry (1961). The response of this species to ozone concentrations as monitored in the Blue Ridge Mountains of Virginia (table 1) has been the subject of several current investigations. Skelly and others (1979) reported that of 315 white pines surveyed by using a modified evaluation scheme as adapted from Miller (1973) 17, 80, and 3 percent were considered to be tolerant, intermediate and sensitive, respectively, to ozone. Of the 315 trees tagged in 1977, 10 were reported to have died following repeated typical clinical symptoms of oxidant induced injury. Subsequent root excavations and isolation of fungi has yielded Verticilladiella procera from the dying trees. The growth rate of trees in each class was also examined by Benoit (1980) and radial increment growth over the period 1955-1978 for the sensitive class was significantly less ($p = .01$) than that of the tolerant class (figure 7). A general decline in growth for all classes was noted.

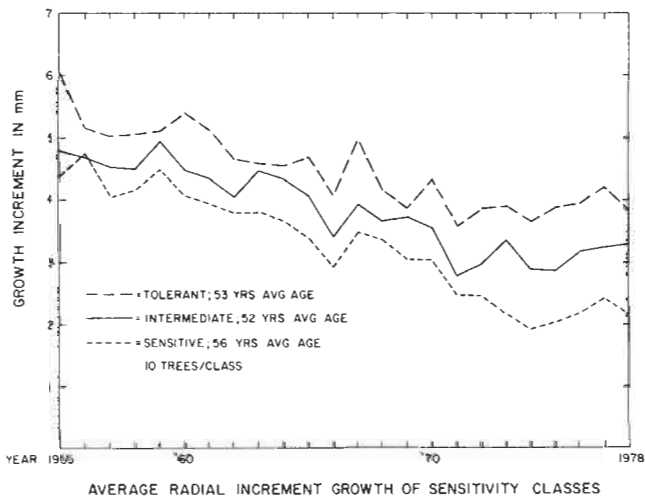


Figure 7--The average radial increment growth of eastern white pine in three ozone sensitivity classes as found in the Blue Ridge Mountains of Virginia. Trees were located in groups of 3 per site with each sensitivity class represented in each of 10 plots (10 trees/class).

Other Eastern Species

Forest trees--the relative O_3 sensitivity of species has been investigated using high dose exposures, e.g. 0.25 ppm O_3 for 8 hour exposure periods. Using such doses Davis and Wood (1972)

exposed 18 conifer species and found Austrian Pine (Pinus nigra, Arnold), jack pine (P. banksiana, Lamb.) and Virginia pine (P. virginiana, Mill.) to be the most sensitive. However, they reported variable symptom response among the different species, among plants within species, and between branches and needles of individual plants. In a series of exposures, numerous deciduous trees were also exposed to similar ozone doses and green ash (Fraxinus pennsylvanica Marsh.), white ash (F. americana L.) and tulip poplar (Liriodendron tulipifera L.) were reported to exhibit foliar injury by Wood (1970). Jensen (1973) determined the sensitivity of 9 deciduous tree species on the basis of height growth during a 5 month exposure to 0.30 ppm O_3 for 8 hours per day. He determined that silver maple (Acer saccharinum, L.), green ash, and sycamore (Platanus occidentalis, L.) were the only species determined to be sensitive using both parameters. Numerous similar studies using high dose exposures have been reviewed by Skelly and Johnston (1979).

Typical oxidant symptoms have been noted by Skelly (unpublished) on several major forest tree species indigenous to Shenandoah National Park of Virginia (fig. 8).



Figure 8--Typical oxidant induced stippling on hickory (Carya spp.) as observed in the Shenandoah National Park of Virginia. Note asymptomatic area of covered over portion of lower leaf in center of photograph (upper leaf pulled back).

Very few low dose exposure studies have been conducted to determine effects due to closer to ambient pollutant concentration or due to ambient exposure conditions. However, several recent studies have attempted to reproduce ambient concentrations of O_3 , SO_2 , and NO_x (and various combinations thereof) and to study their effects on the growth of loblolly pine (P. taeda L.) Kress and Skelly (1980a) American sycamore Kress and Skelly (1980b), and several other eastern tree species (Kress 1980). In the combined pollutant studies using 0.05 ppm O_3 , 0.10 ppm NO_2 , and 0.14 ppm SO_2 for 6 hours per day for 28 consecutive days significant height reductions

were reported as induced by O_3 alone treatments for each species without clinical symptoms present on sycamore and with <5 percent foliar injury on loblolly pine. Kress (1980) reported height growth increases and/or decreases for 10 tree species following exposure to 0.05, 0.10, and 0.15 ppm O_3 for 6 hours/day for 28 consecutive days. Lowest dose exposure significantly reduced the height of loblolly pine and 0.10 ppm O_3 reduced height growth in loblolly pine, green ash, sycamore, pitch pine (*P. rigida* Mill.) and sweetgum (*Liquidambar styraciflua* L.). Slight height growth stimulations were reported for several species following 0.05 ppm O_3 treatment and sugar maple (*A. saccharum* Marsh.) responded positively even at the 0.10 ppm O_3 treatment ($p = 0.05$).

Understory Vegetation--Harward and Treshow (1971) have conducted one of few studies designed to determine ozone effects to understory plants. They exposed 17 representative species of an aspen community to various high and low doses of O_3 5 days per week throughout the growing season for 3 consecutive years. Several species were found more sensitive than expected and sensitivity was so varied between species that the authors suggested major shifts in community composition would be probable following only a year or two of exposure.

Kohut and Krupa (1978) determined the sensitivity of several herbaceous plants of the North-central U.S. forests. They listed a group of plants found in the forests of the North Central region that were also sensitive to 0.08 and 0.15 ppm O_3 for only 4 hours. These plants are also native to the forested areas of the north and southeastern portions of the United States. The sensitive plants listed were wild buckwheat, chicory, daisy, mustard, and *Ribes*. Other work by Skelly (1977, unpublished) has identified symptoms typically induced by O_3 on *Clematis* sp. in the Shenandoah National Park of Virginia.

Figure 9 has been presented to further demonstrate the effect of ambient concentrations of ozone on a species that is widely distributed across North America i.e. common milkweed (*Asclepias* spp.). Duchelle and others (1980) observed severe, moderate, and only slight injury to this species that occurred naturally in open plots and in non-filtered and filtered open top chambers, respectively, as located in the Shenandoah National Park in Virginia. This species is being tested further for injury thresholds and for use as part of a plant bioindicator system.

As another part of the Blue Ridge Mountain Studies, Duchelle and Skelly (1980) established 4 replications of open top chambers receiving charcoal filtered air or non-charcoal filtered air in order to investigate height growth of selected forest trees. Four open plots were also established. By mid-summer it became neces-

sary to clip (to a 1.3 cm height) and remove competing natural vegetation which was then collected for dry weight measurements (table 4).



Figure 9--Oxidant injury to milkweed (*Asclepias* spp.) observed in the Shenandoah National Park, VA as grown in (A) open plots, (B) non-filtered open top chamber and (C) charcoal-filtered open top chamber. Purple stipple was only noted on upper leaf surfaces.

Table 4--The dry weight of foliage of composited clippings as collected from 4 filtered air and 4 non-filtered air open top chambers and 4 open plots established in the Big Meadows Area of the Shenandoah National Park, VA.

Exposure	Clipping Dates ¹			
	Aug. 14, 1979		Oct. 9, 1979	
	Dry Weight (Grams)			
	Total wt ²	Average	Total wt	Average
Filtered	7263	1816	1599	400
Non-filtered	4937	1234	1323	331
Open	3128	782	845	211

¹All species composited as clipped to 1.3 cm height, 10 foot diameter plots.

²4 replications.

Western Species

Ponderosa Pine

Since the decline of ponderosa pine (P. ponderosa Laws.) during the 1950's (and henceforth into the 1980's) in the South Coast air basin and San Bernardino Mountains of California, this species has become the most intensively investigated of all western forest vegetation species. Declining ponderosa pines have most recently been reported in the southern Sierra-Nevada mountains by Miller and Millecan (1971) and in the Sequoia National Forest and Sequoia-Kings Canyon National Parks by Williams and others (1977).

The decline of this species was initially termed "X-disease" by Parameter and others (1962); further study by Miller and others (1963) elucidated ozone to be the direct incitant. Interrelationships of foliar symptoms with increased root disease and increased incidences of bark beetle infestations in injured trees have been determined along with significant growth decreases and mortality (Stark and others, 1968; McBride and others, 1975). Several reviews of the major studies that have dealt with ponderosa pine have recently been published by Brown and others (1979), Miller (1973) and Miller and McBride (1975).

Most recent investigations by Coyne and Bingham (in press) identified characteristics of ecotypic variation in P. ponderosa which varied in their clinical symptom response to ozone under field conditions. Light responses, photosynthetic rates, and stomatal conductances were observed and differences among injury classes were manifest as acceleration of the normal decline of CO₂

fixation and stomatal conductance usually associated with needle aging. Needles of sensitive trees senesced and abscised prematurely thus contributing to a steady decline in tree vigor and increased vulnerability to other sources of stress.

Other Forest Tree Species

Miller (1973) ranked the following species for their decreasing sensitivity to ozone following fumigation tests:

Most sensitive	Western white pine (<u>P. monticola</u> Doug.) Jeffrey x Coulter pine hybrid (<u>P. jeffreyi</u> x <u>P. coulteri</u>) Red fir (<u>Abies magnifica</u> A. Murr.) Monterey x Knobcone pine hybrid (<u>P. radiata</u> x <u>P. altenuata</u>) Ponderosa pine (<u>P. ponderosa</u> Laws.)
Intermediate	Coulter pine (<u>P. coulteri</u> D. Don) Douglas fir (<u>Pseudotsuga menziesii</u> (Mirb.) Franco) Jeffrey pine (<u>P. jeffreyi</u> Grev. & Balf.) White fir (<u>Abies concolor</u> (Gord. & Glend.) Lindl.) Big cone Douglas fir (<u>Pseudotsuga macrocarpa</u> (Vasey) Mayr.) Knobcone pine (<u>P. attenuata</u> Lemm.)
Tolerant	Incense cedar (<u>Libocedrus decurrens</u> Torr.) Sugar pine (<u>P. lambertiana</u> Dougl.) Giant sequoia (<u>Sequoia gigantea</u> (Lindl.) Decne.)

In one of few low O₃ dose exposure studies, 9 western conifer species have been evaluated for foliar injury and growth responses by Wilhour and Neely (1977). They found significant growth reduction in juvenile seedlings of P. ponderosa and P. monticola exposed to 0.10 ppm O₃ for 6 hours/day consecutively for up to 22 weeks. They noted no constant association between growth response and foliar injury.

DIRECT EFFECTS TO FOREST ECOSYSTEMS

A fair number of published reports assessing air pollution induced injury to agricultural crops are available. Damage estimates of yield losses have also been published e.g. air pollution injury to potatoes in the Atlantic Coastal States (Heggstad, 1973). Such evaluations have been based upon extensive knowledge of crop management practices and abundant information exists concerning expected yields. Therefore, yield

losses such as potato tuber number, size, and weight may be easily determined and subsequently correlated with the degree of foliar injury as induced by photochemical oxidants (Heggstad, 1973).

Advancing from a relatively simplistic agro-nomic monocultural system of crop management to a mixed hardwood conifer forest ecosystem poses considerably different problems in determining an effect due to any given single stress factor. As noted previously, an abundance of literature describing symptoms has become available for several major forest species that appear sensitive to photochemical oxidants (table 3). However, the overall impact of the more subtle changes in functional efficiency of the plant expressed, for example, as slightly reduced photosynthetic capability of otherwise asymptomatic foliage or noted trends of slightly shorter pollen tube length in the presence of low doses of O_3 remain relatively little understood.

There have been no major investigations of oxidant impacts to forest communities or forest ecosystems with the single well known exception of the San Bernardino Mountain Studies. The intensity of the various studies and their related interactions within one of many possible areas of investigation has been illustrated in figures 10 and 11. As noted in figure 10, oxidant air pollutants are only one of many natural or anthropogenic stress factors important to the forests of the San Bernardino Mountains and if this figure were to be modified for illustration of an Eastern U.S. or European forest, SO_2 and other forms of atmospheric depositions would necessarily be added for emphasis. Figure 11 illustrates only a few of the complex interactions which may take place within an ecosystem and even a very subtle change in bark characteristics or carbohydrate pools as initially induced by oxidant injury of the foliage may be sufficient to encourage a bark beetle attack of weakened trees. As related in table 3, an evaluation of knowledge concerning injury, damage, and impact to forest species of the San Bernardino Mountains (primarily *P. ponderosa*) is abundant and knowledge is considered to be moderate for the forest community. However, due to the complexity of any given forest ecosystem, including that which has been intensively studied in the San Bernardino Mountains, relatively poor information exists as to the subtle influences to the functioning of any given forest ecosystem.

FUTURE RESEARCH EMPHASIS AND CONSIDERATIONS

Why do we know so little about oxidant induced injury, damage, or impact to forest communities or forest ecosystems? Why have only a few tree species been intensively studied and relatively few others evaluated for foliar symptom response? How does the lack of such information influence decisions made regarding the establishment of National Ambient Air Quality Standards?

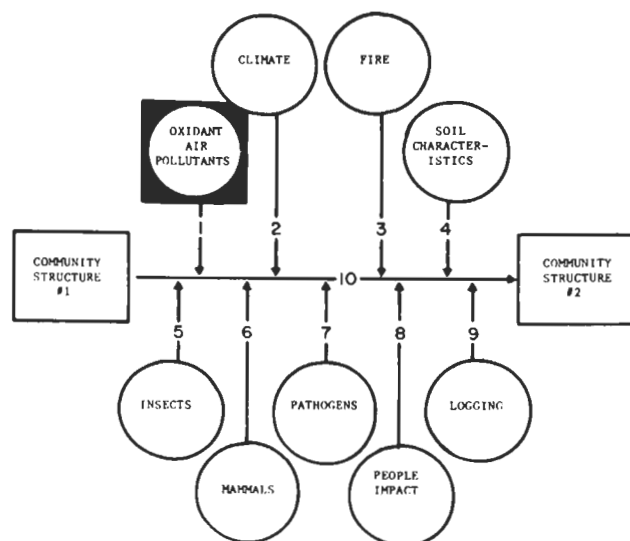


Figure 10--Community-succession interactions in a mixed-conifer forest ecosystem. (From Taylor, 1974).

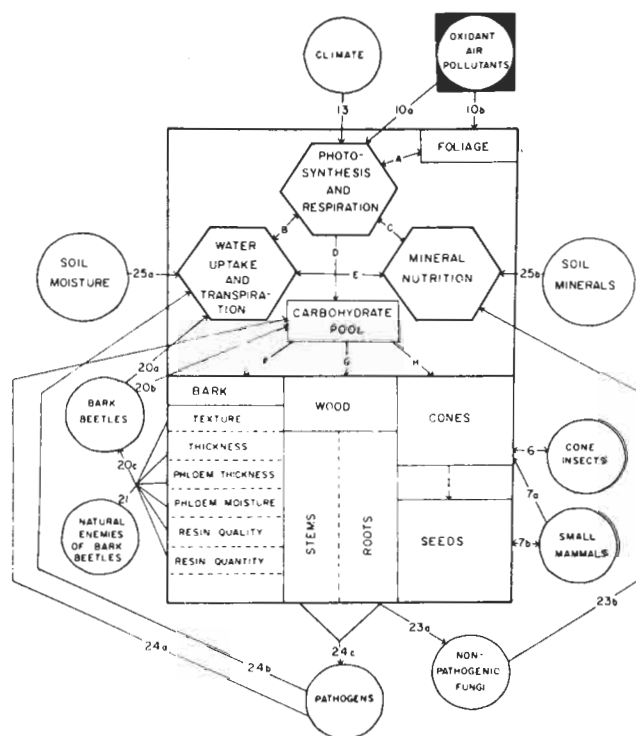


Figure 11--An example of tree-level interactions in a mixed-conifer forest ecosystem. Data from these types of studies would be integrated into overall effect illustrated in figure 11. (From Taylor, 1974).

Intensive management of agronomic crops through selection of tolerant varieties for planting in

high oxidant areas and appropriate changes in cultural practices such as withholding irrigation water during oxidant pollution episodes have all served to reduce the immediate injury (and therefore damage and impact) to such important crops as potatoes, tobacco, soybeans, snapbeans and certain horticultural plants. Through the development of such practices, the agricultural scientist has assisted the immediate grower (as should be the case) but concurrently the economic cost/benefit justifications for pollution abatement enforcement have been lessened since overall impact has potentially been greatly reduced. The evaluation of forest tree species for sensitivity to various pollutants has also been attempted and likewise long term growth losses may be averted. Such investigations must be approached with a certain degree of caution since long term subtle changes most undoubtedly are occurring in natural forest ecosystems but as yet most remain undetected. Natural ecosystems usually have a system of checks and balances but the system remains delicate and trends towards simplification are easily initiated. Changes in primary productivity, energy resource flow patterns, biogeochemical cycles, and species successional patterns may all be challenged by oxidant air pollution but have remained virtually non-studied.

The difficulties encountered in developing research aimed at isolating, identifying, and subsequently integrating the known effects of oxidant air pollutants on the forest ecosystem are too numerous to completely cover in the remaining space available. A partial listing however, must include these important points:

- 1) Photochemical oxidant is insidious over extremely large areas of diverse forest land and the establishment of control (non-pollution exposed) areas has become virtually impossible.
- 2) The introduction of charcoal filtration systems into such areas to establish "controls" is at best artificial and due to physical restrictions the ability of such systems to define a true ecosystem perturbation is likewise limited.
- 3) Long-term investigations must take into account innumerable variables some of which are very transient in their occurrence and others for which relatively little is known even under natural conditions.
- 4) Modeling and related statistical procedures must take into account the diversity and complexity of a forest ecosystem and statistically probabilities of $p = .01, .05, \text{ or } .10$ must not be the only acceptable limits of "biological significance." The prediction of a biological event with 70 percent accuracy may be valid. Observation of only a 1-5 percent decrease in annual radial increment growth or similar decreases in pollen production and viability may not

be statistically significant but may have greater long-term biological significance to the ability of a species to survive over their natural range.

The natural forested ecosystems of the temperate and Mediterranean regions of the world may serve to provide the most invaluable bioindicator of long-term photochemical oxidant air pollution induced effects. National air quality standards must be reasonably developed to protect these natural resources from even minor change but initially adequate financial support and associated quality research must be continued to adequately define the real and potential effect of oxidants to plant species, to plant communities, and to entire forest ecosystems.

LITERATURE CITED

- Benoit, L. F.
1980. Ozone effects on long term radial increment growth and reproduction of eastern white pine. (Abstr.) Proc. Potomac Div. Amer. Phytopath. Soc., Morgantown, W. Va.
- Berry, C. R.
1961. White pine emergence tipburn, a physiogenic disturbance. (Sta. Paper No. 130., 8 p.). Southeast For. Exp. Sta., USDA-For. Serv., Athens, Ga.
- Brown, H. D., P. R. Miller, J. M. Skelly, D. B. Drummond, and C. E. Carlson.
1979. Air pollution effects on forest vegetation and the analysis of the role of Forest Insect and Disease Management. 134 p. USDA-For. Serv., Washington, D.C.
- Cleveland, W. S. and B. Kleiner.
1975. Transport of photochemical air pollution from The Camden-Philadelphia urban complex. Environ. Sci. Tech. 9:869-872.
- Corn, M., R. W. Dunlap, L. A. Goldmuntz and L. H. Rogers.
1975. Photochemical oxidants: sources, sinks and strategies. J. Air Poll. Contr. Assoc. 25:16-18.
- Coyne, P. I. and G. E. Bingham.
1980. Comparative ozone dose response of gas exchange in a ponderosa pine stand exposed to long term fumigations. (Submitted for publication to J. Air Poll. Contr. Assoc.).
- Coyne, P. I. and G. E. Bingham.
1980. Variation in photosynthesis and stomatal conductance in an ozone-stressed ponderosa pine stand: Light response. (Submitted for publication to For. Sci.).
- Davis, D. D. and R. G. Wilhour.
1976. Susceptibility of woody plants to sulfur dioxide and photochemical oxidants. 72 p. USEPA Ecol. Res. Series EPA-600/3-76-102.

- Davis, D. D. and F. A. Wood.
1972. The relative susceptibility of eighteen coniferous species to ozone. *Phytopathology* 62:14-19.
- Duchelle, S. F., J. M. Skelly and L. W. Kress.
1980. The impact of photochemical oxidant air pollution on biomass development of native vegetation and symptom expression of *Asclepias* spp. (Abstr.). Proc. Potomac Div. Amer. Phytopath. Soc. Morgantown, W. Va.
- Galloway, J. N. and J. M. Skelly.
1978. A pollution episode in Virginia. *Virginia Climate Advisory* 2(3):4-6.
- Gerhold, H. D.
1977. Effect of air pollution on *Pinus strobus* L. and genetic resistance. 45 p. USEPA Rep. No. EPA-600/3-77-002. Corvallis, Or.
- Grennfelt, P.
1979. Ozone monitoring in Scandanavia. In Report from the workshop "Ozone effects on vegetation in Europe." M. Eastmond and L. Skarby Eds. p. 5-8. Swedish Water and Air Res. Inst., Stockholm.
- Harward, M. R. and M. Treshow.
1971. The impact of ozone on understory plants in the aspen zone. Proc. 64th Ann. Mtg. Air Poll. Contr. Assoc. Atlantic City, N.J.
- Hayes, E. M. and J. M. Skelly.
1977. Transport of ozone from the northeast U.S. into Virginia and its effect on eastern white pine. *Plant Dis. Repr.* 51:778-782.
- Heggestad, H. E.
1973. Photochemical air pollution injury to potatoes in the Atlantic Coastal States. *Amer. Pot. J.* 50:315-328.
- Husar, R. B., D. E. Patterson, C. C. Paley, and N. V. Gillani.
1977. Ozone in hazy air masses. In International Conference on Photochemical Oxidant and its Control. p. 275-282. USEPA Ecol. Res. Series EPA-600/3-77/001b. Research Triangle Park, N.C.
- Jensen, K. F.
1973. Response of nine forest tree species to chronic ozone fumigation. *Plant Dis. Repr.* 57:914-917.
- Kickert, Ronald, N. and P. R. Miller.
1978. Responses of ecological systems. In Handbook of methodology for the assessment of air pollution effects on vegetation. W. W. Heck, S. V. Krupa, S. N. Linzon, eds. p. 14.1-14.45 Air Poll. Contr. Assoc. Pittsburgh, Pa.
- Kohut, R. J. and S. V. Krupa.
1978. Sensitivity of selected species of native vegetation to ozone. (Abstr.). Proc. Am. Phytopathological Soc. 4:88.
- Kozlowski, T. T.
1980. Impacts of air pollution on forest ecosystems. *Bio. Sci.* 30:88-93.
- Kress, L. W.
1980. Effects of O_3 and $O_3 + NO_2$ on growth of tree seedlings. Proc. Internat'l. Symp. on Eff. of Air Pollutants on Mediterranean and Temperate Forest Ecosystems. June 22-27. Riverside, Ca.
- Kress, L. W. and J. M. Skelly.
1980. Growth impact of O_3 , NO_2 , and/or SO_2 on *Pinus taeda*. (Submitted for publication to *Atmos. Environ.*).
- Kress, L. W. and J. M. Skelly.
1980. Growth impact of O_3 , NO_2 , and/or SO_2 on *Platanus occidentalis*. (Submitted for publication to *Atmos. Environ.*).
- McBride, J. R., V. P. Semino, and P. R. Miller.
1975. Impact of air pollution on the growth of ponderosa pine. *Calif. Ag.* 29(12):8-9.
- Miller, P. R.
1973. Oxidant-induced community change in a mixed conifer forest. In *Air Pollution Damage to Vegetation*. J. A. Naegela Ed. p. 101-117. Adv. in Chem. Series 122. Amer. Chem. Soc., Washington, D.C.
- Miller, P. R. and J. McBride.
1975. Effects of air pollutants on forests. In *Response of Plants to Air Pollution*. B. Mudd and T. T. Kozlowski, eds. p. 195-235. Academic Press, New York.
- Miller, P. R. M. H. McCutchan and H. P. Millegan.
1972. Oxidant air pollution in the central valley, Sierra Nevada foothills, and Mineral King Valley of California. *Atmos. Environ.* 6:623-633.
- Miller, P. R. and A. A. Millecan.
1971. Extent of oxidant air pollution damage to some pines and other conifers in California. *Plant Dis. Repr.* 55:555-559.
- Miller, P. R., J. R. Parmeter, Jr., O.C. Taylor, and E. A. Cardiff.
1963. Ozone injury to the foliage of *Pinus ponderosa*. *Phytopathology* 53:1072-1076.
- National Academy of Sciences.
1977. Ozone and other photochemical oxidants. Comm. on Med. and Biol. Eff. of Environ. Pollutants. 719 p. NAS, Washington, D.C.

- Nicholson, C. R.
1977. The response of 12 clones of eastern white pine (Pinus strobus) to ozone and nitrogen dioxide. 141 p. M.S. Thesis, Virginia Polytechnical Institute and State University. Blacksburg, Va.
- Parmeter, J. R., Jr., R. V. Bega, and T. Neff.
1962. A chlorotic decline of ponderosa pine in southern California. *Plant Dis. Reptr.* 46:269-273.
- Skärby, L.
1979. Elevated ozone levels at the Swedish west coast and in southern Sweden (Skane) using tobacco as an indicator plant. In *Report from the workshop "Ozone effects on vegetation in Europe."* M. Eastmond and L. Skärby Eds. p. 9-12. Swedish Water and Air Res. Inst., Stockholm.
- Skelly, J. M., S. F. Duchelle, and L. W. Kress.
1979. Impact of photochemical oxidant air pollution on eastern white pine in The Shenandoah, Blue Ridge Parkway and Great Smoky Mountains National Parks. *Proc. II Conf. on Sci. Res. in Nat'l. Parks.* San Francisco, CA.
- Skelly, J. M. and J. W. Johnston.
1979. Oxidant air pollution impact to the forests of eastern United States - a literature review 30 p. USEPA Rep. No. EPA-600/3-79-045. Corvallis, OR.
- Stark, R. W., P. R. Miller, F. W. Cobb, Jr., D. L. Wood, and J. R. Parmeter, Jr.
1968. Photochemical oxidant injury and bark beetle (Coleoptera:Scolytidae) infestation of ponderosa pine. I. Incidence of bark beetle infestation in injured trees. *Hilgardia* 39:121-126.
- USEPA.
1977. Photochemical oxidant air pollution effects on a mixed conifer ecosystem - A progress report. 339 p. USEPA Ecol. Res. Series EPA-600/3-77-104. Corvallis, Or.
- USEPA.
1977. International Conference on photochemical oxidant pollution and its control. 1169 p. Vol. I and II. USEPA Ecol. Res. Series EPA-600/3-77-001a, 001b. Res. Triangle Park, N.C.
- Wilhour, R. G. and G. E. Neely.
1977. Growth response of conifer seedlings to low ozone concentrations. In *International Conference on Photochemical Oxidant Pollution and Its Control.* p. 635-645. USEPA Ecol. Res. Series 600/3-77/001b.
- Williams, W. T., M. Brady and S. C. Wilson.
1977. Air pollution damage to the forests of the Sierra Nevada Mountains of California. *J. Air Poll. Contr. Assoc.* 27: 230-234.
- Wolff, G. T., P. J. Lioy, G. D. Wight, R. E. Meyers, and R. T. Cederwall.
1977. An investigation of long-range transport of ozone across the midwestern and eastern United States. *Atmos. Env.* 11:797-802.
- Wood, F. A.
1970. The relative sensitivity of sixteen deciduous tree species to ozone. *Phytopathology* 50:579. (Abstr.).

Primary Productivity, Sulfur Dioxide, and the Forest Ecosystem: an Overview of a Case Study¹

Allan H. Legge²

Abstract: The objective of the West Whitecourt case study was to determine the consequence of chronic long term exposure of a forest ecosystem to low concentrations of sulphur dioxide emissions originating from a "sour gas" processing plant in west central Alberta, Canada. An interdisciplinary ecological approach was utilized. The vegetation and atmospheric environment were characterized. A concept of ecologically comparable sampling site selection was developed and applied in the West Whitecourt study area. Laboratory and field measurements revealed a reduction in photosynthetic rate in lodgepole pine x jack pine (*Pinus contorta* x *Pinus banksiana*) in the field. Reduction of adenosine triphosphate (ATP) concentration in pine tissue during SO₂ fumigation in the field followed by complete recovery after termination of SO₂ fumigation and the disruption of mineral nutrient cycling in the forest ecosystem were observed. Basal area increment measurements of 200 lodgepole x jack pine trees from 5 ecologically comparable sampling sites revealed a decrease in wood production directly related to the presence of sulphur dioxide emissions. It is recommended that the concepts of the assimilatory capacity of the environment for sulphur gas pollutants and irreversible ecological modification be utilized as measures of environmental quality.

Many review articles have been written addressing the problem of air pollutants and forest ecosystems (Tamm and Aronsson, 1972; Smith, 1974; Miller and McBride, 1975; and Linzon, 1978). These reviews documented the extreme examples of acute high concentration long term air pollution stress on ecosystems and were essentially post-mortem studies. Environmental change due to air pollution stress was clearly visible in these cases. With the exception of the San Bernardino Mountain study investigating the effects of photochemical oxidants on a mixed conifer forest ecosystem in California (Miller and others, 1977;

Kickert and Miller, 1979), very little emphasis has been placed upon integrated research programs concerning the impact of chronic long term low concentration air pollution stress on forest ecosystems. The objective of this paper is to present an overview of a four-year integrated forest ecosystem case study designed to determine the consequence of chronic long term exposure of a conifer forest ecosystem to low concentrations of sulphur dioxide emissions originating from a "sour gas" processing plant in Alberta, Canada (Legge and others 1978).

BACKGROUND TO THE CASE STUDY

Potential detrimental environmental impact of sulphur dioxide emissions from the sour gas processing industry upon the environment in Alberta was a major concern of this industry in the early 1970's. This concern led to the formation of the Whitecourt Environmental Study Group (WESG) in 1971, a consortium of eight companies involved

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in the production of saleable natural gas from sour gas (natural gas containing hydrogen sulphide) in the Whitecourt district of west-central Alberta. It was clear at that time that the assessment of the impact of sulphur dioxide on the forest ecosystem was not a simple cause and effect relationship. A five-year environmental research program therefore was initiated in 1972 and was cooperatively funded by both industry and the Alberta Government. The objective of the research program which was called the Whitecourt Environmental Study was to determine the environmental consequences of the operation of sour-gas processing facilities on the forest ecosystem in the Whitecourt district occupied by 11 sour gas processing plants and defined as the Whitecourt study area (figure 1).

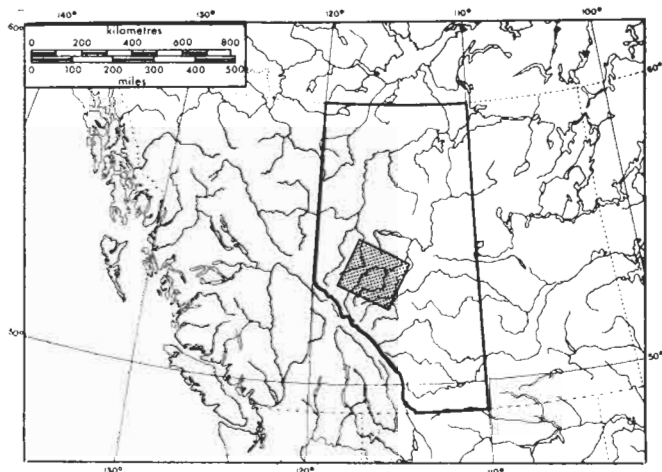


Figure 1. Map of western Canada showing in bold outline the province of Alberta and the Whitecourt Study Area.

The Whitecourt study began in 1972 as a remote sensing airborne environmental survey accompanied by assessment on the ground carried out by INTERA Environmental Consultants Limited to determine if there were any visible large scale environmental disturbances. After two years of general reconnaissance no large scale environmental disturbance was found (Whitecourt Environmental Study 1972 and 1973). Controlled SO_2 fumigation experiments carried out in the laboratory by the Kananaskis Centre for Environmental Research, University of Calgary, on young lodgepole pine (*Pinus contorta* Loud.) seedlings, however, revealed a direct effect of SO_2 on vegetation. These experiments showed that, although plants have the ability to adjust physiologically within certain environmental limits, plants were adversely affected when these limits were exceeded (Legge and Harvey 1974). A conflict therefore arose; the general environmental field survey indicated no large scale modification of the vegetation while the preliminary laboratory

research indicated the potential for environmental change in the field due to sulphur dioxide exposure. To resolve these contradictory research results it became clear that a detailed field case study of a forest ecosystem surrounding a sulphur dioxide source was required.

The AMOCO Petroleum Company Limited West Whitecourt (Windfall) sour-gas processing plant was chosen for the case study. This gas plant had the longest operational history in the Whitecourt study area beginning operation in 1962.

SOURCE OF SULPHUR GAS EMISSIONS

The following is a brief outline of the operation of the West Whitecourt sour-gas processing and sulphur recovery plant to familiarize the reader with the origin of sulphur gas air pollution from the sour-gas processing industry in Alberta.

Hydrogen sulphide is the principle sulphur compound present in sour natural gas and is removed by a series of chemical processes in a sulphur recovery gas plant as elemental sulphur. The H_2S not converted to elemental sulphur is incinerated in a high temperature reaction furnace (580°C) in excess air and methane where it is oxidized to sulphur dioxide (SO_2) and vented to the atmosphere from a tall (122 meter) "candy-striped" incinerator stack. In addition to an incinerator stack, smaller stacks called flare stacks, generally less than 46 meters in height,

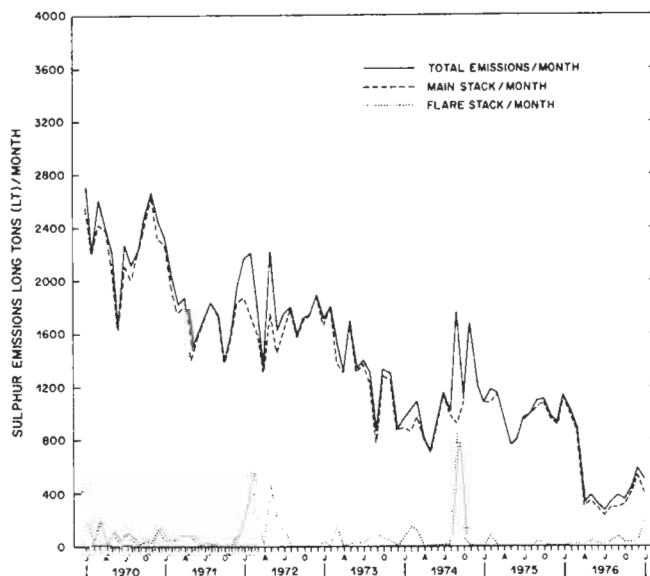


Figure 2. Monthly sulphur emission history of the West Whitecourt Gas Plant from 1970 through 1976.

are used to burn small waste quantities of sulphur recovery gas plant process and compressor gases. Except in the case of a gas plant operational upset, when for short periods the flare stack may contribute more sulphur gas emissions to the atmosphere on a daily basis than the incinerator stack, the incinerator stack is the main source of sulphur gas emissions from the West Whitecourt Plant.

The monthly sulphur emissions (in long tons) of the West Whitecourt Gas Plant for 1970 through 1976 is shown in figure 2; simply double the sulphur emissions to obtain the SO₂ emissions. It is important to note that this gas plant has reduced its sulphur emission output per day an order of magnitude since start-up in 1962 from 150 long tons/day to 18 long tons/day in 1976. This reduction was achieved by enhanced operating procedures and the addition of "tail-gas" recovery units.

SULPHUR: THE NUTRITIONAL CONTROVERSY

Sulphur is an essential nutrient element for normal plant growth and metabolism. It is required in intermediary metabolism and is a constituent of many organic compounds such as amino acids and proteins in plant tissue. Sulphur normally enters the plant via the root system in the form of sulphate, which is biochemically reduced and then converted into numerous organic compounds. Plants, however, can also take-up and utilize SO₂ from the atmosphere via the stomates and utilize it as sulphur source for plant nutrition. Faller (1971), for example, has shown that tobacco plants can utilize SO₂ as a source of sulphur in sulphur deficient soils. This type of information has led many government and industry departments to say that SO₂ emitted from industrial sources is actually beneficial as an aerial fertilizer for plants growing on sulphur deficient soils (Terman 1978; Noggle and Jones 1979). The situation is not quite so simple, however. Though a small amount of atmospheric SO₂ can be nutritional to plants in the short term, the large amount and high frequency of uncontrolled application of sulphur such as SO₂ to an ecosystem by sulphur sources such as smelters, pulp and paper mills, coal-fired power plants, oil sand and oil shale extraction plants and sulphur recovery gas plants can be detrimental in the long term. Different plant species not only have different nutritional requirements for sulphur, but the rate at which plant species assimilate sulphur is influenced by many other variables, such as physiological status, age, time during the growing season, temperature, soil nutrient availability, and light intensity to name a few. When more sulphur is available than can be assimilated it is accumulated in the tissue (Ulrich and others 1967; Legge and others 1977; Cowling and Koziol 1978; Thompson and Kats 1978). This foliar sulphur accumulation can reach toxic levels and adversely affect plant

growth (Katz 1949; Linzon and others 1978). The distinction between the assimilation and the accumulation of sulphur of atmospheric origin by plants must be addressed in any ecosystem study.

CONCEPTUAL APPROACH TO CASE STUDY

To carry out an ecosystem case study one must have a basic understanding of what an ecosystem is. The ecosystem is the basic functional unit of ecology since it includes both the living organism and the non-living environment in which these organisms live. Odum (1971) has defined an ecosystem as "any unit that includes all of the organisms in a given area interacting with the physical environment so that a flow of energy leads to clearly defined trophic structure, biotic diversity and material cycling within the system". Due to the inseparable nature and interdependence of the components of ecosystems upon one another, however, any change that occurs in one component of the ecosystem potentially affects all the components of that ecosystem. The higher the diversity of an ecosystem therefore the more numerous the interrelationships within the ecosystem (Jernelov and Rosenberg 1976). The stability of an ecosystem can be viewed as a function of the balances amongst the components of that ecosystem. An environmental stress such as air pollution can modify the stability of an ecosystem by disrupting the balance amongst the ecosystem components.

Prior to the initiation of the West Whitecourt case study, it was recognized that an ecosystem study was an interdisciplinary undertaking. The term interdisciplinary in this context means the amalgamation of a set of specific disciplinary talents to work together to address a complex environmental problem. An interdisciplinary research team was assembled by the Kananaskis Centre for Environmental Research of the University of Calgary, The University of Alberta, The University of Washington, San Jose State University and the Southern Alberta Institute of Technology. The scientific expertise of the research team was broadly based and ranged from remote sensing, ecology, taxonomy, genetics, plant physiology, analytical chemistry, biochemistry, stable isotope physics and meteorology to statistics and electrical engineering.

A conceptual model was developed to illustrate the dynamic relationship between the sulphur dioxide "source" and the generalized ecosystem "sink" and is shown in figure 3. For purposes of communication among disciplines, the ecosystem was sub-divided into the following four compartments: air, vegetation, soil, and water. The two-way arrows indicate the inter-relationship of the four ecosystem compartments. The expertise of each member of the research team was thus focussed on more than one of the ecosystem compartments at all times. This led to the formulation of cooperatively designed experiments to evaluate not

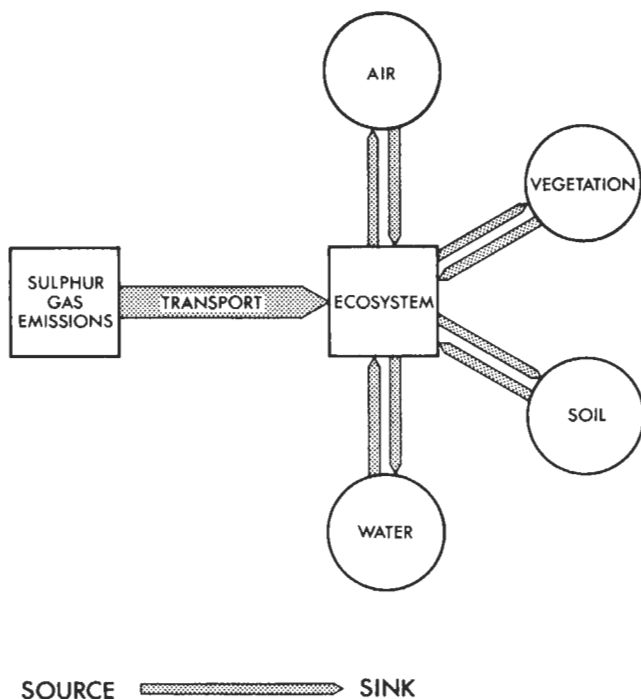


Figure 3. Conceptual model of the forest ecosystem in the West Whitecourt case study area.

only the interfaces between ecosystem compartments but also the processes within the ecosystem compartments. Scientists with different areas of expertise were brought into the case study, however, as a function of the needs of the research program. The selection and the timing of interaction of disciplinary participants were viewed as critically important factors for the success of the program so the case study grew in terms of disciplinary participation from 7 in 1974 to 10 in 1975 and finally to 12 in 1976 and 1977. This evolutionary interdisciplinary approach added a dimension of insight into the fate of sulphur gas emissions in the forest ecosystem that would not have been possible had the separate disciplines of the research team been operating in isolation.

VEGETATION CHARACTERIZATION

The vegetation of the Whitecourt area is included in the predominately forest subregion of the Boreal Forest Region of Canada (Halliday 1937) and is characterized as a transition forest area between the Boreal and Subalpine Forest Regions. The transitional nature of the common species of trees occurring in the area are actually represented by populations of hybrid individuals between lodgepole pine (*Pinus contorta* Loud.) and jack pine (*Pinus banksiana* Lamb.), while the true fir in the area represents hybrids between alpine fir [*Abies lasiocarpa* (Hook.) Nutt.] and balsam

fir [*Abies balsamea* (L.) Mill.] (Legge and others 1977).

A physiognomic classification of the vegetation communities in the West Whitecourt study area identified 13 major cover types out of 24 community types. Ten climax vegetation associations were recognized. The vegetation of the study area was mapped using a combination of LANDSAT imagery, LANDSAT digital data and conventional false colour infrared aerial photography both of which were accompanied by ground based verification. With the aerial colour infrared photography of the West Whitecourt study area as a subsample, computer mapping utilizing LANDSAT digital data was completed on 116,000 hectares to place the study area in a regional perspective. The computer mapping of the study area was eight times faster than conventional photography alone and had a comparative accuracy of 93 percent.

ECOLOGICALLY ANALAGOUS SITES

One of the principle difficulties encountered by air pollution researchers, before collecting samples in the field for analysis, is the selection of sampling locations. Most sampling locations in air pollution studies are chosen solely on the basis of a gradient which is usually a function of the prevailing wind and the distance from a pollution source. Not enough emphasis is placed upon the structure and composition of the plant communities from which samples are taken. Since an ecosystem is composed of many interrelated highly variable biological and physical components, the response of these components to a chronic environmental stress such as air pollution will also be highly variable. If the range of variability of the responses of ecosystem components to an environmental stress is not considered, the expression of the effect of the environmental stress on the ecosystem components may not be detected. There must be a common basis for comparison of ecosystem components therefore to determine both the gross and subtle effects of an environmental stress along a gradient.

The concept and criteria for ecologically analogous sample site selection were developed and applied during the West Whitecourt case study in an attempt to minimize the variability of ecosystem components and hence to minimize the variability of the response of the ecosystem components to air pollution stress. The key to this concept of sample site selection is based upon comparable ecological variability of the ecosystem components and comparable environmental variability at the sampling locations chosen along a distance gradient. The criteria for ecologically analogous sample site selection are summarized in Figure 4. When the ecological and environmental variation of ecosystem components at all the sampling locations are as similar as possible, the sampling sites are said to be ecologically analogous. The major difference amongst the sampling locations

ECOLOGICAL ANALOGUE CONCEPT

ASSUMPTIONS

$$1 = 2 \neq 3$$

1. Ecological Variables (sites $A_1 \rightarrow A_n$)

- slope
- aspect
- soil type
- soil moisture
- species density
- species diversity

2. Environmental Variables (other than pollutants)

- temperature
- wind
- solar radiation
- precipitation

3. Pollutant Variables

- composition
- location
- distance
- concentration/conversion
- frequency/duration

$$[SO_2]_{A_1} > [SO_2]_{A_2} > [SO_2]_{A_3} \\ > [SO_2]_{A_4} > [SO_2]_{A_5} \dots > [SO_2]_{A_n}$$

Figure 4. Summary of criteria utilized for the selection of ecologically analogous sample sites.

therefore is distance from the pollution source. The assumption is that the pollutant variables such as concentration, frequency of fumigation and duration of fumigation will decrease in magnitude with increasing distance from the pollution source. The magnitude of the air pollution stress on ecosystem components at the sampling locations will correspondingly decrease with increasing distance from the pollution source. This procedure ensures that the expression of the air pollution stress on ecosystem components will be maximized.

Meteorological and air quality data were essential in sample site selection. The prevailing winds during the growing season in the study area were shown to have the highest frequency of occurrence from the WNW and the second highest frequency from the ESE. Sulphur dioxide emissions from the West Whitecourt Gas Plant therefore occurred with greatest frequency in an WNW→ESE corridor. Although idealized, the corridor concept provided an essential point of reference for the areas chosen for the selection of sampling locations in the West Whitecourt study area.

The range of distances at which maximum ground level concentrations would occur from sulphur gas emissions from the main incinerator stack and flare stacks was calculated using the simple Gaussian plume model under mean wind conditions and a wide range of stability classes. The maximum ground level concentrations would occur between 1.6 and 34.0 km downwind of the incinerator stack and between 0.4 and 2.6 km

downwind of the West Whitecourt Gas Plant. The intensive experimental site therefore, was principally exposed to sulphur gas emissions from the flare stacks. Only under short-lived meteorological conditions such as would occur during the break-up of an inversion would sulphur gas emissions from the main incinerator stack reach the intensive experimental site.

It must be emphasized at this point that the air quality standards for SO_2 of $0.2 \text{ ppm}/\frac{1}{2} \text{ hr.}/24 \text{ hr.}$ as set by the Alberta Department of the Environment, were only exceeded on three occasions at the intensive experimental site 1.5 km east of the West Whitecourt Gas Plant in over 2500 hours of ambient air monitoring during the 1975 and 1976 growing seasons in the West Whitecourt study area.

Five ecologically analogous lodgepole x jack pine sampling locations were selected in the West Whitecourt case study area. The five sample sites were chosen in locations which were progressively downwind in the main path of sulphur dioxide emission corridor. These sites were chosen in this manner so that the sulphur dioxide emission impact gradient would be very steep across the five sampling locations to maximize the differences in the responses of the ecological analogues to pollution stress. The conceptual ecological model presented in Figure 3 can be generalized to express the relationships amongst the five ecological analogues and is shown in Figure 5. The ecologically analogous sampling

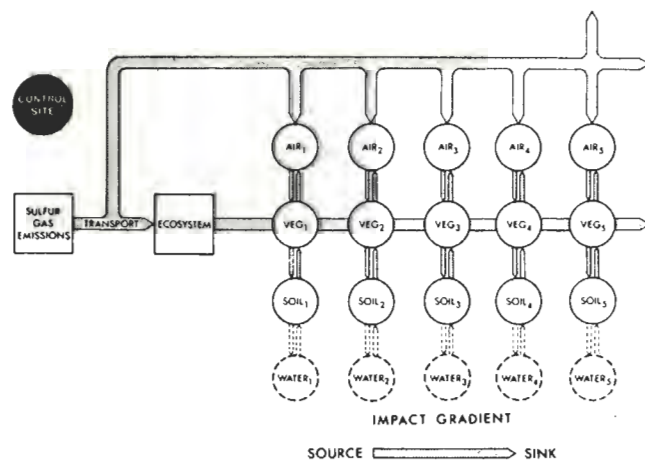


Figure 5. Replicated conceptual ecological model illustrating the five ecologically analogous lodgepole x jack pine sampling locations along a sulphur dioxide concentration gradient.

sites were located at distances of 1.2 km (A_I), 2.8 km (A_{II}), 6.0 km (A_{III}), 7.5 km (A_{IV}) and 9.6 km (A_V) from the Gas Plant. Sampling locations in the study area other than ecological analogues were designated by the letter S.

CASE STUDY DATA OVERVIEW

Foliar sulphate-sulphur concentration in lodgepole x jack pine trees was found to be a better measure of foliar sulphur accumulation than the foliar total sulphur concentration. The method of Johnson and Nishita (1952) was used to determine foliar sulphate-sulphur concentration while a Leco Furnace was used to determine foliar total sulphur. Figure 6 shows a plot of foliar sulphate-sulphur concentration in age-classed lodgepole x jack pine

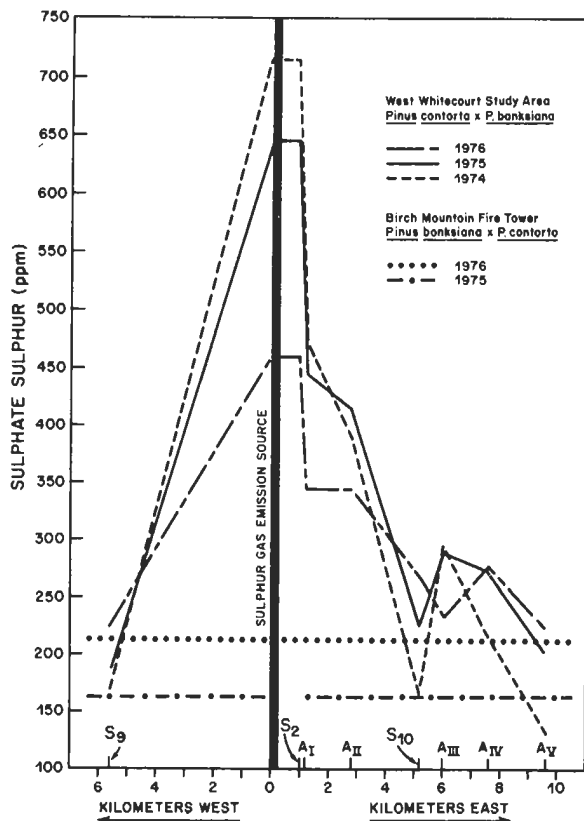


Figure 6. Plot of foliar sulphate-sulphur concentration in age-classed lodgepole x jack pine foliage from eight sampling locations in the West Whitecourt study area as a function of distance from the West Whitecourt Gas Plant.

foliage as a function of distance from the West Whitecourt Gas Plant. The background foliar sulphate-sulphur concentration was determined from age-classed foliage of jack pine x lodgepole pine from the Birch Mountain Fire Tower 111 km (69 mi) NNW of Fort McMurray, Alberta. One can clearly see the decrease in foliar sulphate-sulphur concentration with increasing distance from the sulphur gas emission source. The decrease in foliar sulphate-sulphur concentration with distance was more pronounced with

increasing foliar age. The foliar sulphate-sulphur data strongly support the concept of an environmental stress gradient presented earlier.

The stable sulphur isotopic composition (S^{32}/S^{34} ratio) of sulphur dioxide emissions from sour gas plants in Alberta have been shown to differ from the natural environmental background stable sulphur isotopic composition (Lowe and others, 1971; Krouse, 1977). This is referred to as the $\delta^{34}S$ value. The more positive the $\delta^{34}S$ value the greater the enrichment in S^{34} . The background stable sulphur isotopic composition in the West Whitecourt study area is near 0.

The stable sulphur isotopic composition of sulphur dioxide emissions from the incinerator stack at the West Whitecourt Gas Plant was shown to be $+22.2\text{‰}$ (per thousand). This difference provided an environmental tracer for sulphur of industrial origin.

The mean $\delta^{34}S$ value of 1974-1976 foliage from the ecologically analogous sampling locations A_I through A_V remained close to the mean $\delta^{34}S$ value for the incinerator stack at $+22.2\text{‰}$, while the mean foliar sulphate-sulphur concentration in 1974-1976 foliage decreased from 422 ppm to 185 ppm. These data clearly show that lodgepole x jack pine trees were obtaining some of their sulphur directly from the atmosphere from sulphur gas emissions originating from the West Whitecourt Gas Plant. Needles in the upper crowns of many lodgepole x jack pine trees in the West Whitecourt study area, however, displayed foliar $\delta^{34}S$ values which were greater than those associated with incinerator stack emissions. These data when compared to the laboratory data of Wilson and others (1978) was suggestive of isotopically selective metabolic processes functioning in the lodgepole x jack pine foliage under field conditions.

The 28-meter radio mast tower erected at the intensive experimental site provided a framework for measuring and characterizing SO_2 concentration profiles of ambient air above, within and below the lodgepole x jack pine forest canopy. The vertical SO_2 profiles revealed that the SO_2 concentration minimum measured in the upper crown at 16 meters was primarily due to an aerodynamic effect and was not due to the trees acting as a biological sink. This aerodynamic effect was described as a splitting of the air flow above and below the crowns of the lodgepole x jack pine trees. The measurements of foliar sulphate-sulphur, foliar total sulphur and foliar $\delta^{34}S$ values revealed that the trees, however, were also a biological sink for sulphur gas emissions but that the rate of atmospheric sulphur uptake by the trees was so slow that it was beyond the resolution of the two Thermo Electron Model 43 Pulsed Fluorescent SO_2 analyzers used.

Photosynthetic rates and leaf resistances of lodgepole x jack pine trees in the West Whitecourt study area were shown to be modified. The amount

of this ecological modification was a function of the distance from the West Whitecourt Gas Plant. For example, the seasonal photosynthetic rates of 1976 lodgepole x jack pine foliage were lower and the leaf resistances higher when foliage from S₅ (1.5 km east) was compared with foliage from S₁₀ (5.2 km east) (4.38 ± 1.99 mg CO₂/dry g/hr versus 6.42 ± 1.28 mg CO₂/dry g/hr and 11.3 ± 6.6 s/cm versus 7.8 ± 1.9 s/cm for sample sites S₅ and S₁₀ respectively). This relative reduction in photosynthetic rates, however, was only partially attributable to increased leaf resistance. Additional ecological factors therefore, such as foliar mineral nutrient status and soil pH were considered since these parameters were known to modify plant response.

A detailed analysis of foliar mineral nutrient concentration of N, P, K, Ca, Mg, Mn, Al, Fe and Zn in lodgepole x jack pine foliage from nine sampling locations (including the ecological analogues) in the West Whitecourt study area revealed that the mineral nutrient status of the lodgepole x jack pine trees had been altered. It must be remembered at this point that normal plant growth requires a balance of all essential mineral nutrients within the plant. The foliar concentration of P, K, Fe, Mg, N and Zn tended to increase while the foliar concentration of Ca and Al tended to decrease with distance from the West Whitecourt Gas Plant and distance from the WNW-ESE sulphur dioxide emission corridor. Site type was shown to be a critical factor influencing the concentration of these eight mineral nutrients. Foliar Mn concentration was found to decrease dramatically with distance from the West Whitecourt Gas Plant and distance from the WNW-ESE sulphur gas emission corridor. Variability of site type, however, did not modify this relationship. A low Fe to Mn ratio was found in foliage from sampling locations within 4 km of the West Whitecourt Gas Plant. The low foliar Fe concentration may contribute to the chlorotic appearance of the foliage at these locations. Foliar mineral nutrient analysis of foliage S₂, S₅, A₁ and S₁₀ revealed that foliar K and P were lower in concentration in foliage from S₂, S₅ and A₁ than from S₁₀.

Since foliar K concentration has been linked with stomatal activity, the reduced foliar K concentration may be inhibiting stomatal opening thus increasing leaf resistance which would then limit photosynthetic rate. Reduced foliar P concentration may inhibit phosphorylation and thereby also limit photosynthetic rate. Foliar nutrient deficiencies of either P and K alone or in combination therefore may be partially responsible for the reduced photosynthetic rates observed in lodgepole x jack pine foliage in the West Whitecourt study area.

The alteration of foliar mineral nutrient status in lodgepole x jack pine trees in the West Whitecourt study area therefore is an important ecological factor contributing to the modification of plant response.

Soil pH profiles were measured at the same nine vegetation sampling locations where foliar mineral nutrient concentrations were determined since soil pH is known to affect the availability of mineral nutrients to plants. The general trend or gradient in soil pH over all nine sampling locations was an increase in soil pH with depth and with distance from the West Whitecourt Gas Plant and distance from the WNW-ESE sulphur gas emission corridor. The soil pH gradient was most striking when only the ecologically analogous sampling locations were considered. A direct relationship was found between lowered soil pH and the elevated levels of foliar Mn in lodgepole x jack pine trees. The foliar Mn concentration data and the soil pH data suggest that foliar Mn concentration in lodgepole x jack pine trees could be used as a mineral nutrient indicator of modification of the forest ecosystem by sulphur gas emissions.

Soil total sulphur concentration in the soil profiles at the nine soil sampling locations also decreased with soil depth, distance and direction from the West Whitecourt Gas Plant. It is important to note, however, that there was no correlation between soil total sulphur concentration and soil pH. There was also no direct correlation between a given soil pH value and the soil $\delta^{34}\text{S}$ value. These data suggest that the soil $\delta^{34}\text{S}$ value can be used as an indicator of the presence and penetration of sulphur gas emissions into the soil profile while soil pH and soil total sulphur can be used as indicators of sulphur loading of the soil.

In terms of plant biochemistry and sulphur gas emissions the most significant observation in the field was a transient metabolic effect; the ATP (adenosine triphosphate) concentration of foliage cells from lodgepole x jack pine trees was found to be directly decreased upon exposure to low concentration short duration SO₂ fumigation (75% decrease upon fumigation with 0.14 ppm SO₂ for 15 minutes). It is important to note that when the foliage was no longer exposed to SO₂ the foliar ATP concentration increased to the pre-SO₂ fumigation ATP concentration (see Harvey and Legge, 1979, for details). This decrease and increase in the foliar ATP concentration was also observed with excised lodgepole x jack pine branches from the West Whitecourt study area which were fumigated under controlled conditions at the Kananaskis laboratory. When lodgepole x jack pine trees, which had been grown in the absence of sulphur gas emissions in the laboratory, were fumigated with SO₂ no fluctuation in ATP concentration was observed. It is important to note that the laboratory trees had a foliar ATP content which was over twice the foliar ATP content of the lodgepole x jack pine foliage from the West Whitecourt study area (658 nmoles/dry g versus 1460 nmoles/dry g). The lower foliar ATP concentration of field grown trees compared to laboratory grown trees suggests a partial explanation for the lowered photosynthetic capacities reported for lodgepole x jack pine trees in the West Whitecourt study area.

The photosynthetic rate of lodgepole x jack pine trees grown under controlled conditions in a non-SO₂ environment in growth chambers at the Kananaskis laboratory and lodgepole x jack pine trees grown in the sulphur dioxide emission environment in the field at the intensive West Whitecourt study site S₅ was measured when both sets of trees were exposed to similar low concentration short-duration SO₂ fumigations. The photosynthetic rate of the field grown plant material was not depressed by SO₂ fumigation while the photosynthetic rate of the laboratory grown material was depressed by the SO₂ fumigations. The photosynthetic rate of the laboratory trees was much greater than the trees in the field. There was also no evidence of a plant-water deficit in lodgepole x jack pine trees severe enough to effect photosynthetic rate.

Adenosine triphosphate is the major biochemical intermediate of energy transfer in biological systems. A decrease in foliar ATP content in lodgepole x jack pine trees caused by SO₂ fumigation therefore would also be a decrease in the amount of biochemical energy available for normal metabolic functions. Although the ATP content of the lodgepole x jack pine foliage recovered to the pre-SO₂ exposure concentration after the SO₂ stress was removed, during the SO₂ fumigation there would have been a net loss of biochemical energy. The fact that foliar ATP content increased after SO₂ fumigation, indicated that the trees were coping with the sulphur gas emissions at the cost of a metabolic energy drain.

In summary, the contrast in the biochemical and physiological responses of the lodgepole x jack pine trees fumigated with SO₂ in the field and the laboratory strongly indicates that environmental pre-history and acclimation of the trees to ecological modification of components of the forest ecosystem are the critical factors determining plant response to sulphur gas emissions in the West Whitecourt study area.

A comparison of the mean photosynthetic capacities of 1976 lodgepole x jackpine foliage through the 1976 growing season at sampling locations S₂, S₅ and S₁₀ revealed that S₁₀ had a positive net CO₂ fixation balance two to three weeks prior to the foliage at S₂ and S₅ and is shown in Figure 7. The mean photosynthetic capacity of 1976 S₁₀ foliage was also always higher than the mean photosynthetic capacity measured for 1976 foliage from S₂ or S₅. Additionally in terms of photosynthetically active needle biomass lodgepole x jack pine branches sampled at S₁, S₂, A_I and S₅ were chlorotic in appearance with premature abscission (needledrop) of the third year needles and poor leader growth while branches sampled at S₉ and S₁₀ were comparatively darker green in color with a needle retention of from four to six years and good leader growth. The photosynthetic potential of lodgepole x jack pine trees based upon needle biomass alone therefore was much greater at S₉ and S₁₀ compared to S₁, S₂, A_I or S₅.

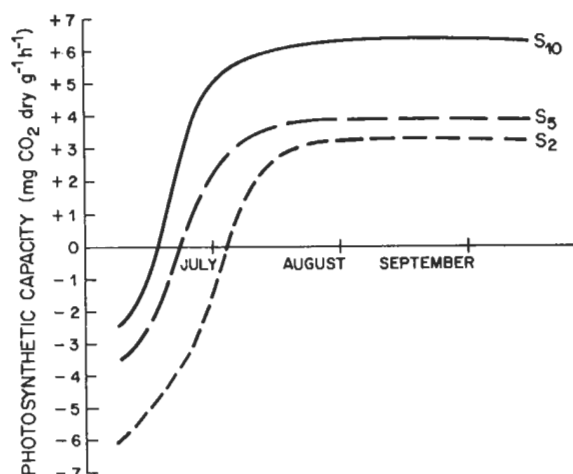


Figure 7. Comparison plot of the mean photosynthetic capacity of 1976 foliage on lodgepole x jack pine branches from sampling sites S₂, S₅ and S₁₀ from early in June to mid-September.

When the observed ecological modifications of the forest ecosystem such as reduced needle biomass, reduced biochemical energy, reduced photosynthetic rates, reduced soil pH, the disruption of mineral nutrient cycling, foliar sulphur loading and the shortened growing season are combined, and considered in the long term time sense the net effect should be measurable as a reduction in forest productivity. This decrease in forest productivity, however, would be expected to decrease with increasing distance from the sulphur gas emission source.

Annual basal area increment measurements were taken from 40 lodgepole x jack pine trees at each of the five ecologically analogous sampling locations A_I through A_V in 1976 to determine if modification of the forest ecosystem was significant enough in the long term sense to be reflected in reduced wood production since the initial start-up of the West Whitecourt Gas Plant in 1961-1962. Statistical analysis of the basal area increment data shown in Figure 8 revealed that distance from the West Whitecourt Gas Plant, time in years, and their interaction had significant effects on the basal area increment of the lodgepole x jack pine trees from the five ecological analogues. An exponential growth curve model was determined for lodgepole x jack pine trees from A_V and the growth curve of the lodgepole x jack pine trees from A_I, A_{II}, A_{III} and A_{IV} were statistically compared to it. The basic underlying assumption was that sulphur gas emissions had not had a significant effect on the growth of the trees at A_V. This analysis statistically revealed that there has been a definite reduction in basal area increment in lodgepole x jack pine trees since 1962 in A_I, A_{II}, A_{III} and A_{IV} compared to the basal area increment model for A_V which was attributable to sulphur gas emissions from the West Whitecourt Gas Plant.

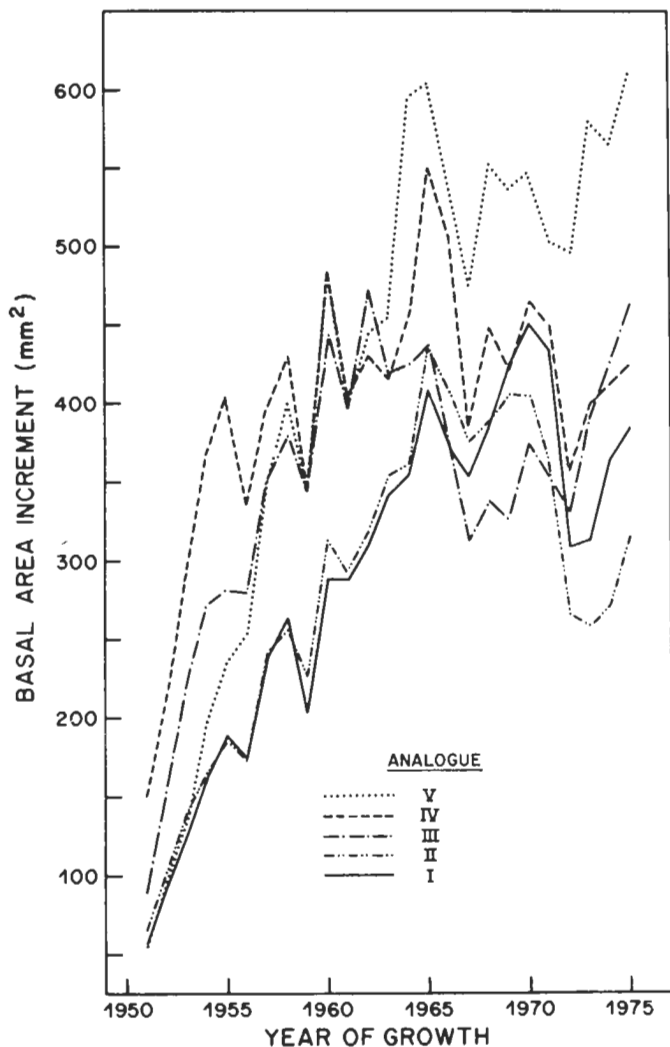


Figure 8. Comparative plots of the mean basal area increments from 40 lodgepole X jack pine trees at each of the five ecologically analogous sampling sites in the West Whitecourt study area.

The maximum reduction in basal area increment occurred at A_I and A_{II} and progressively decreased to zero at A_V . The effect of sulphur gas emissions on basal area increment growth in lodgepole x jack pine trees since 1961 at A_I , A_{II} , A_{III} , and A_{IV} relative to A_V is thus a gradient with the reduction in basal area increment resulting from sulphur gas emission falling to zero at A_V or 9.6 km. If the total basal area increment reduction of A_I relative to A_V is averaged over the fourteen years since the start-up of the West Whitecourt Gas Plant, this would correspond to approximately a one to two percent reduction in basal area increment growth of A_I relative to A_V .

The areal extent of possible modification of components of the forest ecosystem by sulphur gas emission from the West Whitecourt Gas Plant can be estimated using the following assumptions:

tions:

1. sulphur gas emissions reach the forest ecosystem within 17 km. (10.6 mi.) of the source; and
2. the impact of sulphur gas emissions is restricted to areas NW and SE of the West Whitecourt Gas Plant.

The area affected by sulphur dioxide emissions, therefore, is approximately 454 km² (175 mi²) or 45,373 hectares (112,130 acres). This areal extent estimate of impact is conservative because the 17 km distance is only one-half the distance range calculated using the simple Gaussian plume model under all stability classes for maximum ground level concentration of sulphur dioxide emissions from the main incinerator stack.

It is important to bear in mind at this point, however, that this projected impact area has not been uniformly modified by sulphur gas emissions but rather has been modified in terms of an impact gradient extending NW and SE from the West Whitecourt Gas Plant. In other words, the extent of ecosystem component modification will decrease with distance from the sulphur gas emission source.

Another factor must be considered at this point. Sulphur emissions from the West Whitecourt Gas Plant have been reduced almost an order of magnitude since 1970 (refer to Figure 2). This significant reduction in emissions will not only generally decrease the magnitude of the impact of sulphur emissions on the forest ecosystem, it will also decrease the areal extent of the area impacted by sulphur emissions in the past thus allowing a portion of the forest ecosystem to recover from the previous sulphur gas emission stress.

When one uses foliar sulphate-sulphur concentration in lodgepole x jack pine trees as a measure of sulphur accumulation from exposure to the current level of sulphur gas emissions, it appears that a tolerable concentration is reached by 9-12 km (5.6-7.5 mi) of the West Whitecourt Gas Plant. This is indicated by a decrease in foliar sulphate-sulphur concentration with needle age which is within the range of the background foliar sulphate-sulphur concentration. The foliar $\delta^{34}\text{S}$ values, however, could be used to provide a more exact measure of the distance at which the presence of sulphur gas emissions become negligible to components of the forest ecosystem.

CONCLUSION

It is clear from this case study that sulphur gas emissions from the West Whitecourt Gas Plant have modified the forest ecosystem in a number of ways. The main ecological process which has been directly and indirectly affected by sulphur dioxide emissions is mineral nutrient cycling. By progressively altering the mineral nutrient balances of ecosystem components for example, the biological relationships amongst the components and the physiological and biochemical functions of the components are modified. It is these ecosystem

component modifications which are the expressions of environmental deterioration resulting from chronic exposure to sulphur dioxide over time. Despite this measurable deterioration of the forest ecosystem, however, it does not appear at this time that sulphur dioxide emissions from the West Whitecourt Gas Plant have caused irreversible ecological degradation. With the significant reduction in sulphur emissions from the West Whitecourt Gas Plant (See Figure 2) it is not anticipated that there will be significant irreversible ecological modification of the forest ecosystem in the remaining 10 to 20 years of operation of the West Whitecourt Gas Plant.

One philosophical dilemma has resulted from the West Whitecourt case study. There is no relationship between air quality standards and the maintenance of environmental quality since the term environmental quality excludes environmental modification. No effort to date has been made to address or to quantify acceptable limits of environmental modification resulting directly or indirectly from air pollution stress despite the fact that the presence of air pollutants in the atmosphere implies that a certain amount of environmental modification is acceptable.

Since, at the present time it is technologically and economically impossible to remove all air pollutants from industrial processes, it is suggested that irreversible ecological modification of the environment be used as an additional criteria for limiting pollutant emissions to the atmosphere. The assimilatory capacity of the environment, in other words, must be taken into account by both industry and regulatory agencies. The uniform application and enforcement of fixed air quality standards over a geographical area the size of the province of Alberta (661,183 km² or 255,285 mi²) with its physiographically complex terrain, heterogeneous vegetation and diverse climatology is clearly not enough to maintain environmental quality. Future research will be required to determine the assimilatory and accumulatory capacity of the environment to pollutants and to provide the biological monitoring techniques to assure that the assimilatory and accumulatory capacity of the environment is not exceeded.

After the assimilatory and accumulatory capacity of the environment have been considered, flexible air quality standards may be possible. These standards could be adjusted regionally and seasonally in order to minimize pollutant impact on the environment. All emission sources, however, would have to be viewed in the context of their regional location, projected longevity of their operations, composition of their emissions, the proximity of neighboring emission sources as well as regional land use priorities since it is the total pollutant load to the environment which must be considered when one uses assimilatory capacity as a measure of environmental quality.

The conceptual interdisciplinary nature of the West Whitecourt case study has proven to be the basis for the success in unravelling the very complex interrelated consequences of the chronic exposure of the forest ecosystem to sulphur gas emissions from the West Whitecourt Gas Plant. It is suggested that future air pollution research on forest ecosystems follow a similar experimental design if the environmental perturbations caused by air pollution stress are to be understood. Direct extrapolation of the data summarized in this paper to other areas would be misleading unless local environmental factors, vegetation and pollutant parameters, are taken into consideration prior to interpretation.

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LITERATURE CITED

- Cowling, D. W., and M. J. Koziol.
1978. Growth of Ryegrass (*Lolium perenne* L.)
I. Effects on photosynthesis and respiration. *Journal of Experimental Botany* 29(112):1029-1036
- Faller, N.
1971. Plant nutrient sulphur- SO₂ vs. SO₄.
The Sulphur Institute Journal 7(2):5-6

- Halliday, W. E. D.
1937. A forest site classification for Canada. Canadian Forest Service Bulletin 89. 50 p.
- Harvey, G. W., and A. H. Legge.
1979. The effect of sulfur dioxide upon the metabolic level of adenosine triphosphate. Canadian Journal of Botany 57(7):759-764.
- Jernolov, A., and R. Rosenberg.
1976. Stress tolerance of ecosystems. Environmental Conservation 3:43-46
- Johnson, C. M., and H. Nishita.
1952. Microestimation of sulfur in plant materials, soils and irrigation waters. Analytical Chemistry 24(4):736-742.
- Katz, M.
1949. Sulfur dioxide in the atmosphere and its relation to plant life. Industrial and Engineering Chemistry 41(11):2450-2465.
- Kickert, R. N., and P. R. Miller.
1979. Responses of ecological systems. In Handbook of methodology for the assessment of air pollution effects on vegetation. W. W. Heck, S. V. Krupa, and S. N. Linzon, eds. p. 14-1 to 14-45. Air Pollution Control Association.
- Krouse, H. R.
1977. Sulphur isotope abundances elucidate uptake of atmospheric sulphur emissions by vegetation. Nature 265:45-46.
- Legge, A. H., and G. W. Harvey.
1974. Sulphur dioxide and environmental preconditioning of vegetation. In Whitecourt Environmental Study Report 1974. Section IV 1-29. Whitecourt Environmental Study Group AMOCO Canada Petroleum Company Limited, Calgary, Alberta.
- Legge, A. H., D. R. Jaques, R. G. Amundson, and R. B. Walker.
1977. Field studies of pine, spruce, and aspen periodically subjected to sulphur gas emissions. Water, Air, and Soil Pollution 8(1):105-129.
- Legge, A. H., D. R. Jaques, H. R. Krouse, H. M. Brown, E. C. Rhodes, H. U. Schellhase, J. Mayo, A. P. Hartgerink, P. F. Lester, R. G. Amundson, G. W. Harvey, and R. B. Walker.
1978. Sulphur gas emissions in the boreal forest: the west whitecourt case study. II. final report. Submitted to the Whitecourt Environmental Study Group. Kananaskis Center for Environmental Research Report Number 78-10. 615 p.
- Linzon, S. N., P. J. Temple, and R. G. Pearson.
1978. Sulfur concentrations in plant foliage and effects. Presented at the 71st Annual Meeting of the Air Pollution Control Association, Houston, Texas June 25-30, 1978. 15 p.
- Linzon, S. N.
1978. Effects of airborne sulfur pollutants on plants. In Sulphur in the environment. part II: ecological impacts. J. O. Nriagu ed. p. 110-162. John Wiley & Sons, N. Y.
- Lowe, L. E., A. Sasaki, and H. R. Krouse.
1971. Variations in sulphur-34: sulphur-32 ratios in soil fractions in western Canada. Canadian Journal of Soil Science 51:129-131.
- Miller, P. R., and J. R. McBride.
1975. Effects of air pollutants on forests. In Responses of plants to air pollution. J. B. Mudd and T. T. Kozlowski, eds. p. 192-235. Academic Press, New York.
- Miller, P. R., R. N. Kickert, O. C. Taylor, R. J. Arkley, F. W. Cobb Jr., D. L. Dahlsten, P. J. Gersper, R. F. Luck, J. R. McBride, J. R. Parmeter Jr., J. M. Wenz, M. White, and W. W. Wilcox.
1977. Photochemical oxidant air pollutant effects on a mixed conifer forest ecosystem. A progress report 1976. Ecological Research Series EPA-600/3-77-104. U. S. Environmental Protection Agency, Corvallis, Oregon. 339 p.
- Noggle, J. C., and H. C. Jones.
1979. Accumulation of atmospheric sulfur by plants and sulfur-supplying of soils. Interagency Energy/Environment R & D Program Report. TVA/ONR-79/10. EPA-600/7-79-109. U. S. Environmental Protection Agency. Washington, D.C. 37 p.
- Odum, E. P.
1971. Fundamentals of ecology. W. B. Saunders Company, Toronto. 574 p.
- Smith, W. H.
1974. Air pollution-effects on the structure and function of the temperate forest ecosystem. Environmental Pollution 6:111-129.
- Tamm, C. O., and A. Aronsson.
1972. Plant growth as affected by sulphur compounds in polluted atmosphere. A literature survey. Research Notes Number 12. Department of Forest Ecology and Forest Soils. Royal College of Forestry. Stockholm, Sweden. 53 p.
- Terman, G. L.
1978. Atmospheric sulphur-the agronomic aspects. Technical Bulletin Number 23. The Sulphur Institute. Washington, D.C. 15 p.

Thompson, R. C., and G. Kats.

1978. Effects of continuous H₂S fumigation on crop and forest plants. Environmental Science and Technology 12(5):550-553.

Ulrich, A., M. A. Tabatabai, K. Ohki, and C. M. Johnson.

1967. Sulfur content of alfalfa in relation to growth in filtered and unfiltered air. Plant and Soil 26(2):235-252.

Wilson, L. G., R. A. Bressan, and P. Filner.

1978. Light dependent emission of hydrogen sulfide from plants. Plant Physiology 61:184-189.

Effects of Airborne F on Forest Ecosystems¹

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Abstract: Although there are many reports of fluoride (F) injury to forests, there have been no systematic studies on the forest ecosystem. In this paper, we have reviewed the present state of our knowledge on F pollution and effects on physiological processes, tree growth, F accumulation and plant injury, community structure, interaction with pathogens and insects, and distribution of F in the environment. The preparation of this review indicated the many areas of the F-plant interaction on which there is no information, where it is poorly understood, or where available information is highly controversial. In some cases, we have joined the controversy.

Injury to forest tree species by airborne fluoride (F) has been reported in many parts of the world (e.g., Adams and others, 1952; Horntvedt and Robak, 1975; Niklfeld, 1975; EPA, 1973; Flühler and others, 1979), but many F-emitting sources are in agricultural or urban areas and reports of injury to agronomic crops, ornamental and urban trees (e.g., Bolay and Bovay, 1965; Facticeau and Mellenthin, 1976; de Ong, 1946; Leonard and Graves, 1966) or on fluoride accumulation and production of fluorosis in livestock and other herbivores (Suttie, 1977) are also common.

The principal industrial sources of airborne F are primary aluminum smelting; steel manufacture; conversion of fluorapatite to phosphate and phosphorus; and glass, ceramic and brick production. Natural sources of airborne F are principally from soil particles, fumaroles, and volcanoes. The ash from the recent eruption of Mount St. Helens contained 8 ppm soluble F (Stoiber and others, 1980) and 400 ppm total F and its impact on forests in the northwestern U.S. will be

watched carefully.

Results of only a few field studies made near F-emitting sources are available in the scientific literature. One reason for the absence of more reports is that the studies were often routine and not quantitative, making publication in refereed journals difficult. A second reason is that results of a study performed for an industry may be sequestered from publication or other use because of active, pending, or potential litigation. Often, field studies that have been distributed were in a form that was not subjected to peer review, was carelessly assembled, and/or reflected the personal biases of the authors.

Because of these problems, we have not confined this review to works published in journals, but we have tried to judge the reports that we have cited in terms of their pertinence and/or availability, and our personal views are often presented. Most internal reports were avoided, but the lack of published information often left no recourse but to cite them. We hope that we have stated our criticisms of some studies as fairly as possible.

LABORATORY STUDIES ON PLANT PRODUCTIVITY

The productivity of the plant depends upon the coordination and rate of CO₂ assimilation, respiration, transpiration, translocation of photosynthate, mineral nutrition, growth, and reproduction. The amount of information available on

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the impact of airborne F on these processes ranges from virtually none, e.g., on translocation of photosynthates, to a moderate amount, e.g., on CO₂ assimilation (apparent photosynthesis) and respiration.

There are few data on the chemical composition, distribution, and patterns and frequency of exposure of atmospheric F in the field. One reason for this is that air monitors with short averaging times have not been generally available. Because F concentrations in the ambient air have not been characterized and fluctuating F concentrations are difficult to control, the design of meaningful laboratory or controlled field experiments is formidable. The information that is available is generally for averaging times of 12 or 24 hours (McCune and others, 1976) and the peak F concentrations that occur near sources are not known. This information would be very useful since it has been shown for other air pollutants, such as SO₂, that short-term peak exposures are more important in explaining plant damage than average concentrations (McLaughlin and others, 1979). Unfortunately, most laboratory studies have employed continuous exposures at constant concentrations that do not simulate field exposures. For these reasons the data available are of limited value in predicting the impact of airborne F on forest ecosystems.

Gas Exchange

Apparent Photosynthesis

Given the problems outlined above, it is not surprising that there have been so few studies on the effects of F on apparent photosynthesis (AP) of forest tree species (Table 1). Consequently, we have included in Table 1 not only studies on forest trees but also those on horticultural species exposed to hydrogen fluoride (HF) or supplied with sodium fluoride (NaF). We have arbitrarily separated experiments with HF into acute exposures (over 10 $\mu\text{g m}^{-3}$ for a few days or less) and chronic exposures (ca. 5 $\mu\text{g HF m}^{-3}$ or less for a few days to more than a growing season), although we recognize that many exposures classified as chronic could more realistically be classified as acute. Exposures to solutions containing NaF have varied from several hours to months and will be discussed individually.

Acute exposures -- With the exception of cotton, where high concentrations of HF had no effect (Thomas, 1958), acute exposures have consistently reduced AP (Thomas and Hendricks, 1956; Thomas, 1958; Hill, 1969; Bennett and Hill, 1973). Bennett and Hill (1973) exposed alfalfa to HF for 2 hours and found that (1) approximately 120 $\mu\text{g m}^{-3}$ HF were needed to produce foliar necrosis; (2) about 40 $\mu\text{g m}^{-3}$ were necessary to clearly inhibit AP; (3) the depression of AP and subsequent recovery after exposure were slower for HF than for the other major air pollutants tested (SO₂, O₃, NO₂, NO and Cl₂). They also noted that of the pollutants tested, HF produced the greatest reduction

in AP for an equivalent pollutant dose, but stated that the occurrence in the field of a concentration that would produce a 10 percent reduction in AP would be rare. One can conclude that a significant influence of an acute exposure on plant community productivity would likely be preceded by lesions and abscission of foliage.

Chronic exposures -- Several investigators have reported that chronic exposure to HF had no effect on AP if there was no visible injury (Hill, 1969; Hill and others, 1958; Thompson and others, 1967), and when foliar injury occurred, the reduction in AP was proportional to (Hill, 1969; Thomas and Hendricks, 1956; Thomas, 1958) or greater than the amount of foliage injured (Thomas, 1958, for fruit trees; and Woltz and Leonard, 1964, for citrus). Thomas (1958) proposed that there is a threshold of F concentration and duration of exposure for each species above which AP is reduced more than can be accounted for by chlorosis and necrosis.

McCune and others (1976) described experiments in which field-grown sorghum was exposed for 14 days to three concentrations of HF (0.7, 1.7 and 3.5 to > 5 $\mu\text{g m}^{-3}$) and AP of the whole plant canopy was measured three times daily before, during, and after the exposure periods. The lowest HF concentration had no effect on AP; the intermediate concentration reduced AP during the exposure period, but immediate recovery occurred upon cessation of the exposures. Plants subjected to the highest concentration also had reduced rates of AP for the first week. But when the HF concentration was raised to greater than 5 $\mu\text{g m}^{-3}$ on the eighth day, severe foliar injury occurred, the rates of AP dropped drastically, and there was no recovery in the post-exposure period.

In an extensive series of experiments, Keller (1977) placed 11 different tree species (see Table 1) at varying distances from a source of airborne F for several months and measured rates of AP on the whole plants returned to the laboratory. Exposure to F produced foliar injury and abscission, and reduced the rate of AP of the whole plant. The reduction in AP of the whole plant was due primarily to the loss of foliage, because the rate of AP of needles remaining on the plants was as high as those on control plants.

Sodium fluoride -- Navara (1963) reported both depression and stimulation of AP of beans grown in solution culture for 16 days with 0.03 or 0.3 ppm NaF, while those supplied with 3 ppm had depressed rates of AP. When *Picea excelsa* Link. cuttings were watered periodically through the winter and spring with deionized water containing 100 ppm NaF, the AP rates were not only reduced but necrosis was produced on the newly flushed foliage. The F concentrations in the new foliage that exhibited injury contained only 3.7 to 8 ppm when injury first occurred. By the end of July, those needles that survived contained from 31.5 to 52.2 ppm F (Keller, 1980).

McLaughlin and Barnes (1975) exposed cut branchlets of three pine species and leaves of six deciduous trees to 0, 1.9, 19, and 190 ppm NaF for 24 hours and then measured the rates of AP. With 1.9 ppm NaF, the rates of AP of older needles of *Pinus taeda* L. and *P. echinata* Mill. were reduced while the other species were unaffected (see Table 1 for species used). Needles with reduced rates of AP contained less than 10 ppm F. Although low concentrations of foliar F reduced AP and stimulated respiration, the authors warn of the limitations of extrapolating laboratory data to the field situation. However, their data raised several questions: (1) What concentration of HF would be necessary to increase the foliar F concentration 4-8 ppm in a 24-hour period as did the comparable dose of NaF? (2) If upon exposure to HF, a branch on a tree accumulated F at the same rate, would the reduction in AP be permanent or would it recover to the pre-exposure rate after the exposure? (3) Would the 4-8 ppm increase in foliar F associated with the reduction in AP produce visible injury? One could view this kind of exposure as acute, because the comparable dose of HF to accumulate this amount of F in 24 hours could be 4-8 $\mu\text{g m}^{-3}$, assuming an accumulation coefficient of 1 ppm $\mu\text{g}^{-1} \text{m}^3 \text{day}^{-1}$.

Respiration

Respiration (measured as oxygen uptake) was stimulated in intact plants (Applegate and Adams, 1960a; Applegate and others, 1960) or in tissues from intact plants fumigated with HF, in the absence (Weinstein, 1961; Applegate and Adams, 1960b; Yu and Miller, 1967; Miller and Miller, 1974) or presence of foliar lesions (Hill and others, 1959). Fluoride inhibition of oxygen uptake has also been reported and was dependent on plant tissue age (Bejaoui and Pilet, 1975) duration of exposure (Applegate and Adams, 1960a), nutrient status (Applegate and Adams, 1960b), and tissue F concentrations (Applegate and others, 1960). However, the rate of respiration of some tissues is relatively insensitive to F (Hill and others, 1959; Givan and Torrey, 1968). In their experiments with cut branchlets of pines and hardwoods supplied with 1.9, 19 or 190 ppm NaF in solution for 24 hours, McLaughlin and Barnes (1975) found that the lower two concentrations generally stimulated respiration (measured as CO_2 evolution) while the highest concentration both stimulated and inhibited respiration, depending on the species.

Transpiration and Water Use

There are few reports on the effects of F on transpiration. Navara (1963) grew beans in solution cultures containing 0.03, 0.3, and 3.0 ppm F and found reduced rates of transpiration after 12 and 16 days in plants supplied with the two highest concentrations of F. Soybeans fumigated with 12 $\mu\text{g m}^{-3}$ HF had dramatically reduced rates of transpiration within 4 hours (Poovaiah and Wiebe, 1973). This agrees with Navara and Kozinda (1967) who found similar results in bean

and apricot exposed to 70 $\mu\text{g m}^{-3}$ HF. But Thompson and others (1967) did not find significant differences in water use of citrus exposed over a growing season to either ambient levels of F or filtered air with added F (both < 0.5 $\mu\text{g m}^{-3}$) compared to control plants. Amundson and others (in review) exposed corn to 1.5 $\mu\text{g m}^{-3}$ HF continuously for one week and found an increased rate of transpiration over controls. The data available on F effects on transpiration are variable and insufficient to predict possible effects on plant community water relations. However, since F can elicit changes in stomatal aperture, these effects may be important in a forest ecosystem where water deficits limit AP at certain times of the day or year (Larcher, 1975; Kramer and Kozlowski, 1979).

Plant Metabolism

F has long been used as a metabolic inhibitor and the list of published reports of F effects on enzyme systems and metabolic processes is extensive. Many of the effects of F on plant metabolism have been reviewed (McCune and Weinstein, 1971; Chang, 1975) and Horsman and Wellburn (1976) have compiled a useful list of F-induced metabolic responses.

Since F alters normal plant metabolism, efforts have been made to identify metabolites that could be used as indicators of incipient F injury. Yee-Meiler (1975) found that non-specific esterase activity in young Norway spruce (*Picea abies* [L.] Karst.) and European white birch (*Betula verrucosa* Ehrh.) exposed to airborne F was increased late in the growing season without the appearance of injury symptoms. Needles of conifers placed at varying distances from an industrial F source had significant increases in phenols if they came from trees with F injury (Yee-Meiler, 1977). The results were variable for deciduous trees. Keller and Schwager (1971) found increased peroxidase activity in leaves of seven tree species exposed to an industrial source of HF and noted that the enzyme activity increased before or in the absence of development of F injury symptoms. Unfortunately, many environmental stresses and laboratory manipulations can increase peroxidase activity, limiting the usefulness of this assay (Endress and others, 1980).

Mineral Nutrition

Wide differences in the response of peaches to HF led to the first study of the influence of mineral nutrition on HF susceptibility (Brennan and others, 1950). With low or deficient amounts of N, Ca, and P in tomato foliage, there was reduced uptake of NaF by roots or HF by leaves; similar results were found with excessive amounts of N and Ca (Brennan and others, 1950). Other studies have resulted in increased foliar F in P-, K-, or Fe-deficient beans (Applegate and Adams, 1960c); reduced foliar F in Mg-deficient tomato plants (MacLean and others, 1969); smaller fruits in Ca-deficient tomato plants (Pack, 1966); and increased tolerance to HF exposure in tomato plants

Table 1. Reported effects of fluoride (HF and NaF) on apparent photosynthesis (as measured by changes in CO₂ uptake) of higher plants.

Genus or Species	Concentration μg m ⁻³ HF	Duration	Response	Reference
Gladiolus	0.8 - 8.0	7 days	pct. reduction in AP = pct. injury	Thomas & Hendricks 1956
Hordeum	32	4-8 hours	total interruption in AP with recovery in few hours to days	
Medicago	200	4-8 hours		
Fruit trees	16-40	4-8 hours		
Hordeum	32	2 hours	AP reduced during exposure with recovery after exposure	Bennett & Hill 1973
Medicago	32	2 hours		
Lycopersicon	1.4 - 5.2	4 weeks	no effect	Hill & others 1959
	0.9 -11.2	3 weeks	no effect	
Fruit trees	2.1 av.	183 hours	14 pct. reduction in AP 10 pct. injury	Thomas 1958
Gladiolus	3.1 - 5.2	30-205 days	pct. reduction in AP = pct. leaf injury	
Gossypium	13.6	138 hours	no effect	
Citrus	.32 -.77	growing season	no effect	Thompson others 1967
Gladiolus	0.8	39 days	no effect	Hill 1969
	1.2	27 days	3 pct. reduction over injury	
Fragaria	2.3	63 days	no effect	
	38	1 day	50 pct. reduction	
Lycopersicon	5.1/12	17/21 days	no effect	
Prunus	1.6	42 days	no effect	
Zea	7.7	16 days	no effect	
Sorghum	0.7	14 days	no effect	McCune & others 1976
	2.2 then 1.7	12/2 days	reduced with recovery after exposure	
	3.5 then 5+	7/7 days	reduced during 3.5 exposure then severely injured little recovery	
Pinus sylvestris	ambient near source	Nov- April	reduced AP of whole plant due to loss of foliage with visible injury on remaining foliage	Keller 1977
P. nigra				
P. strobus				
Larix leptolepis				
Quercus borealis				
Pseudotsuga menziesii				
Picea excelsa				
Alnus incana				
Sorbus Aria				
Acer pseudoplatanus				
Larix decidua				
Cornus florida	1.9 ppm	24 hours	AP reduced in older needles of <u>P. taeda</u> and <u>P. echinata</u>	McLaughlin & others 1975
Liquidambar Styraciflua	NaF			
Plantanus occidentalis				
Acer rubrum	19 ppm	24 hours	AP reduced in all species	
Liriodendron tulipifera	NaF			
Oxydendrum arboreum				
Pinus strobus	190 ppm	24 hours	AP reduced in all species	
P. taeda				
P. echinata				
Picea excelsa	100 ppm	winter- spring	AP reduced in old foliage/new injured	Keller 1980

grown with excess Mg (MacLean and others, 1976).

There is little information on the effects of F on forest tree nutrition, but there is a considerable amount of information on mineral cycling in forest ecosystems (Grier and Cole, 1972; Bormann and Likens, 1979), and airborne F can influence this cycling in forest vegetation (see "Tree Growth").

Growth and Production

Effects of F on the physiology and metabolism of plants are ultimately manifested as changes in the height, diameter, dry weight, and reproduction of the plant. But most of the available literature describes studies with agronomic crops.

Relatively low concentrations of F have been reported to stimulate growth, but growth can be inhibited by amounts of foliar F that do not produce chlorosis or necrosis in the same species (Treshow and Harner, 1968).

The effects of F on reproduction have been demonstrated and its possible implications discussed by Pack and Sulzbach (1976). They hypothesized that lowered seed production was a result of inhibition of pollen germination or pollen tube growth, inhibiting or preventing fertilization. Growth of pollen tubes in apricot (Facteau and Rowe, 1977) and sweet cherry (Facteau and others, 1973) was reduced by HF fumigation during flowering, but Dinh and others (1973) found no effect on sweet cherry pollen tube growth after exposure to $97 \mu\text{g F m}^{-3}$.

Joint Action with Other Pollutants

Experiments on the joint action of HF with other pollutants have emphasized effects on F accumulation (Matsushima and Brewer, 1972; Mandl and others, 1975, 1980), foliar lesions (Solberg and Adams, 1956; Hitchcock and others, 1962; Mandl and others, 1975, 1980); and growth and yield (Matsushima and Brewer, 1972; Mandl and others, 1980). Field studies that attempt to determine the response of plants or plant communities to F emissions must consider not only environmental and edaphic factors (Treshow and others, 1967), but also the presence of other pollutants (Bunce, 1978; McClenahan, 1978; Carlson, 1978) that complicate assessment of the impact of F alone. McCune (1980) has discussed published and unpublished results of experiments with HF in combination with SO_2 , O_3 , and NO_2 .

FIELD STUDIES

F has many characteristics that make it an ideal toxicant to study in an ecosystem. Firstly, it is an apparently non-essential element that normally occurs in foliar tissues at a concentration of $<10 \text{ ppm}$; thus, the presence and amount of airborne contamination can be measured. Secondly, F is easily identified with specific emission

sources. Thirdly, the source strength is often known and can be applied to dispersion modelling. Fourthly, F is not very mobile in plants and tends to accumulate along the margins and distal end of the leaf. Consequently, most of the F that enters a leaf remains, except for that lost by weathering and perhaps a small amount by translocation. But, as mentioned earlier, one major drawback is the difficulty of monitoring ambient concentrations.

Smith (1974) recognized three broad classes of air pollutant-dose relationships with respect to potential impacts on forest ecosystems. The Class I relationship pertains to a very low dose where the forest acts as a sink for the pollutant and the impact may be unmeasurable or stimulatory. A moderate dose relationship (Class II) is expected to cause significant direct and indirect physiological impairment to individuals resulting in reduced growth, reproduction and/or increased morbidity. With a high dose (Class III), there is acute morbidity resulting in ecosystem simplification with drastic changes in primary productivity, mineral cycling, succession, etc.

All three pollutant-dose relationships have been described in one form or another around F sources (Bunce, 1978; Treshow and others, 1967; Carlson and Dewey, 1971; Wheeler, 1972).

Fluoride Accumulation in Soils

The amount of total F in soils that has been reported ranges up to 8300 ppm but is generally from 20-500 ppm (Weinstein, 1979). In general, plants are poor accumulators of soil F (Hansen, 1958; MacIntire and others, 1949; Merriman and Hobbs, 1962; McClenahan, 1976), but there are some exceptions, notably species of Theaceae, such as tea and camellia (Zimmerman and others, 1957; Zimmerman and Hitchcock, 1956), hickories and flowering dogwood (McClenahan, 1976).

The deposition of fluoride in soils near sources of emission has been the subject of several investigations. McClenahan (1976) examined the geographic distribution of total F in soils at two seasons and at different distances from an alumina reduction smelter. Of course, the highest accumulations occurred in the direction of the prevailing winds and extended about 10 km. In areas where F deposition was lowest, total F increased with depth of the soil profile, but the opposite was true in areas where deposition was heaviest. There was a lower concentration of F in the soil profile in outlying areas than near the source. The total F in the soil profile in low and high impact areas over the two-year study period was consistently different. No attempt was made to correlate soil F with the amount of F accumulated by plants.

Relatively large amounts of F-containing amendments are necessary to increase the accumulation of F in plants (Weinstein, 1977), and Israel (1974) has estimated that each $120 \mu\text{g/g}$ increment

in soil F resulted in a gain in forage F of 1 µg/g.

The accumulation of F in conifer needles and in "soil-humus" samples near a phosphorus plant in Canada has been studied (Thompson and others, 1979; Sidhu, 1977). Severity of damage to vegetation was reported to be correlated with F concentration of foliage and of "soil-humus". Because the distribution of airborne F in soils would be expected to follow the same pattern as in vegetation, it would be difficult to estimate the proportion of F present in foliage that was accumulated from the atmosphere and that from the soil. Water-soluble F from the "soil-humus" was positively correlated with foliar F and because the soils were highly acidic, root uptake could have been an important pathway into the vegetation.

The soil as a source of F to plants has not been adequately investigated and the long-term effect of acidic precipitation in making soil F available to plants, especially in acidic, non-agricultural soils should be investigated. There are many other gaps in our understanding of the cycling of F in forest ecosystems, such as the effects of F accumulation on litter decomposition, on changes in nutrient availability, and on soil structure.

F Accumulation and Occurrence of Injury

In many reports, the authors have presented values for the F content of vegetation at different distances (and sometimes, directions) from a source, but often they did not provide information on the source strength, ambient air concentration, or the forms of airborne F that were present. Often, qualitative or semi-quantitative estimates of injury are given and there is little or no consideration to other possible causes of injury, such as insects, pathogens, environmental stresses, or even the presence of other pollutants. These data are most useful in evaluating the relative sensitivity of different species, the interspecific differences in sensitivity, the most sensitive stages of plant or foliar development, the components of the forest ecosystem most vulnerable to an effect, and, if evaluated carefully, the dose of atmospheric F or the amount of tissue F accumulated to produce a measurable effect, whether it is registered as a metabolic or physiologic change or as a chlorotic or necrotic lesion.

As one would expect, the atmospheric concentration of F and the amount accumulated in vegetation decreases with distance from the source (Treshow and others, 1967; Sidhu, 1977, 1978; Thompson and others, 1979; Roberts and others, 1979; Bunce, 1978, 1979; Wheeler, 1972; Carlson and Dewey, 1971). The amount of F accumulated in foliage, however, will depend upon many factors including the dose and form of F, the species, accessibility of the pollutant to the plant (e.g., screening of understory by overstory species), plant-to-plant variability, etc. In general, broadleaf species will accumulate more F than conifers when they occur together (Sidhu, 1977, 1978); and great differences can occur between

conifer species, ceterus parabus, with the most tolerant ones accumulating the most F (Weinstein, 1977). A likely explanation for this is that when the most sensitive species are injured (metabolically or physiologically) by a given dose of F, continued absorption and accumulation are reduced.

Not only are the most tolerant species the most efficient accumulators, but the amount of accumulation and the threshold for injury within a genus (or even species) may be vastly different in different forest ecosystems. For example, Treshow and others (1967) did not find needle injury in Douglas-fir (Pseudotsuga menziesii [Mirb.] Franco) in Idaho at F concentrations in needles that averaged 150 ppm (composite value for the current year, 2-year-old, and 3-year-old needles), while Carlson and others (1979) reported that "mottling or chlorosis was present at 6-8 ppm" (needles of uncertain age). Obviously, this difference is reflected in part by the environmental differences between the study areas (Idaho and Montana) but other differences between the results of the two studies should be considered. Firstly, Carlson and others (1979) analyzed a number of needle characteristics at different distances from the source of fluoride emissions in Montana. The most common needle injury observed was mottling (presumably chlorotic mottling) and, although significant it was a weak association that did not correlate very closely with F content of needles. The R^2 values for F content of needles and needle mottle in Douglas-fir, lodgepole pine (Pinus contorta var. latifolia Engelm.) and white pine (Pinus monticola Lamb.) were 0.0266, 0.0445, and 0.1604, respectively (Carlson, 1980). But one might conclude from the original report that the occurrence of mottling on these conifer needles was increased greatly by F. The possibility that other pollutants associated with this symptom, such as ozone or another oxidant, would be distributed in the same air mass as F was not considered. Carlson and others (1979) also concluded that there is no threshold content of F in needles above which injury will occur, and that any detectable amount of atmospheric F is detrimental to conifers. This is a simplistic explanation and it ignores several facts. The first is that all conifer needles are not equally sensitive to F injury, as is noted in many compilations (e.g., Weinstein, 1977; Fluorides, 1971; Thomas and Alther, 1966). Secondly, sensitivity to F is related to the age of the needle at the time of exposure. It would be absurd to assert that "adverse effects were visible on needles when their fluoride concentration reached 8-10 ppm" (Carlson and others, 1979) of the F accumulated after the conifer needles had completed their elongation. Thirdly, the form of F to which needles are exposed and whether it is internal or superficial would also determine the kind of effect produced. Finally, if there is no threshold for injury then there are no mechanisms of detoxification in plants and physiologic or metabolic processes, such as photosynthesis or enzyme activity, that have been altered by F should evidence no recovery. There is ample proof that once a fumigation ceases, or if the periods between

successive fumigations are sufficiently separated, recovery processes (repair mechanisms) are active (see Dinman, 1972; McCune and Weinstein, 1971; Thomas and Alther, 1966). Carlson and others (1979) appear to have equated injury from F with such destructive agents as ionizing radiation.

Another interesting contrast in the sensitivity of conifers to F is exemplified by the conditions near a phosphorus plant at Long Harbour, Newfoundland and an alumina reduction smelter at Kitimat, B.C. The Long Harbour area is classified as belonging to the Boreal Forest. Its productive forests are 8-12 m tall and are composed principally of dense stands of balsam fir (Abies balsamea [L.] Mill.) and black spruce (Picea mariana [Mill.] B.S.P.). Non-productive scrub forests are less than 5 m tall and are composed of larch (Larix laricina [DuRoi] K. Koch), black spruce, and balsam fir. Rock-barrens and peatlands are common (Thompson and others, 1979). Soils are generally shallow, precipitation is heavy, and the forests are exposed to high winds containing saline aerosols.

Kitimat is in the Pacific Coastal Rain Forest area. The forest is an uneven-aged, overmature, decadent, and stable climax forest. Logging is an important commercial activity. The forest consists of about 60% western hemlock (Tsuga heterophylla [Raf.] Sarg.), 25% balsam fir (Abies amabilis [Dougl.] Forb.), 7% western red cedar (Thuja plicata Lamb.), 6% yellow cedar (Chamaecyparis nootkatensis [Lamb.] Spach.), and 2% sitka spruce (Picea sitchensis [Bong.] Carr.). The average age of fir and hemlock is more than 300 years, and it is not unusual to observe trees of 1 or 2 m dbh. Total annual precipitation is about 115 inches and occurs on 53% of the days. The site lies in a wide trough that runs north and south, and bisects the Coast Mountains (Reid, Collins, 1976).

Emissions at the Long Harbour phosphorus plant are not known, but are certainly lower than those at Kitimat, which have ranged from 2.5 to 6.6 tons of gaseous F/day between 1955 and 1977 (Alcan Surveillance Committee, 1979). F concentrations in conifers were frequently higher than 100 ppm in late summer without evidence of any foliar lesions. This can be contrasted to the published threshold value for needle injury in balsam fir at Long Harbour of as low as 14 ppm (Sidhu, 1978). Although the same species do not occur in the two areas, the different responses of conifers are so striking that one must conclude that (1) it is not possible to generalize from one site to another, (2) foliar F contents alone may be a poor determinant of injury, and (3) environmental stresses (such as wind, salt, nutrient, or water) are as important predictors of an effect as is F content.

It is difficult to classify conifers and other tree species into groups based upon their relative tolerance to airborne F because most compilations are based upon foliar injury, and not according to effects on timber volume, fruit production, or

other objective criteria related to the intended use of the tree.

There is insufficient information to develop these kinds of lists because existing compilations are based primarily on field and laboratory observations of foliar injury. Sensitivity lists based on foliar injury (Weinstein 1977, 1979) are only a guide and do not provide evidence of relative effects on plant processes.

Tree Growth

Many studies have identified F as the cause of tree mortality around industrial sources (Adams and others, 1952; Scurfield, 1960; Jung, 1968; Robak, 1969). The determination of F as the causal agent usually entailed determination of foliar F concentrations and, occasionally, air quality monitoring. In these class III relationships (Smith, 1974), determination of the area of impact is normally easy to identify. This is not true for class II relationships for several reasons: (1) environmental factors (mainly weather patterns) change from year to year and not only distribute the pollutants randomly but also produce more or less favorable growing conditions for the impacted vegetation; (2) normal biotic factors (insects and pathogens) and abiotic factors (soils) also account for variability in growth; (3) stage of development of the stands of trees also dictate growth rates and the degree of competition between individuals; and (4) all of these factors combined with pollutant exposure produce a given effect. Therefore, to quantify the reduction in growth caused solely by the pollutant, the variability due to the other parameters must be accounted for.

Treshow and others (1967a) measured radial growth, needle length, needle dry weight, and foliar fluoride concentrations in Douglas-fir (Pseudotsuga menziesii [Mirb.] Franco) located at different distances from a fluoride source (also see "F Accumulation and Occurrence of Injury"). They classified the study plots into three groups based on foliar F concentrations of composite samples of four age-classes of needles. The groups were control (average 24 ppm F), intermediate fluoride (average 150 ppm F), and high fluoride (average 225 ppm and with some needle necrosis). Significant reductions in radial growth were found in both groups subjected to elevated F concentration. There was a significant negative correlation of needle length with radial growth, but there were no significant effects on needle dry weights. Thus, they found that (1) radial growth reduction can occur without foliar lesions and (2) Douglas-fir needles can average as much as 150 ppm F without foliar injury.

A study of the impact of F on nutrient cycling in stands of loblolly (Pinus taeda L.) and slash pines (Pinus elliotii Engelm.) and the impact on tree growth was made by Wheeler (1972). No injury symptoms attributable to F exposure were noted in the sample plots but trees at the edge of some stands did show some "burning of needles".

Increased foliar F concentrations (from 13 to 49 ppm in pooled samples) were correlated with increased return rates of Ca and K from greater leaf leaching and of Ca, K, and Mg by greater litterfall. This altered nutrient cycling presumably should alter productivity but no relationship was found between needle concentrations of F, Ca, Mg, and K and productivity as measured by amount of bole wood. Wheeler (1972) concluded that either these fluctuations in needle status did not affect growth or that the sampling was insufficient to detect differences that were present.

Extensive studies on growth and F accumulation have been made at Kitimat, B.C. (Bunce, 1978) and Columbia Falls, MT (Carlson, 1978). Both areas were subjected to F for many years before any scientific assessment of growth reduction due to F were made and each area was subjected to insect infestation (see section "Insects").

Bunce (1978) used "foliage analysis, observations of lichens, air flow patterns and topographic features" to estimate the area of impact and to establish the distribution of his sample plots. Tree ring cores were taken from western hemlock, the dominant species, from all sample locations, and were used to determine the amount of growth reduction due to F emissions. After variability in growth rates due to weather, insect infestation, and another pollutant (SO₂) were accounted for, Bunce (1978) reported the annual loss of wood production to be 950 cunits (95,000 cu. ft.) per year compared to the 800,000 cunits attributed to insect damage. Obviously the cause of the insect outbreak is fundamental to the assessment of the magnitude of the F-related effects on growth and is discussed elsewhere. Although the primary and secondary (bark beetles) insect outbreaks ended by 1968, F emissions have continued at a lower rate since 1975 and trees in the insect damaged zone are regenerating satisfactorily. The question of whether there is a cause-and-effect relationship between F emissions and insect infestation has not yet been answered for reasons discussed elsewhere.

F from an aluminum reduction plant in Columbia Falls, MT caused growth reductions in Douglas-fir, lodgepole pine, and western pine (Carlson, 1978, 1979). However due to questionable assumptions and miscalculations, an overestimate of the loss of usable timber due to F pollution was made. Statistically, the data (Carlson, 1978) showed only a weak correlation between foliar F concentrations and reduced radial growth. The area of growth reduction was substantially smaller than reported previously (Carlson, 1980). It has also been stated (Carlson, 1978) that any increase in foliar F above background concentrations is detrimental to tree growth. This assumption was generated by the implied growth reduction of trees located in areas designated as having reduced growth; but upon closer examination of the data, no growth reduction could be demonstrated (Carlson, 1980). Consequently, the original

assumption, which implies that there is no threshold concentration of foliar F below which injury does not occur, cannot be substantiated. This is not meant to imply that F is not phytotoxic, because it is the most toxic of the common atmospheric pollutants. But an understanding of its effects in the ecosystem, requires much research and the synthesis of an enormous amount of information. Koch's Postulates were not written frivolously.

Community Structure

Large areas of the Eastern United States are subjected to a complex mixture of air pollutants from urban centers and industrial sources. Most of the Eastern Deciduous Forests are subjected to at least moderate air pollution (Class II). McClenahan (1978) studied the effects of a mixture of pollutants derived from industrial sources (containing F, SO₂, NO_x, chloride, and oxidant) on changes in structure and composition of a mixed deciduous forest in the Ohio River Valley. The study sites were arbitrarily divided into overstory, subcanopy, shrub and herb layers and the stands were measured for diversity (Pielou, 1975), evenness (Williams, 1977), and species composition.

In general, the average total stand densities of the overstory and herb layers were found to decrease in proximity to the F source while the subcanopy and shrub layer increased with the shrub layer being the only layer that showed a significant correlation to F exposures. Chloride from another source had a greater influence in the other layers.

Murray (1979) conducted a study of plant community structure around an aluminum smelter in Australia. Although a number of study sites were lost by fire, he was able to ordinate species associations with F stress. More of these kinds of studies are needed to provide data to predict the risk of an effect when an ecosystem is exposed to airborne F.

Incidence and Severity of Diseases and Insects

There is evidence, from laboratory and field experiments or observations, that airborne F may alter the plant-pathogen and plant-insect relationships. The exact relationships between F and these biotic stresses and their underlying mechanisms are only beginning to be understood.

Plant pathogens -- Although there are many industrial sources of F, we are not aware of any field or laboratory reports that link airborne F with incidence or severity of forest tree diseases. It is necessary, therefore, to discuss some laboratory research on the effects of HF on diseases of crop plants in order to evaluate possible forestry effects and to establish research needs.

The plant-pollutant-pathogen interaction was reviewed by Heagle in 1973 and Laurence in 1978. For the kinds of effects that have been found, McCune and others (1973) provided three possible

explanations: (1) there could be a direct effect of the pollutant on growth and development of the organism; (2) the pollutant could affect the susceptibility of the plant to the pathogen; and (3) the pollutant could affect the microbiota or micro-environment of plant surfaces and thereby affect the pathogen.

Tobacco leaves infected with tobacco mosaic virus and containing 200-300 ppm F had a higher titer of virus than control leaves when a local lesion assay was used. The titer was lower at 500 ppm F (Dean and Treshow, 1966; Treshow and others, 1967b). But perhaps the best evidence for a direct effect of airborne F on growth and development of a pathogen was the consistent reduction in bean powdery mildew (*Erysiphe polygoni* DC.) found as a result of HF fumigation, indicating that HF was affecting the infectivity of the pathogen itself, because reduction in disease was proportional to the length of the exposure period, infection was continuous throughout the exposure period, and the pathogen itself is epiphytic.

The most likely mechanism for an effect of F on plant pathogenic diseases would be an alteration in the susceptibility of the host plant to the pathogen. The reduction in the numbers of bean rust (*Uromyces phaseoli* [Pers.] Wint.) uredia by pre- and post-inoculation exposures to HF may have been due to a change in host metabolism by the accumulation of F (McCune and others, 1973). The best evidence available that suggests an indirect effect of F was found in halo-blight of bean (*Pseudomonas phaseolicolus* [Burkh.] Dows.) where stem collapse was affected, but foliar symptoms were not. Thus, the site affected was spatially removed from the site of F accumulation, the leaf (McCune and others, 1973).

There is no reason to believe that crop plants should respond differently than forest species to airborne F and plant pathogens, and laboratory and field studies are needed to determine and evaluate effects on the incidence of disease and possible epidemiological consequences.

Insects -- The controversies associated with the effects of F on plants in general, and ecosystems in particular, also extend to the possibility that F alters the relationship between plants and destructive insects, that F kills beneficial insects, or that accumulation of F in insects makes them a vehicle for the transfer of F in ecosystems. There is ample evidence that an association can exist between F-contaminated vegetation and insects, but the relationship is not understood and it does not occur under all conditions or with all insects.

Pfeffer (1962-1963) reported that attack by bark beetles, snout beetles, and fir leaf rollers were associated with F emissions in a fir forest in Czechoslovakia. Carlson and Dewey (1971) and Carlson and others (1974) have reported that F accumulation in conifer foliage is closely related to infestations by several destructive insects: pine needle scale (*Phenacapsia pinifoliae* Fitch),

pine needle sheath miner (*Zellaria haimbachi* Busck), needle miner (*Ocnorostoma strobivorum* [Zeller]), and sugar pine tortrix (*Choristoneura lambertiana* [Busck]) that ranged from no significance (larch casebearer) to a non-significant trend (pine needle scale) to strong evidence of a weak correlation (needle miners). Only about 6% of the variation in needle damage by needle miners was associated with foliar F concentration. There was an even more remote association between needle miner population and foliar F concentration. Edmunds and Allen (1956) and Compton and others (1961) found no association between pine needle scale (*Nuculopsis californica* [Coleman]) and the extent of F injury or F content of needles of ponderosa pine (*Pinus ponderosa* Laws) and Edmunds (1973) questioned the results of Carlson and Dewey (1971). Thalenhorst (1974) and Wentzel (1965) found positive relationships between spruce galls induced by *Adelges abietes* (L.) and F. But Temple (personal communication) could find no correlation between the F content of washed silver maple foliage and galls induced by the bladder-gall mite (*Vasates quadripes* [deShimer]).

One of the most interesting examples of a possible F-plant-insect association was observed near an alumina reduction smelter in Kitimat, B.C. (see section on "F Accumulation in Plants and Occurrence of Injury" and "Tree Growth"). Between 1960 and 1963, an epizootic of saddleback loopers (*Ectropis crepuscularia* [Denis & Schiff.]) and spruce budworms (*Choristoneura orae* Free.) occurred that killed many trees over a large area that coincided well with the pattern of fume dispersion. In 1961, balsam bark beetles (*Pseudohylesinus grandis* [Swaine] and *P. nebulosus* [Lec.]) appeared as secondary pests throughout the area attacked by the looper and the budworm. We used the word "possible" above in referring to F as the causal agent in this outbreak because (1) there is no way now to establish a cause-and-effect relationship; (2) the emissions were also high in particulate materials, sulfur compounds, pitch volatiles, and even CO₂; (3) the problem was not studied at the time that the outbreaks occurred; and (4) other possible etiologies have been suggested by entomologists from the Canadian Forestry Service. Several of the theories that might explain the insect attacks at Kitimat are: (1) F absorbed by the foliage of the tree alters its metabolism and increases its attractiveness to insects; (2) F weakens the tree, rendering it less able to resist insect attack; (3) gaseous or particular emissions are toxic to parasitic and/or predaceous insects that provide important controls of the population of destructive insects; (4) the emissions have a "blanket" effect that results in a slight temperature alteration and gives the larvae of the loopers and budworm a competitive advantage over parasites and predators; (5) loopers and budworm moths were carried on winds into the Kitimat area and dispersed in the same pattern as smelter emissions; and (6) there were an unusually large number of lights in the valley above the smelter in the early 1960's and they provided light of wavelengths that attracted moths (Alcan Surveillance Committee,

1979).

Although the primary and secondary insect attacks were extremely destructive, amounting to a total net loss of mature timber estimated at 800,000 cunits (80,000,000 cu. ft.) (Reid, Collins, 1976) a considerable number of trees were not damaged and regrowth has been extensive. In some areas near the smelter, F-induced injury was a prominent feature on young hemlock, Sitka spruce, black cottonwood, and even western red cedar (Weinstein, unpublished field reports for 1971 and 1974 cited in Alcan Surveillance Committee, 1979). In the intervening years, especially since 1975, there has been a substantial reduction in total emissions from the smelter (more than 50% between 1975 and 1977), accompanied by greatly reduced foliar injury. Nevertheless, vegetation exhibiting no foliar symptoms often contains 100 ppm F or more, and the incidence of insect attack is no greater than in nearby areas not exposed to the smelter emissions. Because any reduction in emissions would include gaseous F, particles, and other components of the fumes, no cause-and-effect-relationship can be made. From subjective observations, however, particulate emissions have been reduced strikingly, at least since 1971, and we feel that this fraction of the emissions was perhaps of great significance in the original outbreaks. Certainly, the indirect relationship between an increase in insect colonization and particles has been known for many years (Bartlett, 1951). Even before Carlson and his colleagues were attempting to demonstrate a relationship between F and insects on U.S. Forest Service land and in Glacier National Park, an enormous outbreak of mountain pine bark beetle (Dendroctonus ponderosae [Hopkins]) was beginning on the Canadian border many km to the north. It extended throughout the entire Flathead National Forest and the western part of Glacier National Park, and has destroyed many thousands of lodgepole, ponderosa, and white pines.

Although there is little doubt that an association exists between airborne contaminants and insect outbreaks, the evidence for a cause-and-effect relationship with F is unconvincing, and at times it appears that some investigators have forced a relationship. Insect outbreaks occur in unpolluted as well as polluted areas. The question to be resolved is not whether there is or isn't a relationship between airborne substances and insects, but to determine the nature of this relationship and the features it has in common with other stresses.

Evaluation of F Injury in the Field

The most common measures of F injury to forests include (1) assessment of the presence and amount of foliar injury, especially of indicator species; (2) loss or depletion of sensitive receptors and community changes; (3) measurement of biomass production; and (4) the accumulation of F in plant tissues that might produce foliar injury or render the plant unsuitable for indigenous herbivores.

Regardless of the approaches used, the path to useful information can be a difficult and confusing one, as is well-known for other atmospheric pollutants. Some of these problems have been discussed by Weinstein and McCune (1970).

There are a number of indigenous plant species that are sensitive to atmospheric F, including goatweed (Hypericum perforatum L.), common barberry (Berberis vulgaris L.), Oregon grape (Mahonia repens [Lindl.] Don. and M. nervosa [Pursh.] Nutt.), blueberry (Vaccinium spp.), and young needles of many conifers (see Table 2 for a list of F-sensitive higher plants). One general conclusion can be made: field observations of plants can be a good qualitative but is usually a poor quantitative indicator of effects.

In many cases of F pollution, there has been a severe depletion of lichen populations (reviewed by Gilbert, 1973). In areas nearest the F source, a lichen desert may exist, but they appear and increase in frequency and diversity with increasing distance from the source (LeBlanc and others, 1972). Nash (1971) and LeBlanc and others (1971) transplanted several species of lichens into the field in areas of F-emitting industries and found that the species used were injured near the source (but sometimes up to 10 km away) and were effective F accumulators. Corticolous lichens accumulate F more rapidly than saxicolous species, and consequently, demonstrate accelerated damage and reduced abundance (Perkins and others, 1980), but much research remains to characterize and classify the sensitivity of the different lichen types growing on their many kinds of habitats.

Problems associated with the measurement of biomass have been discussed in many treatises on forest mensuration, and Bunce (1978, 1979) and Parker and others (1974) have discussed the problems associated with discriminating between effects of F, insects, and environmental stresses in evaluating effects on tree growth. The relationship between F accumulation and production of foliar lesions or other effects, is discussed elsewhere, and fluorosis in indigenous herbivores is beyond the scope of this review.

F Distribution in the Environment

In its most elementary form, the transfer of F to and from the atmosphere, waters, soils and rocks, and living organisms, due to natural or anthropogenic causes, has been described (Fluoride, 1971; Weinstein, 1977). F that is accumulated in plants enters the food chain through herbivores and passes into the soil in their wastes. The transfer from one animal to another is possible in the case of insects that have accumulated F on or in their bodies and are eaten by birds or other carnivores, but this has not been studied. It is also not known if increased levels of F associated with a variety of insects was due to accumulation by ingestion or by surface contamination. Hughes and others (in preparation) cultured cabbage loopers (Trichoplusia ni [Hubner]) on two kinds of diets. One contained

increasing amounts of F as NaF or KF and equivalent amounts of control cabbage. The other contained F from HF-fumigated leaves and was combined with control leaves to give the same dose curve. Analyses of prepupae and pupae showed that F accumulated in the bodies of the loopers supplied with F salt (ca. 10 pct. of the concentration of the diet on a dry weight basis); no F accumulated in loopers grown with the fumigated cabbage diet. There was evidence that loopers grown with the latter diet developed faster and grew larger than those on the control cabbage NaF or KF diets. These results suggest that the F reported as having accumulated in insects probably was due to surface contamination of particles (Carlson and Dewey, 1971; Dewey, 1973). Ingestion of surface-contaminated insects by carnivores, however, still transfers the F to the next trophic level, but the amount of accumulation that might occur at that level is not known.

There are great differences in the capacities of plants growing in the same soils to accumulate F. Most plants accumulate low concentrations of F (0 - 10 ppm), but some species can accumulate hundreds of ppm from the same soil. In any species, soluble F in the soil solution can be readily absorbed by plants (Romney and others, 1969). When deposited upon plant surfaces, the relatively insoluble forms of F have low phytotoxicity (McCune and others, 1977), but their ingestion can be harmful to herbivores. Consequently, the main source of phytotoxic airborne F is HF.

Once it enters a leaf, F moves in the transpiration stream to the tips or margins of the leaf and stays in a form that can be leached from many leaves (Leone and others, 1956; Ledbetter and others, 1960; Jacobson and others, 1966). Consequently, foliar concentrations of F need not steadily increase (Knabe, 1970). Twigs of deciduous species can accumulate F in the winter, presumably through the lenticels, and elevated concentrations have been found in young foliage in the spring (Keller, 1974, 1978). A small proportion of F entering a leaf can also be translocated to other parts of the plant (Kronberger and others, 1978).

F already present in most soils is in an insoluble form and has little influence on vegetation. The fate of F leached from foliage and of F deposited directly from the atmosphere has not been studied extensively. Any effect will depend upon the nature and chemistry of the soil. Flühler and others (1979) have shown that leaching of particulate F depends upon its water solubility (NaF > powdered pine needles containing F > cryolite > CaF₂). The F in powdered pine needle litter leached nearly as rapidly as did NaF. When NaF solutions were applied to soil columns, organic matter, aluminum, and iron were lost, but the amount depended upon the soil type.

Effects of F on soil composition and structure, mineral cycling, and litter decomposition are

important aspects of the impact of F on forest ecosystems and research in these areas should have a high priority.

LITERATURE CITED

- Adams, D. F., D. J. Mayhew, R. M. Gnagy, E. P. Richey, R. K. Koppe, and I. W. Allan.
1952. Atmospheric pollution in the ponderosa pine blight area, Spokane County, Washington. *Ind. Eng. Chem.* 44:1356-1365.
- Amundson, R. G., L. H. Weinstein, P. van Leuken, and L. J. Colavito.
1980. Joint action of HF and NO₂ on growth and fluorine accumulation in Marcross corn. *Environmental and Experimental Botany* (in review).
- Applegate, H. G., and D. F. Adams.
1960a. Effect of atmospheric fluoride on respiration in bush beans. *Bot. Gaz.* 121:223-227.
- Applegate, H. G., and D. F. Adams.
1960b. Invisible injury of bush beans by atmospheric and aqueous fluorides. *Int. J. Air Pollut.* 3:231-248.
- Applegate, H. G., and D. F. Adams.
1960c. Nutritional and water effect on fluoride uptake and respiration of bean seedlings. *Phyton* 14(2):111-120.
- Applegate, H. G., D. F. Adams, and R. C. Carriker.
1960. Effect of aqueous fluoride solutions on respiration of intact bush bean seedlings. *Amer. J. Bot.* 47:339-345.
- Bartlett, B. R.
1951. The action of certain "inert" dust materials on parasitic Hymenoptera. *J. Econ. Ent.* 44:891-896.
- Béjaoui, M. et P. E. Pilet.
1975. Effets du fluor sur l'absorption de l'oxygène de tissus de Ronce cultivés in vitro. *C. R. Acad. Sc. (D) (Paris)* 280:1457-1460.
- Bennett, J. H., and A. C. Hill.
1973. Inhibition of apparent photosynthesis by air pollutants. *J. Environ. Quality* 2(4): 526-530.
- Bolay, A., and E. Bovay.
1965. Observations sur la sensibilité aux gaz fluorés de quelques espèces végétales du Valais. *Phytopath. Z.* 53:289-298.
- Bormann, F. H., and C. E. Likens.
1979. Pattern and process in a forested ecosystem. Springer-Verlag, New York. 253 p.
- Brennan, E. G., I. A. Leone, and R. H. Daines.
1950. Fluorine toxicity in tomato as modified by alterations in the nitrogen, calcium, and phosphorus nutrition of the plant. *Plt.*

- Phys. 25: 736-747.
- Bunce, H. W. F.
1978. Fluoride emissions and forest growth
Kitimat, B.C., Canada. Paper 78-24.3,
71st Annual Meeting, Air Pollution Contr
Assoc., Houston, Texas.
- Bunce, H. W. F.
1979. Fluoride emissions and forest growth.
J. of A.P.C.A. 29(6):642-643.
- Carlson, C. E.
1978. Fluoride induced impact in a coniferous
forest near the Anaconda Aluminum Plant in
northwestern Montana. Ph.D. Dissertation,
U. of Montana, Missoula. 165 p.
- Carlson, C. E.
1978. Fluoride induced impact in a coniferous
forest near the Anaconda Aluminum Plant in
northwestern Montana. Fluoride 11(4):211.
- Carlson, C. E.
1980. Deposition of November 26-30, 1979 and
January 7-10, 1980, U.S. vs. Atlantic
Richfield Co. and the Anaconda Co., U.S.
District Court, District of Montana.
(Cause No. CV-78-80-M). Vols. I-III.
- Carlson, C. E., N. E. Bousfield, and M. D.
McGregor.
1974. The relationship of an insect infesta-
tion on lodgepole pine to fluoride emitted
from a nearby aluminum plant in Montana.
Insect Disease Report, Report No. 74-14.
21 p.
- Carlson, C. E., and J. E. Dewey.
1971. Environmental pollution by fluorides in
Flathead National Forest and Glacier National
Park. 57 p. USDA, Forest Service, Northern
Region, Division of State and Private Forestry,
Missoula, MT.
- Carlson, C. E., C. C. Gordon, and C. J. Gilligan.
1979. The relationship of fluoride to visible
growth/health characteristics of Pinus
monticola, Pinus contorta, and Pseudotsuga
menziesii. Fluoride 12:9-17.
- Chang, C.W.
1975. Fluorides. In: Responses of Plants to
Air Pollution. J. B. Mudd and T. T.
Kozlowski, eds. pl 57-97. Academic Press,
New York.
- Compton, O. C., L. F. Remmert, J. A. Rudinsky,
L. L. McDowell, F. E. Ellertson, W. M.
Mellenthin, and P. O. Ritcher.
1961. Needle scorch and condition of ponderosa
pine trees in The Dalles area. Ore. Agr.
Exp. Sta. Misc. Paper 120.
- Dean, G., and M. Treshow.
1966. Effects of fluoride on the virulence of
tobacco mosaic virus in vitro. Proc. Utah
Acad. Arts Sci. Letter 42:236-239.
- deOng, E. R.
1946. Injury to apricot leaves from fluorine
deposit. Phytopath. 36:469-472.
- Dewey, J. E.
1973. Accumulation of fluorides in insects near
an emission source in Western Montana. Environ.
Entomol. 2:179-182.
- Dinh, D. L., G. Buchloh, and W. Oelschlager.
1973. The effects of fluorine compounds on
pollen germination and fruit setting of
fruit trees. Erwerbsobstbau 15:154-157.
- Dinman, B. D.
1972. "Non-concept" of "non-threshold": chemi-
cals in the environment. Science 175:495-497.
- Edmunds, G. F.
1973. Ecology of black pineleaf scale (Homoptera:
Diaspididae). Environ. Entomol. 2:765-777.
- Edmunds, G. F., Jr., and R. K. Allen.
1958. Comparison of black pine leaf scale
population-density of normal ponderosa pine
and those weakened by other agents. Proc.
10th Internat. Congr. Entomol. 4:391-392.
- Endress, A. G., S. J. Suarez, and O. C. Taylor.
1980. Peroxidase activity in plant leaves ex-
posed to gaseous HCl or ozone. Environ.
Pollut. 22(1):47-58.
- EPA.
1973. Fluoride in Glacier National Park: A
Field Investigation. EPA-908/1-73-001.
- Facteau, T. J., and W. M. Mellenthin.
1976. Fluoride investigations in the Dalles
area 1968-1974. Tech. Bull. 132, Agric. Exp.
Station, Oregon State Univ., Corvallis, Ore.,
56 p.
- Facteau, T. J., and R. E. Rowe.
1977. Effect of hydrogen fluoride and sodium
fluoride on pollen germination in 'Tilton'
apricot. J. Amer. Soc. Hort. Sci. 102(1):
95-96.
- Facteau, T. J., S. Y. Wang, and K. E. Rowe.
1973. The effect of hydrogen fluoride on pollen
germination and pollen tube growth in Prunus
avium L. cv. 'Royal Ann'. J. Amer. Soc. Hort.
Sci. 98:234-236.
- Flühler, H., Th. Keller, and H. U. Scherrer.
1979. Les dégâts forestieres in Valais. Bull.
Murithienne 96:4-22.
- Fluorides.
1971. Committee on Biologic Effects of Atmos-
pheric Pollutants, National Academy of Sciences,
National Research Council, Washington, DC.

- Gilbert, O. L.
1973. The effect of airborne fluorides. In: Air Pollution and Lichens. B. W. Ferry, M. S. Baddeley, and D. L. Hawksworth, eds. p. 176-191. Univ. Toronto Press.
- Givan, C. V., and J. G. Torrey.
1968. Fluoride inhibition of respiration and fermentation in cultured cells of *Acer pseudoplatanus*. *Physiol. Plant.* 21:1010-1019.
- Grier, C. C., and D. W. Cole.
1972. Elemental transport changes occurring during development of a second-growth Douglas-fir ecosystem. In: Proceedings-Research on Coniferous Forest Ecosystems. A Symposium. J. F. Franklin, L. J. Dempster, and R. H. Waring, eds. p1 103-113. Pacific Northwest Forest and Range Experiment Station.
- Hansen, E. D., H. H. Wiebe, and W. Thorne.
1958. Air pollution with relation to agronomic crops. VII. Fluoride uptake from soils. *Agron. J.* 50:565-568.
- Heagle, A. S.
1973. Interactions between air pollutants and plant pathogens. *Ann. Rev. Phytopathol.* 11: 365-388.
- Hill, A. C.
1969. Air quality standards for fluoride vegetation effects. *J. of A.P.C.A.* 19(5):331-336.
- Hill, A. C., M. R. Pack, L. G. Transtrum, and W. S. Winters.
1959. Effects of atmospheric fluorides and various types of injury on the respiration of leaf tissue. *Plant Physiology* 34:11-16.
- Hill, A. C., L. G. Transtrum, M. R. Pack, and W. S. Winters.
1958. Air pollution with relation to agronomic crops: VI. An investigation of the "hidden injury" theory of fluoride damage to plants. *Agron. J.* 50:562-565.
- Hitchcock, A. E., P. W. Zimmerman, and R. R. Coe.
1962. Results of ten years' work (1951-1960) on the effects of fluorides on gladiolus. *Contrib. Boyce Thompson Inst.* 21:303-344.
- Horntvedt, R., and H. Robak.
1975. Relative susceptibility of eleven conifer species to fluoride air pollution. Rept. Norweg. Forest Res. Inst. 32:185-206.
- Horsman, D. C., and A. R. Wellburn.
1976. Appendix II. Guide to the metabolic and biochemical effects of air pollutants on higher plants. In: Effects of air pollutants on plants. T. A. Mansfield, ed., p. 185-199. Cambridge University Press, Cambridge.
- Israel, G. W.
1974. A field study of the correlation of static lime paper sampler with forage and cattle urine. *Atmos. Environ.* 8:167-181.
- Jacobson, J. S., L. H. Weinstein, D. C. McCune, and A. E. Hitchcock.
1966. The accumulation of fluorine by plants. *J. of A.P.C.A.* 16(8):412-417.
- Jung, E.
1968. Bestandesumwandlungen im Rauchschadensgebiete von Ranshofen. *Miedzynarodowej Konf. wplyw. Zanieczyszczen Powietrza Na Lasy* 6th Katowice. p. 407-413.
- Keller, Th.
1974. Translocation of fluoride by woody plants. *Fluoride* 7(1):31-35.
- Keller, Th.
1978. Wintertime atmospheric pollutants - do they affect the performance of deciduous trees in the ensuing growing season. *Environ. Pollut.* 16:243-247.
- Keller, Th.
1977. Der Einfluss von Fluorimmissionen auf die Nettoassimilation von Walebaumarten. *Eidg. Anst. forst. Versuchsweg. Mitt.* 53:161-198.
- Keller, Th.
1980. The simultaneous effect of soil-borne NaF and air pollutant SO₂ on CO₂-uptake and pollutant accumulation. *Oecologia (Berl.)* 44:283-285.
- Keller, Th., and H. Schwager.
1971. Der Nachweis unsichtbarer ("physiologischer") Fluor-Immissions-schädigungen an Waldbäumen durch eine einfache kolorimetrische Bestimmung der Peroxidase-Aktivität. Sonderdruck aus *European Journal of Forest Pathology* 1(1):6-18.
- Knabe, W.
1970. Natural loss of fluorine from needles of Norway spruce (*Picea abies* Karst). *Staub-Reinhalt. Luft.* 30(9):29-32.
- Kramer, P. J., and T. T. Kozlowski.
1979. *Physiology of Woody Plants.* 811 p. Academic Press, New York.
- Kronberger, W., G. Halbwachs, and H. Richter.
1978. Fluortranslokation in *Picea abies* (L.) Karsten *Angew. Botanik* 52:149-154.
- Larcher, W.
1975. *Physiological Plant Ecology.* 252 p., Springer-Verlag, Berlin.
- Laurence, J. A.
1978. Effects of air pollutants on plant-pathogen interactions. Paper 78-44.5, 24 p., 71st Ann. Meeting, Air Pollution Control Assoc., Houston, Texas.

- LeBlanc, F., G. Comeau, and D. N. Rao.
1971. Fluoride injury symptoms in epiphytic lichens and mosses. *Can. J. Bot.* 49:1691-1698.
- LeBlanc, F., D. N. Rao, and G. Comeau.
1972. Indices of atmospheric purity and fluoride pollution pattern in Arvida, Quebec. *Can. J. Bot.* 50:991-998.
- Ledbetter, M. C., R. Mavrodineanu, and A. J. Weiss.
1960. Distribution of studies on radioactive fluorine-18 and stable fluorine-19 in tomato plants. *Contrib. Boyce Thompson Inst.* 20(5): 331-348.
- Leonard, C. D., and H. B. Graves.
1966. Effect of air-borne fluorides on 'Valencia' orange yields. *Proc. Fla. State Hort. Soc.* 79:81-86.
- Leone, I. A., E. Brennan, and R. H. Daines.
1956. Atmospheric fluoride: its uptake and distribution in tomato and corn plants. *Plt. Phys.* 31(5):329-333.
- MacIntire, W. H., and associates.
1949. Effects of fluorine in Tennessee soils and crops. *Ind. Eng. Chem.* 41:2466-2475.
- MacLean, D. C., O. F. Roark, G. Folkerts, and R. E. Schneider.
1969. Influence of mineral nutrition on the sensitivity of tomato plants to hydrogen fluoride. *Environmental Science and Technology* 3(11):1201-1204.
- MacLean, D. C., R. E. Schneider, and D. C. McCune.
1976. Fluoride susceptibility of tomato plants as affected by magnesium nutrition. *J. Amer. Soc. Hort. Sci.* 101(4):347-352.
- Mandl, R. H., L. H. Weinstein and M. Keveny.
1975. Effects of hydrogen fluoride and sulphur dioxide alone and in combination on several species of plants. *Environ. Pollut.* 9:133-143.
- Mandl, R. H., L. H. Weinstein, M. Dean, and M. Wheeler.
1980. The response of sweet corn to HF and NO₂ under field conditions. *Environ. Exp. Bot.* 20: (In press).
- Matsushima, J., and R. F. Brewer.
1972. Influence of sulfur dioxide and hydrogen fluoride as a mix or reciprocal exposure on citrus growth and development. *J. of A.P.C.A.* 22:710-713.
- McClenahan, J. R.
1976. Distribution of soil fluorides near an airborne fluoride source. *J. Environ. Qual.* 5:472-475.
- McClenahan, J. R.
1978. Community changes in a deciduous forest exposed to air pollution. *Can J. For. Res.* 8(4):432-438.
- McCune, D. C.
1980. Terrestrial vegetation air pollutant interactions: Gaseous pollutants hydrogen fluoride and sulfur dioxide. In: *Air Pollutants and Their Effects on the Terrestrial Ecosystem. Current Status and Future Needs of Research on Effects and Technology.* May 10-17, Banff, Canada.
- McCune, D. C., D. C. MacLean, and R. E. Schneider.
1976. Experimental approaches to the effects of airborne fluoride on plants. In: *Effects of Air Pollutants on Plants.* T. A. Mansfield, ed. p. 31-46. Cambridge Univ. Press, Cambridge.
- McCune, D. C., D. H. Silberman, and L. H. Weinstein.
1977. Effects of relative humidity on free water on the phytotoxicity of hydrogen fluoride and cryolite. *Proc. IVth Internat. Clean Air Congress, Tokyo (JUAPPA)*, p. 116-119.
- McCune, D. C., and L. H. Weinstein.
1971. Metabolic effects of atmospheric fluorides on plants. *Environ. Pollut.* 1:169-174.
- McCune, D. C., L. H. Weinstein, J. F. Mancini, and P. van Leuken.
1973. Effects of hydrogen fluoride on plant-pathogen interactions. p. A146-A149. *Proc. 3rd Internat. Clean Air Congr., VDI-Verlag, Dusseldorf.*
- McLaughlin, S. B., Jr., and R. L. Barnes.
1975. Effects of fluoride on photosynthesis and respiration of some south-east American forest trees. *Environ. Pollut.* 8:91-96.
- McLaughlin, S. B., D. S. Shriner, R. K. McConathy, and L. K. Mann.
1979. The effects of SO₂ dosage kinetics and exposure frequency on photosynthesis and transpiration of kidney beans (*Phaseolus vulgaris* L.). *Environ. and Exp. Bot.* 19: 179-191.
- Merriman, G. M., and C. S. Hobbs.
1962. Bovine fluorosis from soil and water sources. *Tennessee Agric. Exp. Sta. Bull.* 347. Knoxville, Univ. Tenn. 46 p.
- Murray, F.
1979. A study of plant community dynamics around an aluminum smelter. *Annual Rept.* 1978-1979. Dept. of Biological Sciences, The University of Newcastle, New South Wales.
- Miller, J. E., and G. W. Miller.
1974. Effects of fluoride on mitochondrial activity in higher plants. *Physiol. Plant.* 32: 115-121.
- Nash, T. H.
1971. Lichen sensitivity to hydrogen fluoride.

Navara, J.

1963. Zur Frage des Einflusses von Fluor in Substrat auf die Intensität der Stomatär-Kutikularen Transpiration und auf die Photosynthese. *Biologia, Bratislava* 18(1):15-22.

Navara, J. and V. Kozinka.

1967. Wasserhaushalt der Pflanzen in gegenwart gasförmiger Fluorverbindungen in der Atmosphäre. *Biologia, Bratislava* 22(3):210-220.

Niklfeld, H.

1967. Pflanzensozioökologische Beobachtungen im Rauchschadensgebiet eines Aluminiumwerkes Centralbl. Gesamte Forstwest. 84:318-329.

Pack, M. R.

1966. Response of tomato fruiting to hydrogen fluoride as influenced by calcium nutrition. *J. of A.P.C.A.* 16(10):541-544.

Pack, M. R., and C. W. Sulzbach.

1976. Response of plant fruiting to hydrogen fluoride fumigation. *Atmos. Environ.* 10: 73-81.

Parker, M. L., H. W. F. Bunce, and J. H. G. Smith.

1974. The use of x-ray densitometry to measure the effects of air pollution on tree growth near Kitimat, British Columbia. p. 185-204. *Internat. Cong. Air Pollut. and Forestry, Mariánské Lázně, Czechoslovakia.*

Perkins, D. F., R. O. Millar, and P. E. Neep.

1980. Accumulation of airborne fluoride by lichens in the vicinity of an aluminum reduction plant. *Environ. Pollut.* 21:155-168.

Pfeffer, A.

- 1962-1963. Insectenschadlinge an Tannen im Bereich der Gasexhalationen. *Z. Angew. Entomol.* 51:203-207.

Peilou, E. C.

1975. *Ecological Diversity*. John Wiley and Sons, New York.

Poovaiah, B. W. and H. H. Wiebe.

1973. Influence of hydrogen fluoride fumigation on the water economy of soybean plants. *Plt. Phys.* 51:396-399.

Reid Collins and Associates, Ltd.

1976. Fluoride emissions and forest growth. Aluminum Co. of Canada, Kitimat, B.C. 112 p.

Report of Alcan Surveillance Committee.

1979. Environmental effects of emissions from the Alcan smelter at Kitimat, B.C. 151 p. Ministry of Environment, Province of British Columbia.

Robak, H.

1969. Aluminum plants and conifers in Norway.

Roberts, B. A., L. K. Thompson, and S. S. Sidhu.

1979. Terrestrial bryophytes as indicators of fluoride emissions from a phosphorus plant. Long Harbour, Newfoundland, Canada. *Can. J. Bot.* 57:1583-1590.

Romney, E. M., R. A. Wood, and P. A. T. Wieland.

1969. Radioactive fluorine 18 in soil and plants. *Soil Science* 108(6):419-423.

Scurfield, G.

1960. Air pollution and tree growth. *For. Abstr.* 339-347 and 517-528.

Sidhu, S. S.

1977. Fluoride levels in air, vegetation and soil in the vicinity of a phosphorus plant. Paper 77-30.2. 16p. 70th Annual Meeting, Air Pollution Control Assoc., Toronto, Canada.

Sidhu, S. S.

1978. Patterns of fluoride accumulation in forest species as related to symptoms and defoliation. Paper 78-24.7 16 p. 71st Annual Meeting, Air Pollution Control Assoc., Houston, Texas.

Smith, W. H.

1974. Air pollution-Effects on the structure and function of the temperate forest ecosystem. *Environ. Pollut.* 6:111-129.

Solberg, R. A. and D. F. Adams.

1956. Histological responses of some plant leaves to hydrogen fluoride and sulfur dioxide. *Amer. J. Bot.* 43:755-760.

Stoiber, R. E., S. N. Williams, and L. L. Malinconica.

1980. Mount St. Helens. Washington, 1980 volcanic eruption: Magmatic gas component during the first 16 days. *Science* 208:1258-1259.

Suttie, J. W.

1977. Effects of fluoride on livestock. *J. Occup. Med.* 19:40-48.

Thalenhorst, W.

1974. Untersuchungen über den Einfluss fluorhaltiger Abgase auf die Disposition der Fichte für den Befall durch die Gallenlaus *Sacchiphantes abietis* (L.). *A. Pflanzen Krankh. und Pflanzenschutz* 81(12):717-727.

Thomas, M. D.

1958. Air pollution with relation to agronomic crops: I. General status of research on the effects of air pollution on plants, *Agron. J.* 50:545-550.

Thomas, M. D., and E. W. Alther.

1966. The effects of fluoride on plants. *Handbook of Experimental Pharmacology*. Vol. 20. Part 1, p. 231-306. Springer-Verlag, New York.

- Thomas, M. D., and R. H. Hendricks.
1956. Effect of air pollution on plants. In: Air Pollution Handbook. P. L. Magill, F. R. Holden, and C. Ackley, eds. McGraw-Hill Co., N.Y.
- Thompson, L. K., S. S. Sidhu, and B. A. Roberts.
1979. Fluoride accumulations in soil and vegetation in the vicinity of a phosphorus plant. Environ. Pollut. 18:221-234.
- Thompson, C. R., O. C. Taylor, M. D. Thomas, and J. O. Ivie.
1967. Effects of air pollutants on apparent photosynthesis and water use by citrus trees. Environ. Sci. & Technol. 1:644-650.
- Treshow, M., F. K. Anderson, and F. Harner.
1967a. Responses of Douglas-fir to elevated atmospheric fluorides. For Sci. 13(2): 114- 20.
- Treshow, M., G. Dean, and F. Harner.
1967b. Stimulation of tobacco mosaic virus-induced lesions in bean by fluoride. Phytopath. 57:756-758.
- Treshow, M., and F. M. Harner.
1968. Growth responses of Pinto bean and alfalfa to sub-lethal fluoride concentrations. Can. J. Bot. 46:1207-1210.
- Weinstein, L. H.
1961. Effects of atmospheric fluoride on metabolic constituents of tomato and bean leaves. Contrib. Boyce Thompson Inst. 21:215-231.
- Weinstein, L. H.
1977. Fluoride and plant life. J. of Occup. Med. 19(1):49-78.
- Weinstein, L. H., and D. C. McCune.
1970. Field surveys, vegetation sampling, and air and vegetation monitoring. p. G1-G4. In: Recognition of Air Pollution Injury to Vegetation: A Pictorial Atlas. J. S. Jacobson and A. C. Hill, eds. Air Pollution Control Assoc., Pittsburgh, Pa.
- Wentzel, K. F.
1965. Insekten als Immissionsfolgeschädlinge. Naturwissenschaften 52:113.
- Wheeler, G. L.
1972. The effects of fluorine on the cycling of calcium, magnesium, and potassium in pine plantations of eastern North Carolina. Dissertation. North Carolina State Univ. at Raleigh. 71 p.
- Williams, F. M.
1977. Model-free evenness: an alternative to diversity measures. Satellite Program in Statistical Ecology of the International Statistical Ecology Program. The Pennsylvania State University, University Park, Pa.
- Woltz, S. S., and C. D. Leonard.
1964. Effect of atmospheric fluorides upon certain metabolic processes in Valencia orange leaves. Proc. Fl. State Hort. Soc. 77:9-15.
- Yee-Meiler, D.
1975. Über die Eignung von Phosphatase- und Esteraseaktivitäts-bestimmungen an Fichtennadeln und Ritkenblättern zum Nachweis "unsichtbarer physiologischer" Fluorimmissionsschädigungen. Eur. J. For. Path. 5: 329-338.
- Yee-Meiler, D.
1977. Phenole als Indikatoren metabolischen Störungen bei fluorexponierten Waldbäumen. Mitteilungen eidg. Anst. forstl. Vers wes. 53(4):203-229.
- Yu, M. H. and G. W. Miller.
1967. Effect of fluoride on the respiration of leaves from higher plants. Plant Cell Physiol. 8:483-493.
- Zimmerman, P. W., and A. E. Hitchcock.
1956. Susceptibility of plants to hydrofluoric acid and sulfur dioxide gases. Contrib. Boyce Thompson Inst. 18:263-279.
- Zimmerman, P. W., and A. E. Hitchcock, and J. Gwirtsman.
1957. Fluorine in food with special reference to tea. Contrib. Boyce Thompson Inst. 19: 49-53.

Air Pollution—a 20th Century Allogenic Influence on Forest Ecosystems¹

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Abstract: Chronic doses of ozone, sulfur dioxide, nitrogen oxides, hydrogen fluoride and other primary or secondary gaseous air contaminants may cause subtle effects on forest ecosystems. Air pollutants may influence reproduction, nutrient cycling, photosynthesis, predisposition to entomological or pathological stress or quantity of healthy foliar tissue. Forest ecosystem response to chronic air pollution may include alterations in growth rates and successional patterns. The establishment of comprehensive field and laboratory investigations to systematically examine chronic air pollution stress on forest ecosystems in those parts of the world subject to atmospheric contamination is concluded to be of top priority. In the United States, forest ecosystems judged to be at particular risk and in need of more intensive investigation include the Northern Hardwood forest, Central Hardwood forest and Western Montane forest.

The interactions between air contaminants and forest ecosystems are extremely complex, but can be conveniently divided into three major classes (Smith 1980). Under conditions of low dose - Class I relationship - the vegetation and soils of forest ecosystems function as important sources and sinks for air pollutants. When exposed to intermediate dose - Class II relationship - individual tree species or individual members of a given species may be subtly and adversely affected by nutrient stress, impaired metabolism, predisposition to entomological or pathological stress, or direct disease induction. Exposure

to high dose - Class III relationship - may induce acute morbidity or mortality of specific trees. At the ecosystem level the impact of these various interactions would be very variable. In the Class I relationship, pollutants would be exchanged between the atmospheric compartment, available nutrient compartment, other soil compartments and various elements of the biota. Depending on the nature of the pollutant, the ecosystem impact of this transfer could be undetectable (innocuous effect) or stimulatory (fertilizing effect). If the effect of air pollution dose on some component of the biota is inimical then a Class II relationship is established. The ecosystem impact in this case could include reduced productivity or biomass, alterations in species composition or community structure, increased insect outbreaks or microbial disease epidemics and increased morbidity. Under conditions of high dose and Class III relationship, ecosystem impacts may include gross simplification, impaired energy flow and biogeochemical cycling, changes in hydrology and erosion, climate alteration and major impacts

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on associated ecosystems.

This paper is specifically concerned with Class II interactions resulting from forest ecosystem exposure to chronic doses of ozone, sulfur dioxide, nitrogen oxides, hydrogen fluoride or other primary or secondary gaseous air contaminants. It specifically addresses the relationship between these gases, and their mixtures, on forest reproduction, forest metabolism and direct forest stress as detailed by previous contributors to this section and attempts to examine resulting perturbations in ecosystem structure and function.

FOREST REPRODUCTION

Sexual reproduction of forest trees is critically important for maintenance of genetic flexibility and the persistence of most species in natural forest communities. Reproductive strategies, however, are typically beset by a variety of "weak points" and reproductive growth of many forest trees is, at best, irregular and quite unpredictable. Generally there is a very good correlation between tree vigor and the capacity for flowering and fruiting (Kramer and Kozlowski 1979). A variety of environmental constraints impose restrictions on tree reproductive processes. Because air contaminants may reduce tree vigor and in view of the fact that numerous potential points of interaction have been identified between air pollutants and reproductive elements (Smith 1980), it has been hypothesized that air contaminants may impact forest ecosystems by influencing reproductive processes.

Considerable evidence has been presented indicating a potential adverse impact of numerous gaseous pollutants on pollen metabolism. Other papers have indicated reduced cone and fruit production under field conditions (Smith 1980). If one or more of these various reproductive stress mechanisms is operative in natural forest ecosystems, it is possible that changes in species composition may ultimately occur. In their study of ozone impact on the understory vegetation of an aspen ecosystem, Harward and Treshow (1975) concluded that only one or two years of ozone exposure might be sufficient to cause shifts in community composition because of seed production responses to ozone exposure.

FOREST METABOLISM

Photosynthesis is the most fundamental metabolic process of forest ecosystems and is the primary determinant of growth and biomass accumulation. The rate of net photosynthesis of mature trees frequently is within the range of 10-200 mg of carbon dioxide taken up per gram of dry weight per day. The rate is

extremely variable, however, and influenced by genetic, clonal and provenance differences, season of the year, time of day, position within the crown of the tree, age of foliage, climate and edaphic factors.

Studies with a wide variety of agricultural and herbaceous species, under controlled environmental conditions, have indicated that air contaminants must be added to the list of environmental variables that can potentially alter the rate of photosynthesis.

Because of ease of handling and experimental design, investigators studying the relationship between air pollutants and tree photosynthesis have primarily employed tree seedlings for research material and controlled environmental facilities for growth. Evidence has been provided, under the above circumstances, for photosynthetic suppression caused by sulfur dioxide, ozone, fluoride, heavy metals and coal dust. The thresholds of photosynthetic toxicity for tree seedlings vary with individual species and individual pollutants. For several seedlings the threshold of sulfur dioxide photosynthetic influence may approximate 1 ppm (2620 $\mu\text{g m}^{-3}$) or less for an exposure of several hours. For ozone, the threshold of photosynthetic response may approximate 0.5 ppm (980 g m^{-3}) or less for an exposure of several days (Smith 1980).

Considerable risk is associated with extrapolation of seedling photosynthetic data accumulated in controlled environmental facilities to older trees in natural forests. Excised leaf and small chamber techniques, therefore, have been employed to assess the air pollutant influence on photosynthetic rates of trees five-years-old and older. The use of sapling-age experimental material avoids the unique characteristics of seedling metabolism. Evidence for forest tree sapling photosynthetic suppression has been presented for sulfur dioxide, ozone and cadmium. For sulfur dioxide and ozone exposure, the sapling evidence suggests that the threshold of photosynthetic reduction may approximate 0.5 to one ppm for 5-10 hours for one or two days (Smith 1980).

Much of the seedling and sapling evidence suggests that the photosynthetic inhibition caused by sulfur dioxide and ozone is reversible if the pollutant stress is removed. Under the circumstance of variable pollutant concentration in ambient atmospheres, photosynthetic recovery might be common. Synergism, or greater stress resulting from simultaneous pollutant exposure relative to either pollutant alone, appears frequently in the seedling and sapling literature. Evidence for synergistic photosynthetic suppression by sulfur dioxide and ozone and fluoride and cadmium has been presented. Almost all of the studies report photosynthetic depression in the absence, or at least prior to, the appearance of visible

foliar symptoms.

The evidence for air pollution induced photosynthetic suppression in large trees in natural environments is extremely meager and fragile. The seedling - sapling evidence however, demonstrates a threshold of effect that approaches ambient concentrations in numerous temperate environments. Because of the profound importance of the photosynthetic process and the potential for suppression by widespread air contaminants, appropriate field studies must be conducted in spite of their difficulty and cost. The opportunity to examine the impact of contaminants on respiration and transpiration should also be included in experimental designs. Inclusion of one or both of these physiologic processes in seedling - sapling studies has revealed some indication for significant alteration. Increased respiration coupled with reduced photosynthesis could exacerbate growth consequences.

FOREST FOLIAGE

Under conditions of sufficient dose, air pollutants directly cause visible injury to forest trees. The accumulation of particulate contaminants on leaf surfaces or the continued uptake of gaseous pollutants through leaf stomata will eventually result in cell and tissue damage that will be manifest in foliar symptoms obvious to the trained, but unaided eye. This direct induction of disease in trees by air pollutants is the most dramatic and obvious individual tree response of all Class II interactions. It is the only Class II interaction that can be detected in the field by casual observation. Unlike altered reproductive strategy, nutrient cycling, tree metabolism or insect and disease relationships; the degree of foliar symptoms induced by air pollutants can be relatively easily observed, inventoried and quantified. In the presence of sufficient dose, tree damage may be of sufficient severity to cause mortality.

Acute foliar disease may be caused in forest vegetation by widespread air contaminants including; sulfur dioxide, nitrogen oxides, ozone, peroxyacetyl-nitrates, fluoride and several trace metals, and localized air contaminants including acid rain, ammonia, chlorine, hydrocarbons and hydrogen sulfide. The response of woody plants to these atmospheric pollutants is extremely variable and dramatically controlled by genetic factors, plant age and health and environmental conditions. Field symptoms of air pollution injury are not highly specific, are mimicked by a wide variety of other tree stress factors and are useful only to experienced observers familiar with the range of edaphic, entomological and pathological stress factors characteristic of a given flora in a given

location. The dose required to produce acute injury varies widely with pollutant and vegetative type. There has been sufficient work done to enable a generalized ranking of relative forest tree sensitivity to the most important air pollutants (Davis and Wilhour 1976). A summary treatment of general symptoms and injury thresholds for the gaseous contaminants included in this section is contained in Smith (1980).

FOREST ECOSYSTEM RESPONSE

The primary response of a forest ecosystem to sustained intermediate dose and Class II interaction would be reduced growth and consequently reduced biomass. Reduced essential element availability, decreased photosynthesis, increased respiration, increased insect and disease stress and decreased foliar tissue would all contribute to a reduction in tree growth rates and ultimately to lessened forest biomass. Alterations in the reproductive strategies of individual tree species or differential response of these species to reduced nutrition, altered metabolism and pest stress and to direct foliar injury may cause changes in competitive ability and ultimately lead to alterations in tree succession and species composition. Recent reviews of Class II vegetative responses to air pollutants include Heck and others (1977), Jensen and others (1976) and Weinstein and McCune (1979).

Forest Growth

Forest growth is complex in concept and measurement. Addition of woody tissue is the dominant feature of forest growth. The accumulation of woody biomass (living weight) represents gross photosynthetic production less respiratory losses. The most fundamental characteristic of an ecosystem is its productivity. Forest productivity is high relative to other ecosystems and net productivity of 1200 dry g m⁻² year⁻¹ for trees and shrubs together is quite typical for temperate forests (Whittaker 1975). Productivity is strongly controlled, however, by a variety of variables including system age and environmental parameters. The most important of the latter include nutrient availability, water availability and temperature. Because of the variety of Class II interactions identified, air quality also influences forest productivity in certain environments.

Productive forests are critically important, not only for the obvious relationship between wood volume and commercial products in managed forests, but also for the regulation and maintenance of quality for associated ecosystems, amenity functions and general climatic and terrestrial stability. It is disconcerting to realize, therefore, that

there is substantial and impressive evidence to indicate that two widespread air contaminants, sulfur dioxide and ozone, are capable of reducing forest growth. The more localized release of fluoride can also reduce the amount of forest biomass (Smith 1980).

Evidence from a variety of studies examining forest growth in the vicinity of large point sources of sulfur dioxide has indicated significantly reduced growth. Generally the correlation of growth impact with degree of foliar injury caused by sulfur dioxide is not high. Growth retardation occurs in the absence of any visible indication of stress. Most sulfur dioxide studies have accounted for precipitation influence on forest growth over the study periods. Evidence for ozone suppression of forest growth has been provided by the comprehensive oxidant impact study of the Western Montane forest ecosystem in California. Localized reduction of forest growth may also occur in environments subject to elevated levels of fluoride.

There are two serious deficiencies of forest growth - air pollution stress research. The first relates to the paucity of ambient air quality determinations in growth studies. This makes establishment of dose thresholds or correlations of dose with growth influence nearly impossible. The second serious limitation relates to the inability to partition reduced growth to the various Class II interactions that may actually be responsible for it. For example, what percentage of reduced growth may be due to reduced nutrition, reduced photosynthesis, increased insect or disease activity or increased foliar damage?

Future investigations of forest growth, as impacted by air quality, must also include better accounts of growth influencing factors other than precipitation and air pollutants. Better awareness of additional climatic factors, impacts of insect and disease influence, and management strategies must be indicated.

Forest Succession

As a result of the considerable varietal and species variation in relative susceptibility to the various Class II interactions, it is reasonable to suppose that differential tolerance to air pollution influence at the species level may be reflected in altered patterns of succession and species composition at the ecosystem level.

Ecologists recognize two major types of processes that influence ecosystem succession. Autogenic processes are those resulting from biological factors within the system. In forest ecosystems autogenic processes would include site alterations caused by the vegetation, influence of one plant species on

another and impact of native insect or disease microorganisms. Allogenic processes, on the other hand, are abiotic factors that influence succession from without the system. Geochemical and climatic forces are especially important examples of allogenic factors that influence forest ecosystems. Idealized ecosystem development characteristically is portrayed as an orderly change of biological progression occurring in a more or less constant environment (Odum 1969, Woodwell 1974). It has been generally assumed that autogenic processes dominate allogenic processes in terrestrial ecosystem succession. This generalization, however, is quite inconsistent with data generated by recent imaginative studies with forest ecosystems. The importance of fire (an allogenic force) in influencing pre-settlement forest ecosystems in the North Central states of the United States has been substantial (Loucks 1970, Frissell 1973, Heinzelman 1973). The significance of wind stress (an allogenic force) has been suggested to exert substantial control over successional development of forest ecosystems in New England (Stephens 1955, 1956, Raup 1957, Henry and Swan 1974). Forest management practices imposed by man, for example clear-cutting, may simulate the influence of natural allogenic forces on forest development and interrupt progress toward a steady state condition (Bormann and Likens 1979). Conversely other forest management procedures, for example fire control, may eliminate a controlling allogenic force and permit succession to proceed toward an unnatural steady state condition. Class II stresses imposed on forest ecosystems by air pollutants may be considered a 20th Century allogenic process of potential importance to forest ecosystem development. As in the case of clearcutting, this human related force might be expected to alter the attainment of steady state conditions. Air pollution stress would appear to have certain unique qualities that may make it an allogenic influence of particular importance. Length of exposure to this force precludes evolutionary adjustment and its influence, in certain areas, may be quite continuous rather than cyclic as are windstorms and fires. What is the evidence available to support the importance of air pollution as an allogenic force of significance in forest ecosystem development?

In 1968, prior to sophisticated understanding of most Class II interactions, Treshow (1968) provided an excellent review of the impact of air contaminants on plant populations. Treshow's review, along with a variety of additional late 1960's papers, for example Niklfeld (1967), Hajdúk and Ružička (1968) and Trautmann and others (1970), have indicated alterations in successional pattern or species composition in forest ecosystems subject to air pollution exposure.

The forests of the San Bernardino

Mountains in southern California have been subject to oxidant stress from the Los Angeles metropolitan complex for thirty years. Intensive investigations conducted in the San Bernardino National Forest over the years have provided valuable insight and perspective on a variety of forest air pollution relationships. In 1970, Cobb and Stark concluded that if air pollution from the Los Angeles basin continued to increase, there will be a conversion from well stocked forests dominated by ponderosa pine (Pinus ponderosa Doug ex. Laws) to poorly stocked stands of less susceptible tree species in the San Bernardino Mountains. Miller (1973) has provided a thorough discussion of this oxidant induced forest community change. Ponderosa pine is one of five major species of the "mixed conifer type" that covers wide areas of the western Sierra Nevada and the mountain ranges, including the San Bernardino Mountains, in southern California from 1000 to 2000 m (3000-6000 feet) elevation. Other species represented include sugar pine (Pinus lambertiana Dougl.), white fir (Abies concolor [Gord. & Glend.] Lindl.), incense-cedar (Libocedrus decurrens Torr.) and California black oak (Quercus kelloggii Newb.). The response of these five major tree species to oxidant air contaminants in the San Bernardino National Forest has been variable. Ponderosa pine exhibits the most severe foliar response to elevated ambient ozone. A 1969 aerial survey conducted by the U.S.D.A. Forest Service indicated 1.3 million ponderosa (or Jeffrey, Pinus jeffreyi Grev. & Balf.) pines on more than 405 km² (100,000 acres) were stressed to some degree. Mortality of ponderosa pine has been extensive. Actual death is typically attributed to bark beetle infestation of air pollution stressed trees. White fir has suffered slight damage, but scattered trees have exhibited severe symptoms. Sugar pine, incense cedar and black oak have exhibited only slight foliar damage from oxidant exposure. A 233 ha (575 acre) study block was delineated in the northwest section of the San Bernardino National Forest in order to conduct an intensive inventory of vegetation present in various size classes and to evaluate the healthfulness of the forest. Ponderosa pines in the 30 cm (12 inch) diameter class or larger were more numerous than any other species of comparable size in the study area. These pines were most abundant on the more exposed ridge crest sites of the sample area. Mortality of ponderosa pine ranged from 8-10 percent during 1968-1972. The loss of a dominant species in a forest ecosystem clearly exerts profound change in that system. Miller concluded from his investigation that the lower two-thirds of the study area will probably shift to a greater proportion of white fir. It was judged that incense cedar will probably remain secondary to white fir. Sugar pine was presumed to be restricted by lesser competitive ability and dwarf mistletoe infection. The rate of composition change was deemed dependent on the

rate of ponderosa pine mortality. The upper one-third of the study area, characterized as more environmentally severe due to climatic and edaphic stress, supports less vigorous white fir growth. Following loss of ponderosa pine in this area, sugar pine and incense cedar may assume greater importance. Miller judged, however, that natural regeneration of the latter species may be restricted in the more barren, dry sites characteristic of the upper ridge area. California black oak and shrub species may become more abundant in these disturbed areas. Additional and intensive research on forest composition in the San Bernardino National Forest has been reported (Miller 1977). Tree population dynamics were examined on 18 permanent plots established in 1972 and 1973 and on 83 temporary plots established in 1974 to investigate forest development as a function of time since the most recent fire. Generally the data still support the hypothesis that forest succession toward more tolerant species such as white fir and incense cedar occurs in the absence of fire. In the presence of fire, pine may be favored by seedbed preparation and elimination of competing species. These more recent studies suggest a larger number of forest subtypes may exist within the forest ecosystem than initially realized.

The changes in forest composition caused by oxidants in this southern California forest have created a management concern, as well as ecological change, because the forest is intensively used as a recreational resource and the loss of ponderosa pine is judged to reduce aesthetic qualities of the forest.

Other examples, not as dramatic as the San Bernardino example, can be found. Hayes and Skelly (1977) have monitored total oxidants and associated oxidant injury to eastern white pine in three rural Virginia sites between April 1975 and March 1976. Varieties of pine categorized as sensitive and intermediate to oxidant stress were judged to be under stress. The authors speculated that susceptible eastern white pine (Pinus strobus L.) in the Blue Ridge and Southern Appalachian Mountains may be rendered less competitive by air pollution stress. Shifts in species composition away from white pine importance along with other changes in tree distributions may be occurring in certain eastern regions. Brandt and Rhoades (1973) in their investigation of limestone dust impact in deciduous forests in southwestern Virginia predicted changes in species composition resulting from dust influence. Dusty sites had reduced seedling and sapling density of red maple (Acer rubrum L.), chestnut oak (Quercus prinus L.) and red oak (Quercus borealis Michx. f.). This observation along with documentation of reduced mean basal area and lateral growth of these trees, led the authors to suggest that yellow-poplar (Liriodendron tulipifera L.), more resistant to

stress caused by dust accumulation, would increase in importance in these hardwood stands.

Treshow and Stewart (1973) have conducted one of the few studies truly concerned with air pollution impact on an entire vegetative community. Portable fumigation chambers were placed over representative plants in intermountain grassland, oak, aspen and conifer communities. Ozone fumigations were conducted to establish injury thresholds for 70 common plant species indigenous to these communities. Generally injury was evident at varying concentrations above 15 pphm ($294 \mu\text{g m}^{-3}$). Species that were found to be most sensitive to ozone in the grassland and aspen communities investigated included some dominants which were considered key to community integrity. The most dramatic example was aspen (*Populus tremuloides* Michx.) itself. Single two-hour exposure to 15 pphm ozone caused severe symptoms on 30 percent of the foliage exposed. White fir seedlings require aspen shade for optimal juvenile growth. The authors judged that significant aspen loss might restrict white fir development and alter forest succession. In a companion study, Harward and Treshow (1975) pursued their interest in evaluating ozone impact on aspen communities by evaluating the growth and reproductive response of 14 understory species to ozone. Plants were fumigated in greenhouse chambers throughout their growing seasons. It was concluded from these fumigations that plant sensitivities varied sufficiently to make probable major shifts in composition in aspen communities following only a year or two of exposure to ozone above concentrations of 7-15 pphm ($137 - 294 \mu\text{g m}^{-3}$). The authors observed that comparable doses are widespread in the vicinity of urban areas and that widespread impacts on plant community stabilities may be common in nature. The efforts of Michael Treshow and colleagues highlights the importance of examining shrub and herb strata when assessing air pollution impact on forest ecosystems.

McClenahan (1978) has provided a most interesting study with quantitative data on the impact of polluted air on the various strata of a forest ecosystem. Forest vegetation was measured in seven stands on similar sites in a 50 km area of the upper Ohio River Valley. The stands were situated along a gradient of polluted air containing elevated concentrations of chloride, sulfur dioxide, fluoride and perhaps other contaminants. Species richness (number of different species) evenness (dominance index - low values indicate dominance by one or a few species) and Shannon diversity index were typically reduced within the overstory, subcanopy and herb strata near industrial sources of air contaminants. Increasing air pollutant exposure reduced canopy stem density, but abundance of vegetation in other strata tended to increase along the same

gradient. The relative importance of sugar maple (*Acer saccharum* Marsh.) was greatly reduced in all strata with increasing pollutant dose, while yellow buckeye (*Aesculus octandra* Marsh.) appeared tolerant of poor air quality. In the shrub layer the importance of spicebush (*Lindera benzoin* [L.] Bl.) increased with increasing pollutant exposure.

In southern California the predominant native shrubland vegetation consists of chaparral and coastal sage scrub. The former occupies upper elevations of the coastal mountains, extending into the North Coast ranges, east to central Arizona, and south to Baja California; while the latter occupies lower elevations on the coastal and interior sides of the coast ranges from San Francisco to Baja California. Westman (1979) applied standard plant ordination techniques to these shrub communities to examine the influence of air pollution. The reduced cover of native species of coastal sage scrub documented on some sites was statistically indicated to be caused by elevated atmospheric oxidants. Sites of high ambient oxidants were also characterized by declining species richness.

Influence of air pollution stress on succession and ecosystem species composition probably varies with the age and successional status of the forest. Harkov and Brennan (1979) have observed that most woody plants susceptible to ozone injury are generally early successional plant species. Most trees intermediate or tolerant of ozone stress are typically mid- or late successional types. It is not unreasonable to propose, as Harkov and Brennan did, that late successional forest communities may be the most resistant to compositional change as a result of chronic air pollution exposure. Mature ecosystems are also typified by other characteristics that may increase their resistance to air pollution stress. Low net production may reduce potential importance of restrictions imposed by air contaminants on photosynthesis. Closed and slow nutrient cycling may make nutrient capital less liable to loss by air pollutant influence.

There is increasing appreciation of the importance of allogenic forces on forest ecosystem succession. The significance of fire and wind stress on forest development is substantial in certain environments. It is concluded that air pollutant impact may also exert critically important control over forest succession and species composition. Long-term, continual stress tends to decrease the total foliar cover of vegetation, decrease the species richness, and to increase the concentration of dominance by favoring a few, tolerant species.

CONCLUSIONS

Large areas of the temperate forest ecosystem are currently experiencing major perturbation from air pollution. The influence of a variety of air contaminants on biogeochemical cycling, patterns of succession and competition and individual tree health, designated Class II interactions (Smith 1980), are causing significant forest change in the temperate zone. At the ecosystem level the major perturbations include decreased productivity, biomass and diversity; at the community level reduced growth; and at the population level altered species composition. Early and mid-successional forests are concluded to be at particular risk. Temperate forests have historically been subjected to major change resulting from the activities of human beings. For centuries the major influence was gross destruction for agricultural, fuel or other wood-product purposes. In the present Century reduced need for agricultural land and increased forest management has reduced the adverse impact on forests in temperate latitudes. Human activities of primary contemporary importance to forest structure and function have included the introduction of exotic arthropod and microbial tree pests into forest systems lacking evolutionary exposure to these destructive agents, enhancement of native and natural stresses by cultural practices, and the creation of artificial forests of one or a few commercially important species. In the past several decades, however, we have accumulated sufficient evidence to indicate that an additional major anthropogenic modifier of temperate forest ecosystem development is air pollution.

Research

During the last decade forest researchers have outlined numerous Class II interactions by largely utilizing relatively young forest plants grown in controlled environment facilities. During the next decade we must make an effort to perform experiments in natural forest ecosystems to confirm our hypotheses that ambient air pollution is reducing forest productivity and altering species composition.

The very highest research priority is reserved for the establishment of comprehensive investigations to systematically examine Class II interactions in forest ecosystems located in those portions of the temperate zone particularly subject to air pollution stress. These investigations should include analysis of air contaminant influence on soil metabolism and structure, nutrient cycling, tree reproduction, photosynthesis and respiration, important arthropod and microbial pathogens, foliar symptoms of important vegetation in all

forest strata and a careful examination of forest productivity and alterations in successional trends and species dominance. These studies will be of extended term. They will require the participation of numerous scientific disciplines, minimally including pathology, entomology, meteorology, soil science, soil microbiology, ecology and systems analysis. Continuous meteorological and air quality monitoring will be required. Air pollutants measured should include sulfur dioxide, nitrogen oxides, hydrocarbons, ozone and particulates, the latter to include determination of sulfates, nitrates and trace metals. Precipitation acidity will be routinely determined. The objective of these comprehensive studies will be to clarify and quantify various Class I and II interactions. The ecosystems will be evaluated for their ability to resist (inertia) and respond (resilience) to disturbance from air pollution stress. Model development for the various interactions will hopefully allow future projections, given various air quality scenarios, and allow extrapolation of findings to other ecosystems.

In the United States the only research program presently addressing this need is the oxidant study in progress on the San Bernardino National Forest in California. It is imperative that additional investigations be initiated as soon as possible. The studies should be established in those areas judged to be under the greatest stress and they should be initiated, where possible, in association with integrated and comprehensive forest ecosystem studies currently in progress. Priority forest ecosystems in the United States include: 1) Northern Hardwood forest, 2) Central Hardwood forest and 3) Western Montane forest (San Bernardino project in progress). Appropriate locations, in terms of existing research facilities or abundant ancillary information, for the Northern forest are the Hubbard Brook Experimental Forest in New Hampshire, the Isle Royale National Park, Michigan and the Itasca Forest, Minnesota. In the Central forest the Camp Branch Forest watershed in east-central Tennessee and the Coweeta Hydrologic Laboratory in western North Carolina would be appropriate. With regard to location, the Wayne National Forest in Ohio would appear to represent an interesting research opportunity. In addition to the San Bernardino Forest study, the Andrews Experimental Forest, Oregon and the Bitterroot National Forest, Idaho would be other strategically located sites for the Western Montane forest.

Policy

It is recognized that air pollution is one of the most significant contemporary anthropogenic stresses imposed on temperate forest ecosystems. Gradual and subtle change in forest metabolism and composition over wide

areas of the temperate zone over extended time, rather than dramatic destruction of forests in the immediate vicinity of point sources over short periods, must be recognized as the primary consequence of air pollution stress. This realization means that forest interactions with air contaminants must be given consideration in deliberations concerning clean air laws and regulations, alternative energy strategies, industrial and transportation location and forest research funding.

LITERATURE CITED

- Bormann, F.H. and G.E. Likens
1979. Catastrophic disturbance and the steady state in northern hardwood forests. *Amer. Scien.* 67:660-669.
- Brandt, C.J. and R.W. Rhoades.
1973. Effects of limestone dust accumulation on lateral growth of forest trees. *Environ. Pollu.* 4:207-213.
- Cobb, F.W. and R.W. Stark.
1970. Decline and mortality of smog-injured ponderosa pine. *Jour. For.* 68:147-149.
- Davis, D.D., and R.G. Wilhour.
1976. Susceptibility of woody plants to sulfur dioxide and photochemical oxidants. Publica. No. EPA-600/73-76-102, 71 p., U.S. Environ. Prot. Agency, Corvallis, Ore.
- Frissell, S.S., Jr.
1973. The importance of fire as a natural ecological factor in Itasca State Park, Minnesota. *Quat. Res.* 3:397-407.
- Hajdúk, J. and M. Ružicka.
1968. Das Studium der Schäden an Wildpflanzen und Pflanzengesellschaften verursacht durch Luftverunreinigung. In *Air pollution*. Woningen, Pudoc, p. 183-192.
- Harward, M., and M. Treshow.
1975. Impact of ozone on the growth and reproduction of understory plants in the aspen zone of western U.S.A. *Environ. Conserva.* 2:17-23.
- Harkov, R., and E. Brennan.
1979. An ecophysiological analysis of the response of trees to oxidant pollution. *Jour. Air Pollu. Cont. Assoc.* 29:157-161.
- Hayes, E.M., and J.M. Skelly.
1977. Transport of ozone from the Northeast U.S. into Virginia and its effect on eastern white pines. *Plant Dis. Reprtr.* 61:778-782.
- Heck, W.W., A.S. Heagle, and E.B. Cowling.
1977. Air pollution: Impact on plants. In *New directions in century three: Strategies for land and water use*, p. 193-202, Proc. 32nd Annual meeting, Soil Conserva. Soc. Amer., Aug. 7-10, 1977, Richmond, Vir.
- Heinselman, M.L.
1973. Fire in the virgin forests of the Boundary Waters Canal Area, Minnesota. *Quat. Res.* 3:329-383.
- Henry, J.D. and J.M.A. Swan.
1974. Reconstructing forest history from live and dead plant material: An approach to the study of forest succession in southwest New Hampshire. *Ecology* 55:772-783.
- Jensen, K.F., L.S. Dochinger, B.R. Roberts, and A.M. Townsend.
1976. Pollution responses. In *Modern methods in forest genetics*, J.P. Miksche, ed., p. 189-215. Springer-Verlag, New York.
- Loucks, O.L.
1970. Evolution of diversity, efficiency and community stability. *Amer. Zool.* 10:17-25.
- McClenahan, J.R.
1978. Community changes in a deciduous forest exposed to air pollution. *Canad. Jour. For. Res.* 8:432-438.
- Miller, P.R.
1973. Oxidant-induced community change in a mixed conifer forest. In *Air pollution damage to vegetation*, J. Naegle, ed., Adv. Chem. Series No. 122, p. 101-117, American Chemical Society, Washington, D.C.
- Miller, P.R. (ed.)
1977. Photochemical oxidant air pollutant effects on a mixed conifer forest ecosystem. A progress report. Publica. No. EPA-600/3-77-104, p. 338, U.S. Environ. Prot. Agency, Corvallis, Ore.
- Niklfeld, H.
1970. Pflanzensoziologische Beobachtungen im Ruchaschadengebiet lines aluminiumwerkes. *Zentbl. Ges. Forstw.* 84:318-329.
- Odum, E.D.
1969. The strategy of ecosystem development. *Science* 164: 262-270.
- Raup, H.M.
1957. Vegetational adjustment to the instability of the site. In *Proc. 6th Technical Meeting of the Inter. Union for the Protection of Nature*. June 1956, Edinburgh, p. 3-48.
- Smith, W.H.
1980. Air pollution and forests.

Interactions between air contaminants and forest ecosystems. Springer-Verlag, New York (in press).

Stephens, E.P.

1955. Research in the biological aspects of forest production. Jour. For. 53:183-186.

Stephens, E.P.

1956. The uprooting of trees: A forest process. Soil Sci. Soc. Amer. Proc. 20:113-116.

Treshow, M.

1968. The impact of air pollutants on plant populations. Phytopathology 58:1108-1113.

Treshow, M., and D. Stewart.

1973. Ozone sensitivity of plants in natural communities. Biol. Conserva. 5:209-214.

Weinstein, L.H. and D.C. McCune.

1979. Air pollution stress. In Stress physiology and crop plants, H. Mussell and R. Staples, eds., p. 328-341. John Wiley and Sons, New York.

Westman, W.E.

1979. Oxidant effects on Californian coastal sage scrub. Science 205:1001-1003.

Whittaker, R.H.

1975. Communities and ecosystems, 385 p. Macmillan Co., New York.

Woodwell, G.M.

1974. Success, succession and Adam Smith. BioScience 24:81-87.

Secondary and Interactive Effects of Chronic Gaseous Pollutant Exposure of Producers, Consumers, and Decomposers

Influence of Chronic Air Pollution on Mineral Cycling in Forests¹

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Abstract: This paper reviews the literature concerning the impact of chronic air pollution on mineral element cycling in forests. The concept involves the forest trees taking up essential and other elements from the soil and surrounding environment eventually to return them to the soil upon mortality and decay. Chronic pollutants are considered in the context of this cycling as another form of elemental addition to the site subject to cycling in the same manner. The review is organized to assess the addition of the major elements, carbon, nitrogen, and sulfur, and the trace metallic elements. In addition, pollutant effects upon the ecosystem living components in terms of producers and decomposers are considered along with potential changes in the redox and pH state of the different portions of the forest. Some original data on foliar composition of Big Cone Spruce (*Pseudotsuga macrocarpa* (Vasey) Mayr) in relation to pollution exposure, and the evaluation of a soil subject to increments of hydrogen in simulated acid rain leaching are presented.

INTRODUCTION

This paper will present a review of various aspects of mineral element cycling in forests as affected by chronic atmospheric pollution.

The forest will be considered to be the stand of trees and that portion of the atmosphere encompassed by canopy and trunk space, and the portion of the soil encompassed by the root space. Mineral cycling is the process of cycling of elements from the soil through uptake by roots or by foliage, transport within the trees and the eventual return to the forest soil in the processes of foliar leaching, root exudation, foliage drop, etc. Return to the soil storage occurs, and the cycle is continued by uptake again, or it may be broken by loss from the soil, or tieup as insoluble precipitates or compounds. Air pollution exposure will be considered to be the additional atmospheric inputs which enter the nutritional and elemental cycle of forests due to man made effects on atmospheric composition.

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ELEMENTAL CYCLING

The idea of the cycling of elements from vegetation to soil, following uptake from soil probably outdates written records as is apparent from the primitive agricultural practice of shifting cultivation wherein a forest is cut and burned for soil enrichment by the ash. This is followed by the regrowth of a fallow forest crop, which in turn is burned again. The forest vegetation cycles elements essential for vegetative growth as well as unnecessary elements accidental to the particular environment. The 16 or 17 essential elements (C, H, O, P, K, N, S, Ca, Fe, Mg, Mn, Cu, Zn, Mo, B, Cl, and perhaps Se), are cycled by necessity for without them there would be no forest. In addition almost every other element is translocated by plants. This concept is used in geochemical prospecting for many of the more valuable elements, using the plant as a chemical prospector (NASA 1968). These elements which are cycled may be enriched in concentration in various portions of the cycle, as in the foliage, the plant detritus on the ground, or in the surface of the soil beneath the plants. Such enrichment is often the indication of an extraneous element. Similarly the input of extraneous elements in the process of pollution may be evidenced by enrichment over the normal background amounts.

The state of the vegetation and its capacity for storage of elements will determine the relative change induced by the addition of a pollutant element. The state of the vegetation is partly determined by the fact that vegetation is a giant reduction reaction in which photosynthesis reduces carbon from carbon dioxide to carbon in reduced carbon compounds as described by Stumm and Morgan (1970). The forest trees carry out this reduction reaction, followed by their death and the subsequent oxidation of the carbon compounds in the soil. The cycling of the elements on the site occurs in the context of this giant redox reaction (Garrels, et al., 1975), and its state and magnitude determine the rate and amount at which the other elements will be cycled.

The cycling of most of the elements which undergo redox changes in the forest nutrient cycle (C H O N S Mn Fe) will tend to be reduced in the living vegetated portion of the cycle and complete the cycle to an oxidized form in the soil. However, local soil factors favoring anaerobic conditions such as water in excess, may cause the soil to become a sink for some elements in the reduced form, or for other elements in the oxidized form.

Thus pollutants may enter the elemental cycles in forests and vary in mobility depending upon the redox potential at various portions of the elemental cycle. Some pollutants may also be oxidants in relation to the portion of the ecosystem that is absorbing them and thus affect the redox potential at that point according to Haagen-Smit (1958).

STORAGE POINTS AND SINKS

The soil and vegetation of the forest have several points of long term storage of materials added to the elemental cycling system. It has been learned from past fertilization experience that elements added to the forest, either to soil or direct to trees, may be stored for varying periods of time in different portions of the forest. Obviously, the tree trunks provide storage of the materials contained in the wood for the length of life of the tree and its subsequent decomposition time. Deciduous portions of the tree retain materials for lengths of time proportionate to their residence time on the tree; bark for longer periods, leaves and twigs for lesser periods. The storage time in the decomposing or oxidizing portions of the detritus dropped to the soil depends upon local environmental conditions of temperature, redox state, and availability or toxicity of elements to decomposing (oxidizing) organisms. For example, decomposition times for surface detritus on the soil surface in a forest may vary from a fraction of a year to many years.

The soil beneath the forest is a giant fixed-bed ion exchanger, and once elements enter the soil following release from storage in the vegetative portion of the forest and its detritus they may be stored on this exchange complex. The capacity of this ion exchange bed as well as the nature of associated cations will determine the storage probability of elements added to the forest whether as additives (pollutants, fertilizers), or in the normal course of mineral weathering or rainfall additions. The total capacity of this bed for cation exchange in coniferous forest soils ranges from 40 to 300 gm equivalents per meter squared to a meter depth (Zinke, et al., 1979). Whether the pollutant added has a significant effect depends upon the natural base line composition of other cations on the soil column (usually H^+ , Ca^{++} , Mg^{++} , K^+ , Na^+) maintained by the forest and the lyotropic series (leaching precedence) of these elements. The forest soil also has anion exchange capacity although usually of lesser amount. Thus, depending upon the nature of the pollu-

tant additive, and whether it attains ionic form, it may be subject to scrubbing out at the soil exchange complex stage of the elemental cycle. This would apply to the hydrogen and sulfate of acid rain, or the cationic forms of some metallic pollutants such as lead or zinc. Laboratory simulation through leaching with successive increments of the pollutant such as acid rain can be made readily to determine this capacity, and an example of this follows later.

The soil has other capacities for storage of additives in addition to that of the ion exchange capacities. Either following breakthrough or saturation of the exchange capacity, an additive element may be stored on the metallic oxide complex of the soil, or as insoluble precipitates or oxides. The release and uptake of elements into storage in association with the metallic oxides (usually iron and manganese) will depend upon the pH and redox potential of the soil as reviewed by Jenne (1968). Retention will be least under the reducing conditions brought about by large amounts of organic matter and poorly drained - wet conditions.

Thus, the forest soil is a major determinant in the fate of the chronic pollutant element added to the forest. The break through capacity of the soil for the added material will depend upon the current input of similar elements by the natural cycling of cations from the tree cover, the inputs by current mineral weathering, the current additions of hydrogen from rainfall and organic compounds produced by the forest, the depletion of basic metallic cations taken up by the tree growth as the trees age on the site. The greater the storage capacity of the soil either as ion exchange or metallic oxide (hydrous) capacity the greater the buffering capacity of the forest on the input of pollutants.

It will be of interest now to review some of the experience of how the various pollutants behave in relation to these processes of elemental cycles in a forest.

POLLUTANT ADDITIVES TO FOREST CYCLES

Any of the elements added to the forest as pollutants can influence elemental cycling, either by acting as nutrient elements required for the growth of trees, by changing the redox state of any part of the forest, or by changing the pH at some point in the system. Some of this interaction may be physiological in the vegetation, or in the soil micro-

flora, or may be a matter of soil chemistry. Growth rates of the vegetation may be changed, thus affecting the rates of elemental cycling. Each pollutant may have separate effects depending upon its chemical and physiological nature, and there may be synergistic effects with combinations of additives. Some of the major chronic pollutants will be considered separately.

Carbon Compounds and Accompanying Oxidants

Carbon additions to the forest may be important because of the role they play in plant growth and subsequent mineral cycling. Finlayson and Pitts (1976) have reported that carbon may account for 45% of the mass of smog aerosol. Much of this may be the result of secondary reactions creating carboxylic acids, esters, carbonyl compounds, alcohols, peroxidic polymers, long chain alkanes and alkenes, and fatty acids. In addition, carbon in the forms of various compounds is a world-wide atmospheric pollutant due to human activities oxidizing reduced carbon for energy, and to indirect effects such as enhanced oxidation of soil organic matter due to clearing forestland for agriculture. Thus carbon dioxide as a chronic pollutant has been measured by observations at Mauna Loa (Hawaii) to be rising at the rate of from .35 to 1.79 ppm per year to a level of 325 ppm in 1974 (Hobbs, et al., 1974). Peterson (1969) in a review stated that carbon dioxide at its present level in the atmosphere is still limiting to plant growth and that the productivity of plants should increase as the CO₂ content increases. A resulting increase of plant and forest growth of 5% by the year 2000 is predicted, and this would increase the rates of uptake of necessary growth elements from the soil, thus increasing rates of elemental cycling. This all assumes other elements or requirements such as water are not limiting to plant growth. On the other hand, reduction in plant growth may occur where toxic carbon compounds from pollution cause damage to tree foliage, thus reducing rates of mineral cycling. Ozone produced as a secondary product from ultraviolet radiation on hydrocarbons in the atmosphere may reduce plant growth. Evans, et al. (1974) found ozone contents of 0.1 ppm at the top of a smoke plume from a forest fire. However plants synthesize methyl chloride according to Lovelock (1975), and this may destroy such added ozone. The oxidants formed from the organic pollutants introduced to the atmosphere will have adverse effects upon various amino acids in plants resulting in damage

(Haagen-Smit, 1958). Fatty acids on plant surfaces may be altered by the singlet oxygen resulting from NO_2 , benzaldehyde, and polynuclear hydrocarbons, increasing saturated acid content of the foliage according to Dowty et al. (1973). Other oxidants such as peroxyacetyl nitrate (PAN), hydroxyl OH, hydroperoxyl HO_2 , also have similar effects on vegetation. Their effects as oxidants are defined by their capability to oxidize iodide ion in aqueous solution of potassium iodide according to Kuntz, et al. (1973). Also, their action may be synergistic, so that although one or another is in low atmospheric concentration the sum of the oxidants must be considered in effect on the vegetation. Also, the volatile terpenes produced by the forest itself must be taken into account, and according to Rasmussen (1970), as many as 100% of the species in some western coniferous forests may produce such volatile hydrocarbons.

Nitrogen

Nitrogen as it cycles in the forest is in reduced form in the plant and following mineralization to ammonia in decomposing detritus in the soil it is gradually oxidized to nitrate where it is again taken up by plants to be reduced in various nitrogen compounds. Nitrogen is added as a chronic air pollutant as ammonium, and nitrate nitrogen which also is the end product of oxidation of NO by ozone. Where nitrogen is limiting in the forest as an essential element this should stimulate forest growth and decomposition processes provided other elements are not limiting. However, if other elements are limiting this would tend to emphasize the other deficiencies. Additions as a chronic pollutant should show similar symptoms to those of the addition of nitrogen as a fertilizer. Thus foliar nitrogen contents should be higher. A result similar to this encountered in Big cone spruce foliage in the San Bernardino mountains is reported later in this paper.

Sulfur

The sulfur cycle in forests is such that the sulfur is in reduced form in association with organic compounds in the plants, but is oxidized upon decomposition in the organic detritus returned to the soil. The decomposing soil microflora convert the sulfur to sulfate. The degree to which this takes place depends upon the redox potential of the soil. For example, if it is very low, the sulfur will be retained in sulfide form, frequently tying up heavy metals.

The forms of sulfur added in chronic atmospheric pollution have been SO_2 , SO_4^{2-} , with associated cations H^+ , NH_4^+ , and HSO_4^- as determined in the pollutant plume from St. Louis Missouri by Charlson, et al. (1973). They found that tropical air masses were dominated by the more acidic NH_4HSO_4 while northern air masses were less acidic and dominated by $(\text{NH}_4)_2\text{SO}_4$ (1975). Usually the sulfur in the atmosphere will be oxidized to sulfate and this will occur either as ammonium or calcium sulphate dust according to Haagen-Smit (1958), and since trapping surfaces of vegetation are aerobic, any reduced sulfur will be oxidized to sulfate in the foliage.

Since sulfur is an essential element for plant growth, the additions may or may not be deleterious. Actual deleterious effects would occur to trees if sulfur dioxide were the main additive. This would occur only close to the emission source as has happened at Kennett, California; Copper Basin, Tennessee; and Sudbury, Ontario. At greater distance this would be oxidized. Presumably if the pollutant were added in extreme amounts, the titrable acidity produced would break through the soil exchange capacity after depleting other cations present. An example of a simulation of this is offered in table 2 explained later.

Trace Elements as Washout, Particulates, Dusts

Most of the elements other than C, N, and S will be added in dust and rain as chronic pollutants. The particulates in atmospheric pollution contain elements such as lead, sodium, magnesium, aluminium, vanadium, and zinc in Los Angeles type smog according to Finlayson and Pitts (1976). Sedimentary cores taken offshore in the Pacific Ocean by Bruland, et al., (1974), indicated that lead, silver, copper, zinc, chromium, nickel, molybdenum, and cobalt are chronic pollutants in the Los Angeles area. Lead, iron, manganese, nickel, copper and zinc were analyzed in precipitation as common trace element pollutants throughout the U.S. according to Lazarus et al. (1970). Beryllium is common in industrial dusts near sources. The amount of trace element fall out lessens as the square of the distance from sources according to Bertine & Goldberg (1974). Cannon and Boles (1962) found that highways represented linear sources for lead, the amount in vegetation decreasing rapidly with distance from the source. Dedolph et al. (1970) found that this diminution was logarithmic with distance for particulate lead. However, some elements such as cadmium, nickel, lead, and

zinc may be vaporized at the source and carried longer distances according to Lagerwerff & Specht (1970).

The tracing of input of trace elements in the elemental cycles in vegetation has been attempted by many investigators. Chester and Stoner (1973) used an enrichment factor in which a ratio of the element to iron, in the particulate being added is divided by the average ratio of the element to iron in the earth's crust. They found that tin, lead, and zinc most often were the elements enriched by pollution. Peirson, et al. (1974) employed an enrichment factor in the form of the ratio of the element to Scandium content, compared to the same for the local soils. They found the soils enriched for V, Co, Ni, Zn, As, Se, Sb, and Pb this way. Similar enrichment ratios for other pollutants should identify augmentations of the elements in elemental cycles in forests.

The contents of pollutants in the plants should be identifiable by anomalies in composition. Thus, Schacklette and Connor (1973) used the regional variation of vanadium in Spanish moss (*Tillandsia*) along the gulf coast of the U.S. to identify areas where chronic pollution by airborne vanadium occurs. Contents as high as 560ppm V were found in certain areas, and these were assumed to represent the output from oil refineries using Venezuelan crude oil which is high in vanadium. In soil-vegetation cycling, vanadium is frequently held at the soil-root interface due to immobilization in the oxidized state (pentavalent form). Thus, the redox state of the soil-vegetation system and its components in the forest may determine where a chronic polluting element may be immobilized. Elements undergoing changes in solubility with various oxidation states may be immobilized at various interfaces where pH or Redox change. Chromium, vanadium, manganese, and iron are susceptible to this.

Some of the chronic pollutants added to the elemental cycle of forests may later be released from the forest by vaporization. For example, Curtin et al. (1974) found that tin is transpired in vapors from conifers to the amount of 23-80 ppm in the residue of the vapor. They found that some coniferous twigs contained up to 6-40 ppm tin in the ash. The tin was added as atmospheric pollutant in dust from industrial areas. However, no mention was made of this as being deleterious.

The inputs of dry particulate materials as dusts may be washed off by rain. This washoff may be a major process of cycling of the added element to soils. Carlson et al. (1976) found that a simulated rain removed 45% of an applied aerosol of $PbCl_2$, and that light misty rain is most effective. Heichel & Hankin (1972) found that the particulates in which lead adhered to trees averaged 7 micrometers in diameter. Chlorine and Bromine were the main associated negative elements associated with them. The assessment of the intensity of input of chronic pollutants such as these have been made by analyzing concentrations on tree bark by Lotschert (1977), and Grodzinska (1977). Some of these accumulations are washed down the tree trunk by stem flow and may be accumulated in the soil at the tree base.

POLLUTANT INPUT IN PRECIPITATION

Washout as well as input of elements in solution in precipitation is a major path of input to the elemental cycling that occurs in forests. For example, the nitrogen content stored in a 1000 year old redwood forest is about the amount contributed by rainfall during that time period (Zinke, et al., 1979). The geochemistry of precipitation was reviewed thoroughly by Carroll (1962). She found that the cations which balance the ions in rainwater were mostly basic metallic elements near coast lines or arid regions, but that along the storm vector from these sources and over regions with more vegetation, there would tend to be acidification due to less dust or ocean aerosols to provide the basic metallic cations. The control of wind erosion over the Great Plains thus would tend to add to the acidification of rainfall downwind from the area. However the current emphasis on pollution has added the consideration of sources of anions in precipitation that come from burning fossil fuels. As Carroll (1962) mentioned, a pH below 5.7 indicates that hydrogen is ionizing to meet the necessary balance with anions.

The effect of precipitation input upon nutrient element cycling in a forest will tend to be specific to the site conditions. This would be determined by the following local factors: 1. background natural composition of the soil solution. 2. the cation exchange capacity of the soil and the composition of cations occupying this capacity. 3. the volume of water entering as rainfall, 4. the acid base balance of the rainfall, 5. the nature of the vegetation species with

regard to intensity of cycling basic metallic elements, and perhaps others unique to the site. The effects of input in precipitation will most obvious on sites with soils having low exchange matter content soils) and with sparse slow growing vegetation (pines, spruces, heathland), and in climates with large amounts of precipitation.

FOREST ECOSYSTEM EFFECTS

The mineral cycling in a forest takes place in the context of an ecosystem of processes linking the various living organisms and the components of the environment. Chronic atmospheric pollution is one of these components. The elements added will be utilized by the Producers in the forest. These are the organisms which carry out the reduction of carbon to carbon compounds which comprise the forest. This fixed carbon then serves as an energy source for the Decomposers in the forest which also have requirements for the nutrient elements either in the biomass of the producers or furnished directly by the soil. Thus pollutant elements will interact in the cycle between Producers and Decomposers, with storage of elements and possible sinks of unavailable material accumulating in the soil portion of the cycle.

Effects of Chronic Pollution on Producers

These effects can range from enhancement of productivity to a marked decrease depending upon whether the additive is an element currently limiting growth or is a toxic element. If toxic, the effects will show up in foliar damage, or interference with various physiological processes. The symposium proceedings edited by Naegele (1973) contains many descriptions of these toxic effects. Mc Cune in this publication felt that such toxic effects should be specific with regard to the Receptor (leaf, tree, forest), the Pollutant, the Event, and the Environment. Acute effects are relatively easy to determine, as in the death of leaves near a smelter releasing SO_2 . Obviously this would decrease the rate of elemental cycling at the site along with a brief surge of added materials to the soil as the defoliation took place and leaves dropped to the soil to be decomposed. Whether this is a loss to the site depends upon the capacity of the underlying soil as discussed in relation to defoliation by herbicides by Zinke (1974). The effects are more difficult to assess in the case of chronic low level pollution as discussed by Feder (1973). The changes in productivity

would need to be statistically significant before and after the initiation of the pollution. Miller (1973) has made an approach to indices of level of chronic pollution in terms of visual characteristics on the trees; needle retention, needle length, needle chlorosis, branch death in the case of ponderosa pine. Foliage composition should indicate such inputs, and a paper by Arkley and Glauser in this symposium covers this. Also an example in the case of Big Cone Spruce foliage is presented in table 1. McBride, et al. (1975) made measurements of actual growth rate retardation of 26% for height and volume growth of ponderosa pine. In agricultural plants, White (1974) found a decrease in CO_2 uptake due to synergistic effects of NO_2 + SO_2 when each were present at levels not considered inhibiting to growth. Stimulation of primary producers by pollution is not often reported due to the negative connotation of the term. However, Schnappinger (1975) found an increase in growth due to response to zinc contained in fly ash.

There are numerous variables which may affect the intensity with which a given level of pollutant will affect the producers in an ecosystem. Thus the trapping efficiency of the foliage surface is different for various species as discussed by Zinke (1966), the velocity of the air past the trapping surface as has been evaluated by Hori (1953) and Slinn (1976). Following trapping the material may not be absorbed by the surface as found by Motto (1970). This is partially attributable to the particle size according to Natusch & Wallace (1974). There may be synergistic effects at the foliage surface. Lovelock (1974) has alluded to the possible creation of PAN on foliage surface by the combination of O_3 and hydrocarbons as occurs on dry-
ing linen.

Once absorbed by the producer the polluting element will enter the elemental cycle on the site. The disposition from the plant may be not only through leaf drop, but may be by direct root exudation as Olson et al (1962) found with additions of radioactive Cesium added to a tree trunk. However upon being dropped to the soil as detritus or entering in other ways, the polluting element will then be affecting the decomposers in the forest.

Effects of Chronic Pollution on Decomposers

The effects upon decomposers, the soil microflora and fauna that complete

the elemental cycles in the soil, will be variable. As with the producers which fix carbon dioxide, the additives to the site through air pollution may increase or decrease their activities. These organisms require the same nutrient elements as the producers as well as the additional ones of sodium, iodine, and vanadium for the soil fauna. If the pollutants are added in toxic quantities they would presumably retard the decomposition of organic matter in the soil and thus slow or block elemental cycling. They may be sublethal or even mutagenic to these organisms according to Stotzky (1974).

The detritus or litter layer on the forest floor is the first major area of activity of the decomposers. Wittkamp & Frank (1969) found that litter samples impregnated with introduced elements such as Cobalt 60 and Cesium 137 had logarithmic relationship with time in the fraction remaining, and that loss coefficients were different for different elements. These varied with the species of tree yielding the detritus. The storage period of the additive element depended upon relative chemical mobility of the element, composition of other elements present, and the moisture and temperature conditions of the site.

If a pollutant is an element which is limiting rate of decomposition, the effect on the decomposers may be positive. For example leaf litter with too high a Carbon/Nitrogen ratio (nitrogen limiting) decomposes very slowly. Adding nitrogen as a chronic pollutant would accelerate this decomposition and release stored elements more rapidly.

The soil fauna serve a major task in elemental cycling in a forest ecosystem by physically decomposing organic detritus, reducing it in size and increasing surface area to bring about faster rates of fungal and bacterial decomposition. Earthworms of various species were found by Gish & Christensen (1973) to accumulate trace elements from particulate pollution in order of increasing atomic weight: $Ni < Zn < Cd < Pb$. The soil fauna may serve as a link for transfer of nutrient elements to various predators in the forest ecosystem.

Stotzky (1974) characterizes the soil as both an abiotic as well as biotic sink for pollutants. Ethylene and CO are degraded by soil micro organisms. Fungi in soil metabolize volatiles from plant foliage such as ethane, butane, etc. Thus soil by virtue of the various organisms present has a strong capability to

detoxify pollutants. At low concentrations pollutants may bring about an enrichment of the micro organisms which can utilize them.

The ultimate role of the decomposers is thus to cycle elements from the vegetation to the soil, usually oxidizing them as energy sources, and ultimately releasing them to the chemical and physical processes of the soil or renewed uptake by the roots of the producers.

Entry of Chronic Pollutants into Soil Sinks

The final fate of additives to the ecosystem may be to remain in the soil in insoluble form. Thus once processed by the decomposers the elements may become mineral precipitates which may act as sinks (Nat. Res. Council 1977). Thus fluorine added through pollution may be precipitated in the soil as insoluble CaF_2 . Page and Ganje (1970) showed that the top 2.5 cm. of soil was most active in trapping lead entering the soil-vegetation system in southern California. Soils sampled in 1967 were compared with previous samples obtained in 1919. An average increase of 15-36 ppm lead accounted for 1/5 of the lead contained in the 47 billion gallons of gasoline which had been burned during the period encompassed. Keaton (1937) added large amounts of lead nitrate to soil (2784 ppm) and found that very little (17 ppm) remained in soluble form. Lee & Tallis (1973) found it possible to date periods of lead pollution in a peat bog in Great Britain, finding a peak in 500 A.D. with the operation of Roman lead mines, and a subsequent industrial peak beginning about 1460 AD. Similar but recent historical evidence of lead collection in ecosystem sinks was found in wood content of lead dated by annual rings by Sheppard & Funk (1975).

Soil organic matter generally enhances the role of soil as a sink for added trace elements. Furr et al (1971) have noted that pH will determine whether soil is a source or sink. Thus, at neutral to slightly alkaline pH; As, B, Cd, Co, Cu, F, Fe, Pb, Mn, Ni, Sn, and Zn will tend to be immobilized, but Ca, Cr, K Mg, Mo, Rb, Se, and Sr may become more available to plants. Thus anything causing a change in soil pH may release previous pollutants that had temporarily become trapped in the soil as a sink. Various precipitates such as soil carbonates may form, also depending upon pH.

The soil ion exchange capacity will retain elements in a form ready to be

Table 1: Big cone spruce foliage analyses across gradients of decreasing chronic pollution intensity.

WESTERN SAN GABRIEL MTNS. Angeles National Forest								
Location	Weldon Canyon		Woodwardia Canyon		Cloudburst Canyon		Tie Canyon Summit	
NITROGEN								
percent	1.80		1.17		1.31		1.08	
rank	93		55		70		30	
PHOSPHORUS								
ppm	601		771		656		1545	
rank	7		23		12		69	
N/P								
ratio	20.2		14.1		10.5		7.0	
rank	92		78		61		32	
EASTERN SAN GABRIEL MTNS. Angeles National Forest								
Location	Lower San Antonio		Camp Baldy		Snowcrest Camp		Big Pines	
NITROGEN								
pct.	1.15		1.17		1.05		1.25	
rank	41		45		24		64	
PHOSPHORUS								
ppm	1093		1432		1432		1235	
rank	25		59		59		64	
N/P								
ratio	10.5		8.2		7.3		8.2	
rank	61		44		36		44	
SAN BERNARDINO-SAN JACINTO MTNS. San Bernardino N.F. - Cleveland N.F.								
Location	Lower Waterman		Upper Waterman		Crestline	Camp Angeles	Keene	Palomar
	(young)	(old)	(young)	(old)				
NITROGEN								
pct.	2.12	1.80	2.02	2.04	1.22	.99	1.06	1.13
rank	97	92	96	96	52	13	26	38
PHOSPHORUS								
ppm	648	838	1016	1685	1658	1552	1544	1580
rank	1	7	18	81	79	70	69	72
N/P								
ratio	32.7	21.4	19.8	12.2	7.3	6.4	6.8	7.1
rank	99	94	92	70	36	26	31	34
CHRONIC POLLUTION INTENSITY								
	HEAVIEST		HEAVY		MODERATE-LIGHT		LOW	NIL

Table 2: Simulation of acid rainfall addition by leaching a surface soil developed under the influence of a mature ponderosa pine at the arid limit of the occurrence of the species.
(in milliequivalents per 100 gms. fine earth portion except last column as indicated)

Soil Treatment	Exchange Capacity	Exchangeable Cations Remaining on Soil			H ⁺ %
		[Ca ⁺⁺ + Mg ⁺⁺]	[K ⁺ + Na ⁺]	[H ⁺] ^{2/}	
Original Soil ^{1/}	28.80	25.65	2.85	.30	1.0
Leached soils:					
Leachate Composition ^{3/}					
500 ml H ₂ O		22.00	1.79	5.01	17.4
+5.71 meq. H ⁺		18.60	1.02	7.18	25.0
+11.40 meq. H ⁺		15.00	.77	13.03	45.3
+17.09 meq. H ⁺		10.30	.38	18.12	63.0
+22.9 meq. H ⁺		9.00	0.00	19.80	68.8

^{1/} Fine earth portion, 0-2.5 inch sampling horizon.

^{2/} Difference between the sum (Ca⁺⁺ + Mg⁺⁺ + K⁺ + Na⁺) and the Exchange Capacity.

^{3/} The H⁺ was added as 0.1N HCl to the 500 ml of distilled water, since 1 ml of 0.1N HCl = 0.1 meq. H⁺.

recycled back to the living organisms on the site, and also keep them from leaching away from the site. Depending upon this capacity, elements added through chronic pollution may be retained in the elemental cycle of a forest provided they are not trapped in a sink or leached away.

FOLIAGE ANALYSES INDICATING POLLUTION

The effect of chronic air pollution of the type typical of the Los Angeles - San Bernardino basin on foliar analyses of the Big Cone Spruce (*Pseudotsuga macrocarpa* (Vasey) Mayr) was determined. Samples of the foliage were gathered throughout the range of the species, and also on gradients of elevation or distance from the areas typified by "smog" to areas relatively free of "smog". Analyses were made of 9 major elements in the various age classes of the foliage. The elements most obviously related to the intensity of the smog and their rating in a cumulative probability distribu-

tion from lowest to highest value are shown in table 1.

Nitrogen content of the foliage and the ratio of nitrogen to phosphorus content were the most obvious indicators of the effect of chronic air pollution of the Los Angeles type. Where there was ambiguity in the nitrogen values as in San Antonio Canyon, the nitrogen to phosphorus ratio was the best indicator in this species.

SIMULATED ACID RAINFALL LEACHING

A laboratory experiment was conducted in which samples of a nearly cation saturated soil under the influence of ponderosa pine were leached with solutions containing successive increments of H⁺. The soil samples were obtained at the lower rainfall limit of ponderosa pine where the soil was most likely to be base saturated. Five separate ten gram samples of this soil were leached by solutions which were respectively 500 ml of distilled water, and solutions to

which increments of H^+ as HCl equivalent to 20%, 40%, 80%, of the exchange capacity of the sample were added. The results presented in table 2 show the leaching of the exchange complex of soils that takes place with added increments of a cation. Presumably the additions of increments of H^+ in acid rainfall would have similar effects which would have to be considered in the context of other cations being cycled by the forest, and by the composition of cations already on the exchange complex of the soil. Following this leaching, each soil portion was analyzed for remaining exchangeable metallic cations.

These data show that given the stoichiometry of the leaching of cations from the soil exchange complex by H^+ the effects of increments of acid rainfall to a given soil may be simulated with solutions having compositions similar to that of the rainfall. Also, as the increment of H^+ in the precipitation increased, either because of concentration or of volume of rainfall leaching the soil would tend to become hydrogen saturated, displacing basic metallic cations in proportion. In the forest situation these would be leached out of the soil profile in a base saturated soil, or taken up by the vegetation.

CONCLUSIONS

This review of the literature and the presentation of some original work is applicable to the effect of chronic air pollution on mineral element cycling of forests. The literature is general, illustrating processes, but is not site specific except for the location of the studies. Since effects at any specific site may be in small increments each year, but severe in cumulative effects over a number of years, a major remaining problem is to find ways of determining site specific indications of detrimental effects of such chronic pollution. Some pertinent conclusions from the literature reviewed in this paper are:

1. The forest vegetation as is true of any vegetation requires many essential growth elements which it will cycle in addition to others mobilized by local mineral weathering or input through precipitation.
2. To the extent that chronic pollution adds elements normally required by plants, or chemically mobile in the plant, these will be cycled much as the others on the site.
3. The forest is created in a vast reduc-

tion reaction wherein CO_2 is reduced to organic compounds by Producers in the ecosystem, and subsequently oxidized in energy releasing steps by these plants or by the Decomposers in that ecosystem.

4. The cycling of elements takes place in the context of these redox reactions, and those elements which change state with redox potential change are usually in the reduced form in the producer vegetation and subsequently oxidized in energy releasing steps by these plants or by the decomposers on the site.

5. Chronic air pollution additives to the forest affect the cycling of elements not only by direct additions of the polluting elements to the cycles, but also by affecting the redox status of the absorbing plant tissue, and other portions of the soil-vegetation system.

6. Concentration points and sinks for the elements added in chronic pollution exist in the various parts of the vegetation, the surface detritus on the soil, and in the soil capacities for absorption on ion exchange, hydrous oxide complexes, and as insoluble precipitates.

7. The effects of chronic pollutants added to a given forest and its elemental cycling processes will be highly site specific; depending upon climate, degree of weathering development of the soil, the type of minerals from which the soil is forming, the characteristics of the forest species and their stocking densities, the topographic situation with regard to airflow, and the nature of the pollutant.

8. Local site specific indicators of intensity of pollution effect may be visual as in appearance and size of foliage, foliar analyses for pollutant elements and their ratios to background site elements, and soil assessment of capacities to absorb the added elements.

LITERATURE CITED

- Bertine, K. & E. Goldberg. 1977. History of Heavy metal pollution in southern California Coastal Region. *Env. Sci & Tech.* 11:3 297-99.
- Bruland, K.W., K. Bertine, M. Koides, & E. Goldberg. 1974. History of metal pollution in southern California Coastal Zone. *Env. Sci. & Tech.* 8:5 425-432.
- Cannon, H.L. & J. Bowles. 1962. *Science* 137, 765-766.

- Carlson, R., F. Bazzaz, and J. Stukel. 1976. Physiological effects, wind reentrainment and rainwash of Pb aerosol particles deposited on plant leaves. *Env. Sci & Tech.* 10:12 1139-1142.
- Carroll, D. 1962. Rainwater as a chemical agent of geologic processes. U.S.G.S. Water Supply Paper 1535-G 18pp + 1 plate.
- Curtin, G.C., H. King, & E.L. Mosier. 1974. Movement of elements into the atmosphere from coniferous trees in subalpine forests of Colorado and Idaho. *J. Geochem. Explor.* 3:245-263.
- Dowty, B. J. Laseter, G. Griffin, I. Politzer, & C. Walkinshaw. 1973. Reactions of singlet oxygen with pine pollen. *Science* 181 17 Aug. 1973 669-670.
- Evans, L.F., N. King, D. Packham & E.T. Stephens. 1974. Ozone Measurements in smoke from forest fires. *Env. Sci & Tech.* 8:1 75-76.
- Evans, L. I. Weeks, A. Eccleston, & D. Packham. 1977. Photochemical ozone in smoke from prescribed burning of forests. *Env. Sci & Tech.* 11:9 896-900.
- Feder, W.A. 1973. Cumulative effects of Chronic Exposure of plants to low levels of air pollution. pp. 21-30 in *Air Pollution Damage to Vegetation*. Adv. in Chem. Series #122 Amer Chem. Soc. Wash. D.C.
- Finlayson, B. & J. Pitts. 1976. Photochemistry of the polluted troposphere. *Science* v. 192 111-119.
- Friedlander, S.K. 1973. Small particles in air pose a big control problem. *Env. Sci. & Tech.* 7:13 1115-1118.
- Furr, K., T. Parkinson, R. Hinrichs, D. Van Campen, C. Bache, W. Gutenann, L. St John, I. Pakkala, & D. Lisk 1977. National Survey of Elements and Radioactivity in Fly ashes. Absorption of elements by cabbage grown in fly ash soil mixtures. *Env. Sci & Tech.* 11:13 1194-1201.
- Garrels, R.M., F. McKenzie, C. Hunt. 1975. *Chemical Cycles and the Global Environment*. 206pp. Wm. Kaufmann Inc. Los Altos, Calif.
- Gish, C. & R. Christensen. 1973. Cadmium, Nickel, Lead, and Zinc in earthworms from roadside soil. *Env. Sci. & Tech.* 7:11 1060-1062.
- Goldberg, E.D. 1971. Atmospheric dust, the sedimentary cycle, and man. *Geophysics* 1:117-132.
- Grodzinska, K. 1977. Changes in the forest environment in southern Poland as a result of steel mill emissions. Pp. 207-215 in *Vegetation, Science, and Environmental Protection*. Maurzen, Ltd., Tokyo.
- Haagen-Smit, A.J. 1958. *Air Conservation*. Science 128 no. 3329 869-878.
- Heichel, G.H. & L. Hankin. 1972. Particles containing lead, chlorine, and bromine detected on trees with an electron microprobe. *Env. Sci. & Tech.* 6:13 1121-1122.
- Hobbs, P.V., Harrison, H. and E. Robinson. 1974. Atmospheric effects of pollutants. *Science* 183(4128):909-915.
- Hori, J. 1953. On the diffusion by turbulent motion near a forest and its effect upon the capture of fogs by the forest. *Low Temperature Sci., Ser. A, Phys. Sci.* pp. 75-86. Abstr. in *Trans. Amer. Geophys. Union* 35:2 327.
- Jenne, E.A. 1968. Controls on Mn, Fe, Co, Ni, Cu, and Zn concentrations in soils and water. The significance of hydrous Mn and Fe oxides. *Adv. Chem. Ser.* 73: 337-367.
- Lagerwerff, J. & A. Specht. 1970. Contamination of roadside soil and vegetation with Cadmium, Nickel, Lead, and Zinc. *Env. Sci & Tech.* 4:7 583-585.
- Lazarus, A.L., E. Larange, & J.P. Lodge. 1930. Lead and other metal ions in United States Precipitation. *Env. Sci. & Tech.* 4:1 5 - 58.
- Lee, J. & J. Tallis. 1973. Regional and Historic aspects of lead pollution in Britain. *Nature*, Vol. 245 Sept. 28 1973 216-218.
- Linton, R., A. Loh, D. Natusch, C. Evans, & P. Williams. 1976. Surface predominance of trace elements in airborne particles. *Science* vol. 191 27Feb. 1976 852-854.
- Lotschert, W. 1977. Bark of deciduous trees as an indicator for air pollution. Pp. 247-255 in *Vegetation, Science, and Environmental Protection*. Maurzen, Ltd., Tokyo.
- Lovelock, J. 1975. Natural halocarbons in the air and in the sea. *Nature* Vol. 256 July 17 1975 pp.193-194.

- McBride, J., V. Semion, P. Miller. 1975. Impact of air pollution on the growth of ponderosa pine. Calif. Agriculture Dec. 1975 8-9.
- Miller, Paul R. 1973. Oxidant induced community change in a mixed conifer forest. pp. 101-117 in Air Pollution Damage to Vegetation. Adv. in chem. Series #122. Amer. Chem. Soc. Wash. D.C.
- Motto, H.L. R Daines, D. Chilko, C. Motto. 1970. Lead in Plants and Soils. Env. Sci & Tech. 4:3 231-237.
- National Aeronautics and Space Authority NASA. 1968. Application of Biogeochemistry to mineral prospecting. NASA SP-5056 134pp. Office of Technology Utilization. NASA Wash. D.C.
- National Research Council. 1977. Geochemistry and the Environment. Vols I & II.
- Natusch, D., J. Wallace. 1974. Urban aerosol toxicity; the influence of particle size. Science v. 186: no. 4165 695-699
- Olson, J.S., D.A. Crossley, & M. Witkamp. 1962. Mobility of Cs¹³⁷ tracer in a forest of Tulip Poplar (Liriodendron tulipifera). Health Physics Div. Oak Ridge National Laboratory. Ann. Progress report. No. ORNL 3347 1962 (66). Oak Ridge Tenn.
- Page, A. & T.J. Ganje. 1970. Accumulations of lead in soils for regions of high and low motor vehicle traffic density. Env. Sci. and Tech. 4:2 140-142.
- Peterson, E. 1969. Carbon dioxide affects global ecology. Env. Sci & Tech. 3:11 1162-1169.
- Rasmussen, R. 1970. Isoprene: Identified as a Forest type emission to the atmosphere. Env. Sci & Tech. 4:8 667-671.
- Schacklette, H.J. & J.J. Connor. 1973. Airborne chemical elements in Spanish Moss. U.S.G.S. Prof. Paper 574E 46pp. U.S. Gov. Printing Office, Wash. D.C.
- Schnappinger, M. D. Martens, & C. Plank. 1975. Zinc availability as influenced by application of fly ash to the soil. Env. Sci. & Tech. 9:3 258-261.
- Shephard, J. & W. Funk. 1975. Trees as Environmental sensors monitoring long term heavy metal contamination of the Spokane River, Idaho. Env. Sci. & Tech. 9:7 638-642.
- Slinn, W. 1976. Some approximations for the wet and dry removal of particles and gases from the atmosphere. Jour. Of Air, Water, and Soil Pollution. Accepted for Publ.
- Smith, W. 1973. Metal contamination of Woody Plants. Env. Sci & Tech. 7:7 631-636.
- Stotzky, G. 1974. Activity, Ecology, & Population dynamics of microorganisms in Soil. in Microbial Ecology by Las-kin, A.I. & H. Lechevalier. CRC Press Cleveland Ohio. pp. 73-135.
- Stumm, W. & J.J. Morgan. 1970. Aquatic Chemistry. 583pp. Wiley-Interscience N.Y.
- White, K.L., A.C. Hill & J. Bennett. 1974. Synergistic inhibition of apparent photosynthesis rate of alfalfa by combinations of sulfur dioxide and nitrogen dioxide. Env. Sci & Tech. 8:6 574-575.
- Witkamp, M. & M. Frank. 1969. Loss of weight of Co 60 & Cs 137 from tree litter in three subsystems of a watershed. Env. Sci & Tech. 3:11 1195-1198.
- Zinke, P.J. 1966. Forest interception studies in the United States. pp. 137-161 in Forest Hydrology Proc. of a National Science Foundation advanced Science Seminar Penn. State Univ. Penn. Pergamon Press. N.Y.
- Zinke, P.J. 1974. Effect of Herbicides on soils of South Vietnam. Part B. Working Papers -The effects of herbicides in South Vietnam. 39pp. National Ac. of Sci., Nat. Res. Council. Wash. D.C.
- Zinke, P.J., A. Stangenberger, W. Colwell. 1979. The fertility of the forest. California Agriculture 33:5 10-11.

Forest Genetics and Air Pollutant Stress¹

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Abstract: The breeding of trees, particularly conifers, for cultivation in regions under the influence of industry is of considerable economic importance. The role of genetic polymorphism in the process of adaptation to such environments is discussed. Results are presented of research on the genetic variability in Scots pine populations growing under the influence of industry and in regions free of gaseous pollution. Methods of selecting tree populations tolerant to industrial emissions are discussed aimed at the identification of genetic markers of tolerance.

In order to reduce losses in forest ecosystems caused by industrial emissions attempts were made to determine the sensitivity of trees (depending on their genetic properties) to acute injury of the assimilation apparatus by the most common gaseous pollutants. The degree of plant injury by gases is dependent also on the system of ecological factors operating in the environment.

GENETIC VARIATION OF TOLERANCE BETWEEN AND WITHIN SPECIES

So far a wide scale of genetic variability was observed within various species of trees and shrubs in their sensitivity to airborne pollution. The results listed in reviews (Davis et al., 1976, Białobok 1979) are only approximate. It appears that most tolerant are some species of broadleaf trees. Coniferous trees which are of greater economic importance are generally characterised by low tolerance.

It was also found that there is genetic variability in response to SO₂, O₃ and HF within one population of Pinus sylvestris, P. strobus, P. ponderosa and within Picea abies

as well as in some species of broadleaf trees (Gerhold et al., 1977). The greater tolerance of Scots pine and Norway spruce to the air pollution within a population is sometimes associated with the greater resistance to low temperatures diseases and a xerothermic adaptation (Białobok, 1979). A suggestion also exists that the cause of tolerance to some gases may depend on mechanisms that reduce the access of gases to the plant organism (Gerhold et al., 1977).

However, it has been experimentally established or deduced from reliable evidence that there exist genetic mechanisms conditioning higher tolerance to industrial emissions (Niemtur, 1979). The latter investigator studied 58 progenies of Scots pine from 4 provenances obtained from seeds collected on remnant living trees in the highly polluted conditions of the Upper Silesian industrial region. He has shown that these progenies have a greater tolerance than those collected from trees growing beyond the range of pollution.

POLYMORPHISM AND ITS ROLE

Studies on the genetic variability of enzymes in forest trees developed strongly after Stern, (1974) pointed out the great importance of genetic polymorphism for the development of ecological genetics, forest genetics and tree breeding.

Thanks to this method it is possible to recognize that genetic structure of a population (allele frequencies and degree of heterozygosity) in space and time, to conduct studies on inheritance and linkage, to determine the genetic distance

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between individuals and populations, to identify mechanisms of adaptation to an environment and to employ it in practical forestry.

Polymorphism of forest trees which are long-living organisms represents adaptive strategies of populations to a changing environment (Stern et al., 1974). In this genetic system heterozygotes are favored and thus the population is capable of existing in a heterogenic environment surviving numerous and major changes.

Tree populations in industrial regions grow under similar conditions as the herbaceous plants that colonize mine spills having high concentrations of heavy elements (Pb, Zn, Cu). This phenomenon Stern (1974) also considers as being a good example of the developing polymorphism.

In conditions where at least a part of the trees can survive in an industrial environment that has not been too strongly polluted, the population adopts the optimal strategy in reaction to the changes taking place. This, of course, is the result of the evolutionary adaptation process to the new environmental conditions where the genetic system is under a constant selection pressure. Tolerant genotypes appear to have a certain level of fitness to the industrial environment. What is most important to us is that the drastic changes in the industrial environment should not exceed the level of genetic flexibility of the populations of the most important forest forming species. In the acute conditions of change in an industrial environment, the coded genetic system within a population, that conditions its adaptive capacity, gradually ceases to operate since the potential of individual genotypes for adaptation has been surpassed.

GENETIC VARIABILITY OF ENZYMES IN SCOTS PINE POPULATIONS

The use of electrophoretic analysis of enzymes for the identification of variability determining genetic tolerance to industrial pollution is so far only in an introductory phase. The results obtained so far are encouraging. The studies conducted on Scots pine are treated only as a model.

Mejnartowicz (1978) has studied the variability of isozymes of leucylaminopeptidase (LAP) and acid phosphatase (APH) in 19 populations of Scots pine from the whole range of the species in Eurasia. The Poland populations were chosen from regions under industrial influence and from regions beyond pollution. In all the populations APH was coded in locus B with 15 alleles, and there was considerable polymorphism in the studied populations, and LAP was coded in 2 loci, LAP-A having 5 alleles and LAP-B 6 alleles.

It is interesting that the population of Scots pine from Babki (central Poland--strong industrial pollution) has had the lowest degree of heterozygosity on locus LAP-A, $h=0.1124$ and the highest value in LAP-B, $h=0.3970$. Among individuals sensitive to SO_2 the genotype APH-B4/B6 dominated and among individuals tolerant to this gas the allele APH-B5 appeared commonly (Mejnartowicz, 1977-78). Szmidt (1978) when studying the genetic polymorphism of catalase in three populations of Scots pine from industrial regions and 5 populations from regions free of pollution, has found 7 phenotypes of the catalase. When comparing the populations on the basis of heterozygosity he has shown that populations from regions free of pollution were characterized by greater participation of homomorphic individuals of the type C-1/C-1 compared to populations under the influence of emissions.

Mejnartowicz, et al., (1978) having taken material from plus trees in a Scots pine seed orchard established a lack of correlation between the degree of heterozygosity of trees in LAP loci and the degree of their sensitivity to the action of SO_2 in a laboratory test.

These introductory results indicate that there is differentiation in the frequency of some alleles in a population depending on the influence of industrial emissions. The important thing is that an enzymatic marker be found for tolerance which would be independent of environmental influences.

INHERITANCE OF TOLERANCE TO INDUSTRIAL EMISSIONS

Not many investigators have attempted determining in generative progeny the combining ability, heritability and repeatability of tolerance to industrial emissions. In the case of Scots pine heritability and repeatability of tolerance to O_3 is to be found under a low or medium genetic control. However, some provenances have a higher heritability of tolerance to O_3 . Needle injury in these pines was positively correlated with the injuries caused by SO_2 (Gerhold et al., 1977, Demeritt, 1977). Bialobok et al., (1978) have also found a significant correlation between the degree of needle injury on mother trees and on their progenies following treatment with SO_2 . However, these correlations were not found following treatment with O_3 or a mixture of O_3 and SO_2 .

The selection of tree individuals tolerant to industrial emissions is being done by the relatively simple methods of mass selection, without understanding the genetic basis for the population structure. This selection is usually being conducted on seedlings in laboratory conditions. Sometimes positive results were obtained for the practice which with some probability are being utilized in establishing tree plantations or producing trees for ornamental purposes (Demeritt, 1977).

So far it was not possible to solve the problem of breeding various pine species for cultivation in forests under the influence of industrial emissions, and there is no cheap way of propagation of selected individuals.

Gerhold, (1977) has discussed several methods of breeding trees more tolerant to industrial emissions, in which the use of seeds orchards established from tolerant clones is envisaged.

It appears, however, that it would be valuable to expand the investigations of genetic variability of enzymes in populations of coniferous trees endangered by industrial pollution. We have considerable information already on the course of metabolic cycles in the tree organism exposed to the action of some gases. I believe these two sources of information could be combined, (biochemical genetics and metabolic changes) in order to identify the enzymatic genetic marker that would be useful in the selection of tolerant populations.

LITERATURE CITED

Bialobok, Stefan.

1979. Identification of resistant or tolerant strains and artificial selection or production of such strains in order to project vegetation from air pollution. Symp. on the effects of airborne pollution on vegetation, Warsaw (Poland) 20-24 Aug. 1-16, 1979.

Demeritt, Maurice E. Jr.

1977. Genetic Evaluation of Two-Year Height and Ozone Tolerance in Scotch Pine (Pinus sylvestris L.). A Thesis in Genetics. The Pennsylvania State University.

Davis, Donald D. and Raymond G. Wilhour.

1976. Susceptibility of woody plants to sulfur dioxide and photochemical oxidants. EPA-600/3-76-102. p. 72. U. S. Environmental Protection Agency, Corvallis, Oregon 97330.

Gerhold, Henry G., and Raymond G. Wilhour.

1977. Effect of air pollution on Pinus strobus L. and genetic resistance. A Literature Review. EPA-600/3-77-002 p. 45 Environmental Protection Agency, Corvallis, Oregon, 97330.

Mejnartowicz, Leon.

1977-1978. Genetic characteristic of some Scots pine trees susceptible or somewhat resistant to the action of SO₂. Annual Report from Project PL-Fs-74 from U.S. Department of Agriculture under PL-480.

Mejnartowicz, Leon.

1978. Struktura genetyczna populacji sosny zwyczajnej z terenow przemyslowych zanieczyszczonych przez SO₂. In Proc. from: Reakcje biologiczne drzew na emisje przemyslowe. Kornik 4-5 maj 1978.

Mejnartowicz, L., S. Bialobok and P. Karolewski.

1978. Genetic characteristic of Scots pine specimens resistant and susceptible to SO₂ action. Arboretum Kornickie. 23: 233-238.

Mejnartowicz, Leon.

1979. Genetic variation in some isoenzyme loci in Scots pine (Pinus sylvestris L.) population. Arboretum Kornickie. 24: 91-104.

Niemtur, Stanislaw.

1979. Badania nad wewnatzgatunkowym zroznicowaniem odpornosci sosny zwyczajnej (Pinus sylvestris L.) na emisje przemyslowe. Doctoral Thesis, Uniwersytet Slaski, Katowice.

Stern, K. and Peter M. A. Tigerstet.

1974. Okologische Genetic. 98-p. 75 p. G. Fischer. Stuttgart.

Szmidt, Alfred.

1978. Zmiennosc katalazy w populacjach sosny zwyczajnej znajdujacych sie pod wplywem zanieczyszczen przemyslowych. In Proc. from: Reakcje biologiczne drzew na emisje przemyslowe. Kornik 4-5 maj 1978.

Interactions of Air Pollutants and Plant Disease¹

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Abstract: Each interaction between an air pollutant and disease is unique. Pollutants may mitigate disease response or intensify it; and the presence of certain diseases can modify the plants response to a pollutant. Air pollutants can act directly on a fungus or bacteria inhibiting parasitism. They may also act most strongly on the host, modifying its physiology and rendering it either more or less sensitive to a plant pathogen. Where the plant is weakened by the pollutants, it tends to be more sensitive to weak pathogens, but less sensitive to obligate parasites. Where the pollutant physically injures the host, infection may be facilitated. Pollutant interactions have been demonstrated both in the laboratory and the field at ambient pollutant concentrations. Interactions between pollutants and abiotic stress are particularly critical. Water regime, temperature relations, mineral nutrition and other parameters of the physical environment play a major role in the expression of air pollution injury.

Air pollution can kill plants. Even when the concentrations of an air pollutant are not directly lethal, they may be harmful, adversely affecting growth, reproduction and myriads of other biological processes (Miller and McBride 1975). One such process is the interaction that results between the pollutant and plant pathogens.

There are many ways in which such interactions might take place. But do they? The most obvious would be if the pollutant directly affected the fungus, virus, bacteria,

insect or other pathogen. If some stage of an organism's life cycle, or some stage of parasitism, were adversely impaired by a pollutant, the pollutant essentially would be acting as a fungicide, bactericide or other biocide and could be expected to bring about some degree of control.

The second way an air pollutant might act would be in adversely affecting the growth or reproduction of the host plant and thereby influencing its suitability as a host. There are two ways in which such action could be brought about. Most apparently, the pollutant might cause some physical injury to the plant leaving lesions that could serve as infection courts for a fungus or bacteria. Or, the plant might simply be weakened, thus lowering its resistance to a prospective pathogen. Presumably the weaker pathogens, the facultative parasites, would benefit most by such

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conditions. Obligate parasites, on the other hand, preferring a vigorous host, might respond quite differently and their development might even be discouraged in the weakened host.

Conversely, the impact of various diseases or plant stresses on the plant's response to air pollutants has also been considered. It should be quite apparent that adverse climatic or edaphic factors might affect a plant's sensitivity to a pollutant, but the presence of certain biotic diseases may also increase or decrease a plant's sensitivity.

A major question is not so much do such interactions take place, but what concentrations of an air pollutant are required to precipitate such effects, and how widely do such concentrations occur?

Many of the interactions that will be discussed have been demonstrated under laboratory conditions, often at pollutant concentrations known to be fairly widespread. Some of the interactions have been observed in the field.

The literature treating such air pollutant-pathogen interactions has been reviewed most recently in 1978 (Laurence), 1975 (Treshow), and 1973 (Heagle); it is not the intent of this paper to once again provide an exhaustive review of this literature. Rather, I should like to first bring the literature up to date, briefly including that delving into the interactions of abiotic stresses on pollutant effects, and discuss the implication and possible practical relevance of such interactions, especially concerning forest ecosystems. Primarily I should like to develop some of the principles in pollutant-pathogen interactions.

INFLUENCE OF POLLUTANTS ON DISEASE EXPRESSION

Sulfur Dioxide

There is no question that sulfur dioxide interacts with fungi, suppressing their growth and acting fungicidally. The only question concerns the SO₂ concentrations at which such interactions occur.

A few examples exist where SO₂ has acted in such a way in the field, limiting the development of certain plant diseases. The examples provided by Scheffer and Hedgcock (1955) in which parasitic fungi generally appeared to be retarded or inhibited near smelters where SO₂ was present is most often cited (Treshow 1975). Suppressed disease development was especially apparent where rust fungi, Melampsora and Puccinia, or needle cast fungi were involved. Linzon (1973) also discusses the reduced heart rot and blister rust (Cronartium ribicola) near a smelter.

But such incidents are becoming more infrequent as ground level SO₂ concentrations become reduced by modern control technology or use of tall stacks. Higher SO₂ concentrations, often exceeding 1 ppm for extended periods, now tend to be of only historic record. We are now interested in the possible impact of SO₂ concentrations in the 0.1 to 0.5 ppm range or even lower in cases of prolonged duration. Where such concentrations still exist near smelters, power plants, or large urban centers, some pollutant-pathogen interaction might be sought.

There is some evidence from laboratory and field research that certain pathogenic fungi respond to SO₂ in this concentration range. Weinstein and others (1975) for instance, have demonstrated that SO₂ concentrations of 0.14 to 0.15 ppm, continuous for 7 days, while not visibly injuring plant foliage, affected the bean rust fungus, Uromyces. The incidence and severity of the rust as measured by the abundance of uredia was decreased as were both the size and percent germination of the uredospores. Hence the reproductive potential of the pathogen was reduced. The results suggested that the effect on the pathogen was indirect; perhaps involving an SO₂-induced chemical change in the resistance of the host and its suitability as a habitat for the fungus. The authors speculated that production of phytoalexins or supply of metabolites might have been altered, or compounds derived from SO₂ might have accumulated.

One of the most comprehensive field studies of the interactions of industrial air pollutants, especially SO₂, with plant disease was conducted by A. Grzywacz and J. Wazny (1973). The study was notable since it dealt with economically significant diseases of a coniferous forest, specifically Scots pine (Pinus sylvestris L.). Interactions with six fungi were considered: Armillaria mellea, Fomes annosus, Lophodermium pinastri, Microsphaera alphitoides, Melampsora pinitorqua, and Cronartium flaccidum.

In an overall study, comparisons were made of disease presence in industrial areas and forests throughout Poland. In a more detailed approach, disease presence was compared at increasing distances from an industrial town, Torun, with one major SO₂ source.

The general comparisons showed that Armillaria was present in 3.7 percent of the trees in the area damaged by SO₂ compared with only 1.38 percent in all forests. Lophodermium also tended to be more serious in the industrial areas, but the results were indefinite. This was consistent with earlier findings of Donabauer (1966) that high SO₂ concentrations increased the intensity of Armillaria root rot.

Fomes showed the reverse trend and occurred

in 0.87 percent of the areas affected by industrial pollution compared with 1.5 percent in the country as a whole. Fomes, as in the overall study, was least prevalent in areas of highest SO₂ concentrations. Grzywacz and Wazny (1973) found that stumps close to the SO₂ source were sound. Related, decomposition fungi were also less abundant closest to the source. Only 50 percent of the stumps showed signs of decomposing, and 10 percent strongly decomposing, compared with 30 percent strongly decomposing farther out. However, Grzywacz (1978) later reported that stands influenced by air pollution were more subject to decomposition by brown, white or soft rot fungi. Sufficiently high SO₂ concentrations might suppress the fungal growth directly. Such was the case where Mejstrik (1978) showed SO₂ concentrations as low as 84 µg/m³ (.03 ppm) inhibited growth of Flamulina velopes, Nematoloma fasciculare, and Pleurotus ostreatus from 34 to 39 percent.

Lophodermium injury was least significant near the SO₂ source. At distances beyond 1 km injury appeared on up to 70 percent of the needles. There was a slight decrease beyond this. When annual SO₂ concentrations averaged below 0.2 ppm, disease incidence was essentially the same as in the control group. Fungus development was noticeably affected. Within 1 km, reproduction was largely by pycnidia; further distant, apothecia were common. The numbers of needles with both the sexual and asexual stages increased with distances. Closest to the SO₂ source, apothecia were only 15 percent the length of controls deformed, collapsed, underdeveloped, closed or even dried up and with ill-developed asci. Chiba and Tanaka (1968) however reported high SO₂ concentrations of 2 ppm for 14 hours caused increased infection of injured pine needles (P. densiflora) by Rhizosphaera kalkhoffii.

Weidensaul and Darling (1979) inoculated Scots pine seedlings with the fungus Scirrhia acicola. The plants were inoculated either 5 days before or 30 minutes after fumigation for 6 hours with 533 µg/m³ (0.20 ppm) SO₂. After 8 weeks, the seedlings inoculated 5 days before fumigations had more lesions incited by the fungus than those inoculated 30 minutes after fumigations in the control group. It was postulated that since the needle blight fungus penetrates through the stoma, the ability of the SO₂ to keep the stoma open facilitated entry of the fungus.

The pollutant-disease interaction may vary with the host species. Ham (1971) found that when loblolly pine served as the host, the brown spot fungus (S. acicola) was not measurably affected by SO₂ in the 0.5-0.9 ppm range for 2-3 hours even when SO₂ caused visible injury.

Simulated rain, acidified with sulfuric acid has also been shown to influence pollutant-

pathogen interactions (Shriver 1974). Although not involving forest species, the work did demonstrate the way in which plant species might be affected.

Kidney beans (Phaseolus vulgaris var. red kidney) and willow oak (Quercus phellus) were exposed to simulated rain acidified to a pH of 6.0 or 3.2 for 10 minutes each day for 30 to 60 days. The acid "rain" caused an 86 percent inhibition in the number of telia produced by Cronartium fusiforme on willow oak, a 66 percent inhibition in the reproduction of the root knot nematode (Meloidogyne hapla) on beans in the field, and a 29 percent decrease in the percent of leaf area of field grown beans affected by Uromyces phaseoli. Exposures to pH 3.2 "rain" completely destroyed the integrity of the cuticular waxes in both oak and bean. Exposures to the pH 6.0 solution did not visibly alter the cutin. Shriver postulated that damage to the cutin facilitated leaching of carbohydrates from the leaf including those that might inhibit hyphal growth of Botrytis cinerea thus predisposing plants to infection.

This is consistent with the findings of F. A. Last and O. D. Fowler at the Institute of Terrestrial Ecology at Penicuik, Scotland (personal communication) who showed that the cuticle of Scots pine was gradually broken down by low concentrations of SO₂.

Sulfur dioxide, perhaps largely as acid rain, may have an effect on soil microorganisms (Wainwright 1978). Sulfur-oxidizing microorganisms, including Alternaria tenuis, Auriobasidium pullulans and Cephalosporium sp., were isolated from leaves, litter and soils polluted with high levels of SO₂. The predominant autotroph on leaf surfaces was Thiobacillus thioparis.

Hibben and Taylor (1975) found that SO₂-pathogen interaction depended on the infection stage. Conidial germination and the appressorial stages of Microsphaera alni were most sensitive to SO₂. Since infection was not reduced when leaves were previously fumigated, the effect was thought to be mostly fungicidal.

Ozone

There is considerable evidence that photochemical pollutants, most notably ozone, affect plant pathogens and fungus-host interactions. The interaction may be positive or negative depending on the O₃ concentrations and the O₃ sensitivity of the host.

In one such study, Miller and Elderman (1977) showed that fumigation of ponderosa and Jeffrey pine seedlings with ozone enhanced infection by Fomes annosus. Average infection of the two

species increased from 57 percent to 78 percent when the seedlings were exposed to $431.2 \mu\text{g}/\text{M}^3$ (.25 ppm) ozone. They also found that ozone injury increased the susceptibility of pine stumps to colonization by *Fomes annosus*. In the laboratory, wood from trees slightly damaged by oxidants was more decay-susceptible than wood from severely damaged trees.

Far more research has involved powdery mildew (Heagle and Strickland 1972, Heagle 1975, Scheutte 1971), and the negative impact of ozone on disease. Even ozone concentrations in the 5 to 50 pphm range reduced infection when conidia were exposed. According to Scheutte, appresoria formation was largely affected. Also, some epidermal cells, hypersensitive to ozone, were killed thereby reducing infection. Eight hour exposure to 50 pphm ozone reduced formation of secondary hyphae 66 percent and 4 hour exposure reduced 41 percent. Ozone impaired formation of the penetration peg thus inhibiting subsequent development of a functional host-parasite relationship.

While ozone impacted certain critical stages of infection under controlled conditions, in the field such effects would be less apparent since all stages of parasitism would be present simultaneously.

Interactions of facultative parasitic and saprophytic fungi with their hosts may also occur and are most pronounced when the plants are weakened or injured by ozone (Manning 1975). Obligate parasitism appears to be retarded by ozone and ozone-injured host tissue. Colonization of ozone-injured white pine needles by *Lophodermium pinastri* was reduced while colonization by the saprophyte *Aureobasidium pullulans* was increased. Ozone-injured lilac leaves were rarely infected by powdery mildew *Micosphaera alni*.

When the fungus is especially sensitive to ozone the impact may be mostly fungicidal thereby reducing infection. Sporulation and germination of the weak parasite *Botrytis cinerea* was significantly inhibited by 2, 6 hour ozone exposures at a concentration of 30 pphm (Krause and Weidensaul 1978). There was also less infection, based on total lesion area, when plants were exposed to 15 pphm. Ozonation of conidia produced *in vivo* and *in vitro* decreased germination of conidia, germ tube length, and pathogenicity. The authors postulated that the mode of action was to alter the permeability of the conidia membrane. Ozone may increase the conidiophore respiration causing prematurely-formed and non-viable conidia. Ozone also may inhibit, directly or indirectly, enzyme activity of the fungus and cause less maceration of the host cells resulting in a decreased infection.

Curtis and others (1976) found that plants exposed to ozone produced peroxidases and iso-

flavonoids that are toxic to microorganisms. According to Laurence and Wood (1978), ozone inhibited infection of soybean by *Pseudomonas glycinea* presumably because of a bacteriostatic compound, possibly an isoflavonoid compound, produced in response to ozone.

Fluoride

Fluoride is accumulated in the foliage of plants and affects many metabolic processes. Consequently, it is reasonable to presume that pollutant-pathogen interactions can be influenced, perhaps to a greater extent than with any other pollutant. This was confirmed with viruses by some of the early work reviewed by Heagle (1973) and Treshow (1975).

Host-fluoride interactions also have been demonstrated for fungi (McCune and others 1973) although not on forest species. Studying the interactions of 7 to $10 \mu\text{gF}/\text{M}^3$ with 3 diseases of bean and 2 diseases of tomato, HF consistently reduced uredial formation as the foliar fluoride content increased. Tender green beans exposed to HF and inoculated with powdery mildew conidia developed fewer foliar lesions than non-fumigated control plants; 4.4 compared with 48.7 on fumigated leaves having an average of 399 ppm F. Powdery mildew was reduced proportionate to the length of the exposure period. Early blight lesions on tomato were similarly reduced by the presence of fluoride.

INFLUENCE OF DISEASE ON POLLUTANT SEVERITY

Not only may the presence of a pollutant influence pathogenicity, the reverse may be true--presence of diseases influence the degree of pollutant injury. This has best been demonstrated with viruses. None of these involved forest species though, and there has been no new results published since last reviewed by Laurence (1978).

Also, in the forest, variation in pollutant resistance among individual trees would far exceed any disease and pollutant interaction, and any impact of a disease on pollutant severity probably wouldn't be measurable.

ABIOTIC STRESS

Severity of air pollution injury is determined not only by the dose of the pollutant to which plants are exposed but on the predisposition of the plant. Predisposition is determined by any environmental parameter to which the plant is subjected. It has been discussed more in passing than as a major concern (Heck and others 1965, Heck 1968, Treshow 1970, Taylor 1974), but the interactions of abiotic stress or disorders with any pollutant is far more

significant than with biotic pathogens. Moisture stress, the light regime, edaphic factors and temperature relations all influence the sensitivity of plants to every pollutant.

Unusually high or low temperatures prior to exposure tend to reduce plant sensitivity to ozone, for instance. A normal high light intensity renders plants more sensitive. High humidity, by encouraging open stomates, also increases sensitivity. In dry air, plants are more tolerant of ozone than at higher humidities, and PAN injury is most severe in the Los Angeles area when the relative humidity exceeds 50 percent.

Any factor favoring moisture stress and reduced water uptake would cause stomatal closure and favor resistance to pollutant uptake and injury. In this way, plants in more saline soils are more tolerant of certain pollutants (Oertli 1958) and there is little to no ozone or PAN injury when soil moisture is deficient.

Ozone injury may be negligible when ambient air temperatures during and after exposure are below 90° F., yet may be severe when temperatures exceed 90° F. (Taylor 1974).

Effects of low atmospheric and soil moisture on tolerance or sensitivity of plants to SO₂ injury are well known; suffice it to note that plants are far more sensitive under higher moisture conditions.

Mineral nutrition also plays a role in pollutant sensitivity. Vegetable crops are known to be most susceptible to ozone when nitrogen nutrition is adequate although some reports suggest injury is greatest when nitrogen is low. Sulfur dioxide sensitivity is reported to be greatest with increased sulfur nutrition. Stomatal behavior may be involved since a deficiency of nitrogen, potassium or phosphorus may decrease stomatal opening. Plants low in sulfur had the lowest stomatal capacity, thus less capacity for gas absorption (Leone and Brennan 1972).

Toxicity from heavy metals also plays an interesting role with air pollutants. Ormrod (1977) has found that ozone toxicity was enhanced on pea plants growing in elevated concentrations of cadmium or nickel. However, when cadmium concentrations were sufficiently high to adversely affect growth, 100 µmol of CdSO₄, then ozone injury was less than that of control plants.

Harkov and others (1979) found that the amount of ozone injury depended on the cadmium and ozone levels as well as other environmental conditions. Where environmental conditions were not conducive to ozone injury (e.g. bright sunlight), cadmium had little effect on the amount of leaf damage. When environmental

conditions allowed only slight to moderate foliar injury on tomato leaves, damage was enhanced by cadmium treatment especially at higher concentrations.

Along the same line, low concentrations of SO₂ affected metal uptake, increasing zinc and cadmium susceptibility of bean plants. Foliar injury caused by heavy metals was significantly enhanced by SO₂ (Krause and Kaiser 1977).

Lamoreaux and Chaney (1978) also found significant interactions between cadmium and SO₂. While cadmium or SO₂ alone reduced net photosynthesis and transpiration of excised silver maple leaves, the reduction in cadmium-treated leaves was greatest in the presence of SO₂.

CONCLUSIONS

Virtually every environmental parameter, biotic and abiotic can interact with air pollutants to aggravate or mitigate the extent of pollution damage. The significant question then becomes, to what extent and at what concentrations? Is the net impact real, and if so, how meaningful is it under field conditions?

All pollutants don't interact the same. Sulfur dioxide, for instance, appears primarily to act fungicidally reducing disease and decomposition activity. It is also postulated though that SO₂ induces chemical changes imparting resistance to certain pathogens. Ozone tends to act more on the host, weakening it and accentuating disease activity. Research with vegetable crops at low ozone concentrations indicate that pollutant-disease interactions do occur with some diseases and very likely occur in forest stands weakened by photochemical pollutants. Ozone has increased susceptibility to *Fomes* and rendered trees more susceptible to decay. Each interaction is unique since every fungus has its own sensitivity as does each plant. Thus the potential for interaction is enormous.

Pollutants such as fluoride that accumulate significantly in plants would likely have the greatest interaction. While the trend is to inhibit disease development, the concentrations at which this has been demonstrated are now infrequent in the field.

The significance of abiotic stress and pollutant interactions is often taken for granted, but it should be emphasized that this is of paramount importance. Climatic and edaphic factors are often determinants in the expression of pollutant damage. While the abiotic environment strongly influences the severity of air pollution damage, air pollutants influence the severity of plant diseases caused by biotic pathogens. It would appear that as in all life systems, every environmental parameter has some influence on every other. In the case of

pollutants and pathogens, this interaction can be significant.

LITERATURE CITED

- Chiba, O. and K. Tanaka.
1968. The effect of sulfur dioxide on the development of pine needle blight caused by Rhizosphaera kalkhoffii Bubak (I). J. Jap. For. Soc. 50:135-139.
- Curtis, C. R., R. K. Howell and D. F. Kremer.
1976. Soybean peroxidase from ozone injury. Environ. Pollut. 11:189-194.
- Donabauer, E.
1966. Secondary damage to forest caused by industrial exhaust fumes. Mitt. Forst. Bundesvers. 73:101.
- Grzywacz, A.
1978. Interactions between air pollutants and decaying fungi of pine wood. In. 3rd Internat. Congress of Plant Pathology. Munchen. 1978.
- Grzywacz, A. and J. Wazny.
1973. The impact of industrial air pollutants on the occurrence of several important pathogenic fungi of forest trees in Poland. Eur. J. For. Path. 3:129-141.
- Ham, Donald L.
1971. The biological interactions of sulfur dioxide and Scirrhia acicola in loblolly pine. Ph.D. Thesis, Duke Univ., No. Carol. 75 p.
- Harkov, R., B. Clarke and Eileen Brennan.
1979. Cadmium contamination may modify response of tomato to atmospheric ozone. J. Air Poll. Control Assoc. 29:1247-1249.
- Heagle, A. S.
1973. Interaction between air pollutants and plant parasites. Ann. Rev. Phytopathol. 11:365-388.
- Heagle, A. S.
1975. Response of three obligate parasites to ozone. Environ. Pollut. 9:91-95.
- Heagle, A. S. and Alice Strickland.
1972. Reaction of Erysiphe graminis f. sp. hordei to low levels of ozone. Phytopathol. 62:1144-1148.
- Heck, W. W.
1968. Factors influencing expression of oxidant damage to plants. Ann. Rev. Phytopathol. 6:165-188.
- Heck, W. W., J. A. Dunning and I. J. Hindawai.
1965. Interactions of environmental factors on the sensitivity of plants to air pollution. J. Air Poll. Control Assoc. 15:511-515.
- Hibben, C. R. and M. P. Taylor.
1975. Ozone and sulfur dioxide effects on the lilac powdery mildew fungus. Environ. Pollut. 9:105-114.
- Krause, G. H. M. and H. Kaiser.
1977. Plant response to heavy metals and sulfur dioxide. Environ. Pollut. 12:63-71.
- Krause, C. R. and T. C. Weidensaul.
1978. Effects of ozone on the sporulation, germination, and pathogenicity of Botrytis cinerea. Phytopathol. 68:195-198.
- Lamoreaux, R. J. and W. R. Chaney.
1978. Photosynthesis and transpiration of excised silver maple leaves exposed to cadmium and sulphur dioxide. Environ. pollut. 17:259-268.
- Laurence, J. A.
1978. Effects of air pollutants on plant-pathogen interactions. 71st Ann. Mtg. Air Poll. Control Assoc. Houston, Texas. 1978.
- Laurence, J. A. and F. A. Wood.
1978. Effects of ozone on infection of soy bean by Pseudomonas glycinea. Phytopathol. 68:441-445.
- Leone, I. A. and Eileen Brennan.
1972. Sulfur nutrition as it contributes to the susceptibility of tobacco and tomato to SO₂ injury. Atmos. Environ. 6:259-266.
- Linzon, S. N.
1973. The effects of air pollution on forests. Pap. 4th It. Chem. Cong. Conf. 1973 pp. 1-18.
- Manning, W. J.
1975. Interactions between air pollutants and fungal, bacterial and viral plant pathogens. Environ. Pollut. 9:87-90.
- McCune, D. C., L. H. Weinstein, J. F. Mancini, and P. van Leuken.
1973. Effects of hydrogen fluoride on plant-pathogen interactions. Internat. Clean Air Congress, Dusseldorf.
- Mejstrik, V.
1978. Influence of sulfur dioxide on the growth of some fungus species. Zbl. Bakt. II Abt. 133:464-467.
- Miller, Paul R., and Joe R. McBride
1975. Effects of air pollutants on forests. In Responses of plants to air pollution. J. Brian Mudd and T. T. Kozlowski, eds. p. 192-235. Academic Press, New York.

- Miller, Paul, R. and M. J. Elderman.
1977. Photochemical oxidant air pollution effects on a mixed conifer ecosystem. Ecological Research Series EPA 600/3-77-104.
- Oertli, J. J.
1958. Effect of salinity on susceptibility of sunflower plants to smog. Soil Sci. 87:249-251.
- Ormrod, D. P.
1977. Cadmium and nickel effects on growth and ozone sensitivity of pea. Water, Air and Soil Pollut. 8:263-270.
- Saunders, P. J. W.
1971. In Ecology of Leaf Surface Microorganisms, pp. 81-89, T. F. Preece and C. H. Dickinson. Academic Press, London.
- Scheffer, T. C., and G. G. Hedgcock.
1955. Injury to northwestern forest trees by sulfur dioxide from smelters. U.S. Forest Serv. Tech. Bull. 1117, 1-49.
- Scheutte, L. R.
1971. Response of the primary infection process of Erysiphe graminis f. sp. hordei to ozone. Ph.D. Thesis, Univ. Utah, Salt Lake City, 71 p.
- Shriver, D. S.
1974. Effects of simulated rain acidified with sulfuric acid on host-parasite interactions. Ph.D. Thesis, N. Carolina State, Raleigh, N.C.
- Taylor, O. C.
1974. Air pollution effects influenced by plant-environmental interaction. Amer. Chem. Soc. Symp. Ser. 3:1-7.
- Treshow, M.
1970. Environment and plant response. 422 p. McGraw-Hill Co., New York.
- Treshow, M.
1975. Interactions of Air Pollutants and Plant Disease. In Responses of plants to air pollution. J. Brian Mudd and T. T. Kozlowski, eds. p. 307-334. Academic Press, New York.
- Wainwright, M.
1978. Sulphur-oxidizing microorganisms on vegetation and in soils exposed to atmospheric pollution. Environ. Pollut. 17:167-174.
- Weidensaul, T. C. and S. L. Darling.
1979. Effects of ozone and sulfur dioxide on the host-pathogen relationship of Scotch pine and Scirrhia acicola. Phytopathol. 69:939-941.
- Weinstein, L. H., D. C. McCune, A. L. Aluisio and P. van Leuken.
1975. The effect of sulphur dioxide on the incidence and severity of bean rust and early blight of tomato. Environ. Pollut. 9:145-155.

Studies on Relationship Between Air Pollutants and Microorganisms in Japan¹

Kiyoshi Tanaka²

Abstract: A review of the literature on the interaction of air pollutants with parasitic and epiphytic microorganisms is presented with a brief outline of air pollution problems in Japan. Some fungi such as Cercospora sequoiae, Pucciniastrum styracinum, Puccinia kusanoi, Melampsora coleosporioides, Trichoroma matsutake, and some epiphytic bacteria disappeared in the areas affected by air pollutants, and Rhizosphaera kalkhoffii increased its activity due to SO₂ in the air.

Fungi, bacteria, viruses and nematodes, as well as insects are all responsible for bringing a certain species of higher plants into equilibrium with its environment (Treshow 1968). The effect of air pollutants on population trends of various microorganisms, therefore, is extremely important in all ecological systems. Further, air pollutants may affect plant disease development in different ways. Pathogenicity may be influenced through a direct effect of the pollutants on the parasite, or the effects may be indirect through pollution-induced changes in physiology of the host plant (Heagle 1973).

Prior to 1960's, the interaction of microorganisms with pollutants in the air had received limited attention in Japan. But during the past decades, a series of experiments has been conducted in order to explain the effect of air pollution on disease development, and only recently, some attempts have been made to understand the change of populations of microorganisms for use as biological indicators of air pollution.

From the beginning of recognition of the significant and sometimes devastating effects of air pollutants on vegetation, plant pathologists have been

confused with the similarities between the symptoms of many types of pollution damage and those of infectious diseases sometimes leading to incorrect diagnoses. Fortunately, each of the common air pollutants produces a characteristic injury pattern, and the sensitivity of many species differs sufficiently for the trained eye to determine the causative agent (Adams and Salzbach 1961).

This paper will review the literature on: 1) the modification of parasitism and the population changes of epiphytic microorganisms induced by air pollutants; 2) attempts to make reliable guides in distinguishing among symptoms caused by air pollutants and parasites, and those occasionally produced by saprophytic fungi. These subjects are considered in conjunction with a brief outline of air pollution effects on forest ecosystems in Japan.

SULFUR DIOXIDE (SO₂)

In Japan, towards the end of 19th century, air pollution was reported to be harmful to forests around ore smelters that emitted large amounts of SO₂. Plume behavior resulted in pollutants dispersion in the surrounding area, killing all the vegetation (mainly Cryptomeria japonica D. Don and Pinus densiflora Sieb. et Zucc.) and causing erosion of topsoils following the death of the plants (Doi 1919). The concentration of SO₂ of the smoke was reported to be 1 ppm at 1 mile from a large ore smelter located in Ashio, Tochigi Prefecture, and occasionally reached as high as 8 to 10 ppm at the ground level (Doi 1919).

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Raising the height of stacks in order to reduce the concentration of SO_2 at the ground level was the first and primitive control measure, new technology involves the installation of sulfur recovery devices. In 1915, one ore smelter in Hitachi, Ibaraki Prefecture, built a new stack on a hill rising about 350 m above sea level; the stack was nearly 170 m high, so that the smoke was discharged at an elevation of approximately 520 m. Although smelter smoke is released from high stacks, it tends to diffuse rather slowly, and the SO_2 , therefore, reaches the ground in harmful concentrations even after a considerable length of time and distance. Observations made three years after the replacement of stacks revealed the fact that the severity of damage in the area near to the smelter was markedly reduced, while the total damaged area increased as large as 24,000 ha (Doi 1919).

The reduction of the concentration of SO_2 in the area in the vicinity of the ore smelter on account of the change of stack height was good enough to resume the seedling production of *Cryptomeria japonica* in nurseries. Kaburagi (1930) was impressed with the absence of a needle blight of *Cryptomeria japonica* caused by *Cercospora sequoiae* Ell. et Ev. in the nurseries located in the recovered area close to the higher stack. The needle blight is one of the most destructive diseases in forest nurseries producing *Cryptomeria* seedlings, and has spread rapidly throughout Japan except Hokkaido. The causal fungus was introduced from North America in the end of 19th century, and Ibaraki Prefecture including the ore smelter is considered to be the epicenter of the introduced disease (Ito 1976). Kaburagi (1930) suggested that a strong correlation existed between SO_2 in the air and the absence of the disease. This is believed to be the first report on the effect of air pollution on diseases caused by parasites in Japan. Kaburagi's observation is strongly supported by the following hypotheses that: 1) SO_2 might act directly upon the fungus on the surfaces of host plants; and 2) the reduction of disease severity resulted in a decrease of inoculum for new infection.

Some of the acute type of damage by the emissions from smelters still continues in areas near to sources. In the 1960's, widespread SO_2 air pollution damage to ornamental trees had become apparent to the public; this increase was due to urbanization and industrialization. The one most apparent episode in and around Tokyo was the stunting and death of a large number of *Zelkova serrata* Mak., widely used as shade trees throughout Japan (Yambe 1973).

With a considerable increase in public awareness of the chronic type of pollution in many parts, automatic SO_2 recorders were set up by the national and the prefectural agencies to determine the concentrations, frequency, and the duration of atmospheric SO_2 exposures. The number of measuring locations had increased from the beginning at the 15 strategic locations in 1965 to as many as 1473 locations in 1979, forming an extensive sampling system for SO_2 (The Japanese Environmental Agency

1979).

The analysis of air provides a framework of knowledge, and a foundation for developing air quality management strategies, and for prompting the enactment of the Clean Air Act in 1967, and its amendments made in 1970.

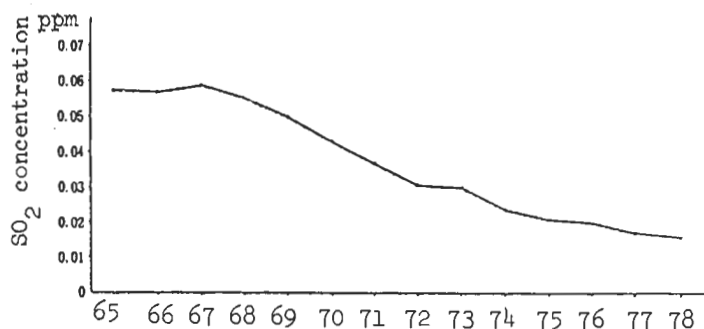


Figure 1. Mean value of annual concentrations at 15 strategic locations.

Figure 1 shows annual changes in average concentrations of SO_2 at 15 strategic locations between 1965 and 1977, the concentration increased until 1967, and then declined gradually. The decline has proved the continuing effectiveness of regulation of air pollution sources by the national and the prefectural governments based on the act.

During the summers in the middle of 1960's, the worst average concentrations of SO_2 were experienced. In this period, an unrecorded needle blight of Japanese red pine, *Pinus densiflora*, occurred throughout the normal range of that species, predominantly in central Japan. Up to that time the disease had been little known and of little concern to forest pathologists. A specific fungus dominantly found on blighted needles was investigated by Kobayashi (1967), and he identified it as *Rhizosphaera kalkhoffii* Bubák which had been known as a causal fungus of a needle blight of spruce and fir in northern Europe and North America.

The results of inoculation experiments to Japanese red pine seedlings proved that the pathogenicity of the causal fungus was considerably weak for pines, and the disease would scarcely occur if pines grew under normal conditions (Tanaka and Chiba 1971). The unusual incidence of the disease occurring in the early summer of 1965 might be related to abnormal weather conditions--extremely little rain fell in early spring followed by heavy rain in May (Chiba and Tanaka 1968). Although the disease has not occurred so widely as in 1965, severe damage has been observed in the vicinity of ore smelters or industrial areas. In addition, Japanese red pine is highly sensitive to SO_2 (Inoue 1973), and the distinction between the needle blight symptoms caused by the disease and those caused by SO_2 is exceedingly difficult (Tanaka 1975, Tanita 1976). Therefore exhaustive experiments were conducted to examine the relationship of SO_2 with disease development.

A total of 163 two-year-old seedlings of Japanese red pine were exposed to 2.0 ppm SO_2 for 1, 2, 3, and 4 hours in a fumigation chamber before or after inoculation in the middle of July, with a water suspension of conidia of the fungus produced on PDA plates.

Damage was more severe in the treatment combining the fumigation with inoculation, especially inoculation before fumigation, compared with those in the treatment of fumigation without inoculation. Numerous pycnidia of the fungus were produced in large lesions on needles treated by fumigation after inoculation as well as fumigation for longer duration before inoculation, no obvious change in the appearance of needles was detected on the seedlings inoculated with the fungus without fumigation. Symptoms produced on the SO_2 treated needles, namely, large lesions with numerous pycnidia, were identical to symptoms of the needle blight occurring in the field (Chiba and Tanaka 1968).

Most pine seedlings displayed acute symptoms of injury following exposure to 2.0 ppm SO_2 for 2 hours. The SO_2 concentration used in this study was several times higher than that found in the ambient air. It was necessary to elucidate the relationship between SO_2 and disease development at lower concentrations because such conditions are frequently encountered in rural and forest regions. Therefore, three-year-old seedlings of Japanese red pine were exposed for 2 weeks in chambers to 0.2 ppm SO_2 , the level of SO_2 is likely to be found in the ambient air (Tanaka 1976a).

In this experiment, special emphasis was placed on the effect of SO_2 fumigation after inoculation in comparison with the treatment before inoculation. The treated needles with SO_2 fumigation after inoculation were more severely affected by the fungus, compared with those fumigated by SO_2 gas before inoculation. This result is consistent with findings of the 2.0 ppm SO_2 fumigation (Chiba and Tanaka 1968). SO_2 injury appeared to facilitate the spread and reproduction of the fungus within tissues of needles rather than to increase the initial rate of infection by the fungus.

To further examine these findings from artificial SO_2 exposure of seedlings in chambers, field experiments were planned and conducted during the summers of 1972 and 1973 by using a transplanting method (Tanaka et al. 1974a) and a filtered air method (Tanaka et al. 1974b) in Hiroshima Prefecture. In the transplanting study, two experimental plots were established. One was in Fukuyama city close to a large steel refinery. Especially in this area, distinct SO_2 damage markings on the needles of Japanese red pine were visible. The other was in a rural area in Miyoshi city. It was located about 60 km north of a highly industrialized area along the Seto Inland Sea including Fukuyama city. In the rural area, the disease caused by *Rhizosphaera kalkhoffii* had not been noted in recent years. A total of 900 potted seedlings of Japanese red pine used in the transplanting study were cultivated previously in the nurseries in the

rural area in Miyoshi city. Seedlings were divided into 12 groups as shown in figure 2 according to the combination of transplanting and inoculation.

SO_2 concentrations of both areas were determined by means of the lead peroxide method. Needles were analyzed for sulfate-S and for total-S dry basis in order to determine the accumulation of sulfur in the needles. Foliar symptoms were rated from 0 to 5, according to the degree of injury, and the average of these individual indexes provides the disease severity for each group.

After being transported from the rural area to the industrialized area, several seedlings exhibited typical symptoms and signs of the disease, while others remaining in the rural area displayed no evidence of injury, even though they received the conidial spraying of the causal fungus (figure 2).

In the experiment using a filtered air method, an experimental plot was established in an area in Mihara city close to a powdered-coal-burning power plant. In this area, a large number of needle blighted Japanese red pines were found, and they displayed gradual reduction in shoot and wood growth. The previous year's annual mean contents of sulfur compounds in the air was 1.04 mg $\text{SO}_3/100 \text{ cm}^2 \text{ PbO}_2$ per day.

Plants were grown in plastic covered chambers. One chamber (2x4x2 m) was equipped with carbon filters and blowers, and the control or ambient air chamber was provided with blowers. The air passed through each chamber at a rate of about one air change / min. SO_2 concentrations of the ambient air were determined by means of an automatic recorder, and SO_2 concentrations in chambers were measured by the lead peroxide method.

The experimental design was similar to the transplanting experiment in Fukuyama and Miyoshi, that is, a total of 300 seedlings were divided into 12 groups according to the combination of exposure to SO_2 and inoculation of the causal fungus, then seedlings were placed in chambers to determine whether some protection from infection could be obtained when the air drawn into the chamber was filtered. There was a large reduction in the severity of the disease in the filtered air compared with that of the disease in the ambient air. These findings by field studies furnished the evidence that SO_2 in the air was responsible for the development of the disease. The sulfur content especially sulfate-S in the needles of the trees exposed to SO_2 was significantly greater than that of untreated needles both in the transplanting and the filtered-air experiments.

An increasing in disease severity may be expected when air pollutants weaken trees, making them more susceptible to infection by weak parasites, or reduced vigor to a degree that plants become predisposed to attack by facultative parasites, such as *Rhizosphaera kalkhoffii*, and *Botrytis cinerea* Pers. (Manning et al. 1970). Conversely, Linzon (1958)

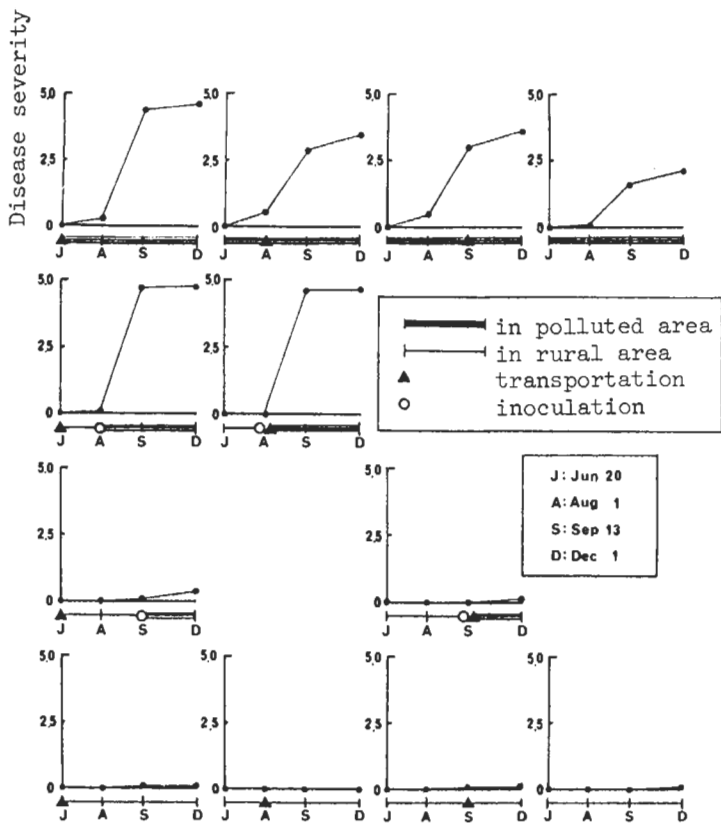


Figure 2. Effect of SO_2 on the development of needle blight of pine caused by *Rhizosphaera kalkhoffii*

found that smelter emissions decreased the incidence of *Cronartium ribicola* C.J.Fis., and there are additional reports that aggressive or obligate parasites, such as *Microsphaera alphitoides* Griff. et Maubl. (Köch 1935), and *Melampsorella cerastii* Wint, *Peridermium coloradense* Arth. et Kern, and some species of *Melampsora* and *Gymnosporangium* (Scheffer and Hedgcock 1955) were less abundant in areas adjacent to smelter zones or to industrialized areas. Ito¹ found a similar case in Tokyo that a close relationship existed between the increase of distance from the metropolitan area and the increase of the incidence of willow rust disease caused by *Melampsora coleosporioides* Dietel and a leaf rust disease of *Styrax japonicum* Sieb. et Zucc. caused by *Pucciniastrum styracinum* Hirat. The complex of air pollutants was brought about by urbanization and industrialization, typical of the mixture of pollutants occurring in and around Tokyo, but SO_2 was considered to be principally responsible for reducing the abundance of these fungi.

OXIDANTS MAINLY OZONE (O_3)

Beginning in the 1970's, changes from solid to liquid and gas fuels, and especially increasing use of motor vehicles, have caused a reduction of the SO_2 -type of urban pollution and an increase of "ozone" pollution.

³ Personal communication from S. Ito, University of Tokyo, January 15, 1980.

On July 18, 1970, *Zelkova serrata* trees in a yard of a senior high school located in Suginami Ward in Tokyo showed unusual shedding of their leaves, and strawberry leaves showed white spots clearly indicating the effects of oxidants. In addition to the damage to plants, 45 students of the school were hospitalized for treatment of irritated eyes, sore throats and difficulty in breathing. From this evidence, the Metropolitan Government issued the statement that the damage to humans and vegetation was caused by photochemical air pollution (The Tokyo Metropolitan Government 1971), and this was thought to be the first episode of "Los Angeles type smog" in Japan. But as a result of later investigations it was found that there were differences between Tokyo and Los Angeles air pollution, since SO_2 concentration and humidity in Tokyo were much higher than in Los Angeles; furthermore, it was confirmed that there was formation of sulfuric acid in the air (The Tokyo Metropolitan Government 1971). These findings suggest that the complex of photochemical oxidants and sulfuric acid might exert a far greater influence on humans and vegetation than any other alone at that time.

In the latter half of 1970's, with continuous increasing industrialization and transportation, every major city in Japan has experienced an increasing in photochemical oxidants even though there was a slow but steady decline in the concentration of SO_2 in the ambient air (figure 1).

Photochemical oxidants, mainly ozone (O_3), continue to cause the defoliation of broad leaf trees. Especially, in the Kanto district, at the center of Tokyo, unusual defoliation of *Zelkova serrata* still continues, since it is sensitive to both SO_2 (Inoue 1973) and O_3 (Kadota and Ohta 1972). While, in the Kansai district, at the center of Osaka, there is a high incidence of defoliation of poplar trees in the early summer in the parks, the gardens or the yards of schools throughout the district.

In addition to the measuring system of SO_2 throughout the country, a number of automatic oxidant recorders were also set up by the national and the prefectural agencies, but there is still a shortage of satisfactory apparatus for monitoring the low ambient concentrations. To make up for the shortage of an instrumented system, plants have been used extensively as bioindicators of air pollution. The use of bioindicators has long been accepted, since damage to green plants is usually one of the first signs that air pollution is becoming a serious problem (Berry 1964); their usefulness in this capacity is based primarily on the sensitivity of selected plant species and varieties for specific pollutants (Heck 1966).

Many clones, species and varieties of poplars were selected as bioindicators because of their high sensitivity to oxidants and their wide distribution, especially in the Kansai district. They also have enormous advantages in that they are fast growing, easy to propagate vegetatively by

cuttings, and are uniformly sensitive to oxidant injury.

Despite these advantages, poplars are frequently plagued by various kinds of insects and diseases that degrade the usefulness of poplar clones as pollution indicators. For example, interpretation of the response to oxidants is made difficult by leaf diseases caused by Marssonina brunnea (Ell. et Ev.) Magnus, Septotinia podophyllina Whetz. and Melampsora larici-popululina Kleb. A reliable guide to distinguish among symptoms caused by air pollutants, insects, and diseases has been developed and published (Tanaka 1975). The defoliation caused by Marssonina brunnea is one of the most serious diseases. It occurs wherever poplars grow, and is not restricted to urban regions, but frequently occurs with severity in urban areas. The disease develops tiny spots with light margins about 1 mm in diameter over the entire surface of the leaf. These superficial spots are not so difficult to distinguish from the symptoms caused by ozone by plant pathologists with a trained eye, but many people who want to use poplars as indicators are confused with the similar symptoms produced by ozone and the disease (Tanaka 1975). In addition, the disease causes premature lower leaf abscission, and the defoliation progresses upward toward the tips of the shoots from the late spring through the summer months. The unsightly appearance of affected trees discourages poplars' use as indicators of oxidants.

A number of investigators suspected that abscission was not due directly to the pathogen but resulted instead from increased rates of ethylene production from the infected plants (Abeles 1973), further, ozone-induced abscission of leaves of varying maturity was closely correlated with rates of ethylene production (Abeles 1973). From these

reasons, Tanaka (1976b) hypothesized that increased rates of ethylene production caused by the synergistic effects of the infection of the fungus and ozone was responsible for the incidence of the disease in the urban area.

Fortunately, there is striking variability in susceptibility of poplar clones to the fungus and ozone. Clones were selected that are sensitive to ozone and resistant to the disease. The selection offers one of the best means of increasing the usability of poplar clones as ozone indicators (Tanaka 1977a). Table 1 presents the relationship between ozone sensitivity and disease resistance of poplar clones (Tanaka 1977a). The data of resistance of poplar clones to the disease are based on the three years' observation made in Osaka using 38 clones of young rooted cuttings (Tanaka 1977a), and the data of ozone sensitivity of them are based on the results from the exposure experiments with 0.15 ppm ozone for 40 hours in chambers conducted by Enoki (1977).

Populus maximowiczii cv. OJ 115 and cv. OJP 1, P. maximowiczii x P. trichocarpa cv. OP 41, P. maximowiczii x P. nigra plantierens cv. OP 52, and P. nigra italica x P. maximowiczii cv. Kamabuchi are promising clones for ozone indicators because they are sensitive to ozone and resistant to the disease (table 1).

Yambe (1978) also attempted to find a possible usefulness of the change of microbial flora on bamboo leaves as indicators for air pollution. He found that Puccinia kusanoi Dietel and epiphytic bacteria on the leaf surfaces of bamboo decreased sharply in number in the urbanized and industrialized areas.

Table 1. Relationship between clonal susceptibility of poplars to O_3 and to Marssonina brunnea.

		Susceptibility to <u>Marssonina brunnea</u>				
		Highly resistant (RR)	Resistant (R)	Moderately resistant (MR)	Susceptible (S)	Highly susceptible (SS)
Susceptibility to O_3	Highly resistant (RR)			NR 6 OP 29	NR 84	
	Resistant (R)			NR 2, OP 26, Serotina	NR 19, NR 32	I 45/51
	Moderately resistant (MR)		OP 285		L 230, L 293, LK 83, LW 42, NR 24B	
	Susceptible (S)	OP 52	OP 1, OP 20		Gerlica I 154, I 214	I 262
	Highly susceptible (SS)	I 72/51, Kamabuchi, OJ 115, OJP 1, OJP 2, OJP 3, OJP 4, OJP 5, OP 41				I 77/51, I 488

OTHER POLLUTANTS

In addition to SO₂ and oxidants, significant and sometimes devastating effects of other air pollutants have long been recognized in Japan. Fluoride compounds have a long history as pollutants in the vicinity of ceramic industry, and more recently, plant injury resulted from fluorides has gradually increased because of the expansion of industries such as aluminum refineries and fertilizer manufactures (Matuura and Kokubu 1972). Chlorine (Tanaka 1977b), ethylene, acidic dust from stacks, and alkaline dust from cement kilns (Inoue 1972a and 1972b) have also been recognized as agents of damage to vegetation, but the damaged areas are relatively restricted because most of them are emitted from point sources or by accidental spills.

The interaction of these minor pollutants with fungi and bacteria has received only minor attention, but Inoue (1972a and 1972b) found that the number of fruiting bodies of Tricholoma matsutake Singer was significantly reduced in the heavily dusted forests in the vicinity of cement works compared to less severely dusted or non-dusted portions of forests. According to the results of hand-dusted experiments, he concluded that the alkaline solution from the dust severely decreased the formation of the fruiting bodies, and reduced the development of the fungus colonies.

CONCLUSIONS

The interaction of pollutants with parasitic and epiphytic microorganisms has received limited attention in Japan. The existing reports are only associated with trees and bamboos and no report with annual plants was found in a search of available literature.

Some pollutants are directly toxic to microorganisms on the leaf surface. They might possibly impair their growth or reproduction, and change the population and community composition of microorganisms on the host plants, since the sensitive species to air pollutants would be gradually replaced by more tolerant species (Treshow 1968). Especially, aggressive or obligate parasites such as Cercospora sequoiae (Kaburagi 1930), Pucciniastrum sty-racinum and Melampsora coleosporioides (Ito⁴), and Puccinia kusanoi (Yambe 1978) are less abundant in polluted areas, and the disease severity they induce, is sharply reduced.

On the other hand, in forest stands under influence of industrial and urban air pollution, some foliar pathogens counted by pathologists in the group of so-called 'facultative parasite' are of greater importance because air pollutants injure leaf tissue and predispose trees to attack by facultative parasites. A close correlation has been found between SO₂ in the air and a high incidence of needle blight of pine caused by a weak

parasite, Rhizosphaera kalkhoffii (Chiba and Tanaka 1968).

Although plant injury caused by specific pollutants such as SO₂, O₃, and fluorides has gradually been abated by engineering improvements or by-products recovery, chronic injury caused by a complex of urban air pollutants has assumed additional importance as urbanization and the use of motor vehicles have increased.

Injury to green plants is usually one of the first signs that air pollution is becoming a serious problem, and severity of specific diseases in polluted areas must be regarded as an additional expression of unbalanced natural conditions. These specific diseases and the change of composition of leaf surface microorganisms provide aids for monitoring air pollutants, especially if they are sensitive to very low concentration of air pollutants. These diseases and microorganisms should be considered a useful supplement to the more expensive techniques and instruments used for the identification of chronic air pollutant effects.

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LITERATURE CITED

- Adams, D.F. and C.W. Sulzbach.
1961. Nitrogen deficiency and fluoride susceptibility of bean seedlings. *Science* 133:1425-1426.
- Abeles, F.B.
1973. Ethylene in plant biology. 302pp. New York.
- Berry, C.R.
1964. Eastern white pine, a tool to detect air pollution. *Southern Lumberman* 209(2609):164,166
- Chiba, O. and K. Tanaka.
1968. The effect of sulphur dioxide on the development of pine needle blight caused by Rhizosphaera kalkhoffii Bubák. I. *J. Jap. For. Soc.* 50: 135-139..
- Doi, T.
1919. (Forest protection). 409pp. Tokyo.
- Enoki, M.
1977. (Clonal susceptibility of poplars to ozone). *Ann. Rep.* 1976., Osaka Agr. Centr.: 117-119.
- Heagle, A.S.
1973. Interactions between air pollutants and plant parasites. *Ann. Rev. Phytopath.* 11:365-388.
- Heck, W.W.
1966. The use of plants as indicators of air pollution. *Air Water Poll. Int. Nat. J.* 10:99-111.

⁴ Personal communication from S. Ito, University of Tokyo, January 15, 1980.

- Inoue, I.
1972a. Studies on the influence of smoke-dust from a cement plant on the vegetation of pine forests, especially on Tricholoma matsutake. I. Trans. Mycol. Soc. Jap. 13:196-214.
- Inoue, I.
1972b. Ditto II. ibid:215-288.
- Inoue, T.
1973. (Tree injury caused by sulfur dioxide). Plant Protection 27:8-12.
- Ito, K.
1976. Blight and canker of Cryptomeria japonica D. Don. caused by Cercospora sequoiae Ellis et Everhart. Ann. Phytopath. Soc. Jap. 42:234-236.
- Japanese Environmental Agency.
1979. (White paper on environment for 1979). 2163pp. Tokyo.
- Kaburagi, T.
1930. (Smoke damage to plants). Iwanami Lectures for Biology. 1-39.
- Kobayashi, T.
1967. Critical revision on the genera Rhizosphaera Mangin et Harriot and Rhizophoma Petrak et Sydow, a little-known fungus group associated with needle disease of conifers. Bull. Gov. For. Exp. Sta. 204:91-112.
- Kadota, M. and K. Ohta.
1972. Ozone injury to some Japanese woody plant species in summer. J. Jap. For. Soc. 54:226-229.
- Kösch, G.
1935. Eichenmehltau und Rauchgasschaden. Z. Pfl.-lanzenkr. 45:44-45.
- Linzon, S. N.
1958. The influence of smelter fumes on the growth of white pine in the Sudbury region. Can. Dep. Agr. Publ., Ontario Dep. Lands Forests. 45pp.
- Matuura, S. and N. Kokubu.
1972. Fluorine, its distribution and role in nature. 150pp. Tokyo.
- Scheffer, T. C. and G. G. Hedgcock,
1955. Injury to northern forest trees by sulfur dioxide from smelters. UDSA, For. Serv. Tech. Bull. 1117:49pp.
- Tanaka, K.
1975. The similarity between symptoms caused by air pollutants, insects, and diseases. Agr. and Hort. 50:Preface.
- Tanaka, K.
1976a. The effect of low-level SO₂ fumigation on the development of needle blight of pine caused by Rhizosphaera kalkhoffii. Ann. Rep. 1975., The range of tolerance of biological indicators to pollutants:5-9.
- Tanaka, K.
1976b. Tree diseases in air polluted areas and their usability as indicators. ibid:51-56.
- Tanaka, K.
1977a. Pest problems in the use of poplar clones as indicator plants for photochemical oxidants. Trans. 88th Mtg. Jap. For. Soc.:419-421.
- Tanaka, K.
1977b. Accidental chlorine gas damage to ornamental woody plants. ibid:415-417.
- Tanaka, K., T. Okada and K. Hako.
1974a. Effect of SO₂ on the development of needle blight of pine caused by Rhizosphaera kalkhoffii, Transplanting experiment. Trans. 86th Mtg. Jap. For. Soc.:290-292.
- Tanaka, K., M. Kuwaki, T. Okada and K. Hako.
1974b. A plastic covered chamber with activated carbon filters. ibid:211-213.
- Tanaka, K. and O. Chiba.
1971. On a needle blight of pine caused by Rhizosphaera kalkhoffii--Life history, physiological characteristics and pathogenicity of the causal fungus. J. Jap. For. Soc. 53:279-286.
- Tanita, S.
1976. Histological effects of air pollutants and pathogens on Pinus thunbergii. Trans. 88th Mtg. Jap. For. Soc.:373-374.
- Tokyo Metropolitan Government.
1971. Tokyo fights pollution. 267pp. Tokyo.
- Treshow, M.
1968. The impact of air pollutants on plant populations. Phytopathology 58:1108-1113.
- Yamabe, Y.
1973. decline of trees in Tokyo. Bull. Gov. For. Exp. Sta. 257:101-107.
- Yamabe, Y.
1978. Decline of trees and microbial florae as the index of pollution in some urban areas. Bull. For. & For. Prod. Res. Inst. 301:119-129.

Sensitivity of Lichens to Air Pollution with an Emphasis on Oxidant Air Pollutants¹

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Abstract: The hypothesis that lichens are sensitive indicators of air pollution is now well established for oxidants, sulfur dioxide, hydrogen fluoride and trace elements. From field studies differential sensitivity of different lichen species around pollution sources is evident. Laboratory studies with the particular air pollutant generally confirm the same degree of differential sensitivity.

LICHENS AND THEIR ECOLOGY

Lichens, which strictly speaking are called lichenized fungi, are composite plants representing a symbiotic union of algae and fungi to form morphological entities unlike either component (Ahmadjian 1967, Hale 1967). Although a few species survive in aquatic environments, most are found in terrestrial habitats, occurring on soil, rocks and as epiphytes on other plants. All lichens are perennial plants with life spans varying from a few decades to reportedly thousands of years (Weber and Andrews 1973). Essentially all lichens are autotrophs because of the photosynthetic activity of the algae. Metabolic activity in general is limited to periods when the lichens are moist, a condition which varies in concert with atmospheric moisture conditions as lichens have no roots by which moisture might be absorbed from the substrate. Thus lichens are prominent examples of poikilohydric plants (Larcher 1973) which also include bryophytes and some lower vascular plants, such as desert species of *Selaginella* and ferns. Because of moisture

constraints on photosynthetic activity, primary productivity assignable to lichens is generally small compared to higher plants. Nevertheless lichens may be important in ecosystem functioning. For example, lichens are generally recognized as important pioneer plants of xeric successional sequences because of their role in biologically controlled weathering (Syers and Iskandar 1973). In addition, lichens are important in mineral cycling, not only because of their ability to alter precipitation chemistry (Lang and others 1976), but also from the ability of blue-green algae containing lichens to fix atmospheric nitrogen. Denison (1973) has estimated that lichens are responsible for 50 percent of the nitrogen fixed in the Douglas-fir forests of the Pacific Northwest.

Lichens are morphologically divided into 3 major growth forms: crustose, foliose and fruticose. Crustose species are generally small, tightly attached to (or imbedded in) their substrate and poorly differentiated. Foliose species are generally larger, loosely attached to their substrate and stratified into several distinct layers. In contrast to the basically 2-dimensional foliose species, fruticose species are strongly 3-dimensional and grow out from their substrate. Most fruticose species are also differentiated into distinct layers. Of these morphological forms fruticose species are generally considered to be the most sensitive to air pollution (Fenton 1964), a fact which may be related to the high surface-to-volume ratio of this form.

LICHENS IN RELATION TO AIR POLLUTION

Different species of lichens are well-known to

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be differentially sensitive to air pollution (Nash 1976). A few species actually grow better in urban environments where air pollution levels are high in contrast to their performance in rural areas. Most species, however, exhibit varying degrees of sensitivity to air pollution and some are more sensitive than higher plants. Thus by documenting variations in abundance and species richness of lichen communities, it is frequently possible to identify pollution sources and to document the magnitude of the air pollution problem. Patterns within lichen communities may be attributable to many environmental factors other than air pollution, of course, and consequently it is extremely important that studies be made within a multivariate context, including both pollution factors and non-pollution factors.

The fact that lichens are differentially sensitive to air pollution is based on a series of field observations and laboratory experiments extending back over the past 130 years. Grindon (1859) and Nylander (1867) noted that lichens were disappearing from city centers. Later Arnold (1892) documented that lichens transplanted from rural areas to the center of Munich were not able to survive. Subsequently this "city effect" has been reported for many cities in Europe and North America (Hawksworth 1971). By air pollution most workers of the 1800's generally meant coal soot and related particulates. However, over the past decades a number of invisible gases, such as sulfur dioxide and ozone, have been identified as the major causes of air pollutant injury to plants. Experiments with these air pollutants over the past two decades have also demonstrated differential sensitivity among different lichen species. In general, the sensitivity patterns demonstrated in the laboratory have remarkably corresponded to the patterns observed in the field around pollution sources. Perhaps the strongest evidence supporting the contention that lichens are sensitive to air pollution comes from studies documenting lichen recovery following pollution abatement. For example, Seaward (1976) has documented that Lecanora muralis fairly rapidly reinvaded an urban complex in southern England following amelioration of air pollution in the area.

In the following sections specific evidence related to various types of air pollutants will be reviewed.

Oxidant Air Pollutants

Oxidant air pollutants, including ozone and peroxyacetylnitrate (PAN), are a group of gases characteristic of the oxidizing atmospheres found in the Los Angeles type smog (Pitts and Finlayson 1975). Although the effects of these compounds have been studied extensively with higher plants (National Academy of Sciences 1977), relatively few studies have been conducted with lichens. Probably the only field study in which lichens have been shown to be sensitive to oxidants is that of Sigal and Nash (1980) for the southern

Californian mountains. The study was centered in the San Bernardino Mountains, utilizing the sites employed by the University of California (Riverside and Davis) ecosystem study of air pollutant effects in contrast to sites in a control region, Cuyamaca Rancho State Park in southern San Diego County. In addition, there was an excellent historical record of the lichens of the San Bernardinos based on collections made by H. E. Hasse at the turn of the century. Of the 91 foliose and fruticose lichen species reported by Hasse (1913) in his lichen flora of southern California, only 34 species were found during three summers of study throughout southern California. Thus there has been a significant reduction in the number of species present. To obtain quantitative data on lichen variation, both conifers (Abies concolor, Pinus ponderosa, Pinus jeffreyi and Pseudotsuga macrocarpa) and California black oak (Quercus kelloggii) were sampled for lichen cover at breast height.

In the case of conifers Hasse (1913) reported the presence of 16 foliose and fruticose lichen species in the San Bernardino Mountains. In our recent study (Sigal and Nash 1980) only 8 of these species were found in the San Bernardino Mountains and 4 of these were only present in vestigial quantities. In contrast, at sites to the north and to the south (Cuyamacas) 15 of the 16 species reported by Hasse were found. Thus in the San Bernardinos which lie adjacent to the Los Angeles Basin, there has been a 50 percent decline in species richness of lichens on conifers. Of the species which do occur in the San Bernardinos only two (Letharia vulpina and Hypogymnia enteromorpha) occur commonly. The latter species, although it is common, is definitely showing signs of deterioration (Table 1) when randomly selected thalli from the San Bernardinos are compared with a similar set from the Cuyamacas. The San Bernardino population has a much higher percentage

Table 1-- Morphological characteristics of recent collections of Hypogymnia enteromorpha from the San Bernardinos and the Cuyamacas.

	n	Percent Bleaching			
		un-bleached	slight-ly	moder-ately	bleached
Forest					
San Bernardinos	66	0	30	27	43
Cuyamacas	38	59	41	0	0
	n	Percent Convolutated			
		unconvolutated	slight-ly	moder-ately	convolutated
Forest					
San Bernardinos	66	0	3	53	44
Cuyamacas	38	38	47	15	0

of bleached and convolutated thalli. In addition, there was a decrease of approximately 50 percent in overall thallus dimensions in the San Bernardino population and a decrease in fertility of 42

percent. Thus there are marked trends between the San Bernardinos and the Cuyamacas of reduced species richness and species vitality in the former area. Within the San Bernardinos themselves, significant variation exists. Below approximately 150 ppm-h oxidant dosage, cover of both *Letharia* and *Hypogymnia* is normal in comparison with other areas. At higher dosages of up to 285 ppm-h lichens are almost completely eliminated from conifers.

Comparisons of trends of lichens occurring on black oak between the San Bernardinos and Cuyamacas also indicated that a deterioration was occurring in the San Bernardinos, but the trends were not as dramatic as in the case of lichens occurring on conifers. No direct comparison with the Hasse material was possible because Hasse did not note the species of oak in his collections. From our sampling 39 species of lichens were found on black oak in the Cuyamacas and 30 species in the San Bernardinos. Fourteen of the species found in the Cuyamacas were not found in the San Bernardinos whereas three species were found exclusively in

the San Bernardinos. Among the 20 species which occurred in common there was a general shift from high frequency in the Cuyamacas to relatively low frequency in the San Bernardinos. In the case of cover of the 6 most common species, no significant difference was observed between the two mountain ranges except for *Collema nigrescens*, the only nitrogen fixing lichen in the group, which was completely absent from the San Bernardinos. Further analysis revealed that *Parmelia subolivacea*, the most common lichen, was a successional species. Because older trees occurred in the Cuyamacas, a significantly higher value for this species in the Cuyamacas was predictable. Further evidence supporting the hypothesis that the dominant species was in fact responding to oxidants was obtained by running ordinations (principal component analysis) of the sites within the San Bernardinos and relating the observed variation in species composition to oxidant dose estimates and other environmental variables (fig. 1). An initial ordination (not shown) included all sites sampled within the San Bernardinos. It exhibited a small cluster of low cover sites which occurred at the

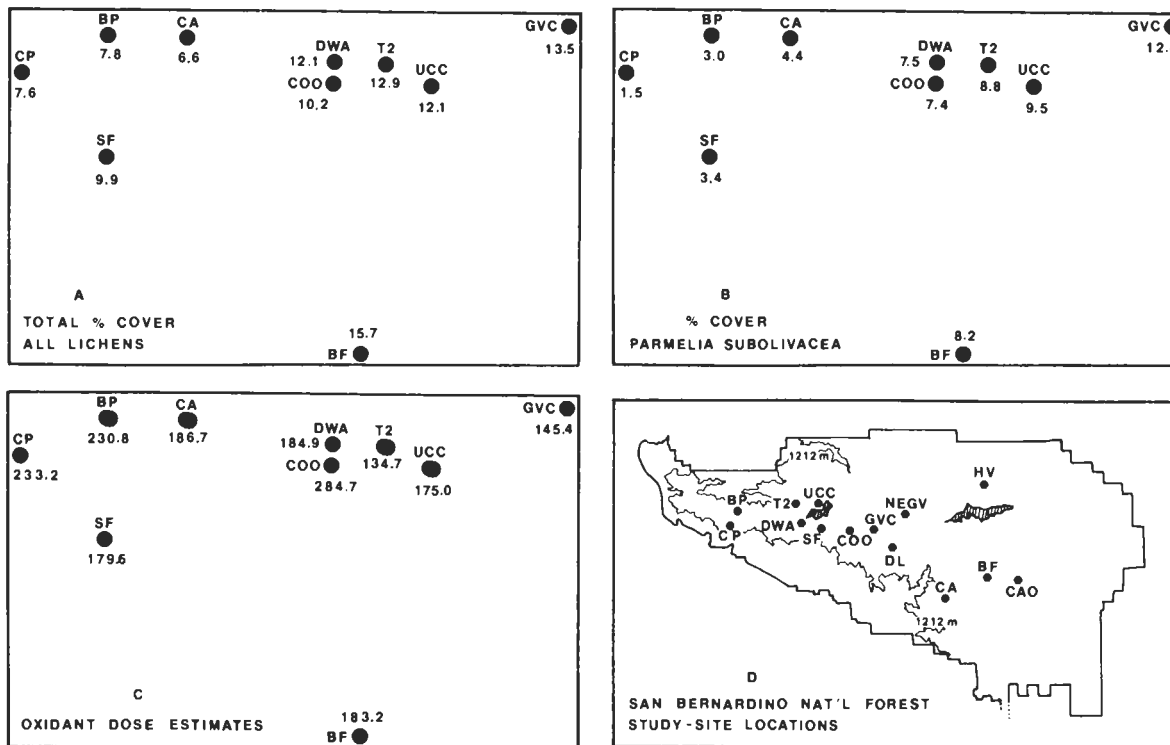


Fig. 1--Principal component analysis ordination of 10 sites in the San Bernardino Mountains using cover data for the 5 most important lichen species occurring on *Quercus kelloggii*. Eighty percent of the variation is explained by axis one (the abscissa). Figure 1A shows the positioning of the sites on the ordination plot with respect to total cover of all lichens; figure 1B, the same ordination but with cover values for *Parmelia subolivacea*; figure 1C, the same ordination, but with oxidant dose estimates (ppm-h); figure 1D, the geographic location of the sites within San Bernardino National Forest.

highest elevations where summer fog is infrequently observed. Consequently these sites and one other site on the desert side of the San Bernardino were excluded from the subsequent analysis. The 10 sites plotted in figure 1 are thus homogeneous with respect to altitude and position along the Los Angeles side of the San Bernardino (fig. 1D). In the subsequent ordination (fig. 1A, 1B and 1C), 80 percent of the variation is explained by axis 1 (the horizontal axis). From figures 1A and 1B it is clear that the primary source of variation is the percent cover of all lichens, which in turn corresponds closely to the percent cover of *P. subolivacea*. Furthermore, there is an inverse relationship between percent cover of *P. subolivacea* (fig. 1B) and oxidant dose estimate (1C). This inverse relationship was substantiated by running a Spearman rank correlation between the two variables. The test was significant with an alpha value of 0.007. For the other 4 species no relationship between oxidant dose estimates and cover values were found. Thus some species have been completely eliminated from the San Bernardino, other species have probably declined in abundance and a few species are apparently unaffected.

Preliminary ozone fumigations have shown differential responses in photosynthesis reduction between *Parmelia sulcata* and *Hypogymnia enteromorpha* with the former species exhibiting greater sensitivity (Nash and Sigal 1979). The *Parmelia*, which grows on black oak, is absent from the San Bernardino whereas the *Hypogymnia* is present, but is exhibiting signs of deterioration. For these species, at least, the pattern of sensitivity observed in the field and the laboratory are complementary. A similar pattern of response for the two species was observed after fumigations with PAN (Sigal and Taylor 1979). Further studies with both ozone and PAN should be run, but the current evidence strongly supports the assertion that oxidants are a major cause of the decline of the lichen flora in the San Bernardino Mountains.

Sulfur Dioxide

Sulfur dioxide is released into the atmosphere by combustion reactions involving products containing the ubiquitous element sulfur, such as coal and oil. It has long been recognized as a phytotoxic agent in general (Daines 1968). Lichens are well known to be sensitive to sulfur dioxide (Nieboer and others 1976). Most of the field studies in northern Europe and North America have involved sites where sulfur dioxide was a major factor (Hawksworth 1971). One of the classic studies by Rao and LeBlanc (1967) demonstrated a severe decline in lichen abundance along a 70+ km transect NNE of Wawa, Ontario where an iron sintering plant is present. Because there is essentially no human development in the affected region, air pollution, and particularly sulfur dioxide, is undoubtedly a major cause of the decline in the region's lichens. Other studies have shown that lichens accumulate sulfur (Olkkonen and Takala 1975, Laaksovirta and Olkkonen 1977) in situations where the presumed

source of sulfur is sulfur dioxide. The gas readily dissolves in water forming a sulfite or bisulfite solution depending on the pH. As a consequence acidification of the substrate frequently occurs, as has been shown in Stockholm where Skye (1968) found that urban tree bark had a pH of more than two units less than corresponding trees from the country-side. Acidification strongly affects the ability of lichens to survive, in part because sulfur dioxide is much more toxic at lower pH's (Tuerk and Wirth 1975). Gilbert (1965 and 1970) has clearly demonstrated the ability of lichens to penetrate into central Newcastle on basic substrates when they are absent from acidic ones. In the laboratory studies, photosynthetic decline in response to short term sulfur dioxide exposures as low as 0.2 ppm have been demonstrated (Tuerk and others 1974). In aqueous experiments with reputedly lower concentrations, effects have also been documented (Baddeley and others 1973, Puckett and others 1974). In addition to reduction of photosynthesis, bleaching of lichen thalli may occur due to phaeophytinisation of the chlorophylls (Nash 1973, Puckett and others 1973). More recently potassium efflux from lichens exposed to sulfur dioxide has been shown to be the most sensitive criterion of response (Tomassini and others 1977). Alteration of membrane permeability may thus be an immediate response to sulfur dioxide exposure. Subsequently it is suggested (Nieboer and other 1976, Puckett and others 1974) that interference in electron flow in photosystem I and II will occur and that proteins will be affected through sulfur dioxide's ability to cleave disulfide linkages. Thus there is very strong evidence from both field and laboratory studies that lichens are sensitive to sulfur dioxide.

Hydrogen Fluoride

Hydrogen fluoride is extremely volatile and is released into the atmosphere during aluminum and rare earth metal refining, fertilizer production and glass and ceramic manufacturing. Patterns of lichen decline are well demonstrated around aluminum factories (Martin and Jacquard 1968, Gilbert 1971, LeBlanc and others 1972, Horntrødt 1975) and around a titanium plant (Nash 1971) and around a phosphate plant (Takala and others 1978). In these areas fluoride accumulation by the lichens has been demonstrated with decline in the lichen communities being associated with elevated fluoride levels. Transplants of healthy lichens into the impoverished zones resulted in fluoride accumulation and subsequent death of the transplants. Gilbert suggested that the critical fluoride level lay between 20 and 50 ppm and Nash independently suggested that the value lay between 30 and 80 ppm. Schoenbeck (1969) found that his transplants died near a fluoride source, but that new transplants survived well for several months following cessation of the fluoride emissions. In a laboratory study with exposure to approximately 5 ppb fluoride over a 9-day period Nash (1971) found that lichens accumulated 84-115 ppm fluoride in contrast to control levels of 14-25 ppm and that the fumigated samples developed chlorotic and necrotic symptoms

similar to the injury observed in the field transplants. Thus fluoride as the probable cause of the injury was confirmed.

Trace Elements

Lichens have a well demonstrated ability to accumulate elements from dilute aqueous solutions (Nieboer and others 1977), a fact probably related to their apparent dependence on atmospheric sources for mineral nutrition. As a consequence of this ability to retain elements, the study of trace element concentrations in lichens is frequently useful in demonstrating "fallout" patterns of particulates and aerosols. For example, mercury is found in elevated concentrations near a chlor-alkali works in Finland (Lodenius and Laaksovirta 1979); lead is elevated in lichens adjacent to highways (Laaksovirta and others 1976, Lawrey and Hale 1979); and a number of elements are present in high concentrations in cities, such as Tel Aviv (Garty and others 1977) and Sendai, Japan (Saeki and others 1977), near the nickel complex at Sudbury, Ontario (Tomassini and others 1976), near a coal-fired power plant (Gough and Erdman 1977) and in coal mine ecosystems of Ohio (Lawrey and Rudolph 1975). In none of these cases is toxicity of the accumulated elements demonstrated. Tolerance to high concentrations of elements may occur if the elements are relatively insoluble and are localized extracellularly. For example, Noeske and others (1970) showed that iron and copper were encrusted on the surface of lichens which grew on metal rich substrates in the Harz Mountains of Germany. Furthermore, Garty and others (1979) have shown that particulates in Tel-Aviv are incorporated extracellularly within the lichen thallus.

In contrast to the above studies "trace" elements do occasionally occur in toxic concentrations. Lawrey and Hale (1979) have shown accumulation of approximately 1000 ppm lead in a lichen growing near an expressway in the vicinity of Washington, D.C. They demonstrated that lead accumulation is correlated with reduction in growth of the species, but they have neither demonstrated experimentally that lead was the cause of the reduced growth nor demonstrated that other factors associated with automobile emissions were unimportant. Probably the most convincing study documenting trace element toxicity is that of Nash (1975) around a zinc smelting complex in eastern Pennsylvania. In the vicinity of the smelting complex, lichen species richness was reduced to 7 in contrast to the 77 species found in a control area (Nash 1972). The zone of lichen impoverishment extended for a distance of 15 km W and 10 km E of the smelting complex (fig. 2). Although detectable and potentially toxic levels of sulfur dioxide were found adjacent to the smelters, the distribution of elevated levels of sulfur dioxide did not extend as far as the lichen impoverishment zone extended. The relative unimportance of sulfur dioxide was further documented by the lack of acidification of the bark of oak trees and by the fact that the

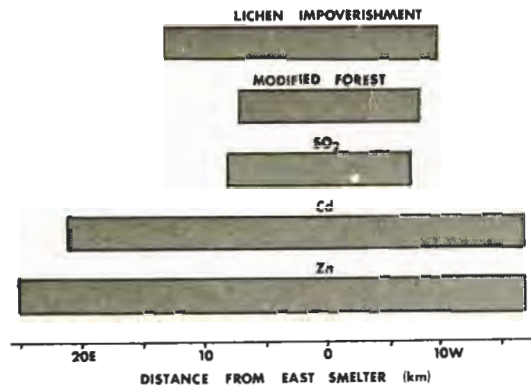


Fig. 2--The distribution of the lichen-impoverished zone (Nash 1972) at Palmerton, Pennsylvania, in relation to the zones of modified forest (Jordan 1975), to detectable levels of ambient sulfur dioxide (Nash 1975), and to elevated levels of cadmium and zinc in surface soils (Buchauer 1973).

company had installed and subsequently continuously used an acid plant at the time that zinc sulfide ores were initially processed. In contrast, metal concentrations in the soil duff were as high as 135,000 ppm zinc and 1750 ppm cadmium at a site adjacent to the smelting complex (Buchauer 1973). The concentration of zinc and cadmium decreased exponentially with distance from the smelters until background concentrations were found at 20 to 25 km E and 16 km W of the smelters (fig. 2). Physiological studies demonstrated that zinc and cadmium were toxic to the lichens when concentrations reached 300-500 ppm in the thallus. Since zinc was present in concentrations 100 times that of cadmium and since zinc is approximately as toxic as cadmium, zinc was inferred to be the most important pollutant. At points beyond the lichen impoverishment zone zinc concentrations ranged from 20-200 ppm and cadmium from 1-30 ppm, where both sets of values were demonstrably non-toxic.

LITERATURE CITED

- Ahmadjian, V.
1967. The lichen symbiosis. 152 p. Ginn Co. (Blaisdell), Boston.
- Arnold, F.
1892. Zur lichenflora von Muenchen. Ber. Bayer Bot. Ges. 2: 1-76.
- Baddeley, M. S., B. W. Ferry and E. J. Finegan.
1973. Sulphur dioxide and respiration in lichens. In Air pollution and lichens. B. W. Ferry, M. S. Baddeley and D. L. Hawksworth, eds. p. 299-313. Athlone Press, London.

- Buchauer, M.
1973. Contamination of soil and vegetation near a zinc smelter by zinc, cadmium, copper and lead. *Environ. Sci. Technol.* 7: 131-135.
- Daines, R. H.
1968. Sulfur dioxide and plant response. *J. Occup. Med.* 10: 516-524.
- Denison, W. C.
1973. Life in tall trees. *Sci. Amer.* 228: 74-80.
- Fenton, A. F.
1964. Atmospheric pollution of Belfast and its relationship to the lichen flora. *Irish Nat. J.* 14: 237-245.
- Garty, J., M. Galun, C. Fuchs and N. Zisapel.
1977. Heavy metals in the lichen Caloplaca aurantia from urban, suburban and rural regions in Israel (a comparative study). *Water, Air and Soil Pollut.* 8: 171-188.
- _____, _____ and M. Kessel.
1979. Localization of heavy metals and other elements accumulated in the lichen thallus. *New Phytol.* 82: 159-168.
- Gilbert, O. L.
1965. Lichens as indicators of air pollution in the Tyne Valley. In *Ecology and the industrial society*. G. T. Goodman, ed. p. 35-47. Oxford Univ. Press, London.
- _____.
1970. Further studies on the effect of sulphur dioxide on lichens and bryophytes. *New Phytol.* 69: 605-627.
- _____.
1971. The effect of airborne fluorides on lichens. *The Lichenologist* 5: 26-32.
- Gough, L. P. and J. A. Erdman.
1977. Influence of a coal-fired powerplant on the element content of Parmelia chlorochroa. *The Bryologist* 80: 492-501.
- Grindon, L. H.
1859. *The Manchester Flora*. London.
- Hale, M. E., Jr.
1967. *The biology of lichens*. 176 p. Edward Arnold Ltd., London.
- Hasse, H. E.
1913. The lichen flora of southern California. *Contrib. U. S. Nat. Herb.* 17: 1-132.
- Hawksworth, D. L.
1971. Lichens as litmus for air pollution: A historical review. *Intern. J. Environ. Stud.* 1: 281-296.
- Horntvedt, R.
1975. Epiphytic macrolichens on Scots pine related to air pollution from industry in Odda, western Norway. *Meddr. Norsk Inst. Skogforsk.* 31: 581-604.
- Jordan, M. J.
1975. Effects of zinc smelter emissions and fire on a chestnut-oak woodland. *Ecology* 56: 78-91.
- Laaksovirta, F. and H. Olkkonen.
1977. Epiphytic lichen vegetation and element contents of Hypogymnia physodes and pine needles examined as indicators of air pollution at Kokkola, W. Finland. *Ann. Bot. Fennici* 14: 112-130.
- _____, _____ and P. Alakuujala.
1976. Observations on the lead content of lichen and bark adjacent to a highway in southern Finland. *Environ. Pollut.* 11: 247-255.
- Lang, G. E., W. A. Reiners and R. K. Heier.
1976. Potential alteration of precipitation chemistry by epiphytic lichens. *Oecologia (Berl.)* 25: 229-241.
- Larcher, W.
1973. *Physiological plant ecology*. 252 p. Springer-Verlag, Berlin. (translated 1975, M.A. Biederman-Thorson).
- Lawrey, J. D. and M. E. Hale, Jr.
1979. Lichen growth responses to stress induced by automobile exhaust pollution. *Science* 204: 423-424.
- _____, _____ and E. D. Rudolph.
1975. Lichen accumulation of some heavy metals from acidic surface substrates of coal mine ecosystems in southeastern Ohio. *Ohio J. Sci.* 75: 113-117.
- LeBlanc, F., D. N. Rao and G. Comeau.
1972. Indices of atmospheric purity and fluoride pollution pattern in Arvida, Quebec. *Can. J. Bot.* 50: 991-998.
- Lodenius, M. and K. Laaksovirta.
1979. Mercury content of Hypogymnia physodes and pine needles affected by a chlor-alkali works at Kuusankoski, SE Finland. *Ann. Bot. Fennici* 16: 7-10.

- Martin, J. F. and F. Jacquard.
1968. Influences des fumées d'usines sur la distribution des lichens dans la vallée de la Romanche (Isère). *Pollut Atmos.* 19: 95-99.
- Nash, T. H., III.
1971. Lichen sensitivity to hydrogen fluoride. *Bull. Torrey Bot. Club* 98: 103-106.
- _____.
1972. Simplification of the Blue Mountain lichen communities near a zinc factory. *The Bryologist* 75: 315-324.
- _____.
1973. Sensitivity of lichens to sulfur dioxide. *The Bryologist* 76: 333-339.
- _____.
1975. Influence of effluents from a zinc factory on lichens. *Ecol. Monogr.* 45: 183-198.
- _____.
1976. Lichens as indicators of air pollution. *Die Naturwissenschaften* 63: 364-367.
- _____. and L. L. Sigal.
1979. Gross photosynthetic response of lichens to short-term ozone fumigations. *The Bryologist* 82: 280-285.
- National Academy of Sciences.
1977. Ozone and other photochemical oxidants. 719 p. Washington, D. C.
- Nieboer, E., D. H. S. Richardson, K. J. Puckett and F. D. Tomassini.
1976. The phytotoxicity of sulphur dioxide in relation to measurable responses in lichens. In T. A. Mansfield, ed. p. 61-85.
- _____, K. J. Puckett, D. H. S. Richardson, F. D. Tomassini and B. Grace.
1977. Ecological and physiochemical aspects of the accumulation of heavy metals and sulphur in lichens. In *Proc. Intern. Conf. on heavy metals in the Environ.* (Oct. 27-31, 1975, Toronto, Ontario), p. 331-352.
- Noeske, O., A. Laeuchli, O. L. Lange, G. H. Vieweg and H. Ziegler.
1970. Konzentration und Lokalisierung von Schwermetallen in Flechten der Erzschlackenhalde des Harzes. *Dtsch. Bot. Ges. Neue Folge* 4: 67-79.
- Nylander, W.
1867. Les lichens du jardin du Luxembourg. *Bull. Soc. Bot. Fr.* 13: 364-372.
- Olkkonen, H. and K. Takala.
1975. Total sulphur content of an epiphytic lichen as an index of air pollution and the usefulness of the x-ray fluorescence method in sulphur determinations. *Ann. Bot. Fennici* 12: 131-134.
- Pitts, J. N. and B. J. Finlayson.
1975. Mechanisms of photochemical air pollution. *Angew. Chem.* 14: 1-15.
- Puckett, K. J., E. Nieboer, W. P. Flora and D. H. S. Richardson.
1973. Sulphur dioxide: its effect on photosynthetic ^{14}C fixation in lichens and suggested mechanisms of phytotoxicity. *New Phytol.* 72: 141-154.
- _____, D. H. S. Richardson, W. P. Flora and E. Nieboer.
1974. Photosynthetic ^{14}C fixation by the lichen *Umbilicaria muhlenbergii* (Ach.) Tuck. following short exposures to aqueous sulphur dioxide. *New Phytol.* 73: 1183-1192.
- Rao, D. N. and F. LeBlanc.
1967. Influence of an iron sintering plant on corticolous epiphytes in Wawa, Ontario. *The Bryologist* 70: 141-157.
- Saeki, M., K. Kunii, T. Seki, K. Sugiyama, T. Suzuki and S. Shishido.
1977. Metal burden of urban lichens. *Environ. Res.* 13: 256-266.
- Schoenbeck, H.
1969. Eine Methode zur Erfassung der biologischen Wirkung von Luftverunreinigungen durch transplantierte Flechten. *Staub* 29: 14-18.
- Seaward, M. R. D.
1976. Performance of *Lecanora muralis* in an urban environment. In D. H. Brown, D. L. Hawksworth and R. H. Bailey, eds. *Lichenology: progress and problems*. p. 323-357. Academic Press, Inc., London.
- Sigal, L. L. and T. H. Nash III.
1980. Lichen communities of southern California mountains: an ecological survey relative to oxidant air pollution. *Ecol. Monogr.* 50: (in press).
- _____. and O. C. Taylor.
1979. Preliminary studies of the gross photosynthetic response of lichens to peroxyacetyl-nitrate fumigations. *The Bryologist* 82: 564-575.
- Skye, E.
1968. Lichens and air pollution. *Acta Phytogeogr. Suec.* 52: 1-114.

- Syers, J. K. and I. K. Iskandar.
1973. Pedogenetic significance of lichens.
In The lichens. V. Ahmadjian and M. E. Hale,
eds. p. 225-248. Academic Press, Inc., New
York.
- Takala, K., P. Kauranen and H. Olkkonen.
1978. Fluorine content of two lichen species
in the vicinity of a fertilizer factory.
Ann. Bot. Fennici 15: 158-166.
- Tomassini, F. D., P. Lavoie, K. J. Puckett, E.
Nieboer and D. H. S. Richardson.
1977. The effect of time of exposure to
sulphur dioxide on potassium loss from and
photosynthesis in the lichen, Cladina
rangiferina (L.) Harm. New Phytol. 79: 147-
155.
- _____, K. J. Puckett, E. Nieboer, D. H. S. Rich-
ardson and B. Grace.
1976. Determination of copper, iron, nickel,
and sulphur by X-ray fluorescence in lichens
from the Mackenzie Valley, Northwest Terri-
tories, and the Sudbury District, Ontario.
Can. J. Bot. 54: 1591-1603.
- Tuerk, R. and V. Wirth.
1975. The pH dependence of SO₂ damage to
lichens. Oecologia (Berlin) 19: 285-291.
- _____, _____ and O. L. Lange.
1974. CO₂-Gaswechsel-Untersuchungen zur SO₂-
Resistenz von Flechten. Oecologia (Berl.)
15: 33-64.
- Webber, P. J. and J. T. Andrews.
1973. Lichenometry: a commentary. Arct.
Alp. Res. 5(4): 295-302.

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Influence of Air Pollution on Population Dynamics of Forest Insects and on Tree Mortality¹

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Abstract: Weakened trees are often predisposed to injury or death by insects, and in forest ecosystems particularly by bark beetles. In the San Bernardino National Forest the interaction between photochemical oxidant weakened ponderosa pine and the western pine beetle (WPB) was examined in detail. The major results from this study suggest that oxidant damaged trees attacked by WPB produce about the same total brood with lower initial attacks compared to healthier trees. This higher productivity trend is most evident in generation 1 trees. Generation 2 trees, both damaged and healthy, are under much greater moisture stress and produce much less WPB brood than generation 1 trees regardless of oxidant damage.

The implication of these results is that in stands with a higher proportion of damaged trees, a given population of WPB could kill more trees and increase at a greater rate than in a stand with a lower proportion of damaged trees. Simulation modelling with these results and other factors that affect ponderosa pine mortality should provide a basis for predicting long term effects of air pollution on the WPB population and pine mortality.

Most of the work on air pollution damage has focused on direct injury to the plant and Kozlowski (1980) gives a good review of the impact of air pollution on forest ecosystems. By comparison, little has been done on the indirect effects such as the predisposing of plants to insects or pathogens. Watt (1969) speculated on the effects of air pollution on population fluctuations of insects and Heagle (1973) reviewed the interaction between air pollutants and plant parasites. More recently the occurrence of diseases and insect pests of trees in air polluted regions of North America has been recorded (Dominik 1978).

There have been some studies on the direct effects of air pollution on insects. Feir (1978) studied the effects of air pollutants on insect growth and reproduction and Hillmann and Benton (1972) looked at the reactions of honey bees to sulfur dioxide.

Bromenshenk (1976, 1978) has studied the effects of coal-fired power plant emissions on a variety of insects. In another study, Gilbert (1971) looked at the indirect effects of air pollution on several bark inhabiting insects. Air pollution may be the cause for the scarcity of all orders of insects in New Jersey (Muller 1971).

The effects of insects in pine stands influenced by air pollution, particularly xylophagous insects, have been studied (Sierpinski 1972, 1977). In California on the San Bernardino Mountains, ponderosa pine shown to have advanced symptoms of oxidant injury were most frequently infested and killed by western pine beetle (Dendroctonus brevicornis), and mountain pine

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beetle (*D. ponderosae*) (Stark et al., 1968). The purpose of this study was to look a little closer at the interaction between the western pine beetle and oxidant affected ponderosa pine, to see what effect diseased trees would have on the dynamics of bark beetle populations and to look for differences between beetle population parameters in diseased and healthy ponderosa pine.

MATERIALS AND METHODS

Field Procedures

Beginning about the first week in July, the San Bernardino mountain areas with substantial ponderosa pine stands were searched for first generation western pine beetle attacked trees. These areas are generally on the southern side of the mountains at about 1500-1800 m elevation, and are areas where high oxidant air pollution levels occur. Second generation trees were located in mid-August to mid-September, depending on the timing of the first generation. Trees under attack were detected by pitch tubes or frass in bark crevices. Attacked trees were checked with an axe at the base to see which species were attacking and to check the stage. Trees with mixed brood (*D. ponderosae* and *D. brevicornis*) and trees with strip attacks (one side only) were relatively uncommon and were not selected for this study. Trees with *Ips* spp. in the tops were selected, but sampling for western pine beetle was stopped at the base of infestation of the *Ips*. Each suitable tree located was used in the study until the required number were found (12 per generation in 1973 and 1974, 6 per generation in 1975 and 1976). In the time period allowed for sampling the initial WPB stages there was no practical way to locate all the attacked trees in the area and then pick a random subsample for the study.

Sampling

Prior to sampling, the major tree characteristics were recorded: location, height, DBH, height of first green branch, and estimated tree damage due to air pollution. The air pollution damage was rated by rating upper and lower crown needle retention, needle condition, needle length, and branch mortality. Needle condition and retention were difficult to rate on the sample trees because in many cases, fading and needle loss due to the bark beetle attacks had begun. For 1973 only one needle length rating was made for the entire tree, so the needle length rating for the other years was combined to be consistent in the analysis (in nearly all cases the upper and lower crown needle length was the same).

Samples for all procedures except where noted later were taken at 1.5 meter intervals over the length of the western pine beetle infestation. Two 88 cm² discs taken on opposite sides of the tree at 1.5 meter intervals gave the desired level of precision.

The samples were taken from the tree using a modified portable 4.5 kg gas powered drill (Drilgin[®], Precision Multiple Control, Ridgewood, New Jersey). A standard circular hole saw with an 11 cm diameter blade was used in the chuck of the drill. This saw cuts an area of approximately 88 cm².

At each sample height the circumference was measured, then the two discs were cut. With the egg discs it was desirable to take a portion of the xylem with each disc to protect the galleries. After a disc was removed it was examined by the climber for insects, which were identified and called down to the notetaker.

During the last larval sample emergence cartons were placed on the trees at each sample height. These cartons were not removed until well after brood emergence.

Laboratory Procedures

Once the samples had been returned to Berkeley, they were placed in cool storage until analyzed. X-rayed samples were placed into rearing immediately after the radiographs were taken. Basically there were four laboratory analysis procedures for sample discs.

Egg Disc Analysis

To determine egg mortality discs were taken at 3.0 meter intervals approximately two to three weeks after the mass attack period. Sampling at every 3.0 m interval gave 4 or more heights for each tree. Egg discs were taken only once, at the time of the first larval sample. This avoids an additional climbing of the sample tree, assures that maximum egg hatch has occurred and mortality can still be determined.

If xylem tissues still remained on the disc it was carefully removed so that the galleries in the phloem tissue were not disturbed. Gallery length was measured with a metric map reader and all attacks were recorded. Each disc was then examined with a dissection scope (10-20x) along the length of all the adult galleries. Each egg niche was evaluated as empty or as containing a viable egg or nonviable egg in addition to recording larval eclosion. The difference between the total number of egg niches recorded and the total number of larvae hatched was calculated as egg mortality.

X-ray Analysis

The first samples to be x-rayed were taken concurrently with the egg discs (egg discs are destroyed during analysis so they can not be placed into rearing). For generation one, an additional x-ray sample was taken late in the brood development period, usually in mid to late July. For the second generation, x-ray samples were removed on two later occasions: once in mid-September to October to check for early emergence of brood and

adult re-emergence, and finally in the following spring (March to April) after brood development was nearly complete, but before spring emergence.

The discs to be x-rayed were taken from cold storage as soon as possible and removed from the plastic bags. The average bark thickness of each disc was measured with a metric caliper and the percentage woodpeckering on each disc was recorded. The two 88 cm² discs from each height were placed on one sheet of 8" x 10" Industrial Kodak® AA-2 film and exposed with a Picker® radiograph machine. The samples were then re-bagged and transported to a large storage shed for rearing (see below).

X-rays were interpreted for western pine beetle, live and dead parent adults, brood adults, pupae, and larvae, parasitoids, predators and miscellaneous. The miscellaneous category included buprestid and cerambycid larvae, weevils, *Ips*, and unknowns, of which there were many judging from the rearing of these discs. A light table was used for interpretation and all inclusions were marked with a grease pencil by the interpreter, counted and then erased. A second interpreter then repeated the process, thereby reducing error and bias. The x-rays from each tree were then stored in envelopes as a permanent record.

Laboratory Rearing

Each x-rayed disc was placed in a 1/2 gallon ice cream carton with a small 2 or 3 dram glass vial pushed through the lid. Since the insects in the discs are positively phototropic at least in the adult stage, and since very few beetles or other insects bore through the sides of the cartons, this was a suitable rearing procedure. The cartons were stored in racks in a large unheated warehouse where Berkeley's mild, cool climate provided an ideal rearing environment.

The emerging insects were collected from the vials three to five times per week. Most of the specimens could be identified with a 10x hand lens and then recorded on a data sheet by collection date. Only the parasitoids and the western pine beetle were sexed. All specimens have been identified by taxonomic experts and specimens were sent for verification regularly; the collection is updated as necessary. Approximately 130 different insects were recorded. In this paper, only WPB, 4 predators (*Enoclerus lecontei*, *Temnochila chlorodia*, *Aulonium longum*, *Medetera aldrichi*) and 4 parasites (*Roptrocerus xylophagorum*, *Dinotiscus burkei*, *Eurytoma conica*, *Coeloides* sp. nr. *brunneri*) are considered.

Sample discs were kept in rearing for nine months and at the end of this period the cartons were examined for any remaining insects; the discs were then discarded. The gallery length and number of attacks were recorded for each disc prior to discarding. Attack holes can be distinguished from emergence holes as they are

oblique to the surface of the bark and often have pitch as a remnant of a pitch tube.

Field Emergence Cartons

As an additional check on emergence, a procedure for rearing *in situ* on the sample trees was also used.

One quart squat ice cream cartons that cover 88 cm² of bark were painted silver on the outside and prepared with a screened ventilation hole. Stikem Special® was placed on the inside of the cartons to discourage insects from boring out or from going back into the bark. A groove was cut in the bark with the sampling saw into which the lip of the carton was placed; a nail was driven through the bottom of the carton and into the tree to increase stability.

The cartons were placed on the tree at the time the last x-ray or brood sample was taken (to coincide with the pupal and/or callow adult stages of the brood) and were left in place for up to three months, until well after beetle emergence and the collection of all associated insects had taken place. The second generation cartons were not placed on the tree until early spring since cartons do not weather well and therefore cannot be left on the trees for four to six months over winter. The cartons were split open and examined under the microscope and then discarded. The same insects that were recorded for laboratory rearing were recorded for the sticky emergence carton.

RESULTS AND DISCUSSION

Attacked tree characteristics and variation by year and generation:

The tree heights of the 71 attacked trees sampled over the four year period varied from 13 to 44 meters, but most trees were in the 20-30 meter range (Fig. 1). Tree DBH, stem volume, and infested bark area varied more widely, but exhibited no significant trends with regard to year and generation. Western pine beetle attack and emergence density, however, both showed significant differences ($P < .05$) by generation. Year differences were not significant for these variables except for a lower attack density for generation one trees of 1976 compared to the generation one trees of the other three years.

Effect of tree oxidant damage on beetle population:

In order to determine if damage to the sample trees had an effect on the western pine beetle populations, the individual components of the damage score were analyzed with respect to beetle attack density. Each individual component of the score can be regarded as an interval level variable, so multiple regression techniques were used to determine their relationship to attack density. The technique used was a full screen analysis

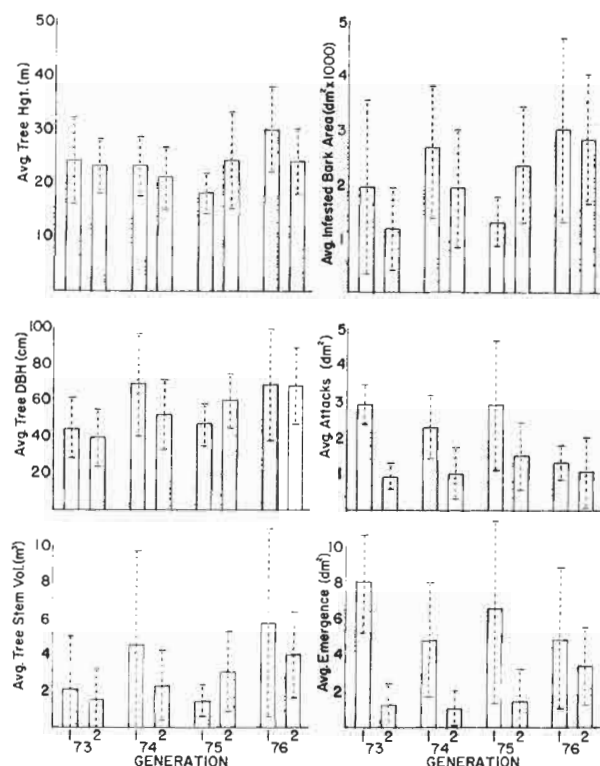


Fig. 1.

Western pine beetle attacked tree characteristics, attack density and emergence density by generation and year for 71 ponderosa pines in the San Bernardino National Forest, California. (dashed lines = mean $\pm$ one standard deviation)

Most other generation one beetle variables were not significantly different between short and long NL trees, except for eggs per attack and emerged brood per attack (Table 1). This probably indicates that the attacking beetles for generation one were more productive when the trees had lower attack density and less competition within the trees. The short NL generation 1 trees, with lower attack densities, had essentially equal numbers of eggs and emerging brood compared to the long NL trees.

The trend toward high productivity in the lower attack density, short NL trees for generation 1 was not evident for generation 2. The beetles in the short NL trees had essentially the same eggs/attack and emerged brood/attack as those in long NL trees (Table 1). Total eggs, hatched eggs, and late larval densities were all significantly lower in the lower attack density, short NL trees. Mortality proportions for each life stage sampled seemed relatively independent of NL although they were higher in generation 2 compared to generation 1.

which calculated R^2 values for all possible linear combinations of the score components, year, and beetle generation for the attack density dependent variable. The results of this analysis indicated that only the needle length (NL) component (upper and lower crown combined) of the score was significant ($P < .05$) along with beetle generation. The effect of any other components was insignificant when added to the equation containing NL and generation. Some of the other components which have been significant in other studies - needle retention and needle condition - probably were unreliable for these trees as they were examined in a partly faded condition some weeks after being mass attacked by bark beetles. In subsequent analyses the needle length (NL) was used to distinguish between trees highly affected by air pollution (NL=0) and less affected trees (NL=1). Table 1 gives mean values of the attack density and other western pine beetle variables for each generation and needle length category. In our sample of trees significantly fewer short NL trees were found in generation one (7 of 36) compared to generation two (16 of 35), but due to the practical limitations of our sample selection procedure, it is uncertain whether this difference holds true for the entire population of attacked trees.

The attack density was significantly higher for long NL trees compared to short NL trees for both generations, while generation two trees had lower attack densities than generation one regardless of NL.

Table 1--Western pine beetle variable means by generation and tree air pollution injury (needle length)

Injury:	Generation 1		Generation 2	
	Short NL	Long NL	Short NL	Long NL
No. of trees	7	29	16	19
Attacks/DM ²	1.69*	2.65	0.72*	1.34
Gallery length/DM ²	65.8	72.8	37.1	43.4
Total eggs/DM ²	71.6	67.8	39.7*	60.1
Eggs/attack	62.3*	30.1	64.7	57.9
Hatched eggs/DM ²	58.3	55.1	30.4*	44.5
(1st instar larvae/DM ²)				
Egg mortality	0.20	0.19	0.25	0.29
Late larvae/DM ²	12.0	17.2	2.37*	4.84
Larval mortality--early to late larvae (LM)	0.78	0.69	0.91	0.85
Emerged WPB ₂ (REAR)/DM ²	6.08	6.16	1.24	1.67
Mortality-late larvae to emergence (REAR) (BM)	0.42	0.58	0.53	0.71
Mortality-eggs to emergence (REAR)	0.91	0.91	0.97	0.97
Emerged brood per attack	4.12*	2.53	2.53	1.93

*Significant differences between NL classes, $P < .05$.

Needle length was also used to compare differences in predator and parasite densities for each generation (Table 2). Predators included the 4 common species: Enoclerus lecontei, Temnochila chlorodia, Aulonium longum, and Medetera aldrichii. Parasites included Roptrocercus xylophagorum, Dinotiscus burkei, Eurytoma conica and Coeloides sp. nr. brunneri. For generation 1, only emerged (REAR) predators/dm² were significantly higher for the long NL trees. For generation 2, initial parasitized larvae/dm² were higher for short NL trees, but late parasitized larvae/dm² were higher for long NL trees.

A much larger increase

in numbers of

parasitized larvae occurred for the gen. 2, long NL trees. Increased woodpeckering, along with a higher final larval density for the long NL trees may explain the increased numbers of parasitized larvae. A similar proportion (about 1%) of initial larvae were parasitized in both cases.

Table 2--WPB predator and parasite means by generation and tree air pollution injury class (needle length).

Injury:	Generation 1		Generation 2	
	Short NL	Long NL	Short NL	Long NL
No. trees	7	29	16	19
Initial pred./DM ²	0.962	1.910	0.53	0.30
Initial parasitized lar./DM ²	0.20	0.36	0.18*	0.08
Late predators/DM ²	1.83	2.75	0.80	0.84
Late parasitized larvae/DM ²	0.56	0.61	0.26*	0.48
Emerged (REAR) predators/DM ²	0.73*	1.50	0.25	0.27
Emer. (REAR) para/DM ²	0.48	0.42	0.22	0.28
% area samples wood-peckered	0.1%	0.8%	7.0%	11.0%

*Significant difference between means of smog injury classes, P < .05 .

Table 3--Ozone effects table: western pine beetle in ponderosa pine.

Ozone dose (1)	Tree needle length	Beetles into trees/dm ² (2)	Beetles out of trees/dm ² (3)	Total beetles (4)	Beetle caused tree mort. rate (4)
low	long	4.0	3.9	constant or decreasing	constant or decreasing
high	short	2.4	3.7	increasing	increasing

Notes:

- (1) Effect depends on individual tree characteristics.
- (2) Based on both generations combined and multiplying female adult attacks by two to account for males.
- (3) Based on both generations combined.
- (4) Assuming constant no. of damaged pines/stand.

The consequences of bark beetle activity in an area are summarized in Table 3 and show that attack rates are lower on oxidant affected trees, but the output of brood is essentially the same in healthy and diseased trees. Bark beetles, assuming no direct deleterious effects of air pollution, should increase in areas with high ozone damage and tree mortality will also increase in these areas. The possible interactions of air pollution, ponderosa pine, and western pine beetle and the effects on forest succession are shown in Table 4.

Table 4--Interaction table: air pollution, ponderosa pine, and western pine beetle.

	Causes a change in		
	WPB pop. level	Ponderosa pine mortality caused by WPB	Forest succession
An increase in:			
air pollution	+	+	+
soil water	?	?	?
foliar injury	++	+	+
soil nutrients	?	?	?
mature tree growth	?	?	?
root disease	+	+	+
forest succession	-	-	0
WPB pop. level	0	++	+
pine mort. caused by WPB	?	0	+

Key:

- 0 = unrelated or not possible
- +
- ++ = large increase
- = a large decrease
- ? = unknown, needs more investigation

LITERATURE CITED

- Kozlowski, T. T.
1980. Impacts of air pollution on forest ecosystems. *BioScience* 30:88-93.
- Watt, K.E.F.
1969. Prospective effects of air pollution on insects. *Can. Entomol.* 101:1235-1238.
- Heagle, A. S.
1973. Interactions between air pollutants and plant parasites. *Annu. Rev. Phytopathol.* 11: 365-388.
- Dominik, J.
1978. Investigations of the occurrence of diseases and insect pests of North American trees in the regions of different degrees of air pollution caused by the industry. Final Report of Investigations covering the period of August 1, 1973-June 30, 1978. Warsaw: Sin., 1978. 91, 21 leaves: Ill.

- Feir, D.
1978. Effects of air pollutants on insect growth and reproduction. *Physiologist* 21:36.
- Hillman, R. C., and A. W. Benton.
1972. Biological effects of air pollution on insects emphasizing the reactions of the honey bee, Apis mellifera to sulfur dioxide. *J. Elisha Mitchell Sci. Soc.* 88:195.
- Muller, J.
1971. Is air pollution responsible for melanism in Lepidoptera and for scarcity of all orders of insects in New Jersey, U.S.A. *J. Res. Lepid.* 10:189-190.
- Gilbert, O. L.
1971. Some indirect effects of air pollution on bark living invertebrates. *J. Appl. Ecol.* 8: 77-84.
- Bromenshenk, J. J.
1976. Biological impact of air pollution on insects. *Ecol. Res. Ser.* 600/3-76-013: 295-312.
- Bromenshenk, J. J.
1978. Investigation of the impact of coal-fired power plant emissions upon insects: entomological studies at the zonal air pollution systems. *Ecol. Res. Ser.* 600/3-78-021:473-507.
- Sierpinski, Z.
1972. The economic importance of secondary noxious insects of pine on territories with chronic influence of industrial air pollution. *Mitt. Forstl. Bundes-Versuchsanst Wein.* 97: 609-615.
- Sierpinski, Z.
1977. Economic significance of noxious insects in pine stands under the permanent impact of the industrial air pollution. *Slyvan.* 64:59-71.
- Stark, R.W., P.R. Miller, F.W. Cobb, Jr., D.L. Wood, and J.R. Parmeter, Jr. 1968. Photochemical oxidant injury and bark beetle (Coleoptera: Scolytidae) infestation of ponderosa pine. I. Incidence of bark beetle infestation in injured trees. *Hilgardia* 39:121-126.

Air Pollutants and Their Effects on Wildlife with Particular Reference to the House Wren (*Delichon urbica*)¹

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Abstract: Injury and death to wildlife from air emissions have been recorded since before the turn of the century. Air pollution can directly affect wildlife (i.e., cause death), or can indirectly affect wildlife (i.e., cause habitat loss). A recent investigation on the chronic and sublethal effects of air pollution on the house martin (*Delichon urbica*) has shown that air emissions can significantly affect the nesting ecology of this species.

INTRODUCTION

Since the end of the 19th century, over 100 air pollution episodes have been reported involving injury and death to animals (Newman 1975, 1979). Although the majority of the incidents involved domesticated animals, a number of incidents have been reported for wildlife since the 1880's (table 1). Overall, the few reports involving wildlife appear to be more a function of economic bias to report injury and death of domestic animals than some innate resistance of wildlife to air pollution. For example, in the early 1900's arsenic emissions from a smelter caused the injury and death of several thousand cattle and horses in rural Montana (Harkins and Swain 1908). No mention was made of injury or death to deer or other wildlife plentiful in the area, although deer and rabbits are known to be very sensitive to arsenic

emissions (Prell 1936; Hais and Masek 1969). Eighty-five percent of the wildlife incidents have been recorded in the last 25 years. The increase in the number of wildlife incidents appears to be due not only to an increase in air emissions but to a better understanding of air pollution effects, broader communications, and a greater interest in reporting such problems.

Table 1--The number of reported air pollution incidents involving wildlife and affected groups (adapted Newman 1979)

Period	Number of Wildlife Groups			Consumer Groups	
	Inci dents	Game	Nonegame	Prim.	Secondary
Before 1900	1	1		1	
1900-1950	3	3	2	5	
1951-1970	8	4	7	8	7
1971 to Present	<u>17</u>	<u>7</u>	<u>15</u>	<u>15</u>	<u>8</u>
TOTAL	29	15	24	29	15

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Recent Federal legislation, namely the 1977 Clean Air Act Amendments, now require formal consideration of the effects of air emissions on wildlife and other components of ecological systems.

The purpose of this paper is to summarize some of the known effects of air emissions on wildlife and present preliminary findings on the ecological effects of air emissions on the house martin (Delichon urbica), an insectivorous bird. For this paper, discussion is limited to terrestrial wildlife and vertebrate consumers. Previous speakers here discussed the effects on invertebrate consumers.

GENERAL EFFECTS OF AIR EMISSIONS ON WILDLIFE

One of the earliest wildlife incidents (1887) involves the death of fallow deer (Dama dama) from arsenic emissions from a silver foundry in Germany (Tendron 1964). As early as 1900 the Royal Society of London was charged to investigate the phenomenon of industrial melanism. This incident is recognized more for its demonstration of "natural" selection than as an environmental problem. Recently, genetic changes in small mammals because of air pollution have been reported (Newman 1980).

One of the earliest detailed descriptions of the harmful effects of air emissions on wildlife is from Germany. Arsenic emissions were the cause of widespread death of game animals in the Tharandt forest of Germany in 1936. Sixty to seventy percent of the red deer (Cervus elephas), roe deer (Capreolus capreolus), and wild rabbits (Oryctolagus cuniculus) died. The deer exhibited defective hair growth and antler formation, cirrhosis of the liver and spleen, and emaciation (Prell 1936).

Adverse effects of air emissions on wildlife have been reported from North America, Europe, and Asia, and have included game and nongame animals; birds and mammals; primary and secondary consumers; and herbivores, omnivores, and carnivores (table 1). The effects on wildlife have included die-offs and other population reductions, physiological abnormalities, disease, physical injury, and bioaccumulation. The majority of wildlife incidents reported involve deer and small birds (Newman 1979). These groups are not necessarily more sensitive but are more likely to be monitored, especially deer, which is a widely managed game species. The status of small birds is not only of scientific concern, but also of more widespread public interest.

Pathways of contamination by air emissions for wildlife are: inhalation, adsorption, and ingestion. Wellings (1970) reported the occurrence of pulmonary anthracosis in urban sparrows (Passer domesticus) compared to rural populations in California. One of the most startling examples of contamination by inhalation involved the die-off of 200 to 500 songbirds near

a British Columbia pulp mill which emitted high concentrations of H_2S and other pollutants. The dead birds showed internal hemorrhaging in the lungs and liver.³

Adsorption of air emissions involves the adhesion of gases or particulates to the external surfaces or external membranes, e.g., cornea of eyes. Light (1973) reports a high incidence of blindness in bighorn sheep (Ovis canadensis) found in mountain areas of California with heavy oxidant levels. Oxidants are known eye irritants. Investigators in Czechoslovakia⁴ have observed early aging of the cornea in hares (Lepus europaeus) found in areas with heavy SO_2 and particulate deposition from power plants and other industries.

Ingestion is the most commonly reported mode of contamination for wildlife. Injury and death to wildlife from ingestion of contaminated food and water have been reported for numerous animal species including rabbits and deer from arsenic emissions (Prell 1936), deer from fluoride emissions (Karstad 1967; Robinette and others 1956; Newman and Yu 1976), and sparrows from cadmium emissions (Nishino and others 1973). There are many reports of the bioaccumulation of air pollutants in wildlife. With a few exceptions, the effects of this bioaccumulation are not known (Newman 1980).

Air emissions may not be lethal to wildlife but may lower the health and resistance of wildlife so that natural stress such as cold and shortage of food may result in further injury or death. Studies on the health of wild hares (Lepus europaeus) living in areas of high air pollution in Czechoslovakia showed physiological responses similar to animals with infections or allergic reactions. There were also changes in the normal age structure of the hare populations (Novakova 1969). The general health of deer populations affected by fluoride emissions is poor (Robinette and others 1957; Karstad 1967; Newman and Yu 1976; Newman and Murphy 1979). The survival of these deer during times of natural stress is questionable. Hais and Masek (1969) report that arsenic contamination of red and roe deer caused emaciation and loss of hair so that many of the animals froze to death during the winter.

Injury or death to vegetation caused by air emissions can have significant indirect effects on wildlife. The loss of food resources and habitat can occur as the result of injury or

³Unpublished report, R.D. Harris 1971. Birds Collected (Die Off) at Prince Rupert, B.C. Canadian Wildlife Service, Prince Rupert, British Columbia.

⁴Personal communication from Eliska Novakova, Institute of Landscape Ecology, Prague, Czechoslovakia.

death to vegetation which provides cover, reproductive habitat, or food for wildlife. Many examples exist of extensive damage to ecosystems from air emissions. Large areas of the mixed coniferous forest of southern California (over 40,000 hectares) have been killed or injured from photochemical oxidants (Taylor 1973). No overall assessment has been made as to the loss of wildlife whose forest habitat has been destroyed. Similar large wildlife habitat losses have occurred in Montana, Tennessee, British Columbia, and Ontario, Canada (Newman 1980).

ECOLOGICAL EFFECTS OF AIR EMISSIONS ON THE HOUSE MARTIN

With few exceptions, there is little information available on the effects of chronic exposure of wildlife to air emissions (Newman 1980). Since 1976, RNDr. Eliska Novakova, CSC, from the Institute of Landscape Ecology, Prague, and myself have been studying the ecological responses of the house martin (Delichon urbica) to chronic air pollution. In this portion of the talk I would like to present some of our preliminary findings.

The house martin belongs to the family Hirundinidae. It is an insectivorous bird which feeds primarily on aphids and diptereans. A migratory species, the martin summers throughout Europe and winters in Africa. As a colonial nesting species, it builds mud nests on the walls of buildings. The nesting locations are used each year, often by the same birds.

Previous studies have indicated that D. urbica is sensitive to air pollution. Feriancova-Masarova and Kalivodova (1965) observed changes in the species diversity of birds, including the house martin, in the area of an aluminum plant in Czechoslovakia. Cramp and Gooders (1967) observed a correlation with increased nesting of D. urbica and a decrease in smoke pollution in London. A follow-up study (Newman 1977) in the vicinity of the same aluminum plant studied by Feriancova-Masarova and Kalivodova in 1965 showed that D. urbica decreased its nesting density with increased air emissions.

In 1978 direct counts of active nests were made in 141 villages and towns located in industrialized (contaminated) and non-industrialized (control or background) areas. Of the 141 nesting localities censused, 101 villages and towns were influenced by air emissions from various industrial sources. Power plants and associated surface mines, chemical plants, local heating plants (coal burning), cement plants, and general urban emissions sources were the major air emission influences. Censusing was conducted at pre-determined upwind and downwind locations from an emission source. This census represented observations of over 20,000 buildings. Records were kept of the number of active nests. A

standard nesting density measure was derived as well as measures of the occupancy of a nesting location and colony size. The environmental attributes of each nesting locality was characterized. Comparisons of background nesting localities were then made to ecologically similar, but contaminated, nesting localities.

The mean nesting density of D. urbica from control areas was 0.565 ± 1.94 nests per side. The range in colony size was 1 to 33 active nests per occupied side, with an average colony size of 2.7 ± 1.1 active nests per occupied side. The average occupancy (the number of occupied sides per suitable side) for the 32 background nesting areas was 21.0 ± 8.1 percent.

The overall nesting density for D. urbica from contaminated areas was 0.393 ± 0.734 active nests per suitable nesting side. This nesting density (30 percent) was significantly lower ($t_{0.002} = 3.674$, $df = 8968$) when compared to background nesting sites. Occupancy was only 13.6 percent. Attempts at nesting were also lower, with only 0.74 attempts per contaminated village compared to 1.12 attempts per background village. Colony size was significantly lower (2.1 ± 1.5 active nests per occupied side).

The house martin responds also to the level of air emissions. The overall nesting density for downwind locations was 0.237 ± 0.378 ($n = 4891$) nests per side compared to 0.415 ± 1.603 ($n = 1969$) nests per side for upwind localities. This difference was significant ($P < 0.001$; $t = 7.263$; $df = 6808$). The overall upwind nesting density was not significantly different from background areas.

Colony size and occupancy also changed with proximity to emission sources. Upwind colony sizes averaged 2.5 ± 1.0 nests per occupied side compared to 1.8 ± 0.8 nests per occupied side for downwind locations. These differences were significant ($P < 0.001$; $t = 3.492$ $df = 99$). Upwind occupancy (18.8 $\pm$ 8.9 percent) was also significantly higher ($P < 0.001$, $t = 5.70$, $df = 99$) when compared to downwind occupancy (9.1 $\pm$ 7.7 percent).

The nesting density for downwind locations decreases significantly as the distance to the emission source decreases (table 2). For upwind locations, the closer to the source, the higher the nesting density; for increased distances, nesting densities declined. The upwind nesting densities closest to emission sources were not significantly different from background nesting densities. Colony size was lower at all downwind distances, but only significantly lower at the two closest distances. Occupancy was significantly lower for all distances censused downwind. For the two closest upwind distance groups (out to 6 km), both colony size and occupancy were not significantly different from background conditions.

Table 2--Comparison of nesting density of *D. urbica* at various distances upwind and downwind from emission sources

Location	Sample Size (No. of Sides)	Mean Nesting Density	Percent Difference From Background
<u>Downwind</u>			
0 to 3 km	1451	0.187 + 1.046	1 -66
3.1 to 6 km	1162	0.213 + 0.966	1 -62
6.1 to 9 km	773	0.260 + 1.323	1 -54
9.1 to 12 km	252	0.286 + 1.425	1 -49
<u>Upwind</u>			
0 to 3 km	531	0.589 + 2.508	2 + 4
3.1 to 6 km	493	0.550 + 1.771	2 - 3
6.1 to 9 km	583	0.293 + 1.020	1 -42
9.1 to 12 km	59	0.372 + 1.193	1 -33

¹Significant, $P < 0.025$.

²Not Significant, $P < 0.05$.

The air dispersion patterns associated with the stack heights influenced the observed effects. For sources with high stacks (greater than 100 meters) the adverse effects on nesting were observed in both downwind and upwind localities. For downwind locations, the lowest nesting density did not occur closest to the emission source, but at an intermediate distance (3 to 6 km). Nesting density was significantly depressed out to 12 km. For low stack sources (less than 100 meters), the lowest nesting density, colony size, and occupancy occurred at the closest distances (0 to 3 km). Average nesting densities were 20 percent of background. In contrast to high stack sources, the nesting density influenced by low stack sources was near normal at the 9.1 to 12 km distance categories. For upwind localities, emissions from high stack sources have an adverse influence, especially close to the source. For low stack localities, upwind nesting densities were either at or above background levels at all distances censused.

When the house martin nests in optimum nesting conditions, such as apartment buildings and feedlots in proximity to water, the effects of air emissions were mitigated. The greatest adverse effects on the nesting ecology of *D. urbica* were observed surrounding power plants and open surface mines. Chemical plant emissions had the next most significant effect on the house martin.

In summary, sublethal and chronic levels of air emissions from industrial sources, such as power plants and chemical plants, adversely affect the nesting ecology of *D. urbica*, including a reduction of the nesting density, colony size, and occupancy of this species. The greatest effects on the nesting pattern of *D. urbica* were observed in areas downwind from the emission sources and at decreasing distances from the air emission sources (fig. 1). For low stack emission sources, greatest effects were observed in the immediate vicinity of the stacks where the highest emission levels are expected to occur. For high stack sources, greatest effects were observed at intermediate distances from the stacks and a greater distance than observed in low stack localities.

Evidence suggests that wildlife populations living in optimal habitat conditions are less affected by air emissions than wildlife populations living in less than optimal conditions. This observation has particular importance for the management and preservation of endangered species.

Overall the effects of air pollutants on wildlife can be dramatic, such as the die-offs, or quite subtle, such as shifts in the age structure of populations. In many cases the effects have been debilitating injuries, often contributing to the death of animals during periods of natural stress. Air emissions can also reduce wildlife populations not only directly, but indirectly through loss of habitat.

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LITERATURE CITED

- Cramp, S., and J. Gooders.
1967. The return of the house martin.
Lond. Bird Rep., No. 31:93-98.
- Feriancova-Masarova, Z., and E. Kalivodova.
1965. Niekoľko poznámok vplyvu fluorových exhalátov v okolí Hliníkarne v Ziari nad Hronom na kvantitu hniezdiacich vtakov (The effects of exhalations from the aluminum plant in Ziar nad Hronom on the spectrum of bird species in the vicinity of the plant). Biologia (Bratislava) 20:341-346.
- Hais, K., and J. Masek.
1969. Vcinky nektých exhaláci na hospodarska zvirata (Effects of some exhalations on agricultural animals). Ochr. Ovzduzi 3:122-125.

Harkins, W.D., and R.E. Swain.

1908. The chronic arsenical poisoning of herbivorous animals.

J. Am. Chem. Soc. 30:928-946.

Karstad, L.

1967. Fluorosis in deer.

(*Odocoileus virginianus*).

Bull. Wildl. Dis. Assoc. 3:42-46.

Light, J.T.

1973. The effects of oxidant air pollution on forest ecosystems of the San Bernardino

Mountains. In Oxidant Air Pollution Effects

on a Western Coniferous Forest Ecosystem.

O.C. Taylor, ed. Part B, pp. 1-12. Task B

Rep. Air Pollution Research Center,

University of California, Riverside,

California.

Newman, J.R,

1977. Sensitivity of the house martin

(*Delichon urbica*) to fluoride emissions.

Fluoride 10(2):73-76.

Newman, J.R.

1979. Effects of industrial air pollution on

wildlife. Biol. Conserv. 15:181-190.

Chronic Effects of Acidic Precipitation and Heavy Metals on Forest Ecosystems

The Acidity Problem—Its Nature, Causes, and Possible Solutions¹

Lowell Smith²

Abstract: Interest within the scientific community in North America and Europe about the nature, effects, and causes of atmospheric acid deposition has grown rapidly over the past decade. This interest has recently intensified because of the explosion in public awareness of, and concern over, the acid deposition problem, and a growing political will to address the problem within appropriate national and international forums. This paper sketches the nature of the acid deposition problem; describes the atmospheric processes that convert precursor emissions into acidic compounds as these are transported over distances ranging from a few to more than a thousand kilometers; discusses past and possible future trends in geographical distribution and rate of acid deposition; and summarizes the governmental activities which have been initiated to address the problem.

The scientific study of acid deposition is archtypical of many contemporary environmental problems, in that it necessarily covers a wide spectrum of disciplinary interests. Simply listing the many subdisciplines involved would fill more than a page. It is important for the active research worker in the field to recognize the many interconnections between her or his own endeavors and other research areas. This involves a careful balance because, at the same time one is encouraging a cross-fertilization of ideas among various disciplinary efforts, one must guard against extending scientific judgements beyond one's own limits of competency. Since the author is fully aware of this hazard, he

invites reaction from any who take exception to the summaries presented in this paper.³

The major features of acid deposition are:

- ° acid deposition results primarily from the combustion of fossil fuels which releases sulfur dioxide (SO₂) and nitrogen oxides (NO_x) in the form of nitric oxide (NO) and nitrogen dioxide (NO₂) to the atmosphere;
- ° depositing acidic material is formed out of these precursor emissions by means of a large number of chemical reactions as the emissions are transported away from their source region;

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³The views expressed in this paper are those of the author and do not necessarily reflect those of the Environmental Protection Agency.

- ° acidic material is deposited dry in the form of fine particulate matter and is incorporated into all of the possible physical forms of precipitation;
- ° transport distances between source and receptor regions can exceed a thousand kilometers, although a major source can, on occasion, significantly affect the rate of acid deposition within the first few kilometers downwind of the source; and
- ° the effects of the deposited acid can vary markedly, depending upon the form in which the acid is deposited, the biologic, geologic and hydrologic pathways between deposition site and the receptor of interest, and the sensitivity of the receptor.

This paper explores these major features of acid deposition.

Nature of Acid Deposition

Acidic compounds may be deposited by several forms of precipitation (rain, snow, hail, dew, rime, and mist) as well as by fine particles that settle out of the atmosphere on to biologic, mineral, and aquatic surfaces, and on to man made materials, buildings, and artistic objects. Because monitoring techniques are not well developed for measuring and characterizing the dryfall component of acid deposition, comparatively little is known about its extent and variability (Durham and Hicks, 1980). Monitoring data from eastern North America indicates that the sulfate ion is the predominant anion present in acidic precipitation; the nitrate ion is associated with about half as much acidity as the sulfate ion, and chloride and other anions make substantially lower contributions (Hales, 1980). In some areas of the West the nitrate ion contributes an equal or larger share than the sulfate ion (McColl, 1980; and Morgan and Liljestrand, 1980).

Some regional scale atmospheric models estimate that up to half of the sulfate component may be deposited dry, while other models estimate a lower percentage (Whelpdale and Galloway, 1979). The variance results from a basic lack of understanding of the physical, as well as biological processes, and their variation over space and time, as these processes transfer gases and particles across the atmosphere/surface interface (Durham and

Hicks, 1980). In addition, nitric acid depositing onto surfaces out of its vapor phase may at times be an important contributor to dry deposition; but even less is known about its ambient concentration and geographical distribution (Altshuller, 1979). Thus, until monitoring techniques are developed for routine reliable measurements of dryfall deposition, we are compelled to rely on isolated spot measurements and tentative inferences to characterize this important phenomenon.

The characterization of wet deposition is a more tractable problem. While it is doubtful that precipitation at any place on the globe is entirely free of anthropogenic contaminants, relatively "clean" rainfall is generally found on the windward edge of continental land masses and where frontal storm systems have traversed large expanses of sparsely populated land surfaces (Granat, 1978). Because of the formation of carbonic acid from the hydrolization of background atmospheric carbon dioxide, such clean rainfall is thought to have a pH of about 5.6, approximately 25 times more acidic than a neutral pH of 7 (Likens, *et. al.*, 1979). Yet even the pH of rainfall relatively unaffected by anthropogenic emissions can vary by several pH units due to the entrainment of alkaline soil particles, reaction with atmospheric ammonia, and possibly other little understood factors (Stensland, 1979).

The concentration of acidity in precipitation is observed to be quite episodic (Smith and Hunt, 1979), which combined with the episodicity of precipitation rates, leads to great variability over space within a particular rain event and over time at a particular location of the rate of acid deposition (Hales, 1980). A range of over seven orders of magnitude of acidity (pH 2 to pH 9) has been recorded for various isolated rainfall events in North America. Averaged over time, spatial variations tend to vanish at locations distant from large sources. An exception is the orographic effect produced by high terrain features on precipitation rates. This effect can be further augmented by the increasing acidity of cloudwater at higher elevations within a cloud structure (Falconer and Falconer, 1980). Terrain subject to orographic precipitation in North America and Europe, such as the Northern Alps, that is downwind of high emission areas has been observed to sustain substantially increased quantities of deposited acidic material than similarly situated low lying terrain (Schrumpff, 1980).

Averaging wet acid deposition rates over an annual cycle for monitoring stations in eastern North America produces a pattern of depressed pH values in the northeast United States (Pack, 1980) extending northward into southern Ontario and Quebec, westward to the eastern Midwest, and southward along the Appalachian ridge into Tennessee and the Carolinas. Less depressed average pH levels extend deep into Florida (Brezonik, *et. al.*, 1980), west to Arkansas and Missouri, and northward an undetermined distance into Canada (Likens and Butler, 1980). The average acidity of precipitation in the central portion of this pattern approaches pH 4, approximately a factor of forty more than what many consider to be the "normal" value of pH 5.6 (Likens and Butler, 1980).

Due to the spottiness of available monitoring information less is known about acid deposition rates west of the Mississippi River. Measurements made in the Boundary Waters Canoe Wilderness Area (northern Minnesota) (Glass and Loucks, 1980), the Colorado Rockies (northwest of Denver) (Lewis and Grant, 1980), and California (the Sierra Nevada, and the Los Angeles and San Francisco Bay basins) (McColl, 1980; and Morgan and Liljestrand, 1980) all suggest that this problem is not unique to the eastern half of the continent.

Effects of Acid Deposition

Acid deposition creates a public policy problem to the extent that potentially sensitive receptors are harmed by the deposition rates they sustain. Research results on acid deposition effects are multiplying rapidly in North America and Europe (International Conference on the Ecological Impact of Acid Precipitation, 1980). An impressive body of information now exists as to the impact of acidifying surface waters on the ecosystems they support (Gorham, 1976). Some fish species show adverse effects below pH 6; while other more tolerant fish species do not evidence serious effects until pH 4 or lower has been reached. Eggs and especially the fry of sensitive species are more susceptible than are adult fish (Johannson, *et. al.*, 1977). Thus, during the initial melting of a snowpack, acid pulses may be released which abnormally skew subsequent fish population distributions to the extent that one or more generations may be completely absent in some instances. Other portions of aquatic food chains can also be affected to the detriment of wildlife species at the top of these chains, including fish eating birds (Peakall, 1980) and freshwater wading birds (Loucks, 1980).

Recent evidence from Europe suggests that leaching of calcium and other nutrients from the lower horizons of sensitive soils and the concomitant mobilization of Al^{3+} ion within these soils (Ulrich, 1980) is promoted by atmospheric acid deposition. Also observed is a reduction of the decomposition rate of forest litter and decreased numbers of micro-organisms (Baath, *et. al.*, 1978) within the

upper soil horizons of forest soils subjected to heavy rates of acid deposition. In extreme cases ground water quality may be adversely affected (Hultberg and Wenblad, 1980). Abnormally low pH for drinking water supplies drawn from wells in some of these regions has been observed. This can result in unacceptably high concentrations of copper and lead in drinking water in those homes which are equipped with copper plumbing systems (Hultberg and Wenblad, 1980).

Agricultural crop species cultured in pots and subjected to lowered pH simulated acid rain irrigation water have displayed mixed responses in yield (Lee, *et. al.*, 1980). Some cultivars exhibit enhanced yields and others depressed yields while many cultivars show no discernible effect on yield. The effect of acidic deposition on forest growth rates is more uncertain. Early stages of acidification can accelerate growth rates for a few years, possibly due to the increased mobilization of nutrients within the soil structure. But sustained elevated rates of acid deposition are hypothesized to depress forest growth rates in geologically sensitive areas due to nutrient depletion and elevated concentrations of Al^{3+} ion within the root zone. Other ecological effects are discussed elsewhere (Overrein, 1980, Norton, *et. al.*, 1980, and Cowling, 1980).

Mechanisms of Atmospheric Formation

As previously stated, acid deposition is primarily the result of the anthropogenic release of SO_2 and NO_x to the atmosphere where these react through several available chemical pathways to sulfuric and nitric acids. To a lesser extent hydrochloric acid can also be involved. Coal and oil fired electric utility generating stations, predominantly in the central and eastern portions of the U.S. are responsible for two-thirds of the national SO_2 emission inventory (U.S. EPA, 1979a). Nine-tenths of these emissions are from coal-fired stations. Industrial and commercial boilers account for nearly half of the remaining emissions. SO_2 emissions in the U.S. are likely to increase slowly over the next few decades as emissions from well-controlled new sources slightly overbalance emission reductions expected to be achieved by the control or retirement of existing sources (Altshuller and McBean, 1979). Canadian SO_2 emissions come predominantly from their non-ferrous smelter industry at present, but by the end of the century coal fired utility boiler emissions are expected to about equal those from the smelter industry (Choquette, 1980).

Two-fifths of the nation's NO_x emissions come from motor vehicles. Geographically, these mobile sources are clustered around large population centers. Nine-tenths of the remaining NO_x emissions are released by the same large utility and industrial boilers which are responsible for the predominant portion of the SO_2 inventory (U.S. EPA, 1979a). NO_x emissions are expected to increase significantly in the U.S. and Canada over the remainder of the century,

unless some promising breakthrough in NO_x emission controls is achieved and rapidly implemented by industry (Altshuller and McBean, 1979).

Although no emission inventory exists for chloride ions, these emissions are likely to be primarily the result of burning high chloride coals produced from many coal seams in the midwestern and eastern U.S. Therefore, chloride emissions are likely to be colocalized with large SO₂ and NO_x emitting sources. Natural emissions of SO₂ (Adams, *et. al.*, 1980) and NO_x⁴ make only modest additions to the anthropogenic sources in eastern North America, though natural emissions may make an appreciable contribution to the global background of these atmospheric constituents (Husar, *et. al.*, 1978).

These precursor emissions react in the atmosphere with water vapor, other minor atmospheric constituents and sunlight to produce fine sulfate particles, nitric acid vapor and dilute acids in cloud droplets. These submicron particles agglomerate into larger particles until a natural barrier to further growth is reached at slightly above one micron mean diameter. The growing particle may be deposited in dry form onto a surface, incorporated into a cloud droplet (rainout), or scavenged by a falling raindrop (washout) (Hales, 1980). The hydroxyl ion in conjunction with sunlight is believed to promote the formation of particulate sulfate (Davis, *et. al.*, 1974), while the peroxy ion is believed to promote the conversion of SO₂ to sulfuric acid in cloud droplets.⁵

In the summer months, substantially enhanced concentration of sulfate ion are found in deposited rainwater while nitrate ion concentrations tend to be more constant throughout the year (Hales, 1980). There is some evidence to suggest that nitrogen oxides are deposited more rapidly from the atmosphere on the average than are sulfates (Mueller, *et. al.*, 1979), so nitric acid deposition may be relatively less important for receptor sites far from emission regions than is sulfuric acid deposition.

Another complicating feature in the atmospheric chemistry of acid deposition is the atmosphere's ability to partially neutralize its acid load. Alkaline fine wind blown soil particles, particularly over arid regions, appear to neutralize the acid load (Eisenreich, *et. al.*, 1980), or create aerosols with basic chemical properties. The ammonium ion is also effective in partially neutralizing dry and wet atmospheric acidic materials. Many natural and anthropogenic sources produce ammonia, but a complete ammonia emissions

inventory has yet to be constructed. Stockyards, municipal waste water treatment plants, certain industrial processes, and decaying vegetable matter are among its important sources.

Some investigators have hypothesized that fly ash from coal combustion has historically played a major role in reducing atmospheric acidity and that recent efforts to control fly ash emissions have noticeably worsened the acid deposition problem (Frohlinger and Kane, 1975). Such an effect is unlikely to have been nearly as important as was once supposed. As will be discussed later in this paper it is unclear what the historic trends for deposited acid are in the Northeast. Thus, such an *ad hoc* explanation may not be relevant to explain what is an inconclusive trend. More importantly, considerations such as the size of fly ash particles relative to the size of sulfate particles, the chemical composition of fly ash from midwestern and eastern coals, and the change from stoker-fired coal boilers to pulverized coal boilers during this period, all suggest that the neutralizing effect of emitted fly ash could only have been important in the immediate locality of a relatively few heavy fly ash emitting sources.

Mechanisms of Atmospheric Acid Transport

Several physical processes are important to the atmospheric transport of acid precursors and their acidic products. Particularly important for the SO₂/sulfuric acid/sulfate complex is the elevated height of injection into the atmosphere level for SO₂ emissions from most major power plant sources. A nocturnal inversion layer frequently isolates these tall stack plumes from the ground (Smith, *et. al.*, 1978) until after sunrise when the incident solar energy begins to mix the atmosphere through the activation of convective cells. In the Midwest during summertime conditions, a nocturnal bulge in the wind speed vertical profile is frequently observed at normal tall stack plume heights. This condition can transport emissions from a tall stack several hundred kilometers overnight (Smith, *et. al.*, 1978). The gas to particle conversion of the emissions in this displaced plume can be greatly accelerated the next day as these transported emissions are mixed with a polluted urban air mass.

The highest ambient concentrations of particulate sulfate are observed under summertime conditions when a synoptic scale high pressure system stalls for more than a day over a region of high SO₂ emission density, such as the Ohio River Basin (Hidy, *et. al.*, 1978). In this situation weakly circulating winds can trap a large air mass while it is being continually filled with precursor emissions (Vukovich, 1979). The higher temperature, moisture and sunlight levels generally encountered under these conditions tend to increase the chemical reactivity of the atmosphere, so the higher concentration of precursor emissions can more rapidly be converted to particulate sulfate.

⁴Personal communication from Dr. Rudolph Husar, August 15, 1980, Washington, D. C.

⁵Personal communication from Dr. A. L. Lazrus, May 21, 1980, Washington, D. C.

A common atmospheric cleansing mechanism for this condition is for a cold front to approach the high pressure center from the north or northwest (Whelpdale, 1978). This creates a strong pressure gradient which sweeps much of the polluted air mass parallel to the line of the front for many hundreds of kilometers, frequently to the northeast. (LaFleur and Whelpdale, 1977). Frontal storm activity can further remove considerable amounts of pollutants as rainout and washout. Similarly, large convective storms are believed to be an efficient mechanism for processing and removing pollutants from the large volume of air they entrain. These storms are capable of pumping large quantities of polluted air from the planetary boundary layer to high elevations where these can be left as isolated patchy layers to be transported considerable distances and eventually deposited.

Alternating periods of stagnation and ventilation over a high emission area produce episodic concentrations of pollutants (Ottar, 1978) which result in highly variable acid deposition rates in downwind regions. Likewise, the variability in wind direction, as it guides and mixes isolated plumes from major sources and the regional scale plumes described above, adds to the temporal variability of deposition rates at any monitoring site. Thus, acid deposition must be viewed as a stochastic process which, in areas not under the influence of a major source and not affected by orographic terrain effects, is temporally and spatially chaotic over a small scale but is rather spatially homogeneous within a given region when averaged over a large number of events.

Historical Trends in Acid Deposition Rates

While the routine measurement in a scientifically reliable manner of wet deposited acid has only recently, with the exception of a very few monitoring stations, been undertaken in this country, it has been practiced in several European countries over a longer period of time (Granat, 1978). Both the European and North American experiences have demonstrated the need for strict quality assurance procedures for collecting and analyzing the rain water samples. Failure to establish sufficiently stringent quality assurance procedures early in monitoring programs has created questions about the validity of much of the early data (Tyree, 1980).

The need to calculate retroactively the acidity of monitored rainfall using one of several ion balancing procedures, has compounded these measurement difficulties since acidity or pH was usually not measured directly (Kramer, 1978). Such procedures can propagate the experimental uncertainties, which were introduced by the collection and laboratory procedures employed, into rather sizable uncertainties in the calculated hydrogen ion concentration. Further careful analysis of all

available rainwater chemistry monitoring data is required in order to establish the level of certainty with which historical trends of acid deposition in North America can be determined.

Fortunately, it may be possible to describe the gross features of deposition trends by relying on other related physical phenomena for corroborative support. Since sulfate aerosols are highly efficient light scatterers due to their characteristic submicron size, and since sulfate is the dominant component of Eastern aerosols (U.S. EPA, 1979b), visibility trends probably serve as a useful indicator for dry sulfate deposition trends. The ratio of dry to wet sulfate deposition should have remained relatively constant for a particular area unless a climate change has occurred.

Visibility measurements have been routinely made at medium and large size airports in the U.S. for several decades. Recent analysis of these data shows that summertime visibility has significantly deteriorated throughout large portions of the eastern U.S. (Trijonis and Shapland, 1979). Some regions such as the Tennessee Valley appear to have sustained nearly a factor of two decrease in average summertime visibility over a twenty-five year period (Husar, *et. al.*, 1979). Similar trends in summertime solar insolation are also suggestive of a increase in atmospheric turbidity during this same period. Further analysis effort is required to assess the possible causal relationships between trends in precursor emission rates and these trends in environmental conditions.

Although the trends of such surrogates for acid deposition may only be used to corroborate an inconclusive record for monitored acid deposited, other studies strongly support the conclusion that anthropogenic emissions are deposited on ecosystems far from their point of origin. Dated lake bottom cores from remote lakes in North America and Greenland icecap cores show a marked increase in deposition rates for fossil fuel combustion-related pollutants shortly after the beginning of the industrial revolution.

A review of emission trends from U.S. sources over the past forty years indicates several clear trends. First, SO₂ emissions increased about forty percent. Although SO₂ emissions have decreased from most economic sectors during this period, the electric utility sector's emissions increased by more than a factor of six during this same period (EPA, 1978). Second, the increase in SO₂ emissions from this one sector occurred concurrently with a substantial increase (by approximately a factor of five) in the stack height for utility sources. Third, SO₂ emissions from coal burning changed from a wintertime peak to a summertime peak in emission rate (Husar, *et. al.*, 1979). Fourth, the precursor emissions for photochemical oxidants increased markedly during this time (EPA, 1978). At the beginning of the period photochemical smog was hardly recognizable as a problem, whereas, currently urban plumes of photochemical oxidants now frequently blanket the Northeast during the summer months (Altshuller, 1978). Fifth, total NO_x emissions approximately quadrupled during this period (EPA, 1978).

These trends suggest a situation in which the atmosphere has become chemically more reactive, and for which greater quantities of acid-forming precursors are added to the atmosphere during its most reactive period. Further, a substantially greater quantity of these emissions is now injected high into the mixed layer where the emissions and their reaction products have much longer residence times as they travel to areas remote from their point of origin.

Possible Solutions

At present there are no regulatory requirements that are primarily directed at reducing emissions to control acid deposition. However, the Clean Air Act's requirement to reduce ground level SO₂ concentrations has achieved modest overall reductions in SO₂ emissions during the first part of the past decade (U.S. EPA, 1979a). But, average stack heights continued to increase, as did the emission rates for NO_x, over this period. Regulatory requirements established for new coal-fired utility boilers mandate control of seventy to ninety percent of their SO₂ emissions and require NO_x emission reductions of approximately forty percent (U.S. Federal Register, 1979). EPA is currently developing new source control performance standards for industrial boilers which could require similar levels of control for this important source category. From a regulatory perspective the principal problem for SO₂ emission control is the control of emissions from existing sources. In addition, there are large opportunities for improved NO_x control requirements for new and existing sources.

Others have maintained that the costs of emission control are so high that the only cost-effective mitigation measure is to raise artificially the pH of affected lake water by the addition of lime (Barnes, 1979). Repeated treatments would be required as long as acid deposition rates exceeded the geologically controlled release rates for a given area (Horn, *et. al.*, 1980). Recent Swedish experience indicates that liming the entire watershed for a lake may be necessary, while others believe that even though it is possible to raise the pH of an acidified lake, it is not possible to restore the lake to a natural condition containing its original food chains. Others have observed that the potentially sensitive areas in North America cover millions of hectares, and question the feasibility of liming such a vast area for hundreds of years.

Federal government efforts to address the acid deposition problem include:

- ° monitoring activities to establish more fully the geographical variation and temporal trends in rainwater chemistry;

- ° research activities to determine more completely the range of environmental effects produced by acid deposition and the atmospheric processes which transport and convert emissions into acid deposition;
- ° assessment activities to determine the potential seriousness of the acid deposition problem in North America and the most cost-effective measures that could be employed as first steps to combat the problem;
- ° the creation of a ten year inter-agency Federal Acid Rain Assessment Program to coordinate the above activities;
- ° working with the states in order to promote a mutual understanding of the nature and causes of the acid deposition problem, and to encourage the states to engage in collective problem solving on this issue; and
- ° becoming a signatory to an international Convention on Long-Range Transboundary Air Pollution, developed under the auspices of the United Nations Economic Commission for Europe, and pursuing bilateral discussions with Canada which are expected to evolve into formal negotiations on a U.S.-Canada bilateral transboundary air pollution agreement.⁶

Such efforts cannot be expected to bring about immediate and complete relief from the acid deposition problem - nor are they designed to achieve this objective. Rather the goals are to establish as quickly as is feasible a scientific basis for understanding the full range of receptors at risk and the extent of risk to each receptor category; to determine the extent of the control measures which are required to reduce the acid deposition problem to an acceptable level; to develop new air pollution control policies and strategies as necessary and appropriate; and to promote full cooperation and understanding across interstate and international borders to deal effectively with this complex and challenging problem.

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LITERATURE CITED

- Adams, D. F., S. D. Farwell, M. R. Pack, and E. Robinson.
1980. Biogenic Sulfur Emissions in the SURE Region. EPRI Report No. EA-1516, Palo Alto, Calif.
- Altshuller, A. P.
1978. Association of Oxidant Episode With Warm Stagnating Anticyclones, J. Air Poll. Control Assoc. 28 (2): 152-155.
- Altshuller, A. P.
1979. Transport and Fate of Sulfur and Nitrogen Containing Pollutants Related to Acid Precipitation. Draft Report submitted to Council of Environmental Quality, Washington, D. C.
- Altshuller, A. P., and G. A. McBean, co-chairman.
1979. The LRTARP Problem in North America: A Preliminary Overview, Report prepared by the United States-Canada Research Consultation Group on the Long-Range Transport of Air Pollutants, released jointly by the U.S. and Canadian governments on October 15, 1979, Ottawa, Ont. and Washington, D. C.
- Baath, E., B. Berg, U. Lohm, B. Lundgren, H. Lundkvist, T. Rosswall, B. Soderstrom and A. Wiren.
1980. Effects of Experimental Acidification and Liming On Soil Organisms and Litter Decomposition in a Scots Pine Forest. Pedobiologia 20:(in press).
- Barnes, R. A.
1979. The Long Range Transport of Air Pollution: A Review of European Experience. J. of Air Pollution Control Assoc. 29(12):1219-1235.
- Brezonik, P. L., C. D. Hendry, and E. S. Edgerton.
1980. Acid Precipitation and Sulfate Deposition in Florida, Science, 208: 1027-1029.
- Choquette, P. J.
1980. Acid Precipitation: Trends and Projections in Precursor Pollutant Emissions in Canada and in The United States - A Comprehensive Survey, Report prepared by the Pollution Data Analysis Division of the Air Pollution Control Directorate, Environment Canada, 24 p., Ottawa, Ontario.
- Cowling, Ellis B.
1980. Effects of Acidic Precipitation on the Health and Productivity of Forests. This symposium.
- Davis, D. D., G. Smith, and G. Glauber.
1974. Trace Gas Analysis of Power Plant Pollutants Via Aircraft Measurements: O₃, NO_x, SO₂ Chemistry. Science 186: 733-736.
- Eisenreich, S. J., J. D. Thornton, E. Gorham, and W. Munger.
1980. Acidity and Trace Element Composition and Loading from Event Rain and Snow to the Sensitive Remote Areas of Northern Minnesota. In Proc. of the Conf. on the Ecological Impact of Acid Precipitation (March 11-14, 1980, Sandefjord, Norway), in press.
- Falconer, R. E., and P. D. Falconer.
1979. Determination of Cloud Water Acidity at a Mountain Observatory in the Adirondack Mountains of New York State. Report No. 741, Atmospheric Sciences Research Center, State Univ. of N.Y., Albany, N.Y.
- Frohlinger, J. O., and R. Kane.
1975. Precipitation: Its Acidic Nature. Science 189: 455-457.
- Gibson, J. H. and C. V. Baker.
1979. NADB First Data Report, July 1978 through February 1979. Natural Resource Ecology Laboratory, Colo. State Univ., Fort Collins, Colo.
- Glass, Gary E. and Orie L. Loucks.
1980. Impacts of Airborne Pollutants on Wilderness Areas Along the Minnesota-Ontario Border. EPA-600/3-80-044, 187 p., Environmental Research Laboratory, U.S. Environmental Protection Agency, Duluth, Minn.
- Gorham, E.
1976. Acid Precipitation and Its Influence Upon Aquatic Ecosystems: An Overview. Water, Air, Soil Pollution 6:457-481.
- Granat, L.
1978. Sulfate in Precipitation As Observed by the European Atmospheric Chemistry Network. Atmos. Environ. 12: 413-424.
- Hales, J. M.
1980. Precipitation Chemistry: Its Behavior and Its Calculation, Proc. of

- the Int. Conf. on Air Pollutants and Their Effects on Terrestrial Ecosystems, (May 10-17, Banff, Alberta), John Wiley & Sons (to be published).
- Hidy, G. M., P. K. Mueller, and E. Y. Tong.
1978. Spatial and Temporal Distributions of Airborne Sulfate in Parts of the United States. *Atmos. Environ.* 12: 735-752.
- Horn, Edward, James Colquhoun, and Harry H. Hovey.
1980. LRTAP: Acid Precipitation. *J. of Air Pollution Control Assoc.* 30(5): 462-464.
- Hultberg, H. and A. Wenblad.
1980. Acid Groundwater in the Southwestern Area of Sweden. In *Proc. of the Int. Conf. on the Ecological Impact of Acid Precipitation* (March 11-14, 1980, Sandefjord, Norway), in press.
- Husar, R. B., D. E. Patterson, J. M. Holloway, W. E. Wilson, and T. E. Ellestad.
1979. Trends of Eastern U. S. Haze Since 1948, *Proc. of the Fourth Conference on Atmospheric Turbulence, Diffusion, and Air Pollution*, (January 15-18, Reno, Nevada), American Meteorological Society, Boston, MA.
- Husar, R. B., J. P. Lodge, Jr., and D. J. Moore, eds.
1978. *Sulfur in the Atmosphere: Proceedings of the International Symposium on Sulfur in the Atmosphere* (September 7-14, 1977, Dubrovnik, Yugoslavia), Pergamon Press, Oxford, Eng.
- International Conference on the Ecological Impact of Acid Precipitation.
March 11-14, 1980. Sandefjord, Norway, proceedings in press.
- Johannson, N., P. Runn, and G. Milfrink.
1977. Early Development of Three Salmonid Species in Acidified Water. *Zoon.* 5:127-132.
- Kramer, J. R.
1978. Acid Precipitation, Chapter 9 in *Sulfur in the Environment-Part 1: The Atmospheric Cycle*. Wiley Interscience Series, John Wiley & Sons, New York.
- LaFleur, R. J. and D. M. Whelpdale.
1977. Spatial Distribution of Sulfates Over Eastern Canada During August 1976, Presented at the 70th Annual Meeting of the Air Pollution Control Association, June 20-24, Toronto, Ontario.
- Lee, Jeffrey, J., Grady E. Neely, and Shelton C. Perrigan.
1980. Sulfuric Acid Rain Effects on Crop Yield and Foliar Injury. Report No. EPA-600/3-80-016, 20 p. Environmental Research Laboratory, U.S. Environmental Protection Agency, Corvallis, Ore.
- Lewis, William M., Jr. and Grant, Michael C.
1980. Acid Precipitation in the Western United States. *Science* 207:176-177.
- Likens, G. E., and Butler, T. J.
1980. Recent acidification of precipitation in North America. *Atmos. Environ.* 14, in press.
- Loucks, Orie L.
1980. Acid Rain: Living Resource Implications and Management Needs. Presented at 45th North American Wildlife and Natural Resources Conference (March 22-26, 1980), Miami Beach, Florida.
- McColl, John G.
1980. A Survey of Acid Precipitation in Northern California. CARB Contract A7-149-30 and UCB Project CARB-SPN-3664-H, 94 p., California Air Resources Board, Sacramento, Calif.; and Agricultural Experimental Station, University of California, Berkeley, Calif.
- Morgan, J. J., and Liljestrang, H. M.
1980. Measurement and Interpretation of Acid Rainfall in the Los Angeles Basin. Report No. AC-2-80, 54 p., W. M. Keck Laboratory of Environmental Engineering Science, California Institute of Technology, Pasadena, Calif.
- Mueller, P. K., G. M. Hidy, K. Warren, T. L. Lavery and R. L. Baskett.
1979. The Occurrence of Atmospheric Aerosols in the Northeastern United States. In *Proc. of Conf. on Aerosols: Anthropogenic and Natural Sources and Transport*, New York Academy of Sciences, New York, 1980.
- Norton, Steve A., Richard J. Campena, and Harold E. Young.
1980. The Impacts of Acidic Precipitation and Heavy Metals on Soils in Relation to Forest Ecosystems. This symposium.

- Overrein, Lars.
1980. Results of the Norwegian Study of the Effects of Acidic Precipitation on Forest and Fish Resources. This symposium.
- Ottar, B.
1978. An Assessment of the OECD Study on Long Range Transport of Air Pollutants (LRTAP). Atmos. Environ. 12: 445-454.
- Pack, D. H.
1980. Precipitation Chemistry Patterns: A Two-network Data Set. Science 208: 1143-1145.
- Peakall, David B.
1980. Acid Precipitation and Wildlife. In Proc. of Action Seminar Acid Precipitation (Nov. 1-3, 1979), Toronto, Ontario.
- Schrimpf, E.
1980. The Relationship Between Relief and the Precipitation Impact of Polycyclic Aromatic Hydrocarbons, Pesticides, Trace Metals and Some Macro-Elements in Northern Bavaria. In Proc. of the Int. Conf. on the Ecological Impact of Acid Precipitation (March 11-14, 1980, Sandefjord, Norway), in press.
- Smith, T. B., D. K. Blumenthal, J. A. Anderson, and A. H. Vanderpol.
1978. Transport of SO₂ in Power Plant Plumes: Day and Night. Atmos. Environ. 12: 605-611.
- Smith, F. B., and Hunt R. D.
1979. The Dispersion of Sulfur Pollutants Over Western Europe. Phil. Trans. R. Soc. Lond., A. 290: 523-542.
- Stensland, G.
1979. Precipitation Chemistry Trends in the Northeastern United States. Proc. of Twelfth Annual Rochester Inter. Conf. on Environmental Toxicity: Polluted Rain (May 21-23), Rochester, New York.
- Trijonis, J. and T. Shapland.
1979. Existing Visibility Levels In the U.S. Report prepared for the U.S. EPA, Research Triangle Park, NC.
- Tyree, S. Y., Jr.
1980. Rainwater Acidity Measurement Problems. Atmospheric Environment (to be published).
- Ulrich, B.
1980. Deposition, Production and Consumption of Hydrogen Ions In a Beech and Spruce Ecosystem In the Solling District. In Proc. of the Int. Conf. on the Ecological Impact of Acid Precipitation (March 11-14, 1980, Sandefjord, Norway), in press.
- U. S. Environmental Protection Agency.
1978. National Air Pollutant Emission Estimates, 1940-1976, Report No. EPA-450/1-78-003 Office of Air, Noise and Radiation, Research Triangle Park, N.C.
- U. S. Environmental Protection Agency.
1979a. National Air Pollutant Emission Estimates, 1970-1978, Monitoring and Data Analysis Division Report, EPA-450/4-80-002, Research Triangle Park, N.C.
- U. S. Environmental Protection Agency.
1979b. Protecting Visibility: An EPA Report to Congress. Report No. EPA-450/5-9-008 Office of Air, Noise, and Radiation, Research Triangle Park, N.C.
- U. S. Federal Register.
1979. Standards of Performance for New Stationary Sources: Electric Utility Steam Generating Units (June 11, 1979) 44:33580.
- Vukovich, T.
1979. A Note On Air Quality in High Pressure Systems. Atmos. Environ. 13: 255-265.
- Whelpdale, D. M.
1978. Large Scale Atmospheric Sulfur Studies. Atmos. Environ. 12: 661-670.
- Whelpdale, D. M., and J. N. Galloway.
1979. An Atmospheric Sulfur Budget for Eastern North America, Proc. of the WWO Symposium on Long Range Transport, October 1-5, Sofia, Bulgaria, World Meteorological Organization, Geneva, Switzerland.

Acid Precipitation Impact on Terrestrial and Aquatic Systems in Norway¹

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Abstract: In recent decades the acidity of rain and snow has increased sharply over wide areas. The principal cause is the release of sulphur and nitrogen oxides by the burning of fossil fuels. The air quality in any one European country is measurably affected by emissions in other European countries. Strong acids have lowered the annual mean pH of precipitation in much of northern Europe to between 4 and 5. In southern coastal areas of Norway, the annual mean acidity in precipitation is now 4.3 pH-units, or even more acidic.

Acid precipitation has increased leaching of nutrients from the uppermost soil layers. These losses of nutrients may be expected to decrease plant growth, but field evidence in Norway and elsewhere, has not yet been obtained. It is possible that polluted air and precipitation over a period of years can influence plant production.

Atmospheric transport of sulphur and other acidifying components has led to extensive regional acidification of water courses in areas with very little neutralization capacity. Acidification of watercourses had had major effects on life in rivers and lakes. Lakes in an area of 13,000 km² in southern Norway have become empty of fish in recent decades, and a further area of approx. 20,000 km² contains lakes with significantly reduced fish stocks.

The ecological impact of acid precipitation has been a matter of growing concern over the last decade particularly in the industrialized countries of the Northern Hemisphere.

The Norwegian Interdisciplinary Research Programme "Acid Precipitation - Effects on Forest and Fish", (The SNSF-project) was initiated in 1972.

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In Norway acid precipitation was at that time seen as a possible cause of increasing acidity of the watercourses in the southern part of the country, and of the gradual disappearance of valuable fish populations from many lakes and rivers. It was also feared that the inputs of acid might over time reduce forest growth particularly through increased leaching of nutrient elements from the soil.

The SNSF-project has this year marked the conclusion of eight years of research by organizing the International Conference on the Ecological Impact of Acid Precipitation. This Conference, which was held in Sandefjord, Norway, March 11 - 14, 1980, attracted more than 300 participants from some 20 countries.

The present report is mainly a summary and brief

discussion of the ecological impact of acid precipitation in Scandinavia with particular reference to Norway.

EMISSIONS, TRANSPORT, AND DEPOSITION

The concept of acid precipitation is now used to denote precipitation with high amounts not only of the hydronium ion (H^+ for short), but with enhanced concentrations of the acidifying anions sulphate, SO_4 and nitrate, NO_3 , which predominantly stem from anthropogenic sources. "Acid" precipitation regularly also contains high amounts of ammonium, NH_4 , and of various heavy metals and trace elements including organic micropollutants.

It should also be kept in mind that comparable amounts of anthropogenic pollution may be deposited by dry deposition processes from the same air masses giving acid precipitation. In studies of effects of acid precipitation the contribution from dry deposition is often difficult to quantify. In areas remote from the main industrial centres, like in Southern Norway, acid precipitation is more or less synonymous with long-range transported air pollution, but the distinction should always be observed. In our context, sulphur and nitrogen compounds giving rise to the acidifying properties of precipitation, are of prime interest, and their sources should be described with a view to finding their geographical distribution, emission rates and seasonal variation. This is necessary both for enabling modelling of their transport, and for formulating abatement policies against their negative effects.

Sulphur Emissions

Most of the man-made sulphur emissions occur as SO_2 from combustion of coal and petroleum products. Comparatively less stems from smelting of sulphur-containing mineral ores and other industrial processes, on a global basis about 10 percent. Knowledge of the location of industrial and powerproducing units in addition to population distribution has allowed quantification of annual emissions in a 150 km x 150 km grid over Europe (Semb, 1979). The main area of SO_2 emissions corresponds to the industrial belt from the Midlands in U.K., the Netherlands and Belgium, central and southern parts of Germany and Poland. Parts of northern France, Czechoslovakia and USSR also have industrial concentrations with very high emission rates. On a country basis, total SO_2 emissions in 1973 were estimated at e.g. $2.8 \cdot 10^6$ tonnes S from UK, 2.0 from the Federal Republic of Germany and 1.6 from France (OECD, 1977). Norwegian annual emissions were estimated at 91,000 tonnes of S.

Seasonal variations in SO_2 emissions occur as the demand for energy fluctuates through the year. At high latitudes the variable component of emis-

sions is related to space heating and illumination during winter. In warmer climates, the maximum demand may occur during summer due to the use of air conditioning. In most of Europe, the fuel consumption will probably be at its peak in January - February. The seasonal variation is about 30 percent of the annual mean emission.

Nitrogen Emissions

Both nitrogen oxides, NO_x , and ammonium, NH_4 , play important roles for the composition of acid precipitation. Although global budgets of nitrogen compounds are even more uncertain than those of sulphur, natural sources seem to be larger than the man-made. However, major man-made emissions also of nitrogen compounds occur in Europe and thus influence strongly precipitation chemistry in Scandinavia. The main anthropogenic source of nitrogen oxides is combustion of fossil fuels, by oxidation of nitrogen compounds in the fuel and oxidation of nitrogen in the combustion air.

It is well known that the NO_x emissions depend on fuel types, combustion chamber design and operating conditions. High combustion temperatures favour the emissions. There is, naturally, a high spatial correlation between SO_2 and NO_x emissions. OECD studies indicated almost a doubling of anthropogenic nitrogen oxides emissions in Europe from 1959 to 1973, thus increasing more than sulphur dioxide emissions in the same period (OECD, 1977).

Trace-Element Emissions

Acid precipitation contains a wide range of minor and trace elements associated with the major chemical constituents. Heavy metals, other trace elements and organic micropollutants of man-made origin are receiving increasing interest as some of these are enriched in living organisms.

Emission rates for trace elements and micropollutants are largely unknown, but the main sources are known. Many of the organic micropollutants in the atmosphere are products of human activity, including industrial and waste products, and also chemicals used in industry as solvents or intermediate products. The emissions are very complex mixtures of chemical compounds.

Transport and Deposition

Of particular interest for long-range pollutant transport is the build-up of high concentrations in stagnant air near the ground during inversion situations. Observations show that such parcels of contaminated air may subsequently move over long distances without much dilution. A much used technique in analyzing source areas and transport directions of air pollutants, is the sector analysis, grouping together trajectories belonging to the same sector.

Of considerable interest for the proportion of sulphate in acid precipitation to dry deposited sulphur components, is the oxidation rate from sulphur dioxide to sulphates. The oxidation takes place both by absorption of SO_2 in cloud droplets with subsequent oxidation, and by oxidation in the gas phase with oxygen compounds in the atmosphere. High concentrations of ozone and photochemical oxidants, which are observed over large areas of Europe, will increase the transformation rate. The transfer of gases and particles from the air to natural surfaces, and the adsorption, are usually described in analogy with the theory of electrical resistance. The transport from the atmosphere to the boundary layer close to the surface, takes place by turbulent diffusion. The aerosol particles in question, i.e., sulphur and nitrogen aerosols, are mostly in the 0.1 - 1.0 μm size range with low gravitational setting, and the turbulent transport will depend on meteorological conditions.

Model calculations of wet and dry deposition patterns over Europe show that in Scandinavia, particularly in Norway, the wet deposition outweighs the estimated dry deposition. For southern Norway dry deposition is estimated to account for about 30 percent of the total deposition of excess sulphate.

The precipitation in Norway is largely determined by polar front lows bringing moist maritime air in westerly to south-easterly directions. Of particular importance is the orographic enhancement of precipitation, caused by the lifting and subsequent cooling of the air masses when flowing across the Scandinavian mountain chain. This gives rise to a maximum zone of elevated precipitation some 40 - 50 km from the coastline. In this zone, annual mean precipitation exceeds 1000 mm along the SE coast increasing to an absolute maximum of perhaps more than 5000 mm in north western Norway. Still further north, in northern Norway, annual precipitation in the maximum zone exceeds 2000 mm.

The group of macrocomponents typical of acid precipitation, i.e. H^+ , NH_4^+ , SO_4^{2-} and NO_3^- , has a marked north-south gradient. The correlations between sulphate, nitrate and ammonium are high, and there are roughly equivalent amounts of the principal cations H^+ + NH_4^+ and anions SO_4^{2-} and NO_3^- . The content of strong mineral acid in precipitation is strongly correlated with excess sulphate. The correlation coefficient is 0.7 - 0.9 at Norwegian stations. At Norwegian stations NO_3^- makes about 30 percent of the sum SO_4^{2-} + NO_3^- . Ammonium and nitrate occur in Norwegian precipitation in about equivalent concentrations, lowest at mountain stations. The concentrations of SO_4^{2-} and H^+ in precipitation are highest along the south-east coast. The mountain plateau in northernmost Norway is affected by air transport from the south, which gives acid precipitation (pH 4.5 - 5.0) as far north as 70°N. lat. In southernmost Norway about 10 percent of the pre-

cipitation has a pH below 4.0 and about 5 percent above pH 5.0. Precipitation acidity below 3.5 has been observed several times at several places, Dovland and Semb, (1980).

Present knowledge of atmospheric deposition of inorganic trace elements in Norway is compiled by Semb (1978). They are found in the watersoluble fraction of aerosols, or absorbed on the other particles, and are mostly contained in the aerosol size fraction with aerodynamic mass diameter below 2 μm . Available data for lead, zinc and cadmium in precipitation shows a deposition pattern similar to that of excess sulphate. Also antimony, arsenic, selenium and vanadium seem to have similar deposition patterns. Influence from metallurgical industrial centres is evident for instance for chromium (western Norway) and arsenic, selenium, nickel, chromium and copper (smelting industry in the Murmansk area, U.S.S.R.). In northern Norway trajectory analysis shows that highly polluted episodes are often associated with the return flow pattern discussed by Rahn and McCaffrey (1980). Several studies of organic micropollutants in precipitation and in aerosols have been performed within the SNSF project. They have identified a wide range of compounds in the same air masses bringing acid precipitation to Norway.

ECOLOGICAL IMPACT

The effects of air pollution have historically been considered local problems, occurring near pollutants sources, usually urban areas. This concept of polluted cities versus clean rural areas is no longer applicable. The increase in anthropogenic emission sources coupled with the increased height of emissions have enhanced the phenomena of air pollution effects on rural areas. The most startling effects discovered so far of the long-range transmission of pollutants, have appeared in relatively remote, pristine areas of Norway, Sweden and the Eastern United States and Canada.

Despite the fact that sulphur dioxide emissions to a large extent contribute to acid precipitation the two pollutants show great differences in effects. Sulphur dioxide is a primary air pollutant as well as a primary toxicant. Acid precipitation on the other hand, is a secondary pollutant causing mainly indirect effects on ecosystems.

Forest Ecosystems

The effects of air pollutants on plants is extremely diversified; it depends upon species-linked tolerance or susceptibility, and is a function of many exposure parameters (frequency, time, concentration, etc.). Many types of response have been described on the basis of laboratory experiments, where known chemicals were tested under controlled conditions with different plant species.

However, the inverse operation - namely identifying and estimating the nature and importance of an existing source on the basis of response symptoms - is often difficult, except in the case of acute injury and when a pollution source is known or suspected in the vicinity.

Vegetation damage due to the emission of acid and poisonous substances has long been observed in the vicinity of emission sources. Visible symptoms have been described and are often associated with the decrease in growth. Recently, however, concern has been expressed that forest growth may also be affected far away from emission sources. Even if the direct evidence is meagre, Tamm, (1976), there is fairly substantial indirect evidence that continued exposure to acid rain has a growth-decreasing effect. The most significant indirect evidence is the positive correlation between forest yield and the soil base status. Jonsson and Sundberg (1972) classified areas in southern Sweden as relatively resistant to acid rain and relatively susceptible to acid rain, respectively, and compared the growth trends in both areas by measuring annual rings on increment cores from groups of trees which were otherwise as identical as possible. They found a statistically significant difference and "found no reason for attributing the reduction in growth to any cause other than acidification." These results however, have not been confirmed by Norwegian researchers, Abrahamsen and others, (1976); Abrahamsen, (1980), Strand, (1980).

When evaluating the effect of acid precipitation on the supply of plant nutrients in a forest, a basis could be to consider the nutrient cycle in a terrestrial ecosystem. Plant available nutrients are generally supplied to the system from two sources; from the atmosphere, as for N and S, and from the minerals, as for Ca, Mg, P, K, S, and the micronutrients. In natural systems not harvested by man, nutrients are also lost in two ways: To the atmosphere by volatilization and to the sea by leaching. Evaluation of the effect of acid precipitation on the amount of plant nutrients in a forest ecosystem can therefore be restricted to the consideration of four processes; deposition from the atmosphere, weathering, volatilization and leaching from the soil. Obviously many processes in the soil and the plants can affect the accessibility of plant nutrients. Acid rain may affect some of these processes.

Experimental studies on tree growth in relation to acid rain have been conducted in recent years in several countries. As yet no conclusive evidence of decreased growth has evolved. On the contrary, a slightly positive growth effect on the seedling, which was explained as a nitrogen fertilizer effect, was reported by Wood and Bormann, (1975). Such increases, though, are likely to be temporary, as depletion of nutrient cations through accelerated leaching

should eventually retard growth. Experiments on the effect of artificial acidification on forest growth under field conditions have been carried out in Sweden and Norway. The Swedish experiments have shown that increasing application of dilute H_2SO_4 has significantly increased the basal area growth, Tamm et al., (1980). The Norwegian studies consist of five field plot experiments where artificial rain has been produced by mixing groundwater and H_2SO_4 to pH values from 6 to 2. In one experiment with Scots pine, increased height and diameter growth was observed in 1976 and 1977 at the plots supplied with 250 mm of water per year of pH 3, 2.5 and 2. In 1979 however, the most acidified plots showed significantly less growth than the other experiments. (See Abrahamsen 1980).

The experiments thus show increased growth the first couple of years in the acidified plots, followed by decreased growth the last year. Similar patterns, though not significant, have been found in some of the other experiments. Chemical analyses of the foliage have revealed that the most likely explanation of the increased growth is increased N uptake. The decrease in growth observed in 1979 might be related to reduced availability of Mg as the foliar concentration is close to values giving visual deficiency symptoms.

Short-term growth results from acidification experiments must be treated with caution. They indicate, however, that tree growth may be reasonably stable when the plant-soil system is stressed by acid rain. Another difficulty to be kept in mind is that part of the acidity of rain is due to nitric acid or nitrogen oxides, which means that the positive fertilizer effect of nitrogen may partly or fully compensate for any harmful effects.

Theoretically there might be cases where acidity caused by sulphur oxides is counteracted by fertilizer effects, since sulphur is an indispensable plant nutrient. However, deficiency in sulphur has never been observed in forest trees under natural conditions in Scandinavia, and considering the rather tight nutrient circulation in the forest ecosystem, it is not likely to occur except possibly on very extreme sites.

A number of possible effects of acid rain on biological processes in the forest soil has been considered by Tamm, (1976), Abrahamsen, (1980). Most forest soils have a considerable buffer capacity. Therefore, we may assume that the supply of acidity, measured either as hydrogen ions or as "strong acid" of industrial origin, cannot yet have affected the entire soil profile, except possibly in the immediate vicinity of emission sources. Still, effects may be found on processes occurring in the top-soil or on the surface of soil particles. Soil organisms, including roots, in the upper soil horizons may also be affected. Soil respiration, nitrogen

turnover which is intimately connected to organic matter decomposition in soil, nitrification, nitrogen fixation and nitrogen immobilization are some of the processes apparently affected by increasing soil acidity. There are a number of other biological processes which may be affected by a change in soil acidity or sulphur supply which have not yet been studied.

Several comparative and experimental investigations have yielded evidence in support of the theoretical assumption that acidified precipitation, like any other change in climate, will result in changes in the properties of soil. Influences from this changed chemical climate on soil conditions have been indicated through decreases in pH and base saturation as well as increased leaching. From the studies performed up to now, however, it is difficult to draw any definite conclusions on the time required for the reactions and their intensities. Many soils are far from the stable and mature stage, and it is a well-known fact that considerable changes due to factors other than acid precipitation may simultaneously be affecting the properties of the soil. The great variation in soil types and in their susceptibility to acid precipitation make detection even more difficult before the expected effects have become extensive.

The relative significance of strong acids and associated heavy metals found in heavily polluted areas has not been clearly established in terms of toxic effects on plants and soil organisms. The most serious consequence of regional acidification at currently observed levels may be the increased rate of leaching of major elements and trace metals from forest soils and vegetation. This is true for the forest ecosystem and also has a bearing on the aquatic systems receiving these effluents.

Aquatic Ecosystems

Freshwater bodies in many areas of northern Europe and eastern North America, that today lie in and adjacent to the areas where precipitation is most acid, are threatened by the continued deposition and further expansion of acid precipitation. Many of these bodies of fresh water are poorly buffered and vulnerable to acid inputs. These ecosystems appear fated to suffer acidification and loss of fish populations. Equally as serious as damage to fish are the less conspicuous effects of the acidification of fresh water including changes occurring in communities of aquatic organisms such as microdecomposers, algae, aquatic macrophytes, zooplankton and zoobenthos.

Water chemistry

The composition of the lakes which are discussed in connection with acidification depends

on three principal sources of chemical components - atmospheric inputs of sea-water salts, atmospheric inputs of acid precipitation and terrestrial inputs of chemical-weathering products.

Unpolluted, soft water lakes are generally dilute solutions of Ca and Mg bicarbonate. The bicarbonate system constitutes the main buffering system in the water. Lakes in regions underlain by highly resistant, carbonate-poor rocks have lower buffer capacities, and are vulnerable to the input of acid precipitation. A major number of the lakes in Scandinavia fall within this category, especially above the postglacial marine limit, where the bedrock over large areas is covered by only thin glacial deposits. A continuous supply of acid substances to lakes and streams eventually leads to the depletion and loss of the normal buffer system. The pH falls to below 5.0, and sulphate becomes the major anion. Such lakes have only minimal capacity to neutralize additional inputs of acid; and new inputs of acid cause sharp drops in pH, Wright and Gjessing, (1976), Henriksen, (1980).

Acid precipitation also causes other changes in lake water chemistry as well. The acidic, high sulphate lakes also have high aluminum concentrations. Since precipitation contains very little Al, the Al in the lake water must come from the drainage basins. That has been shown to be the case in investigations conducted on 9 small drainage basins in southern Norway. It has been shown that loss of calcium, magnesium and aluminum from the basins is partly due to natural weathering processes, but a major fraction probably results from the massive inputs of acid precipitation. In granitic basins there is approximately equivalence between net H^+ input and $Ca + Mg + Al$ output.

The deposition of acid precipitation occurs episodically. Acid precipitation generally causes two seasons of increased acidity in streams and rivers - the fall, a season of frequent rain, and the spring, when pollutants stored in the snowpack are released in the first part of snowmelt. Laboratory and field studies of polluted snow have shown that the first fractions of meltwater due to concentration effects within the snowpack contain higher concentrations of pollutants than the bulk snow. The first 30 per cent of the meltwater contains up to 70 - 80 per cent of the total amount of H^+ , NO_3^- and SO_4^{2-} .

The episodic deposition of air pollutants and a certain temporary accumulation of sulphate in the summer results in major short-term increases in the acidity of lakes and rivers and these changes are most frequent in the fall and spring. From a biological point of view these periods are often critical because they are spawning and hatching seasons for many aquatic organisms.

Regional surveys

A large number of surveys have been conducted to give a picture of the acidification of Scandinavian lakes and rivers. A systematic survey of 155 lakes in southern Norway was conducted in October 1974, and repeated every year since, on varying number of lakes.

Excess sulphate is sulphate that does not come from sea water salts. Although in some lakes, this excess sulphate comes from a terrestrial source in the drainage basin, the regularity of a SE-NW gradient most probably is due to chronic inputs of anthropogenic sulphur through precipitation and dry deposition. Indeed, the distribution of excess sulphate in lakes is remarkably similar to the weighted average concentration of excess sulphate measured in precipitation over southern Norway.

The pH levels in the lakes can be largely explained by inputs of acid precipitation in the SE-NW gradient superimposed upon the variations in buffer capacities due to the geology of the drainage basin.

Aquatic organisms

Acid stress on life in rivers and lakes has effects on all stages in the food chain. Primary producers communities, like phytoplankton, are simplified by a reduction in the number of species. The composition of a community may alter with a shift to more acid-tolerant species.

Macrophytic vegetation has been observed to change in acidified lakes. In some lakes Sphagnum occurs in dense mats, and epiphytes are well developed. Generally, the vegetation in acidified lakes is poor in species.

The same tendency of an acid-tolerant shift is observed in diatom communities in 7 locations in southern Norway described in 1949 and revisited in 1975. There was an increase in the proportion of species which prefer or require acid water.

In some acidified lakes, an increased accumulation of organic bottom sediment has been observed, indicating a reduced rate of decomposition. There are strong suggestions that microbial decomposition is reduced in acid water and that slow-acting fungi take over. This will influence the nutrient exchange with the lake sediments.

In the invertebrate freshwater fauna the same trends are observed: a reduced number of species and a total reduction in biomass in the acid locations. Experiments on the tolerance of certain crustaceans which are important as fish food (Gammarus lacustris and Lepidurus arc-ticus) have shown a direct mortality of eggs at pH 5.5 or lower. There are also delays in the

development from stage to stage of surviving animals. Exposure below pH 5.5 will kill a majority of adult individuals even after a short time, 1 - 2 days. In Norwegian lakes, snails are rare at pH below 5.8 and disappear below 5.2.

In spite of the strong effects on fish food organisms, the indications are that changes in fish food supply play a small role in the elimination of fish from acid rivers and lakes. Instead, lack of recruitment seems to be the dominant factor. The tolerance against acid water is lowest in newly hatched larvae. This fact makes the spring flood a particularly critical time for the fish population.

The physiological mechanism or mechanisms responsible for fish death in acid water are not fully understood, but it has been well established that acid stress is accompanied by a failure in salt regulation within the fish body. Metabolism and osmotic cell regulation seem to be affected. It is also quite clear that the salt uptake and loss is influenced by the ion content of the water. Elevated concentrations of aluminum, manganese, zinc, cadmium, lead, copper, and nickel have frequently been observed in acidified lakes. The abnormally high concentrations are apparently due in part to direct deposition with precipitation as well as to increase release (solubility) from the sediments. These metals may represent a major physiological stress for various aquatic organisms.

Fish population statistics from nearly 1000 lakes in southern Norway show that when both pH is low (e.g. pH 4.7) and salinity is low (e.g. $\leq 10 \mu\text{S cm}$) almost all lakes are empty of fish. At higher salinity ($\leq 20 \mu\text{S/cm}$) several lakes have sparse populations and a few even good.

The recent acidification of freshwater in parts of Europe and eastern North-America has profound impacts on aquatic life. It can be stated with reliability that all trophic levels are affected. Of immediate concern to the people living in the acidified regions is the major decline in fish populations. In the four southernmost counties in Norway more than half of the fish populations have been lost during the 1940 - 1980 period. Today, lakes in more than 13,000 km^2 of south Norway are practically devoid of fish, and in an additional 20,000 km^2 the fish stocks are reduced. Continued water acidification is a threat to hundreds of lakes still harbouring valuable fish populations (Muniz and Leivestad, 1980).

LITERATURE CITED

- Abrahamsen, G.
1980. Acid precipitation, plant nutrients and forest growth. In Proc. International conference on the ecological impact of acid precipitation, Norway, March, 1980. SNSF, 1432 Aas-NLH, Norway (In press)
- Abrahamsen, G. , Bjor, K. , Horntvedt, R. and B. Tveite.
1976. Effects of acid precipitation on coniferous forest. In SNSF-project FR 6/76, SNSF, 1432 Aas-NLH, Norway
- Dovland, H. and A. Semb.
1980. Atmospheric transport of pollutants. In Proc. International conference on the ecological impact of acid precipitation, Norway, March, 1980. SNSF, 1432 Aas-NLH, Norway (In press)
- Henriksen, A.
1980. Acidification of freshwaters - a large scale titration. In Proc. International conference on the ecological impact of acid precipitation, Norway, March, 1980. SNSF, 1432 Aas-NLH, Norway (In press)
- Jonsson, B. and R. Sundberg.
1972. Has the acidification by atmospheric pollution caused a growth reduction in Swedish forests? A comparison between regions with different soil properties. In Rapp. Uppsatser Inst. för skogproduktion, Skoghögskolan, Nr. 20, 48 p.
- Muniz, I. and H. Leivestad.
1980. Acidification - effects on freshwater fish. In Proc. International conference on the ecological impact of acid precipitation, Norway, March, 1980. SNSF, 1432 Aas-NLH, Norway (In press)
- OECD
1977. The OECD Programme on Long Range Transport of Air Pollutants. OECD, 2 rue André Pascal, Paris
- Rahn, K.A. and R.J. McCaffrey.
1980. On the origin and transport of the winter Arctic aerosol. In Proceedings of the conference on Aerosols. New York Academy of Sciences, New York City, 9-12 January 1979 (In press)
- Semb, A.
1978. Deposition of trace elements from the atmosphere in Norway. SNSF-project FR 13/78, SNSF, 1432 Aas-NLH, Norway
- Semb, A.
1979. Sulphur emissions in Europe. Atm. Env. 12 p. 455-460.
- Strand, L.
1980. The effect of acid precipitation on the growth. In Proc. International conference on the ecological impact of acid precipitation, Norway, March, 1980. SNSF, 1432 Aas-NLH, Norway (In press)
- Tamm, C. O.
1976. Acid precipitation: Biological Effects in soil and on forest vegetation. Ambio, 5,6: 235-238.
- Tamm, C. O., Farell, E. P. , Nilsson, J. , and G. Wicklander.
1980. Effects of artificial acidification with sulphuric acid on tree growth and soil chemistry in Scots Pine forest. In Proc. International conference on ecological impact of acid precipitation, Norway, March, 1980. SNSF, 1432 Aas-NLH, Norway (In press)
- Wood, T. and F. H. Bormann.
1975. Increases in foliar leaching caused by acidification of an artificial mist. Ambio, 4: 169-171.
- Wright, R. F. and E. T. Gjessing.
1976. Acid precipitation: Changes in the chemical composition of lakes. Ambio, 5: 219.
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- SNSF-contribution FA 115/80

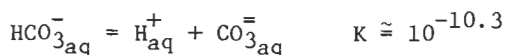
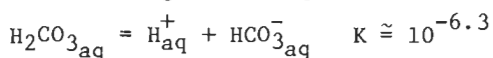
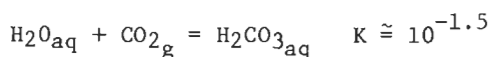
The Impact of Acidic Precipitation and Heavy Metals on Soils in Relation to Forest Ecosystems¹

Stephen A. Norton, Denis W. Hanson, and Richard J. Campana²

Abstract: Normal terrestrial cycling of metals in eastern North America and the Pacific Coast states has been altered by the increasing acidity of precipitation, and associated heavy metal deposition and mobilization. Pb and chemically similar metals are accumulating in soils. Al, Ca, K, Mg, and Mn are being leached from soils. The mobilities of Fe, Zn, and P vary with site characteristics. Biological recycling of nutrients by decomposition and uptake is impeded by lowered pH and elevated levels of toxic metals in soils. Increased leaching of nutrients in the O and A horizons, caused by increased H⁺ inputs, decreases percent base saturation and thus decreases nutrient pools for shallow rooted plants, especially seedlings. Deeper rooted plants are subjected to elevated, potentially toxic, concentrations of dissolved metals (e.g., Al and Mn).

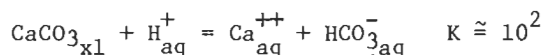
In many contemporary forest ecosystems, nutrient availability is barely adequate for sustained yield with bole harvesting techniques. Our work indicates that nutrient pools are diminishing in the northeastern United States, suggesting that decreases in forest productivity will occur.

Large areas of the northern hemisphere are receiving precipitation which is more acidic than would be predicted by equilibration of rain and atmospheric CO₂:



The resulting pH should be about 5.6. This may be modified by the hydrolysis of particulates or the addition of naturally occurring organic and inorganic acids. Precipitation pH's in the U.S., prior to pollution of the atmosphere, probably

ranged up to about 8.2 (corresponding to semi-arid to arid regions where CaCO₃ dust, or its equivalent, dominates the rain chemistry).



Atmospheric concentrations of CO₂ should result in a pH of about 8.2. In eastern North America, where vegetation cover minimizes particulate injection into the atmosphere, unpolluted precipitation would probably have a pH of about 5.6. Associated with the precipitation are numerous metals and plant nutrients (e.g., Na, K, Ca, Mg, NO₃⁻, NH₄⁺, H₂PO₄⁻, Pb, Zn, etc.).

Precipitation (wet and dry) is one of three inputs into the nutrient budgets for forest ecosystems. The others are chemical weathering of inorganic soil and nutrient cycling within the canopy/root space. Both are closely linked to precipitation chemistry.

Simply put, from a nutritional point of view:

Input - Output = Net accumulation of organic material.

Output consists of leaching to groundwater (below

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the root zone), injection of particulates into the atmosphere, volatilization of certain elements, and loss of particulates "downstream", including harvesting. If the mass balance (above) is positive for all limiting nutrients, growth and accumulation of organic matter will occur as living biomass or as soil organic matter. If the mass balance is negative, growth can occur only so long as the decreasing reservoirs of organic matter and nutrients from precipitation and chemical weathering can supply necessary nutrients; organic matter must decrease. The amount of organic material on the forest floor and contained in the rooting zone of the mineral soil is generally of the same order of magnitude or larger than the organic material contained in the above ground biomass. Removal of this biomass disrupts the recycling of nutrients. Continued growth of new biomass must occur from the reservoir within the soil, forest floor, and from precipitation.

A forest ecosystem can subsist on nutrients delivered solely by precipitation, both wet and dry (Art et al. 1974). However, it is doubtful that this precipitation-based forest ecosystem could be harvested periodically and still have sustained growth. If there are changes in the chemistry of precipitation, specific inputs/outputs of the forest nutrient budget may be altered so as to affect both sustained growth yields and net organic matter accumulation.

Many forest ecosystems are not forced to subsist on atmospheric inputs alone. They receive additional primary inputs of nutrients from chemical weathering of mineral matter. However, in unglaciated areas of the eastern U.S. where chemical weathering dominates over mechanical weathering or in glaciated granitic (nutrient poor) terrains characteristic of large areas of eastern North America, nutrient pools are largely contained within the organic litter of the forest floor. Thus, these areas are the most vulnerable to depletion of nutrient pools due to acidic precipitation.

Nutrient availability is commonly classified as deficient (where addition of the limiting nutrient elicits a positive response), adequate (where addition of a nutrient does not elicit a response), and excessive (where addition of the nutrient elicits a negative response). This paper investigates some of the consequences of increased acidity and metal availability, associated with acidic precipitation to nutrient availability (and thus to forest productivity). Excellent reviews of potential problems are given by Voigt (1979) and in Hutchinson and Havas (1980).

EFFECTS OF DECREASED pH

Although early precipitation chemistry is sparse for North America, Cogbill and Likens (1974) constructed 3 pH isopleth maps spanning 1952 to 1972 which suggested increasing acidity for precipitation in the eastern U.S. and a broadening of the

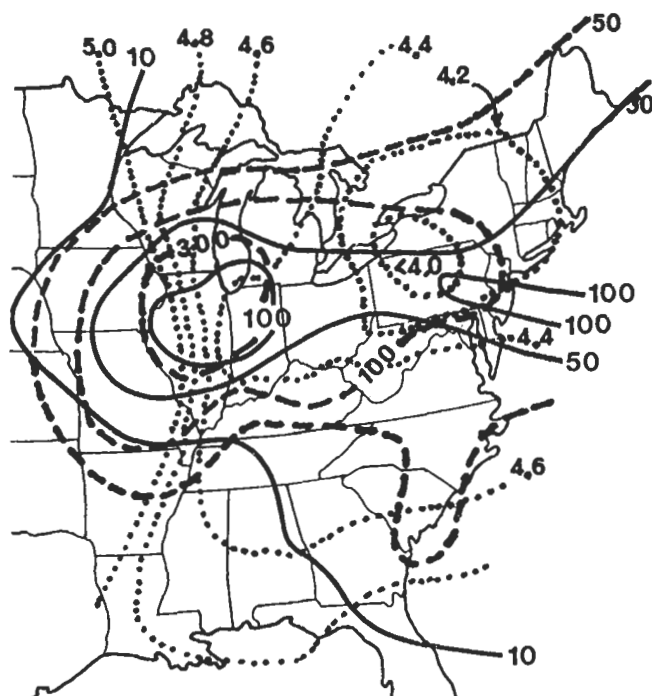


Figure 1--Generalized pH (dotted) (NADP 1979, 1980; CANSAP 1979), Pb (solid) and Zn (dashed) isopleths (Davis and Galloway 1980) for eastern U.S. precipitation. Fluxes are g/h/mo.

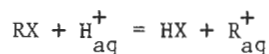
area receiving acidified precipitation. Recent data (NADP 1979; NADP 1980 a,b; CANSAP 1980) shows a continued decrease in the pH to average values of less than 4.0 for the "bull's eye"; the eastern half of the U.S. and the eastern half of southern Canada is receiving precipitation with a pH less than 5.0 (fig. 1).

Most of the lowering of the pH of precipitation from "normal" to present levels occurred over the last 40 years. In North America, there are no long term regional studies of the acidification of soils showing trends related to regional pH gradients. Linzon and Temple (1980) (in Ontario) found acidification of soils over a 16 year observation period. Many studies have been made of soil acidification and nutrient status in areas heavily affected by point sources.

Expected changes in the inorganic aspects of soil chemistry associated with acidification include desorption of metals from organic and inorganic cation exchange surfaces, increased solution of "mineral" colloids and crystalline minerals, and changes in metal speciation and thus biological availability.

Desorption of metals

Regardless of the nature of the substrate, ion exchange (for a monovalent ion) may be represented as



where X is an exchange site, R^+ is any monovalent metal, H^+ is a proton, and aq refers to an aqueous specie. This reaction can be forced in either direction by changing H^+ or R^+ activities.

Pre-pollution precipitation in eastern North America entered the soil with a pH probably in the range 5.5 to 6.0. Microbial activity in the organic litter produces molecular CO_2 which can reduce the pH considerably (as low as 4.5 to 5.0). The production of various organic weak acids (e.g. fulvic and humic) may depress the pH further to 4.0 to 4.5. To maintain electrical neutrality in these solutions to offset the H^+ production, either anions must be gained (HCO_3^- activity is reduced by the lowering of pH; organic anions may be produced) or cations must be lost from the solution. This is most effectively accomplished in the litter and at root surfaces where H^+ is exchanged for percolating cations. Thus nutrients are gained by the soil.

If the precipitation has a pH of 4.0 due to the strong acids H_2SO_4 and HNO_3 , the exchange reaction is forced strongly to the right, stripping cations (particularly Ca and Mg) from the litter, reducing percent of base saturation. Continued production of CO_2 , organic acids, by microbial activity assures that the cations are lost from the system.

No long term studies of soils exist to demonstrate the switch from accumulation to loss of cationic nutrients as a result of low pH precipitation. Indirect evidence for the switch consists of long term changes in surface water quality such as conductivity (Malmer 1976 [in Sweden]), alkali and alkali earth concentrations (Malmer 1976), and the commonly observed relationship between non-dystrophic low pH waters and elevated Ca, Mg, Al, Mn, and other metals. Ulrich (1980) and Linzon and Temple (1980) have shown loss of base saturation in soils over 8 and 16 years, respectively. Abundant experimental evidence (e.g., Hutchinson 1980; Abrahamsen and Stuanes 1980) and studies of soils adjacent to large point source emitters of SO_x and NO_x suggest what long term results might look like. Ca, Mg, K, Zn, Cd, and Mn are readily leached from litter. However, the pH levels employed for experimental work are commonly well below what we might expect on a regional basis. Processes operating at a pH < 4.0 may not be effective even over long periods of time at pH > 5.0.

A transect of "equivalent" soil sites across a regional pH gradient (fig. 2), in effect, is a time study of the effects of low pH precipitation. Table 1 indicates a progressive decrease in

$\frac{CaO}{Al_2O_3}$ and $\frac{MnO}{Al_2O_3}$ in a southwesterly direction, toward lower pH precipitation. We interpret this as a preferential leaching (desorption) of Ca and Mn from the litter.

Organic matter in lake sediment is derived from both the watershed and the lake water column. Organic-rich lake sediments in acidified lakes in New England (Williams 1980) are depleted of CaO,

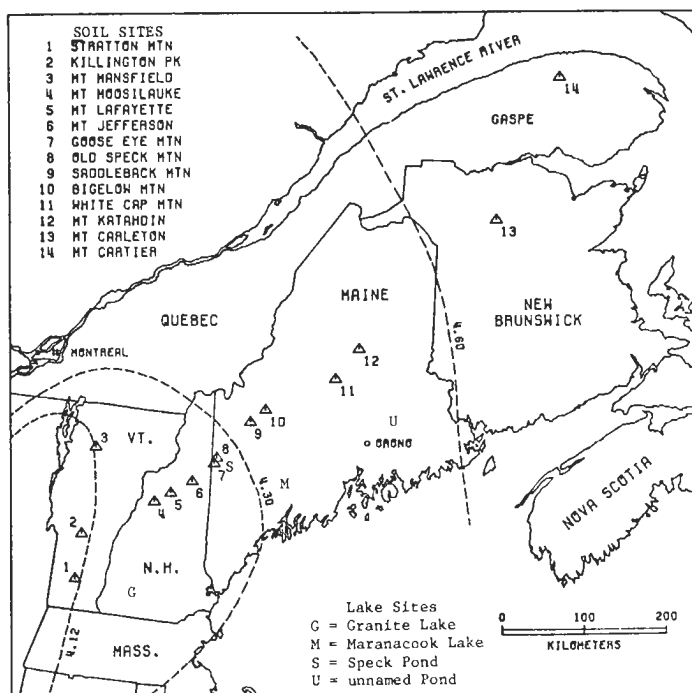


Figure 2--Location of soil localities reported in Table 1 (Hanson 1980) and location of lakes for which Figure 4 is developed. pH isopleths are for 1975/6 (Likens et al. 1979).

also suggesting that forest litter is being leached before it is transported to the lake and/or that leaching continues while the sediment is in contact with acidic lake water.

Solution of Metals from Minerals or Colloids

The release of cations from minerals due to acidic soil water weathering may be represented

Table 1--Chemistry of forest litter from high altitude fir forests. Sample sites are shown on figure 2. Note: Site 6 is anomalous for all parameters. Site 12 had abundant admixed mineral soil and the bedrock is very low in MnO. Details of collection and analysis are in Hanson (1980).

Site	CaO	MnO	Pb(ppm)	Zn(ppm)
	Al_2O_3	Al_2O_3		
1	7.62	0.21	189	72
2	4.08	0.09	204	59
3	5.44	0.16	187	61
4	7.43	0.25	171	77
5	5.36	0.34	178	84
6	11.87	0.63	85	81
7	3.91	0.32	129	78
8	9.21	0.63	136	83
9	12.36	0.45	137	86
10	10.22	0.45	112	78
11	20.71	1.00	140	86
12	10.22	0.32	86	81
13	15.29	1.06	118	94
14	18.68	0.99	63	68

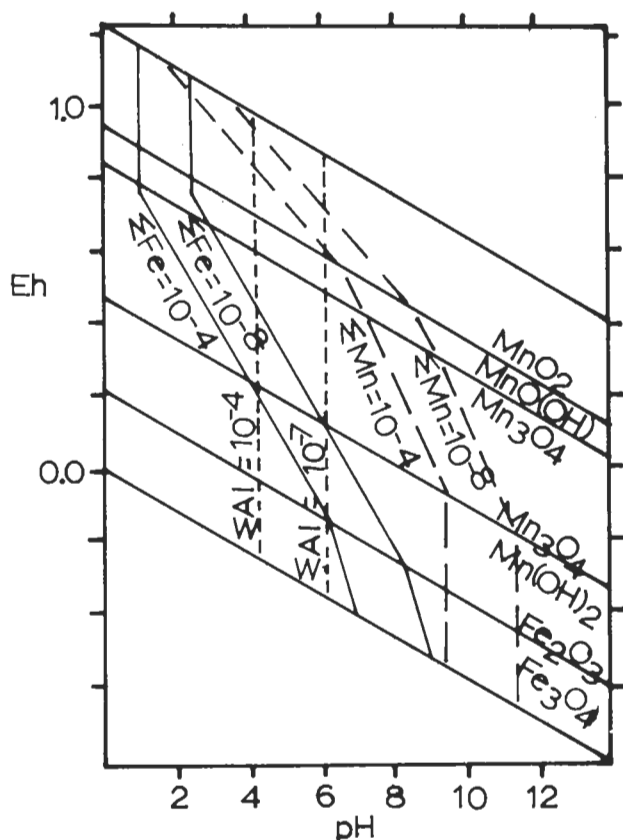
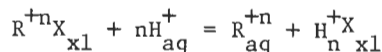


Figure 3--Eh-pH diagram for the system Al-Fe-Mn-O-H₂O at 25°C and 1 atmosphere. Solubility in moles/l.

by



where R is a metal, n is the valence of the metal, X_{xl} is the stoichiometric formula for the mineral (minus R), H⁺ is a proton, and aq refers to aqueous species. Reactions of this type are forced by elevated H⁺ levels. Mn⁺²-, Fe⁺²-, Fe⁺³-, and Al⁺³-bearing minerals are somewhat unique in that the respective metal's mobility is a function of [H⁺]² or even [H⁺]³ activity and the transition from geochemical immobility to mobility under oxidizing conditions takes place at a pH between 4 and 6 (fig. 3).

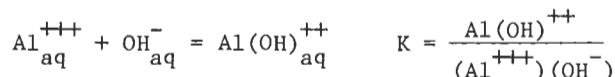
Increased mobilization of Al from inorganic soils has been documented by Cronan and Schofield (1979). Both experimental and field studies (Table 1) suggest that Mn is leached from upper levels of soils. Data are sparse but indicate that acidic non-dystrophic surface waters have elevated Mn levels. Al and Mn have toxic effects on plants via root effects, and on aquatic animals. Fe, although leached somewhat by acidic percolating solutions (Cronan and Schofield 1979), is relatively enriched in the litter in leaching experiments and in the field (Hanson 1980). Downstream reduction in concentration of metals may be accomplished by dilution (Ca, Mg, K), precipitation (Fe, Al, Mn), and adsorption (P, Zn).

Overall, acidic precipitation accelerates podsolization, expanding depths of upper soil horizons, and depleting the nutrient pool in the upper soil.

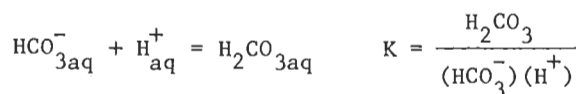
Changes in Speciation of Dissolved Metals

Biological uptake of nutrients and toxicants is via molecular diffusion through semi-permeable membranes which are somewhat specific to the species involved. Biological response is highly specific to the dissolved form ("biological availability") of the element.

Decreasing pH affects speciation in two ways. Elements which complex with OH groups will be preferentially partitioned into a less hydroxylated form. For example:



Lowered pH also results in protonation of weak acid radicals (e.g. HCO₃⁻, fulvic⁻, humic⁻) causing a decrease in ligands for the cation of interest. For example:



Because toxicity is generally greater for the uncomplexed metal (Hg is an exception), acidification of soil waters should result in greater direct toxicity to roots and micro-organisms or foliage after uptake, and on "downstream" ecosystems.

INCREASING TRACE METAL MOBILITY/AVAILABILITY

Trace metal levels, particularly heavy metals, are intimately affected by acidic precipitation in that their flux to the forest ecosystem is greatly increased over pre-pollution values and their mobility is in some cases greatly altered.

Historic data to evaluate the changing atmospheric flux of heavy metals in North America is absent. Changes in fluxes with time in remote regions have been evaluated from snow/ice cores (Cragin et al. 1975). Anthropogenic emission rates may be compared with natural emission rates to obtain an estimated percentage increase or mobilization factor. For Cd, Mn, Pb, and Zn the factors are 13, 0.48, 180, and 13 respectively. However, these factors may not relate closely to deposition values (Galloway et al. 1980), because of spatially non-homogeneous emission, dispersion, and deposition.

Limits on pre-pollution metal concentrations in precipitation may be established using modern data for precipitation chemistry from modern remote sites. Metal levels for eastern North America must have been between the remote (Antarctica)

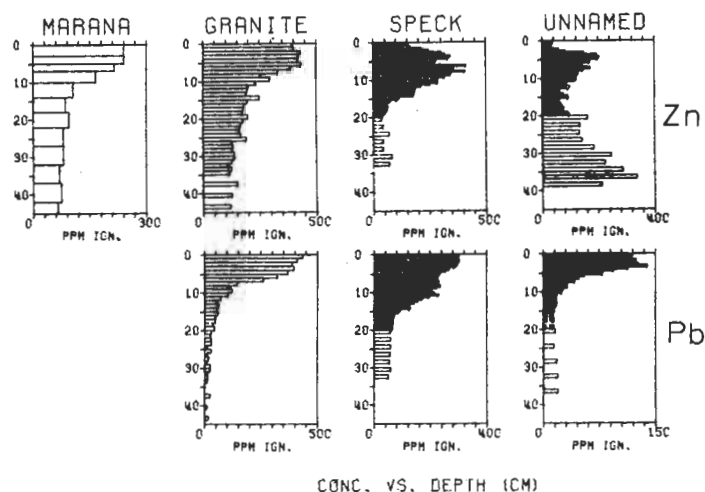


Figure 4--Pb and Zn profiles for sediment from a circum-neutral (Maranacook), slightly acidic (pH, 5-6) (Granite), and acidic (pH<5) (Speck) lake with surface inlets and outlets and an acidic (pH <5) kettle pond ("unnamed" Pond). Ponds are located on Figure 2.

and North Atlantic values. (We know that Greenland [Cragin et al. 1975] has been receiving polluted precipitation for approximately 200 years.) Modern deposition rates for Pb and Zn are roughly known (fig. 1). However, our poor knowledge of pre-pollution deposition rates doesn't permit a good assessment of the increases that have occurred for eastern North America.

Lake sediments record changes in atmospheric deposition but normally not in a straight forward manner because of watershed effects, sediment focusing, and diagenesis (Norton et al. 1980). Figure 4 (unnamed Pond profile) suggests a minimum of an 800% and 200% increase for the deposition rate for Pb and Zn, respectively, over the last 100 years. These figures assume that the background levels (below 15-30 cm, depending on the lake) are due to atmospheric deposition. However, most of the background concentrations are probably bedrock contributions. Based on increases for Pb and Zn concentrations in sediments from remote lakes in New England, we estimate that atmospheric deposition rates in New England have increased at least by a factor of 30X. Pre-pollution concentrations of heavy metals in soils are unknown. Short term studies (Siccama et al. 1980) of heavy metals in soils indicate that concentrations are increasing with time. Our data (fig. 2, Table 1) shows a strong relationship between the pH gradient and Pb accumulation, consistent with the Pb deposition gradient (fig. 1). These concentrations are of concern with respect to toxic effects for soil microbial activity and nutrient cycling. High Pb concentrations in litter are also considered as possible controls on the biological availability of phosphorus in soils (Cox and Raisons 1972).

However, even with elevated deposition rates it appears that Zn (and other elements with similar chemical behavior) is not accumulating in for-

est litter (Table 1) although biologically available B-horizon accumulation occurs (Conrad, pers. comm.). This appears to be related to increased leaching of Zn from litter due to decreased pH of precipitation; lake sediments from New England exhibit similar behavior (fig. 4). Acidified watersheds have lake sediments being deposited which have less Zn than sediments deposited prior to their acidification.

SUMMARY

Acidic precipitation, experimentally and empirically, accelerates podsolization and depletes nutrient pools in forest litter and shallow inorganic soils. Some heavy metals (e.g. Pb) accumulate to concentrations which may impede biological soil processes and immobilize phosphorus. Other metals are mobilized (Al, Mn, Zn) and may affect ecosystems "downstream". Paleolimnological and soils data from New England indicate that acidification of drainage basins (including soils) has occurred and nutrient depletion is underway.

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LITERATURE CITED

- Abrahamsen, Gunnar, and A.O. Stuanes.
1980. Effects of simulated rain on the effluent from lysimeters with acid, shallow soil, rich in organic matter (abs.). Int. Conf. on The Ecological Impact of Acid Precipitation, The SNSF Project. 1:27.
- Art, Henry W., F.H. Bormann, G.K. Voigt, and G.M. Woodwell.
1974. Barrier island forest ecosystem: Role of meteorologic nutrient inputs. Science. 184: 60-62.
- Bunzl, K.
1974. Kinetics of ion exchange in soil organic matter. II Ion exchange during continuous addition of Pb²⁺-ions to humic acid and peat. Jour. Soil. Sc. 25:(3).
- Cox, W.J., and D.W. Raisons.
1972. Effect of lime and lead uptake by five plant species. Jour. Envir. Qual. 1:167-169.
- Cragin, J.H., M.M. Herron, and C.C. Langway, Jr.
1975. The chemistry of 700 years of precipitation at Dye 3, Greenland: CREEL Research Report 341, 18 p.
- Cronan, Christopher S., and C.S. Schofield.
1979. Aluminum leaching response to acid precipitation: Effects on high-elevation watersheds in the northeast. Science. 204:304-305.
- Davis, Anthony O., and J.N. Galloway.
1980. Atmospheric trace metal deposition into lakes of the eastern United States. In Input of Atmospheric Pollutants to Natural Waters. Steven J. Eisenreich, ed. Ann Arbor Science Publishers, Michigan.
- Environment Canada.
1979. CANSAP data summary. 15 p.
- Galloway, James N., H.L. Volchok, D. Thornton, S. A. Norton, and R. McLean.
1980. Trace metals: a review and assessment. In Toxic substances in atmospheric deposition. James N. Galloway, S.J. Eisenreich, and B. Scott, eds. National Atmospheric Deposition Program, Fort Collins, Col.
- Hanson, Denis W.
1980. Acidic precipitation-induced changes in sub-alpine fir forest organic soil layers. M. S. unpublished thesis, University of Maine.
- Hutchinson, Thomas C.
1980. Effects of acid leaching on cation loss from soils. In Effects of acid precipitation on terrestrial ecosystems. T.C. Hutchinson and M. Havas, eds. p. 481-497. Plenum Press, New York.
- Hutchinson, Thomas C., and M. Havas, eds.
1980. Effects of Acid Precipitation on Terrestrial Ecosystems. 654 p. Plenum Publishers, New York.
- Lazrus, A.L., E. Lorange, and J.P. Lodge, Jr.
1970. Lead and other metal ions in United States precipitation. Envir. Sci. Tech. 4:55-58.
- Likens, Gene E., R.F. Wright, J.N. Galloway, and T.J. Butler.
1979. Acid Rain. Scient. Amer. 241:43-51.
- Linzon, S.N., and P.J. Temple.
1980. Soil resampling and pH measurements after an 18-year period in Ontario (abs). Int. Conf. on The Ecological Impact of Acid Precipitation, The SNSF Project. 1:42.
- Malmer, Nils.
1976. Acid precipitation: chemical changes in the soil. Ambio. 5:231-234.
- National Atmospheric Deposition Program.
1979. NADP first data report, July 1978 through February 1979. Fort Collins, Col. Natural Resource Ecol. Lab., Col. State University.
- National Atmospheric Deposition Program.
1980a,b. NADP Data Report. Fort Collins, Col. Natural Resource Ecol. Lab., Col. State University, I and II:(1)&(2).
- Norton, Stephen A., C.T. Hess, and R.B. Davis.
1980. Rates of accumulation of heavy metals in pre- and post-European sediments in New England lakes. In Input of Atmospheric Pollutants to Natural Waters. Steven J. Eisenreich, ed. Ann Arbor Science Publishers, Michigan.
- Siccama, T.G., W.H. Smith, and D.L. Mader.
1980. Changes in lead, zinc, copper, dry weight, and organic matter content of the forest floor of white pine stands in central Massachusetts over 16 years. Envir. Sci. Tech. 14:54-56.
- Ulrich, B.
1980. Deposition, production and consumption of hydrogen ions in a beech and spruce ecosystem in the Solling District (abs.). Int. Conf. on the Ecological Impact of Acid Precipitation, The SNSF Project. 1:50.
- Voigt, Garth K.
1979. Acid precipitation forest ecosystems and intensive harvesting. In Impact of Intensive Harvesting on Forest Nutrient Cycling. p. 33-48. College of Environmental Science and Forestry, School of Forestry, Syracuse, N.Y.
- Williams, John S.
1980. The relative contributions of local and regional atmospheric pollutants to lake sediments in northern New England. M.S. unpublished thesis, University of Maine at Orono.

Impact of Heavy Metals on Terrestrial and Aquatic Ecosystems¹

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Abstract: The high toxicity of many metals and metalloids to a wide range of biota, coupled with their long residence times in the soils, in sediments and in the oceans has led to real concern about their role in environmental deterioration. Residence times in watersheds are commonly measured in hundreds of years, while residence of metals in air is rarely as long as several days. Overall industrial activity and transportation leads to widespread metal dispersion. Major elevations in many metals occur around mines and smelters and for lead especially, alongside highways. Coal-burning and applications of fertilisers and pesticides add metals to agricultural soils and to natural ecosystems. The surface organic layers of both soils and sediments act as adsorption and exchange sites so that major accumulations may occur. Yet, this shallow organic layer is the critical site for many microbial activities, including those essential for nutrient cycling, nitrogen fixation and for pathogens. Genetic and physiological tolerances are shown in a wide array of different organisms which have survived in metal-stressed habitats. Most recently, acid precipitation has mobilised Al, Mn, Fe and Zn from the soil and sediment. These are now producing particular stresses for aquatic biota.

The considerable toxicity of many metals to biota is well known e.g. lead, mercury, cadmium, arsenic. The quantities mined and smelted for innumerable uses continues to rise each year. It is apparent that both natural and man-made eco-

systems are the recipients of increasing quantities of these elements and that we have to be continuously vigilant to ensure that we do not either poison ourselves, our fellow biota, or our agricultural and natural ecosystems. Major human poisonings have occurred such as that by mercury in Japan and in Iraq, and that of cadmium in Japan. The beer-deaths in Birmingham, England at the turn of the century were also believed to be metal-related, being variously ascribed to arsenic and/or selenium. Concern has also been expressed about not only the potential for food chain contamination leading to man but also to direct aerial inputs to man via the respiratory tract. Lead from automobile emissions where it is used as an anti-knock in gasoline, and from primary and secondary smelters, as well as many other smelter-

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emitted metals such as arsenic, copper, nickel, zinc, cadmium, antimony and selenium, all have potentially harmful consequences due to their persistence in the body, and their ability to interfere with specific enzyme systems. It should also be noted that many instances are known of synergistic and antagonistic interactions between metals, both in aquatic and terrestrial systems and in the human body. Notable amongst these are the ameliorative effects of selenium and arsenic on both mercury and cadmium toxicity in mammals and the recently described synergistic effects of ozone on cadmium and nickel toxicity in certain crop plants, e.g. Parizek 1978, Levander 1977, Groth and others 1973, Czuba and Ormrod 1974. Nickel and copper synergisms have been described for a variety of biota, including freshwater unicellular algae, floating aquatic plants, and tree seedlings, Hutchinson 1973, Hutchinson and Stokes 1975, Hutchinson and Czyrska 1972, and Hutchinson and Whitby 1974.

Despite what sometimes seems to be a gloomy picture with respect to metal accumulations in the environment, it ought to be borne in mind that concentration of airborne particulates in urban and industrial areas of Europe and North America have often been much worse in the past. Cohen and Ruston (1925) reported very high arsenic levels in the air of Leeds, England in 1902-1910 due to coal burning, while the overall levels of SO_2 and acidic aerosols were much higher than presently occur.

Residence Times and Watershed Loss

Rather little attention has been paid to the residence times of metals in components of the biosphere. The strong retention of metals on the organic surface layers of the soil and of the sediments is of great importance, in that it causes long residence times as well as in allowing accumulation of metals to potentially toxic concentrations. Since it is precisely in these surface zones that the major populations of microbes are located, and where the essential processes take place of decomposition of organic matter, of nitrogen fixation and of elemental cycling etc. Thus, the persistence and accumulations of toxic elements is of real concern. Litter in urban areas contains elevated lead concentrations.

The retention of the metals themselves is an exchange process, with the elements behaving in a reasonably predictive way, based on such properties as ionic radius and electronegativity. Acid leaching can cause a downward movement of heavy metals through the profile, so that they may enter ground water or watershed streams. (e.g. Abrahamsen, Stuanes and Bjor 1979, Cronan and Schofield 1979, Hutchinson 1980, Bacon and Maas 1979.) Both rock surfaces and soils can contribute and such clay mineral constituents as

aluminium, manganese, zinc and ferric iron enter drainage water in this way. Sediments can similarly lose these same elements to the water bodies of lakes under acidifying conditions (Schindler and others 1980). The damaging effect of the resultant aluminium concentrations to fish have been described by various authors, including Schofield 1976, Baker and Schofield 1980, at levels as low as 0.1-0.2 mg/l. Increased transport of aluminium into aquatic systems can also affect phosphorus availability (Cronan and Schofield 1979).

The residence time of metals in the air is always very much shorter than that in soil, water, sediments or oceans. This is illustrated by Table 1, which emphasises the rather larger residence time of lead in air than that of a wide range of other metals. This partly explains the elevations of lead noted at remote locations, such as in the arctic, in glacial ice in Greenland and at mountain tops in California (see National Academy of Sciences Lead Review 1980). It is also a function of particulate size and partial vapour pressure.

Table 1. Residence times¹ of metals in the atmosphere at La Jolla and Ensenada, from Hodge, Johnson and Goldberg, 1978.

	days	
	La Jolla	Ensenada
Pb	7	8
Cd	0.7	0.5
Ag	0.2	0.1
Zn	0.4	0.3
Cu	0.5	1
Ni	3	0.8
Co	1.2	0.2
Fe	1.0	0.4
Mn	0.8	0.2
Cr	0.8	0.4
V		0.6
Al	1.0	0.2
Pb 210	5	---
Pb 239 + 240	1	---

¹Standing crop of metals on particulates in 1,000 m X 1 cm² column of air (filter data--Table 1) divided by the flux to 1 cm² of ground surface (bucket data --Table 2). Filter data averaged over period during which buckets exposed. Filter concentrations of Co, Fe, Mn, Cr and Al have been multiplied by 2 in order to account for the discrimination against large particles by the Hi-Vol sampler.

The contrasting data for watershed soils are illustrated by Table 2, taken from Bowen 1975, and from which it is apparent that soil-watershed residence times are measured in hundreds of years.

Table 2. Inputs and outputs in $\text{mg X/m}^2 \text{ yr}$, and residence times in years, for nine elements in soils of the Upper Thames basin.

X	Rain input	Fertilizer input	Rock input	Drainage output	Cropping output	Residence time/ years
As	1.6	0.3	0.014	0.04	0.2	2000
Cd	0.5	0.6	0.0004	0.5	0.2	280
Cr	3.8	3	0.06	0.02	0.08	6300
Cu	11	0.8	0.04	1	5	860
Hg	0.08	0.002	0.0004	0.008	0.005	920
Ni	5	0.9	0.1	1	1	2300
Pb	37	0.8	0.13	0.4	1	400 - 3000
Sc	0.44	0.03	0.003	<2	0.07	<2500
Zn	43	5	0.12	1	58	2100

Clearly the potential for accumulation to toxic levels is much greater. The excess of lead and chromium in input over output is a feature of systems subjected to industrial deposition. The volatilization of some of these heavy metals from the foliage of vegetation e.g. zinc, mercury, and selenium (Beauford and others 1975) may re-mobilise small quantities of these metals and increase atmospheric residence times but it will not influence loss into drainage waters. Allen and Steinnes (1979) determined the regional distribution of lead, zinc, cadmium, copper, arsenic, antimony and selenium in Norwegian surface soils, utilising 500 humus samples. Lead levels were 10-fold higher in the south than in the arctic areas and also higher along the coast than inland. Cadmium, arsenic, antimony and selenium showed a similar north-south trend. All of these elements are highly volatile, low boiling point components of the atmospheric load from industrial and urban centres.

Residence times in soil water are affected to a great extent by pore size. The water in the large pores infiltrates into lower layers and air enters again behind it. The residence time in these pores is not more than hours. Meanwhile, the water in the narrow pores is displaced only centimetres or millimetres. An example is shown in Table 3 from the work of Frissel (1978), with residence times as high as 5000 years. The implications for ground water contamination are apparent.

Targets for H^+ and Heavy Metals

I should like to emphasize that one very useful way of considering the potential threats to ecosystems is through a consideration of targets in the ecosystem. Clearly, all surface interfaces fall into this category. Surfaces present areas of potential accumulation or residence. In terrestrial ecosystems, including

agricultural ones, the surface of the leaf is one such interface. Higher plants are covered by a rather impermeable waxy cuticle, which reduces gas and water flow to a minimum but is perforated by numerous stomata, often on the underside of the leaf especially. While particulates can accumulate on such a surface, they are also rather easily washed off by rain or blown off by wind. Frequently, however, small particles can be incorporated into the cuticle or enter the stomata. The leaf surface of many plants also are covered by numerous fine branched hairs or glands. These can act as traps for particulates so that very dirty leaf surfaces can occur in areas of high dustfall. Nevertheless, the directly toxic effects of particulate metal constituents are limited as they are kept away from metabolically active sites.

In the mosses, liverworts and lichens, the cuticle is effectively absent. The exposed cell wall surface is at the air interface and it consists of charged sites, which can exchange both cations and anions. The metals are selectively exchanged onto this surface and are held there. Large accumulations can take place. Lichens in polluted regions attest to this, as do mosses (e.g. Rühling and Tyler 1970). The use of *Sphagnum* moss bags as air monitors is based on this cation exchange capacity. The special sensitivity of many lichens to air pollutants is due to the ready entry of the pollutant to metabolically active sites.

Soil Surface Layers as Target Areas at Risk

The surface of the soil as a critical site for accumulation of airborne metallic contaminants has already been referred to here. It is in this upper few centimetres of the soil that nutrient uptake into plant roots takes place and in which new root hairs are developed. Seeds germinate in this layer amongst the forest litter and seedlings establish there. The microbial sequences essential

Table 3. Residence times of water in the saturated zone of the soil.

Discharge per year (mm)	System and pathways	Estimated residence time depending on the place of infiltration and on porosity of the soil or rock	Watershed
A. 250 mm	About constant	100-500 years	Merkenfritzbach (Federal Republic of Germany, Land Hessen, Main area) 1600 ha
350 mm	slow discharge 17		
160 mm	partly 12		
	slow discharge 12	1-500 years	
	fast discharge 10	1 day-10 years	
	partly 12		
B. 150 mm	(12, 17)	10-1000 years	Okkenbroek (The Netherlands, IJssel area) 443 ha
100 mm	(12)	1 day-10 years	
50 mm	(4, 10)	1 hour-1 day	

for effective litter decomposition take place, in this zone pathogens strive to infect seedlings or root systems, and the mycorrhizal fungi essential for effective nutrition of many forest trees, especially in the boreal forest, develop here. The rhizobial bacteria which act as nitrogen fixers in legumes and the actinomycetes and blue-greens which fulfill this role in other shrubs and grasses also have to infect roots in these upper few centimetres of the soil. Yet, onto this surface is being deposited increasing loads of toxic heavy metals, of acidifying substances and also of gaseous pollutants. The threat to the safe functioning of such ecosystems and to the well-being of man are focused on this zone. Indeed, we can consider that the reduction or elimination of just a few key processes could put the whole system at risk. The enzyme aryl sulphatase which produces the plant-available sulphate from the non-available organic sulphur in soils is one such step, and it is known that soil-extracted aryl sulphatases are susceptible to a wide range of heavy metals including aluminium. The ability of rhizobial bacterial to infect legume roots is also known to be acid sensitive and heavy metal susceptible. The conditions for seedling establishment might be affected if the atmospheric inputs of wet and dry deposition were to acidify surface soils so as to favour fungi at the expense of bacteria.

The threat to litter decomposition may be a long time in developing in most areas but in those where intense heavy metal accumulations have occurred from smelter emissions, examples of this have already been demonstrated. In both the remnant forest in the major Sudbury smelting area, where nickel and copper concentrations have reached up to 2000 ppm in the past and in the New Lead Belt of Missouri, where lead, zinc, cadmium and

copper are now high, abnormal accumulations of litter on the forest are reported (Freedman and Hutchinson 1980, Watson and others 1976). At the zinc smelter of Palmerton in Pennsylvania, Strojjan (1978) reported reduced decomposition of the foliage of a number of species and ascribed it to elevated zinc and cadmium concentrations.

The reports have also included detrimental effects on a number of soil enzyme activities, on overall microbial respiratory activity and on micro-arthropod and earthworm action. Indeed, many reports are now available which show sensitivity of earthworms to heavy metal accumulations.

Sediments as Sites of Risk

It should be noted that a rather similar but parallel case can be made for effects on surface sediments. Again, in this zone, much of the microbial activity takes place, the aquatic plants have to root, the benthic organisms live and reproduce and the important gas exchanges take place with the water column. This is the zone of deposition of the dead planktonic organisms, of incoming particulate matter and of pollutant material equally. The sediments are often highly organic and have a large cation exchange capacity. Toxic levels can develop (mercury in polluted sediments of the Detroit River and Lake St. Clair), and benthic-feeding fish, clams, crayfish etc. come in contact with pesticides, PCB's and heavy metals which have been initially transported by air. The methylation transformation which creates organic mercury compounds nine to ten times more toxic than inorganic equivalents take place here in the sediment surface layers.

Reproduction: crucial steps at risk

A number of essential sequential steps can be identified, the interference with which places the whole reproductive process at risk. These include a) the health of pollinators, especially the insects essential for those plants with specialised mechanisms and appropriate floral guides, trip mechanisms, nectar production, pollen positioning etc., b) the ability of the pollen to germinate successfully on the stigma and to then successfully produce a pollen tube which can reach the unfertilised ovules c) the ability of sperm to successfully reach and fertilise the egg d) the ability of plants and animals to successfully disperse their progeny to suitable new habitats.

Interestingly, it is known that steps a-c can all be affected by elevated levels of heavy metals. For example, bees are known to be very susceptible to airborne arsenic. In the region of the Novatny power station in Czechoslovakia bee hives have been wiped out. The power-station burns coal with high levels of arsenic i.e. several hundred ppm, and consequent elevated air levels of As_2O_3 occur. Bees also will pick up and accumulate selenium when pollinating high selenium plants, such as some of the loco-weeds (*Astragalus* species).

The successful growth of the pollen tube is reported to be pH-dependent and also to be influenced by the presence of toxic heavy metals such as zinc and copper. The fertilization of fern archegonia has been reported to be inhibited by acid solutions and acid rain, L. Evans, (personal communication). L. Schlichter, a graduate student in botany at University of Toronto, has recently shown that the successful post-fertilization steps in embryo development of frog eggs is inhibited by even quite minor decreases in pH below 6.0 and that multiple-fertilization of an egg, which normally are precluded, can occur under these acidic conditions. The consequence is an early aborted embryo. Effects on trout eggs, on New Jersey frogs and on reproductive success in planktonic crustaceans have been reported by a number of workers, e.g. Krishna (1953), Gosner and Black (1957), Havas (1980).

Other Factors which Influence the Outcome of Metal Impacts

It does seem to be the case that the damage to ecosystems from airborne metals, as well as from gaseous pollutants, is greatest when the individuals in the ecosystem are metabolically most active. Thus, the damage to forest ecosystems in temperate zones is clearly greatest in the summer growing season. Damage during the day when stomata are open is greater than at night when they are closed. The lichens and mosses are most susceptible when they are moist and photosynthesizing actively. In dry or arid habitats,

such as the deserts of Arizona, the damage from large smelter-emitted SO_2 and copper particulates is minimal in contrast to that of wetter areas such as Palmerton Pennsylvania, Ducktown Tennessee or Sudbury, Ontario. This probably partly relates to metabolic activity including the percentage of time stomata remain open, but also to the activity of the root systems. If one follows through with this generalization, then we can predict that arctic regions with very short growing seasons and very long dormant periods will be less affected by equivalent metal pollutant inputs (or SO_2 , O_3 or F1 inputs) than would Temperate or especially Tropic Rain Forest systems. Equally, one can predict that the more arid an area, the less susceptible this area's vegetation will be to toxic damage.

The importance of the longevity of the individual also needs to be stressed. Damage to long-lived trees may take a long time to become apparent and finally perhaps only through their inability to reproduce. Equivalent reproductive failure in an annual will obviously be very rapidly apparent. Metal stresses often seem to favour perennial plants with largely vegetative reproduction, such as grasses and sedges. The equivalent aquatic examples would be that of fish compared with planktonic algae or crustaceans.

Finally, it must be emphasised that the initial sensitivities of the species, populations and individuals of an area when first subject to metal stress, are not the final response. While pre-adaptation or pre-sensitivity may allow an initial selection and sorting, the stressed environment represents a changing habitat in which evolutionary change occurs. The occurrence of metal-tolerant grasses on mine waste sites is well known. The ability of some of these grasses to evolve multiple-metal tolerances and co-tolerances is now also receiving attention (Tatsuyama and others 1975, Cox and Hutchinson 1980). Even in habitats such as the Smoking Hills of arctic Canada, where pond water pH's may reach as low as 1.8 and soil pH's to < 3.0 from the initial values of pH > 7.0, some organisms do survive. These even have the ability to tolerate the high acidity, low nitrogen and phosphorus availability and the extremely elevated levels of normally toxic metals such as aluminium, manganese, ferric ion and zinc, Hutchinson and others (1978). At the extreme, some organisms seem to have evolved specifically on areas of very elevated metal levels. The occurrence of a legume *Becium homblei* on copper mineralizations is an example of this, in which the plant accumulates enormous levels of copper and also requires concentrations which would be lethal to other plants (Reilly 1967).

While this allows a certain re-assurance about the ability of life to thrive under even most adverse conditions, it does not at all influence

the fact that we must be extremely concerned with the accelerating liberation of heavy metals to our agricultural and natural environments. Economic and social pressures do compel us to take stock. Proposals for disposal of metal-contaminated sewage sludge on farm or forest lands have already caused agencies and scientists to think in a much longer time frame than we are used to do.

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LITERATURE CITED

- Abrahamsen, G., R. Horntvedt, and B. Tveite. 1977. Impacts of acid precipitation on coniferous forest ecosystems. *Water, Air and Soil Pollution* 8:57-73.
- Allen, R.O. and E. Steinnes. 1979. Contribution from long-range atmospheric transport to the heavy metal pollution of surface soil. In: *Internat. Conf. Management and Control of Heavy Metals in the Environment*. Ed. R. Perry. Pub. CEP Edinburgh. 271-274.
- Bacon, J.R. and R.P. Maas. 1979. Contamination of Great Smoky Mountains trout streams by exposed Anakeesta formations. *J. Environ. Quality*. 8:538-43.
- Baker, J.P. and C.L. Schofield. 1980. Aluminium toxicity to fish as related to acid precipitation. In: *Internat. Conf. on The Ecological Impact of Acid Precipitation*. Ed. L. Overrein. SNSF Project (In Press).
- Beauford, W., J. Barber and A.R. Barringer. 1975. Heavy metal release from plants into the atmosphere. *Nature* 256:35-37.
- Bowen, H.M.J. 1975. Residence times of heavy metals in the Environment. In: *Intern. Conf. on Heavy Metals in the Environment*. Symp. Proc. vol. 1. Ed. T.C. Hutchinson, Pub. Inst. Environmental Studies, Univ. Toronto. 1-19.
- Cohen, J.B. and A.L. Rushton. 1925. *Smoke. A Study of Town Air*. London.
- Cox, R.M. and T.C. Hutchinson. 1980. Multiple metal tolerances in the grass *Deschampsia cespitosa* (L.). Beauv. from the Sudbury smelting area. *New Phytol.* 84:631-647.
- Cronan, C.S. and C.L. Schofield. 1979. Aluminium leaching response to acid precipitation: effects of high-elevation watersheds in the north-east. *Science*. 204:304-305.
- Czuba, M. and D.P. Ormrod. 1974. Effects of cadmium and zinc on ozone-induced phytotoxicity in cress and lettuce. *Can. J. Bot.* 52:645-650.
- Freedman, B. and T.C. Hutchinson. 1980. Smelter pollution near Sudbury, Ontario and effects on forest litter decomposition. In: *Effects of Acid Precipitation on Terrestrial Ecosystems*. NATO Eco-Sciences Conf. Ecology Series 4. Ed. T.C. Hutchinson and M. Havas. Pub. Plenum N.Y. 395-434.
- Frissel, M.J. 1978. *Cycling of Minerals Nutrients in Agricultural Ecosystems*. Pub. 43 of series Mededelinger KNHM. Developments in Agriculture and Managed-Forest Ecology 3. Elsevier. Amsterdam.
- Gosner, K.L. and I.H. Black. 1957. The effects of acidity on the development and hatching of New Jersey frogs. *Ecology* 38:256-262.
- Groth, D.H., L. Stettler and G. Mackey. 1976. Interactions of mercury, cadmium, selenium, tellurium, arsenic and beryllium. In: *Effects and Dose-Response Relationships of Toxic Metals*. Ed. G.F. Nordberg. Elsevier. Amsterdam. 527-544.
- Havas, M. 1980. Studies of the invertebrates inhabiting strongly acidified ponds at the Smoking Hills, N.W.T. in arctic Canada. Ph.D. Thesis. Department of Botany, University of Toronto.
- Hodge, V., S.R. Johnson and E.D. Goldberg. 1978. Influence of atmospherically transported aerosols on surface ocean water composition. *Geochem. J.* 12: 7-20.
- Hutchinson, T.C. 1973. Comparative studies of the toxicity of heavy metals to phytoplankton and their synergistic interactions. *Water Polln. Res. in Canada* 8:68-90.
- Hutchinson, T.C. 1980. Effects of acid leaching on cation loss in soils. In: *Effects of Acid Precipitation on Terrestrial Ecosystems*. NATO Eco-Sciences Conf. Ecology Series 4. Ed. T.C. Hutchinson and M. Havas. Pub. Plenum N.Y. 481-497.
- Hutchinson, T.C. and H. Czyrska. 1972. Cadmium and zinc toxicity and synergism to floating aquatic plants. *Water Polln. Res. in Can.* 7: 59-65.
- Hutchinson, T.C. and L.M. Whitby. 1974. Heavy metal pollution in the Sudbury mining and smelting region of Canada. I. Soil and vegetation contaminated by nickel, copper and other metals. *Environmental Conservation* 1: 123-132.

- Hutchinson, T.C. and P.M. Stokes. 1975. Heavy metal toxicity and algal bioassays. In: Water Quality Parameters. Special Tech. Pub. 573 of A.S.T.M. Philadelphia:320-343.
- Hutchinson, T.C., W. Gizyn, M. Havas and V. Zobens. 1978. Effect of long-term lignite burns on arctic ecosystems at the Smoking Hills, N.W.T. In: Trace Substances in Environmental Health. Ed. D.D. Hemphill. Pub. University of Missouri, Columbia. 7:317-332.
- Krishna, D. 1953. Effect of changing pH on developing trout eggs and larvae. Nature 171:434.
- National Academy of Sciences. 1980. Lead in the Human Environment. N.A.S. Washington.
- Levander, O.A. 1977. Metabolic interrelationships between arsenic and selenium. Environ. Health Perspect. 19:159-163.
- Parizek, J. 1978. Interactions between selenium compounds and those of mercury or cadmium. Environ. Health Perspect. 25:53-58.
- Reilly, C. 1967. Accumulation of copper by some Zambian plants. Nature 215:667-669.
- Rühling, A. and G. Tyler. 1970. Sorption and retention of heavy metals in the woodland moss Hylecomnium splendens. Oikos 21:92-97.
- Schindler, D.W., R.H. Hesslein, R. Wagemann and W.S. Broecker. 1980. Effects of acidification on mobilization of heavy metals and radionuclides from the sediments of a freshwater lake. Can. J. Fish. Aquatic Sci. 37:373-377.
- C.L. Schofield. 1976. Acid precipitation: effects on fish. Ambio. 5-6:228-230.
- Strojan, C.L. 1978. Forest leaf litter decomposition in the vicinity of a zinc smelter. Oecologia 32:203-212.
- Tatsuyama, K., H. Egawa, H. Yamamoto and H. Senmaru. 1975. Tolerance of cadmium resistant microorganisms to other metals. Trans. Mycol. Soc. Jap. 16:79-85.
- Watson, A.P., R.I. Van Hook, D.R. Jackson and D.E. Reichle. 1976. Impact of a lead-mining complex on the forest-floor arthropod fauna in the New Lead Belt region of S.E. Missouri. Oak Ridge Natl. Lab. Environ. Sci. Div. Publ. 881. 161 pp.

Effects of Acidic Precipitation on Health and the Productivity of Forests¹

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Abstract: Acid precipitation has become a dominant feature of man-induced change in the chemical climate of the earth. But acid precipitation is only one special feature of the changing chemistry of atmospheric deposition in developed and developing regions throughout the world. In recent decades, human activities (mainly increased combustion of fossil fuels and decomposition or combustion of waste products) have greatly increased the total emissions and deposition of beneficial nutrients and injurious substances (such as strong mineral acids) from the atmosphere. Projected increases in the use of fossil fuels, and especially in the use of coal, will add still further to the total burden of beneficial and injurious substances deposited on forest and rangeland ecosystems from the atmosphere. The purpose of this brief paper is to summarize certain important principles concerning the phenomena of acid precipitation and atmospheric deposition and their beneficial and detrimental effects on the health and productivity of forests.

The supply of both beneficial nutrient elements and injurious substances in the atmosphere influence the health and welfare of forests. Plant life as we know it would be impossible without atmospheric sources of carbon dioxide for photosynthesis, nitrogen for biological fixation and proteins synthesis, oxygen for respiration and synthesis of carbohydrates, and water for

transpiration and the maintenance of cell turgor.

Some epiphytic plants, such as orchids, spanish moss, and certain lichens, obtain essentially all their nutrients and water from the atmosphere. Although these plants represent an extreme case of dependence on atmospheric resources, many forest trees and some herbaceous plants also derive a significant portion of their nutrients from the atmosphere.

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Plants suffer when the concentrations of injurious substances in the atmosphere exceed the amounts they can tolerate. Injurious gases can enter through the stomata of leaf tissues and poison the photosynthetic system of living cells. Toxic particles can accumulate on plant surfaces and injure plant cells. Strong acids can dissolve in rain drops or adsorb to snowflakes and then be deposited in precipitation. Dissolved substances can accumulate in snow where they may be concentrated and released with the first meltwater. In

all these ways substances transferred from the atmosphere to the biosphere can influence plant growth, beneficially (as in the case of beneficial nutrient elements) or harmfully (as in the case of toxic gases, aerosols, dry particulate matter, and injurious substances dissolved in precipitation).

The growth and productivity of forests are determined by the availability of sixteen elements that are essential for growth and a few that are toxic to plants. The essential elements include nine major elements: carbon, hydrogen, oxygen, nitrogen, phosphorous, potassium, sulfur, calcium, and magnesium and seven minor elements: iron, copper, zinc, manganese, molybdenum, boron, and chlorine.

Some elements are both essential and injurious to plants. For example, sulfur and nitrogen are needed for synthesis of protein, nucleic acids, and other substances; but gaseous sulfur and nitrogen oxides and sulfuric and nitric acid aerosols are also injurious to plants at very low concentrations. Similarly, excess amounts of the minor nutrient elements also can injure plants. Atmospheric fluoride is toxic to plants at 25-50 ppm. Aluminum is the most abundant potentially toxic element in soils. Its availability (and thus its toxicity) is influenced greatly by the acidity of soils, which in turn is influenced by the abundance of acid precipitation.

Uptake of nutrients from atmospheric sources is especially important in natural ecosystems such as lakes, estuaries, wetlands, forests, and rangelands where nutrients from other sources are scarce and where fertilization is not a normal management procedure. But this capacity also increases the vulnerability of terrestrial and aquatic organisms to injury by acid precipitation and toxic aerosols and gases (Galloway and others 1978).

ACID PRECIPITATION AS PART OF A GENERAL PHENOMENON OF ATMOSPHERIC DEPOSITION

Air-borne substances that influence terrestrial plants include sea spray from oceans and large lakes; dust resulting from wind erosion of soil as well as from volcanic and cosmic sources; gases such as CO_2 , NH_3 , SO_2 , H_2S , CH_4 , released from decomposing organic matter and volcanoes; biogenic particles such as spores, hyphal fragments, bacteria, and pollen; particulate matter, aerosols, and gases produced by wild fires and controlled burning of agricultural, forest and urban wastes as well as from industrial, agricultural residential and commercial heating, and transportation operations (Tamm 1958).

Rain and snow change in chemical composition within, as well as between, precipitation events. In cold climates, acid substances accumulate in the snowpack where they are released in concentrated form with the first melt water and thus

cause very sudden increases in acidity of surface soils, vegetation, and surface waters. Thus a given plant may be subject to beneficial atmospheric influences at one time and to negative influences at another time within a given day, month, growing season, or the years of its development in the case of perennial plants and animals. Even a given substance, such as sulfur or nitrogen dioxides, may be absorbed and utilized as a beneficial nutrient at one concentration in the atmosphere. At another, higher concentration, even on the same day, however, the same substance may be absorbed and found to be toxic or even injurious to the very same plant.

Forests and rangelands cover a larger fraction of the total land area of the United States than all other uses of land combined. For this reason, terrestrial vegetation, soils, and surface waters are the primary deposition sites for precipitation and airborne particulate matter of all types. Trees develop very large canopies of leaves and branches that extend high into the air. Thus, forests and range plants provide an extremely large surface for deposition and assimilation of both beneficial nutrient elements and injurious substances dispersed in the atmosphere.

Direct injury to vegetation is most likely when a particularly vulnerable life form is exposed at a particularly vulnerable life stage, and is growing in a poorly buffered environment during a season of the year when acid precipitation is most likely. For example, a tender young plant, at the earliest stage of reproduction, growing on a poorly buffered sandy soil, during a heavy spring rain is especially vulnerable to acid rain.

Both herbaceous and perennial plants are subject to changes in atmospheric deposition within a given growing season. In addition, perennial shrubs and trees live in the same environment for many years or even decades. As a result, they are subject to very long-term changes in the chemistry of the air and precipitation.

The effect of acid precipitation on plants is only one facet of the much larger subject of atmospheric/plant/soil interactions. Acidity in precipitation should be understood as a reflection not only of the amounts of substances yielding hydrogen ions (such as sulfuric, nitric, hydrochloric, and organic acids) but also of the total balance between all the cations and anions dissolved in precipitation. These major anions and cations include H^+ , NH_4^+ , NO_3^- , SO_4^{2-} and many others including K^+ , Na^+ , Ca^{++} , Mg^{++} , CO_3^{--} , Cl^- , and PO_4^{--} .

For all of the above reasons, it is difficult to assess the effects of acids in rain or snow in isolation from the general chemistry of precipitation and atmospheric deposition. Also, the effects of a given "acid rain" or a prevailing condition of "acid rains" are very complex,

variable in time, and involve significant interactions that are only partially understood.

POSSIBLE DETRIMENTAL EFFECTS OF ACID PRECIPITATION ON VEGETATION

A partial list of theoretical effects of acid precipitation on vegetation was developed earlier by Tamm and Cowling (1977) and is reproduced in Table 1. The effects are classified as either direct or indirect, although most direct effects will have many indirect consequences as well. A decreased rate of growth is the expected consequence of most of the effects postulated in Table 1 but unequivocal evidence of significant growth effects have yet to be demonstrated in forest or range ecosystems. Specific biological effects that have been proven to occur in at least one experimental plant are marked with an asterisk (*) in Table 1.

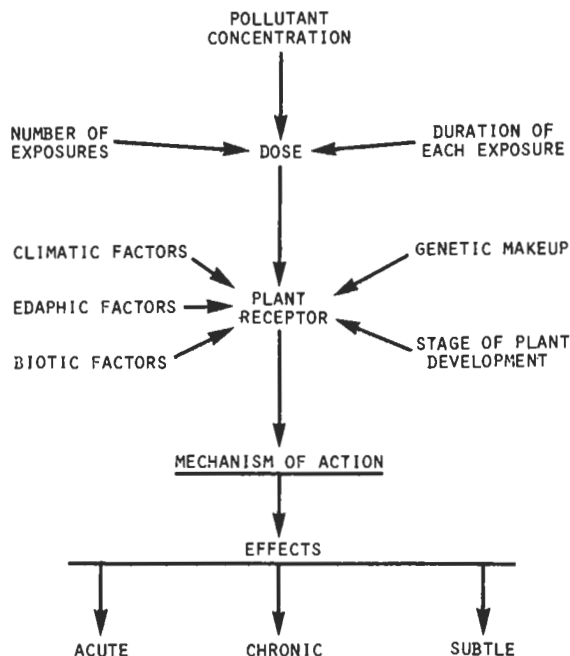
Many factors (i.e., genetic composition, biotic and abiotic stress factors, dose of pollutant, and pollutant combinations) affect the impact of acid precipitation and other pollutants on terrestrial plants and animals. Variation in any one factor can result in variation in the nature and magnitude of pollutant effects. This is shown simplistically in Figure 1.

Previously, it was believed that the essential and potentially toxic elements listed above were taken up by plants almost entirely from the soil solution. Now, it is recognized that airborne gases, particulate matter, and aerosols significantly augment the supply of both essential and injurious elements. All of the substances

listed above can be taken up readily through foliar organs as well as by absorption from the soil solution (Tamm 1958; Wittwer and Bukovac 1969).

Much larger amounts of essential nutrients are required for sustained-yield agriculture than for sustained-yield range management or hardwood or softwood forestry. This is true in rangelands because biomass yields are very low and in forests because the parts of trees that usually are harvested (the wood and bark of tree stems) contain much less of most essential elements than the seeds and fruits that are commonly harvested in agriculture. This is a major reason why fertilization is so common in agriculture and so rare in forestry and range management. In some forested regions, atmospheric deposition alone is more than adequate to permit harvesting of crop after crop of trees without fertilizing the forest. This is much less likely to remain so, as more and more of the nutrient-rich branches, foliage, and roots of trees are harvested in so-called "whole-tree chipping" and other modern harvesting practices.

Some scientists believe that acid rain and snow are deposited directly onto soils where acid substances can be neutralized in well-buffered soils or by applications of lime. This is true on some agricultural lands, especially after harvest, but is not true in forests, rangelands, or even on most agricultural lands during the growing season. Most raindrops are intercepted by the foliage of plants where substances dissolved in rain can induce various physiological changes before reaching the soil (see Table 1). In a mature forest, for example, rain will wash over at least three tiers of foliage before reaching the soil.



SOURCES, AMOUNTS, AND DISTANCES OF TRANSPORT OF BENEFICIAL AND INJURIOUS SUBSTANCES IN THE ATMOSPHERE

Forest, range, and aquatic biologists are becoming increasingly concerned about atmospheric transport and deposition of both nutritionally beneficial and potentially injurious substances for three major reasons:

- (1) vegetation, soils, and surface waters are the primary deposition sites for precipitation and airborne particulate matter;
- (2) atmospheric deposition constitutes an important source of nutrients and potentially injurious substances that affect the productivity and stability of agricultural, forest, and aquatic ecosystems; and
- (3) human activities are steadily increasing the amounts and variety of substances dispersed in the atmosphere (Oden 1968; Bolin and others 1972).

Figure 1. Conceptual model of factors involved in air pollution effects on vegetation (Heck and others 1977).

Table 1--Potential Effects of Acid
Precipitation on Terrestrial Vegetation

DIRECT EFFECTS

- *1. Damage to protective surface structures such as cuticle. Damage to surface structures may occur due to accelerated erosion of the cuticular layer that protects most foliar organs. It also could result from direct injury to surface cells by high concentrations of sulfuric acid and other harmful substances that are concentrated by evaporation or adherence of soot particles on plant surfaces.
- *2. Interference with normal functions of guard cells. Malfunction of guard cells will lead to loss of control of stomata and thus altered rates of transpiration and gas-exchange processes and possibly increased susceptibility to penetration by leaf-attacking plant pathogens.
- *3. Poisoning of plant cells after diffusion of acidic substances through stomata or cuticle. This could lead to development of necrotic or senescent spots on foliar organs including leaves, flowers, twigs, and branches.
- *4. Disturbance of normal metabolism or growth processes without necrosis of plant cells. Such disturbance may lead to decreased photosynthetic efficiency, altered respiratory patterns and intermediary metabolism, as well as abnormal development or premature senescence of leaves or other organs.
- *5. Alteration of leaf- and root-exudation processes. Such alterations may lead to changes in populations of leaf-surface and root-surface microorganisms, including nitrogen-fixing organisms.
- *6. Interference with reproduction processes. Such interference may be achieved by decreasing the viability of pollen, interference with fertilization, decreased fruit or seed production, decreased germinability of seeds, etc.
- 7. Synergistic interaction with other environmental stress factors. Such reinforcing interactions may occur with gaseous sulfur dioxide, ozone, fluoride, soot particles, and other air pollutants as well as drought, flooding, etc.

INDIRECT EFFECTS

- *1. Accelerated leaching of substances from foliar organs. Damage to cuticle and surface cells may lead to accelerated leaching of mineral elements and organic substances from leaves, twigs, branches, and stems.
- 2. Increased susceptibility to drought and other environmental stress factors. Erosion of cuticle, interference with normal functioning of guard cells, and direct injury to surface cells may lead to increased evapotranspiration from foliar organs and vulnerability to drought, air pollutants, and other environmental stress factors.
- *3. Alteration of symbiotic associations. Changes in leaf- and root-exudation processes and accelerated leaching of organic and inorganic substances from plants may affect the formation, development, balance, and function of symbiotic associations, such as mycorrhizae, nitrogen-fixing organisms, lichens, etc.
- *4. Alteration of host-parasite interactions. Resistance and/or susceptibility to biotic pathogens, parasites, and insects may be altered by subjecting plants to any environmental stress. Acid precipitation may increase the susceptibility of plants to these injurious agents, alter their capacity to tolerate disease or injury, or alter the virulence of pathogens. The effects of acidic precipitation may vary with the following: the nature of the pathogen involved (whether a fungus, bacterium, mycoplasma, virus, nematode, parasitic seed plant, insect, or multiple-pathogen complex); the species, age and physiological status of the host; and the stage in the disease cycle in which the acidic stress is applied, for example, acidic rain might decrease the infective capacity of bacteria before infection and increase the susceptibility of the host to disease development after infection.

Source: Tamm and Cowling, 1977

Recent increases in the deposition of substances on terrestrial vegetation are due mainly to increases in combustion of fossil fuels in industrial enterprises, residential heating, transportation, and agricultural operations. Previously, it was believed that most of these materials fell out of the atmosphere near the site of emission. Now it is recognized, particularly with increased use of tall stacks at power plants, that atmospheric processes can lead to extensive mixing and both chemical and physical interactions and transformations of atmospheric particles, aerosols, and gases. Furthermore, these substances and their reaction products are dispersed by meteorological processes and finally are deposited on vegetation or soils as much as several hundreds of kilometers from the original sources of emission. The recent fallout of radioactive materials in the eastern United States as the result of atomic explosions in the Peoples Republic of China provides a dramatic reminder of the long-distance transport and deposition of pollutants.

The amounts of substances introduced deliberately or inadvertently by man are becoming so large that man is becoming a major force in the biogeochemistry of the earth (Kovda 1975). This is shown in table 2 which contains a tabulation of data on annual output of fertilizers, industrial dusts, garbage and other urban wastes and by-products, mine refuse, and discharges of aerosols and gases mainly from the combustion of fossil fuels. All these categories of matter are becoming comparable in magnitude to the discharges of dissolved and suspended substances in all the rivers of the world, the annual yield of photosynthetic products, or the cycling of inorganic elements in the earth as a whole. Man-made emissions into the atmosphere are also

Table 2--Biogeochemical and Technological Forces in the Biosphere of the Earth

Biosphere Components	Tons/Year
Biogeochemical processes:	1×10^{10}
Global yield of photosynthesis	1×10^{10}
Cycle of inorganic elements	1×10^{10}
River discharges:	
Dissolved substances	3×10^9
Suspended substances	2×10^{10}
Anthropogenic sources:	
Output of fertilizers	3×10^8
Industrial dust	3×10^8
Garbage, urban wastes and byproducts	2×10^{10}
Mine refuse	5×10^9
Aerosols and gas discharges	1×10^9

Source: Kovda, 1975

very large, as shown in table 3. Most gases, carbon oxides, and aerosols result from the combustion of fossil fuels. A very large part of these global emissions are produced in the United States.

If the United States continues to add to the amount of substances dispersed in the atmosphere and deposited into the biosphere of the earth, it is essential that we measure the amount and chemical form of the deposited matter and understand the biological consequences of that deposition. Regrettably our understanding of these processes in the United States is very fragmentary. Fortunately, however, more extensive measurements of atmospheric deposition and its biological consequences have been made in Europe, where an atmospheric-deposition network has been maintained since the late 1940s (Oden 1968).

The European Air Chemistry Network began in Sweden and has gradually spread to include most of western Europe and parts of eastern Europe, including Poland and the Soviet Union. Since the mid 1950's, a network of about 100 stations has made monthly measurements of changes in the chemistry of precipitation. The substances analyzed at most of these stations include the following major cations and anions: NH_4 , Na, Ca, K, Mg, SO_4 , NO_3 , PO_4 , Cl as well as pH, conductivity, and titratable acidity and alkalinity. These data have shown various long-term trends. For example, the amount of nitrate nitrogen in precipitation (an important fertilizer element) increased markedly in many parts of Europe during the fifteen years between 1955 and 1970. Nitrate nitrogen helps plants grow. Thus, the nitrogen added in precipitation probably increased yields of agricultural and forest crops.

But not all the substances detected in precipitation were beneficial. Long-term trends of injurious sulfate and hydrogen ions also were detected from 1955-1970. The latter changes were attributed to strong acids formed in the atmosphere, mainly from oxides of sulfur and nitrogen produced during combustion of fossil fuels. More recent data show that these trends of increasing acidity are

Table 3--Anthropogenic Emissions into the Atmosphere

Types of Emissions	Tons/Year
Dust	2.5×10^8
Gases (mainly SO_2 , HC and NO_x)	6.5×10^8
Carbon oxides ($\text{CO} + \text{CO}_2$)	2.0×10^9
Aerosols	1.0×10^9
Note: Doubling about every 7-10 years	

Source: Kovda, 1975

continuing although the relative contribution of sulfuric and nitric acids is changing (Likens 1976).

CHANGES IN THE CHEMISTRY OF PRECIPITATION IN THE UNITED STATES

Some monitoring of the chemistry of precipitation has been carried on in the United States (Feth and others 1964; Lodge and others 1968). Many of these studies provide excellent and reliable information about the acidity of precipitation. But most studies in this country have suffered from three major shortcomings:

1. The data were collected for a limited land area--typically only a single point or a few points in one or two states (Gambell and Fisher 1966);
2. The data were collected for very limited periods of time--typically only one or two years;
3. Very few direct measurements of acidity have been made.

There is only one location in the United States--at Hubbard Brook Experiment Forest in New Hampshire--where the acidity of rain has been measured directly and consistently for as long as 10 years. The longest-term national monitoring program was operated by the U.S. Public Health Service for 6 years, from 1960 to 1966 (Lodge and others 1968). These data showed that precipitation generally is acidic east and generally alkaline west of the Mississippi River, the latter because of alkaline dust in the air.

Using fragmentary bits of information, obtained indirectly and in limited areas and periods of time, Cogbill and Likens (1974) managed to calculate the probable changes in average acidity of rainfall in various parts of the eastern United States from 1955-1973. Precipitation in a large portion of the eastern United States was less than pH 5.6 in 1955-56; the zone of greatest acidity (lowest pH) was generally consistent with the zone where sulfur emissions were high--parts of Ohio, Pennsylvania, West Virginia, New York, and New England. By 1973, however, the area with an average pH of rain below 4.5 had extended to include parts of Missouri, Arkansas, Mississippi, Alabama, Georgia, South Carolina, Virginia, Kentucky, Illinois, Michigan, and further north into New England and Canada. Essentially, it embraces most of the area east of the Mississippi River. Individual rainstorms with pH values between 2.1 and 3.6 have been reported in New York, Illinois, Indiana, New Hampshire, Massachusetts and North Carolina--in some cases many hundreds of kilometers from major sources of air pollution (Likens 1976).

The relative contribution of sulfate and nitrate to the total acidity of precipitation

apparently changed markedly during the years since 1964-65. At Hubbard Brook, New Hampshire, the ratio of sulfate to nitrate changed from 83:15 in 1964 to 66:30 in 1974. During this same decade, the total input of hydrogen ions increased by 36 pct. Thus, most of this increase appears to be due to increased deposition of nitric acid.

EFFECTS OF ACID PRECIPITATION ON TERRESTRIAL ECOSYSTEMS

Cowling (1980a, 1980b) has recently completed 2 historical analyses of progress in scientific and public understanding of acid precipitation and its biological consequences. Several publications are worthy of special notice in this connection. The pioneering researches by Robert Smith, Eville Gorham and Svante Odén dealt with effects on lake waters, aquatic vegetation, terrestrial vegetation, and human health (Smith 1872; Gorham 1958, 1976; Odén 1968). In 1971, Bolin and his co-workers completed the Swedish Case Study Contribution to the United Nations' Conference on the Human Environment (Bolin and others 1972). In 1972, three Norwegian research organizations established a special research project called Acid Precipitation: Effects on Forests and Fish, with an annual budget of 10,000,000 Norwegian kroner (U. S. \$2,000,000). The first International Conference on Acid Precipitation and the Forest Ecosystem was held at Ohio State University at Columbus in May of 1975 (Dochinger and Seliga 1976a). In June of 1976, an International Conference on Effects of Acid Precipitation was held at Telemark, Norway, and the major papers assembled for this meeting published by Braekke (1976) and in a special issue of Ambio (1976). In November, 1976, Gene Likens published his summary report in Chemical and Engineering News (Likens 1976). In May, 1978, a NATO Advanced Research Institute on Ecological Effects of Acid Precipitation was held at Toronto, Canada (Hutchinson and Havas 1980). In September, 1978, the Central Electricity Generating Board in England and the Electric Power Research Institute in the United States sponsored an international symposium on the biological effects of acid precipitation (Howells 1979). In March, 1980, the Norwegian special project on acid precipitation sponsored an International Conference on Effects of Acid Precipitation in Sandefjord, Norway (SNSF 1980).

The effects of acid precipitation on terrestrial ecosystems generally have been less well documented than those on populations of freshwater fish and other aquatic organisms (Ambio 1976; Braekke 1976). Nevertheless, certain definite effects have been reported. The most striking of these effects was the development of peat moss (Sphagnum sp.) as a submarine, rather than a terrestrial plant in acidified lakes and streams in Sweden. Dense mats of Sphagnum and heavy felts of algae develop on the bottom of these lakes in water as deep as 18 m. This growth is reported by Grahn and others (1974) to induce oligotrophication (opposite of eutrophication)--a self-accelerating

process that leads to a substantial nutrient impoverishment of lake waters.

Analyses of forest growth in southern Sweden from 1896 to 1965 showed a 2 to 7 percent decrease in growth between 1950 and 1965. Johnsson and Sundberg (1972) "found no good reason for attributing [this] reduction in growth to any cause other than acidification." Similar attempts to quantify possible effects on growth of forests in the United States have been inconclusive.

Both direct and indirect damage to crops and forests have been reported by various investigators in laboratory, greenhouse, and field experiments in which synthetic rain equivalent in chemical composition and rate of deposition to natural rains has been applied. The biological effects recorded in these experiments include the following (Cowling 1980c):

- Induction of necrotic lesions on foliage;
- Loss of nutrients due to leaching from leaves and other foliar organs;
- Predisposition of plants to infection by bacterial and fungal pathogens;
- Accelerated erosion of waxes on leaf surfaces;
- Inhibition of nodulation of legumes leading to decreased fixation of nitrogen by symbiotic bacteria; and
- Reduced rates of decomposition of leaf litter leading to decreased mineralization of organically-bound nutrients.

Abrahamsen (1980) has recently summarized many years of research showing both positive and negative effects of acid precipitation on forest growth. He concludes with the following general statements: "Apart from possible direct effects of acid precipitation on forest trees, the effects on forest growth can be considered a nutrition problem . . . increased deposition of N and S can be regarded as a . . . fertilization effect, and the increased leaching of nutrient cations . . . as an oligotrophication or acidification effect . . . the general hypothesis that acid precipitation significantly will decrease forest production over large areas must be revalued. The deposition of N and to some extent S . . . is likely to increase forest production. Reduced growth may be expected where or when nutrients like Mg and possibly K are the growth limiting elements."

RECENT INITIATIVES DEALING WITH ACID PRECIPITATION AND ITS BIOLOGICAL EFFECTS

In 1975, the National Academy of Sciences' Committee on Atmospheric Sciences published its report on Atmospheric Chemistry: Problems and Scope (NAS 1975). Growing awareness of important influences of acid precipitation on fish populations and potential effects on forest and crop plants led the U. S. Forest Service to sponsor the First International Symposium on Acid Precipitation and the Forest Ecosystem at Columbus, Ohio in May, 1975. The proceedings of this Symposium and the Associated Workshop Report were

published by Dochinger and Seliga (1976a, 1976b). At Congressional hearings in July, 1975, Cowling (1976) testified on the inadequacy of research in the United States on Acid Precipitation and its biological consequences. Specifically, the lack of a coordinated program of research on ecological effects and lack of a stable monitoring network were recognized as primary causes of our profound ignorance of acid precipitation. In the spring of 1976, however, a cadre of scientists in various institutions and agencies throughout the United States began the process of creating the National Atmospheric Deposition Program (NADP) to meet these two critical needs (Kennedy 1977; Galloway and Cowling 1978).

In the fall of 1977, the President's Council on Environmental Quality contracted with the NADP for the drafting of "A National Program for Assessing the Problem of Atmospheric Deposition (Acid Rain)." This publication (Galloway and others 1978) provided the basis for a Presidential Initiative on acid precipitation which President Carter announced on August 2, 1979 in his Second Environmental Message (Carter 1979). This initiative calls for a 10-year long, \$10,000,000 per year program of research on the causes and consequences of acid precipitation. A standing Acid Rain Coordinating Committee was established by the President to plan and manage the program. Leadership for the Committee is provided by co-chairmen from the Department of Agriculture and the Environmental Protection Agency. At the present time, the Acid Rain Coordinating Committee is drafting a coherent program of research on atmospheric chemistry and transport, chemical and biological monitoring, ecological and materials-damage effects, economic assessments, and public-policy options for control of acid precipitation and/or amelioration of its ecological effects.

Wetstone (1980) has recently summarized the biological and materials-damage effects of acid precipitation in relation to the pollution-control laws in North America.

In conclusion, the Presidential Initiative on Acid Precipitation, coupled with growing Congressional, public, and private-industrial interest in acid precipitation research, provide a basis for increasing hope that the United States will do its part, together with Sweden, Norway, England, Canada and other nations, to meet the challenge of continuing economic development with adequate safeguards for the quality of life and the long-term productivity of ecosystems on which our good life critically depends.

LITERATURE CITED

- Abrahamsen, G.
1980. Acid precipitation, plant nutrients and forest growth. *In* Proc. Int. Conf. on Ecological Impacts of Acid Precipitation [Sandefjord, Norway].

- Ambio.
1976. Report of the International Conference on the effects of acid precipitation in Telemark, Norway. Ambio 5:200-252.
- Bolin, B., L. Granat, L. Ingelstam, M. Johannesson, F. Mattsson, S. Oden, H. Rodhe, and C. O. Tamm.
1972. Air pollution across national boundaries. United Nations Conference on the Human Environment. 97 p. P. A. Norstedt and Sons, Stockholm.
- Braekke, F. H., ed.
1976. Impact of acid precipitation on forest and freshwater ecosystems in Norway. Res. Rep. 6: Acid precipitation - effects on forests and fish. Aas, Norway. 111 p.
- Carter, J.
1979. The President's Environmental Program 1979. 57 p. The President's Council on Environmental Quality. Washington, D. C., USA.
- Cogbill, C. V., and G. E. Likens.
1974. Acid precipitation in the northeastern United States. Water Resources Res. 10:1133-1137.
- Cowling, E. B.
1976. Testimony on research and development relating to sulfates in the atmosphere. In Research and development relating to sulfates in the atmosphere - Hearings before the Subcommittee on the Environment and the Atmosphere of the Committee on Science and Technology. U. S. House of Rep., 94th Congress, First Session, p. 398-440. Washington, D. C.
- Cowling, E. B.
1980a. From research to public policy: progress in scientific and public understanding of acid precipitation and its biological effects. In Proc. Action Seminar on Acid Precipitation [Nov. 1-3, 1979, Federation of Ontario Naturalists, Don Mills, Ontario, Canada], 27 p.
- Cowling, E. B.
1980b. An historical resumé of progress in scientific and public understanding of acid precipitation and its biological consequences. In Proc. Int. Conf. on Ecological Impacts of Acid Precipitation [Sandefjord, Norway].
- Cowling, E. B.
1980c. Acid precipitation and its effects on terrestrial and aquatic ecosystems. In Aerosols: anthropogenic and natural, sources and transport. T. J. Kneip and P. J. Liou, eds. Ann. New York Acad. Sci. Volume 338:540-555.
- Dochinger, L. S., and T. A. Seliga.
1976a. Proceedings First International Symposium on Acid Precipitation and the Forest Ecosystem. USDA Forest Service, Gen. Tech. Rep. NE-23, 1079 p. Northeast. For. Exp. Stn., Upper Darby, Penn.
- Dochinger, L. S., and T. A. Seliga.
1976b. Workshop report on acid precipitation and the forest ecosystem. USDA Forest Service. Gen. Tech. Rep. NE-26, 18 p., Northeast. For. Exp. Sta., Upper Darby, Penn.
- Feth, J. H., S. M. Rogers, and C. E. Roberson.
1964. Chemical composition of snow in the northern Sierra Nevada and other areas. Geol. Survey Water Supply Paper 1535-J. 39 p.
- Galloway, J. N., and E. B. Cowling.
1978. The effects of precipitation on aquatic and terrestrial ecosystems: a proposed precipitation chemistry network. J. Air Pollution Control Association 28:229-235.
- Galloway, J. N., E. B. Cowling, E. Gorham, and W. W. McFee.
1978. A national program for assessing the problem of atmospheric deposition (acid rain). 97 p. National Atmospheric Deposition Program, Natural Res. Ecology Laboratory, Fort Collins, Colorado.
- Gambell, A. W., and D. W. Fisher.
1966. Chemical composition of rainfall, eastern North Carolina and southeastern Virginia. Geol. Survey Water Supply. Paper 1535-K. 41 p.
- Gorham, E.
1958. The influence and importance of daily weather conditions upon the supply of chloride, sulphate, and other ions to fresh waters from atmospheric precipitation. Phil. Trans. Royal Soc. Lond. Biol. Sci. 241:147-178.
- Gorham, E.
1976. Acid precipitation and its influence upon aquatic ecosystems - an overview. Water, Air, Soil Pollution 6:457-481.
- Grahn, O., H. Hultberg, and L. Landner.
1974. Oligotrophication - a self-accelerating process in lakes subjected to excessive supply of acid substances. Ambio 3:93-94.
- Heck, W. W., A. S. Heagle, and E. B. Cowling.
1977. Impact of air pollutants on soil, water, flora and fauna: impact on plants. In Proc. Soil Conservation Society of America p. 193-207.
- Howells, G.
1979. Ecological effects of acid precipitation. Report of a workshop held at Galloway, Scotland [September 1978]. Prepared by the Central Electricity Research Laboratories, Leatherhead, Surrey, United Kingdom.
- Hutchinson, T. C., and M. Havas, eds.
1980. Effects of acid precipitation on terrestrial ecosystems. 654 p. Plenum Press, New York.

- Johnsson, B., and R. Sundberg.
1972. Has the acidification by atmospheric pollution caused a growth reduction in Swedish forests? Research Note No. 20. 48 p., Department of Forest Yield Research, Royal College of Forestry, Stockholm, Sweden.
- Kennedy, V. C.
1977. Research and monitoring of precipitation chemistry in the United States - present status and future needs. 75 p., Federal Interagency Work Group on Precipitation Quality. Office of Water Data Coordination. U. S. Geological Survey, Reston, Va.
- Kovda, V. A.
1975. Biogeochemical cycles in nature, their disturbance and study. (In Russian). Kuaka Publishing House, Moscow, USSR.
- Likens, G. E.
1976. Acid rain. Chem. & Eng. News 54(48):29-44.
- Lodge, J. P., J. B. Pate, W. Basbergill, G. S. Swanson, K. C. Hill, E. Lorange, and A. L. Lazrus.
1968. Chemistry of United States Precipitation. 66 p., Final report on the National Precipitation Sampling Network, National Center for Atmospheric Research, Boulder, Colo.
- National Academy of Sciences.
1975. Atmospheric Chemistry: Problems and Scope. 130 p., National Academy of Sciences, Washington, D. C.
- Odén, S.
1968. The acidification of air and precipitation and its consequences in the natural environment. Ecology Committee Bulletin No. 1., 117 p., The State National Science Research Council, Stockholm. Translation Consultants, Ltd., Arlington, Va.
- Smith, R. A.
1872. Air and rain: the beginnings of chemical climatology. 600 p., Longmans, Green, London.
- SNSF.
1980. Final report and bibliography. SNSF Project. As, Norway. In preparation.
- Tamm, C. O.
1958. The atmosphere. Handbuch für Pflanzenphysiologie 5:233-242.
- Tamm, C. O., and E. B. Cowling.
1977. Acidic precipitation and forest vegetation. Water, Air, Soil Pollut. 7: 503-511.
- Wetstone, G.
1980. Air pollution control focus in North America and the problem of acid rain and snow. Environ. Law Reprtr. X(2):50001-50020.
- Wittwer, and Bukovac.
1969. The uptake of nutrients through leaf surfaces. In Handbuch der Pflanzenernährung und Düngung. K. Scharrer and H. Linser, eds. p. 235-261. Springer-Verlag, New York.

Simulation Modeling of the Effects of Chronic Pollutant Stress on Plant Processes and Plant Community Dynamics

Modeling Pollutant Uptake and Effects on the Soil-Plant-Litter System¹

R. J. Luxmoore²

Abstract: Five coupled models of water, carbon, and chemical dynamics in a soil-plant-litter system are outlined. Algorithms defining gaseous and particulate pollutant uptake are described along with functions for chemical effects on plant growth and litter decomposition. Some simulation results of a deciduous forest illustrate the importance of diurnal and annual cycles of environmental conditions on pollutant movement in vegetation. This modeling approach has provided (1) insights into plant physiological processes and their interactions, (2) identification of plant properties important in pollutant uptake, (3) alternative hypotheses about pollutant effects, and (4) a unified basis for assessment of diurnal and long-term pollutant impacts on plant communities.

"and all the king's horses and all the king's men couldn't put Humpty together again."

from Humpty Dumpty, Anon.

The discouraging words of the nursery rhyme suggest that the synthesis of bits of an egg to a whole will not happen at least while horses and men are in charge! Our task of trying to couple together bits and pieces of mechanistic information about the physiology of trees and responses to soil and atmospheric environments is no less awesome a challenge. Simulation modeling is a remarkable tool for meeting this challenge, since through mathematics coupled relationships may be quantified. In this paper,

paper, an outline is presented of five models that link together and provide a framework for study of pollutant uptake and effects in the whole plant environment complex. Some applications are shown and the use of models in analysis of experiments is explored. Lastly, some speculations are presented about pollutant impacts on whole plants and their diurnal metabolism.

MODELING THE SOIL-PLANT-LITTER SYSTEM

The development of a unified approach to the modeling of terrestrial processes has been undertaken at Oak Ridge. Five component models of water, carbon, and chemical dynamics in a soil-plant-litter system were constructed and linked together (Baes et al. 1976). The models (table 1) are deterministic. The flow processes are dependent on gradient terms calculated by the models to provide the flow driving forces and empirical inputs are used to represent pathway resistances or conductivities. Flow directions are not predetermined and the models can be applied to a range of different soil-plant systems (e.g., coniferous, deciduous forest) by changing the empirical properties in the input data. The reader is referred to the documentation reports (table 1) for further details.

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Table 1--Some attributes of coupled models describing carbon, water, and chemical dynamics in the soil-plant-litter system.

ORNL-DWG 78-13073R

COMPONENT	WATER	SOIL EXCHANGE	CARBON	ROOT SOLUTE UPTAKE	SOLUTES
NAME	PROSPER (TEHMI)	SCEHM	CERES	DIFMAS	DRYADS
TIME STEP	15 OR 60 min.	15 OR 60 min.	60 min.	15 OR 60 min.	15 OR 60 min.
ATTRIBUTES	EVAPOTRANSPIRATION BY COMBINATION OF EQUATION. SOIL WATER FLOW BY DARCY FLOW EQUATION. USES EMPIRICAL RELATIONSHIP BETWEEN SURFACE RESISTANCE AND SURFACE WATER POTENTIAL. EMPIRICAL DATA FOR SOIL HYDRAULIC PROPERTIES.	USES EMPIRICAL DISTRIBUTION COEFFICIENT (Kd) FOR SOIL OF INTEREST. TRANSLOCATION EQUATION FOR POTENTIAL GROWTH OF LEAF, STEM, ROOT, FRUIT. EMPIRICAL LITTER DECOMPOSITION RELATIONSHIPS.	CO ₂ DIFFUSION EQUATION FOR NET PHOTOSYNTHESIS. SUBSTRATE GRADIENT EQUATION FOR TRANSLOCATION. USES INPUT VALUES FOR POTENTIAL GROWTH OF LEAF, STEM, ROOT, FRUIT. EMPIRICAL LITTER DECOMPOSITION RELATIONSHIPS.	IMPLEMENTS MODEL OF DIFFUSION AND MASS FLOW OF SOLUTES TO ROOTS BY BALOWIN, NYE AND TINKER (1973). GRADIENT EQUATION FOR PHLOEM TRANSLOCATION. TRANSPORT FLUX USED FOR XYLEM TRANSPORT. PLANT CEMANO FUNCTION DETERMINED BY POTENTIAL SOLUTE CONCENTRATION INPUT VALUES.	SOLUTE UPTAKE BY ROOTS AND LEAVES. DIFFUSIVE GAS UPTAKE BY LEAVES. GRADIENT EQUATION FOR PHLOEM TRANSLOCATION. TRANSPORT FLUX USED FOR XYLEM TRANSPORT. PLANT CEMANO FUNCTION DETERMINED BY POTENTIAL SOLUTE CONCENTRATION INPUT VALUES.

REFERENCE HUFF et al. (1977) BEGOVICH AND JACKSON (1975) DIXON et al. (1978) LUXMOORE et al. (1978) LUXMOORE et al. (1978)

The coupling between models (fig. 1) shows that every model has information transfer with at least two other models, and these take place on either an hourly time step or every 15 minutes during storm events. Hourly values of stomatal resistance and plant water potential from PROSPER are used in CERES to determine photosynthesis and growth respectively. Leaf and root growth in turn influence transpiration and thus soil water flow. During rainfall, infiltration and the movement of water between soil layers (calculated in PROSPER) is used in the soil chemistry model (SCEHM) to calculate chemical fluxes. Chemical concentration and root water uptake information are used in DIFMAS to calculate chemical uptake into root by diffusion and mass flow. Chemicals within the plant are moved up in the transpiration stream and down in the phloem pathway.

This set of models can be run for simulation periods of several years and annual budgets for water, carbon and chemicals can be evaluated as well as detailed results for hourly periods of interest. The algorithms defining gaseous and particulate pollutant uptake and effects on plant growth and litter decomposition are outlined in the next two sections along with example simulation results.

AIR POLLUTANT UPTAKE

The uptake of air pollutants by vegetation may occur directly through leaves (gaseous and particulate) or indirectly through roots after the pollutants have been incorporated into soil. Gaseous uptake is represented by a diffusion equation (same form as the photosynthesis equation). Thus

$$U_g = \frac{(g_e - g_i) * g_d}{r_a + r_s + r_m} ,$$

where g_e is the external pollutant concentration (ml/ml)

g_i is the internal pollutant concentration (ml/ml)

g_e is gas density ($\mu\text{g/ml}$)

r_a is boundary layer diffusion resistance (sec/cm)

r_s is stomatal resistance (sec/cm)

r_m is mesophyll resistance (sec/cm)

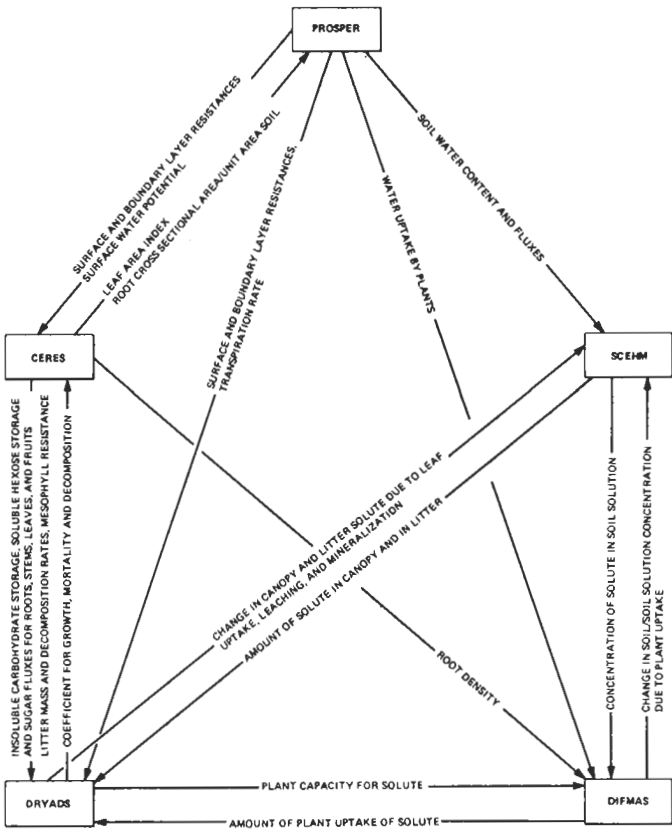
U_g is uptake ($\mu\text{g/cm}^2$ leaf/sec)

The value of g_i is made to vary between zero and g_e depending on the level of pollutant in leaf storage (E_i) as follows,

$$g_i = g_e * (1 - \frac{E_m - E_i}{E_m}) .$$

E_m is the maximum allowable level of pollutant in leaf storage, an input parameter. Operationally this is the pollutant level at which the leaf tissue becomes necrotic.

ORNL-DWG 75-15812R2



- PROSPER soil-plant-atmosphere water flow model
- CERES carbon dynamics of vegetation and litter
- SCEHM soil chemistry model
- DIFMAS diffusion and mass flow of chemicals to roots
- DRYADS chemical dynamics of vegetation and litter

Figure 1--Coupling of five process models that describe hourly carbon, water, and solute dynamics of the soil-plant-litter system.

Sulfur dioxide uptake by an oak-hickory forest in the vicinity of a lead mining and smelter complex in southeastern Missouri was simulated and results illustrate the behavior of the model. Cumulative sulfur levels in leaves (fig. 2) show a rapid increase on the 25th of August, a day in which the atmospheric SO₂ level was increased 10 fold above ambient. The translocation of sulfur from leaf to stem (fig. 2) clearly shows a diurnal pattern and at elevated rates on the 25th of August. Some of the sulfur material that was transported to the roots, leaked into the transpiration stream and returned from the roots to the stem, albeit in trace amounts. The phloem and xylem transport pathways can allow considerable mobility of solutes between plant tissues according to the simulation. The cumulative sulfur levels in the leaf, stem and root components (fig. 3) show that the majority of sulfur remained in the leaves. The value of $8 \times 10^5 \mu\text{g S/m}^2$ is equivalent to a leaf concentration of 180 ppm.

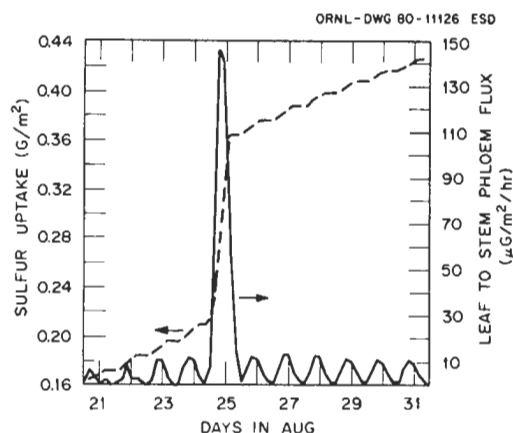


Figure 2--Simulated cumulative sulfur dioxide uptake by vegetation (g/m^2) and leaf to stem phloem translocation rate ($\mu\text{g/m}^2/\text{hr}$) for 11 days in August.

The uptake of pollutants from particulates deposited on leaves (U_l) is represented by a gradient equation using empirical input values for the cuticular conductivity (k_l) and thickness (W). Thus,

$$U_l = k_l \frac{(S_e - S_i)}{W},$$

where S_e is the external pollutant on leaf surface (g/m^2 land)

S_i is the internal pollutant within foliage (g/m^2 land)

The amount of dissolved pollutant on leaf surfaces is calculated as the lesser of either the product of solubility and the water volume on leaves (interception) or the current amount of pollutant on leaves. The soluble pollutant within leaves (S_i) is assumed to be uniformly

distributed and has one of two fates. It may be transported to other plant parts or be incorporated in the leaf in an immobile form. The cuticular uptake process is considered reversible in the model. Thus during rainfall, wash-off occurs and if S_e becomes less than S_i , then leaching of pollutant out of leaves will occur.

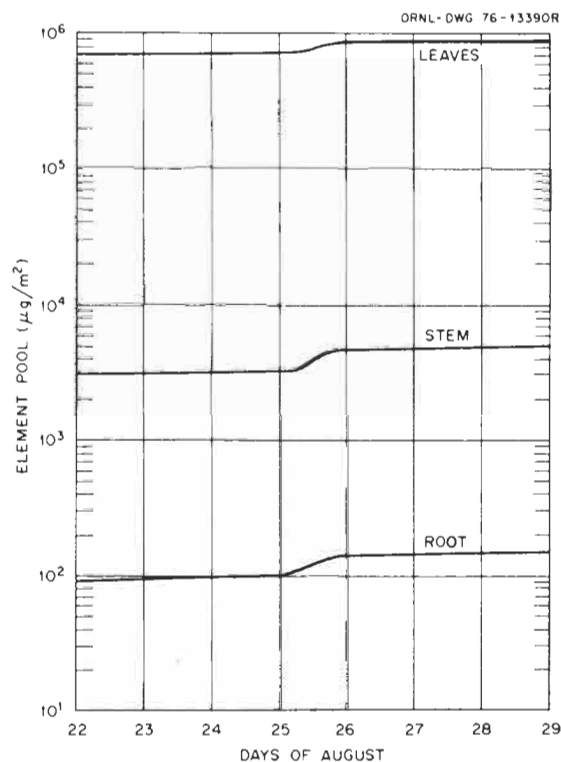


Figure 3--Simulated sulfur elemental accumulation in leaf, stem, and root tissue ($\mu\text{g/m}^2$) resulting from gaseous uptake.

Sensitivity analysis of the leaf cuticle conductivity (figs. 4a,b) shows that greater conductivity is associated with greater chemical (zinc in the example) uptake by leaves and a slightly reduced uptake of zinc from the soil solution (Begovich and Luxmoore 1979). This latter and more subtle effect is induced by the higher zinc level in the plant with higher conductivity which feeds back a reduced chemical demand in the root uptake algorithm. It is possible that subtle effects may become significant when integrated over long time periods. Cuticular conductivity and the equivalent property at the root-soil interface (root conductivity, k_r) were shown to be very sensitive parameters in the model, and yet these are perhaps the least well characterized experimentally. Results from a sensitivity analysis of root conductivity on lead uptake (table 2) show large increases in uptake by roots and lead concentration in tree tissues with increase in k_r from 10^{-10} cm/sec to 10^{-6} cm/sec. The simulations also show that pollutants accumulate preferentially in the leaf and root, the sites of pollutant entry. A modification has subsequently been added to the model to allow chelation of chemical within the

plant (Luxmoore and Begovich 1979) which has the effect of increasing the mobility of pollutant within the plant. Thus, the site of pollutant entry may not be the site of accumulation.

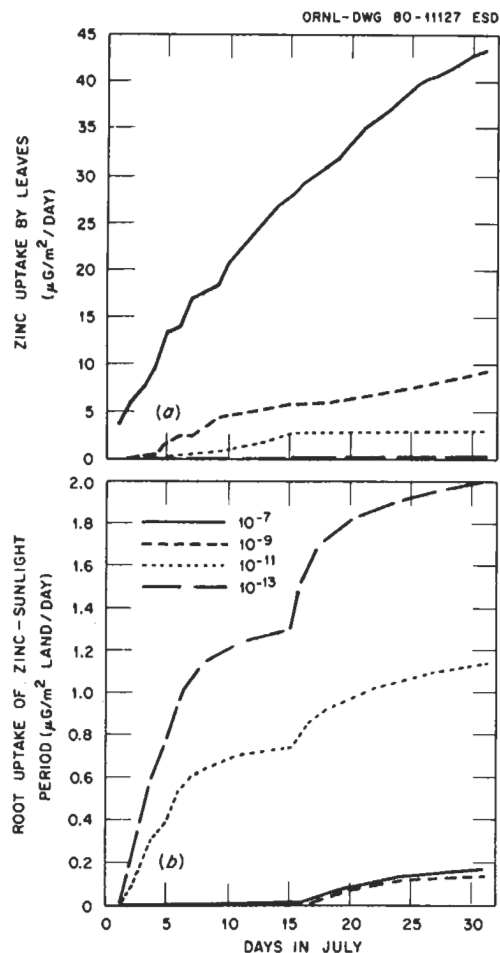


Figure 4--a. Influence of leaf cuticle permeability on zinc uptake by leaves.

b. Influence of leaf cuticle permeability on zinc uptake by roots.

The monthly pattern of lead uptake by roots and foliage simulated for an oak forest near a mine-smelter complex during the first year of operation shows that uptake corresponds with the growing season (table 3). The major proportion (88%) of root uptake occurred during the day chiefly due to two complementary transportation processes; the mass flow of pollutant to roots and mass flow of pollutant from roots to shoots. The latter was the controlling process in the simulations. Overall, leaf uptake was more than double that simulated for roots for the first year of smelter operation.

Table 3--Simulated root and leaf uptake, (mg Pb/m² land/month) of lead by oak vegetation in the vicinity of a mine-smelter complex.

Month	Root		Leaf
	Day	Night	
Jan.	0	0	0
Feb.	0	0	0
March	57	4	250
April	213	13	410
May	243	28	890
June	196	45	1190
July	286	41	1130
Aug.	285	37	890
Sept.	315	40	420
Oct.	226	38	20
Nov.	0	0	0
Dec.	0	0	0
Total	1821	246	5200

Table 2--Sensitivity of annual root lead uptake and tissue concentration (prior to leaf fall) in an oak forest to change in the root solute conductivity parameter (k_r).

k_r (cm/s)	Annual root uptake ($\mu\text{g}/\text{cm}^2/\text{year}$)	September tissue concentration (ppm)					
		Leaf	Stem		Root		Fruit
			Sapwood	Heartwood	Sapwood	Heartwood	
10^{-4}	156.1	309	854.0	3.30	967	10.2	264.0
10^{-5}	143.3	258	774.0	2.80	961	9.9	191.0
10^{-6}	128.0	242	683.0	2.32	993	11.7	251.0
10^{-8}	20.7	113	5.06	0.01	432	3.1	1.2
10^{-10}	0.52	112	0.48	0.003	11	0.1	0.1

Pollutant Impacts

Simple ramp functions are used to determine pollutant effects on the growth and decomposition of leaf, stem, root and fruit components. Separate ramp functions for either growth effects (fig. 5) or control of decomposition in the litter (same form as for growth effects) represent ranges of chemical deficiency, sufficiency, and toxicity as the chemical concentration increases. Hypotheses concerning beneficial (fertilizer) and toxic pollutant effects can thus be examined. The product of the growth coefficient and tissue growth rate (from CERES) provides a modified growth rate due to pollutant effects.

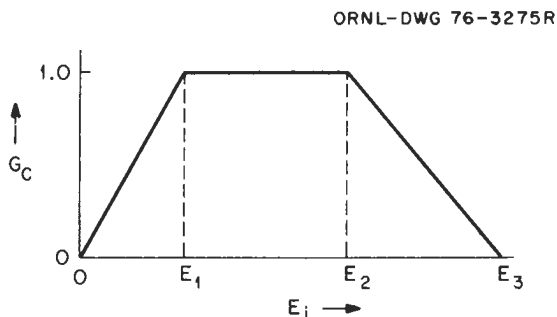


Figure 5--The relationship between the growth coefficient (G_C) and the amount of element in tissue (E_i) used to represent deficiency ($E_i < E_1$), sufficiency ($E_1 < E_i < E_2$), and toxicity ($E_i > E_2$) effects of the elements on tissue growth rate.

A six-year simulation of heavy metal deposition, transport, and uptake in an oak-hickory forest in southeastern Missouri showed that the lead accumulation was greatest in the litter (Luxmoore et al. 1978). Root uptake of lead increased through the six-year period, whereas leaf uptake was a constant for the repetitive annual deposition of 25 g Pb/m². Due to the buildup of lead in the plant tissues, the mortality of plant parts returned increasingly greater amounts of lead to the litter system. The litter dry weight increased through the six-year period by 949 g/m². This compares reasonably with a difference of 1130 g/m² between the litter mass at a control site and a site exposed to equivalent heavy metal deposition (Watson et al. 1976). The simulation results pose an alternative hypothesis to the experimental inference of reduced rates of litter decomposition at the elevated levels of heavy metal accumulation (Jackson and Watson 1977), by showing that the same effect could be obtained with increased mortality of plant parts.

Next Step

The previous sections outline one particular set of models and show some simulation results including sensitivity analysis of selected

parameters. The work presented is best viewed as "equipment"; the subroutines being component parts which collectively form a package of hypotheses, theories, or knowledge in mathematical form. We need to thoroughly test models through applications to experimental studies as much as possible to, hopefully, invalidate parts of the model structure. The deviations of model predictions from experimental findings provide the key to new insights - in this way models facilitate the analysis and synthesis of complex interactions. Putting models to work in this way requires data from well-documented experiments. For example, the uptake and physiological effects of gaseous pollutants have been documented for several tree species (Jensen and Kozlowski 1975, Thompson et al. 1967, Roberts 1974, Lawhon 1973, Houston and Stairs 1973), and these experimental data can be used in leaf physiological models (Kercher 1977) or in the models outlined in the earlier sections. A considerable body of experimental data has been developed for air pollutant effects on plants, and it is timely to apply modeling techniques in the research analysis of impacts. An alternative approach is one of conceptual extrapolation of the model behavior. Some speculations are presented in the next section.

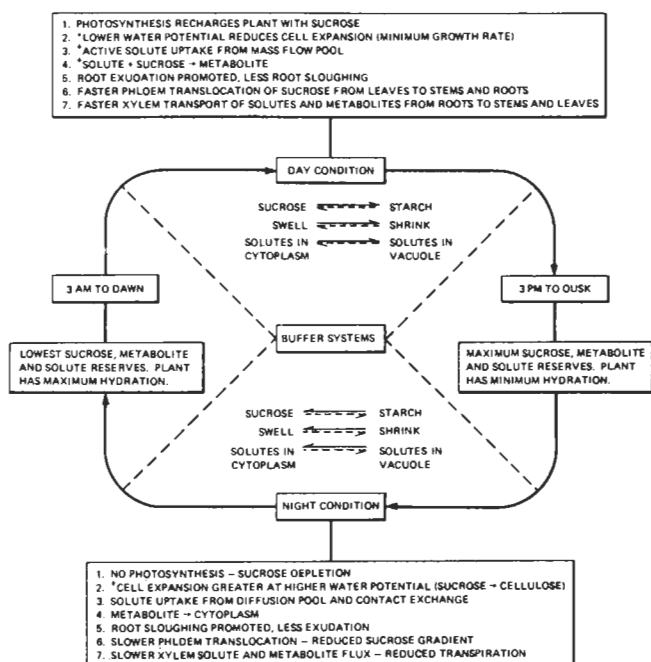
Pollutants and the Diurnal Cycle

The modeling of water, carbon, and chemicals as coupled components in soil-plant-litter systems has stimulated the development of a conceptual framework for the diurnal cycle in plants (fig. 6) that can be used to invent hypotheses of pollutant effects on whole plants. In the diurnal cycle, plants change between two relative states: (a) lowest sucrose, metabolite, and solute reserves at maximum hydration (dawn state); and (b) highest sucrose, metabolite, and solute reserves at minimum hydration (dusk state). These states are relative and apply to a given day. Photosynthesis recharges the plant with sucrose and increases starch storage (or equivalent) during the day. At the same time, the plant is also recharging with nutrients and undergoing dehydration. The loss of water can reduce the rate of cell expansion processes during the day with greater growth being favored with rehydration. Thus plants may need to solve a timing imbalance between carbon gain and utilization by changes in internal storage. The higher internal carbon status of leaves during the afternoon may reduce the significance of pollutant impacts on leaves during this part of the day. Photosynthesis may be already slowed by product accumulation, or alternatively detoxification mechanisms using readily available carbon metabolites and/or energy may more easily cope with pollutant insult than during early morning when internal carbon status is lower.

The diurnal pattern of behavior (fig. 6) also suggests that root exudation of carbon compounds could be facilitated during the day. In the

same way, the carbon supply to mycorrhizae and root nodules may be facilitated. Disruption of these processes through the impact of air pollutants may be of great importance to understanding whole plant responses. Pollutant stress that causes reduced photosynthesis and/or greater respiration in forest ecosystems may decrease the carbon leakage to mycorrhizal associations with roots, potentially decreasing the extent and efficiency of the fungi in supplying nutrients back to the tree. Thus, it may be further hypothesized that phytotoxic air pollutants may cause forest ecosystems to be less efficient in nutrient retention (i.e., become more leaky, see also O'Neill et al. 1977) and conversely beneficial air pollutants may increase nutrient retention of forest ecosystems. Elevated atmospheric CO₂ levels may be an example of the latter.

ORNL DWG 75-3870R



*DOMINANT ENERGY UTILIZATION FOR TIME PERIOD
 *INDUCED BY ROOT RESISTANCE TO TRANSPIRATION

Figure 6--Diurnal pattern of carbon, water, and solute dynamics showing relative tendencies and relative states in vegetation.

Perhaps, like Humpty Dumpty, these attempts at deriving whole system understanding from the pieces involved shows many cracks and flaws. Nevertheless, we give it a go! The key test is our answer to the question "Did we learn something that we didn't know before?"

SUMMARY

Modeling of pollutant interactions with whole plant processes has provided:

1. Insights about the processes and their interrelationships, e.g.,
 - (a) Transpiration may facilitate pollutant uptake by transporting chemical from roots to stem thus maintaining a favorable chemical gradient for continuing uptake.
 - (b) Phloem and xylem may provide ready transport pathways for pollutant movement between plant parts (fig. 3).
2. Identification of plant properties important in pollutant uptake. In particular, leaf and root chemical conductivity have great influence on pollutant uptake (fig. 5, table 2).
3. Alternative hypotheses, e.g., increased forest litter in areas polluted with heavy metals could be due to increased mortality of plant parts in addition to reduced decomposition rate.
4. A basis for short-term (diurnal) and long-term speculation or pollutant impacts, e.g.,
 - (a) Hourly changes in water, carbon, and nutrient status of plants may influence physiological sensitivity to pollutant insult.
 - (b) Pollutant disruption of carbon allocation to belowground processes may have long-term nutrient cycling impacts.

LITERATURE CITED

- Baes, C. F., C. L. Begovich, W. M. Culkowski, K. R. Dixon, D. E. Fields, J. T. Holdeman, D. D. Huff, D. R. Jackson, N. M. Larson, R. J. Luxmoore, J. K. Munro, M. R. Patterson, R. J. Raridon, M. Reeves, D. C. Stein, J. L. Stolzy, and T. C. Tucker.
 1976. The unified transport model. In Ecology and analysis of trace contaminants progress report October 1974-December 1975. R. I. Van Hook and W. D. Shults, eds., pp. 13-62. ORNL/NSF/EATC-22. Oak Ridge National Laboratory, Oak Ridge, Tennessee 37830. 200 pp.
- Baldwin, J. P., P. B. Nye, and P. B. Tinker.
 1973. Uptake of solutes by multiple root systems from soil, III. A model for calculating the solute uptake by a randomly dispersed root system developing in a finite volume of soil. Plant Soil 38:621-635.
- Begovich, C. L., and D. R. Jackson.
 1975. Documentation and application of SCEHM. A model of soil chemical exchange of heavy metals. ORNL/NSF/EATC-16. Oak Ridge National Laboratory, Oak Ridge, Tennessee. 67 pp.

- Begovich, C. L., and R. J. Luxmoore.
1979. Some sensitivity studies of chemical transport simulated in models of the soil-plant-litter system. ORNL/TM-6791. Oak Ridge National Laboratory, Oak Ridge, Tennessee. 97 pp.
- Dixon, K. R., R. J. Luxmoore, and C. L. Begovich.
1978. CERES - A model of forest stand biomass dynamics for predicting trace contaminant, nutrient and water effects. I. Model description, II. Model documentation. Ecol. Model. 5:17-38, 93-114.
- Houston, D. B., and G. R. Stairs.
1973. Genetic control of sulfur dioxide and ozone tolerance in eastern white pine. For. Sci. 19:267-271.
- Huff, D. D., R. J. Luxmoore, J. B. Mankin, and C. L. Begovich.
1977. TEHM. A terrestrial ecosystem hydrology model. ORNL/NSF/EATC-27. Oak Ridge National Laboratory, Oak Ridge, Tennessee. 153 pp.
- Jackson, D. R., and A. P. Watson.
1977. Description of nutrient pools and transport of heavy metals in a forested watershed near a lead smelter. J. Environ. Qual. 6:331-338.
- Jensen, K. F., and T. T. Kozlowski.
1975. Absorption and translocation of sulfur dioxide by seedlings of four forest tree species. J. Environ. Qual. 4:379-382.
- Kercher, J. R.
1977. GROW1: A crop growth model for assessing impacts of gaseous pollutants from geothermal technologies. UCRL-52247. Lawrence Livermore Laboratory, California.
- Lawhon, W. T.
1973. Radial growth and wood density of white pine in relation to coal-derived environmental pollutants. Ph.D. dissertation. Graduate program in ecology, University of Tennessee, Knoxville, Tennessee. 110 pp.
- Luxmoore, R. J., and C. L. Begovich.
1979. Simulated heavy metal fluxes in tree microcosms and a deciduous forest. Internat. Soc. Ecology. Modelling. J. 1:48-60.
- Luxmoore, R. J., C. L. Begovich, and K. R. Dixon.
1978. Modeling solute uptake and incorporation into vegetation and litter. Ecol. Model. 5:137-171.
- O'Neill, R. V., B. S. Ausmus, D. R. Jackson, R. I. Van Hook, P. Van Voris, C. Washburne, and A. P. Watson.
1977. Monitoring terrestrial ecosystems by analysis of nutrient export. Water, Air, and Soil Pollut. 8:271-277.
- Roberts, B. R.
1974. Foliar sorption of atmospheric sulfur dioxide by woody plants. Environ. Pollut. 7:133-140.
- Thompson, C. R., D. C. Taylor, M. D. Thomas, and J. O. Ivie.
1967. Effects of air pollutants on apparent photosynthesis and water use by citrus trees. Environ. Sci. Technol. 1:664-650.
- Watson, A. P., R. I. Van Hook, D. R. Jackson, and D. E. Reichle.
1976. Impact of a lead mining-smelting complex on the forest floor litter arthropod fauna in the New Lead Belt region of southeast Missouri. ORNL/NSF/EATC-30. Oak Ridge National Laboratory, Oak Ridge, Tennessee. 163 pp.

Data-Based Ecological Modeling of Ozone Air Pollution Effects in a Southern California Mixed Conifer Ecosystem¹

Ronald N. Kickert and Barbara Gemmill²

Abstract: The purpose of this research was to determine the effects of ozone air pollution on a mixed conifer forest ecosystem in the San Bernardino National Forest, California.

We used an ecological systems modeling approach in concert with various biological specialists. This required conceptual model development, computer programming, and the analysis of original project data for model calibration.

We found that this process led to the investigators conducting new research of an integrative nature. A structure for complex interactions of forest effects was produced. Insights on changes in ecosystem dynamics and a worst-case scenario of future forest changes were derived.

We conclude that sudden qualitative changes in conifer forest composition can occur under the influence of ozone air pollution and the exclusion of natural fire events.

If it were known that air pollutants did not affect people and their environments, society would be likely to have little interest in those pollutants. The central issue is "What are the effects?"

INSTITUTIONAL SETTING

In the United States, National Ambient Air Quality Standards for ozone have been legally established with a view for effects on humans, the primary standard, and separately for the effects

on the biological, ecological, and physical environment, the secondary standard. Recently, the standards were raised from 0.03 to 0.12 ppm for one hour per year (U.S. Environmental Protection Agency 1979). In view of the fact that knowledge of pollutant effects continues to develop, the criteria for justifying the legal standard is expected to be re-evaluated every few years.

THE PROBLEM

In evaluating criteria for deciding upon the secondary standard for ozone, it has been recognized throughout the 1970's that biological and ecological effects information was biased toward the more reductionistic levels, i.e., biochemistry, plant science, plant physiology, and, because of logistical problems with larger spatial and time scales, biased against, or at least failing to consider, effects on "natural" ecological systems in the landscape. Biological effects criteria have been based on data for individual organisms, but the direct and indirect effects on plant and animal communities have been mostly speculative (U.S. Environmental Protection Agency 1978).

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History of The Study

Objectives of the Entire Project

In 1973, the EPA established a several-year study of oxidant effects on the mixed conifer ecosystem in the San Bernardino National Forest. The interpretations derived to date from this project have constituted the major source of information for the ecosystem chapter on air quality criteria published by the EPA; however, the majority of the data integration remains to be completed. As a further focus of the project, two years after it was initiated, the senior author was brought in to introduce computer simulation as a tool in guiding the collection, integration and interpretation of data on forest responses to oxidant stress. The potential use of this study in future policy-making required an emphasis in the modeling effort particularly on long-term effects, projected effects at different theoretical levels of oxidant flux, and effects on the behavior of the natural community as opposed to individual organisms.

Objectives of the Modeling Activity

Structural Simplification--The goal of the ecosystem modeling effort is twofold, and equally diverse in each direction. Due to the nature of the SBNF project, it was required that the modeler begin with a localized, real-world situation and make extensive use of the large data-base in constructing the model. The real world situation, from which the data are derived is extremely variable, consisting of an east-west trending mountain range which increases in elevation and changes in species composition along the same gradient of oxidant flux, such that essentially no control areas are possible. Given such a complex system, the first goal of the modeling activity was to break down this system structurally into its simplified, basic components and driving factors.

Experiments and Model Behavior-- The other half of this goal was to provide answers to the question: how might one use a simulation model for computer experiments to assess the totality of these effects, acting alone or synergistically, on ecosystem structure and function? A list of effects does not help policy makers very much when they are in the position of making decisions in the face of uncertainty - even less does it inform biologically knowledgeable people who rightfully suspect that interactions occur between items on the list that will affect future outcomes as much or more than a summary of simple effects could ever express. Thus, the modeling effort has been developed to address the following questions:

The EPA/SBNF Project has attempted to establish effects of ozone air pollution on tree stem growth, foliar injury, tree mortality, regeneration, cone production, nutrient cycling, and insect and disease occurrence. What is the consequence of these effects when combined together in a simulated ecosystem? What time scale is necessary to use to see the

full long-term ecosystem effects? Is there potential in this system for sudden jumps and irreversible trends?

The modeling methods and philosophy used in the project have been described in previous publications (Kickert 1977a, 1977b, 1980).

RESULTS

Because the modeling activity is still being conducted, the results presented here are not based on experiments performed on the computer using the models. Rather, they are based on insights gained during the model development process, from conceptualization, to mathematical formulation, to computer coding, and analysis of original data toward the goal of calibrating the models for tree species and sites within the SBNF and then applying those models in experiments of air pollution effects.

How the Systems Modeling Process Aided the Project

There are two ways in which the total study was improved. Model development aided the project investigators in viewing their own work as a part of an integrated conceptual structure. Also, with the design of a graphic model of various subsystems, discussions with investigators led to the identification of questions subsequently turned into research which otherwise would not have been done. A mixed tree species population dynamics approach led to seedling establishment experiments, study plot seedling regeneration surveys, and a comprehensive pest damage inventory, to determine mortality patterns. Data needed for calibrating a stand moisture model led to a seismograph survey for plot soil depths which indicated soil water monitoring profiles were too shallow on several plots. Information required for root disease and bark beetle dynamics led to more cautious use of the smog injury scoring procedure, as well as to the dendrochronological analysis of tree radial growth.

A Structure for Complex Interactions

The effects of air pollution in the forest ecosystem are not only the direct visible effects that a casual observer might notice by discolored foliage on the trees, but also less apparent, but nonetheless real, indirect effects that are subtly transferred through the system. Such indirect chain reactions can occur at the level of individual trees, at the population level of trees of a certain species, and because of changes in the mixtures of the latter over the long-term, changes which occur at the whole community level. As a map of how such changes can be transferred throughout the system, figure 1 displays some significant portions of a forest ecosystem which must be considered. The reference numbers associated with each component in this diagram pertain to various kinds of environmental conditions and biological organisms important to understanding changes occurring in a forest ecosystem. These numbers also reference particular

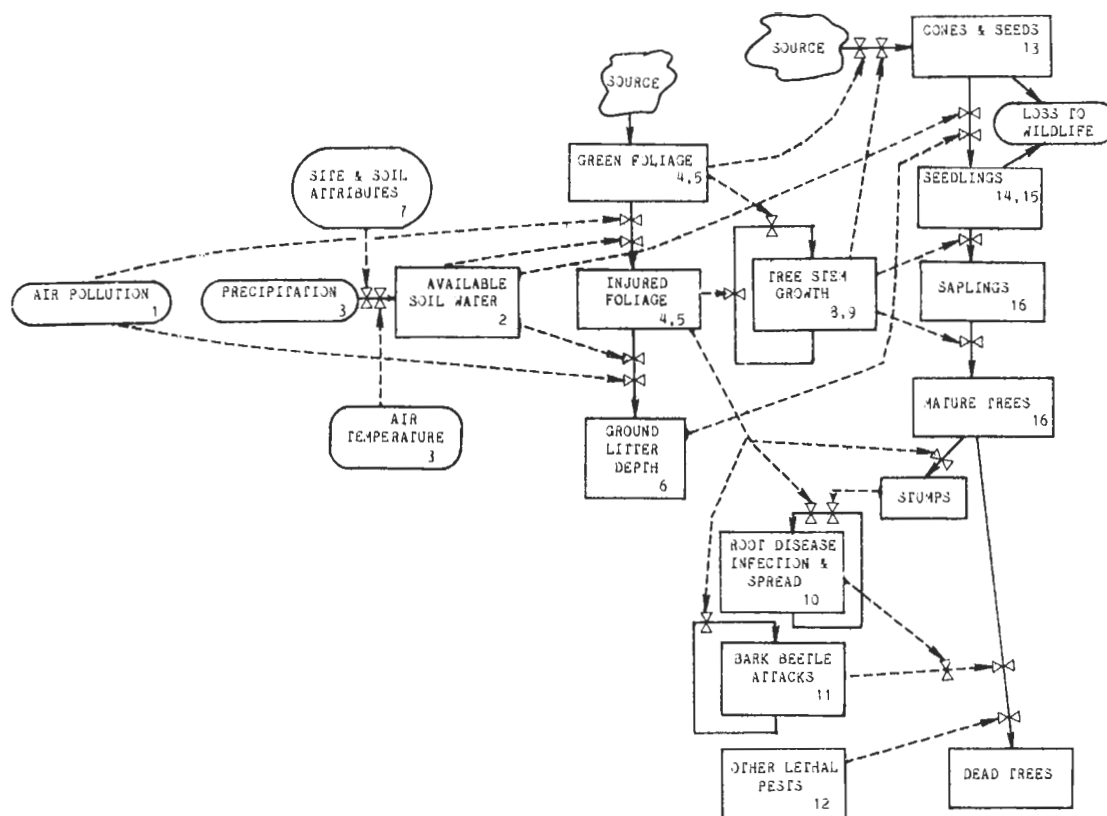


Figure 1--Components of the forest ecosystem directly and indirectly affected by photochemical air pollution.

research topics which have been studied between 1973 through 1980 in the San Bernardino National Forest.

The purpose of this overview is to present an integrated, simplified frame of reference within which the discoveries, results, and conclusions from the project may be viewed as a whole.

While a forest is more than simply a group of trees, the latter is by definition the dominant life form of such a system. Reference will be made to the numbers in various parts of figure 1. To view the effect of air pollution (1) on a community of trees, it is necessary to consider several relatively static site and soil properties (7), as well as very dynamic meteorological conditions such as air temperature and precipitation (3), since all of these may contribute to a synergistic effect of a long-term, chronic air pollution exposure in terms of tree response. Some of the precipitation (3), depending on site and soil characteristics (7), enters the soil as available soil water (2) for tree growth (3,9). For lack of better data, we envision air temperature (3) as a rough index of heat available for enabling available soil water to be depleted through water lost from tree leaves to the atmosphere through transpiration.

Over time, and depending on sensitivity between and within various tree species, some of the green foliage (4,5) on trees becomes injured, discolored foliage (4,5), and some of that is dropped from the trees. This, added to normal amounts of needle shed affected by the availability of soil moisture (2), becomes a part of ground litter (6).

The relative amount of foliage that changes from green to injured is thought to have a bearing on the rate of stem wood growth (8,9) of trees. These three responses are thought to be associated with the rate of production of cones and therefore seeds (13) for regeneration of new trees. As pollutants lead to a greater degree of foliage injury for some tree species, and stem growth is reduced, otherwise mature individuals of these species produce less and less cones, if any.

It has been mentioned how air pollution can increase the ground litter depths (6). This is significant because seeds (13) of some tree species have a biological behavior which is adapted to little or no ground litter for sprouting and surviving as seedlings (14,15) during dry summers (the effect of available soil water (2) once again). Many cones, seeds, and small seedlings are lost to wildlife of various forms under natural conditions. Any further reduction in this reproduction chain because of air pollution effects can make continued replacement of some tree species a very precarious circumstance.

Those seedlings that do survive the first few years eventually grow to a larger size often called saplings (16) in the population structure. The extent to which the effect of air pollution retards stem growth (8,9) simply tends to keep trees in this size range for a longer time, subject to the many causes of death which can occur. Eventually, some saplings grow into larger sizes which are mature (16) and potentially capable of producing cones, as well as being valued for esthetic purposes and as

potentially merchantable timber.

Those mature trees that develop a significant degree of visible foliage injury (4,5) are sometimes cut down whether legally or by poaching. This produces stumps. It has been discovered that the degree of foliage injury (4,5), namely the younger the age of the oldest needles, is significantly correlated with the incidence of infection and colonization of such stumps by the root rot disease (10) *Fomes annosus*, if the fresh stumps are not treated when the live tree is first cut. One would hardly be concerned about this if it were not for the observation that such diseases can spread from dead stump root systems into adjacent live tree root systems, from seedlings, up to large, mature, pollution-resistant trees, depending on the spatial density of the stand. When this does happen with the latter, and those trees already have a considerable degree of injured foliage (4,5), then, especially on ponderosa pine, bark beetles (11) appear better able to successfully attack and kill such trees. Mortality surveys show numerous other patterns of disease and insect combinations (12) also act to kill trees. Reduced growth cannot be sustained indefinitely, and the length of time of reduced growth prior to death appears to be age-dependent.

Over time, the complexion of the forest ecosystem will change according to which tree species are best able to resist the agents that cause death (16), whether natural or human-caused, and are also capable of providing new young seedlings (14,15) able to survive to maturity. In many cases, the evidence seems to indicate that the balance between various tree species is shifting dramatically in the San Bernardino National Forest.

Given three direct effects of air pollutants on forest growth, namely foliar injury and accelerated needle cast, woody growth reduction, and indirectly increased mortality, these effects might be tied together in the following scenarios of ecosystem-level effects.

Ecosystem Dynamics Under a Single Stress

Foliar Injury Consequences-- Foliar injury and premature shedding of past year's needles will slow down the natural development of canopy closure in a stand. It has been shown and repeatedly confirmed that as a stand grows, leaf area expands until it reaches a plateau, at which it remains for the remainder of the life of the stand (Grier and others 1978) (fig. 2). Stand growth, developmental patterns and time to maturity are entirely dependent upon the rate of canopy closure. Net production by coniferous forests is related to leaf area (Whittaker and Niering 1975), and all other things being equal, the greater this leaf area, the greater is the productivity. Once maximal leaf area (canopy closure) is obtained, other ecosystem functions begin to make major qualitative changes, as discussed below. If time to canopy closure is increased, not only are productivity rates reduced, but also qualitative ecological changes may occur.

Since forest foliage always tends toward forming a continuous, complete surface area, the degree of completion represents the degree of occupancy of the site. A vigorous, healthy stand of trees will

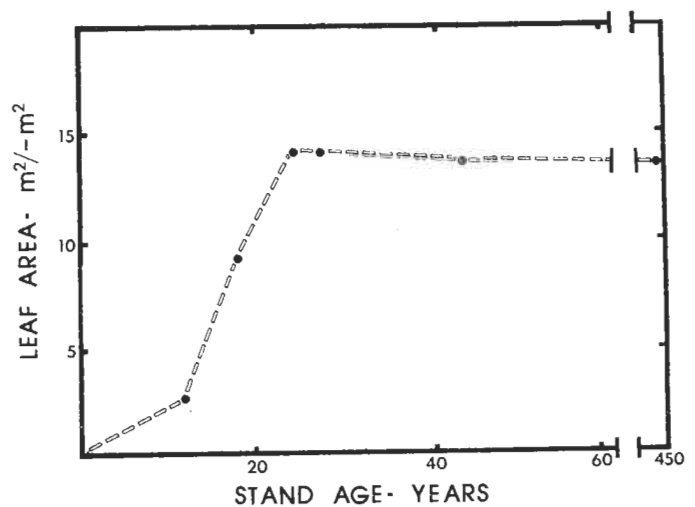


Figure 2--Natural forest stand canopy closure.

achieve occupancy of an open site at about 12 years of age, establishing dominance over competing vegetation, and will maintain occupancy until at least middle age (Smith 1962). A given generation of trees ultimately loses command of the site, giving way to younger members and/or other species. A forest canopy experiencing pollution-caused injury might not be able to fully establish occupancy of the site. The amount of accompanying vegetation, especially of an understory nature, might indicate the degree to which the main stand falls short of full occupancy.

Growth Reduction Consequences-- Leaf area is directly related to woody production in a forest stand, as mentioned. Impediments to leaf area expansion and canopy closure might affect the overall pattern of woody growth in a stand. Studies have shown that second-year needles are more important in providing photosynthate for stem growth, while current year needles contribute primarily to shoot and needle elongation (Walker and others 1972); the second year needles are the most impacted by air pollution. Trees might continue to put on height growth at the expense of diameter growth for a longer period of time than with a normally closing canopy, both because of a lack of photosynthate for stem growth and because the open canopy fosters rapid height growth rates. However, because growth and productivity is suppressed, both height and diameter growth rates in general would be much slower than normal.

It is a tenet of silviculture that changes in stand density do not significantly alter the total amount of dry matter or stem wood produced by a stand. Thus, precommercial thinnings do not markedly change production rates but rather add the same amount of wood to a lesser number of trees (Smith 1962). Mathematically, mean plant size multiplied by density tends toward a constant. This relationship has been confirmed to hold true for many vegetation types, including forest stands (Cooper 1961). In a pollution stressed forest, however, one might expect the relationship either to be weak, or to break down altogether (fig. 3). At low densi-

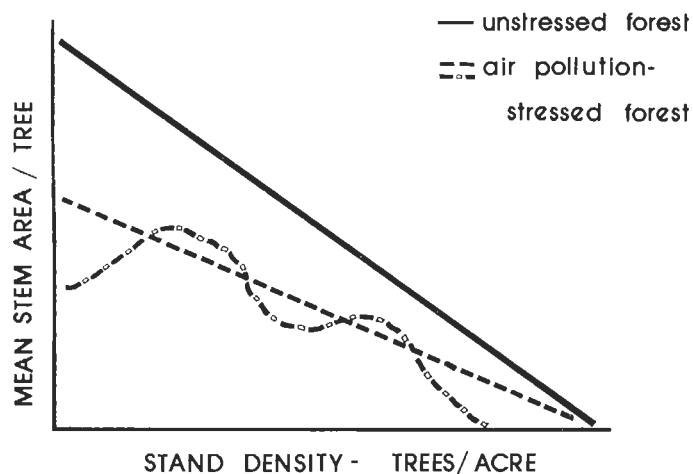


Figure 3--Relation between stand density and average tree size with and without air pollution.

ties, mean plant size is not remarkably large as the trees are not in good vigor; at high densities, plant size is suppressed even more than would be expected due to density effects alone. In especially damaged and open stands, the relationship might be expected to break down. The functioning of the relationship is dependent upon trees exerting an influence over each other. If all trees are generally weakened in competitive ability, their growth will depend more on limitations imposed by the physical environment and less on inter-tree influences. An open stand with highly variable tree sizes might be the result.

Tree Mortality Consequences-- Typically, a stand of trees achieves a density in balance with the physical environment by means of death of less competitive trees in the stand. Thus, there is generally a continuous decline in stand density with increasing stand development. In a pollution-stressed forest, one finds the phenomenon of selective mortality. However, the magnitude of the mortality can be greater and the cause is other than simple competitive stress. As young trees prematurely age, taper off in growth, and die, more openings are created in the canopy. While in a healthy forest this provides room for the dominant trees to expand, in pollution-stressed forests the open canopy is opened further. Even if the pollution should be removed from the system, a lower stand density because of increased mortality rates requires a greater time to develop a full canopy than would a stand with a more normal stocking rate.

Potential Responses Under Multiple Stresses

Dolan and Hayden (1978) classified types of changes in nature reserve park ecosystems as either steady state, eddy state, or trend state. Steady state changes include diurnal and seasonal environmental changes under which the system has evolved. Eddy state changes are discrete pulses of environmental disturbances. Trend state changes are long-term changes that are often the most subtle to detect, as well as the most difficult from which to

protect natural landscape ecosystems if those changes are associated with human influences. Effects of air pollution on forests can have this trend state nature.

A Worst-Case Scenario of Future Forest Change-- It could be insightful to integrate the present state of ecological understanding on combinations of trend state changes in an attempt to see what a possible worst-case scenario might be for vegetation in western coniferous forests. The relative order of sensitivity, conceived as probability of mortality associated with a particular environmental stress, is often different (even opposite) between various species for one stress, compared to another. Ranking tree species of mature individuals in terms of likely mortality to fire would place ponderosa and lodgepole pine as "low", while white fir and incense cedar would often be rated "high". The latter would be killed by a moderate intensity surface fire (not a prescribed burn necessarily).

In contrast, research on ambient oxidant air pollution sensitivity has shown ponderosa pine as very sensitive, while white fir and incense cedar might have a low sensitivity to this stress. Air pollution weakens certain tree species which are subsequently hit by biotic diseases and insects, and produces a decreased competitive advantage, compared to less sensitive species. In the community, this can lead to decreased longevity of the sensitive species.

As a worst case condition, one could envision that our present western forest heritage from pristine decades ago under a natural fire frequency shifted the balance of tree species composition such that it was heavily proportioned with what are now air pollution sensitive species. If the air pollution problem intensifies over the years, these species can be expected to be decimated. The combined effect of both of these trend state changes, each working on different tree species, might make it impossible to preserve and protect coniferous forests in certain locations (fig. 4).

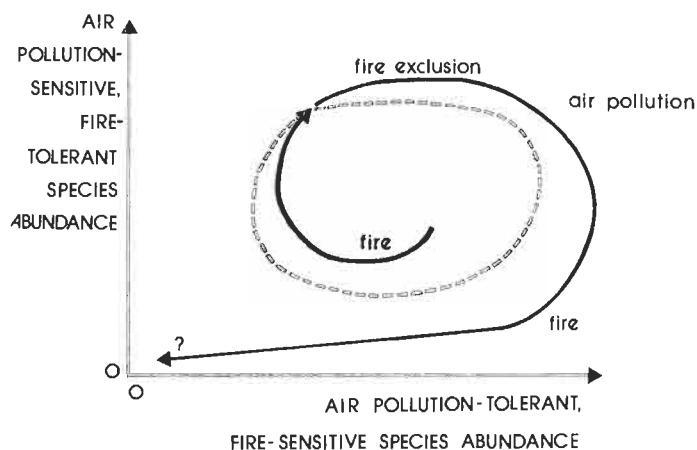


Figure 4--Possible future course of forest species composition under combined stress of air pollution and natural fire exclusion.

CONCLUSIONS

In the case of air pollution, there could be the gradual elimination of many fire-tolerant tree species from forests. Whenever a natural fire does occur under such a future scenario, the proportion of forest stand species in the fire-sensitive category could be much higher than normal, and sudden qualitative changes, or ecosystem catastrophes, in conifer forest species composition could be expected. There is a strong likelihood that conifer stands might change into mixtures of deciduous tree and shrub communities at mid-elevations, and perhaps scrub field ecosystems at higher elevations which presently contain conifer forests. This would represent a qualitative change from one successional pattern to another, and is a possibility which forest management has a responsibility to try to evaluate.

Acknowledgments:

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LITERATURE CITED

- Cooper, C.F.
1961. Equations for the description of past growth in even-aged stands of ponderosa pine. *For. Sci.*:72-80.
- Dolan, R. and B.P. Hayden.
1978. Environmental dynamics and resource management in the U.S. National Parks. *Environ. Manag.* 2(3):249-258.
- Grier, C.C., R.L. Edmonds, R.H. Waring and D.W. Cole. 1978. Forest management implications of productivity, nutrient cycling and water relations research in western conifers. p.96-106 In: *Proc., Jnt Conv. of S.A.F. and Can. Inst. For.*, 1978.
- Kickert, R.N.
1977a. Toward forecasting alternative future ecosystem responses: ecosystem modeling. p.63-85 In *Photochemical Air Pollutant Effects on Mixed Conifer Ecosystems*. Progress Report 1974-75. EPA-600/3-77-058. U.S. Environmental Protection Agency, Corvallis Environmental Research Laboratory, Corvallis, Oregon.
- Kickert, R.N.
1977b. Definition of the conifer forest ecosystem as a group of coupled ecological models. p.71-105 In *Photochemical Oxidant Air Pollution Effects on a Mixed Conifer Forest Ecosystem--A Progress Report, 1976*. Paul R. Miller and Michael J. Elderman, eds. EPA-600/3-77-104. U.S. Environmental Protection Agency, Corvallis Environmental Research Laboratory, Corvallis, Oregon.
- Kickert, R.N.
1980. Ecosystem simulation modeling. Chpt. 2 In *Photochemical Oxidant Air Pollution Effects on a Mixed Conifer Forest Ecosystem*. O.C. Taylor (ed.). U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, Oregon. EPA-600/3-80-002. 195 p
- Smith, D.M.
1962. *Practice of Silviculture*. John Wiley and Sons, New York. 578p.
- U.S. Environmental Protection Agency.
1978. Air quality criteria for ozone and other photochemical oxidants. EPA-600/8-78-004. Volume II. Office of Research and Development. Washington, D.C. 341p.
- U.S. Environmental Protection Agency.
1979. EPA changes ozone standard to 0.12 ppm. *Environmental News*, January 26, Office of Public Awareness, Washington, D.C. 3p.
- Walker, R.B., D.R.M. Scott, D.J. Salo and K.L. Reed.
1972. Terrestrial process studies in conifers: a review. pp.211-225 In: *Proc., Res. on Coniferous Forest Ecosystems Symp.*, Bellingham, WA. March 23-24, 1972.
- Whittaker, R.H. and W.A. Niering.
1975. Vegetation of the Santa Cruz Mountains, Arizona. V. Biomass, production and diversity along the elevation gradient. *Ecol.* 56:771-790.

Response of Plant Communities to Air Pollution¹

R. Guderian and K. Kueppers²

Abstract: Under the influence of air pollution two retrogressive processes are set in motion in plant communities: By means of direct and indirect effects, changes occur in structure and function of the community leading up to total destruction. Parallel to this degradation (retrogression) is a spontaneous or man initiated process during which the original adaptive resistant members of the existing community as well as new arrivals undergo secondary succession. The causes and mechanisms for air pollution-induced changes in plant communities are demonstrated by means of literature analysis and the interaction of dose response determining factors are summarized. In order to emphasize the existing potential danger and to set remedial procedures in motion, research themes are pointed out that must receive immediate attention.

Thus far research on the effects of air pollutants on plants has been centered on homotypical populations of economically important species. With the development of long-term loading of extensive areas entire ecosystems are also increasingly being influenced by air pollutants. From this, the question arises as to the possible reactions of phytocoenoses to changed air quality as a new habitat factor.

REACTIONS OF PLANT COMMUNITIES RELATED TO AIR POLLUTANT CONCENTRATIONS

Single or repeated observations in the vicinity of single sources as well as within and outside of extended regions subjected to air pollution load can only provide momentary records or sequences of changes under the respective local conditions. In general, however,

certain inherent patterns of the confrontation between plant communities and air pollution may be deduced.

For the following comparative study of the influence of varying concentrations of air pollutants on vegetation, a division into the following levels, derived from Smith's classification (1974) seems practical: high, intermediate and low dosage effects.

High Pollution Dosage

A characteristic of the relationship between high dosage and the reaction of a plant community is a breakdown of community structure -- more or less obvious depending on the complexity of the ecosystem. The degradation of the system is characterized by a rapid change in structure, including composition. It is accompanied and finally replaced by a secondary succession which can lead to a new equilibrium under sustained load.

Direct acute and chronic injury appearing especially on leaves, but not always correlated with their pollutant content, (Guderian, 1970; Linzon, 1979) will first affect the most sensitive species of the tree stratum in a forest and can lead to the total destruction of the canopy. Without protection from the freely

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entering air masses of polluted air (Bennett and Hill, 1975) shrub, herb and moss or lichen layers are destroyed one after another, until a barren zone results (Gordon and Gorham, 1963; Woodwell, 1970). As an example of such conditions, the zonation under the influence of approximately 10 tons of SO₂ per day from an iron ore roasting furnace in Biersdorf (Germany) will be described briefly.

The denuded zone in the immediate vicinity of the emission source is surrounded by the transition zone with isolated clusters of grass (Deschampsia flexuosa) and resistant ground cover (Erica cinerea, Galium mollugo, Veronica officinalis, Rumex acetosa, and Convallaria majalis). Now and then one also encounters the relatively resistant Sambucus racemosa and Rhamnus frangula in this zone. In the grass cover of vegetation consisting mostly of Deschampsia flexuosa the first shoots of Quercus petraea are established in the shelter of the herbaceous vegetation. The populations of this oak and Fagus silvatica which adjoin the stunted forest zone show signs of disintegration starting at the periphery. As SO₂ loading decreases the species diversity increases, until finally it reaches the combination typical for the acidic soil-mixed forest (Fago-Quercetum).

A comparison with other studies (Treshow, 1968; Smith, 1974; Miller and McBride, 1975; and, Linzon, 1978) shows that this is the typical picture of the break-down of a plant community, that is, a change in species composition toward a simplification of the system. In principle it does not differ from that caused by gamma radiation (Woodwell, 1963, 1970). The secondary succession which sets in as soon as the original vegetation begins to change leads in time under constant loading to the formation of new, less complex stable structures. Thus, in an old manufacturing district of Upper Silesia (Poland), in locations once stocked with native coniferous or mixed deciduous forests, Wolak (1977 and 1979) described a stable zonation in relation to different loads of SO₂, zinc, and lead. Under heavy loads an industrial wasteland is followed by a grass zone with associations dominated by Deschampsia flexuosa on oligotrophic sand, by Calamagrostis epigeios on mesotrophic sites, and by Calamagrostis villosa on damp organic soil. In the adjacent shrub zone one finds both cultivated and volunteer scrub tree species. It is remarkable that Pinus silvestris can take on the shape of a creeping shrub or of a tree with branches projecting horizontally up to 5 m from the stem. These dwarf forms are no more than 2 m high at an age of 30 to 50 years. On low grade sands groups of the described Pinus silvestris forms and Solanum dulcamara were found which were not found in similar locations without the strong influence of air pollution. Those plant communities are called industrio-climax communities (Wolak, 1971). They represent spontaneous associations with relatively constant species composition which have developed gradually through industriogenous (secondary)

succession, both from species present before pollutant loading as well as from new arrivals, under the combined influence of the habitat factors; climate, soil, insects and parasites--dominated by the factor air pollution.

Intermediate Pollution Dosage

Intermediate air pollution dosage conditions are ecologically significant because their subtle, direct and indirect effects on the individual species can set the stage for changes in the structure of the community with possibly irreversible consequences. In plant communities experiencing intermediate pollution dosage, interruption of growth and reproduction processes as well as impairment of the vitality of individual plants, among other factors through increased vulnerability to abiotic and biotic stress, become particularly important (Wentzel, 1965; Huttunen, 1979; and Laurence, 1980).

In pine and spruce populations in the Lower Main Region (Germany), which indeed show an increased sulfur content in the leaves, but did not yet display an abnormal loss of individuals, morphological changes such as thin crowns coupled with shorter needles were detected in older stands (Wentzel, 1979). Such changes occur slowly and only the accumulation of annual effects gradually leads to higher morbidity (Wentzel, 1980). In this context the carry-over of accumulated toxicants in the new shoots of the next growing season should be mentioned (Keller, 1978; Preston, 1979). A slight change of the horizontal structure in the canopy will influence such habitat factors as the supply of light and precipitation for the lower-lying vegetation. Frequently, the interaction of changed soil reaction--pH--and toxicant content brings about a restructuring of the shrub and herb strata over extended areas sometimes influencing natural reproduction of woody species (Lux, 1964; Wentzel, 1971; Harward and Treshow, 1975).

Such changes in the composition of plant communities were detected through vegetation surveys caused by a complex of factors. For example, certain species were found in dense clusters, while others were evenly distributed and still others were totally absent, depending on dosage (Borgsdorf, 1960; Gordon and Gorham, 1963; Niklfeld, 1967; Ionescu, et al., 1971; Trautmann, et al., 1971). Hajduck (1961) talks about positive or negative phytoindicators, while Anderson (1966, quoted in Treshow 1968) employs the terms "increaser" or "decreaser." The concept of phytoindicators is essentially the same as bioindication with lichen or moss species (Le Blanc and Rao, 1975; Taoda, 1977; and Pilegaard, 1978). Kaleta (1972) was able to demonstrate in addition the dynamics of change of whole plant associations under the influence of magnesite.

Brandt and Rhoades (1972, 1973) took tree species of several strata into consideration in their study on the influence of limestone dust on a forest community. This method made

it possible to assess the trend of future succession, especially through shifts detected in the species diversity of the seedling and sprout data. Thus, *Quercus coccinea*, *Quercus velutina* or *Tilia americana* can drop out as members of the oak-chestnut association and *Liriodendron tulipifera*, *Acer saccharinum* and possibly *Quercus muehlenbergii* could become dominant species. Through this example it also becomes evident how air pollutants can influence the makeup of phytocoenoses by influencing reproduction (Wentzel, 1963; Karnosky, and Stairs, 1974; Keller, 1976).

McClenahan (1978) utilized community composition as a means to investigate changes in plant communities along a pollution gradient in the Ohio Valley (USA). In this study, an eastern deciduous forest experiencing intermediate dosages was shown to decline in species richness, evenness, and Shannon diversity index within all strata of the community, particularly in those locations experiencing the highest relative dose. Simultaneously, the similarity in composition decreased with increasing dosage. Thus, the relative importance of *Acer saccharinum*, a species slightly stimulated by limestone dust (Brandt and Rhoades, 1972), showed distinct decline in all strata, while the importance of *Aesculus octandra* increased. Opposing tendencies in density were observed in some strata. A decline in the tree and herb strata was accompanied by an increase of the subcanopy and the shrub strata. This can be attributed to better light conditions in the lower strata combined with a relative increase of herbs intolerant to shade.

Low Pollution Dosage

The effects of low dosages on vegetation lie in the border zone between the fluctuating states of normal, i.e., unaffected vegetation on the one hand, and significant injurious effects on the other hand. Depending upon the respective pollutant, its concentration and duration of action, as well as the affected object and the local conditions, these effects can range from increases to reductions in growth, reproductive capability or quality of plants. Under practical conditions such effects can be detected to only a certain degree of the actual intensity. The detection limit has been lowered through the development of new exposure systems with filtered and unfiltered air (Mandl, et al., 1973; Lee, et al., 1973; Miller, et al., 1979; Shinn, et al., 1979). However, measurement of pollutant effects on plant communities occupying large regions presents particular difficulties because the necessary pollution-free control areas with comparable soil and climate are not available.

Before detectable reductions occur in productivity or alteration of environmental conditions can be observed, there are various changes that are induced at the plant biochemical, physiological or substructural level

(Keller, 1974; Jager and Klein, 1977; Horsman and Wellburn, 1977; and, Raabe and Kreeb, 1979). Of course the question of how much such findings reveal about the economic and ecologic performance of a particular plant species remains of central importance here. Some of the reactions undoubtedly have no effects on the total organism; even if significant effects are found, it is very difficult to establish a causal link to the primary responses mentioned above. The possible effects of low dosage on plant communities, for example, through changes in interspecific competition, are almost totally unresolved. The filtering effects of vegetation is an important process but this topic will only be introduced here.

As shown in the Solling project (Ulrich, et al., 1978) or the Hubbard Brook study (Bormann and Likens, 1979) vegetation can filter large amounts of pollutants out of the atmosphere without showing signs of external injury or growth depression. Particulate and gaseous pollutants enter an ecosystem through adsorption and absorption mainly on leaf surfaces as well as soil and water surfaces (Hill, 1971; Bennett and Hill, 1975; and, Olsen, 1976). The specific behavior of the substance is important for the possible long-term effects of low dosage on plant communities. Pollutants which are subject to rapid decomposition such as ozone or PAN take effect through the summation of direct effects. NO_x , NH_3 , or sulfur compounds can be channeled into the nutrient cycle and may destroy the balance of especially sensitive ecosystems, such as moors, through eutrophication (Porter, et al., 1972; Cowling and Lockyer, 1976). The importance of acid precipitation in this context is not yet clear (Braekke, 1976; and, Tamm, 1976). Accumulating substances such as heavy metals, represent a special danger for ecosystems (Kraemer, 1976; and Guderian, 1980). Soil samples and analyses of moss specimens have revealed that a constant input into ecosystems is offset by only a limited export, and that this is now occurring over wide areas (Huckabee, 1973; Ruhling and Tyler, 1973; and Grdzinska, 1978). Such components can also endanger the nutrient cycle (Mags, 1977 and Uba, 1977); further, they reduce the number and activity of decomposers, thereby impairing remineralization as a requirement for uninterrupted biogeochemical cycles (Taylor, 1975; and, Greszta, et al., 1979). Especially, with accumulating substances and under continuous loading, it is only a question of time before the direct and indirect effects described above begin to interrupt the structure and function of plant communities.

THE INFLUENCE OF POLLUTANTS ON THE FUNCTION OF PLANT COMMUNITIES

Changes in plant communities caused by pollutants can lead to more or less lasting impairment of economic and ecologic functions depending on the dosage. The damage to agriculture

through growth reduction, loss of quality and higher labor costs have long drawn considerable attention, but only now is an attempt being made to take the effects on performance of ecosystems into account. In this connection, an especially important question is how pollutants affect such functions of vegetation as filter effect, stabilization of climate, regulation of water and nutrient cycles, soil conservation as well as the preservation of living space for polymorphic zoo- and phytoceonoses. Extensive changes in vegetation cover are probably linked to interruption or even total breakdown of all the above-mentioned functions. For example, the function of plant communities as protection against erosion or as a factor in counterbalancing excessive temperature fluctuations and the accompanying danger from light frost during bud break is considerably more impaired in areas experiencing high pollutant dosage where woods have been replaced by sparse vegetation than in unpolluted areas. Currently, to what extent intermediate and low dosages affect the functions of vegetation mentioned, can, at best, be deduced to an order of magnitude (Materna, 1980) from the known real effects on vegetation.

Causes for the Observed Responses of Plant Communities

The effects of a given pollutant on plant communities, as illustrated by several examples, are determined by: the genetically predetermined degree of resistance of the companion species (Dochinger et al., 1965; Rohmeder et al., 1965), the modifying influence of environmental conditions of resistance, and the changes in intra- and interspecific relations caused by pollutants.

Populations of certain plant species, varieties, and clones, as well as individuals within the respective populations studied, react to a given air pollution stress with varying degrees of sensitivity. In contrast to certain phytopathogenic organisms (Baumann, 1951; Grossmann, 1970), there is no absolute resistance, as demonstrated by the existence of vegetation-free zones. The schematic diagram (Figure 1) is an attempt to demonstrate which factors influence the response to air pollution stress.

Individual and Species Specific Responses

According to Levitt (1972), two mechanisms determine a plant's resistance to stress: stress avoidance and stress tolerance. In the first case the stress, caused here by a specific pollutant dose, is prevented from taking effect--it is excluded. A multitude of factors determines the resistance of a plant organism to the entry of pollutants into the cell. Morphological properties such as

shape and surface structure including wax layers (Rentschler, 1973; Shriner, 1980) as well as the number, distribution, and aperture of the stoma (Meidner and Mansfield, 1968) must be mentioned. According to Taylor (1978), whose definition was taken into consideration in the corresponding section of Figure 1, stress tolerance presupposes the entry of the respective pollutant into the cell. As long as the entering substance is tolerated, assimilated or buffered, and consequently no morphological or physiological change takes place, one speaks of "strain avoidance." Above certain intracellular concentrations, for example, after certain biochemical threshold values have been exceeded, injury occurs which is either reversible (elastic strain), such as subtle changes in photosynthetic performance (Sij and Swanson, 1974), or irreversible (plastic strain), such as injury to leaves in the form of necrosis.

Thus, the biochemical threshold values, which characterize the transitions from strain avoidance to strain tolerance as well as from elastic strain to plastic strain, determine the tolerance of a plant. It follows that the individual response can manifest itself in terms of indifference, modification or death of the affected plant depending on the level of ambient stress, and the respective resistance.

As is apparent from Figure 1, a multitude of organismal and environmental factors before, during, and after the pollutant impact is responsible for the sometimes very great differences in the "resistance series" or "resistance groups" of various authors (Stoklasa, 1923; Bredemann, 1956; Thomas, 1961; Garber, 1967; Mooi, 1974; Davis and Wilhour, 1976; Guderian, 1977). All climatic factors, for example, that regulate the number, size and aperture of the stomata (Bronte and Conguet, 1975; Hall and Kaufmann, 1975), such as light, optimal water supply, high relative humidity or adequate temperature, determine the rate at which pollutants are absorbed (Guderian, 1970; Jones and Mansfield, 1970; McLean and Schneider, 1971).

The influence of edaphic factors is demonstrated with two examples: Copper-beech (Fagus sylvatica) is considerably more resistant on soils with high lime content than on sandy soil low in nutrients; elm (Ulmus campestris) proved to be one of the most resistant species in alluvial forests, but in less suitable habitats it was one of the most vulnerable of all the deciduous species (Wentzel, 1968). This shows the difficulty in setting up generally accepted resistance series, as recent studies with various soybean cultivars under changing environmental conditions have clearly shown (Heagle, 1979a, b).

The larch serves as a typical example for changes in resistance in relation to levels of concentration (Guderian and Stratmann, 1962; Wentzel, 1963). Under high, acute SO₂ concentrations, both Larix europea and Larix leptolepis show signs of necrosis before spruce

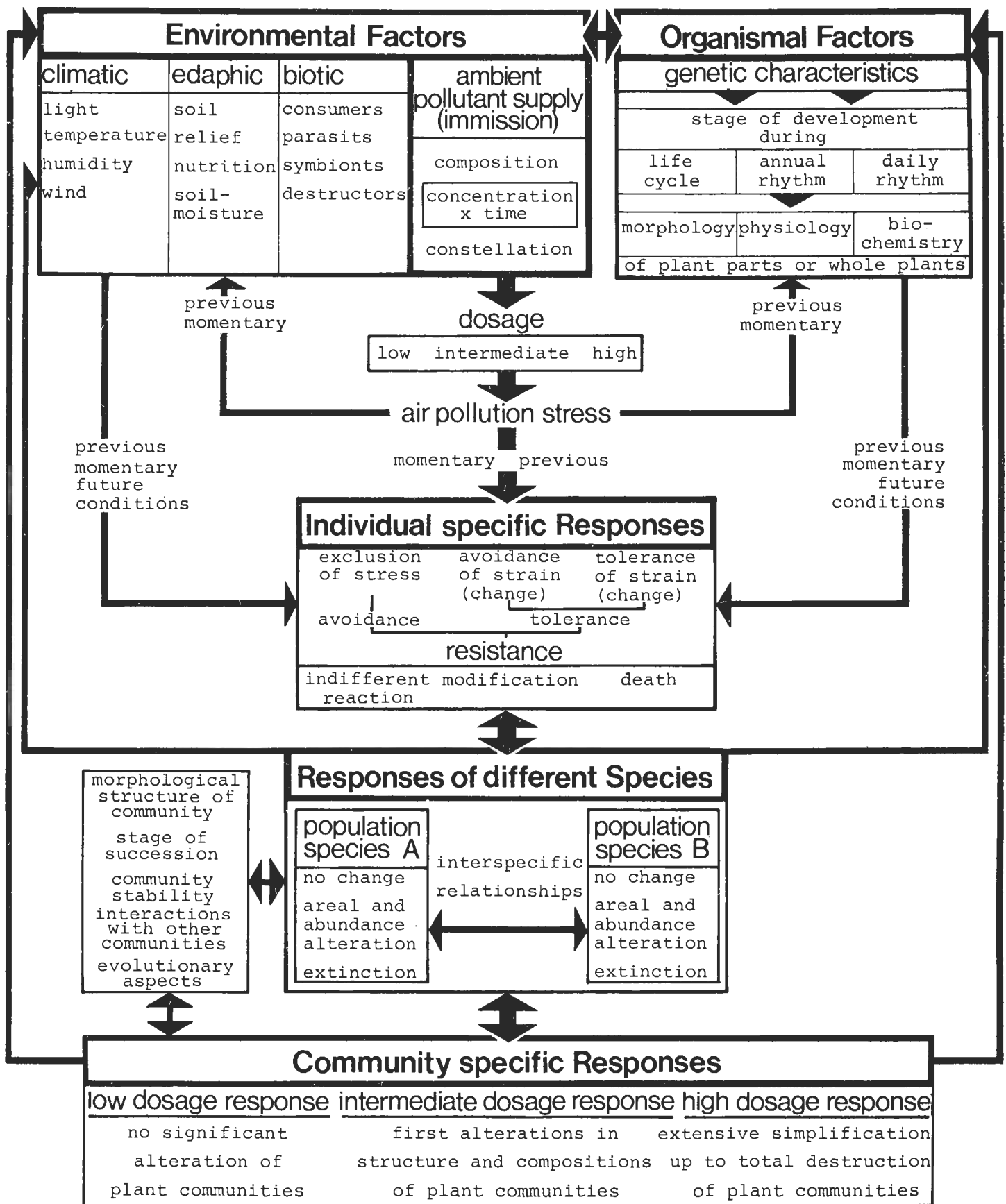


Figure 1: Effect-determining-factors for various responses of plants on individual-, species- and community-levels.

(*Picea abies*) and pine (*Pinus silvestris*). On the other hand, the larch is among the most resistant conifers under continuous low levels of concentration (Wentzel, 1963), and it is widely used to reestablish tree populations after the degradation of spruce and pine forests in regions of chronic stress.

Regarding organismal factors, the significance of age for sensitivity should be stressed. According to observation in the field, conifers such as *Picea abies* and *Pinus silvestris* remain particularly susceptible from the late pole timber stage (at the time of accumulating growth) through to the tree timber stage. During these periods of development, especially strong reductions in growth and the widespread degradation of entire stands occur (Wentzel, 1962; Materna et al., 1969), while deciduous populations respond in a much weaker form. In plantings, however, under mostly chronic SO₂ concentrations, the conifers mentioned and the copper-beech (*Fagus silvatica*) and pedunculate oak (*Quercus pedunculata*) exhibited nearly equal resistance (Guderian and Stratmann, 1968).

One aspect not often taken into account when judging the resistance of plants, besides environmental, pollutant, and organic influences, is the criteria used to interpret the effect. Various deciduous tree species, such as linden (*Tilia cordata* and *Tilia platyphyllos*) and beech (*Fagus silvatica*) respond to acute SO₂ concentrations with leaf necrosis earlier than spruce (*Picea abies*) or Scots pine (*Pinus silvestris*). Nevertheless the deciduous species can still grow in polluted regions where spruce and Scots pine die out (Wentzel, 1968). Thus resistance must first of all be characterized by the differences in the reduction of growth and yield of the specific species. The functions of the plant species being considered here determine the criteria used to evaluate the effects (Guderian, 1977). Through the short description of factors determining the resistance of an individual or a species it can be seen what degrees of variation must be expected in the responses. The use of these results to forecast the behavior of single species under air pollution stress is necessarily very difficult, especially when studying plant communities.

In the following model, supported by experimental results, an attempt is made to illustrate the possible responses of two plant species to increasing air pollution stress (Fig. 2). A specific response of two plant species (A and B) is shown in percent of control. Up to a specific concentration labeled A1 and B3 no significant deviation in the responses of the exposed plants and the control plants could be detected. Important qualitative differences in response between the two species exist above concentration A1. With Species A, the specific response is initially stimulated by sulphur dioxide; a reduction of performance only occurs at a higher concentration, while this Species B lacks stimulating effect. The further slope of the curve shows

the degree of reduction in response. The concentration A4/B4 should be emphasized, as here the resistance relationship of the two species changes (Wentzel, 1963). In the concentration range A1 to A4/B4 species A would have an advantage over species B, even under pollutant concentrations which do not yet have an adverse effect on species B. Accordingly, changes in the composition of plant communities must be expected even if SO₂ concentrations are so low that they do not yet have a direct harmful effect. This is a significant aspect for ecological research.

Community Specific Responses

The previously demonstrated relationships between heredity, environment and resistance in individuals or homotypical populations are naturally also valid for plant communities. Community specific aspects must be given additional consideration when ascertaining pollutant effects. The importance of the relationship between two or more populations shown in Fig. 1 is underlined by the few existing results from experiments on the influence of air pollutants to plant communities. Thus, according to experimental analysis of pure and mixed seedings, consisting of rye grass (*Lolium multiflorum*) hairy vetch (*Vicia villosa*) and crimson clover (*Trifolium incarnatum*) shifts in the composition of plant communities cannot be explained exclusively through the direct effect of pollutants on various species of different sensitivity (Guderian, 1966, 1977). Under SO₂ the influence of interspecific competition was altered. As a result, the primary effect on the more susceptible members was magnified to such a degree that they could no longer compete effectively for vital growth-determining factors. As a result of changed competition in the community, the decline of the more sensitive members allowed improved growth of the more resistant species. The total community yield decreased less than would have been expected from the loss of the more susceptible species. Similar results were found under the influence of ozone (Bennett and Runeckles, 1977), ultraviolet radiation (Fox and Caldwell, 1978) and ionizing radiation (McCormick, 1963). The last of these studies shows the importance of stress during the seedling and sprout stages.

The extent of shifts in plant communities as a reaction to a given load is also dependent to a large degree on the condition of the community itself. The importance of the building of strata, of morphological structures, as well as relief and uniformity of the vegetation cover were already pointed out as was the interconnection between stages of succession and system responses. The stability of a community greatly influences the response of its individual members as well as the whole to a given pollutant load. In the presence of small disturbances, highly productive, complex systems can usually

Special responses of
plant species A and B
control = 100 %

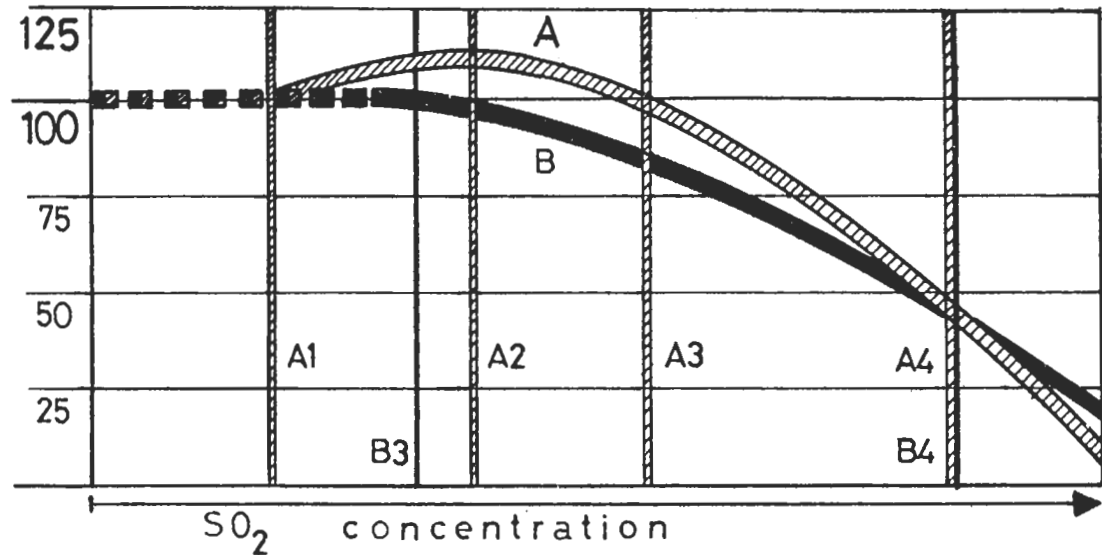


Figure 2 :

Typical reactions of two plant species depending
on the sulfur dioxide content of air

regain their state of balance quickly because of their complex feedback systems. Under heavier loads on the other hand, drastic changes must be expected particularly if certain key species are very sensitive to the respective air pollutant. But even very low pollutant concentrations can produce quite considerable effects in plant communities, especially if synecological amplitudes of the natural community species are far apart. Additional stress through pollutants of lower dosage can lead to drastic reduction in vitality of plants already living outside their ecologic optimum. The relatively high susceptibility of spruce (*Picea abies*) in the Erz mountains (Materna, 1972) and in the boreal coniferous forests in Finland (Huttunen, 1979) might well be caused by their unsuitable habitats. Keeping this in mind, it seems problematic to transfer those dose-effect relationships determined from "production ecosystems"--in which the food plants generally encounter favorable conditions--to natural ecosystems.

The relationships enumerated up to this point show clearly why emissions induce degradation in plant communities. On the other hand spontaneous adaptation to air pollution stress may be observed--adaptation which does not ensure the survival of the species through stunting, but rather seems to have genetic origins. When *Marchantia polymorpha* (Briggs, 1972) was exposed to lead, and various grass species (Bradshaw, 1971, 1972, 1976) were affected by copper and zinc, more tolerant populations developed in a short time through directed selection. Bell and Clough (1973), Bell and Mudd (1976), and Horsman and Wellburn (1977) mention similar processes with *Lolium perenne* and *Rumex obtusifolius* subjected to decades of SO₂ loading. Finally, the results of long-term fumigation of native grassland (Preston and Bullett, 1978) also point to the fact that under anything less than acute concentrations spontaneous adaptation may occur in the course of the formation of a new secondary equilibrium, which may also interrupt possible long-term injury (Preston, 1979a).

Accordingly, the dose and its rate of change should be adjusted over the long-term such that plant communities retain their capability--even by the evolutionary method mentioned above--to fulfill their function in natural and agrarian ecosystems to the fullest (Guderian and Kueppers, 1979).

The genetically fixed variation in populations which is expressed in the described spontaneous adaptation, provides the basis for breeding of pollutant resistant plants through selection and reproduction of relatively resistant individuals (Bialobok, 1979).

The responses of individuals and homotypic populations taking into account interspecific relations discussed in this section, lead to the community specific responses shown in Fig. 1 which range from insignificant changes to the total destruction of plant communities.

CONCLUSIONS

Contamination of extensive areas due to increasing emissions and control strategies using tall stacks for dilution has made the study of plant communities and ecosystems especially necessary. To aid in recognition of possible risks and in making decisions regarding control measures at the source and in the affected area, the following points must be clarified:

1. Under which doses do changes occur in structure and function of plant communities of different complexity?

2. To what extent do plant communities show more sensitive responses than the individual species composing them?

3. What are the mechanisms of these changes?

- Points of impact for air pollutants in the ecosystem.
- Location of pollutants in the ecosystem (assimilation, accumulation, break down).
- Direct stimulatory or injurious effects on the individual species of the community.
- Causes and mechanisms of changes in competition equilibrium.
- Secondary succession with particular attention to adaptation and compensation.

4. How are risks determined for plant communities?

- Development of experimental designs and intensifications of epidemiological studies for the determination of effects to highly structured systems.
- Establishment of permanent study areas to investigate succession.
- Use of model plant communities as indicators in eco-toxicological tests.
- Analysis of the condition of ecosystems before and after start-up of a pollutant source.

5. What measures are necessary for the protection of vegetation?

- Determination of dose-response relations for plant communities as a basis for risk predictions and the establishment of standards for ecosystems.
- Collection of genetic resources in natural reserves and in gene banks.
- Protection of endangered plant communities, especially in existing natural reserves, from effects of air pollutants.
- Development and maintenance of air pollutant control strategies allowing pollutant dose and its rate of change be so controlled that the structural diversity, and the ecologic and economic functions of the vegetation, as well as its function as a gene pool, are fully protected.

LITERATURE CITED

- Anderson, F. K.
1966. Air pollution damage to vegetation in Georgetown Canyon, Idaho, M. S. Thesis Univ. Utah, 102 p., cited in: Treshow, M., 1968. *Phytopathology* 58: 1108-1113.
- Bell, J. N. B., and W. S. Clough.
1973. Depression of yield in ryegrass exposed to sulfur dioxide, *Nature* 241, 5384: 47-49.
- Bell, J. N. B., and C. H. Mudd.
1976. Sulphur dioxide resistance in plants. A case study of *Lolium perenne*, In: Effects of air pollution on plants, T. A. Mansfield, ed. p. 87-104, Society for Experimental Biology Seminar Series 1, Cambridge University Press.
- Bennett, J. H., and A. C. Hill.
1975. Interaction of air pollutants with canopies of vegetation, In: J. B. Mudd, T. T. Kozlowski, eds. Responses of plants to air pollution, p. 273-306, Academic Press, New York.
- Bennett, J. P., and V. C. Runeckles.
1977. Effect of low levels of ozone on plant competition, *J. Appl. Ecol.* 14, p. 877-880.
- Bialobok, S.
1979. Identification of resistant or tolerant strains and artificial selection or production of such strains in order to protect vegetation from air pollution, Symposium on the effects of air-borne pollution on vegetation, August 20-24, Warsaw (Poland).
- Bonte, J., and P. Louguet.
1975. Interrelations between sulphur dioxide pollution and stomatal movement in *Pe-largonium hortorum*: Effects of relative humidity and CO₂ concentrations in air, *Physiologie Vegetale* 13, p. 527.
- Bormann, F. H., and G. E. Likens.
1979. Pattern and Process in a Forested Ecosystem, Springer, New York.
- Borsdorf, W.
1960. Beitrage zur Fluorschadensdiagnostik, I. Fluorschaden-Weiserpflanzen in der Wildflora, *Phytopath. Z.* 38, p. 309-315.
- Bradshaw, A. D.
1971. Plant evolution in extreme environments, In: Ecological genetics and evolution, R. Creed, ed., p. 20-50, Oxford.
- Bradshaw, A. D.
1972. Some of the evolutionary consequences of being a plant, *Evolut. Biol.* 5, 1-10.
- Bradshaw, A. D.
1976. Pollution and evolution, IN: Effects of air pollution on plants, T. A. Mansfield, ed., p. 135-159, Society for Experimental Biooogy Seminar Series 1, Cambridge.
- Braekke, F. H., (ed.).
1976. Impact of acid precipitation on forest and freshwater ecosystems in Norway, Research report No. 6, SNSF project of the Agricultural Research Council of Norway
- Brandt, C. J. and R. W. Rhoades
1972. Effects of limestone dust accumulation on composition of forest community, *Environ. Poll.* 3, p. 217-225.
- Brandt, C. J., and R. W. Rhoades
1973. Effects of limestone dust accumulation on lateral growth of forest trees. *Environ. Poll.* 4:207-213.
- Bredemann, G.
1956. Biochemie und Physiologie des Fluors, Akademie-Verlag, Berlin.
- Briggs, D.
1972. Population differentiation in *Marchán-tis polymorpha* L. in various lead pollution levels. *Nature (London)*. 238: 166-167.
- Cowling, D. W. and D. R. Lockyer
1976. Growth of perennial ryegrass (*Lolium perenne* L.) exposed to a low concentration of sulphur dioxide. *J. Exp. Bot.* 27:411.
- Davis, D. D. and R. G. Wilhour
1976. Susceptibility of woody plants to sulfur dioxide and photochemical oxidants. A literature review, Corvallis Environmental Research Laboratory, U.S. Environmental Protection Agency, EPA-600/3-76-102.
- Dochinger, L. S., C. E. Seliscar and F. W. Bender.
1965. Etiology of chlorotic dwarf of eastern white pine, *Phytopath.* 55:1055.
- Dunikowski, S.
1977. Relationship between increase in air-pollution toxicity and elevation above ground, Cz. IX. Meteorological studies, Wyd. Instytut Badawczy, Lesnictwa.
- Fox, F. M. and M. M. Caldwell
1978. Competitive interaction in plant populations exposed to supplementary ultraviolet B-radiation, *Oecologia* 36 (2):173-190.
- Gaumann, E.
1951. Pflanzliche Infektionslehre, Birkhauser Verlag, Basel.
- Garber, K.
1967. Luftverunreinigung und ihre Wirkungen, Gebr. Borntrager Verlag, Berlin.

- Gordon, A. G. and E. Gorham
1963. Ecological aspects of air pollution from an iron-sintering plant at Wawa, Ontario, Ca. J. Bot. 41:1063.
- Greszta, J., S. Braniewski, K. Marczyńska-Galkowska, and A. Nosek
1979. The effect of dusts emitted by non-ferrous metal smelters on the soil, soil microflora and selected tree species, Ekol. pol. 27:397-426.
- Grodzinska, K.
1978. Mosses as bioindicators of heavy metal pollution in Polish national parks, Water, Air, Soil Poll. 9:83-97.
- Grossmann, F.
1970. Einfluss der Ernährung der Pflanzen auf den Befall durch Kränkheitserreger und Schädlinge, Landwirtsch. Forsch. 18(25).
- Guderian, R.
1967. Reaktionen von Pflanzengemeinschaften des Feldfutterbaues auf Schwefeldioxydeinwirkungen, Schriftenreihe der Landesanstalt für Immissions- und Bodennutzungsschutz des Landes Nordrhein-Westfalen in Essen, Heft 4:80-100.
- Guderian, R.
1970. Untersuchungen über quantitative Beziehungen zwischen dem Schwefelgehalt von Pflanzen und dem Schwefeldioxyidgehalt der Luft, Z. Pflanzenkrankh. Pflanzenschutz 77: 200-220, 289-308, 387-399.
- Guderian, R.
1977. Air Pollution, Phytotoxicity of acidic gases and its significance in air pollution control, Springer, New York.
- Guderian, R.
1980. Terrestrial vegetation--air pollutant interactions: non-gaseous pollutants, International conference air pollutants and their effects on the terrestrial ecosystem, May 10-17, Banff, Alberta, Canada.
- Guderian, R. and H. Stratmann
1962. Freilandversuche zur Ermittlung von Schwefeldioxydwirkungen auf die Vegetation, I. Teil: Übersicht zur Versuchsmethodik und Versuchsauswertung Forsch. Ber. d. Landes Nordrhein-Westfalen Nr. 1118, Westdeutscher Verlag, Köln und Opladen.
- Guderian, R. and H. Stratmann
1968. Freilandversuche zur Ermittlung von Schwefeldioxydwirkungen auf die Vegetation, III. Teil: Grenzwerte schädlicher SO₂-Immissionen für Obst- und Forstkulturen sowie für landwirtschaftliche und gärtnerische Pflanzenarten, Forsch. Ber. d. Landes Nordrhein-Westfalen Nr. 1920, Westdeutscher Verlag, Köln und Opladen.
- Guderian, R. and K. Kuppers
1979. Problems in determining dose-response relationships as a basis for ambient pollutant standards, Symposium on the effects of air-borne pollution on vegetation, August 20-24, Warsaw (Poland).
- Hajduk, J.
1961. Quantitative und qualitative Änderungen der Phytozonosen verursacht durch Exhalationsprodukte von Fabriken, Biologia, Bratisl XVI, 6:404-419.
- Hall, A. E. and M. R. Kaufmann
1975. Stomatal response to environment with *Sesamum indicum* L., Plant Physiol. 55:455.
- Harvard, M. and M. Treshow
1975. Impact of ozone on the growth and reproduction of understory plants in the aspen zone of western USA, Environ. Conserv. 2: 17-24.
- Heagle, A. S. D. E. Body, and W. W. Heck
1973. An open top chamber to assess the impact of air pollution on plants, J. Environ. Qual. 2(3):365-368.
- Heagle, A. S.
1979a. Effects of growth media, fertilizer rate and hour and season of exposure on sensitivity of four soybean cultivars to ozone, Environ. Poll., 18(4):313-322.
- Heagle, A. S.
1979b. Ranking of soybean cultivars for resistance to ozone using different ozone doses and response measures, Environ. Poll. 19:1-10.
- Hill, A. C.
1971. Vegetation: A sink for atmospheric pollutants. J. Air Poll. Contr. Ass. 21: 341-346.
- Horsman, D. C., and A. R. Wellburn
1977. Effect of SO₂ polluted air upon enzyme activity in plants originating from areas with different annual mean atmospheric SO₂ concentrations, Environ. Poll. 13(1):33-40.
- Huckabee, J. W.
1973. Mosses: Sensitive indicators of air-borne mercury pollution, Atmos. Environ. 7: 749-754.
- Huttunen, S.
1979. The integrative effects of airborne pollutants on boreal forest ecosystems, Symposium on the effects of air-borne pollution on vegetation, August 20-24, Warsaw (Poland).
- Ionescu, A., U. Sanda, E. Grou, and I. Buiculescu
1971. Considerations on the effects of atmospheric impurification upon the flora and

- vegetation of the Copsa Mica region, Rev. Rom. Biol. Botanique 16:125-139.
- Jager, H. J. and H. Klein
1977. Die Bedeutung stoffwechselphysiologischer Reactionen von Pflanzen als Kenngrößen für SO₂-Immissionswirkungen, Phytopath. Z. 89(2):128-134.
- Jones, R. J. and T. A. Mansfield
1970. Increases in the diffusion resistances of leaves in a carbon dioxide-enriched atmosphere, J. Exp. Botany 21:951-958.
- Kaleta, M.
1972. Die Wirkung von Magnesit-Immissionen auf die Änderung von Pflanzengemeinschaften, Mitteilungen d. forstl. Bund.vers.anst. Wien 97(2):569-584.
- Karnosky, D. F. and C. R. Stairs
1974. The effects of SO₂ on in vitro forest tree pollen germination and tube elongation, J. Environ. Qual. 3(4):406-409.
- Keller, Th.
1974. Die Peroxidase--Aktivität als Indikator unsichtbarer Immissionsschädigungen, Anz. Schädlingsschde. PflzSchutz, Umweltschutz 47:86-89.
- Keller, Th.
1976. Zur Phytotoxizität von Fluorimmissionen für Holzarten, Mitt. eidg. Anst. Forstl. Versuchswesen 51(2):301-331.
- Keller, Th.
1976. Wintertime atmospheric pollutants--do they affect the performance of deciduous trees in the ensuing growing season, Environ. Poll. 16:243-247.
- Kramer, F.
1976. Erste Untersuchungen zur Erstellung eines Bundesbelastungskatasters (Pb, Zn, Cd, Cu) in Raume Duisburg, Dinslaken, Schriftenreihe der Landesanstalt für Immissions- und Bodennutzungsschutz des Landes Nordrhein-Westfalen, Heft 39:45-48.
- Laurence, J. A.
1978. Effects of air pollutants on plant-pathogen interactions, 71st Ann. Meeting of APCA, June, Houston.
- Le Blanc, F. and D. N. Rao
1975. Effects of air pollutants on lichens and bryophytes, In: Responses of plants to air pollution, J. B. Mudd and T. T. Kozlowski, eds., p. 237-272, Academic Press, New York.
- Lee, J. J. and R. A. Lewis
1978. Zonal air pollution system: Design and performance, In: The bioenvironmental impact of a coal-fired power plant, Third Interim Report, Colstrip, Montana, E. M. Preston and R. A. Lewis, eds., EPA-600/3-78-021, p. 322-344, Oregon.
- Levitt, J.
1972. Response of plants to environmental Stresses. Academic Press, New York.
- Linzon, S. N.
1978. Effects of air-borne sulfur pollutants on plants, In: Sulfur in the environment, Part II, Ecological impacts, J. O. Nriagu, ed., p. 109-162, Wiley, New York.
- Linzon, S. N., R. G. Pearson, P. J. Temple and W. D. McIlveen
1979. Effects on vegetation by power plant emissions of sulphur dioxide and soot, 72nd Annual Meeting of APCA, June 24-29, Cincinnati, Ohio.
- Lux, H.
1964. Beitrag zur Kenntnis des Einflusses der Industrieexhalationen auf die Bodenvegetation in Kiefernforsten (Dubener Heide), Archiv für Forstwesen 13:1215-1223.
- Mags (Minister für Arbeit, Gesundheit und Soziales des Landes Nordrhein-Westfalen)
1977. Umweltbelastung im Raum Dorsten, Analyse und Problemlösung, Düsseldorf.
- Mandl, R. H., L. H. Weinstein, D. C. McCune and M. Keveny
1973. A cylindrical open-top chamber for the exposure of plants to air pollutants in the field, J. Environ. Qual. 2:371-376.
- Materna, J.
1972. Einfluss niedriger Schwefeldioxydkonzentrationen auf die Fichte, Mitt. forstl. Bund.Vers.Anst. Wien, 97 (I):219-232.
- Materna, J.
1979. Effects of pollution on the capacity of vegetation to perform such functions as water retention, soil protection, wildlife habitat, etc., Symposium on the effects of air-borne pollution on vegetation, August 20-24, Warsaw (Poland).
- Materna, J., J. Jirgle, and J. Kucera
1969. Vysledky mereni koncentraci kyslicniku siriticeho v lesich Krusnych hor, Ochrana ovzduši 6:84-92.
- McClenahan, J. R.
1978. Community changes in a deciduous forest exposed to air pollution, Can. J. For. Res. 8(4):432-438.

- McCormick, R.
1963. Changes in herbaceous plant community during a three year period following exposure to ionizing radiation gradients, 1st Nat. Symp. Radioecol., Proc. Radioecology, Reinhold Co., p. 271-275, New York.
- McLean, D. C. and R. Schneider
1971. Fluoride phytotoxicity: Its alteration by temperature, Proc. 2nd Intern. Clean Air Congr., p. 292-295, New York.
- Miller, P. R. and J. R. McBride
1975. Effects of air pollutants on forests, In: Responses of plants to air pollution, J. B. Mudd and T. T. Kozlowski, eds., p. 196-235, Academic Press, New York.
- Miller, J. E., D. C. Sprugel, R. N. Muller, H. J. Smith and P. B. Xerikos
1979. An open-air fumigation system for studies of SO₂ effects on crops, Environmental Research Contribution No. 79-21, Argonne National Laboratory, Argonne, Illinois.
- Mooi, J.
1974. Onderzoek naar de gevoeligheid en opnamecapaciteit van houtige gewassen voor HF, Groen 30(10):321-327.
- Murphy, G. E., Jr., T. R. Sinclair and K. R. Knoerr
1977. An assessment of the use of forests as sinks for the removal of atmospheric sulphur dioxide, J. Environ. Qual. 6:388-396.
- Niklfeld, H.
1965. Beobachtungen im Rauchschaadensgebiet eines Aluminiumwerkes, Cbl. ges. Forstwesen 84(2-6):318-329.
- Olsen, R. A.
1976. Absorption of sulphur dioxide from the atmosphere by cotton plants, Soil Science 84:107.
- Pilegaard, K.
1978. Airborne metals and SO₂ monitored by epiphytic lichens in an industrial area, Environ. Pollut. 17:81-92.
- Porter, L. K., F. G. Viets, Jr., and G. L. Hutchinson
1972. Air containing nitrogen-15 ammonia: Foliar absorption by corn seedlings, Science (N.Y.) 175:759-761.
- Preston, E. M.
1979. The ecological implications of chronic sulfur dioxide exposure for native grasslands, 72nd annual meeting of the Air Pollution Control Association, Cincinnati, Ohio, June.
- Preston, E. M., and T. L. Gullet (eds.)
1978. The bioenvironmental impact of a coal fired power plant, Fourth Interim Report, Colstrip, Montana, U.S. EPA, Corvallis Environmental Research Laboratory, Corvallis, Oregon.
- Raabe, R. and K. H. Kreeb
1979. Enzyme activities and chlorophyll and protein content in plants as indicators of air pollution, Environ. Poll. 19:119-137.
- Rentschler, I.
1973. Die Bedeutung der Wachsstruktur auf den Blättern für die Empfindlichkeit der Pflanzen gegenüber Luftverunreinigungen, Proc. 3. Intern. Clean Air Congr., Dusseldorf, p. A 139-A 142.
- Rohmeder, E., and A. von Schoenborn
1965. Der Einfluss von Umwelt und Erbgut auf die Widerstandsfähigkeit der Waldbaume gegenüber Luftverunreinigungen durch Industrieabgase- Ein Beitrag zur Zuchtung einer relativ rauchresistenten Fichtensorte, Forstw. Cbl 84:1-13.
- Ruhling, A. and G. Tyler
1973. Heavy metal deposition in Scandinavia, Water, Air and Soil Poll. 2:445-455.
- Siccama, T. G. and W. H. Smith
1978. Lead accumulation in a northern hardwood forest, Environ. Sci. Technol. 12:593-594.
- Shinn, J. H., B. R. Clegg, and M. L. Stuart
1979. A minimum, field fumigation method for exposing plants to controlled gradients of air pollution levels, Contr. Environ. Sci. Div., Lawrence Livermore Laboratory.
- Shriner, D. S.
1980. Terrestrial vegetation- air pollutant interactions: Non-gaseous pollutants, wet deposition, Int. Conf. air pollutants and their effects on the terrestrial ecosystem, May 10-17, Banff, Alberta, Canada.
- Sij, J. W.
1974. Short-term kinetic studies of the inhibition of photosynthesis by sulfur dioxide, J. Environ. Qual. 3:103.
- Smith, W. H.
1974. Air pollution effects on the structure and function of the temperate forest ecosystem, Environ. Poll. 6:111-129.
- Stoklasa, J.
1923. Die Beschädigung der Vegetation durch Rauchgase und Fabrikexhalationen Berlin.
- Tamm, C. O.
1976. Acid precipitation: Biological effects in soil and on forest vegetation, Ambio 5: 235-238.

- Taoda, H.
1977. Bryophytes as indicators of air pollution, In: Science for better environment, Y. Fukushima, ed., p. 292-301, Proc. Int. Congr. on the Human Environment, Kyoto 1975.
- Taylor, O. C.
1975. Air Pollution injury to plant processes, Hort. Sci. 10:501.
- Taylor, G. E., Jr.
1978. Plant and leaf resistance to gaseous air pollution stress, New Phytol. 80(3): 523-534.
- Thomas, M. D.
1961. Effects of air pollution on plants, In: Air pollution, WHO, Geneva, Monog. Series, Vol. XXXXVI, p. 233-278.
- Trautmann, W., A. Krause, and R. Wolff-Straub
1971. Veränderung der Bodenvegetation in Kiefernforsten als Folge industrieller Luftverunreinigungen im Raum Mannheim-Ludwigshafen, Schriftenreihe Vegetationskunde 5:193-207.
- Treshow, M.
1968. The impact of air pollutants on plant populations, Phytopath. 58:1108-1113.
- UBA (Umweltbundesamt der Bundesrepublik Deutschland, Berlin).
1977. Berichte 4/77, Luftqualitätskriterien für Cadmium.
- Ulrich, B., R. Mayer, P. K. Khanna, and J. Prenzel
1978. Ausfilterung von Schwefelverbindungen aus der Luft durch einen Buchenbestand, Z. Pflanzenernährung Bodenkunde 141:329-335.
- Wentzel, K. F.
1962. Konkrete Schädwirkungen der Luftverunreinigung in der Ruhrgebietslandschaft, Natur Landschaft 37:118-124.
- Wentzel, K. F.
1963. Waldbauliche Massnahmen gegen Immissionen, Allg. Forstz. 18:101-106.
- Wentzel, K. F.
1965. Die Winterfrost-Schaden 1962/63 in Koniferen-Kulturen des Ruhrgebietes, und ihre vermutlichen Ursachen, Forstarch. 36:49-59.
- Wentzel, K. F.
1968. Empfindlichkeit und Resistenzunterschiede der Pflanzen gegenüber Luftverunreinigung, Forstarch. 39 (9):189-194.
- Wentzel, K. F.
1979. Die Schwefel- Immissionsbelastung der Koniferenwälder des Raumes Frankfurt Main, Forstarch. 6:112-121.
- Wentzel, K. F.
1980. Weisstanne - immissionen empfindlichste ein heimische Baumart, Allgemeine Forst Zeitschr. 14:373-374.
- Wolak, Jr.
1971. Studies on the industrioklimax in Poland, Methods for the identification and evaluation of air pollutants injurious to forests, Proc. XV IUFRO Congr., Wien.
- Wolak, Jr. (ed.)
1977. Relationship between increase in air pollution toxicity and elevation above ground, Wyd. Instytut Badawczy Lesnictwa, Warsaw.
- Wolak, J.
1979. Reaction des ecosystemes a la pollution subnecrotique, Symposium on the effects of air-borne pollution on vegetation, August 20-24, Warsaw (Poland).
- Woodwell, G. M.
1963. The ecological effects of radiation, Scientif. American 208 (6):40-49.
- Woodwell, G. M.
1970. Effects of pollution on the structure and physiology of ecosystems, Science 168: 429-433.

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Forecasting Effects of SO₂ Pollution on Growth and Succession in a Western Conifer Forest¹

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Abstract: A simulator has been developed for the mixed conifer forest type of the Sierra Nevada, California to forecast the effects of SO₂ on forest growth and succession. The model simulates recruitment, growth, and death of each tree and is based on a northeastern USA simulator with extensive modifications. These modifications include the introduction of fire ecology, temporal seed crop patterns unique to the Sierra, and water stress. Pollutant stress is modeled as an effect on tree growth. The model simulates the shift from the ponderosa pine dominated forest type to the white fir dominated mixed conifer type as elevation increases from 5000 to 6000 ft. It also simulates the fire-suppression of white fir and the fire-climax of ponderosa pine. For a 10% growth reduction of ponderosa pine from pollutant stress and with growth reductions in other species as determined by their relative sensitivities, standing crops of ponderosa pine were reduced and white fir increased.

It is anticipated that extensive fossil fuel energy development will occur in the United States over the next several decades with increased emissions of phytoactive effluents. It has long been recognized that many of these pollutants have deleterious effects on the growth and behavior of vegetative communities. The effects of SO₂ in particular have been extensively studied and occur at all levels of resolution from the metabolic process level to the ecosystem level. In the work reported on here, we wanted to predict the effects at the population and community levels given the results of the effects at the whole plant level. We have developed other models to forecast effects at the process level (Kercher 1977; Kercher 1978). The model, SILVA, uses an empirical dose-response relationship for the effects of pollutants at the tree-level. By virtue of the ecological interactions contained in the model, the effects at the tree level are translated into effects at the community level.

We have followed the modeling approach developed by Botkin and others (1972) who developed

JABOWA, a simulator of forests of the northeastern USA. For a case study, SILVA has been applied to the ponderosa pine and mixed conifer forest types of the Sierra Nevada, California, USA. The associated species in these forests are ponderosa pine (*Pinus ponderosa*), white fir (*Abies concolor*), Douglas-fir (*Pseudotsuga menziesii*), sugar pine (*Pinus lambertiana*), incense-cedar (*Libocedrus* or *Calocedrus decurrens*), California black oak (*Quercus kelloggii*), and Jeffrey pine (*Pinus jeffreyi*).

MODEL DESCRIPTION

SILVA calculates environmental parameters of the stand and initializes number and sizes of the trees from environmental and control data respectively. A table of good and bad seed crop years and a list of fire years is generated. The effect of pollution on trees is calculated. The number of new seedlings for that year, the growth of each tree, and mortality are then determined for each year. Growth is modeled as a difference equation in the tree dbh and as a function of environmental variables. The killing is done stochastically depending on the probability of death as determined by ecological risk, lack of growth, and fire damage. The dynamics of fuel accumulation (litter and brush) are also modeled.

Temporal Seed Crop Patterns--For the conifers of the Sierra Nevada, there can be significant temporal variations in the annual cone production. We modeled the phenomenon of high and low yield seed years as a Bernoulli random process with blocking. If the species is in an unblocked state, the probability of a good crop is p and of a poor crop is $(1-p)$. If a good seed crop occurs, the

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the process is assumed to be blocked for $r-1$ years. The parameters p and r were taken from cone crop data.

Fire Ecology--Fire is a critical factor in the population dynamics of western forests. The most important aspect of fire is fire-induced mortality. The occurrence of fire was also modeled as a Bernoulli random process with blocking and p and r are based on fire incidence data. The blocking in this case arises from the time required for fuel to build back up to levels capable of supporting fire propagation. Fire kills by raising the temperature inside the tree and by damaging the crown. Fire intensity is calculated in kilowatts/meter of fireline length using FIREMOD (Albini 1976) and probability of death is determined as a function of dbh, bark thickness, and scorch height. Scorch height is calculated from fire intensity, ambient temperature, and windspeed.

Moisture Stress--The effects of moisture stress are modeled by multiplying the difference equation for growth by a moisture stress factor. Parameters for this function are taken from published ranges of tolerance data. The moisture stress factor is a function of the ratio of actual evapotranspiration to potential evapotranspiration.

MODELING SO_2 -POLLUTANT EFFECTS

It has long been held that chronic injury results from sulfate accumulation in plant tissues. Guderian (1977) has suggested that in most cases involving a single point source, chronic injury results from the "short-term action of relatively high concentration peaks". Thus the long-term average air concentration can be quite low due to the large number of pollution-free time periods. Because two different perspectives exist, i.e., (1) measuring average annual concentration or accumulated dose or (2) regarding injury as arising from episodes and making detailed measurements of episode parameters, we have two different pollutant-effects submodels.

Seasonal Average Submodel--This approach assumes that growth reduction is a simple function of the SO_2 concentration averaged over the growing season, or equivalently, of the integral of SO_2 concentration over time. We use a dose-response function in which growth decreases linearly with increasing accumulated dose based on the preliminary study of the tree-ring data of Lathe and McCallum (1939) for ponderosa pine grown near the smelter at Trail, B.C.

Successive Episode Model--An alternative approach is to calculate the accumulated damage caused by successive short episodes separated by time intervals with no or negligible pollution. One method to implement this approach would be to use a process model (Kercher 1978). The method used here is an empirical dose-response where the dose is that accumulated from successive episodes (Kercher and Axelrod 1980). We use the empirical dose-response of Larson and Heck (1976) which mo-

dels the probit of the effect being proportional to the log of the generalized dose.

SIMULATION RESULTS

Fire Ecology--Figure 1 shows the response of ponderosa pine and white fir with fire occurring at the natural frequency and with complete fire suppression. Ponderosa pine is well adapted to fire and dominates where undergrowth is thinned by fire. The model reproduces this result and indicates white fir would eventually outcompete ponderosa pine in the absence of fire. The model suggests that a significant factor in the fire adaptation of ponderosa pine is its growth rate and growth form which allow it to evade fire by minimizing the time that the crown is exposed to fire. The effects of fire on tree mortality is shown in figure 2a. Note the shift in age of death to the lower ages in the presence of fire.

Pollution Simulations--As an example of effects of pollution, consider the minimally significant case of 10% growth reduction in ponderosa pine. We scaled the response of the remaining species according to their published relative sensitivities. These calculations used the seasonal average model. The results for ponderosa pine and white fir (fig. 3) indicate that while white fir undergoes a nominal growth reduction of about 1 to 2% per tree with pollution, total basal area actually shows a dramatic increase. This is due to the much greater growth retardation that the dominant species experiences. Tree mortality of ponderosa pine (fig. 2b) indicates the trees are at higher risk at higher ages under pollution. The older, slower growing, pollution-stressed trees have size-dependent risks comparable to those of the younger unstressed trees. We can summarize (fig. 4) the results for ponderosa pine,

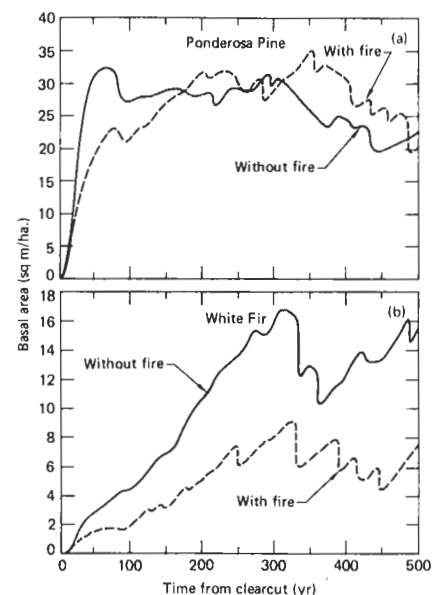


Figure 1--Average of basal area from 25 simulations showing effects of fire. (a) Ponderosa pine (b) White fir.

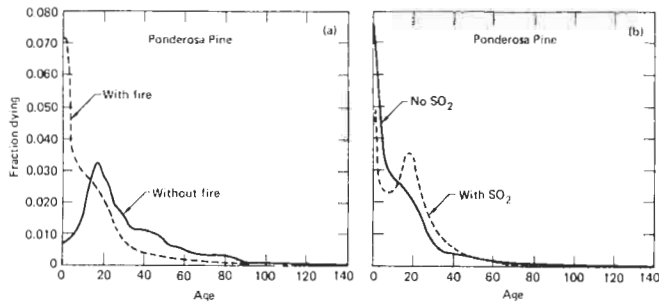


Figure 2--Fraction of trees which died in simulations of figure 1 plotted against age at death. (a) With and without fire. (b) With and without pollution.

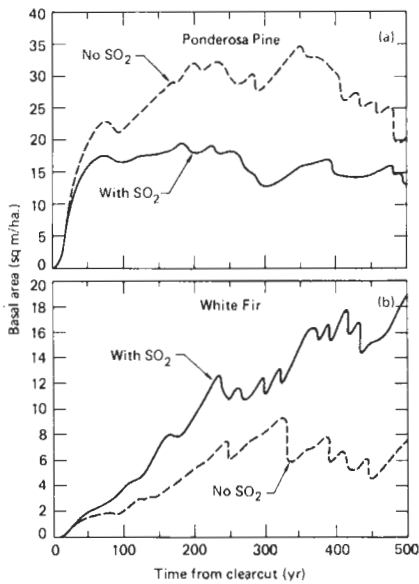


Figure 3--Basal area growth with and without pollution for (a) pine and (b) fir.

white fir, and Douglas-fir by using boxplots of the distributions of the 500 annual data points of each species fraction of the total basal area. Note the decrease in pine and the increase in fir with pollution. The basal area of Douglas-fir is extremely reduced. The environmental conditions were poor for Douglas-fir even in the absence of SO_2 . The competitive disadvantage for Douglas-fir is made worse by pollution because Douglas-fir is sensitive to SO_2 and carries its needles longer than ponderosa pine. Thus the growth reduction for an individual tree (greater than that for ponderosa pine) translates into a much larger effect on basal area.

LITERATURE CITED

Albini, F.A.

1976. Computer-based models of wildland fire behavior: a user's manual. 68 p. USDA For. Serv., Intermt. For. and Range Stn., Ogden, Utah.

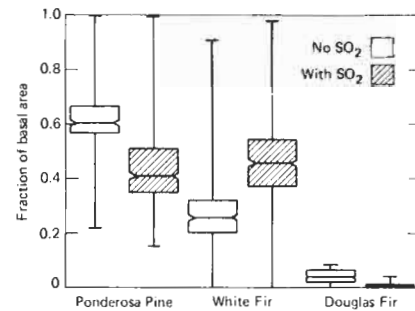


Figure 4--Boxplots of polluted and unpolluted cases. Median is line at notches. Top of box is 75th percentile; bottom of box is 25th. Range is vertical line. Non-overlapping notches indicate significance at 95% level.

Botkin, D.B., J.F. Janak, and J.R. Wallis.

1972. Some ecological consequences of a computer model of forest growth. *J. Ecol.* 60: 849-872.

Guderian R.

1977. Air pollution. 127 p. Springer-Verlag, New York.

Kercher, J.R.

1977. GROWL: A crop growth model for assessing impacts of gaseous pollutants from geothermal technologies. UCRL-52247, 33 p. Lawrence Livermore Natl. Laboratory, Livermore, Calif.

Kercher, J.R.

1978. A model of leaf photosynthesis and the effects of simple gaseous sulfur compounds (H_2S and SO_2). UCRL-52643, 37 p. Lawrence Livermore Natl. Laboratory, Livermore, Calif.

Kercher, J.R. and M.C. Axelrod.

1980a. A model for forecasting the effects of SO_2 pollution on succession in a western coniferous forest: Interim Report. UCID-18537. 58 p. Lawrence Livermore Natl. Laboratory, Livermore, Calif.

Kercher, J.R. and M.C. Axelrod.

1980b. SILVA: a model for forecasting the effects of SO_2 pollution on growth and succession in a western coniferous forest: Final report. UCID (to be published). Lawrence Livermore Natl. Laboratory, Livermore, Calif.

Larson, R.I. and W.W. Heck.

1976. An air quality data analysis system for interrelating effects, standards, and needed source reductions: Part 3. Vegetation injury. *J. Air Poll. Control Assoc.* 26:325-333.

Lathe, F.E. and A.W. McCallum

1939. The effect of sulphur dioxide on the diameter increment of conifers. In *Effect of sulphur dioxide on vegetation*. National Research Council of Canada. p. 174-206. N.R.C. No. 815. Ottawa, Canada.

Forest Models: Their Development and Potential Applications for Air Pollution Effects Research¹

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Abstract: As research tools for evaluating the effects of chronic air pollution stress, forest simulation models offer one means of integrating forest growth and development data with generalized indices of pollution stress. This approach permits consideration of both the competitive interactions of trees in the forest stand and the influences of the stage of stand development on sensitivity of component species. A review of forest growth models, including tree, stand, and gap models, is provided as a means of evaluating relative strengths, weaknesses, and limits of applicability of representative examples of each type. Data from recent simulations with a gap model of eastern deciduous forest responses to air pollution stress are presented to emphasize the potential importance of competition in modifying individual species' responses in a forest stand. Recent developments in dendroecology are discussed as a potential mechanism for model validation and extended application.

Atmospheric emissions from widespread industrial and urban sources have now significantly altered the air quality of extensive forested regions of the world. Wolak (1971) described the influence of industrial emissions on forested areas of Poland as an abiotic para-natural ecological factor. He viewed the results of these emissions on forest succession as the establishment of a new final seral stage termed the industrio-climax. Assessing the impacts of these changes and those which may ensue as we rely increasingly on fossil fuels in

the future is a challenge made considerably more difficult by the complex nature of forest ecosystems. The perennial growth habit of forest trees and the nature of their competitive interactions in a forest community make difficult the evaluation of chronic exposures of forests to atmospheric pollutants. Treshow (1970) pointed out that terrestrial ecosystems are delicately balanced with a structure that may depend on a few critical species. He indicated the response of vegetation may be slow, but once natural balances are sufficiently disrupted, subsequent alterations may occur much more rapidly because of irreversible alterations of essential system functions or species interactions.

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Traditionally, studies of responses of forest trees to air pollution stress have focused primarily on species level responses, seedlings, a few selected physiological processes, and generally simplistic exposure regimes. While valuable information has been gained on specific plant-pollutant interactions, we still know very little about the potential effects of pollutants on forest communities. For instance, how are individual species effects integrated over space and time into responses of the forest community? What are the probable limits of impacts on

forests based on our current knowledge of sensitivity of individual species responses?

To address these questions necessitates that we combine both autecological and synecological approaches. The former we can derive in large part from dose-response data for individual species. In the latter task, we can derive from the experiences of two decades of experimentation with mathematical simulation of the growth and development of forest communities (Reichle and others 1973, Munro 1974, Shugart and West 1980). The purpose of this paper is to review the basic components of these models with a view toward understanding their strengths and weaknesses and their potential utility as tools for studying community-level responses to air pollution stress.

Computer Models of Forest Dynamics

In the mid-1960's, foresters and ecologists independently began to develop extremely detailed computer models of forest growth and development. Foresters realized that certain changes in forest practice (e.g., change in trees due to genetic improvement, use of fertilizer in forests) would render less useful the stand yield tables that had been laboriously developed over the prior several decades. Some foresters began to develop models of forest growth and yield that could be calibrated on the extant, stand-table data sets and could also be used to incorporate some of the changes in forestry practice (fig. 1). At the same time, ecologists became dissatisfied with the static notion of forest typology and developed intensive investigations (e.g., the International Biological Program) of the dynamic aspects of ecosystems. This increased interest in ecosystem dynamics led naturally to the development of forest models. By the mid-1970's (fig. 1), three approaches evolved to modeling the long-term dynamics of forests (table 1). We will discuss the utility of each of these approaches in terms of its applicability to assessing the consequences of air pollution effects over long time scales. The approaches are:

- (1) Forest models consider the forest as the focal point of the simulation model. Forestry yield tables constitute a highly data-dependent subset of these forest models.
- (2) Tree models take the individual tree as the basic unit of a forest simulator. The degree of complexity ranges from simple tabulation of the probabilities of an individual tree of one kind being replaced by an individual of another kind to extremely detailed models that include 3-dimensional geometry of different species at different sizes.
- (3) Gap models dynamically simulate particular attributes of each individual tree on a prescribed spatial unit of relatively small

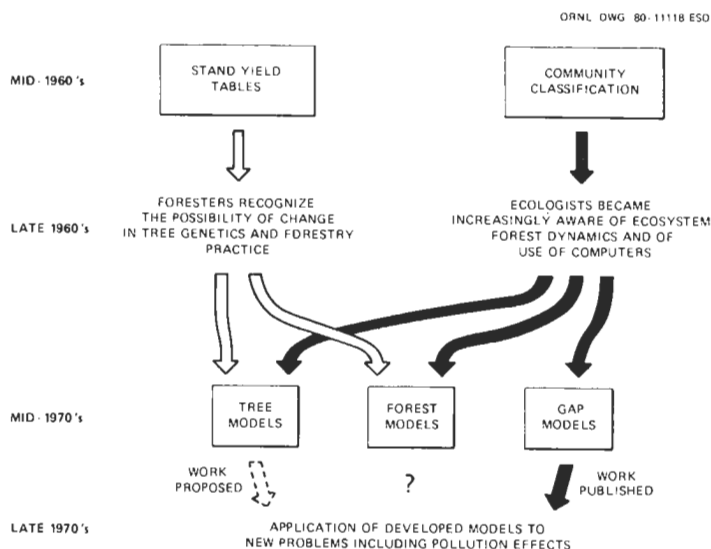


Figure 1--Recent historical origins of computer models used for pollution effects assessment at the forest ecosystem level.

size. The spatial unit is usually either a gap in the forest canopy or a sample quadrat.

In general, the model type used is based on the problem considered, the data available, and the desire to develop a flexible model. The tree and forest model categories correspond to the tree and stand model categories used in a recent review of forestry models (Munro 1974). In the present review, gap models (which might be considered a special case of three models) are recognized as a category developed exclusively for use in studying ecological succession.

Forest Models

Yield tables used in forestry management are, in fact, empirical models of expected responses of an even-aged forest of (usually) a single species. In this context, a forest is taken as a larger spatial dimension than either single tree or gap models considered explicitly.

Comparable succession models have been developed using a variety of mathematical approaches. Most of these models consider the landscape to be composed of a number of mosaic elements that change in response to successional processes. These changes may be viewed as probabilistic (e.g., Wilkins 1977, Hool 1966) or continuous (Shugart and others 1973), depending on modeling assumptions relating to the actual size of the landscape considered. Forest models tend to be data-dependent concerning changing rates of the mosaic elements assumed to comprise the forests, and the actual mechanisms that cause the changes in the forests do not appear explicitly in the models. All of the forest models listed (table 1) require little computer time and can be solved analytically in many

Table 1. Classification and characterization of forest simulation models as tools for evaluating stress effects.

Model category	Age-structure	Space	Examples	Assessment potential	
				Limitations	Advantages
Forest	Even (usually)	Nonspatial	Most yield tables in use in forestry today	1. Usually calibrated on long-term data sets on many different sites. Slow to develop.	1. High degree of realism because of data input. 2. Familiar to the forestry industry.
	Mixed	Nonspatial	Hool 1966 Olson and Christofolini 1966 Moser and Hall 1969 Shugart and others 1973 Johnson and Sharpe 1976 Wilkins 1977	1. Usually requires data or insights that are collected over a long time period.	1. Provide a regional inventory of effects. 2. Mathematically simple and could be coupled with economic models.
		Spatial	Newnham 1964 Lee 1967 Mitchell 1969 Lin 1970 Bella 1971 Hatch 1971 Hegy 1974 Lin 1974	1. Require extremely detailed growth data and other detailed parameters. 2. Commercial forests only are considered. 3. Establishment may not be considered.	1. Tremendous detail. 2. Economic variables (e.g., board feet, products) simulated directly.
Tree	Even				
		Nonspatial	Clutter 1963 Curtis 1967 Dress 1970 Goulding 1972 Sullivan and Clutter 1972 Burkhart and Strub 1974 Solomon 1974 Clutter 1974 Elfvig 1974	1. Require extremely detailed growth data. 2. Commercial forests only are considered. 3. Establishment may not be considered.	1. Economic variables (e.g., board feet) simulated directly. 2. Fast computationally; could be interfaced with economic models.
	Mixed	Spatial	Adlard 1974 Arney 1974 Ek and Monserud 1974 Mitchell 1975	1. Detailed data requirement. 2. Computationally slow.	1. Tremendous detail. 2. Have been proposed for use in long-term pollution assessment.
		Nonspatial	Leak 1970 Bosch 1971 Namkoong and Roberts 1974 Forcier 1975 Suzuki and Umemura 1974 Horn 1976 Noble and Slatyer 1978 Waggoner and Stephens 1970	1. Lack of detail in output. 2. Require clever validation procedures.	1. Models have been explored for their theoretical aspects. Level of abstraction is both an advantage and disadvantage.
Gap	Mixed	Vertical	Botkin and others 1972 Shugart and West 1977 Mielke and others 1978 Tharp 1978 Shugart and Noble 1980 Shugart and others 1980 Doyle and others 1980	1. Spatial in the vertical dimension only. 2. Require clever validation procedures.	1. Have been used in long-term pollution assessment. 2. Complex parameters can be inferred from ecological principles.

cases. All of these models could be used for assessing the consequences of some inferred pollution effect on a region's forests assuming that the primary problem of estimating the forest stand response could be overcome.

Spatially Explicit Tree Models

Two categories of models in table 1 (even-aged or mixed age) are used almost exclusively in sophisticated evaluations of planting,

spacing, and harvesting schemes in commercial forests. These models produce information used primarily by large governmental or industrial land managers which is as a consequence, normally communicated by direct means that do not necessarily involve the scientific literature (e.g., internal reports). The models we listed in these categories (table 1) are probably only a subsample of such models that are actually in use.

These models function by incrementing individual trees (usually tree diameter, crown volume, and various form and shape parameters) periodically and are usually solved in 1- to 5-year time steps. To illustrate the degree of detail used in such models, Mitchell's (1969) model of white spruce (*Picea glauca*) uses branch-pruning of trees that overlap to determine competition interaction.

The models explicitly consider the crowding of trees and can be easily adapted to either even- or mixed-age stands. In fact, Hegyi's (1974) even-aged model is derived from Arney's (1974) mixed-age model, and Mitchell's (1969, 1975) models are derived in the converse manner. The models are designed for commercial forestry operations and do not include phenomena that ecologists would expect in a succession simulator. They generally ignore establishment of invading seedlings and often use functions for geometry of trees that could only be expected to hold in young, vigorously growing trees. The models sometimes use thinning or harvest as a surrogate for mortality. Because of the level of detail needed, these models synthesize great amounts of autecological data that are usually only available for commercial species and are difficult to extend to mixed-species forests. Nonetheless, the FOREST model (Ek and Monserud 1974) does simulate mixed-species, mixed-age northern hardwood forest in Wisconsin. This model is also being considered for use in a pollution effects assessment problem (fig. 1). There is also a potential to apply the other models of the commercial species that should be explored.

Even-aged, Nonspatial Tree Models

Even-aged, nonspatial models have been used in commercial forestry also and are logical nonspatial alternatives to models in the previous category. Nonspatial models have been used almost exclusively in pine (*Pinus* spp.) plantations and are usually in the form of differential equations with basal area, stocking density, and volume (biomass) of a forest stand changing with respect to time. Because these relationships are functions of the size of the average tree, the models contain parameters derived from the expected growth of trees. The even-aged, mono-species character of the simulated forests allows the assumption that mathematical functions for the expected response of an average or typical tree are sufficient to express these relationships among volume, stocking, and basal area. These models work best if the trees tend to be the same size, which helps to explain the use of these models in the more genetically optimized, short-rotation, crop-like *Pinus* plantations. The underlying assumptions of these models limit their applications to even-aged stands, and the development of mixed-aged models using this approach is difficult. Unlike the spatial mono-species models we discussed previously, these models can, in some

cases, be solved analytically and, in all cases, require only a moderate amount of computer time.

Mixed-age, Nonspatial Tree Models

These models simulate ecological succession in naturally regenerated forests. Their emphasis is on birth/death processes affecting individual trees, and the importance of tree growth and form is greatly deemphasized. They are not particularly complex (i.e., birth and death of trees might be treated as simple stochastic processes; replacement of trees as a first-order Markov process), but frequently it is the stated objective of the authors to attempt to capture the salient aspects of succession with a minimal model representation. In this objective, the models are actually explorations into the consequences of theories and assumptions on the nature of ecological succession based on the attributes of the species involved (Gleason 1926, Drury and Nesbit 1973).

The models can provide considerable insight into patterns of ecosystem dynamics and can be solved analytically without resorting to digital computation. An example of this modeling approach (Noble and Slatyer 1978) uses the vital attributes of species to determine the expected patterns of community successions generated by competition among the species. Vital attributes considered are the modes that a species uses to persist at a site, the modes for establishment, the availability of a method or persistence (e.g., seeds, vegetative sprouts) at different life stages of the plants (propagule, juvenile, mature, extinct), and longevity of individuals. Using these species attributes, they construct schematic diagrams of changes that can be compared with observational data from a given area.

Gap Models

Gap models simulate year-to-year changes in diameters of each tree on a plot of known area. These models do not account for the exact location of each tree but use tree diameters to determine tree height and then use simulated leaf area profiles to devise competition relationships due to shading. These models are spatial in the vertical but not the horizontal dimension. This simplification greatly reduces the cost of running these models and also eliminates the consideration of complex spatial patterns of trees, should this be important in a given application. The vertical gap models are probably best used in studies of successional dynamics of natural forests considered over long time spans. Gap models have also been the first detailed succession simulators applied to air pollution effects research.

Current Model Applications

Most models built strictly for forestry use are usually intended as applications in a restricted set of specified circumstances.

Given the great specificity of the models, they still simulate commercially important forest types, and it is unfortunate that they have yet to be used in any pollution effects studies. Several of the succession models presented in table 1 have been used in evaluating environmental impacts on naturally occurring forests. Botkin (1973, 1977) considered the effects of CO₂ enrichment on plant growth and subsequent effects on forest dynamics. He found that an arbitrarily assumed percentage change in rate of photosynthate production at the individual plant level in CO₂-enriched atmospheres was not manifested directly as a change in forest growth. Other effects such as plant competition and shading tended to lower the magnitude of the system response. McLaughlin and others (1978) and West and other (1980) performed model experiments on chronic air pollution stress expressed as a change in growth rates of pollution-sensitive trees. They noted that the response of growth over the long term and in natural forests might vary in direction as well as in magnitude from what one might predict from laboratory or greenhouse studies. Kickert (this symposium) and Kercher (this symposium) have also used these gap models of western forests to investigate long-term pollutant effects. All of these studies identify a common problem; namely, in natural forests where trees vary in spacing, size, and competitive responses, one cannot extrapolate directly from laboratory studies to field conditions. Forest succession models can provide and have provided a necessary adjunct to laboratory-based assessments of environmental effects. We will provide a detailed example of such an application in the following section.

Gap Model Application

As used in the following example, the model (the FORET model, Shugart and West 1977) considers 33 forest tree species native to the southern Appalachian region and simulates growth of individual trees on a circular 1/12-ha plot. The growth of each tree on a plot is incremented yearly as a function of (1) total annual growing degree days (5.6°C base), (2) the total leaf area of taller trees on the plot, (3) total number of trees on the plot, and (4) the size of the tree. A typical simulation is illustrated in figure 2.

The selection of a species for the plot and subsequent initiation and growth of the tree are based on silvicultural characteristics of each species. These characteristics include:

- (1) site requirements for germination,
- (2) palatability of seedlings for browsers,
- (3) sprouting potential, (4) shade tolerance,
- (5) germination and growth temperature requirements, (6) inherent growth potential,
- (7) longevity, and (8) sensitivity to crowding stress (fig. 2). The initial trees established on a plot with bare soil are those having shade-intolerant growth requirements and germination affinities for mineral soil. As the simulation

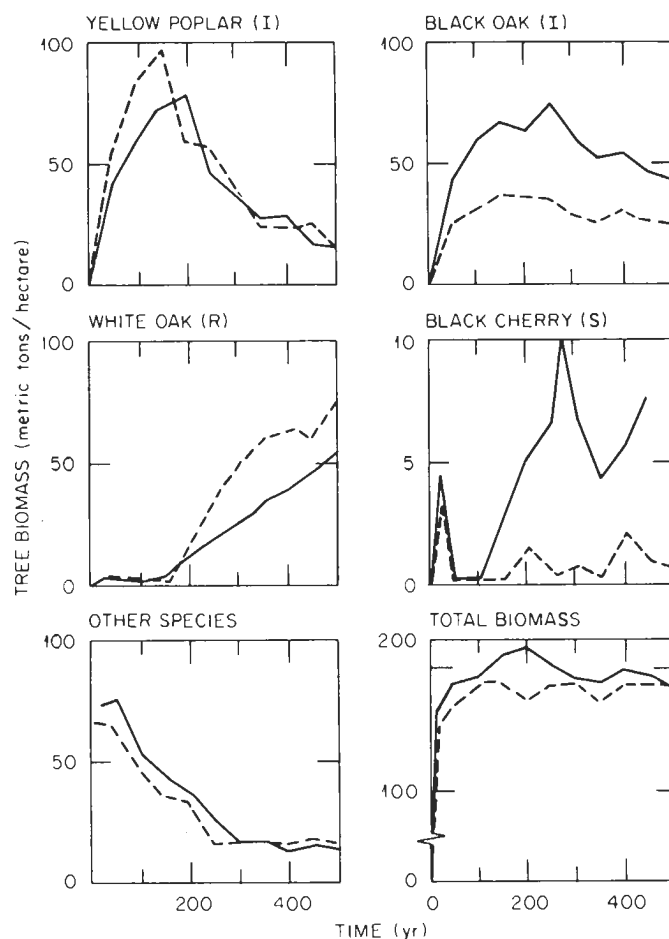


Figure 2--Species and stand dynamics of a forest with and without continuous exposure to air pollution stress (— unaffected; ---- affected).

proceeds, trees that have the ability to germinate in leaf litter and grow under shaded conditions are selected by the model. Leaf litter is assumed to have accumulated to a level commensurate with the total tree biomass for the plot. The amount of shade cast by each tree is a function of leaf area of the tree and is calculated allometrically from its diameter by totaling the leaf area of all taller trees on the plot. Under optimal conditions, tree growth is assumed to occur at a rate that will produce an individual of maximum recorded size (dbh) for that species during the period of maximum recorded age and is based on a curvilinear function that grows a tree to two-thirds its maximum dbh at one-half its age. Modifications reducing this optimal growth are imposed on each tree by some additive combination of shading and crowding from other trees on the plot and the stochastic variation from optimum climate. Optimum climate is defined as the means of the minimum and maximum growing degree-days within an individual species range. Death is a stochastic process with the probability of dying inversely related to the yearly growth increment. Total

stand density characteristics are calculated from dbh. Ingrowth occurs by germination of seeds and sprouting, and simulation may be initiated either from a bare plot or an existing stand of a predetermined composition and structure.

Validation of the FORET model was accomplished by simulating a deciduous forest stand with and without American chestnut as a viable species (Shugart and West 1977). Simulations with chestnut removed produced forests of similar composition to the contemporary, post-chestnut blight forest. With chestnut included, the model produced a forest similar (Spearman rank correlation - $r = 0.83$, see Siegel 1956) in composition to the relatively undisturbed southern Appalachian forest which existed around 1890 to 1910. All simulations were typically repeated for a large number of plots (≥ 100), and interpretations were based on average biomass of individual species and the forest stand determined from the multiple runs.

By utilizing this type model, we investigated the results of the interaction of forest tree competition and air pollution stress. In doing this, the following relevant questions concerning the response of forests to a pollutant were considered:

- (1) What level of air pollution stress would be required to significantly alter forest growth and development?
- (2) How are stress effects integrated over time?
- (3) How important is competition in moderating or enhancing induced stresses on individual species?
- (4) How are species responses integrated into the response of forest systems?

Application of the model to the study of the effects of air pollution stress on growth and development of eastern forests necessitated (1) developing a rationale for classifying species in terms of their relative sensitivity to this stress and (2) incorporating growth reductions into the model which reflected species' sensitivity ranking and a range of impacts which might be expected under field conditions.

Addressing the first task assumes that species vary measurably in their growth responses to chronic air pollution stress. Such a conclusion is intuitively obvious from a wealth of data from controlled laboratory and field studies where obvious differences in sensitivity of foliage to visible injury from air pollution have been demonstrated. Data on relative sensitivity of forest trees to growth reduction from chronic air pollution stress are limited, however. In this application, we made the assumption that trees most sensitive to foliar injury

would also be most sensitive to growth inhibition. We group the 32 species into 3 sensitivity classes (resistant, intermediate, and sensitive), based on their relative sensitivity to visible injury. The sensitivity classification was based on 10 years of field survey data of vegetation near a coal-fired electric plant (McLaughlin and Lee 1974) and an extensive summary of field and laboratory data on susceptibility of woody plants to SO_2 and photochemical oxidants reported by Davis and Wilhour (1976). This classification then formed a framework for addressing the second task, determining appropriate levels of growth reduction to introduce into the modeled forest. For eastern forests, this task must also rely on the rather limited data currently available from the literature. However, one advantage of mathematical models is that a range of stress levels may be simulated. While not providing exact quantitative answers, such an approach does permit one to bracket the range of likely responses based on the best available data.

In the FORET approach, both the influence of varying stress levels and the stage of forest maturity at which stress was initiated were examined. Results of a typical simulation are presented in figure 2. Here, responses of selected species are shown from a simulation in which annual growth inhibitions of 20, 10, and 0 percent were imposed on seedlings in sensitive, intermediate, and resistant sensitivity classes, respectively. Increases in biomass of 4 major species [yellow poplar (intermediate), white oak (resistant), black oak (intermediate), and black cherry (sensitive)], the collective "other" species category, and total stand biomass were compared with and without simulated air pollution stress as the forest developed over time.

The results indicated that competition within the forest stand may greatly modify responses predicted from individual species' sensitivity to stress. Both enhanced growth suppression (black oak and black cherry) and reduced suppression (yellow poplar) were demonstrated. These responses were attributed to shifts in the competitive potential of these species induced by differential stress applied within the forest stand. An examination of total biomass of all species indicated that suppression could be greater than (as high as 20 percent) or less than (< 5 percent) that of the weighted average suppression (7 percent) imposed in the simulation.

Another useful capability inherent in simulation approaches is that variations in stand age and, relatedly, stand composition may be introduced for the time of stress initiation. In the FORET test, stage of stand development was also identified as an important modifier as shown in figures 3 and 4. Yellow poplar, a fast-growing, shade-intolerant species which showed growth stimulation when the seedling forest was stressed (initiation time - year 0),

LIRIODENDRON TULIIFERA

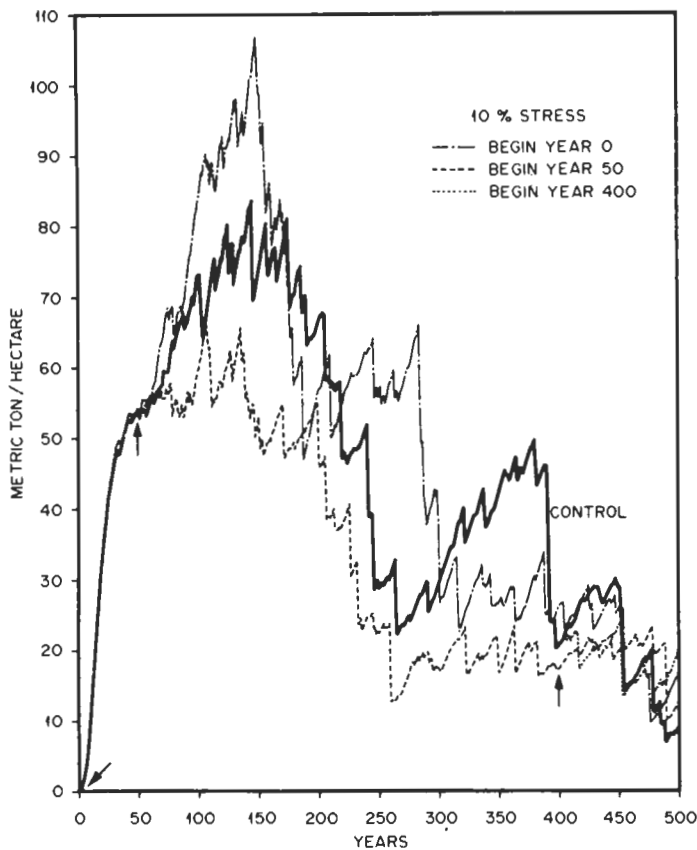


Figure 3--Response of yellow poplar (*Liriodendron tulipifera*) to a 10% reduction in growth. Growth reducing stress is applied at year 0, year 50 or year 400.

showed growth reduction in the more mature forest (initiation time - year 50) where other species compete more favorably in the closing forest canopy. Black oak, on the other hand, when stressed in the seedling forest showed a greatly enhanced growth reduction. When stress was initiated at year 50, however, the response was greatly delayed until other more resistant species such as white oak began to dominate (see fig. 2).

The effects of differential levels of sensitivity on growth and competition of forest trees which we have shown in figure 2 are supported by the field responses of deciduous trees measured by Brandt and Rhodes (1972, 1973). In their studies of the effects of 25 years of limestone dust deposition on a deciduous forest, they found changes in composition, with increased dominance of yellow poplar, white oak, and red oak at the site of heavy dust accumulation. Reduced lateral growth (≥ 18 percent) of sensitive species such as red maple, chestnut oak, and red oak was accompanied by a 76 percent increase in lateral growth of yellow poplar at the test site near the limestone quarry (Brandt and Rhodes 1973). Evidence of the amplification of effects of abiotic stress by both inter- and intra-specific competition has also been

QUERCUS VELUTINA

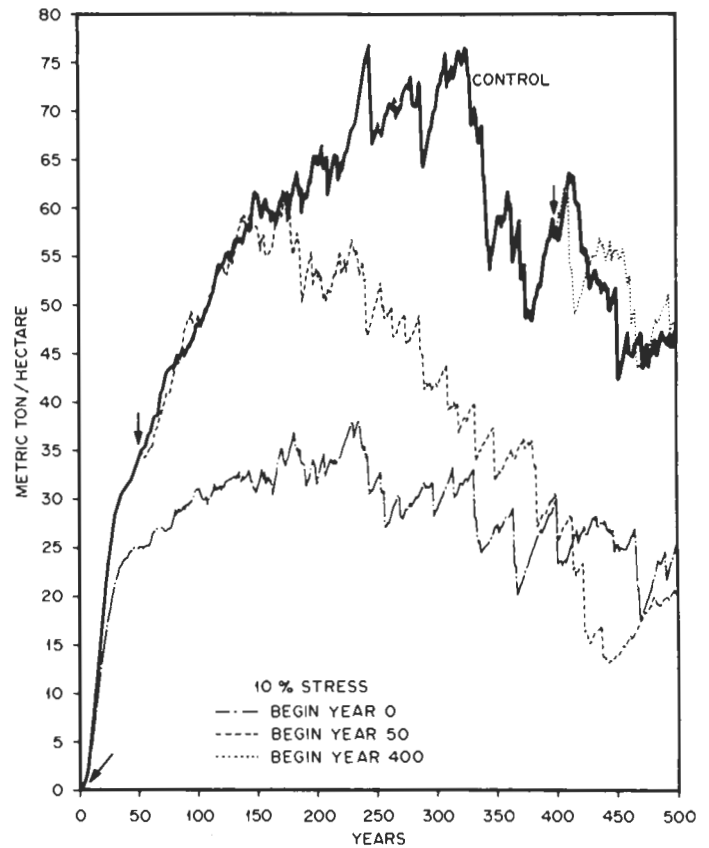


Figure 4--Response of black oak (*Quercus velutina*) to a 10% reduction in growth. Growth reducing stress is applied at year 0, year 50 or year 400.

demonstrated by Fox and Caldwell (1978) in studies with UV-B radiation. In situations of severe mutualistic competition, some species showed improved growth under the UV-B treatment, a response attributed to improved competitive status. Other examples of changes in plant competition under air pollution stress were reviewed by Guderian and Koppers (1980) in the preceding paper in this session.

Validation of Forest Community Response to Stress

While the validity of model results may be readily checked against actual growth and development patterns of "normal" forests of a region, evaluation of responses of disturbed forests becomes a much more difficult task. It implies developing a capability to clearly distinguish differences among measured values of parameters of stand growth and composition and those which would have occurred in the absence of pollutant stress. Accomplishing this necessitates either obtaining measurements on comparable stands over a variety of stress levels or documenting the growth characteristics of the stand in question before the stress was initiated. In either case, the investigator is faced with measuring pollutant effects in the face of the wide variety of biotic and abiotic variables controlling

growth of individual trees and forest communities.

Historically, documentation of forest responses to rather high levels of gaseous pollutants, primarily SO₂ and HF, from smelting processes was facilitated by the typical occurrence of well-defined gradients of stress with distance from the source. Gordon and Gorham (1963), for instance, were able to measure increased numbers of higher plant species along a 63-km gradient from the smelters at Sudbury, Ontario. These changes followed a generalized pattern of replacement of more highly evolved species of later successional stages by the more broadly adapted, stress-tolerant generalists which Woodwell (1970) reported following point-source radiation stress of a deciduous forest community.

Present-day air pollution stress regimes can generally be characterized as induced by generally lower levels of pollutants contributed by multiple sources. High-level point sources have been largely replaced by area sources where local topography and meteorology combine to concentrate multipoint effluents. Classic examples are the Los Angeles Basin in the West and numerous industrial corridors along river valleys in the East. These areas provide good possibilities for examining species and community responses to chronic and occasionally acute stress regimes.

Community-level effects of oxidants on forests of the San Bernardino Mountains near Los Angeles were described originally by Miller (1973) and have formed a basis for a broadly based study of a variety of ecosystem processes at this site. Kickert and Gimmel (1980) used these data in parameterizing a forest simulation model to describe these changes. In the East, McClenahan (1978) examined 7 deciduous forest stands located along a gradient of chronic air pollution stress on a 50-km portion of the heavily industrialized Ohio River Valley. Species richness, evenness, and Shannon diversity index were generally depressed for both overstory and understory layers in the forest as proximity to industrial air pollution sources increased. Stem density in the overstory decreased, while lower strata showed increased abundance of species along this same gradient. Shifts in relative species' importance were also noted.

Studies of the latter type provide very valuable data for describing the types of changes that may occur under moderate pollution stress, but are limited in their utility for predicting rates of change over time or at varying stress levels. Information of this type may be contained in the chronology of tree growth at that and other sites, however. Recent developments in tree-ring analysis provide a potentially powerful tool for analyzing both the rate and direction of within-community changes.

Dendroecology is a discipline of dendrochronology, the science of dating annual growth rings of woody plants (Fritts 1971). It can be considered a companion tool with dendroclimatology to examine changes in tree growth in relation to local and regional environment. The basic concepts, applications, and limitations of dendroecology have been discussed by Fritts (1971). In general, it relies on multivariate statistical analysis to identify principal variables influencing tree growth. Resultant equations are in themselves models of individual tree growth over time. As a tool for studying air pollution effects, dendroecology permits separation of effects of tree age and local climate from those induced by air pollution (Nash and others 1975). Phillips and others (1977a,b) have used this approach to correlate growth reductions in stands of loblolly and white pine with production levels near an army munitions plant. More relevant to the challenges of providing reliable predictions of species and community-level changes is the potential utility of this technique for detecting growth responses in our eastern regional environment. Measurements of growth reductions of white oak in apparent response to chronic stress of this type have been reported near LaPorte, Indiana, by Ashby and Fritts (1972). In this case, the decade during which anomalous growth reductions occurred was associated with a heavy incidence of smoke and haze in that region.

Documentation of pollutant histories in the broader regional context represents a more difficult task but one of great importance to efforts to eventually develop a predictive potential. A greatly expanded network of air quality trends; however, data for the past 40 years, during which emissions in the Eastern United States increased sharply, are lacking. One potentially useful tool for obtaining histories of exposure to general air pollution stress is heavy metal analysis in the individual rings (Lepp 1977). This approach has been used in Sweden (Symeonides 1979) to construct histories of heavy metal pollution, although Tian and Lepp (1975) caution that factors such as radial transport and soil uptake must be fully understood to use this technique accurately. In the Swedish study, both lead and copper showed little lateral movement and were useful in constructing a decade-level history of metal pollution at the study site. Recent developments coupling x-ray emission spectroscopy (Valkovic and others 1979) with growth-ring analysis show promise for using a variety of trace elements for historical analyses. As these techniques are developed further, they may provide useful data for constructing historical indices of regional-scale chronic stress.

The tools for validating or modifying forest simulators as predictive tools appear to be either available now or close at hand. We feel that dendroecological approaches have tremendous potential for unlocking a wide variety of

species/community/environment interactions which will make this task ultimately possible. Probably, the greatest value of the forest simulators is in predicting the consequences of sets of "most logical" assumptions regarding pollution effects on trees. Other assumed relationships can be tested easily, and new information may be added as it is developed (Kozlowski 1980). The model is merely a tool to be used in this synthesis and refining process.

LITERATURE CITED

- Adlard, P. G.
1974. Development of an empirical competition model for individual trees within a stand. In *Growth models for trees and stand simulation*. J. Fries, ed. p. 22-37. Res. Notes 30. Department of Forest Yield Research, Royal College of Forestry, Stockholm.
- Arney, J. D.
1974. An individual tree model for stand simulation in Douglas-fir. In *Growth models for trees and stand simulation*. J. Fries, ed. p. 38-40. Res. Notes 30. Department of Forest Yield Research, Royal College of Forestry, Stockholm.
- Ashby, W. C., and H. C. Fritts.
1972. Tree growth and climate near LaPorte, Indiana. *Bull. Am. Meteorol. Soc.* 53:246-251.
- Bella, I. E.
1971. Simulation of growth, yield and management of aspen. Ph.D. Thesis, University of British Columbia, Vancouver.
- Bosch, C. A.
1971. Redwoods: A population model. *Science* 162:345-349.
- Botkin, D. B.
1973. Estimating the effects of carbon fertilization on forest composition by ecosystem simulation. Proc., 24th Brookhaven Symposium in Biology. In G. M. Woodwell and E. V. Pecan. p. 384-444. CONF-720510. National Technical Information Service, Springfield, Virginia.
- Botkin, D. B.
1977. Forest, lakes, and the anthropogenic production of carbon dioxide. *BioScience* 27:325-331.
- Botkin, D. B., J. F. Janak, and J. R. Wallis.
1972. Some ecological consequences of a computer model of forest growth. *J. Ecol.* 60:849-872.
- Brandt, C. J., and R. W. Rhodes.
1972. Effects of limestone dust accumulation on composition of a forest community. *Environ. Pollut.* 3:217-225.
- Brandt, C. J., and R. W. Rhodes.
1973. Effects of limestone dust accumulation on lateral growth of forest trees. *Environ. Pollut.* 4:207-213.
- Burkhart, H. E., and M. R. Strub.
1974. A model for the simulation of planted loblolly pine stands. In *Growth models for trees and stand simulation*. J. Fries, ed. p. 128-135. Res. Notes 30. Department of Forest Yield Research, Royal College of Forestry, Stockholm.
- Clutter, J. L.
1963. Compatible growth and yield models for loblolly pine. *For. Sci.* 9:354-371.
- Clutter, J. L.
1974. A growth and yield model for *Pinus radiata* in New Zealand. In *Growth models for trees and stand simulation*. J. Fries, ed. p. 136-160. Res. Notes 30. Department of Forest Yield Research, Royal College of Forestry, Stockholm.
- Curtis, R. O.
1967. A method of estimation of gross yield of Douglas-fir. *For. Sci. Monogr.* 13:1-24.
- Davis, D. D., and R. G. Wilhour.
1976. Susceptibility of woody plants to sulfur dioxide and photochemical oxidants. EPA-600/3-76-102. 71 p. USEPA, Corvallis, Oregon.
- Doyle, T. W., H. H. Shugart, and D. C. West.
1980. A model of Puerto Rican tropical montane forest. M.S. Thesis, University of Tennessee, Knoxville, Tennessee.
- Dress, P. E.
1970. A system for the stochastic simulation of even-aged forest stands of pure species composition. Ph.D. Thesis, Purdue University, Lafayette, Indiana.
- Drury, W. H., and I. C. T. Nesbit.
1973. Succession. *J. Arnold Arbor. Harvard Univ.* 54:331-368.
- Elfving, B.
1974. A model for the description of the structure in unthinned stands of Scots pine. In *Growth models for trees and stand simulation*. J. Fries, ed. p. 181-189. Res. Notes 30. Department of Forest Yield Research, Royal College of Forestry, Stockholm.
- Ek, A. R., and R. A. Monserud.
1974. FOREST: A computer model for simulating the growth and reproduction of mixed forest stands. Research Report A2635. p. 1-14 + 3 appendices. College of Agricultural and Life Sciences, University of Wisconsin, Madison.

- E.P.A.
1977. National Air Quality, Monitoring, and Emissions Trends Report. 1977. EPA-450/2-78-052. Environmental Protection Agency. Research Triangle Park, North Carolina. 22 p.
- Forcier, L. K.
1975. Reproductive strategies and the co-occurrence of climax tree species. *Science* 189:808-809.
- Fox, F. M., and M. M. Caldwell.
1978. Competitive interaction in plant populations exposed to supplementary ultraviolet-B radiation. *Oecologia* 36:173-190.
- Fritts, H. C.
1971. Dendroclimatology and dendroecology. *Quat. Res.* 1:419-449.
- Gleason, H. A.
1926. The individualistic concept of the plant association. *Bull. Torrey Bot. Club* 53:7-26.
- Gordon, A. G., and E. Gorham.
1963. Ecological aspects of air pollution from an iron sintering plant at Wawa, Ontario. *Can. J. Bot.* 41:1063-1078.
- Goulding, C. J.
1972. Simulation techniques for a stochastic model of the growth of Douglas-fir. Ph.D. Thesis, University of British Columbia, Vancouver.
- Guderian, R., and K. Kuppers.
1980. Responses of plant communities to air pollution. In *Proc., USFA Symposium on Effects of Air Pollutants on Temperate Forest Ecosystems*. In press.
- Hatch, C. R.
1971. Simulation of an even-aged red pine stand in northern Minnesota. Ph.D. Thesis, University of Minnesota, St. Paul.
- Hegy, F.
1974. A simulation model for managing jack-pine stands. In *Growth models for trees and stand simulation*. J. Fries, ed. p. 74-87. Res. Notes 30. Department of Forest Yield Research, Royal College of Forestry, Stockholm.
- Hool, J. N.
1966. A dynamic programming - Markov chain approach to forest production control. *For. Sci. Monogr.* 12:1-26.
- Horn, H. S.
1976. Succession. In *Theoretical ecology: Principles and applications*. R. M. May, ed. p. 187-204. Blackwell Scientific Publishers, Oxford.
- Johnson, W. C., and D. M. Sharpe.
1976. An analysis of forest dynamics in the Northern Georgia Piedmont. *For. Sci.* 22:307-322.
- Kozlowski, T. T.
1980. Impacts of air pollution on forest ecosystems. *BioScience* 30:88-93.
- Kickert, R. N., and B. Gimmel.
1980. Data based forest ecosystem simulation implications of possible future trends of ozone air pollution in Southern California. In *Proc., USFS Symposium on Effects of Air Pollutants on Temperate Forest Ecosystems*. In press.
- Leak, W. B.
1970. Successional change in northern hardwoods predicted by birth and death simulation. *Ecology* 51:794-801.
- Lee, Y.
1967. Stand models for lodgepole pine and limits to their application. *For. Chron.* 43:387-388.
- Lepp, N. W.
1975. The potential of tree-ring analysis for monitoring heavy metal pollution patterns. *Environ. Pollut.* 9:49-61.
- Lin, J. Y.
1970. Growing space index and stand simulation of young western hemlock in Oregon. Ph.D. Thesis, Duke University, Durham, North Carolina.
- Lin, J. Y.
1974. Stand growth simulation models for Douglas-fir and western hemlock in the Northwestern United States. In *Growth models for trees and stand simulation*. J. Fries, ed. p. 102-118. Res. Notes 30. Department of Forest Yield Research, Royal College of Forestry, Stockholm.
- McClenahan, J. R.
1978. Community changes in a deciduous forest exposed to air pollution. *Can. J. For. Res.* 8:432-438.
- McLaughlin, S. B., and N. T. Lee.
1974. Botanical studies in the vicinity of the Widows Creek Steam Plant. Review of air pollution effects studies, 1952-1972, and results of 1973 surveys. TVA I-EB-74-1. 62 p. Tennessee Valley Authority, Chattanooga, Tennessee.
- McLaughlin, S. B., D. C. West, H. H. Shugart, and D. S. Shriner.
1978. Air pollution effects on forest growth and succession: Applications of a mathematical model. In *Proc., 71st Meeting of the Air Pollution Control Association*. H. B. H. Cooper, Jr., ed. In press.

- Mielke, D. L., H. H. Shugart, and D. C. West.
1978. A stand model for upland forests of southern Arkansas. ORNL/TM-6225. Oak Ridge National Laboratory, Oak Ridge, Tenn.
- Miller, P. R.
1973. Oxidant-induced community change in a mixed conifer forest. In Air pollution damage to vegetation. J. A. Naegele ed. p. 101-117. Adv. in Chem. Ser. 122. American Chemical Society, Washington, D.C.
- Mitchell, K. J.
1969. Simulation of growth of even-aged stands of white spruce. Yale Univ. Sch. For. Bull. 75:1-48.
- Mitchell, K. J.
1975. Dynamics and simulated yield of Douglas-fir. For. Sci. Monogr. 17:1-39.
- Moser, J. W., and O. F. Hall.
1969. Deriving growth and yield functions for uneven-aged forest stands. For. Sci. 15:183-188.
- Munro, D. D.
1974. Forest growth models - a prognosis. In Growth models for trees and stand simulation. J. Fries, ed. p. 7-21. Res. Notes 30. Department of Forest Yield Research, Royal College of Forestry, Stockholm.
- Namkoong, G., and J. H. Roberts.
1974. Extinction probabilities and the changing age structure of redwood forests. Am. Nat. 108:355-368.
- Nash, T. H., H. C. Fritts, and M. A. Stokes.
1975. A technique for examining non-climatic variation in widths of annual tree rings with special reference to air pollution. Tree-Ring Bull. 35:14-24.
- Newnham, R. M.
1964. The development of a stand model for Douglas-fir. Ph.D. Thesis, University of British Columbia, Vancouver.
- Noble, I. R., and R. O. Slatyer.
1978. The effect of disturbance on plant succession. Proc. Ecol. Soc. Aust. 10. In press.
- Olson, J. S., and G. Christofolini.
1966. Model simulation of Oak Ridge vegetation succession. p. 106-107 in ORNL/TM-4007. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Phillips, S. O., J. M. Skelly, and H. E. Burkhart.
1977a. Growth fluctuation of loblolly pine due to periodic air pollution levels: Interaction of rainfall and age. Phytopathology 67:716-720.
- Phillips, S. O., J. M. Skelly, and H. E. Burkhart.
1977b. Eastern white pine exhibits growth retardation by fluctuating air pollutant levels: Interaction of rainfall, age, and symptom expression. Phytopathology 67:721-725.
- Reichle, D. E., R. V. O'Neill, and J. S. Olson.
1973. Modeling forest ecosystems. Report of International Woodlands Workshop. U.S. International Biological Program [August 14-16, 1972, Oak Ridge National Laboratory], 339 p.
- Shugart, H. H., T. R. Crow, and J. M. Hett.
1973. Forest succession models: A rationale and methodology for modeling forest succession over large regions. For. Sci. 19:203-212.
- Shugart, H. H., and D. C. West.
1977. Development of an Appalachian deciduous forest succession model and its application to assessment of the impact of the chestnut blight. J. Environ. Manage. 5:161-179.
- Shugart, H. H., A. T. Mortlock, M. S. Hopkins, and I. P. Burgess.
1979. A computer simulation model of ecological succession in Australian subtropical rainforest. ORNL/TM-7029. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Shugart, H. H., and I. R. Noble.
1980. A computer model of succession and fire response of the high altitude Eucalyptus forests of the Brindabella Range, Australian Capital Territory. Austr. J. Ecol. In press.
- Shugart, H. H., and D. C. West.
1980. Forest succession models. BioScience (in press).
- Solomon, D. S.
1974. Simulation of the development of natural and silviculturally treated stands of even-aged northern hardwoods. In Growth models for trees and stand simulation. J. Fries, ed. p. 327-352. Res. Notes 30. Department of Forest Yield Research, Royal College of Forestry, Stockholm.
- Sullivan, A. D., and J. L. Clutter.
1972. A simultaneous growth and yield model for loblolly pine. For. Sci. 18:76-86.
- Suzuki, T., and T. Umemura.
1974. Forest transition as a stochastic process II. In Growth models for trees and stand simulation. J. Fries, ed. p. 358-379.

- Symeonides, C.
1979. Tree-ring analysis for tracing the history of pollution: Application to a study in Northern Sweden. *J. Environ. Qual.* 8:482-486.
- Tian, T. K., and N. W. Lepp.
1977. Further studies on the development of tree-ring analysis as a method for constructing heavy metal pollution histories. *Proc., Univ. Mo. Annu. Conf. Trace Substances.* *Environ. Health* 11:440-447.
- Treshow, M.
1970. The impact of air pollutants on plant populations. *Phytopathology* 58:1108-1113.
- Tharp, M. L.
1978. Modeling major perturbations on a forest ecosystem. M.S. Thesis, University of Tennessee, Knoxville. 52 p.
- Valkovic, V., D. Rendic, E. K. Biegert, and E. Andrade.
1979. Trace element concentrations in tree rings as indicators of environmental pollution. *Environ. Int.* 2:27-32.
- Waggoner, P. E., and G. R. Stephens.
1970. Transition probabilities for a forest. *Nature* 225:1160-1161.
- Watt, K. E. F.
1966. The nature of systems analysis. In *Systems analysis in ecology.* K. E. F. Watt, ed. p. 1-14. Academic Press, New York.
- West, D. C., S. B. McLaughlin, and H. H. Shugart.
1980. Simulated forest response to chronic air pollution stress. *J. Environ. Qual.* 9:43-49.
- Wilkins, C. W.
1977. A stochastic analysis of the effect of fire on remote vegetation. Ph.D. Thesis, University of Adelaide, South Australia.
- Wolak, J.
1971. Studies on the industrio-climax in Poland. *Proc., IV Congress of IUFRO - Methods for the identification and evaluation of air pollutants injurious to forests.* p. 233-244.
- Woodwell, G. M.
1970. Effects of pollution on the structure and function of ecosystems. *Science* 168:429-433.

Poster Summaries

Effects of Oxidant Air Pollutants on Pine Litter-fall and the Forest Floor¹

Rodney J. Arkley and Rudolph Glauser²

Oxidant injury to western yellow pines (*Pinus ponderosa* and *P. jeffreyi*) in the San Bernardino Mountains results in needle injury followed by increased fascicle mortality and abscission, decreased needle length, branch mortality and finally tree mortality. The degree of injury has been estimated each year for 6 years by a scoring system based on observation of all of these factors with binoculars and combining them into a single oxidant injury score (OIS). Foliage density is directly proportional to OIS as expected. The number of annual needle-whorls retained (W) is related to the score by $W = 0.171 \times OIS$ with $r = .75^{**}$ and the grams of needles per twig by $F = 2.71 \times OIS$ with $r = .62^*$. Note that OIS decreases with increasing injury and a score of 0 indicates a dead tree. Scores greater than 35 indicate no obvious injury.

EFFECT ON NEEDLE-FALL

The oven-dry weight of needles collected on 0.209 m² screens placed under pines of varying OIS is shown in Figure 1. The average annual needle fall increases from 131 gm/m² under healthy trees to 357 gm/m² with OIS of 9 to 14, and decreases as the tree nears death.

The weight per needle fascicle in the litter-fall decreases progressively with increasing injury as shown also in Figure 1. (Sig. < .001).

The increased litter-fall (170 percent) can be expected to increase the thickness of loose dry litter on the forest floor with consequent increased fire hazard and decreased seedling germination.

Plant Nutrient Content

Litter-fall samples were analyzed to determine the effect of oxidant injury on the plant nutrient of the litter. The results are shown in Figure 2. The scatter of points (not shown) represented by the regression lines is wide, but 187 samples were analyzed and the trends indicated are clearly real, since they are highly significant ($P = < .001$). Magnesium was also analyzed but

showed no trend whatsoever.

The response of these elements is perhaps due to declining cell wall thickness with its calcium pectate and an inverse dilution effect on N, P, and K.

The data for the regression lines shown in Figure 2 are as follows:

$$N(\text{pct}) = .598 - .00255 \text{ OIS}, r = -.264, n = 187$$

$$P(\text{pct}) = .0707 - .00034 \text{ OIS}, r = -.28, n = 185$$

$$K(\text{pct}) = .346 - .00331 \text{ OIS}, r = -.37, n = 187$$

$$Ca(\text{pct}) = .303 + .00431 \text{ OIS}, r = .58, n = 147$$

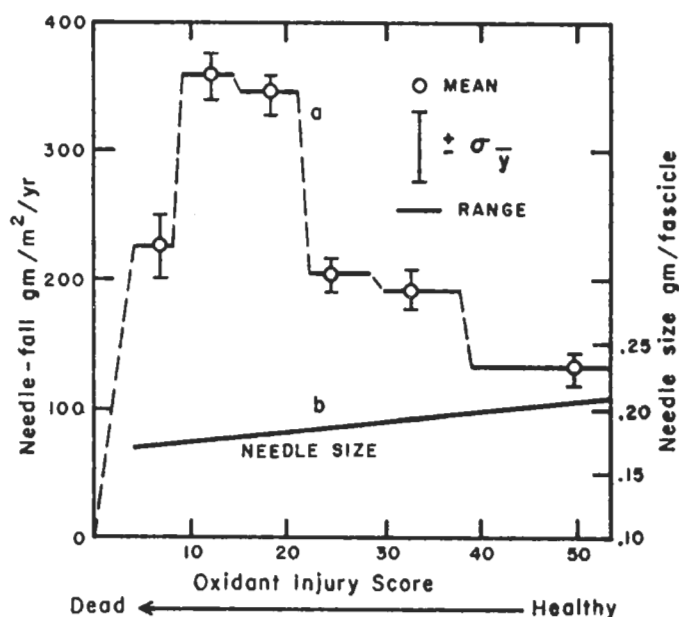


Fig. 1. Effect of oxidant injury on pine needle fall and needle size.

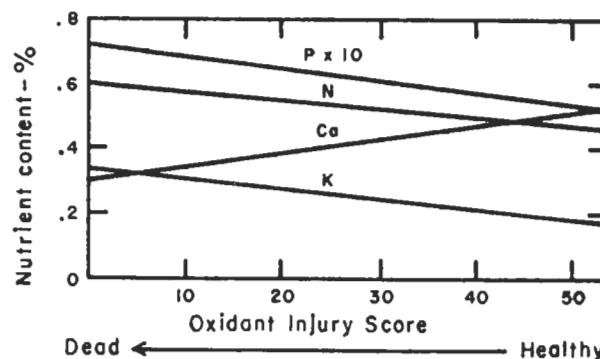


Fig. 2. Needle-fall content of plant nutrients.

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Regional Air Pollution Impacts on Forest Growth¹

Thomas V. Armentano, Orie L. Loucks, and Wayne T. Williams²

Recent studies underway in the Ohio River Basin Energy Study (Loucks 1980) have shown that chronic air pollution levels may be reducing regional growth over much of eastern North America. During June through August, 1977, monitoring stations representative of forest areas recorded hourly maxima ≥ 0.10 ppm on 14 to 27 percent of the days, and maxima ≥ 0.05 ppm on 70 to 93 percent of the days.

A survey of eastern white pine (*Pinus strobus* L.) stands in rural and urban locations throughout central and southern Indiana showed widespread ozone damage symptoms: chlorotic mottling, chlorotic dwarfing and premature needle senescence, on a scale from moderate to severe. Sycamore (*Plantanus occidentalis* L.), silver maple (*Acer saccharinum* L.), yellow poplar (*Liriodendron tulipifera* L.), and the black oak group (*Quercus* spp.) also are somewhat sensitive. The 37.3×10^6 acres of forests in the Ohio Basin yield about $40 \text{ ft}^3/\text{acre}^{-1}$ of wood annually, but an estimated 25 percent of the forest consists of O_3 -sensitive species, indicating a yield reduction from oxidant effects (and interactions with other gases and pathogens), ranging from 3 to 6 percent annually (Table 1).

Table 1--Total loss in wood production in Ohio River Basin forests estimated to result from direct and indirect air pollutant effects upon forest growth and mortality rates. Total normal wood yield for the region in 1970 was $1.5 \times 10^9 \text{ ft}^3$. Data expressed in millions of cubic feet.

Year	Wood Production Loss		Annual Mortality Loss		Total Loss
	(Pct.)	(10^6 ft^3)	(Pct.)	(10^6 ft^3)	(Pct.)
1970			2.7	40.3	
1977	3-6	45-90	5.4	80.6	8.3-11.3
1985	5-11	75-166	10.8	162.8	15.8-21.9
2000	5-11	75-166	10.8	162.8	15.8-21.9

Independent studies indicate O_3 levels will increase in the Ohio River Basin over the next 20 years, depending on increases in utility NO_x emissions. A conservative energy development scenario

suggests that O_3 levels will continue upward at least through 1985. At the current rate of O_3 increase, 0.02 ppm per decade, wood losses could reach $166 \times 10^6 \text{ ft}^3$, twice the current estimated losses (Table 1).

A pattern of increasing tree mortality in the relatively young eastern forests (30 percent mortality increase in the Northeast, and 10 percent in the South from 1962 to 1970) has been reported during a period of significant air pollution increases in the Northeast (U.S. Forest Service 1978). This mortality may be attributable, at least in part, to degradation of air quality. This hypothesis is supported by nearly constant mortality in western forests where air pollution is generally low, despite overmaturity in these forests. If mortality losses in the Ohio Basin were intermediate between those of the Northeast and South from 1962 to 1970, the loss of wood would be $4.08 \text{ ft}^3/\text{acre}/\text{yr}^{-1}$. Proportionately greater losses are indicated for 1985 and 2000 (Table 1).

Other studies underway indicate these forest losses could be significant for the global CO_2 balance (Armentano and Ralston 1980). Because of a favorable stand age distribution brought on by past harvest patterns, temperate zone forests now store around 10^9 tons of carbon annually in long-lived tree components, 20 percent as much as the carbon released from fossil-fuel combustion. This storage could continue for the next two decades, but increased harvest, forest maturation, and air pollution effects can reduce carbon sequestering and wood production rates. Thus, only management of forests focused on a balance between wood production and wood accumulation can provide optimum economic and ecological benefits. If present air pollution trends continue, and if relationships to forest growth suggested in this paper are substantiated by further research, the long-term productivity of forests will be threatened in several regions of the United States.

LITERATURE CITED

- Armentano, T.V., and C.W. Ralston.
1980. The role of temperate zone forests in the global carbon cycle. *Can. J. For. Res.* 10:53-60.
- Loucks, O.L. (ed.).
1980. Crop and Forest Losses Due to Current and Projected Emissions from Coal-fired Power Plants in the Ohio River Basin. TIE Report in review. Indianapolis.
- U.S. Forest Service.
1978. Forest Statistics of the U.S. 1977. Department of Agriculture.

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Canopy Analysis of Pollutant Injured Ponderosa Pine in the San Bernardino National Forest^{1,3}

M.C. Axelrod, P.I. Coyne, G.E. Bingham, J.R. Kercher, P.R. Miller, and R.C. Hung²

Nine trees were selected from a ponderosa pine (*Pinus ponderosa* Laws.) stand which was established after a fire in the mid-1950's. These trees were classified into three injury groups, [(1) slight, (2) moderate, and (3) severe injury] in accordance with a scoring devised by P.R. Miller. During the summers of 1978 and 1979, a detailed inventory was made of the canopy on each tree. The lengths of all main stem internodes were measured along with the number of primary branches radiating from each of these internodes. At roughly every other main stem internode, several of the primary branches received a detailed inventory. The length of the primary branch and the number of internodes were recorded. This inventory was carried on through to the secondary, tertiary, and quaternary branch levels. The compass angle of each of the inventoried primary branches was also recorded. Whenever needles were found on an internode, the following information was recorded: (1) needle age, (2) needle condition on a scale of 0-4, (3) the number of fascicles, (4) the average needle length, (5) the average needle chord width, and (6) the length of the internode bearing the needles. Note that while not all the primary branches at a selected main stem internode were inventoried, the ones selected did receive a complete inventory.

A preliminary analysis of the 1978 canopy data has been completed, characterizing the distribution of needle surface area for each tree. The total needle surface area for each inventoried primary branch was computed separately for each needle age. In order to determine the distribution of needle surface area for a whole tree, we estimated the needle area at the main stem internodes where no data was taken. The curve representing needle area as a function of height is approximately bell-shaped. We have developed an algorithm designed to yield estimates of the missing points and a smooth curve. The algorithm is iterative and uses linear interpolation be-

tween known points to produce a set of initial estimates. The algorithm first computes the cumulative needle surface area as a function of height, then a linearizing transformation is made. Linear interpolation is carried out on this new curve. The transformation is then inverted and first differences taken, resulting in a reproduction of the original data and a new set of estimates for the missing data. The new estimates are used as initial values each iteration. A weighted average of the needle conditions for each primary branch was also computed. These averages are then combined to yield an index of the condition of the needles on the whole tree.

In table 1A, we see the total leaf area index for each needle age class, with the trees grouped into injury classifications. We can see the leaf area indices decrease across injury classes within age groups. There is a pronounced decrease in the retained leaf area with needle age even in the slightly injured group; the severely injured group has essentially no needle area except in the 1 year age category. Table 1B shows the (weighted) average needle condition for the inventoried trees and presents further evidence that ozone injury is dose accumulative with young needles being less affected by ozone than older needles. Since the current year needles were still growing at the time the initial inventory was taken, they were inventoried separately the following year. This data is still being processed and is not yet available, but is expected to add proportionately to the numbers presented here. The trend in leaf area decline indicates the competitive disadvantage of an injured tree.

Table 1--Needle properties by needle age for nine trees under pollution stress.¹

Injury Class	(A) Leaf Area Index		
	1 Year	2 Year	3 Year
Slight	9.53	4.13	0.70
Moderate	5.23	0.73	0.00
Severe	3.55	0.03	0.00
	(B) Average Needle Condition ² of Canopy		
	1 Year	2 Year	3 Year
Slight	0.96	1.31	1.28
Moderate	2.18	2.42	NA
Severe	2.63	NA	NA

¹Does not include data for current year needles. Averages for 3 trees in each injury class.

²Needle condition scale: 0 = green; 2 = chlorotic mottle. 4 = uniform yellow with necrosis.

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Photosynthesis and Stomatal Behavior in Ponderosa Pine Subjected to Oxidant Stress: Water Stress Response^{1,2}

Gail E. Bingham and Patrick I. Coyne³

Light response curves for net and/or gross photosynthesis, stomatal conductance, and needle xylem potential of ponderosa pine (*Pinus ponderosa* Laws.) subjected to water and/or ozone stress were studied in the field and laboratory. In the field, measurements were made on a stand of ponderosa pine in the San Bernardino National Forest, which has experienced long-term oxidant fumigations from the south coast air basin since their establishment following fire in the mid-1950's. These trees were stratified for comparative studies into three groups (slight injury, moderate injury, and severe injury) having similar oxidant symptoms, on the basis of the scoring system of P.R. Miller.

Controlled studies using ten healthy, uniform, Oregon-grown, six-year-old saplings growing in 55 l containers were conducted to elucidate specific field responses. These trees were approximately 2 m tall, and were randomly allocated to positions in two naturally-lighted, mylar-covered, air and humidity conditioned exposure chambers. One chamber was supplied charcoal filtered air and the other with air containing 0.01 ppm ozone for six hours/day during the midday period.

Measurements were made at regular intervals from May through October at the forest site and during the fumigation and water stress cycles in the laboratory. During the laboratory study, net photosynthesis (P_n) and stomatal conductance (C_s) measurements were made at constant humidity and temperature using the LLNL developed minicuvette system, with only the fascicle being measured and a few surrounding needles exposed to light. The rest of the tree was surrounded with heavy black cloth from before dawn until after light response curves had been measured on three fascicles.

The relationship between needle xylem potential (ψ_x) and maximum stomatal conductance (C_{max}) was not single valued, and depended on predawn

ψ_x . When predawn ψ_x was in the range from -3 to -5 bars, C_s (and P_n) decreased only slightly during the normal daily decrease in ψ_x due to changes in the diurnal course of irradiance. Minimum ψ_x under these conditions seldom exceeded -14 bars. However, when ψ_x was forced below -15 to -17 bars (by severing the fascicle from the branch) a sharp decrease in C_s occurred, with complete stomatal closure ($C_s < 0.01$ cm/s) in the range of -36 to -40 bars. When predawn ψ_x started in the -5 to -8 bar range, however, a bilevel relationship between C_s and ψ_x was observed, with C_{max} reaching an initial full light value, followed at some later time by a step decrease to a lower value. Conductance and P_n remained at this significantly lower level throughout most of the day, with an accelerated closing trend toward the late afternoon. Late afternoon values were usually only 60 to 70 percent of their light corrected morning opening value.

Care had to be taken when interpreting pre-dawn ψ_x measurements in the field. It was often observed that at a predawn ψ_x of -3 to -7 bars, the stomata would partially open in the predawn hours, resulting in C_s values approaching one-tenth of their full light value. Under these conditions, sufficient transpiration occurred to make predawn ψ_x very sensitive to wind speed. Differences between predawn ψ_x on calm and windy mornings as large as 3 bars were observed.

The second factor depended solely on the predawn xylem potential and controlled the maximum stomatal conductance (C_{max}) observed during the diurnal cycle. This relationship could not be adequately examined in the forest due to untimely late season rains during the three years that field studies were conducted, and was investigated using potted trees. As predawn ψ_x decreased below -7 bars in control trees and about -10 bars in fumigated trees, a drastic reduction in C_{max} and P_n was observed. In trees kept in filtered air, C_{max} was decreased from 0.36 to 0.036 cm/s by the decrease of predawn ψ_x from -5 to -15 bars. Since the majority of the tree was kept in the dark during the measurement, the relationship between C_s and ψ_x at ψ_x values greater than can normally be observed in nature (due to root and xylem resistance) were examined. Conductance and P_n of fumigated trees in this region was not significantly higher than that observed at potentials associated with full illumination. Reductions in C_{max} of 25, 31, and 44 percent were measured for slight, moderate, and severely injured needles. Net photosynthesis for the same needle injury classes was reduced by 38, 54, and 69 percent from the 12.9 mgCO₂/dm²-h rate measured in trees which had not been exposed to ozone.

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Oxidant Impact on Ponderosa and Jeffrey Pine Foliage Decomposition¹

J. N. Bruhn, J. R. Parmeter, Jr., and F. W. Cobb, Jr.²

Litter decomposition was studied on four plots located along the oxidant dosage gradient in the San Bernardino Mountains (SBNF). Precipitation decreased with decreasing oxidant dosage along this gradient, while forest floor-level solar radiation and temperature increased. Ponderosa (*Pinus ponderosa* Dougl.) and Jeffrey pine (*Pinus jeffreyi* Grev. & Balf.) dominated the most and least severely impacted sites, respectively. Nylon mesh envelopes of fresh-fallen litter were exchanged among healthy and sick trees on the four study sites. Decomposition was measured as percents overall and nutrient (N, P, K, Ca and Mg) weight loss.

During the first two years of decomposition, its rate at each site was directly related to precipitation and oxidant dose. While site moisture apparently dominates litter decomposition, evidence suggests that oxidant injury to live needles is directly related to the rate at which they decompose. One year weight loss of ponderosa pine litter was negatively correlated ($P < .05$) with the oxidant injury scores (O.I.S.) (Miller 1973) of litter source trees. One year weight loss by litter of both species was negatively correlated ($P < .05$) with the O.I.S. of litter destination trees.

Live needle internal microflora may be initiated in the foliar bud and can be thought of as pioneers in a succession of microorganisms responsible for litter decomposition. Microbial populations involved in foliage decomposition were studied via incubation of surface-sterilized live and litter needles on water agar. Eight trees from each of two central Sierra Nevada sites, Stanislaus National Forest (SNF) and Blodgett Experimental Forest (BEF), were included for comparison with the 15 SBNF study trees. Both taxonomic richness and population density increased with needle age. Both parameters increased similarly with age on all four SBNF sites. However, both parameters increased faster on the somewhat less oxidant-impacted BEF, and both parameters developed fastest on the least oxidant-impacted site (SNF).

Because not all fungi recorded in incubation studies were identified, they are individually referred to as categories rather than species. Live SNF foliage yielded approximately twice as many fungus categories (60) as foliage from any other site. The two BEF plots harbored a few more categories (29 and 32) than did HV (27), the healthiest SBNF site. HV, in turn, provided more categories than any other SBNF site (15 to 20). Similar effects were not evident in forest floor litter. Oxidants apparently affect live needle microflora in two ways. By reducing needle longevity, internal microflora development is prematurely truncated. It also seems likely that oxidants further reduce the variety of fungi colonizing live pine foliage by eliminating susceptible species. Insofar as community functional properties are stabilized by a combination of species adaptability and community diversity, reduction of live foliage microfloral diversity by oxidants is viewed as weakening the functional stability of these communities. The significance of such weakening is unknown.

In a growth chamber experiment, propylene oxide-sterilized pine needles were incubated on moist forest floor organic matter from either the SNF or one of the SBNF sites. No meaningful differences in weight loss developed between treatments over 22 weeks, showing that oxidants to date had not significantly impaired the abilities of study-site microflora to cause litter weight loss. In a second growth chamber experiment, propylene oxide-sterilized pine needles were incubated on a uniform moist forest floor organic matter mixture in either filtered air or filtered air enriched with 20pphm ozone 8 hours daily for 14 weeks. The weight loss difference between treatments did not reach significance ($.05 > P > .01$). Any effect in the field would be slight and probably limited to surface litter during moderate to severe oxidant episodes.

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LITERATURE CITED

- Miller, P. R.
1973. Oxidant-induced community change in a mixed conifer forest. In *Air pollution damage to vegetation*. p. 101-117. *Advances in Chemistry Series*, Number 122. American Chemical Society.

Integrated Lake-Watershed Acidification Study¹

Carl W. Chen and Robert A. Goldstein²

OBJECTIVES

The Integrated Lake-Watershed Acidification Study (ILWAS) was designed to determine the ecological effects of acid rain under natural conditions (EPRI, 1979, Goldstein *et al.*, 1980). Since the most widely reported effect of acid rain has been the acidification of lake water leading to elimination of fish, it is of interest to learn how and why the ecosystem becomes acidified by acid rain.

Three forested watersheds (Panther, Woods, and Sagamore) within 15 km of each other in the Adirondack Park region of New York were selected for investigation. Each watershed has different configurations and characteristics. Principle hypothesis of the study is that these differences may lend to different pH dynamics, i.e., Panther Lake alkaline, Woods Lake acidic, and Sagamore Lake in-between.

This research will produce a comprehensive data base for the three watersheds covering a period of almost 4 years, a series of interpretive reports, and a calibrated and verified mathematical model. The wealth of understanding gained and the mathematical model developed will be readily applicable to other lake basins.

APPROACH

ILWAS couples field investigation with theoretical modeling. The interactions between the model and the field research are practiced in an interactive manner, each influencing and strengthening the other. Other research findings on mechanisms and rates of acidification processes are integrated into the model formulations.

FIELD PROGRAM

The watershed ecosystem is envisioned to comprise a cascade of basic compartments: atmosphere, canopy, snowpack, catchment, soil layers, bogs, stream segment and lake. These are the compartments that the acid rain must pass through before it reaches lake outlet. As it passes through each compartment, biogeochemical processes acting in

series and in parallel will produce or consume acids and will release chemicals that shift the pH and other chemical equilibrium.

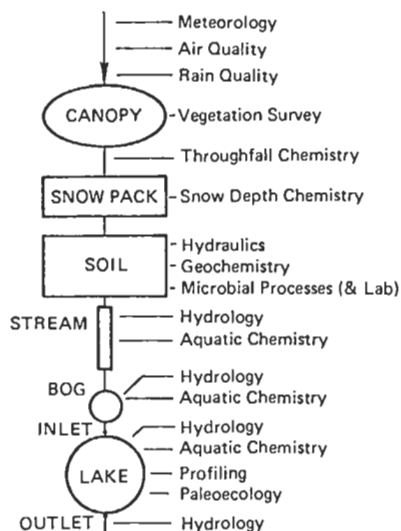


Figure 1--Field Program Components

Field surveys are being conducted to characterize the properties of the basic ecosystem compartments in each watershed. At selected locations, measurements are made for ambient air quality, the quantity and quality of waters that move through the system from tree top to lake outlet (see fig. 1). Data are collected monthly, weekly, synoptic, and once only, depending on the parameters and their temporal variability. The field program began in 1977 and will be completed in 1981.

MODELING

The model organizes the data into an integrated theoretical framework (Goldstein *et al.*, 1980). The model also serves as a vehicle to check the consistency of theory and data from rainfall quantity to lake outlet quality. Eventually, the model may provide scientific answers to such management questions as: What will and will not happen if a certain air quality standard is imposed, and if the acidity of precipitation is increased or decreased.

The model flowchart shows the computation sequence (fig. 2). The model calculates dry deposition as a function of ambient air quality and simulates the quantity and quality of water in throughfall, soil horizons, bogs, stream, lake, and lake outlet (Chen *et al.*, 1978). All important acidification processes are included in the model.

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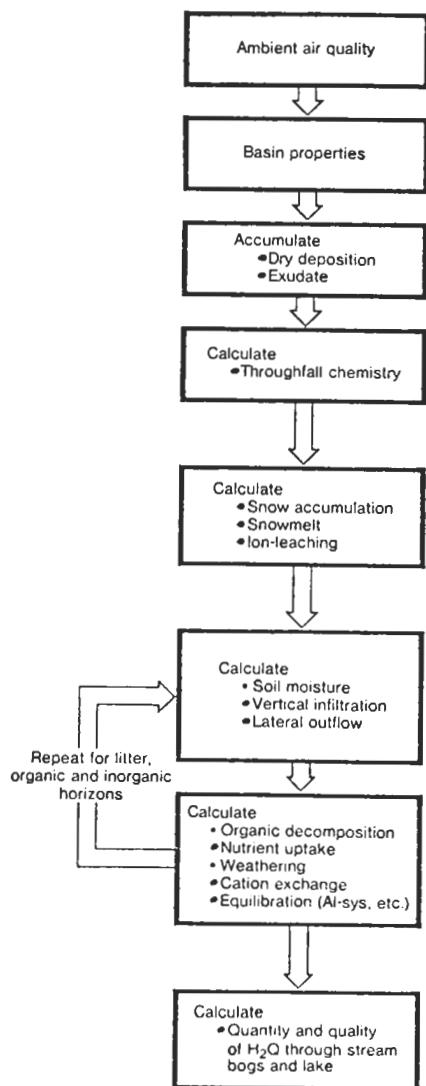


Figure 2--Model Flowchart

The model uses a network of compartments to account for spatial variability of ecosystem. It updates meteorological conditions daily and ambient air quality weekly. The calculations are performed on a daily time step to simulate the dynamic responses of ecosystem throughout years. Graphic outputs are provided to facilitate interpretation.

PRELIMINARY FINDINGS

Data indicate that the H^+ deposition rates are seasonal and are approximately the same for all three watersheds (Johannes and Altwicker, 1980). The seasonal pattern of H^+ deposition seems to follow that of SO_4 deposition (Johannes, 1980). The deposition rates of various ions in acid rain as measured at seven ILWAS stations correlate well with those measured at the nearby MAP3S stations. This is significant because it suggests that acid rain data from regional monitoring stations may be used to perform preliminary calculation of acid rain effect for a new site.

The pH at Panther Lake inlet is normally 7.3 to 7.5 throughout the year. The pH at the outlet is similar except during the snowmelt periods. During that period, pH drops to as low as 5.0. Something must have happened in the lake. Alternatively, it was argued that the inlet was only a small spring, not representative of all inflows.

The pH profiles measured in the lake show that only surface water is acidified during the period of snowmelt (Hendrey *et al.*, 1980). What is the source of H^+ ions that acidify the lake surface?

To resolve the puzzle, the model was used to help trace the source of water at the outlet. The model was first calibrated to the Panther Lake basin (Chen and Goldstein, 1980). After that, precipitation falling directly on lake surface is set to zero. This allows estimation of the contributions of this input to the total observed outflow. Approximately two-fifths of the peak flow can be accounted for by the direct precipitation to the lake surface. During that period, lake water is inversely stratified with respect to temperature. Direct precipitation which has a pH of 3.8 to 4.2 quite possibly is deposited right on the surface to acidify the lake surface water (pH 5.0). Another possible explanation is surface runoff resulting from snowmelt which has a high acidity (Galloway *et al.*, 1980).

The significance of the capability to manipulate the model to examine the effect of a single process on the integrated response of the ecosystem should not be overlooked.

LITERATURE CITED

- Chen, C.W., S.A. Gherini, and R.A. Goldstein.
1978. Modeling the Lake Acidification Process. In *Ecological Effects of Acid Precipitation*, Report of a Workshop, Galloway, United Kingdom, September 4-8, 1978, ed. M.G. Wood, Electric Power Research Institute, Palo Alto, CA, U.S.A., EA-79-6-LD.
- Chen, C.W., and R.A. Goldstein.
1980. "Techniques for Assessing Ecosystem Impacts of Air Pollutants." In *Proc. International Conference Air Pollutants and their Effect on the Terrestrial Ecosystem* [May 10-17, 1980, Banff, Alberta, Canada].
- EPRI Technical Work Statement for the Integrated Lake-Watershed Acidification Study RP 1109.
1979. Electric Power Research Institute, Palo Alto, California.
- Galloway, J.N., C.L. Schofield, G.R. Hendrey, and A.J. Johannes.
1980. "Sources of Acidity in Three Lakes Acidified During Snowmelt." In *Proc. of International Conference on the Ecological Impact of Acid Precipitation*, [March 11-14, 1980, Sandefjord, Norway].

Goldstein, R.A., C.W. Chen, S.A. Gherini, and J.D. Dean.

1980. A Framework for the Integrated Lake-Watershed Acidification Study. In Proc. of International Conference on the Ecological Impact of Acid Precipitation, [March 11-14, 1980, Sandefjord, Norway].

Hendrey, G.R., J.H. Galloway, and C.L. Schofield.

1980. "Temporal and Spatial Trends in the Chemistry on the Ecological Impact of Acid Pre-

cipitation." In Proc. of International Conference on the Ecological Impact of Acid Precipitation, [March 11-14, 1980, Sandefjord, Norway].

Johannes, A.H., and E.R. Altwicker.

1980. "Atmosphere Inputs to Three Adirondack Lake Watersheds." In Proc. of International Conference on the Ecological Impact of Acid Precipitation, [March 11-14, 1980, Sandefjord, Norway].

Photosynthesis and Stomatal Response to Light and Temperature in Ponderosa Pine Exposed to Long-Term Oxidant Stress¹

Patrick I. Coyne and Gail E. Bingham²

Seasonal courses (May to October 1977) of gross photosynthesis (calculated from $^{14}\text{CO}_2$ uptake) and stomatal conductance were characterized as a function of light and gross and net photosynthesis were characterized as a function of temperature (May and July 1978) in a stand of ponderosa pine (*Pinus ponderosa* Laws.) in the San Bernardino National Forest. The CO_2 diffusion pathway was partitioned into its stomatal and residual (mesophyll, carboxylation, excitation) resistance components for conditions of light saturation and 20°C. These trees have experienced long term oxidant fumigations from California's South Coast Air Basin since their establishment following fire in the mid-1950's. Nine trees were stratified for comparative studies into three chronic injury classes (I - slight injury, II - moderate, III - severe) having similar oxidant injury symptoms based on the scoring system of P. R. Miller.

Maximum or light saturated gross photosynthetic rates (P_{max}) and photochemical conversion efficiencies (dP/dI)_{I=0} were highest in the current needles and decreased with increasing needle age and with season. Differences among needle age classes within an injury class diverged as the season progressed indicating an acceleration of senescence by ozone. Maximum stomatal conductances (C_{max}) and stomatal sensitivity to increasing light (dC/dI)_{I=0} during opening followed a similar trend to P_{max} and dP/dI except in the current needles in which C_{max} and dC/dI were highest in the severely injured trees. This suggests a possible factor contributing to differential ozone sensitivity in this stand. The ratio of the stomatal resistance for CO_2 (r'_s) to the total resistance (r'_t ; stomatal+residual) decreased with oxidant injury, increasing needle age, and season suggesting that loss of photosynthetic capacity resulted more from limitations at the chloroplasts than from resistance to CO_2 diffusion through the stomata.

Temperature optima (T_{opt}) for photosynthesis were similar in all injury classes and averaged

about 20°C in May and 25°C in July. Light respiration (estimated as $P_{\text{gross}} - P_{\text{net}}$) was highest in healthy young needles and increased with temperature from 5°C to T_{opt} and then leveled off between T_{opt} and 35°C. Although light respiration was inversely related to oxidant injury, the ratio of $P_{\text{net}}/P_{\text{gross}}$ tended to decrease with oxidant injury. Apparently oxidant stressed trees not only had reduced rates of CO_2 fixation, but retained a smaller proportion of assimilated carbon after respiration losses. Summary data for select parameters are compared in Table 1.

Table 1--Comparison of select parameters normalized by dividing each mean by the maximum mean value in each column.

Injury Class	Needle Age	1977 Means			July 1978
		P_{max}	C_{max}	r'_s/r'_t	$P_{\text{gross}} - P_{\text{net}}$
I	0	1.00	0.89	1.00	1.00
	1	0.61	0.84	0.80	0.60
	2	0.32	0.62	0.59	0.35
II	0	0.92	0.92	0.82	0.95
	1	0.53	0.75	0.71	0.51
	2	0.17	0.38	0.40	0.18
III	0	0.79	1.00	0.59	0.94
	1	0.35	0.57	0.45	0.35
Max. Value ¹		8.30	0.24	0.29	2.46

¹Parameters defined in text. Units: P, mg CO_2 $\text{g}^{-1}\text{h}^{-1}$; C, cm s^{-1} ; r'_s/r'_t , s cm^{-1} ; age in years.

The differential response in photosynthesis and stomatal conductance among these field-grown ponderosa pine trees growing in a common environment indicated the presence of ecotypic variation in ozone sensitivity. The differences among injury classes were manifest as an acceleration of the normal decline in CO_2 fixation and stomatal conductance associated with needle aging. Particularly evident were the premature senescence and abscission of needles occurring at about the time gross CO_2 uptake dropped to 10 percent of the potential for class I current needles without foliar injury symptoms. This occurred at integrated incident ozone doses as low as 450 ppm-h in severely injured trees (class III) or as high as 800 ppm-h in slightly injured trees (class I).

A probable scenario for oxidant effects can be described. As foliar injury symptoms increase, photosynthetic capacity and net carbon accumulation per unit leaf mass or area, mass and area per needle, needle mass per unit area, and number of needle whorls retained per tree decline. These factors contribute to the steady loss of tree vigor, weakening them to the point of vulnerability to pathogenic organisms such as root rotting fungi and bark beetles.

¹Presented at the Symposium on Effects of Air Pollutants on Mediterranean and Temperate Forest Ecosystems, June 22-27, 1980, Riverside, California, U.S.A.

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The Effect of Air Pollution on Western Larch as Detected by Tree-Ring Analysis¹

Carl A. Fox and Thomas H. Nash III²

The number of publications in recent years dealing with the effects of air pollution on plants has been voluminous. A vast majority of these have dealt with the effects of short-term (hours or days) fumigations on plant response. In contrast to these studies, there is relatively little information on the long-term, cumulative effects of air pollution over decades. To examine these long-term effects, a dendroecological (tree-ring) analysis was employed to assess the growth response of western larch (Larix occidentalis Nutt.) to sulfur dioxide. Utilization of dendroecological techniques provided a means for quantitatively removing the effect of climate on the growth response of larch and, thus, permitted an examination of the residual growth response in terms of local site factors, specifically sulfur dioxide.

Five western larch study sites were located in the Columbia River Valley near the lead-zinc smelter at Trail, British Columbia. This particular location represents a unique study area in that the sulfur dioxide gradient has been well documented, both over space and time, since smelting activity began in 1896.

Sulfur emissions from the smelter increased in the early 1900's, reached a maximum in 1930, and resulted in a concomitant decrease in annual tree growth. After 1930, sulfur emissions decreased dramatically with the implementation of pollution abatement measures. However, the growth response of western larch to the decreased sulfur emissions was not immediate, and reflected the low frequency nature of the variance of the sulfur dioxide effect on tree growth.

Correlation and regression analyses were utilized to develop multivariate models for the larch study sites sampled at varying distances from the Trail smelter. Climatic models developed for the control site (tree-ring) chronology were applied to site chronologies located within the sulfur dioxide affected area to remove the effects of climate on tree growth and examine the residual response of the system. The pattern of the residuals closely resembled the sulfur emissions from the smelter with the most negative residuals occurring when sulfur emissions were greatest.

Further regression modeling identified the relative importance of sulfur emissions, prior year's growth, temperature, and precipitation to annual tree growth. In those sites closest to the smelter, sulfur emissions accounted for the greatest proportion of the variance calibrated by the regression models. As distance from the smelter increased, the variance attributable to sulfur emissions decreased in each site model. In all of the site models, temperature, particularly summer temperature, appeared to be a primary limiting climatic factor. Prior growth also accounted for considerable variance in the models with precipitation variables appearing to be of lesser importance in explaining the variance of the site chronologies.

The results of this study demonstrate the applicability of tree-ring analysis in identifying and quantifying the long-term effects of air pollution on forest communities. It also provides a basis for examining the interrelationships between air pollution, climate, and tree growth.

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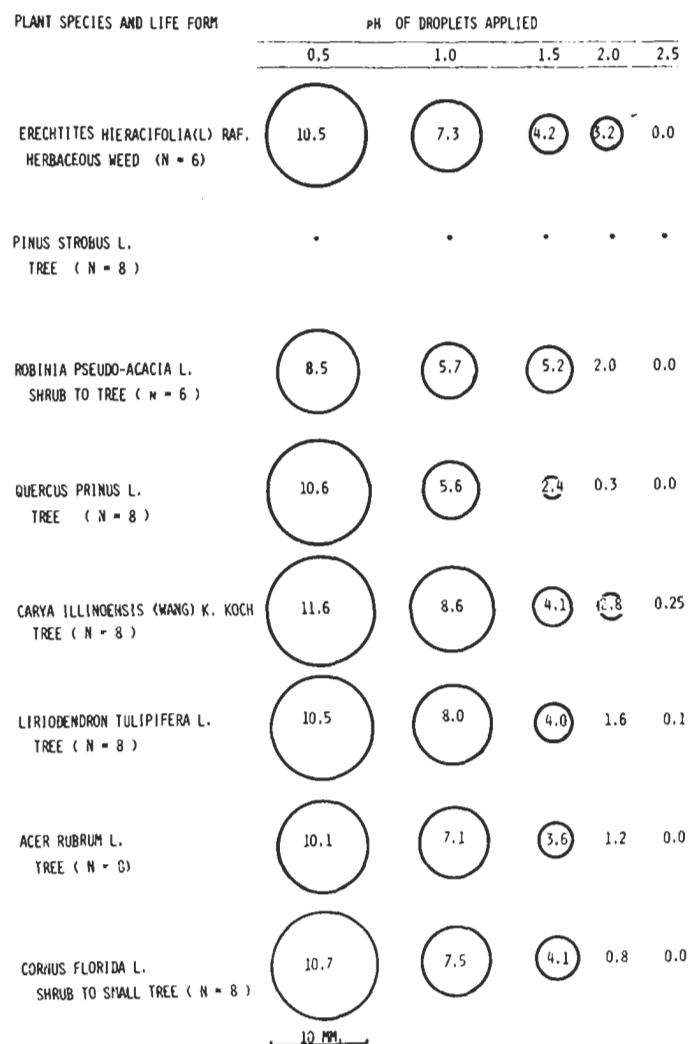
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Acid Rain: Threshold of Leaf Damage in Eight Species from a Forest Succession¹

B.L. Haines, M. Stefani, and F. Hendrix²

Eight plant species were subjected to artificial acid rains of pH 2.5, 2.0, 1.5, 1.0, and 0.5 in order to determine the threshold for and symptoms of damage. In a previous study artificial acid rains of pH 5.5, 4.5, 3.5, and 2.5 failed to produce symptoms of damage. The present study was designed to extend the pH range. The plants were *Erechtites*, *Robinia*, *Pinus*, *Quercus*, *Carya*, *Liriodendron*, *Acer*, and *Cornus* from the Forest Service's Coweeta Hydrologic Laboratory near Franklin, North Carolina. Duplicate 0.01 ml drops of each of the 5 simulated acid rain solutions were applied to a single mature leaf on each of two plants of seven species. The entire experiment was performed twice. In *Pinus* needle tips were immersed in the solutions. The sizes of necrotic spots are shown in Fig. 1. Droplets of pH 2.0 produced brown necrotic spots on all species except *Pinus* while droplets of pH 1.0 produced necroses on leaves of all species examined. *Pinus* needles were damaged at pH values between 1.0 and 0.5. Damage was confined to younger needles and was evidenced by browning and collapse. For angiosperms the sizes of necrotic spots increased with decreasing pH. This suggests that the leaf has some buffering capacity which was progressively overcome by increasing droplet acidity. Comparison of results with literature suggests that developing leaves are more easily damaged than are mature leaves used in this study. No successional trends in susceptibility were observed.

The volume weighted average rainfall pH for Coweeta is 4.6 with observations ranging from 3.2 to 5.9. Because the pH scale is logarithmic with a decrease in 1 pH unit corresponding to a 10-fold increase in the H^+ concentration, it is apparent that a 100-fold increase in the volume weighted average concentration of H^+ at Coweeta would change the pH from 4.6 to 2.6 which is near the threshold of damage. However, the extremes may be far more critical than the volume weighted average. With the lowest pH value recorded for Coweeta being pH 3.2, merely a 10-fold increase in acidity to pH 2.2 in a single spring or summer storm seems likely to bring damage or death to mature leaves of the flowering plants at Coweeta.



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Figure 1. Average diameter in mm of necrotic spots on leaves subjected to droplets of 5 pH levels. (*) Spot diameters not measured on needles, see text for description of damage.

Prioritization of Research on Air Quality Related Resources of the National Parks¹

E. A. Howard, L. M. Boyd, W. M. Brock, D. W. Morse, S. S. Shin and K. L. Steel-Sabo²

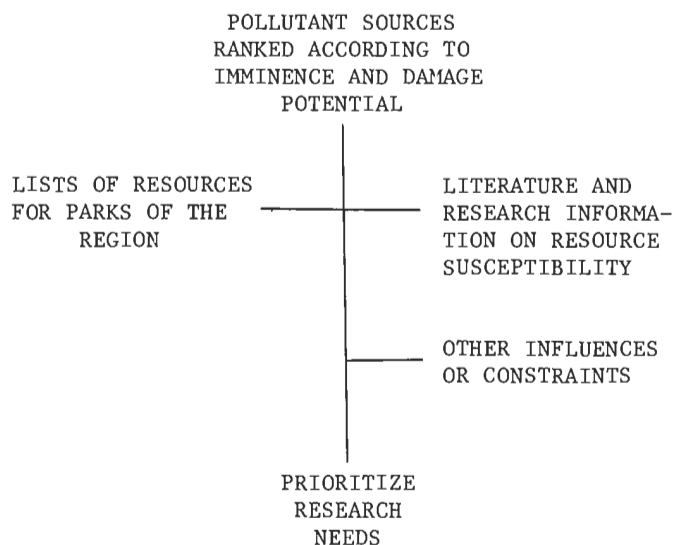
The National Park Service (NPS) has the legal responsibility and authority to preserve and protect the resources of lands under its jurisdiction. Air pollution has been recognized as a significant threat to the parks and the 1977 Amendments to the Clean Air Act provided the Service the authority and responsibility to protect resources that are air quality related. The NPS Air Quality Office intends to conduct research programs to determine air pollution effects, to emphasize the use of sensitive receptors to detect effects, as well as to ensure that scientific evidence is readily available when NPS must make determinations of adverse effects. These efforts are subject to time and funding limitations and therefore guidelines on setting research priorities are being developed.

The first step in establishing cause and effect relationships is to identify the pollutants that may affect park resources (fig. 1). Pollutants from existing or planned sources may be identified by monitoring or estimated from permit requirements. If sources are not known, pollutants must be identified through some early warning process or on the basis of the potential development of regional natural resources. Existing sources of air pollutants that may affect the natural or cultural resources of park units must be ranked according to pollutant type, concentration, frequency, and duration of exposure of the resource. The ranking must also incorporate the estimated lead time before new pollutant sources begin operation. The identification and ranking of pollutant sources is necessary to economize the search for potential effects.

The identified pollutants are then used as a basis of comparison between the list of resources present on a park unit and the literature or current research information on resource susceptibility (fig. 1). The presence or absence of information on park resources must be incorporated into the setting of priorities. If these data are unavailable, obtaining information on air quality related resources may be the most pressing need. If the resource inventories are available, then the information from these inventories is used in the comparison between resources present and those

that are susceptible to the effects of air pollution. The availability of current research information or literature on susceptibility must be similarly considered in establishing research priorities. Time constraints on recovering information or research results on air quality effects on natural ecosystems has prompted the development of a quick access annotated bibliography that uses a coding system based specifically on natural and cultural resources of NPS units. The bibliography has been used on several occasions to provide lists of references in support of litigation and testimony for hearings.

Setting priorities for research on air quality effects is altered by other influences and constraints (fig. 1). Funding and time limitations must be incorporated into the prioritization. Insufficient funding may cause postponement of the highest priority research until these requirements can be integrated into the budget cycle. Time constraints also have considerable influence on priority setting. Substantive scientific data on air quality effects cannot be produced within the short time schedules of hearings or litigation. The timing of the budget cycle creates difficulty in obtaining sustained funding for long-term effects research. Political requirements at the Washington level may override the regional prioritization. The setting of priorities must be continually updated in conjunction with any change imposed by these influences or constraints.



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Figure 1--Considerations in setting research priorities for air quality related resources.

Estimation of Adverse Effects of Air Pollution on Danish Forests¹

Ib Johnsen²

Effects of air pollution on Danish forests do probably only occur under conditions, where SO₂, NO_x and/or O₃ are present together, thus resulting in synergistic action on the leaves. The maximum observed monthly average urban immission of NO_x and SO₂ are within the range for synergistic action hereof. In rural regions only the most sensitive species may be affected by combinations of SO₂, NO_x and O₃. Oxidants, and O₃ in particular, probably play a stronger role in rural areas situated in the outer periphery of cities. Here levels exceeding the values at which effects on rather sensitive species occasionally occur during summer time. Unstable superadiabatic conditions combined with high insolation lead to high O₃ formation rates, and high ground level SO₂/NO_x levels are observed when high stack emissions are transported to the ground relatively close to the source.

Fluorides are only of importance in very restricted areas around brick factories, fertilizer industries etc., and of minor significance in relation to Danish forestry.

Heavy metals and hydrogen ions result in indirect effects as adverse effects on the nutrient status of the soil and the soil microbial processes. The map shows areas of Denmark (shaded areas) where the soil is believed to be most vulnerable to acid precipitation; it is reasonable to believe that these areas are coincident with those expected to be most affected by increasing heavy metal levels in top soil due to atmospheric fallout. The most vulnerable soils are intermediate between the very podzolised soils of the alluvial plains (Western Jutland) and brown earths/clayish soils with high buffer capacity, and are characterized by their high content of moraine sand.

Areas of woodland in Denmark related to species, 1976. (Numbers in 1000 ha.)

<i>Fagus silvatica</i>	75	Res.
<i>Quercus robur</i>	25	Res.
<i>Fraxinus excelsior</i>	9	Sens.
<i>Acer psuedoplatanus</i>	5	Res.
Other deciduous spp.	24	
Deciduous total	138	
<i>Picea abies</i>		Sens.
<i>Picea sitchensis</i>	173	Sens.
<i>Abies</i> spp.	27	Sens.
<i>Pinus mugo</i>	30	Res.
Other coniferous spp.	46	
Coniferous total	276	
Woodland total	414	

Immission levels and threshold values

	SO ₂	NO _x	O ₃	F ⁻
	(µgm ⁻³ , diurnal means)			
Urban	40-100	50-100	25-50	0.5-5
Rural	10-40	5-20	25-200	0.2-0.5
Single	250	500	100	1
	100	50		
Comb.		50	50	
	100		50	



Figure 1--Areas of Denmark (shaded areas) dominated by moraine sand deposits.

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Changes in Southern Wisconsin White Pine Stands Related to Air Pollution Sensitivity¹

David F. Karnosky²

Fifteen eastern white pine (*Pinus strobus* L.) sample plots consisting of a total of 1523 trees were established within a 13 km radius of the coal-burning 1054 MW Columbia Generating Station, located 40 km north of Madison, Wisconsin. The sample plots were established in 1971 and observed frequently during the growing season for 4 years and then annually at the end of each growing season for the next 5 years. These plots consisted of plantations with trees having uniform ages within each plantation but with trees ranging in age from 10 to 40 years old across the 15 plots. Baseline study during the 4 years before the plant began operation in 1975 showed that some 10 percent of the white pine trees were sensitive in some degree to ambient air pollution as determined by the presence of needle tipburn and/or chlorotic dwarf symptoms in one or more years. The sensitive trees occurred randomly in the plots.

The most common type of air pollution symptom found in the baseline study was tipburn consisting of reddish brown dieback (0.1 to 3 cm in length) on first-year needle tips. The severity of symptoms varied from tree to tree and from year to year. The most severely affected trees had stunted tops, short needles, poor needle retention, and were characteristic of an air pollution-induced syndrome called the chlorotic dwarf disease. However, chlorotic mottling of new needles, common to the chlorotic dwarf disease, did not occur. Tipburn symptom development usually began during the early summer when the new needles were elongating. Thus, trees began showing symptoms in early to mid June, and symptoms developed throughout the growing season.

Continuous air monitoring for sulfur dioxide (SO₂) and ozone (O₃), begun in 1973 and continued to the present time, revealed the common occurrence of SO₂ and O₃ concentrations in the range of 0.4 to 5.0 pphm for 1 to 3 hours during the summer months. Maximum one-hour averages recorded during the study were 11 pphm SO₂ and 13 pphm O₃. These levels, while low in terms of air quality standards, have been shown to be within the range of concentrations of these pollutants reported to injure genetically sensitive eastern white pine trees in controlled fumigations.

During the 5 years since the power plant began operation, there have been no detectable differences in the occurrence or severity of symptoms. With only a few exceptions, the same trees have shown symptoms after the plant began operation as before it started. However, mortality has reduced intermediate genotypes (those injured in 1 or 2 years of the 9 year study) by 4.7 percent and the sensitive genotypes (those injured in 3 or more years) by 10.4 percent (Table 1). In comparison, only 2.0 percent of the tolerant trees have died during the course of the study. Most of the mortality of the sensitive genotypes was apparently due to their being slower growing and thus unable to compete for light, water, and nutrients with their neighboring trees. As a result, the white pine stands studied are undergoing a gradual transition towards having air-pollution tolerant genotypes dominate.

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Table 1--Mortality over a 9-year period of eastern white pine trees differing in air pollution sensitivity.

Air Pollution Sensitivity ¹	Original Number Of Trees	Number That Have Died	Percent Mortality
Tolerant	1369	27	2.0 pct.
Intermediate	106	5	4.7 pct.
Sensitive	48	5	10.4 pct.
Totals:	1523	37	2.4 pct.

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¹The sensitivity rankings were based on the following: tolerant trees did not have air-pollution-induced tipburn symptoms during the course of the study; intermediate trees had symptoms for 1 or 2 years; and sensitive trees had symptoms for 3 or more years.

Effect of O₃ and O₃ + NO₂ on Growth of Tree Seedlings¹

Lance W. Kress²

Seedlings of 10 eastern forest tree species were exposed to 0.05, 0.10, or 0.15 ppm O₃ and seedlings of 7 tree species were exposed to 0.10 ppm O₃ and/or 0.10 ppm NO₂ in 6 hr/day exposures for 28 consecutive days. The exposures were performed when the seedlings were 2–4 wk-old in indoor exposure chambers of the CSTR design.

The Environmental Protection Agency has determined that the threshold for significant growth effects due to extended O₃ exposures (7 hr daily avg. for 2 mo) for sensitive vegetation is between 0.06 and 0.10 ppm. Three species in this study exhibited a threshold for significant suppressions in that range, while three other species exhibited a threshold for significant effects between 0.10 and 0.15 ppm (Table 1). However, two species exhibited a threshold for significant growth suppressions at or below 0.05 ppm. Those two species, loblolly pine and American sycamore, are probably the most important of the species tested to the forest industry. The significant effects at 0.05 ppm O₃ were not accompanied by foliar injury.

In previous studies there have been indications that low concentrations of NO₂ might be stimulatory to plant growth or alleviate O₃ phytotoxicity. Similar indications were noted for some of the tree species in this study. Two species (white ash and green ash) exhibited greater growth in the NO₂ treatment than the control, and the only significant interactive effects were significantly less than additive (sweetgum and white ash).

Stimulations of growth at low O₃ concentrations have been noted in the past, and some species exhibited growth stimulations in this study (Table 1). However, apparent growth stimulations appear to be dependent in part on the plant species and the parameter being evaluated.

The relationship that these data have to field conditions is not clear. This study demonstrates the potential for adverse effects at pollutant concentrations below the current NAAQS, but future research will have to determine whether such effects can be detected in the field.

Table 1. Height growth and dry weight expressed as percent of control for seedlings of 10 tree species exposed to 0.05, 0.10, or 0.15 ppm O₃ for 6 hr/day for 28 consecutive days.

Species	Treatment	Percent of control		
		Height growth	Dry weight	
			Top	Root
Loblolly Pine (<i>Pinus taeda</i> L.)	C	100a	100a	100a
	5	82 b	85ab	90ab
	10	73 b	79 b	72 bc
	15	59 c	74 b	64 c
Pitch Pine (<i>P. rigida</i> Mill.)	C	100a	100a	100a
	5	96ab	94a	84ab
	10	87 b	83a	77ab
	15	74 c	78a	68 b
Virginia Pine (<i>P. virginiana</i> Mill.)	C	100a	100a	100a
	5	95a	98a	120a
	10	89a	97a	93a
	15	86a	87a	86a
Sweetgum (<i>Liquidambar styraciflua</i> L.)	C	100a	100a	100a
	5	109a	91ab	88ab
	10	71 b	76ab	65 bc
	15	55 b	60 b	52 c
Sycamore (<i>Platanus occidentalis</i> L.)	C	100a	100a	100a
	5	104a	84a	57 b
	10	73 b	43 b	27 b
	15	79 b	36 b	19 b
Green Ash (<i>F. pennsylvanica</i> Marsh.)	C	100a	100a	100a
	5	98ab	86ab	86a
	10	76 bc	71ab	75a
	15	70 c	64 b	75a
White Ash (<i>Fraxinus americana</i> L.)	C	100a	100 b	100ab
	5	112a	125a	111a
	10	91a	92 b	87 b
	15	85a	83 b	81 b
Willow Oak (<i>Quercus phellos</i> L.)	C	100a	100a	100a
	5	99a	99a	94a
	10	96a	92a	83a
	15	81 b	89a	83a
Sugar Maple (<i>Acer saccharum</i> L.)	C	100 b	100a	100ab
	5	95 b	88a	161a
	10	108a	90a	116ab
	15	88 c	58 b	69 b
Yellow Poplar (<i>Liriodendron tulipifera</i> L.)	C	100 b	100 b	100a
	5	160a	147a	133a
	10	108 b	111ab	96a
	15	88 b	108 b	121a

¹Values for each column/species followed by the same letter are not different at p = 0.05 according to the Duncan's New Multiple Range Test.

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Impact of Oxidant Air Pollution on Ponderosa and Jeffrey Pine Cone Production¹

Robert F. Luck²

Cone production by ponderosa and Jeffrey pine trees 10 cm or more in dbh was assessed on 19 plots located in the San Bernardino Mountains of southern California. These plots were established along a gradient of oxidant air pollution. Each tree was rated annually using an oxidant air pollution index based upon the number of year classes of needles retained, their chlorotic condition, their length and the amount of branch mortality present. This rating was made independently in the upper and lower crown for each tree. Other tree characteristics, e.g., age, dbh, height, were also measured once during the course of the 6 year study. Cones were visually counted within the crown of each tree each September and October.

Although severe oxidant air pollution injury was associated with reduced cone production, the most important correlate with cone production was crown class, i.e., the position of a tree's crown relative to those of its neighbors. Dominant trees bore the greatest proportion of cones. In ponderosa pine dominant trees comprised 32 percent of those present on the 19 plots but bore 80 percent of the cones. When the dominant and codominant crown classes were combined they comprised 58 percent of the ponderosa pines but bore 96 percent of the cones. Jeffrey pine showed a similar pattern. In both Jeffrey and ponderosa pine, cone production increased significantly with age ($H_0: b=0$: ponderosa pine; dominant: $\hat{y} = -92.33 + 1.86 (\text{Age})$, $r^2 = 0.811^{**}$; codominant: $\hat{y} = -74.48 + 1.077 (\text{Age})$, $r^2 = 0.4216^{*}$; Jeffrey pine; dominant: $\hat{y} = 4.74 + 0.434 (\text{Age})$, $r^2 = 0.765^{**}$; codominant: $\hat{y} = -8.28 + 0.243 (\text{Age})$, $r^2 = 0.7456^{**}$).

Three patterns were observed: (1) In dominant Jeffrey pines 130 years or older fewer severely injured trees bore cones than uninjured ones (1b $F_{(4,25)} = 11.98$, $p < 0.05$); (2) in dominant ponderosa pines 130 years or older severely injured trees bore fewer cones/tree than uninjured ones (2a $\chi^2_{(4)} = 21.07$, $p < 0.005$); and (3) in both dominant and codominant Jeffrey and ponderosa pines severely injured trees bore significantly fewer cone crops than uninjured ones during the 6 years of the study (χ^2 's values < 0.01 in 5 cases, < 0.05 in 1 case).

¹Presented at the Symposium on Effects of Air Pollutants on Mediterranean and Temperate Forest Ecosystems, June 22-27, 1980, Riverside, California, U.S.A.

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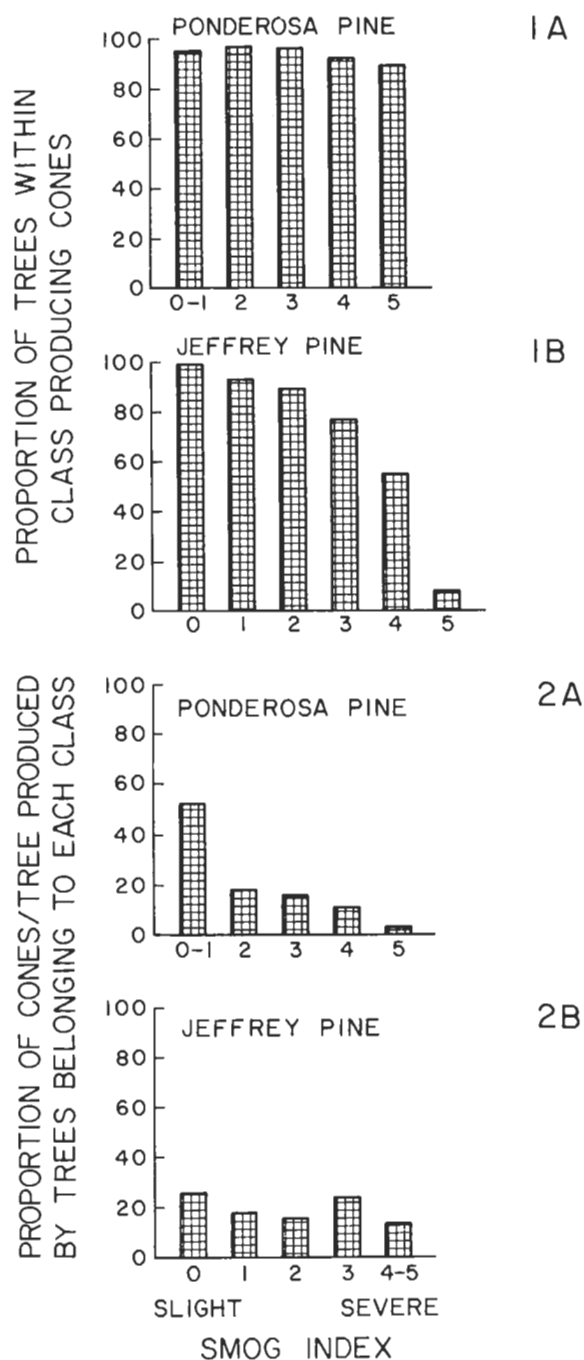


Fig. 1. Proportion of trees producing cones in a given smog class. (a) Ponderosa pine; (b) Jeffrey pine.

Fig. 2. Proportion of cones/tree borne by tree belonging to a given smog class. (a) Ponderosa pine; (b) Jeffrey pine.

Lichens as Air Quality Monitors¹

P.D. Lulman
R.J. Fessenden²
S.A. McKinnon²

The Athabasca tar sands, located in a remote region in northeastern Alberta, in the boreal forest region of Canada, contain vast reserves (approx. 600 billion barrels) of bitumen. At the present time there are only 2 companies operating in the region, producing synthetic crude oil from this bitumen, Suncor Inc. and Syncrude Canada Ltd. Suncor, which began production in 1967, emits approximately 200-300 tonnes of SO₂ per day. Syncrude, which began production in 1978 and which is located only 10 km from Suncor, has been emitting about 50-100 tonnes per day. In the next 10-20 years there is a good possibility that many more companies will begin operating in the region as well. This has led to a concern about the effects of changes in air quality on the vegetation of the region. Since lichens are known to be more sensitive to air pollutants, particularly SO₂, than most higher vegetation, it was decided to monitor changes in lichen growth as an early warning of the impact of pollutants on all components of the vegetation.

In 1976, Syncrude Canada Ltd. installed a network of 56 permanent plots in a radiating pattern centered on the 2 operations and at distances of up to 47 km. Within each plot 20, 200 cm² permanent quadrats containing thalli of Parmelia sulcata or Hypogymnia physodes, 2 abundant bark lichens, were established and photographed. Most of the quadrats were established on white spruce (Picea glauca), although a smaller number were established on balsam fir (Abies balsamea) and white birch (Betula papyrifera). The photographs were projected and the surface areas of selected lichen thalli were measured. In 1979, all of the quadrats in 12 selected plots were rephotographed and the photographs analyzed as before. The objectives of this partial resurvey were (a) to measure changes in the surface area of thalli of P. sulcata, the dominant lichen species, in the 3 year period 1976-1979, (b) to test the significance of these changes as a function of distance

and/or direction from the emission sources, and (c) to assess the practicability of the lichen network as a long term monitoring system for detecting air quality impacts on vegetation.

There was a significant increase in projected thallus surface area in 6 plots, a significant decrease in 2 plots, and no significant change in 4 plots. In most cases the net change in lichen surface area during the 3 year period was less than 10%. However, within-plot variability was very large with coefficients of variation commonly between 300-400%. The changes in projected thallus surface area were significantly but weakly, related to distance; the thalli closest to the emission sources increasing more in area than those at a distance, where in fact there was a net decrease in surface area on the average. Field observations indicated that there was no relationship between lichen color and/or vigor and distance and/or direction from the emission source. It was evident in a comparison of the 1976 and 1979 paired photographs that there was a significant loss of whole thalli and portions of thalli from most of the quadrats. It was not unusual for 10-20% of the thalli present in 1976 to be missing in 1979. This appeared to be primarily related to the natural process of bark exfoliation, although animal activity and insect grazing are also suspected causes.

This technique is capable of detecting significant changes in the surface area of P. sulcata thalli over a 3 year period, however, it requires careful standardization and a large number of samples due to the large within-plot variability in this characteristic. The high variability is related to the rather non-uniform growth of the P. sulcata thallus as well as the variety of natural processes causing losses of portions of the thallus. Perhaps other attributes of the thallus would make better indices of growth (or lack of growth), however, none were investigated. The extensive losses of whole thalli raise questions about the permanence of this system and hence its value as a long term monitoring system. Because the predominant reason for thallus loss is bark exfoliation, this problem is very much related to tree species. Spruce are particularly bad in this regard. There was very little thallus loss from birch or fir. Notwithstanding the above difficulties, it was concluded that there was no air quality caused damage to P. sulcata between 1976 and 1979 as inferred from the relationships between the net change in thalli surface area and distance and/or direction from the emission sources. This conclusion is supported by field observations of lichen color and vigor.

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Acid Precipitation in California and Some Ecological Effects¹

John G. McColl and Mary K. Firestone²

Wet and dry precipitations were monitored on an event basis in 1978-9 at Berkeley and San Jose (pollution source areas in the San Francisco Bay area), Davis and Parlier (in the central agricultural and rangelands), Challenge (lower Sierran forest), and Tahoe City (on the shore of Lake Tahoe). Concentrations of thirteen ionic species and specific conductance and volume were measured (Table 1).

Acid rain (pH < 5.6) was common at all eight sites. Mean pH of storms varied from 4.24 at San Jose to 5.20 at Davis, and the lowest pH of any storm was 3.71 at San Jose. The primary cause of the acidity was probably the air pollutant NO_x, following its dissolution in wet precipitation. NO₃⁻ was the anion most closely correlated with H⁺, and NO₃⁻ generally occurred in greater concentration than SO₄²⁻.

Total dry depositions of chemical constituents between rainstorms were of the same order as total wet depositions during storms. Dry deposition during summer would greatly increase the amounts recorded in this study which was conducted in the wet season only. However, more research is needed in procedures for quantifying dry atmospheric deposition.

Although NO₃⁻ concentration (μg/l) and acidity (H⁺ concentration, μg/l) of wet precipitation were greatest in pollution source areas, total deposition (kg/ha) of NO₃⁻ and H⁺ were greatest in the non-urban receptor areas of Napa and Challenge; this was largely a function of the greater precipitation volumes at these two sites (Table 1). Thus ecological effects may be expected in the coast ranges and Sierras within the general east-to-west "wash-out fan" of wet precipitation, as well as within pollution-source areas.

Effects of these acid inputs to California soils are currently being assessed. These investigations on soil effects include research on both the inorganic phase of soil and on the organic/biological components.

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Table 1--Mean ionic concentrations of wet precipitation during the study period in 1978-9 (μeq/l).

Ion	SITE ¹							
	BE	TC	KE	CH	SJ	HO	DA	NA
H ⁺ (Lab.)	22.1	6.8	10.9	13.0	38.0	7.9	6.3	14.6
Na ⁺	36.9	11.8	15.3	12.8	33.1	14.5	15.1	22.0
K ⁺	1.5	0.6	1.4	2.1	1.5	1.1	1.1	1.4
Ca ²⁺	6.0	4.3	8.0	10.9	12.6	3.3	5.6	4.6
Mg ²⁺	9.6	1.7	3.6	7.0	9.7	3.8	5.7	5.3
Fe ³⁺	0.3	0.0	0.6	0.4	0.8	0.2	0.4	0.4
Mn ²⁺	0.1	0.0	0.1	0.6	0.1	0.1	0.1	0.1
Cu ²⁺	0.1	0.1	0.1	0.2	0.3	0.1	0.2	0.7
Zn ²⁺	0.3	0.1	0.2	0.2	0.6	0.1	0.1	0.2
NH ₄ ⁺	8.0	4.1	40.0	11.9	19.1	9.7	35.5	12.1
NO ₃ ⁻	13.7	6.7	43.4	19.9	16.4	11.1	22.6	16.4
Cl ⁻	40.1	3.7	11.2	7.7	38.9	14.2	14.0	23.1
SO ₄ ²⁻	10.2	13.3	13.8	8.6	10.0	6.2	19.0	11.7
Cond. (μmho/cm)	13.9	3.9	17.4	10.6	16.2	6.4	9.6	10.7
pH (Lab.)	4.7	5.2	4.7	4.9	4.4	5.1	5.2	4.8
Vol. (cm)	52.7	78.0	19.7	110.7	21.0	64.9	39.7	62.0

¹BE Berkeley, TC Tahoe City, KE Kearny (field station at Parlier), CH Challenge, SJ San Jose, HO Hopland, DA Davis, and NA Napa.

In the inorganic soil-chemical studies, soils covering a wide range of parent materials and age are being treated with acid inputs, and the subsequent leaching patterns of ions (including Al³⁺, H⁺, Na⁺, K⁺, Mg²⁺, Ca²⁺) are being determined.

The biological investigations include assessment of possible effects on the soil-plant nutrient system. The first 10-week pot-trial, using a Yolo series soil and growing barley and clover, has just been completed. Preliminary results indicate marked growth increases in treatments where the plants were sprayed with solutions of pH 2.0 and 3.0. This result is attributed to greater additions of SO₄²⁻ and NO₃⁻, and/or to increased availability of soil nutrients in these lower-pH treatments. However, spotting or leaves by acid droplets also occurred in the pH 2.0 treatments. Nitrification, denitrification and nitrogen fixation activities are being measured in both the rhizosphere and non-rhizosphere soil.

Leaf Litter Decomposition in the Vicinity of a Zinc Refinery¹

W. D. McIlveen²

Emissions from a large zinc refinery in northern Ontario have been monitored for 10 years and an accumulation of zinc, copper, cadmium, arsenic and lead in the soil and vegetation surrounding the refinery complex has been documented. It was considered that decomposition of leaf litter as a part of the nutrient cycling process might be a convenient method to monitor the initial impact of these metals on the environment. This investigation was undertaken in the fall of 1977. Nylon mesh bags containing 10 g (oven-dry weight) of trembling aspen (*Populus tremuloides*) were set out at several locations around the refinery. Two sources of aspen foliage were utilized including foliage from a control location and contaminated foliage collected near the refinery. The exposure locations for the litter bags included three sites in close proximity to the refinery, (Plot A, B, C) one site 2.2 km from the refinery (Plot D) and one at a control location (80 km southwest of the refinery) (Plot E).

The litter bags were allowed to over-winter and triplicate bags of each litter type were collected in May, June and August of 1978 and in May, July and August in 1979. Microarthropods were extracted from the bags in modified Tullgren funnels in the laboratory. The litter was then dried to determine the leaf weight loss and processed for chemical analysis.

It was found that the rate of decomposition of the leaf litter was lower at sites nearest to the refinery and that contaminated foliage decomposed more slowly than control foliage at all sites. The microarthropod population was dominated by mites and springtails. The number of microarthropods was generally lower at the sites nearest to the refinery and was also lower in the litter bags containing the contaminated foliage. The numbers of microarthropods appeared to fluctuate with moisture content of the litter. It was found that the concentrations of zinc, copper, cadmium, lead, arsenic, iron, sulphur and selenium in the leaf litter increased with time at the sites nearest to the zinc refinery but only a slight increase was noted at the more distant sites.

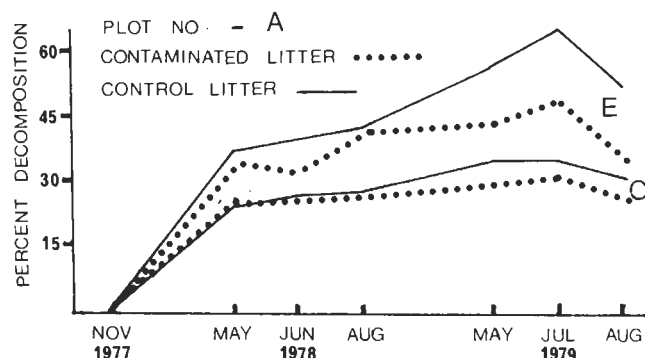


Figure 1. Pattern of decomposition of two types of leaf litter at two sites.

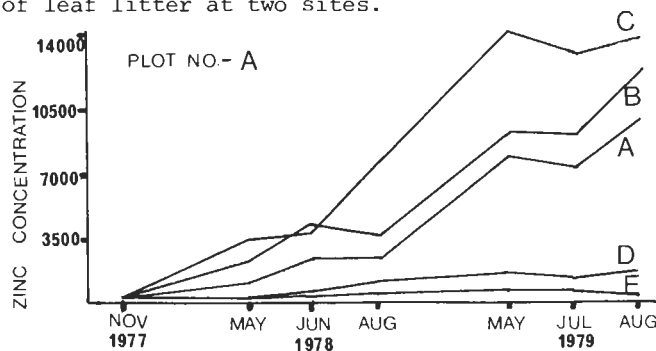


Figure 2. Zinc accumulation is typical of metal accumulation in leaf litter.

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Effects of Chronic Air Pollution Stress on Allocation of Photosynthate by White Pine¹

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Allocation of ¹⁴C-photosynthate by *in situ* branches of nine field-grown white pine trees was studied to determine whether distribution patterns differed between trees with apparent differences in sensitivity to air pollution stress. Three trees were selected in each of three sensitivity classes which were differentiated on the basis of needle length, mottling, and duration of retention. Previous studies (Mann *et al.* 1980) indicated that photosynthetic potential of foliage from trees in these three classes was not significantly different. Growth ring analysis of increment cores indicated that average annual increment of intermediate and sensitive trees was 98% and 47%, respectively, of that attained by the tolerant trees (7.8 mm yr⁻¹) over the past 18 years (Fig. 1). Sensitive trees showed a marked decline in annual growth during the past 10 years.

Foliage was labelled with ¹⁴CO₂ four times during the growing season (June, July, August and November). Of the paired branches from each tree labelled on each date, one was harvested after seven days and the remainder in November at the end of the growing season. Photosynthate allocation patterns were compared by determining levels of foliar retention and allocation to nearby branches. In June when elongating needles were approximately 50% of their final length, ¹⁴C movement patterns in needles of three age classes (Fig. 2) indicated that contribution of ¹⁴C-photosynthate by old needles to new needle growth was occurring. This process was most rapid in tolerant trees which retained needles from two prior years and least significant in sensitive trees. Lower levels of incorporation of photosynthate into foliar tissues occurred subsequently ($\bar{X}$ = 35% in June, 27% in July, and 5% in August after seven days). There were no distinct differences in foliar retention of ¹⁴C between the three sensitivity classes which could be associated with the distribution of high levels of ozone (1h avg > 0.08 ppm) near the study area. Significant 2-year-old needle retention on tolerant trees did not extend beyond July.

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Generally higher levels of transfer of ¹⁴C from foliage into branches were noted in the tolerant trees throughout the growing season. Higher levels of translocation of ¹⁴C-photosynthate out of 1-year-old needles than current-year needles were also found for all sensitivity classes. Enhanced current needle elongation (+25% sensitive, +10% intermediate, and +3% tolerant) compared to the previous year was associated with lower than average ozone levels and higher than average rainfall during the growing season. Data support the hypothesis that growth limitations in sensitive trees are a function of stress-induced reductions in photosynthate availability which result from reduced needle length (a function of decreased availability of carbohydrates for needle growth) and premature needle senescence. The resulting chronic decline of sensitive trees is likely a result of interactions between air pollution stress and site factors which may include secondary belowground pathogens.

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Literature Cited

Mann, L. K., S. B. McLaughlin, and D. S. Shriner. 1980. Seasonal physiological responses of white pine under chronic air pollution stress. *Env. Exp. Bot.* (in press).

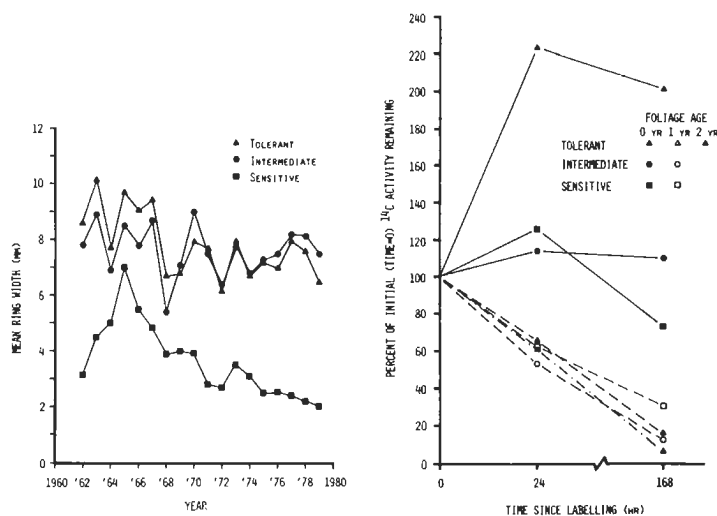


Figure 1 (left)--Average annual diameter growth of 3 trees in each of 3 sensitivity classes.

Figure 2 (right)--Distribution of ¹⁴C photosynthate in foliage 0, 1, and 7 days after ¹⁴CO₂ uptake in June.

Effects of SO₂ and Ozone on Photosynthesis and Leaf Growth in Hybrid Poplar¹

Reginald D. Noble
and
Keith F. Jensen²

Plants in the natural environment are often simultaneously exposed to a combination of many atmospheric pollutants. Thus it is of interest to ascertain the nature of interaction of these pollutants on plant development and metabolism. Two pollutants commonly encountered in the atmosphere which are particularly toxic to plants and would appear to offer potential for interaction are SO₂ and O₃.

METHODS

Plants of hybrid poplar #207 (*Populus deltoides* Bartr. X *P. trichocarpa* Torr. & Gray) were grown from cuttings under greenhouse conditions. Six week old plants were fumigated in controlled environment chambers for 12 hours per day for 24 consecutive days. Four sets of plants were used as follows: Set 1-Controls (no fumigation); Set 2-Fumigated with SO₂ at 0.5 ppm; Set 3-Fumigated with O₃ at 0.25 ppm; Set 4-Fumigated with SO₂ at 0.5 ppm plus O₃ at 0.25 ppm. At the end of the 24-day fumigation period, number of leaves per plant, leaf fresh and dry weight and leaf area were determined. During the later stages of the fumigation, photosynthesis measurements were determined for single attached leaves from plants fumigated with SO₂ and those fumigated with SO₂+O₃. Photosynthetic measurements were made at ambient CO₂ levels (300 ppm CO₂) and at 1000 ppm CO₂.

RESULTS

Plants fumigated with SO₂ showed little visible evidence of injury while those fumigated with O₃ or a combination of the two began to develop tiny necrotic spots after 5 to 7 days of exposure. Soon after lower leaves began to abscise and by 14 days extensive injury was apparent on leaves below the 7th node from the apex. Both O₃ and SO₂+O₃ treatments caused a reduction in the number of leaves per plant; however, the rate of leaf formation was not affected for the four groups. The number of leaves per plant was lower in the O₃ treatment group than in the SO₂+O₃ group, suggesting an antagonistic

relationship. Fumigation tended to cause a decrease in leaf area, leaf dry weight and leaf fresh weight. This was increasingly true as the distance from the plant apex increased. Pollutant interaction evaluations revealed an antagonistic relationship between the responses to O₃ and SO₂. Ozone which retarded growth most had little or no effect on area, dry or fresh weight of leaves above node 5; however, these parameters were reduced by 25 to 40 percent in leaves fumigated with O₃ at node 9. These and related observations demonstrate that O₃ retards growth of leaves in this plant only during the later stages of their development. Ozone fumigated leaves with visible injury comparable to those fumigated with both SO₂ and O₃ contained less dry matter per unit area.

Photosynthetic measurements on controls as well as plants fumigated with SO₂ and SO₂+O₃ revealed little difference in rate of CO₂ assimilation at 300 ppm CO₂. Even in leaves of SO₂+O₃ fumigated plants where visible injury was extensive the photosynthetic rate was reduced by only 10 to 20 percent. When the CO₂ concentration was elevated to 1000 ppm CO₂ photosynthetic rates for controls, SO₂ fumigated and SO₂+O₃ fumigated leaves (with no visible injury) were elevated three-fold to approximately 35 mg CO₂ dm⁻²hr⁻¹. In leaves in which injury was apparent the photosynthetic rate increases were usually less than two-fold, to a level of approximately 18 mg CO₂ dm⁻²hr⁻¹ (whether the injury was severe or barely discernable).

DISCUSSION

Ozone, and O₃ in combination with SO₂ hasten senescence and promote rate of leaf drop to the extent that control plants have approximately 50 percent more leaves than fumigated ones. Fumigation with SO₂ causes no such effect. Analysis of attached leaves for treatment effects clearly shows an O₃ effect but little or no SO₂ effect. In combination these pollutants at this concentration interact in an antagonistic manner in relation to growth parameters.

Measurements of effects of fumigation on photosynthesis indicate that leaf photosynthesis is not drastically reduced even in the presence of considerable injury. It appears that injury tends to reduce leaf potential to respond to enhancement conditions more so than the ability to fix CO₂ under conditions where CO₂ may be limiting.

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Behavior of Airborne Fluorides in Soils¹

Janina Polomski, Hannes Flühler, and Peter Blaser²

Accumulation of airborne fluorides in soils is said to be a side effect of minor importance if it is compared with the phytotoxicity of man-made gaseous fluoride. When soil fluoride was considered in the past, F-uptake of plants, leaf injury or yield reduction was of prime concern. Not much is known about the fate of F-pollutants after being deposited onto the soil surface or incorporated in the plant litter layer. Our study aims at providing experimental tools and data for a better understanding of long-termed effects. The objective is to pin down the most significant, controlling mechanisms of the F-mobility in soils.

Field experiments: In the vicinity of a 30000 t Aluminum smelter, built 1908, we observed that F actually accumulates in the soil (Table). The soils sampled at various distances differ in many respect but are all calcareous. The F-contents do depend upon distance from the emission source.

distance from Al-smelter (km)	total F ($\mu\text{g F per gram soil}$)	watersoluble F	dissolved F ($\mu\text{g F per ml soil solution}$)	F-content of 1 1/2-year old pine needles (ppm)
0.5	2360	159	8.2	472
0.8	1760	153	—	443
1.0	1500	46	—	242
1.8	696	26	3.1	126
5.0	640	24	—	108
6.0	860	14	5.1	60
10.0	520	13	0.3	15

In soils, fluoride coexists in many different forms which makes the total F-content to be a hardly interpretable soil characteristics.

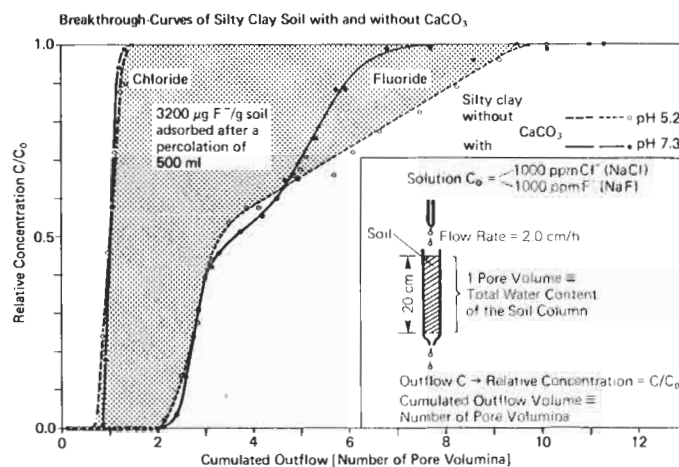
In a pine stand, located 1.8 km from the smelter the magnitude of the annual F-input and leaching losses were determined (kg F per y and ha) : 1-3 kg within dead plant material, 6-10 kg atmospheric washout, 40-80 kg adsorption and sedimentation. The leaching losses out of the root zone were estimated to be some 80-100 kg.

Under pine canopies the F-contents in the surface layer (humus) are systematically higher than in the subsoil (mineral horizon) whereas in uncultivated open land under grass cover fluoride accumulates in the subsoil.

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Laboratory experiments: What controls the fluoride transport in soils? In case of a non-calcareous soil (pH 5.2) the amount of fluoride adsorbed within a mixture of soil and NaF-solution being equilibrated on a shaker (3340 $\mu\text{g F}^-/\text{g}$) soil equals approximately the F-adsorption in the course of a percolation through a soil slab (Fig.)



The close agreement between the two experiments indicates that adsorption and precipitation are fast reactions as compared with the velocity of the percolating soil solution. In calcareous soils, however, the exchange kinetics between the liquid and solid phase controls the extent of accumulation and depletion. The exchange proceeds too slow to keep up with the pace of the moving water. This explains the amazingly high F-mobility observed in case of the F-contaminated calcareous soils.

In the course of the percolation experiments the leachates from the soil columns contain appreciable amounts of solubilized organic matter and aluminum. For a given soil type the leaching losses depend in the first place upon the F-content of the infiltrating solution and further upon soil type and the F-compounds used to make up the percolating solution. Under field conditions with 8-10 ppm F in the soil solution such phenomena may also occur. A significant amount of soluble organic material can be lost which may represent an energy substrate shortage for the microorganisms, and furthermore the Al-concentrations possibly reach phytotoxic levels. At this point the experimental evidence rather backs up questions than provides definite answers.

Multiple Pollutant Fumigations Under Near Ambient Environmental Conditions Using a Linear Gradient Technique¹

P.B. Reich, R.G. Amundson, and J.P. Lassoie²

Investigations of the impact of atmospheric pollutants on plants normally involve artificially raising the concentration of certain gases around study plants. Such experiments typically utilize enclosures which greatly modify other environmental factors. Hence, these experiments may not accurately test plant responses under ambient conditions. The development of open-top, field fumigation chambers (Mandl and others 1973) and a new linear gradient exposure system (Shinn and others 1977) have been attempts to solve this problem.

In 1979 we constructed a modified linear gradient system and 6 open-top chambers. The gradient system enabled the simultaneous exposure of numerous study plants to a concentration gradient of SO₂ and O₃. Since the system does not use enclosures, plants are easily accessed and experience near-ambient environmental conditions. A plot of soybeans (*Glycine max.* var. Hark) in Ithaca, NY was used in an attempt to answer the following questions.

1) Can the gradient system be used to expose plants in the field to controlled levels of air pollutants without significantly altering their environments?

2) Is plant growth significantly different in open-top chambers versus in a linear gradient system?

3) What reductions in soybean seed yield will result from exposure to moderate levels of SO₂ and O₃ during the seed maturation period?

During the seed maturation period, the plants were exposed to SO₂ and O₃ for about 55 h over a 20-day period. On all occasions, a linear concentration gradient was created by the system. Mean concentrations of SO₂ and O₃ at the "high" end of the gradient were 0.16 and 0.06 ppm, respectively, while ambient concentrations of both pollutants were about 0.02 ppm. Maximum hourly means at the "high" end of the gradient were 0.50 and 0.17 ppm SO₂ and O₃, respectively. By monitoring the plot in a grid design, we were able to account for the effects of wind on pollutant levels. In the open-top chambers, plants were

exposed to 0.30, 0.10, and 0.02 (ambient) ppm SO₂ for 72 h, over a 30-day period, during the pod-filling stage.

Comparisons of treatments revealed that the gradient fumigations reduced total bean yield per plant and dry mass per bean by as much as 39 and 14 percent, respectively. In the open-top chambers, total bean yield per plant and dry mass per bean were reduced by as much as 10 and 7 percent, respectively. It is of interest to note that both the "high" and "medium" (mean SO₂-0.09 ppm, mean O₃-0.04 ppm) treatments in the gradient system caused much greater reductions in yield than did the 0.30 ppm SO₂ treatment in the open-top chamber. In interpreting these data, one must remember that the plants in the gradient system did experience short-term peaks much higher than their mean concentration exposures.

In any case, O₃, although present at moderately low levels, was (alone or synergistically with SO₂) probably responsible for the greatest reductions in seed size and yield.

In open-top chambers, one can accurately control pollutant levels. In contrast, the linear gradient system allows fumigations which vary with time and wind, as under ambient conditions. The linear gradient system proved to be an effective means of fumigating plants with multiple pollutants under ambient environmental conditions. Another modification of the gradient system is currently underway so as to allow for analysis of the interactions between SO₂ and O₃, and the effects of such on various hardwood tree seedlings.

ACKNOWLEDGMENTS

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LITERATURE CITED

¹Presented at the Symposium on Effects of Air Pollutants on Mediterranean and Temperate Forest Ecosystems, June 22-27, 1980, Riverside, California, U.S.A.

²Graduate student of Forest Science, Cornell University, Ithaca, NY; Post Doctoral Associate, Boyce Thompson Institute, Ithaca, NY; and Assistant Professor of Forest Science, Cornell University, Ithaca, NY.

Mandl, R.H., L.H. Weinstein, D.C. McCune, and M. Keveny.

1973. A cylindrical, open-top chamber for the exposure of plants to air pollutants in the field. *J. Environ. Qual.* 2:371-376.

Shinn, J.H., B.R. Clegg, and M.L. Stuart.

1977. A linear-gradient chamber for exposing field plants to constant levels of air pollutants. DOE-Preprint UCRL-80411.

Changes in Plant Communities with Distance from an SO₂ Source¹

Paul R. Scale²

Investigations were carried out to assess the effects on plant community composition of sulphur dioxide (SO₂) emitted from an iron sintering plant in Wawa, Ontario, Canada. Diversity indices and the ordination techniques of Correspondence Analysis (C.A.) and Principal Component Analysis (P.C.A.) were used to discern community level changes. Of special interest were the subtle changes in areas noted to be only slightly affected by SO₂ emissions. Over 60 upland birch stands were accessed along a north-easterly transect away from the source at distances of 12 to 55 km. Species specific data was collected on the tree, shrub and ground flora.

Changes in species composition are dramatic as the 30 year SO₂ source is approached. The total percent cover of the dominant, predominantly boreal, ground flora species are shown in Figure 1. They reflect the complex pattern of species abundance which occurs along the transect. Characteristic is the transitory increase in abundance which occurs in most species. What is particularly evident is the ability of many species to take advantage of the increasing damage caused first to the tree canopy and subsequently to the shrub layer. This

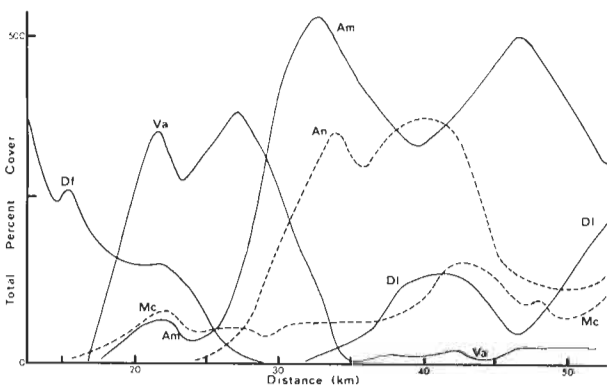


Figure 1--Changes in ground flora species abundance with distance using a running average of percent cover: Am, *Aster macrophyllus*; An, *Aralia nudicaulis*; Df, *Deschampsia flexuosa*; Dl, *Diervilla lonicera*; Mc, *Maianthemum canadense*; Va, *Vaccinium angustifolium*.

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occurs despite their own greater exposure to SO₂. The low-bush blueberry (*Vaccinium angustifolium* Ait.) for example, becomes very abundant at distances of 20 to 30 km yet shows considerable SO₂ damage in that area. The first substantial change in species abundance occurs at a distance of 40 km where beaked hazel (*Corylus cornuta* Marsh.) becomes very abundant before a very rapid decline. This undoubtedly contributes to the drop in the abundance of the large leaved aster (*Aster macrophyllus* L.) at that distance.

Dealing collectively with all species of a community to discern changes caused by a pollutant is, for theoretical reasons and from experience, a more satisfactory approach than changes in individual species abundance. Diversity indices have been used extensively for this purpose. However, the data from Wawa indicates that although diversity indices can deal with spatial/structural changes in the community, they do not adequately reflect the subtler changes in species composition. Ordination techniques, such as C.A. and P.C.A., are better suited for this purpose. For example, the Shannon-Weiner Diversity index has a transitory increase between 20 and 30 km before declining at distances below 20 km. In contrast, the first axis of C.A. using ground flora data shows a simple, linear change from 12 to 33 km (Figure 2). C.A. is most effective in reflecting overall trends whereas P.C.A. is particularly useful in discerning differences in stands of very similar composition. On the basis of P.C.A. using shrub density data, the separation of sites in intermediate areas from controls can be achieved. The boundary between the two groups in Figure 2 corresponds well with the line designated in 1973 and 1974 as separating areas which do and do not show visible leaf damage.

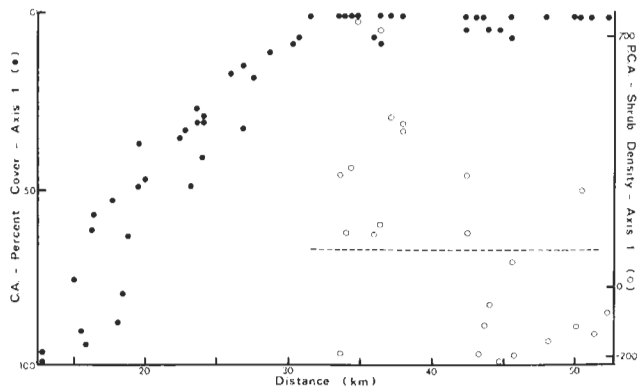


Figure 2--Distance vs Axis 1 of ground flora (percent cover) and shrub density (number of stems under 5 cm diameter at breast height) data using C.A. and P.C.A., respectively.

Lichens as Ecological Indicators of Photochemical Oxidant Air Pollution¹

Lorene L. Sigal and Thomas H. Nash III²

From the late 1960's to the present, both the concentration and dose of photochemical oxidant air pollutants (ozone and peroxyacetyl nitrate or PAN) have continued to increase in the mountainous areas surrounding the Los Angeles Basin. Since 1972, a multi-disciplinary team of ecologists has monitored and analyzed the ecological consequences of this pollution in the conifer forest ecosystem of the San Bernardino Mountains. Our lichen study parallels and expands the parameters of the aforementioned study and allows a comparison of the relative sensitivity of lichens versus higher plants to oxidants.

Historically, lichens have been demonstrated to be sensitive indicators of air pollutants such as sulfur dioxide. More recently, there is strong evidence that lichens are also sensitive to hydrogen fluoride and heavy metals. The present study documents for the first time that lichens are also sensitive to ozone and peroxyacetyl nitrate.

Field investigations were conducted in four mountain ranges surrounding the Los Angeles Basin. A fifth site in Cuyamaca Rancho State Park, east of San Diego, was chosen as the control area. Results of sampling lichens on *Quercus kelloggii* Newb. and conifers showed overall species richness decreased by 38 percent in the highly impacted San Bernardino Mountains. Cover values for species on *Q. kelloggii* decreased by 16 percent; for lichens on conifers the decrease was 78 percent. The latter decrease is large due to the fact that the "sensitive" fruticose lichen species are found mainly on conifers. There was an inverse relationship between the cover of lichens at breast height on conifers and the oxidant dose estimates at sites in the San Bernardino Mountains (fig. 1). A threshold is seen at 180 ppm-hrs above which the lichens are almost absent.

Fumigation studies in the laboratory provided a definitive way of testing the response of field sensitive and field tolerant species to controlled concentrations of ozone and PAN similar to those

occurring in the field. Injury was documented as a significant reduction in gross photosynthesis. *Parmelia sulcata* Tayl. exhibited greater sensitivity to ozone and PAN than *Hypogymnia enteromorpha* (Ach.) Nyl. These results were consistent with field observations.

As a result of collections made by H. E. Hasse and the subsequent publication of his "Lichen Flora of Southern California" in 1913, the present distribution and vitality of a number of lichens was compared to their distribution at the turn of the century. There was a 50 percent decrease in species richness. In addition, marked morphological deterioration was seen in the thalli of existing foliose species such as *Hypogymnia enteromorpha*. Forty-two percent of the thalli collected in the San Bernardinios were bleached and 44 percent were convoluted in contrast to no bleaching or convolution in collections made in the Cuymacas. Thallus dimensions were decreased by approximately 50 percent in the San Bernardino collections.

The importance of lichen studies lies in their sensitivity to air pollution and their potential use as bioindicators. By monitoring the abundance and distribution of sensitive lichens, it is possible to estimate the magnitude of air pollution. Use of lichens in this matter may be feasible not only as currently applied in northern Europe with sulfur dioxide air pollution, but also over large geographical areas impacted by photochemical oxidant air pollution.

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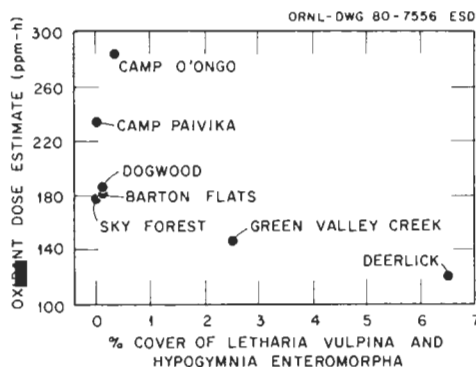


Figure 1--Percent linear cover at breast height of *Letharia vulpina* (L.) Hue and *Hypogymnia enteromorpha* (Ach.) Nyl. as a function of the oxidant dose estimates (ppm - hrs. = the mean oxidant concentration in ppm multiplied by the time of exposure) at sites in the San Bernardino Mountains.

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The Effects of Air Pollutants on Forest Ecosystems in S.R. Slovenia¹

Marjan Šolar

INTRODUCTORY DATA

Slovenia with its surface of 20,251 km² represents 8% of the territory of Yugoslavia (255,804 km²). It is situated in the extreme northwestern part of the state between the Alps, the Pannonian plain, the Adriatic sea and the Dinaric orographic system. Half of the surface is covered by forests (1,000,000 ha), 2/3 of which are privately owned. The portion of conifers amounts to 56%, the average wood mass is nearly 200 m³/ha. The total increment per year is 4,000,000 m³, the cutting 3,000,000 m³. Two-thirds of the wood harvest possess a technical value. The portion of forestry within the national income is 4% and indicates well the level of the industrial development of Slovenia.

Slovenia is divided up into 15 forest management areas and each of those into the socially and privately owned sectors. The principles of management are unified. The basic principle is to manage as much pronaturally as possible to secure the permanency of yield and functions of the forest. The forest service enjoys in some places a 200-year tradition.

The study of effects of air pollutants on the forests goes back to the year 1926, and systematic investigations were started in 1969.

OBJECTIVES OF INVESTIGATION

We wish to determine the parameters, intensity and perspectives of the effects of air pollution on the forests in Slovenia, with the aim to secure reliable foundations for the planning of the industrial development running parallelly with the preservation of the multifunctional forest, for the regulation of indemnities and the elaboration of correct normative concerning the maximum tolerable concentrations valuable for this specific forest area.

PRELIMINARY FOUNDATIONS REQUIRED

Ecology (geology, soils, phytocoenology, climate) as bases for the determination of forest ecosystems not stressed by emission, forest inventory as a basis for the determination of the normal management

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situation of the forests and a register of emitters.

METHODS OF ASSESSING THE EFFECTS OF POLLUTED AIR ON FORESTS

Symptomatics, chemical analyses, increment analyses, bioindicators, presence of differently aged needles, presence of secondary pests, infracolor-aerophotographs.

SPECIFIC FEATURES OF THE TERRITORY OF SLOVENIA

Accentuated configuration, industry often located in narrow valleys, temperature inversion, extremely high emission values (shocks), high proportion of susceptible conifers, erodible soils, predominancy of SO₂ and HF.

RESULTS OF INVESTIGATION

The total surface of visibly damaged forests in S.R. Slovenia amounts to 22,000 ha i.e. 2.2% of the total forest area (situation in 1977). The internal division of damaged forests is the following:

Group A-the narrower emission area- the forest destroyed and existentially threatened, or surfaces under the impact of critical emission conditions entailing the total destruction of all forest functions-destruction of the forest ecosystem. Surface 4,000 ha.

Group B-the wider emission area-forest under permanent emission impact, yet the existence of the forest is not threatened, its functions are however reduced, or surfaces under to heavy emission conditions to allow a normal thriving of the forest. The emission causes an instability of forest ecosystems. Surface 16,000 ha.

Group C-Periodic emission area-periodic occurrence of damages, possibly acute ones, but between individual emission influences in every case longer normal situations are intercalated during which the forest can recover. Surface 2,000 ha.

Based on the chemical analyses of Spruce needles as to the increased content of sulphur and fluoride, the surface of forests with hidden injuries is estimated to 25,000 ha. Survey and detailed maps of injured forest have been made for all bigger emissions areas. On the basis of relative resistance capacity of forest trees and of their proportional presence in the forest vegetation communities a categorization of forests with respect to their susceptibility to air pollution has been carried out.

The process of a further improvement of the emission damages in the forests observed during the last three years is not likely to proceed because of the increase of consumption of coal possessing high contents of combustible sulphur.

CONCLUSIONS

On the basis of investigations carried out in the total forest area of Slovenia influenced by emission, of knowledge concerning the ecology of this territory and the particularities of effects of the polluted air, we are able to predict exactly the future of a certain forest under certain emission influence.

The standpoint of foresters is always the following: Wherever the forest has been destroyed or is on the way of destruction, the air is locally polluted to a relatively critical extent. The air pollution is too high also in places where the forest is hampered in its development. Appealing to the normatives fixed by law and concerning the maximum emission values allowed

means nothing but that the normatives are set too high and that the foresters are bound to lower them.

The Resolution issued on the Xth Meeting of the IUFRO Group 2.09-Air Pollution, Ljubljana 1978 is proposing the normatives insuring the normal thriving of all kind of forests on average and extreme sites. The values for SO₂ are following (in µg/m³/air): 50 as yearly average, 100 as 24 h average, 150 as ½ h value, and for HF (in µg/m³/air) 0.3 as yearly average and 0.9 as ½ h value. For extreme sites the normatives are twice as sharp.

If we compare the normatives fixed by Resolution to those fixed by law in different countries we realise the last ones are much too high.

Population Differences in Response to Sulfur Dioxide: a Physiological Analysis¹

G. E. Taylor, Jr. and D. T. Tingey²

The environment exerts a profound influence on a plant's ability to survive and reproduce, and consequently vegetation in a given area will possess a set of morphological and physiological traits that enhance fitness. At the level of populations, these site-specific attributes arise through either a modification of the phenotype (phenotypic plasticity) or a change in the gene pool (ecogenetic adaptation). Both responses are common strategies among natural populations experiencing disparate climatic, edaphic and biotic stresses. It is hypothesized that elevated levels of atmospheric pollutants have elicited in a comparable manner a variety of traits that enhance survival and reproduction in native plants inhabiting pollution-stressed areas.

One example of ecogenetic adaptation in response to air pollution is the evolution of sulfur dioxide resistance within populations of *Geranium carolinianum*, an herbaceous winter annual common in disturbed habitats in the Southeastern United States. In comparison with their counterparts from pollution-free regions, populations sampled from areas experiencing variable SO₂ stress for 31 years were consistently more resistant to SO₂ under controlled exposure conditions. This infraspecific variation is genetically determined and quantitatively controlled. Plants of contrasting SO₂ resistance were used to investigate the physiological basis of this adaptation. Individual plants were placed in a whole-plant gaseous exchange system in which concurrent steady state measures of leaf resistance to water vapor efflux and SO₂ influx were monitored in the dark and light at pollutant concentrations of 0.4, 0.6, and 0.8 $\mu\text{l l}^{-1}$. For resistant and sensitive plants at each concentration, estimates of total SO₂ flux ($\mu\text{g cm}^{-2}\text{hr}^{-1}$) as a function of leaf resistance to H₂O efflux were modeled using linear regression techniques. From estimation procedures, total flux was partitioned into leaf surface and internal fractions.

Total SO₂ flux varied as a function of leaf resistance and did not differ among resistant and sensitive plants at 0.4 and 0.6 $\mu\text{l l}^{-1}$ SO₂. Conversely, at the highest concentration total SO₂ flux was not the same for the 2 plant groups as a consequence of disparate slope parameters. Irrespective of concentration the leaf surface and interior were major sinks for SO₂ (Fig. 1). Each fraction increased linearly with concentration,

however the ratio of internal to total flux decreased steadily. The absolute values for internal SO₂ flux were strikingly similar for both plant groups at each concentration. Therefore, in spite of the variation in total SO₂ flux to the plant, the rate of SO₂ absorbed into the leaf interior was equivalent for resistant and sensitive plants. Since leaf resistance to water vapor efflux is the same for all plants in both ambient and SO₂-polluted atmospheres, overt plant-to-plant differences in their response to SO₂ reflect disparate internal biochemical processes affecting pollutant toxicity, perturbation or cellular repair.

The fact that plant differences in response to SO₂ are genetically controlled and not a result of pollutant exclusion is relevant to an understanding of the rapid evolution of population resistance in *G. carolinianum*. This species is an annual that thrives for six months as a winter rosette during which new leaf growth is minimal and yet transpiration remains active. This habit coupled with the plant's inability to track and exclude elevated levels of SO₂ may predispose this species to accumulate more SO₂ derivatives than other co-occurring species that are either more ephemeral or capable of avoiding SO₂ uptake via increasing stomatal resistance. These species' attributes may explain the consistent field observations that show *G. carolinianum* to be a sensitive biological indicator of elevated SO₂ levels.

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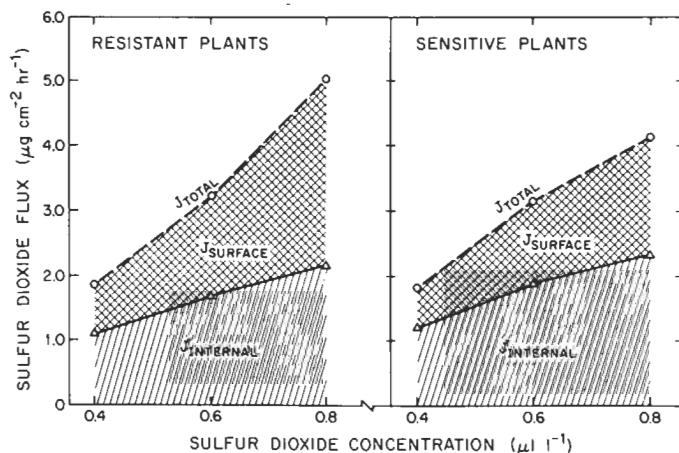


Figure 1--Absolute values for total, surface and internal leaf flux of SO₂ as a function of pollutant concentration in resistant and sensitive plants.

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Ozone Injury to Pines in the Southern Sierra Nevada of California¹

Detlev R. Vogler and John Pronos²

The Forest Service began evaluating the impact of ozone on Sierra Nevada forests in 1974. Evaluation efforts were expanded and intensified in 1977, and since then the primary methods of assessment have included, (1) monitoring ambient ozone levels, and (2) rating injury to pine foliage in permanent trend plots.

Air pollution injury to commercial pine stands located between 4000 and 8000 feet elevation occurs quite uniformly along most of the southern Sierra Nevada front range adjacent to the San Joaquin Valley and up into the major river drainages. Symptoms are not confined to localized areas downwind from major pollutant sources. Rather, ozone is carried down the entire San Joaquin Air Basin and eastward into the Sierra. Metropolitan areas contributing to the ozone dose include Stockton, Modesto, Merced, Fresno, Visalia, Bakersfield, and perhaps even Sacramento and the San Francisco Bay Area.

OZONE MONITORING

Five forested locations in the southern Sierra were monitored for season-long ozone dosages between 1977 and 1979. One site -- Whitaker Forest -- was monitored continuously during this 3-year period. The sites ranged in elevation from 5400 feet to 7540 feet, and all were at least 50 miles from suspected metropolitan sources of air pollution.

Ozone levels at each site exceeded both the Federal (0.12 ppm) and State (0.10 ppm) Standards each year they were recorded. Based on the number of hours exceeding the Standards, 1977 was the worst year for ozone, while levels declined steadily in 1978 and 1979. During the summer of each year daily peak ozone values commonly ranged between 0.10 ppm and 0.14 ppm. These values can be compared to those in the San Bernardino Mountains of southern California, where maximum daily ozone levels frequently range from 0.20 to 0.33 ppm, and where injury to pine forests is correspondingly more severe.

TREND PLOTS

¹Presented at the Symposium on Effects of Air Pollutants on Mediterranean and Temperate Forest Ecosystems, June 22-27, 1980, Riverside, California, U.S.A.

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Trend plot data were collected for three consecutive years beginning in 1977. In 1977 and 1978 ozone injury ratings of plots scattered throughout a sampling area of over 1 million acres ranged between no injury and moderate injury. Although the majority of 1978 plots showed more oxidant symptoms than 1977 plots, there was no statistically significant trend evident. The change between 1977 and 1979, however, was more dramatic. Nineteen of 27 plots evaluated in 1979 showed additional injury over 1977 levels, and seven of these plot differences were statistically significant ($P = 0.05$). In 1979, for the first time, several plots fell into the severe injury category. The trees in these plots averaged 2 years of needle retention with ozone symptoms on 2-year-old needles.

This obvious increase in injury was not expected because measured ozone levels in 1979 were generally lower than in either 1978 or 1977. Considerable physiological stress from a 2-year drought (1976-1977) could account for part of the observed increase in foliar injury. One visible response of pines to the drought was a dramatic decrease in needle retention, which tended to produce more severe ozone injury ratings. Future recovery of trees from drought stress, in the form of increased needle retention, may result in a relative reduction of measureable air pollution injury during the next few years.

CONCLUSIONS

The overall level of forest-wide injury in the southern Sierra Nevada can be termed slight, with some local populations of susceptible pines showing moderate or worse injury. Unexpectedly high amounts of injury detected in certain areas in 1979 placed some plots into the severe category. It is presently unclear whether this is a permanent trend of increasing impact, or just a short-term result of drought stress.

With only 3 years of monitoring data, trends in annual ozone dose are difficult to assess. Ozone levels seemed to decrease slightly each year between 1977 and 1979, even though visible injury to pines increased. Measured variations in seasonal ozone dose may be associated more with weather patterns than with decreased pollutants at the source. Maximum daily ozone values in the Sierra still remain about 1/2 or less of those occurring in southern California. As ozone continues to be transported into the western slopes of the Sierra Nevada, sensitive pines will continue to show additional injury and will slowly decline.

Modifications of Chemical Contents of Precipitation by Passage through Oak Forests¹

George T. Weaver and Jon D. Jones²

Bulk precipitation monitored within oak forests in the Ozark Physiographic Province in Illinois was strongly acidic ($\text{pH} < 4.5$) throughout most of 1978. Exceptions occurred during late summer and early autumn when pH rose to 5.8. This pattern with similar pH values reoccurred during the autumn and winter of 1979-80 and differs from seasonal trends usually reported from the northeastern United States where periods of greatest and least acidity are summer and winter, respectively. During the 1979-80 period, the highest concentrations of SO_4^{2-} occurred during autumn, and on an equivalence basis, exceeded H^+ concentrations by a factor of 10. Concentrations of Ca^{2+} , Mg^{2+} , and K^+ were also high during autumn. It is hypothesized that considerable neutralization of strong acids occurred in the atmosphere due to the presence of airborne dust which normally exists during autumn.

Additional neutralization occurred as precipitation passed through the forest canopy, especially during autumnal leaf senescence. At this time canopy drip pH exceeded bulk precipitation pH by up to 0.7 units. The concentrations of Ca^{2+} , Mg^{2+} , K^+ , SO_4^{2-} and PO_4^{3-} in canopy drip increased markedly compared to bulk precipitation, particularly during autumn. During the winter, canopy drip pH decreased to values as low as 3.8 and differed little from bulk precipitation pH. In 1978, 68 percent of the precipitation reaching the forest floor as canopy drip was acidic ($\text{pH} < 5.6$). The removal of nutrient ions from the canopy also de-

creased during winter. In the spring of 1980 maximum bulk precipitation pH was 6.6 but passage through the canopy caused a decrease in pH as great as 0.8 units. Soluble organic compounds, apparently leached from some oaks, were present in sufficient quantities to impart a dark stain to canopy drip during this period and may be associated with the reversal in H^+ exchange in the canopy.

The concentrations of ions in soil water formed three distinct patterns relative to concentrations in bulk precipitation and canopy drip. Hydrogen ion concentration was decreased markedly by passage through these ecosystems and the quantity remaining in soil water was only about 10 percent as great as in bulk precipitation. The concentrations of four ions - SO_4^{2-} , NO_3^- , Ca^{2+} , and Mg^{2+} - increased as water passed through these ecosystems. The levels of these ions in soil water were as great as 37 times and 9 times the levels in bulk precipitation and canopy drip, respectively, although major differences occurred among seasons and between ions. The contents of K^+ and PO_4^{3-} in bulk precipitation also were increased by up to 25 times by passage through the forest canopy. However, concentrations of these particular ions either decreased or remained similar to concentrations in canopy drip after passage through the soil.

Between October 1979 and April 1980, Ca^{2+} and SO_4^{2-} were the predominant ions (equivalence basis) associated with meteorologic and hydrologic processes in these ecosystems. The importance of Ca^{2+} in these ecosystems was anticipated since it is selectively accumulated by some species of oaks in these forests. It is also apparent that elevated levels of H^+ are being deposited in association with SO_4^{2-} in these ecosystems, but the impacts remain undetermined.

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²Associate Professor of Forestry, Southern Illinois University, Carbondale, Ill.; Forestry Aide, Baruch Science Institute, Georgetown, S.C.

Seasonal Variation of Inorganic and Organic Sulfur in Coniferous Needles Intensified by SO₂ Pollution¹

Karl Friedrich Wentzel und Günther Gasch²

Method:

Needles from 50 - 70 year old Norway spruce from 1. gardens and parks of Wiesbaden town (slight damage), and 2. Wiesbaden forests (no visible injury) were used as bioindicators for SO₂-uptake. In october we picked 0.5, 1.5 and 2.5 year old needles, in may we picked 1, 2 and 3 year old needles from the tops of the same trees. Inorganic and organic sulfur was determined according to Jäger und Steubing (1970).

Results:

1. Air pollution measurement 1976 - 1979. Datas in microgramm SO₂ per cbm air. I1 = annual average, I2 = 95 percentile of 30 min. values:

Zone	I 1	I 2
Industrial Area	120 - 140	400 - 500
Town	70 - 90	200 - 300
Forests	< 50	< 120

2. Total sulfur content increases with needle age as shown in Fig. 2. In town the S contents of the needles are 100 - 200 ppm higher than those from the forests.

3. The inorganic fraction of all needle samples surmounts the organic fraction. The difference is greater in autumn than in spring and more apparent in town than in the forests.

4. The validity of using coniferous needles as indicator of air pollution effects is best when october datas of inorganic S are used.

5. Organic sulfur content decreases during the vegetation period while the inorganic fraction increases. During winter time the opposite occurs (Fig. 3). From this it is suggested that in spring time a part of the organic sulfur demand is covered by removing the inorganic amount, which partially is of air-borne origin.

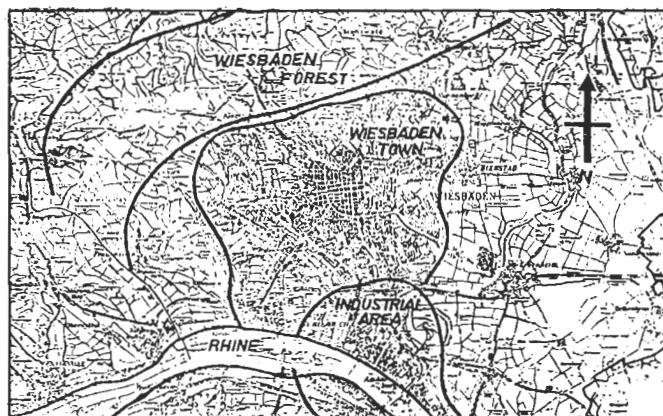


fig. 1 : Map of Wiesbaden/FRG

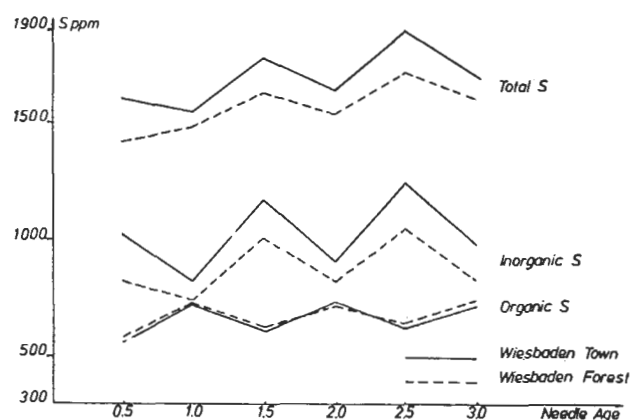


Fig. 2 : Sulfur content (ppm of dr. wt.) in spruce needles from Wiesbaden town (means of 39 tree samples) and forests (means of 16 tree samples).

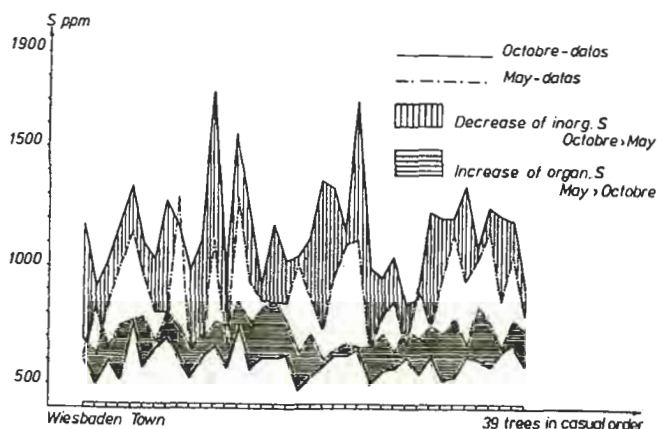


fig. 3 : Seasonal alteration of inorganic and organic S in spruce needles from Wiesbaden town (means of 6 semi-annual needle sets).

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Sulfur Dioxide and Oxidant Effects on Californian Coastal Sage Scrub¹

W.E. Westman and K.P. Preston²

Two field studies have related oxidant and sulfur dioxide pollution to a deterioration in the structure and function of southern California's drought-deciduous shrublands (coastal sage scrub). In a study of 67 sites (0.63ha) from San Francisco to El Rosario (Baja California), data were collected on species cover and 43 habitat variables describing topography, soil, climate, vegetation structure, age since fire, grazing intensity, and air pollution of sites. The variable which showed the highest significant correlation with percent foliar cover of native species was the mean annual oxidant concentration ($r = -.58$, $P < .001$). Elevation and mean maximum temperature of the warmest month also showed highly significant correlations ($r = -.52$). The partial correlation coefficients of oxidants with percent cover remained high when covariations with elevation, mean maximum temperature of the warmest month, and distance from the coast were extracted ($r = -.41, -.35, -.42$ respectively). The interrelationships of these variables were investigated further by means of path analysis. A path model (chi-square probability = .87) related environmental factors to a reduction in the percent cover of native species. The path model suggested that other factors correlated with declining cover were acting primarily through their influence on oxidant concentration in predicting the decline in cover of native scrub species.

Increasing concentrations of oxidants were also associated with a decline in species richness ($r = -.23$; $P < .05$) and increase in equitability (Whittaker's E_c index; $r = .24$; $P < .05$). Diversity in floristically similar sites of high and low annual oxidant concentrations was compared. Highly polluted sites have fewer species per abundance class and a lower total species richness ($\bar{X} = 18$ v. $\bar{X} = 29$ at less polluted sites). Concentration of dominance increases in the more polluted sites. Major results have been reported in Westman (1979).

In a second field study, the effects of sulfur dioxide emissions (up to 0.13 ppm for a period of 25 years) from an oil refinery were studied near Santa Maria, on the rural central coast of California. Stands of coastal sage scrub with black sage (*Salvia mellifera*) downwind of the refinery were compared with stands in relatively pollution-

free upwind sites. Injury to individual *Salvia* shrubs and changes in community structure and floristic composition were recorded. Stomatal resistance was found to be significantly lower on the polluted sites with a concomitant 35 percent increase in mean transpiration rates of *Salvia*. Flowering capacity of *Salvia*, measured by the number of flower whorls per flower spike, was also significantly reduced. Significant decreases in the height/width ratio of *Salvia* were hypothesized to be caused by sulfite-mediated destruction of indole acetic acid (Yang and Saleh, 1973) and possible inhibition of apical dominance. Significant reductions in photosynthetically active tissue of *Salvia* resulted from increased defoliation and reduction in leaf size associated with SO_2 stress. The evidence indicates that these factors decreased the shrubs' ability to compete with the more r-selected annuals. As such, the number of species, primarily annuals, increased considerably in the most polluted sites ($\bar{X} = 27$ v. $\bar{X} = 18$ on control sites). These SO_2 -associated changes in community structure and floristic composition suggest that retrogression is occurring, causing the 26-year old stand to resemble a 7-year old post-fire seral stage. Results are reported more fully in Preston (1980). Laboratory studies are in progress to assess sensitivity of sage scrub species to SO_2 injury.

Preston, K.P.

1980. Effects of sulfur dioxide pollution on coastal sage scrub. M.A. Thesis. Department of Geography, University of California, Los Angeles.

Westman, W.E.

1979. Oxidant effects on Californian coastal sage scrub. Science 205:1001-1003.

Whittaker, R.H.

1972. Evolution and measurement of species diversity. Taxon 21:213-251.

Yang, S.F. and S.A. Saleh.

1973. Destruction of indole-3-acetic acid during the aerobic oxidation of sulfite. Phytochemistry 12:1463-1466.

¹Presented at the Symposium on Effects of Air Pollutants on Mediterranean and Temperate Forest Ecosystems, June 22-27, 1980, Riverside, California, U.S.A.

²Associate Professor of Geography and graduate student, respectively, University of California, Los Angeles, California.

Miller, Paul R., technical coordinator.

1980. **Proceedings of the symposium on effects of air pollutants on Mediterranean and temperate forest ecosystems, June 22-27, 1980, Riverside, California, U.S.A.** Gen. Tech. Rep. PSW-43, 256 p. Pacific Southwest Forest and Range Exp. Stn., Forest Serv., U.S. Dep. Agric., Berkeley, Calif.

These proceedings papers and poster summaries discuss the influence of air pollution on terrestrial and related aquatic ecosystems. They describe single species-single pollutant relationships; interactions of producers, consumers, and decomposers under pollutant stress; and the use of ecological systems models for interpreting and predicting pollutant effects.

Retrieval Terms: air pollution injury, acidic precipitation, pollutant stress, terrestrial habitats, species-pollutant relationship, mathematical models.

Synthesis Session

Opening Remarks and Summary of Panel/Audience Discussion¹

Walter E. Westman²

Abstract: In the opening remarks by W. E. Westman, three major questions were raised: (1) How can resilience of ecosystems to air pollution damage be predicted? Models constructed from key physiognomic, physiological or life-history attributes of dominants within a community could form the basis of an aut-ecological approach. Using a synecological approach, at least four distinct components can be recognized (elasticity, amplitude, hysteresis, malleability) which reflect different aspects of the recovery process. These could be measured in the field or derived from modeling, using such community-level attributes as components of diversity, foliar cover, and similarities in composition. (2) What indicators of effects of air pollution on ecosystem nutrient cycling are most reliable? Observing changes in the mineral composition of fresh litterfall during peak litterfall periods may be preferable to measuring foliar nutrient concentrations in relatively small samples of forest species. (3) What is the role of cultural values in air pollution research? What are the social responsibilities of scientists, and how can they be discharged? Examples were given of the role of cultural values and perceptions in the conduct of air pollution research and interpretation of its results. Scientists may assist decision-makers in interpreting the significance of results of air pollution effects on forests by illustrating the external costs generated in the economy by loss of ecosystem functions, as well as structure, due to air pollution damage. In the open discussion that followed, participants discussed the use of air pollution simulation models in making decisions about land use. Modelers indicated that reparameterization of existing vegetation models to local conditions could provide an efficient means of applying existing models to local siting decisions. Field biologists and managers expressed some reservations about the level of precision to be achieved from such a procedure. Some attributes of the forest ecosystem which are most indicative of pollution stress, and hence most usefully incorporated in such models, were detailed, including physiognomic attributes, visible foliar injury symptoms, wood growth rates and lichen composition. At least nine areas in which research is needed on the effects of air pollutants on forests were suggested.

¹Presented at the Symposium on Effects of Air Pollutants on Mediterranean and Temperate Forest Ecosystems, June 22-27, 1980, Riverside, California, U.S.A.

²Associate Professor of Geography, University of California, Los Angeles, California

OPENING REMARKS

The purpose of this final discussion session is to identify ecosystem-level concepts pertinent to the study of air pollutant effects of forests, and to provide an opportunity for discussion between participants at the Symposium

regarding key questions arising from the Symposium proceedings. To assist in this effort, Symposium session chairmen (Patrick Coyne, Joe McBride, Samuel McLaughlin, Jr., Joseph Shinn, William H. Smith, and David Tingey) will serve as a panel to field discussion questions from the audience. I have been asked to open the session with some "provocative" questions regarding ecosystem-level studies of air pollution effects, arising from the Symposium presentations and field trips of the last four days. I cannot guarantee that anything more than my garish red tie will be provocative, but I would like to pose three questions for your consideration.

HOW CAN RESILIENCE OF ECOSYSTEMS TO AIR POLLUTION DAMAGE BE PREDICTED?

The term "resilience" has been used to refer to the degree, manner and pace of restoration of initial structure and function in an ecosystem after disturbance (Westman, 1978). Most of the papers in this Symposium have discussed not resilience, but what has been termed ecosystem "inertia" (Orians, 1975; Westman, 1978). "Inertia" is the ability of an ecosystem to resist displacement in structure or function when subjected to a disturbing force. In the case of air pollution, inertia can be measured by determining the minimum concentration of a pollutant at which impact to an ecosystem occurs. We have considerable information on the levels of chronic or acute air pollution that will initiate injury to individual species, and a less ample body of evidence regarding the levels of pollutants necessary to initiate community-level changes.

Ecosystem models, once effectively validated, can potentially serve as tools for the prediction of resilience. Since it is impractical to model all features of an ecosystem, however, modelers need to know which attributes of a species make the organism most vulnerable to pollution injury. Thus, to be of maximum assistance to modelers, experimentalists need to determine which physiognomic and/or physiological attributes of a species (or species association) are most useful indicators of the inertia and resilience of species. Noble and Slatyer (1976) and Grime (1979) have made progress recently in identifying attributes of species which may be useful in predicting their ability to recolonize burned or cleared sites.

We might be able to speed the building of efficient models of air pollution effects on ecosystems by examining species for those attributes which are most vulnerable to pollution damage, or which enhance recovery following damage. During this symposium, we heard that lichens and mosses tend to be more vulnerable to air pollutants because of their lack of waxy cuticles. Beyond this, it would be useful to know more about the relative effects of such leaf attributes as mesophylly, sclerophylly and succulence in providing resistance to ab-

sorption of pollutants. What is the predictive value of phenological attributes, such as evergreenness vs. deciduousness, or life-cycle attributes such as annual vs. perennial reproductive cycles? What are the effects of crown-sprouting ability, or lack thereof, on ecosystem recovery following air pollution damage?

These questions form part of what might be called the "autecological approach" to ecosystem resilience, since they focus on species attributes which, when incorporated in an ecosystem model, can be used to synthesize key properties of ecosystems.

It is possible, however, that progress may be made more quickly by attempting generalizations of ecosystem resilience by studying community-level processes of recovery in particular biomes. In order to organize a study of community-level response to disturbance it is useful to recognize at least four distinct components of resilience (Westman, 1978):

Elasticity

The rapidity of restoration of a stable state following disturbance. To use the analogy of a metal coil, elasticity of the coil is the time required to spring back to initial size after stretching a certain distance.

Amplitude

The zone from which the ecosystem will return to a stable state. By analogy, amplitude is the distance beyond which a coil cannot be stretched without being permanently deformed.

Hysteresis

The degree to which the path of restoration (succession) is an exact reversal of the path of degradation (retrogression). By analogy, hysteresis is the degree to which the region temporarily occupied by a coil in springing back differs from the region through which the coil moved when initially stretched.

Malleability

The degree to which the stable state established after disturbance differs from the original steady state. Similarly, malleability is the degree to which a stretched coil remains stretched after the deforming force is removed.

These components of resilience are subject to measurement (Westman, 1978). In the case of the study of oxidant effects on pines in the San Bernardino Mountains, inertia was determined by observing the levels of oxidants at which damage to pines first appeared. Amplitude might be established by identifying the sample plots, among the several used, at which pines are no longer replacing themselves (if these are taken to be climax species for the region), and determining the lowest mean and

peak concentrations of oxidants at which this threshold effect is observed. Elasticity could be measured as the time necessary for recovery of such a site once pollutant stress is removed. Malleability could be measured, using a percentage similarity index, by comparing the community composition of the new steady state which was established following pollution stress to the pre-stress composition. In examining hysteresis, one would ask whether the first species to disappear from the ecosystem were the last to return, using, for example, a rank correlation coefficient.

It is obvious that in many situations recovery cannot be observed because the chronic stress continues (as is the case in the San Bernardino Mountains). Further, the post-stress recovery period may be on the order of centuries, in which case ecosystem models using autecological approaches must be relied upon for quicker predictions. Nevertheless, there are situations in which a polluting source has been removed or reduced, and recovery can be observed. In these situations, compilation of the components of resilience for a particular plant formation may aid us in generalizing about the impact of new pollution sources on as yet unimpacted areas of vegetation of similar type.

Thus in considering the prediction of ecosystem resilience, we may be wise to focus both on sensitive attributes of individual species or species associations, and of community-level changes in species richness, composition, foliar cover, etc. which can form the bases for observing community-level components of resilience.

WHAT INDICATORS OF EFFECTS OF AIR POLLUTION ON ECOSYSTEM NUTRIENT CYCLING ARE MOST RELIABLE?

The nutrient budget of an entire ecosystem represents one ecosystem level attribute that can reveal much about growth-potential and functioning at the supraorganismal level. Determining the nutrient budget for even a small portion of the landscape, however, is a very costly and time-consuming process. Hence some speakers (Bruce Wiersma and K. W. Brown, Allen Legge) described attempts to determine effects of air pollutants on nutrient cycling by measurement of concentrations of mineral elements in foliage, as possible bioindicators of pollution-induced ecosystem-level damage. Paul Zinke described some of the many ecosystem compartments which must be considered in conceptualizing ecosystem nutrient cycling.

I would like to issue a caution against the use of foliar nutrient concentrations, in the absence of considerable context, for the study of pollution stress. Soil scientists have long recognized that foliar nutrient concentrations are dependent on soil nutrient concentrations, and have used the foliage analyses as indicators of "available" concentrations of the elements in the soil in a number of instances. Thus it

becomes very important for the air pollution researcher to characterize soil heterogeneities in his or her study. Secondly, ecologists know that foliar nutrient concentrations vary temporally as the leaf passes from early growth stages to senescence and leaf fall. The changes are due to the change in tissue and cellular component ratios with age, to the changing ratio of photosynthate to mineral elements, and to withdrawal of more mobile nutrients into stems before leaf fall. Thus the harvest of leaves at different times of year makes nutrient analyses of these leaves inappropriate for use as samples from a single population. Furthermore, species differ in their abilities to assimilate, retain or accumulate to luxury levels, particular mineral elements. The ability of some species to accumulate certain heavy metals, for example, is the basis for biogeochemical prospecting. Luxury accumulation of potassium by many species is well known. Calcium, being immobile, tends to increase in concentration in leaves with age, but the initial ability to assimilate calcium differs from species to species. Thus the analysis of foliar nutrient concentrations without regard to species or ecotype is to be avoided.

The multiple axes of variation presented by differences in species, mineral properties, time of year and soil concentrations implies that a much more massive sampling program must be undertaken to observe meaningful trends from foliar analyses than has characterized some of the air pollution studies reported.

Short of a full nutrient budget analysis, changes in the mineral composition of fresh litterfall during peak litterfall periods may provide a more suitable indicator of nutrient changes due to pollution stress, as this component is standardized in time, and weighted to the foliar biomass composition of the forest. Even so, large sample sizes are needed, and much caution in extrapolation of results will still be necessary.

WHAT IS THE ROLE OF CULTURAL VALUES IN AIR POLLUTION RESEARCH?

WHAT ARE THE SOCIAL RESPONSIBILITIES OF SCIENTISTS, AND HOW CAN THEY BE DISCHARGED?

As natural scientists, we tend to relegate the social context of our research to other segments of society. We do so at risk, however, because there are both social issues upon which we are most qualified to comment, and sociopolitical forces which affect the choice and conduct of our research problems. In the course of the present Symposium, I was amused to note the variation in perception of the air quality in Riverside during the period, by various participants. Some thought the smog light, others oppressively heavy. The smog concentration was a constant; cultural values were at work in influencing perceptions. As a second example,

some speakers emphasized the role of acid rain as a fertilizer, while others emphasized its toxic properties. Of course, whether any pollutant will exercise its toxic properties will depend upon rate and duration of application, and concentration, as well as upon the physiological state of the receptor organism and its ecosystem. In the case of characterizing the properties of acid rain, however, the attributes chosen for emphasis were chosen for reasons having to do with the cultural attitudes and values of the speakers, and not because of disagreement over empirical observations. As a third example, we heard from one Forest Service representative during the field trip that the damaged portions of the San Bernardino National Forest were not experiencing difficulty in reproduction. But, the E.P.A. sponsored, University of California-Forest Service research team provided evidence to the contrary. Were these two parties disagreeing over empirical observations, or was their difference one of cultural values and perceptions applied in evaluating the significance of the same body of data? As a fourth example, recall that about 50% of the adult American population smokes, thereby bringing into their lungs several times the ambient levels of particulates and toxic gases present in polluted urban air. Will these people perceive the human health hazards of outdoor air in the same way as nonsmokers? Will they assign the same weights to their importance?

At the very least, scientists have a responsibility to differentiate clearly between empirical observations and normative (value) judgments. But does our responsibility to decision-makers and the public stop there? Much has been said regarding the appropriateness of scientists in offering value judgments to society, and I will not enter the debate here. I would, however, like to suggest a way in which scientists can help to clarify in socially-meaningful terms the social costs of air pollution damage.

I believe decision-makers would profit from a fuller understanding of the effects of air pollution not only on the marketable aspects of ecosystem structure (standing timber, crop yield, tourist revenue), but also on those aspects of ecosystem functioning which create hidden costs in the market place. To take an example from the San Bernardino National Forest oxidant study which I have previously discussed (Westman, 1977), consider the dollar costs to society of the loss of the soil binding function from air pollution damage to pines in this forest. As of 1972, 57% of the trees over a 4000 hectare area will be replaced by a retrogressive vegetation of forbs and grasses, and that erosion losses from the latter will increase in the proportion observed when chaparral was converted to grassland by the U.S. Forest Service at San Dimas in the neighboring San Gabriel Mountains, it is possible to estimate erosion losses from the stressed forest. At current rates of cleanup of sediment from streets, sew-

ers and debris basins, the cost of damage from loss of the soil binding function of the San Bernardino Mountain pines is \$27 million per year. These costs are being absorbed currently by the general public not only as direct tax collection to local governments cleaning up the sediment, as only a portion of the sediment is being recovered in this way. The costs are being absorbed also in terms of losses to fisheries in coastal waters where spawning areas are smothered by sediments, by public works allocations for new dams, by flood damage following storms in silted flood channels. Rarely if ever are these costs attributed to smog and considered in the cost-benefit analysis of proposals to install emission control devices. Further, soil binding is only one of the functions destroyed through death of the pines. Loss of the functions of nutrient capture and retention, pollution absorption, climatic regulation and energy fixation all have their social costs, capable of at least partial enumeration and evaluation (Westman, 1978).

Complex as this process of social cost identification is, I suggest that it is a topic in which scientists can play a larger role than we have to date, and a topic to which, it may be argued, we have a responsibility to contribute.

SUMMARY OF AUDIENCE PANEL DISCUSSION

Use of air pollution simulation models in decision-making

Discussion ensued on the current applicability of computer models to such immediate questions as how to site power plants to minimize damage to vegetation. Those with experience in building models expressed considerable confidence that these could be used, upon reparameterization, to help resolve such questions in a number of parts of the country. Experimental biologists and land managers expressed concern that the models were still too broadly conceived to provide accurate estimates of differences between sites in a single vegetation type. A number of the specific issues that were broached as part of this discussion are as follows:

Transferability of models between vegetation types

What level of resolution of a model is needed in order to provide accurate predictions of pollution effects on vegetation, of use to land managers? Are single general models for each biome sufficient, or do we need a model for each of 400 or 500 American vegetation types? How should a model be constructed to maximize its transferability between regions? Several suggested responses were offered:

1. The work of ecologists, air pollution research scientists and modelers could be

coordinated in a nationwide research laboratory consortium to construct working models of air pollution effects on vegetation for the major regional plant formations. Land managers concerned with particular siting or forestry questions in a region of the country could submit a request to the consortium for adaptation of a regional model to his or her particular problem. One advantage of this approach is that biologists and modelers who are most familiar with the assumptions in the model would be available to operate the model. Repetitious research efforts could be avoided, and experience accumulated from various regional efforts.

2. Models which incorporate the mechanistic basis for air pollution effects on vegetation may be most easily adapted to new vegetation types with accuracy. "Mechanistic" bases may consist of physiological models of the effects of pollutants, temperature and moisture on nutrient assimilation and photosynthesis, or they may consist of ecosystem-level models in which sensitive attributes of the ecosystem (litterfall, evergreenness vs. deciduousness, species longevity) are modeled.

3. The National Power Plant Team of the United States Fish and Wildlife Service has available a power plant siting model which, although considering various constraints, does not incorporate a model of air pollution effects on vegetation in detail. They have recently obtained, however, a copy of Kercher's (Lawrence Livermore Laboratory) forest growth model, and plan to make this available for public use, and perhaps ultimately link it to their existing siting model.

4. A forest growth model for a particular vegetation type appears to require 1 - 2 person-years of effort to construct from scratch. Reparameterization of an existing model is regarded as a less costly and time-consuming way to provide a model for a new vegetation type. An important limiting factor to this effort is the paucity of field and laboratory fumigation data on effects of air pollutants, alone and in combination, on species, especially over extended periods. In the absence of such field data, sensitivity analysis of the computer model may be used to establish the likely components of the ecosystem which will be most adversely affected by a particular stress, and to produce qualitative scenarios of worst case events.

5. Existing models for many vegetation types do not incorporate long-term cyclical events such as fires, seed cycles, etc. However, models such as those of Kercher do incorporate these effects. The forest growth model available through Oak Ridge National Laboratory is considered capable of application to all eastern U. S. forest types except Southern pine forests.

Ecosystem-level indicators of pollution stress

What attributes of species or of ecosystems can be used by field ecologists and modelers as sensitive indicators of likely pollution-induced changes?

1. Physiognomic attributes (leaf cuticle thickness and chemistry), phenologic attributes (age to reproduction, duration of foliage), and life history attributes (annual vs. perennial nature, longevity of generations) are seen as community-level indicators. Litterfall may be a useful indicator of nutrient budget processes.

2. Growth rings (dendrochronology) may be used as a long-term record of forest growth responses. In Pennsylvania, for example, tree ring analysis in forest in the vicinity of power plants has been conducted.

3. Visible injury symptoms, such as the foliar injury index used by Paul Miller and co-workers, may be useful. The Tennessee Valley Authority has recorded visual injury symptoms on trees surrounding a number of its power plants.

4. The use of particularly sensitive species, such as lichens, was illustrated earlier in the Symposium.

Use of models in setting air pollution standards

Can air pollution models be used by agencies concerned with establishing minimum concentrations of ambient exposure that will cause damage to vegetation?

1. To the extent that standards incorporate social values as well as scientific criteria, a computer model of the type being discussed cannot be used to set a standard.

2. On the other hand, computer models have been used to determine levels of forest growth reduction from particular levels of pollutants. This information alone, or converted to economic loss figures, can be useful as information for criteria documents used in standard setting.

3. Models have been used to calculate radiation dose to people in the vicinity of nuclear power plants. The Indian Point power plant was modified to reduce damage of effluent to fish life, based on information obtained through a computer model.

4. Models of forest growth have illustrated that even a 5-10% decrement in tree growth due to air pollution can have severe long-term effects on forest growth and composition. This is a prediction that would have been difficult to make with precision in the absence of a computer model.

A Research Agenda

During the course of the synthesis session several suggestions were made regarding areas in which important information is particularly lacking. These research topics are listed below.

Patterns of Recovery in Stressed Vegetation-- While many ecosystem studies have documented the 'inertia' of the systems, fewer have observed the recovery process. The emission source at Trail, British Columbia, is an example of a point source whose emissions decreased dramatically in the late 1930's. Studies of subsequent recovery of damaged vegetation could tell us much about resilience of that forest ecosystem.

Precipitation Chemistry-- Much needs to be known about the chemical transformations in rain and snow as they pass over the surfaces of vegetation and soil, and how these in turn affect the flux of nutrients in the ecosystem.

Mechanistic Models For Particulate (including heavy metal) Pollutants-- Models for the injury to plants from toxic gases are more advanced than those for particulates. How do particulates enter plants and at what rates? How are their effects registered? How can they be modeled?

Synergisms-- Much is unknown about the interactive effects of several air pollutants on individual species and on ecosystems. Fumigation studies should incorporate pollutant combinations as well as test the effects of single pollutants.

Long-term Effects of Pollutant Exposure-- There is a need for laboratory studies of long-term exposures (greater than one month) of air pollutants to species. In the field, exposures to forest species occur over many years. It is difficult to understand long-term effects on reproduction and growth in the absence of chronic exposure studies.

Monitoring Data For Ambient Pollutant Concentrations-- More of these data are needed for all parts of the United States if use of computer models with realistic ambient air concentrations is to be achieved. The same data are needed for interpretation of field observations.

Interaction of Air Pollution Stresses With Management Practices-- How does the effect of an air pollutant on a forest differ under different thinning regimes? This question may be examined in the next phase of research in the San Bernardino Mountain pine forests.

Carbon Dioxide Enrichment in the Atmosphere-- Rising ambient CO₂ levels are affecting forest

growth. How do these effects interact with ambient air pollution levels? How will the affect of CO₂ on global climate further affect pollution response of forests?

Funding of Research on Forest Growth Models Specifically Built to Incorporate Air Pollution Effects-- Most existing models simulate forest growth in the absence of pollution effects, and were funded by agencies other than those concerned with air pollution.

REFERENCES

- Grime, J. P.
1979. Plant strategies and vegetation processes. 222 p. John Wiley and Sons, New York.
- Noble, I. R., and R. O. Slatyer.
1976. Post-fire succession of plants in Mediterranean ecosystems. *In* Proc. Symp. Environmental Consequences of Fire and Fuel Management in Mediterranean Ecosystems. H. A. Mooney and C. E. Conrad, eds. p. 27-36. Forest Service, U. S. Dept. Agric., Washington, D. C.
- Orians, G. H.
1975. Diversity, stability and maturity in natural ecosystems. *In* Unifying Concepts in Ecology. W. H. van Dobben and R. H. Lowe-McConnell, eds. p. 64-65. Junk, The Hague.
- Westman, W. E.
1977. How much are Nature's services worth? *Science* 197: 960-964.
- Westman, W. E.
1978. Measuring the inertia and resilience of ecosystems. *BioScience* 28: 705-710.

Integration: a Role for Adaptive Environmental Assessment and Management¹

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Abstract: Adaptive Environmental Assessment and Management (AEAM) is a proven methodology for integration and analysis of environmental research and management. Simulation modelling workshops are an essential part of the process. They act as a catalyst to focus analysis, stimulate discussion and foster communication amongst managers, planners and scientific disciplinarians.

Some of the philosophy and methodology are described along with a case study example on the Alberta Oil Sands Environmental Research Program (AOSERP). The emphasis is on how AEAM can help foster integration and communication of scientific information.

Over the last decade we have seen phenomenal growth of environmental science. It has spawned a proliferation of public and private institutions dedicated to environmental research and protection. Environmental research and management have become highly skilled, influential, and more often than not, respected professions. Governments have created such well meaning provisions as the National Environmental Policy Act (NEPA), the Clean Air Act and the Water Quality Act in the United States, and the Environmental Assessment and Review Process (EARP) in Canada.

These gallant efforts to research and protect the environment have over the years generated voluminous amounts of information; information intended ultimately to provide knowledge to help mankind better manage the world's environment and resources.

The challenge today is to analyze and make sense of the vast amount of data and knowledge. Unfortunately most research has been disciplinary

based, and inspite of a desire for interdisciplinary coordination most large research programs have failed to integrate their results. One reason for this is the lack of a proper forum for communication amongst scientific specialists; another is the absence of a systematic framework for synthesizing results. While some programs have been successful at pulling the individual studies together few have been able to make the results relevant to environmental management. Many argue that this can be remedied by raising the quality of environmental research. Unfortunately there is a dichotomy between environmental research and environmental management. Research science cannot effectively guide managers because scientists do not readily comprehend management concerns; management cannot adequately direct research science since managers often do not have scientific knowledge or the breadth of understanding such knowledge provides. This inability of scientists and managers to interact effectively has been a major stumbling block in developing progressive policies and attitudes towards the environment both in the public and private sector.

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Adaptive Environmental Assessment and Management (AEAM), well documented in Holling (1978), has evolved over the last ten years into a proven methodology for integrating environmental research while linking research and management. AEAM uses computer simulation modelling in a unique and novel way. Within a structured modelling workshop, the

task of constructing a simulation model acts as a catalyst to focus analysis, stimulate discussion and foster communication amongst managers, planners, and scientific disciplinarians.

BIASES OF AEAM

Ecologist's understanding of ecosystem structure and behaviour have come from four basic properties which determine how ecological systems respond to change (Holling, 1978).

1. The parts of an ecological system are connected to each other in a selective way (this has implications for what should be measured).
2. Events are not uniform over space. (This has implications for how intense impacts will be and where they will occur.)
3. Sharp shifts in behaviour are natural for many ecosystems. (Traditional methods of monitoring or assessment can misinterpret these and make them seem unexpected or perverse.)
4. Variability, not constancy, is a feature of ecological systems that contributes to their persistence and to their self monitoring and self correcting capabilities.

Underlying each of these properties is the fact that environmental systems are characterized by overwhelming uncertainty. Man's understanding of the underlying biological, social and physical processes and interactions is minimal, and will remain so for the foreseeable future. The complex and pervasive nature of environmental and social issues guarantees our ignorance will always exceed our knowledge.

The many sources of error in environmental systems ensure that no matter how broad or deep an analysis inevitably something outside will influence the results and violate the predictions. The conclusion is that environmental and social systems are fundamentally unpredictable. If you accept this hypothesis then, how can environmental science produce effective research and management.

First and foremost we must never promote research and analysis as the panacea for prediction of the fate of society and its environment. Second, we must recognize that decisions are always made under uncertainty, and ignorance. Realizing this we must capitalize on methods that help focus ideas and information, integrate concepts, and guide decision making. Better decisions will usually be made with a clear picture of both knowledge and ignorance, and a broad appreciation of the consequences of action.

The key tool of AEAM is the computer modelling workshops. These short intense meetings circumvent the natural scientific tendency for reductionism and the eternal cry for more studies. Participants in these workshops are forced to recognize that all components of natural resource systems

are not of equal importance and judgment is the tool of management not exhaustive research.

MODELLING WORKSHOPS: THE CORE OF AEAM

A modelling workshop is a 3 to 5 day meeting of a group of scientists, planners, and managers involved in the design and execution of an environmental study. No papers are presented, there is no keynote speaker, there are simply 3 to 5 days of focussed activity on the problems at hand. The development of a computer simulation model of the physical, biological, and social aspects of the problem serves as the focus for the workshop.

Participants in the workshop do not need any knowledge of computers or modelling to contribute to the workshop and to gain from its results. Workshop facilitators translate participant input into quantitative relationships that can be programmed into the simulation model. The facilitators can be viewed as information translators for it is the participants who conceptualize the model. Therefore the model that evolves from a workshop is as much a product of the ideas and concerns of people unfamiliar with modelling as it is a product of those familiar with simulation techniques.

The obvious objective of a modelling workshop is to build and run a computer simulation model of the bio-physical system of interest. However, the resultant model is not an end in itself. Usually its predictions are not very precise and it often lacks obvious features of the actual system. Rather, the model is a focus for communication promoting objectivity and honesty. Building the model forces the participants to formalize their understanding of the system components and interactions. This facilitates easier evaluation of the importance of interactions to the system and the workshop objectives. Often favoured factors turn out to be irrelevant for predictions, therefore requiring less future effort both in model development and data-gathering programs.

As with many modelling studies, a workshop generated model confers the ability to test hypotheses, research plans and different management policies without risk. However, that is where the similarity usually ends. Since a workshop model is designed and built by all the participants its structure and resultant dynamics are "transparent" to the user. The model is comprehensible. This inspires trust and increases insight thereby promoting generalization of the model projections. It also facilitates easier evaluation of those factors left out of the model.

The modelling workshop style prevents the building of a sophisticated state of the art simulation model. The workshop model is invariably simple in structure and inefficient in operation. Further by striving to simplify the problem the need for sophisticated tools such as complex implicit (or explicit) finite difference methods for approximating differential equations are usually avoided.

Although the precision of the model's predictions may suffer simplification does not deter from the objectives of participant involvement, communication and understanding. People are the key components in the modelling workshop not the model. This does not have to be the case after the workshop. Often the model structure established serves as an excellent guide for future modelling efforts when more consideration can be given to the "art" of simulation modelling.

By definition any simulation model is "wrong" since it must be a simplification of reality. However, complex models do not necessarily make better predictions. As the number of variables increases, so does the number of assumptions about how they are related and the chances of making a critically wrong assumption rises rapidly. Therefore parsimony is the underlying workshop modelling theme. Interactions and relations should make sense when interpreted in terms of physics and biology. Logical consistency and clarity are stressed in the building of the workshop model and go far in maintaining its "transparency" to the participants.

The incorporation of modelling workshops in the AEAM process is designed to be iterative. By locating them between sequential field programs the researchers and managers have the opportunity to adapt research plans and management policies in light of new insights emerging from the modelling workshop exercise. The AEAM process, and specifically modelling workshops, can be successfully implemented at any point in a study, right up to the end. The integration and coordination aspects of modelling workshops provide a useful vehicle for communication among those responsible for preparing environmental overviews and assessments. This ensures they are pertinent, credible, and address the questions being asked. Further the resultant model (after some refinement) provides a very effective device for summarizing and presenting the results of a study to policy makers, administrators and/or funding agencies. A workshop held in 1979 on the Alberta Oil Sands, is a good example of such an application.

ASSESSMENT OF THE ENVIRONMENTAL EFFECTS OF ALBERTA OIL SANDS DEVELOPMENT

Setting

"The development of the Athabasca Deposit, one of several oil sands deposits in Alberta, has been the subject of intense interest for several decades. The Athabasca Deposit contains more than 600×10^9 barrels of bitumen reserves, and constitutes about 88 percent of known oil sands in Alberta. Consequently its potential to augment the oil supply of Canada has been a driving force in present development, and will continue to generate pressure for further development...

...The Government of Alberta has an established policy of environmental legislation which allows for the orderly development of resources with

a minimum of environmental damage. This policy resulted in particular attention being paid by regulatory agencies to the need for development of an environmental research program for the Athabasca Oil Sands region. Consequently, late in 1973, officials of Alberta Environment (Research Secretariat) and Environment Canada (Environmental Management Service) separately produced internal reports recommending a comprehensive environmental research program. Projections in each of the reports favoured an environmental research program lasting 10 years, with total costs estimated in the range of \$30 million to \$40 million." (Smith 1979).

The Alberta Oil Sands Environmental Research Program (AOSERP) began in April, 1975. Early investigations were used primarily to establish large data bases. The data bases were intended to facilitate the construction of models to aid in predicting physical, chemical, biological, and social impacts of Oil Sands development.

In the fall of 1979, AOSERP sponsored an AEAM modelling workshop. By this time the program had been organized into four systems: Air, Land, Water and Human. The objectives of the workshop were: (1) to construct a simulation model that would provide a mechanism of integration of the plethora of AOSERP data and information; (2) to delineate the interrelationship between the systems (Air, Land, Water, Human) that are basic to a general understanding; (3) to identify and evaluate data gaps and uncertainties about system function; and (4) to evaluate and recommend approaches to environmental management in the oil sands areas, including mechanisms for technology transfer from AOSERP to Alberta Environment regulatory branches (Staley and others 1979). The participants in the workshop included the director of AOSERP, the heads of the Air, Water, Land and Human systems, the chairman of the Research Secretariat of Alberta Environment, as well as numerous government scientists and planners, and private consultants.

Simulation Model

During the bounding exercise, the workshop decided to consider the entire AOSERP study area which comprises approximately 2.86×10^4 square kilometers of northeastern Alberta, Canada with the spatial resolution based on the area's 13 water drainage units. To assess impacts over a meaningful period, a thirty year time horizon with a yearly time step was used. Important system phenomena operating on a shorter time scale were represented implicitly within the one year step. The simulation model was divided into four inter-related submodels: human, physical transport, aquatic biology, and terrestrial.

The human submodel consisted of three components:

1. An industrial component that generated a number of different development scenarios.

2. A population component that estimated the population based on background growth and development related growth scenarios.

3. A land component that calculated the land requirements for urban, industrial and transportation needs.

The transport submodel was concerned with the physical transport of water, air, and associated pollutants throughout the AOSERP area. The water component was a simple hydrological model of the flow within the 13 water drainage units. Relevant water pollutants were selected as water quality indicators and their concentrations were calculated for each water drainage unit.

The aquatic biology submodel was concerned with the impacts of commercial and recreational fishermen, instream flows, and instream pollutants on four common fish species.

The terrestrial submodel was responsible for representing the vegetation and wildlife dynamics. The model concentrated on the economically important species moose and beaver and their habitat.

RESULTS

A detailed description of the simulation model and its results are found in Staley et al. (1979). However, the results of applying the AEAM process are far more important than the output of the model. Each submodel made explicit a number of data and information gaps. At the conclusion of the workshop it was apparent that many of the data gaps revealed by the simulation modelling exercise could be filled by appropriate reorganization of the AOSERP data base. However, it also became clear that a further conceptual understanding of the environmental system under study was needed before a detailed analysis of the effects of oil sands development could be made. This has important implications for future research since the data baseline has been established. Further research should concentrate on understanding the dynamics of the system. In other words the focus needs to be on those things that cause variation and change in the system, and not on the current state of the system.

The simulation model itself provided a vehicle for integration of five years of environmental research. The process of building the model in the workshop revealed a number of important relationships between the (Air, Water, Land, Human) systems that are basic to an overall understanding.

The application of the AEAM process to AOSERP is not complete. The next phase which should be completed in the fall of 1980 will concentrate on developing the model as a tool to aid environmental management in the oil sands area. The many conceptual weaknesses and bad data in the simulation model will be remedied through a series

of technical meetings to be held with the staff of each of the Human, Land, Water and Air systems. The results of these meetings will guide model refinement. The final model will be put together in an integration workshop where the participants of the original workshop and the technical meetings will interactively game with the model. Once the model goes through this "trial by fire" before its creators the model will be used to evaluate a number of environmental management strategies. This will be done in an one day policy workshop focussed on the evaluation of the model's projections. It is the results of this latter stage that will measure the degree of success of this application of AEAM.

CONCLUSIONS

The AEAM process represents the combined learning of a number of international scientists and practitioners and is in a state of "dynamic equilibrium", continually adapting in light of new experience. But the term "adaptive" stresses a more important lesson, the need for research and management to be open to change and to be adapt in both style and content when new information becomes available. While AEAM itself is in a continual state of change, its two underlying themes, expect the unexpected, and learn to plan and plan to learn, never change.

Utilizing the modelling workshop provides many benefits to the success of the AEAM exercise. It forces participants to focus on the relevant issues, promotes interdisciplinary communication, identifies information needs, provides a framework for evaluation of existing information and management actions, and is a guide for environmental policy design.

AEAM through its use of simulation in the workshop setting provides a mechanism for integrating information and facilitating the analysis of impacts rather than massaging baseline data. It is through thoughtful synthesis, analysis and effective communication of environmental information that environmental managers and researchers will make effective use of their resources: time, money and expertise.

LITERATURE CITED

- Holling, C.S. (editor). 1978. Adaptive Environmental Assessment and Management. 377 p. Wiley Inter-Science, Chichester.
- Smith, S.B. (editor). 1979. Alberta Oil Sands Environmental Research Program Interim Report covering period April 1975 to November 1978. Prepared by A.S. Mann, R.H. Hursey, R.T. Seidner, and B. Kasinska-Banas. Edmonton, Alberta.
- Staley, M.J., and others. 1979. Report of a simulation modelling workshop on the environmental effects of Alberta Oil Sands development. 73 pp.