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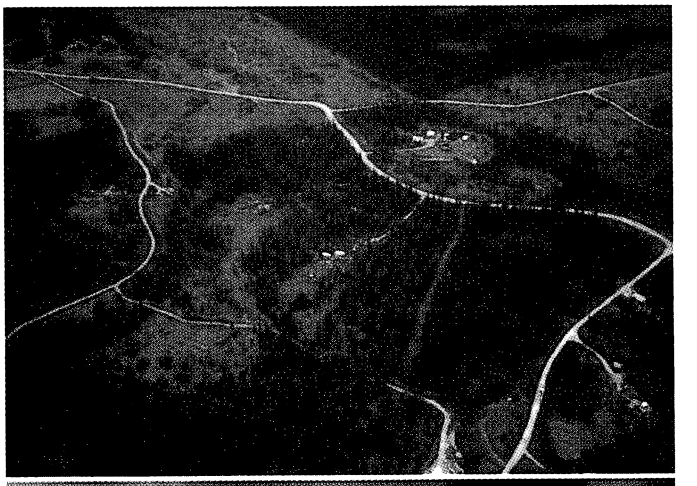

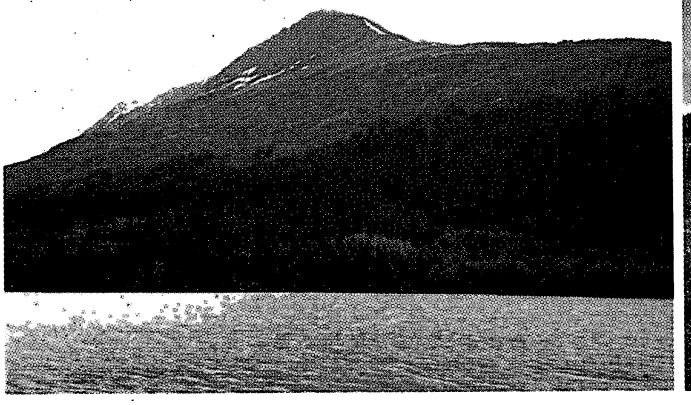

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The remaining three—Susan G. Conard

North Central Research Station
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1992 Folwell Avenue
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Disturbance in Boreal Forest Ecosystems: Human Impacts and Natural Processes

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Duluth, Minnesota, USA

Edited by
Susan G. Conard

PREFACE

Establishment of the International Boreal Forest Research Association (IBFRA) was proposed in 1990 at the International Conference on Northern Forests in Arkhangelsk, Russia. The first organizing committee meeting was held in June, 1991, in the Ukraine, and by September, 1993, IBFRA membership included all major boreal forest countries: Russia, Canada, the United States of America, Finland, Norway, and Sweden. The current president of IBFRA is Anatoly I. Pisarenko of Russia.

The major purposes of IBFRA are:

- To support sustainable development of boreal forests by promoting basic and applied research and other activities relating to forest ecosystems, forestry, and forest industry;
- To promote and coordinate research to increase understanding of the role of circumpolar boreal forest in the global environment and the effects of environmental change on that role;
- To promote the dissemination and transfer of research results and new technology and the adoption of new methods of forest use, planning, and management, with a view to ensure conservation and sustainable development of boreal forests.

The primary formal activity of IBFRA is to promote exchange of research and management information through international scientific conferences. The first IBFRA conference held in Anchorage, Alaska, USA, in 1992 focussed on "Managing Forests to Meet People's Needs". Since that time, annual conferences have brought together hundreds of scientists and managers interested in the problems of forest management in the boreal zone. Conferences have rotated among the major boreal forest countries and topics have included: "Classification, inventory and monitoring of boreal forests" (1993; Norway, Sweden); "Boreal forests and global change" (1994; Canada); "Climate change, biodiversity, and boreal forest ecosystems" (1995; Finland); and "Sustainable development of boreal forests" (1996; Russia). In response to growing recognition of the importance of understanding disturbance processes, the 1997 conference reported in this volume focussed on "Disturbance in boreal forest ecosystems: human impacts and natural processes". Over 100 researchers and managers from eight countries attended the conference. The papers in this volume represent material that was presented at the meeting as either oral talks or poster presentations. The major conference topics were Forest Dynamics and Structure, The Role of Disturbance, and Boreal Forest Management.

Many people contributed to the success of this conference. William Sommers (USDA FS, Washington, DC (WO)) was the overall meeting chairperson. Members of the program committee were: John Brissette, Susan Conard, Alan Ek, Eric Kasischke, Michael Lalich, Joel Levine, Roberta Balstad Miller, Jon Ranson, Charles Slaughter, Keith van Cleve, and John Zasada. Anatoly Shvidenko organized and chaired a panel session on International Cooperation (not reported on here). Elizabeth Schmucker's efforts in meeting planning and coordination were extensive and invaluable. Meeting arrangements were handled by the USDA FS North Central Research Station (NCRS) (Linda Donoghue, Mike Prouty, Diane Veilleux), with support from the Superior National Forest (Duane Hanson, Mark van Every, and James Sanders), the Chippewa National Forest (Steve Eubanks), the Northeastern Research Station (Andrew Gillespie), the Minnesota Department of National Resources (Jerry Rose and Joyce Nyhus), the University of Minnesota (Alan Ek), and the Natural Resources Research Institute (Michael Lalich).

Production of this volume would not have been possible without the able assistance of Mary Peterson (NCRS) in developing the cover and finalizing the overall format and structure of the proceedings. Bill Bennett (WO) converted the manuscripts to two-column format, inserted tables and figures, and typed in revisions; Sharon Parker (WO) attended to many of the details involved in final formatting of papers. Papers were edited as necessary for format and to enhance clarity. The overall quality and consistency of the proceedings has been improved vastly as a result of the cooperation of the authors in this process, but the final responsibility for any errors that may have resulted from editing and reformatting is mine.

Readers will find that this volume contains many excellent papers and a great deal of new information on disturbance and its management in the boreal zone. The high participation of Russian authors provides access to a number of important papers previously referenced only in the Russian literature. Many of the personal contacts made in Duluth continue to bear fruit in growing international research cooperation on boreal forest issues.

Susan G. Conard, editor
Washington, DC
20 March, 2000

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CHALLENGES FOR THE BOREAL FOREST ZONE AND IBFRA

Sten Nilsson[1] [2]

ABSTRACT.--This paper gives a simple overview of the boreal forest sector, and addresses a few of the challenges confronting the boreal zone. These challenges are: how to deal with sustainable boreal forest management, the structural changes in the boreal industrial sector due to globalization, and the imbalance between political commitments and real actions with respect to the boreal forest sector.

BACKGROUND

Boreal forests are increasingly becoming a high-profile topic on the international political agenda. Unfortunately, there are limited possibilities to discuss all of the problems associated with the boreal forests in one condensed presentation. I also do not feel that this would be a task for me to accomplish. Instead I will try to highlight a few issues which I regard crucial with respect to the future development of the boreal forests. I am especially aiming at issues that are not on the boreal forest agenda today. But before addressing these issues, I will present a very aggregated summary of some features of the boreal forests.

SOCIO-ECONOMIC IMPORTANCE OF THE BOREAL FORESTS

Currently, boreal forests constitute roughly one-third (1.4 billion hectares – depending on what is included in the definition of boreal forests) of the global forested areas (Nilsson 1996a). Forests have historically played a vital role in the development of the economies and societies in the boreal zone. Settlements were only possible through the support of wood resources. Wood was used for buildings, heating, cooking, tools and transportation. Mining, ship-building, charcoal production and tar extraction were major consumers until the end of the 19th century. The evolving timber industry during the 20th century resulted in the rapid exploitation of the boreal forests. Today, the boreal countries are economically dependent on their forest sectors (table 1).

The forest sector's importance to the national economy in these countries can also be approached by assessing the role of the so-called forest cluster, which includes other industries and services that are directly connected to the forest sector. An example of a forest cluster is illustrated in figure 1.

Table 1.--*Economic importance of the forest sector in boreal countries.*

Country	Percentage of Gross Export Earnings	Percentage of Direct Contribution to GDP
Canada	13	4
Nordic	9-34	4-7
Russia	5	3

[1] *Professor Sten Nilsson, Leader, Sustainable Boreal Forest Resources Project, International Institute for Applied Systems Analysis (IIASA), A-2361 Laxenburg, Austria.*

[2] *Acknowledgements: I am extremely grateful for the help I have received from Drs. Tim Williamson and René Samson, Canadian Forest Service, Dr. Sven Svensson, The Swedish National Board of Forestry, and Prof. Matti Palo, Finnish Forest Research Institute, for supplying some of the basic data. In addition, I have also used a report by Zasada et al., (1997) in writing this paper and would like to thank the respective authors for the use of their work.*

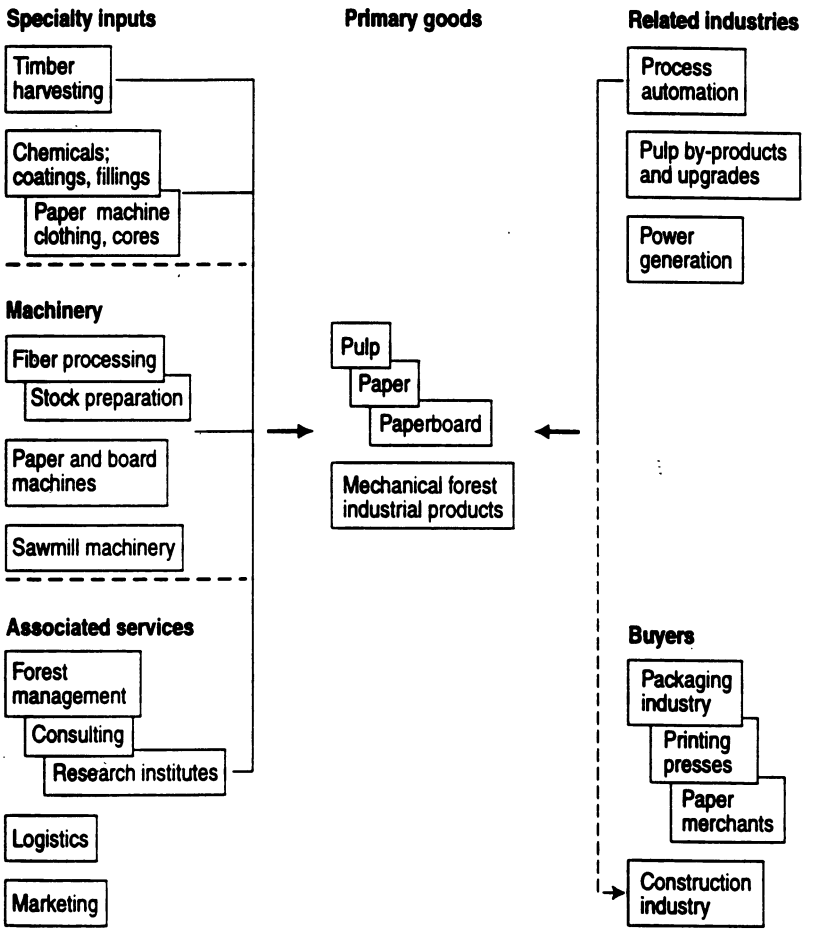

Figure 1.--*Example of a forest cluster. After Hernesniemi* et al. *(1996).*

With this approach, the forest sector's contribution to GDP is estimated to be 10%-15% in Canada. The forest cluster share gross export earnings is 41% in Finland, compared with 34% as a direct impact by the forest sector; and 24% in Sweden, compared to 17% as direct impact (Lammi 1994). The important factor is that net export earnings are some 80% of gross export earnings as most input factors are produced domestically (Swedish Pulp and Paper Association 1996). These estimates are based on the forest products' values and do not consider non-wood functions. Hultkrantz (1992) and Solberg and Svensrud (1992) have estimated the contribution of non-wood functions to GDP to be 35%-75% of the value of industrial products in the Nordic countries. Costanza *et al.* (1997) have tried to estimate the value of ecosystem services and functions (17 different functions) and have developed an aggregated annual ecosystem services value of US \$302 $ha^{-1}yr^{-1}$ (1994 US$) for the boreal forests. This value can be seriously debated, since only oceans and grasslands have lower values of the biomes analyzed.

The economic importance, as a proportion of overall economic activity, of the boreal countries' forest sectors has declined since the post-war period. Despite this decline, boreal forests continue to play a substantial role in the economies of these countries. There are very few, or for that matter, any other, sectors in the boreal countries that can provide a similar contribution to net export earnings as the forest sector.

An outlook on the future consumption and production of industrial roundwood based on conventional estimates is shown in table 2. These conventional estimates indicate a stable increase in consumption of industrial wood. The demand will be difficult to meet without increased forest production, utilization and higher prices on forest products. Boreal countries supplied some 40% of the industrial coniferous supply in 1992. According to these conventional scenarios, the output of coniferous industrial wood from boreal forests will have to increase by some 140 million m^3-390 million m^3 within the next 25-year period. However, the share of the world's industrial coniferous supply has also been estimated to remain constant in 2010 and 2020 (Nilsson 1996b).

The amount of timber resources is not a limiting factor for the supply from the boreal forests estimated above. It should be pointed out that, even if the calculations identifying shortages in the global wood balance are accurate, realistically these shortages will never appear. A number of balancing measures will occur on the market to achieve an equilibrium including decreases in demand due to increased prices, introduction of new and more fiber-efficient technology, substitution of products, increased supply, etc. (McNutt 1995).

The development of the energy market will influence future utilization of forest resources. The contribution of forests in the future to electricity, heat, and gaseous and liquid fuels will depend on markets and incentives, on Research and Development progress, and on environmental requirements. Today, the costs for biomass energy production are compatible with the cost of several traditional fuels. There is a large potential estimated, and strong political pressure (e.g. Sweden), for wood for energy in the future. Forest sector managers in the boreal countries have the necessary knowledge to intensify utilization of forest resources for energy production.

There are more than one million Aboriginal people living in the boreal zone.

The cultures and economies of these people are intricately adapted to their natural environment and depend upon it for self-perpetuation. The *lebensraum* (living space) of Aboriginal people will probably be degraded with increased utilization of the boreal forests unless special actions are taken in the future.

ECOLOGICAL IMPORTANCE OF THE BOREAL FORESTS

The boreal forests are finite whose productivity is restricted by the boundaries set by their physical environment and the ability of biota to survive and grow within that environment. Silvicultural activities can enhance productivity within the limits of biological constraints and economic realities. Understanding the biotic and abiotic characteristics of the present boreal forests and their dynamics under the current environmental conditions is necessary to determine the potential for biological productivity and resource management. This information is also critical for predicting future paradigms that will result as environmental, socio-economic, and political conditions evolve. In the following, I will briefly discuss the physical and biological factors determining the potential for sustainable development of the boreal forests.

Climatic Attributes

All climatic attributes combined serve to distinguish boreal forests from the more temperate areas, but the temperature control over above and below-ground biological processes and the characteristics of the snowpack are particularly important for this distinction. The below-ground temperature affects all aspects of soil biology and nutrient cycling. Forest floor conditions, overstory and understory density, and soil temperature are closely related. The amount of heat available for growth during the growing season, as well as frost events, greatly influence phenology and the development of vegetative and reproductive growth. More than 50% of the annual precipitation occurs during the growing seasons.

Table 2.--*Global industrial wood balance, expressed in million m^3, (Nilsson, 1996b).*

	1993		2010		2020		
	Coniferous	Non-coniferous	Coniferous	Non-coniferous	Coniferous	Non-coniferous	Total
Demand	939	540	1,210-1,680	730-990	1,400-1,540	850-1,010	2,250-2,550
Supply	939	540	950-1,210	545-745	1,080-1,330	585-800	1,665-2,130
Balance	--	--	0 - -470	+15 - -445	-70 - -460	-50 - -425	-120 - -885

Land and Water Base

Under certain conditions of evaporation an accumulation of organic matter will take place, which drives the development of organic soils. Snow modifies local climate through its high albedo and high thermal conductivity. The boreal forests are mosaic of upland forests and wetlands with lakes and rivers interspersed. The current mosaic of floristic and forest stand patterns are maintained by the combined effects of site conditions, disturbance factors and climatic regime of the region. This mosaic is an extremely important landscape element in terms of biodiversity.

Soils and Permafrost

Features commonly referred to in a discussion of boreal forest soils are low temperatures with the formation of permafrost as the maximum expression of this characteristic, poor drainage, thick organic layers with deep organic soils in wetlands, low soil biological activity and low nutrient availability. Due to these features productivity is often limited.

Forest Composition and Distribution

Although the boreal forest is often viewed as one large cold-dominated area occupying the northern part of the globe, this simplistic view is far from realistic. Rowe (1992) identifies 45 sections of the Canadian boreal forest based on climate, physiography and tree species composition. At the scale of the stand there can be large stand-to-stand variations in species' composition and structure. The distribution of species within the landscape varies within the boreal region even if the boreal forests have relatively few tree species (some 15 dominating species in North America and some 35 in Siberia). Species composition, species diversity, and productivity of shrubs and herbaceous plants also vary in the boreal region. Shrubs are key elements for wildlife habitat, providing both food and cover. Nonvascular plants, such as mosses and liverworts, are ecologically important boreal plants and often contribute more to plant species diversity in a stand than do vascular plants.

Biodiversity

Biological diversity may be defined as the number, variety and variability of living organisms on the earth (WCED 1992), including the sum of diversities found at the genetic, species, ecosystem, and landscape levels (Wilson 1988). It is estimated that the boreal forests contain over 100,000 species, 95% of which are arthropods and microorganisms. Viruses are particularly abundant with over 40,000 species. Currently, only some 20% of the taxa in the boreal forests have been identified taxonomically.

The dominant tree species have varied from one interglacial period to another, and many species have gone extinct during the Pleistocene climatic fluctuations. There is also growing evidence that the current composition of plant associations is contingent upon historical accidents.

Forest Succession

As discussed earlier, boreal forests are described as a mosaic with relatively few tree species arranged in pure and mixed stands of various sizes and shapes owing to site and soil conditions, distribution of lakes and rivers, species characteristics, and disturbance history (e.g., Suffling 1993). Within any landscape, there are both long- and short-term scale processes that shape current vegetation and influence potential future vegetation. Here I will concentrate on the short-term factors (years to several centuries). The primary natural short-term disturbances that influence landscape patterns in boreal forests are fire and insects. Wind and snow breakage and diseases can also be important, but tend to follow the first two mentioned factors. Another group of disturbances is linked to flooding.

Table 3.--*Global Change of the Environment-Applied to the Boreal Forests (adopted from the World in Transition: Basic Structure of the Global Man-Environment Relationship, German Advisory Council on Global Change, 1993)*

Atmosphere	**Climate Change**	**Hydrosphere**	**Lithosphere and Pedosphere**	**Biosphere**	**Population**	**Commerce and Traffic**	**Psycho-Social Impact Factors**
Depositions in relation to carrying capacity in different ecosystems	Current and possible Fixation of greenhouse gases by the ecosystems under different management regimes Identification of regions and ecosys-tems which are sensitive to climate change	Depletion and pollution of fresh water Impact on the water cycle by deforestation	Identification of sustainable land-use Pollution of soil	Ecologically sustainable use of different forest types Identification of sustainable structure of landscapes Management of ecosystems and landscapes for a sustainable biodiversity The role of northern forests as a climatic stabilization factor	Sustainable livelihood for people (with emphasis on indigenous people)	Identification of structural changes in the economy for reduction of pressure on the Environment Identification of sound environmental policy regimes (e.g. property rights, subsidies) Economic analysis taking environmental aspects into account	Intergenerational policies Assessment of social structures and processes that positively influence the environment Improvement of culture specific strategies for promotion of the environment Forest sector impact on regional development

Forest Productivity

Both the primary and secondary productivity are important, but I will concentrate on the primary productivity in this discussion. It is important to take into account the tangible values of the primary productivity such as water protection, water supply, soil protection and climatic roles.

Forest biomass and annual bioproductivity of the boreal forests are of particular importance to the boreal forests in the global carbon budget (Kurz *et al.* 1993, Shvidenko *et al.* 1996). Total biomass, distribution of biomass and productivity differs depending on site conditions, species composition and stand age. For example, on cold sites the moss component may have levels of annual productivity, equal to or greater than, that of the overstory woody species.

Man-Made Disturbances

The conversion of forests to agriculture, community development, mining, oil and gas exploitation and forest management have historically affected the boreal forests and will continue to do so in the future. The conversion of boreal forests to agricultural land has: decreased the area of forest, especially on rich soils; fragmented the forest resulting in remnant forest patches and increased edge effects; and changed the forest stand community structure by introducing new species.

The forest management activities that have most influenced the boreal forests as a whole are forest harvesting and forest fire management. Both of these activities change forest dynamics at all scales of resolution, from altering the landscape structure to microsite characteristics and have introduced externally operated dynamics versus natural dynamics.

Sustainability

The rational for sustainability of the boreal forests has both a pragmatic and a more idealistic dimension. The pragmatic reason is that the boreal forests are an important source of employment and economic development, as well as a source of food, fiber, water and forest products upon which people depend. The more idealistic and long-term dimension is of an ecological nature and includes the role boreal forests play in the greenhouse gas budget, maintenance of biodiversity, wilderness, wildlife habitat and spiritual values. The idealistic dimension is difficult to substantiate to the satisfaction of the general public, politicians and policy-makers. Often there is a direct conflict between the ecological issues and the pragmatic issues.

In order to achieve sustainability, all of the ecological factors contributing to sustainability discussed above must be taken into account in a satisfactory manner in any future management system.

GLOBAL CHANGE AND THE BOREAL FORESTS

An array of global changes will influence the potential responses of the boreal forests and the ways the boreal forests will be used and managed in the future. Many people instantly think of biophysical processes when global change is the subject of interest, e.g., climate change. These processes are indeed central to examining the boreal forests, but of parallel significance today, and in the future, will be a set of human-oriented forces or socioeconomic forces (Forester *et al.* 1994). The German Advisory Council on Global Change (1993) has stressed the width of the global change concept. This concept is illustrated in table 3 in an application revised for the boreal forests. It can be seen from this table that Global Change is much more than merely climate change.

The meta-drivers of global change are human population and human behavior. There has been, and continues to be, a serious concern regarding an exploding population development. But the most recent results (Lutz *et al.* 1997) indicate that it is unlikely that the population will be doubled by the year 2050, as earlier expected. The median estimate is a total global population of 10 billion by the year 2050, compared to 5.7 billion in 1995 (Lutz *et al.* 1997). Population growth will probably be most rapid in the Middle East, sub-Saharan Africa and North Africa, with the population tripling by 2050 in these regions. There will also be significant increases in the proportion of the population above 60 years of age. Western Europe, Eastern Europe and Russia are likely to experience little, if any (most likely negative in Eastern Europe and Russia), increases in the population. But in Western Europe and Pacific OECD, the proportion above 60 years of age will probably double compared to the situation today (it could be as high as 40% of the total population). In North America, a population increase of 25% seems likely with a substantial aging of the population, but not as dramatic as in Europe. The population in Latin America is likely to double with the proportion of the population above 60 years of age increasing by 20%.

Central Asia is estimated to grow by 37% and have an increase in the 60 and over age group from 9% to 25%. Southern Asia (mainly India) will double its population by the year 2050 (Lutz *et al.* 1997).

This increase in population is still dramatic and the population development in the regions outside the boreal region will cause transboundary effects impacting the boreal forests. But the most dramatic message in this new information is the high proportion of aging people in the global population. Therefore, I expect much of the demographic research in the future will address problems connected with the aging population instead of the population growth as of today.

Most important for boreal forests in the future will be human behavior and how it affects the global environment – economic behavior (e.g., consumption patterns), social behavior (e.g., attitudes), and political behavior (e.g., cooperation or conflict). Human behavior will probably change dramatically over time, especially with the shift in age distribution illustrated above, we do not know what direction these changes will take.

NON-STATEMENTS

Until now, I have taken a rather conventional approach to my presentation. In the following, I will address a few issues that normally are not on the agenda for the future of boreal forests. To give ammunition to this convention (the IBFRA-meeting), I could state that climate change is not a major threat to the boreal forests. The continuing, steady decarbonization of energy systems means that concentrations of carbon dioxide will not rise above 500 ppm. It has been stated that a climate warming over the next 75 years "Will not significantly effect diet, health, income or environment. Most of the economy has moved indoors and much that has not will do so. Climate simply matters less and less (Ausubel 1996)." Or I could state that the assumed climate change is only good for the boreal forests. Perez-Garcia *et al.* (1996) estimate that the boreal zone will have a substantially increased net primary production (+15%-25%), an increased output of forest industrial products (+13%-20%), increased export, and improved economics due to forecasted climate change. Mendelsohn *et al.* (1997) claim that the expected climate change will have positive market implications for the boreal region in the range of 15-29 billion US$ per year. There are indications that we have already started to move on this path with a significant increase in plant growth in the northern high latitudes during 1981 and 1991 (Myneni 1997). In this situation, I could argue that instead of worrying about protecting existing production and forest resources we should worry about how to take care of this new production in the boreal zone. I could argue that the loss of biodiversity is an issue more of the past than the future. Ausubel (1996) reminds us that the Americans cleared forests not only for food production but also to keep down the snakes, chiggers and ticks. A similar pattern occurred in Europe and Russia with clearing of forests for health, safety, and mental security. Mankind has passed this stage by now and the snakes and bugs are back in the forests. But I do not intend to make these statements. Instead I will concentrate on a few other issues.

IS SUSTAINABLE BOREAL FOREST MANAGEMENT POSSIBLE?

After reading some 50,000 titles indicated in library indexes and through the internet using the keyword "sustainable forest management," I should of course be wise enough not to touch upon this topic at all, but I am not that bright.

Quite recently, Rice *et al.* (1997) raised the question of "Can Sustainable Management Save Tropical Forests?" They came to the conclusion that this is probably not the case. The conclusions were that current logging practices cause considerably less damage than some forms of sustainable management and the authors state, "substainability is, in fact, a poor guide to the environmental harm caused by timber operations." I will deviate Rice *et al.*'s (1997) original question to "Is Sustainable Boreal Forest Management Possible?".

A tremendous effort has been put into development of criteria and indicators for temperate and boreal forests through the so-called Montreal and Helsinki processes. Criteria is defined as: The essential components of forest management against which sustainability may be assessed. Each criterion relates to a key element and can be characterized by one or more quantitative, qualitative, or descriptive indicators. The development of criteria and indicators was identified as being one of the major tasks for the Intergovernmental Panel on Forests. The basic idea is that by monitoring these indicators the degree of sustainability can be measured. FAO (1997a) states that it seems plausible to arrive at a globally-agreed upon core set of national-level criteria and indicators. A further step in this direction is *certification.* Roughly defined, certification is the process of establishing whether or not a particular forest area is

achieving objectives based upon criteria and indicators of sustainable forest management.

The Helsinki Process includes 6 criteria and 27 quantitative indicators (of which 6 are difficult to implement; FAO, 1997a) and Montreal Process contains 7 criteria and 67 indicators (of which only 9 can be measured in e.g., the USA; FAO, 1997a). Russia has developed its own preliminary set of 6 criterion and 35 indicators (Strakhov 1997). This entire development is excellent and shows a way to move towards sustainable forest management. But will it happen?

First, it is certainly necessary that the signatories of the agreements monitor the indicators they have agreed upon. Second, currently the agreements are non-binding national indicators. There is a strong need to demonstrate and quantify what does it mean for the sustainable development at a sub-regional national level in concrete form by implementing these indicators – usage of just national indicators will only mask the problems. To my knowledge, no one has done this so far. Even if these two hurdles are overcome I still think there exist great difficulties with the entire concept.

I will use Russia as an illustration. I have selected one indicator from each criterion in my illustration and you should remember that the agreements deal with 27-67 indicators. By merely overlaying my 6 selected indicators for Russia one can see that the indicators point in different directions[3]. The indicators of the agreements are selected in a manner where there will always be conflict between the same. The questions one may ask in this situation are: what is acceptable sustainability, what is good sustainability, and what is bad sustainability? Who is going to decide this? In the end it will, of course, be a political decision at some level and at that stage a whole array of important socioeconomic factors or indicators (which are not included in the current indicators) will be the driving forces for the decision-making process.

The problems with the sustainability concept, and by that the criteria and indicator approach, have been illustrated in an excellent way by Heilig (1997). I will use his paper as a platform for my further discussion.

The concept ignores the fundamental diversity of interests. One of the major characteristics of our world is that people fundamentally disagree about objectives, values and lifestyles. *What is environmentally sound may not be acceptable for our social structure, our economy or our culture.* A most serious obstacle to a universal concept is the fact that human life has to deal with more than one dimension. Stability of our ecosystems is only one of many concerns.

The concept is based on a social concept of harmony and ignores the fact that human development is driven by conflicts and fierce competition. The whole evolution of the human species indicates that we are part of the ecosystems (Behan 1997), but we are not just happy being part of a sustainable ecosystem. We want to dominate and grow. Through all kinds of interventions, we try to shape our environment for our benefit – even if at the cost of other species and our human neighbor. By nature, we are competitive animals.

It can be demonstrated that a strategy of simple-minded cooperation will always lose in an environment where the other players can gain a slight advantage by not cooperating (Nowak *et al.* 1995). Human interaction is based on bargaining, open threats, economic pressure, coercion, or a strategy of limited retaliation against non-cooperation. This diplomacy of violence (Schelling 1966) is one of the fundamental principles of all social, economic and political development, which is neglected in the current concept. This is especially crucial at a time when the world is swept by, international, global and neo-liberal economies.

The concept reduces the analysis of social, economic, cultural and political processes to a simplistic Bio-Physical Framework. It has been demonstrated that the most important social processes are not at all similar to those in ecosystems (Etzioni 1968).

Thus, I do not think that we can achieve sustainable forest development in the boreal zone as thought, due to the fact that the current concept *does not address the fundamental social problem of power imbalance between societies and social groups. Furthermore, it does not identify which social, economic and political structures and processes a society should use in order to promote sustainable development.* This has been illustrated for the tropical forests by Kamari (1998) and Rice *et al.* (1997) and more generally by Jaeger (1995), we have the same situation in the boreal zone.

What counts is what the local society wants to happen in and for their forests. The moment that occurs is when we may have a sustainable boreal forest (Duinker 1997). The important task is to find the mechanism for this consensus-seeking by the

[3] *Illustrations are not presented in this paper.*

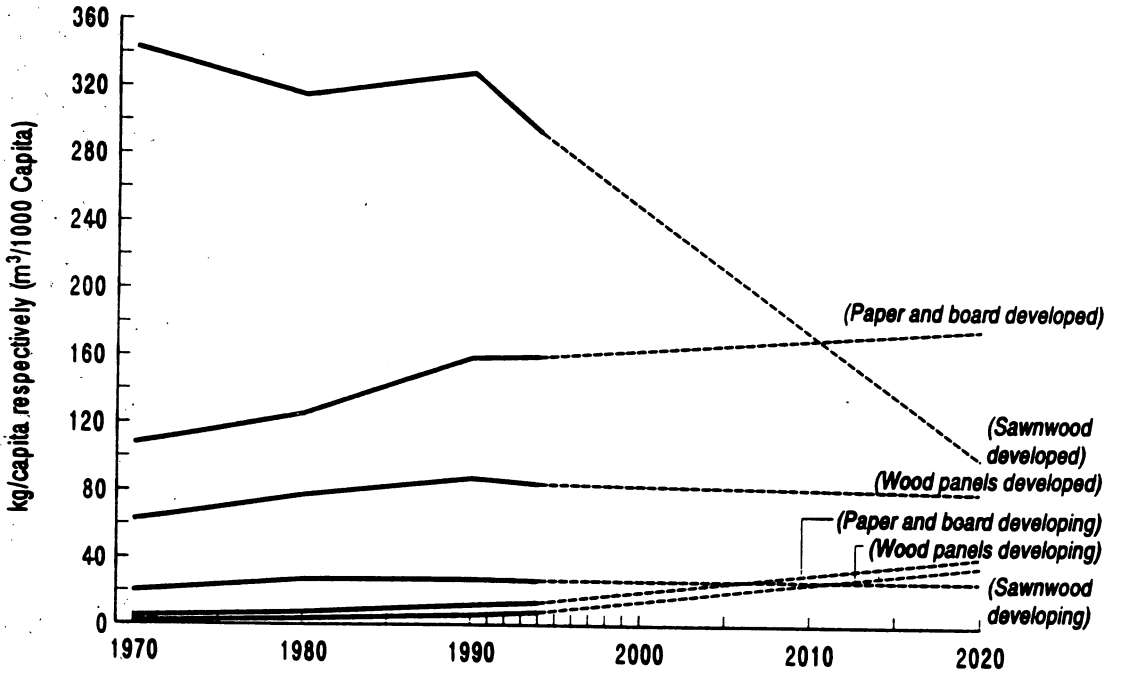

Figure 2.--*Consumption of industrial wood products expressed in kg/capita (paper and board) and m^3/1000 capita (wood panels and sawnwood).*

local societies in the boreal forests. Thus, the first recommendation to IBFRA would be:

- *To broaden its current concept with a rather basic ecological approach to include social, economic, cultural and political processes. Otherwise, the organization will not be successful in the very important work it is carrying out in order to contribute to the sustainable development and management of the boreal forests.*

IS THERE A FUTURE FOR THE TRADITIONAL BOREAL FOREST SECTOR?

As stated earlier, the world is swept by international, global and neo-liberal economies. The development is going very fast, the driving criteria are to be competitive and it is very powerful. This strongly influences the future of the boreal sector. Concerns are raised that this "competition fundamentalism" is incapable of integrating social justice, economic efficiency, environmental sustainability, political democracy and cultural diversity (Emmerij 1997). However, this is how the world is ruled today.

In order to illustrate in what direction we are going, I use FAO data (FAO, 1997a and b) (fig. 2 and table 4). In figure 2, I have used a simple prolongation of the trend breaks to illustrate my message (these trend breaks are not accounted for in earlier estimates on demand in table 2). The key message is that sawnwood consumption per capita in the developing world has decreased dramatically between 1970 and 1994. In the same countries, per capita consumption of wood panels has also started to decline. There is still an increase in consumption of paper and paperboard, however this increase is weak. The increase in per capita consumption takes place in panels and paper and paperboard in the developing countries. If we look at the expected paper and paperboard consumption changes, based on FAO

estimates (1997b), we get a development as illustrated in table 4.

What does this mean? It means that the pulp and paper industry in the boreal zone will move to areas with a rapid growth in consumption, mainly Asia and South America, to be profitable. There will still be a substantial increase in paper and paperboard consumption in the developed world, but in the future it will be less dependent on virgin fiber, as increased utilization of wastepaper will take care of most of this expansion. It means the boreal pulp and paper industry will become increasingly more global, and increasingly less dependent on boreal fiber and as a result we will not see any new mills established. This process has already started (e.g., Financial Times 1997), the result will be a handful of big companies in the boreal zone. At the same time we have a domestic sawmilling industry in the boreal zone facing a dramatic drop in per capita consumption and a stagnating aging population in traditional markets, with the non-traditional markets incapable of absorbing this difference in consumption. In addition, with a boreal pulp and paper industry going abroad and becoming more and more dependent on wastepaper, it will decrease prices on pulplogs and sawmill chips. Income from sawmill chips for most of the sawmilling industry constitutes a profit margin of about 15% today.

A friend of order asks of course, what does this have to do with sustainable forest management and the boreal forests? The anomaly is that the overall objective with most boreal forestry is to make money. The way forestry brings in money to the forest landowner, and different operators of the forests is through sawlog production. Therefore, the overall objective in the boreal zone has been, and still is, to produce high quality sawlogs. But the world is changing rapidly and it seems that very few will be interested in sawnwood in the future.

- If this happens, we should work with quite different objectives for the boreal forests in the future, which will result in quite different ecological and sustainability concerns (many of our current concerns may also disappear). A key issue will be who will pay for the forestry and ecological management under these new conditions. Based on the above, I would like to make two recommendations to IBFRA's future activities:

- *To include analyses of the development of the industrial boreal forest sector in the current framework. Without a strong sector I see limited possibilities to achieve sustainable development in the boreal forests, and*

- *To expand analyses to include links to, and impacts on, the sustainable development (both from a socioeconomic and environmental point of view) of the boreal forests with the development of the resources and forest sector at a global level.*

Table 4.--*Estimated increase in paper and paperboard consumption and wastepaper consumption between 1994 and 2010. Expressed in million tons.*

	Paper and Paperboard	**Wastepaper**	**Relation of Wastepaper to Paper and Paperboard (in percentage)**
Africa	1.9	0.8	42
North/Central America	28.9	22.4	78
South America	4.6	2.4	52
Asia	67.2	36.2	54
Oceania	1.7	1.2	71
Europe	23.0	17.9	78
Former USSR	2.3	1.0	42
Developed	66.3	49.9	75
Developing	63.3	32.0	51

IS THERE A SUSTAINABLE FUTURE FOR THE BOREAL FORESTRY COMMUNITIES?

The future of human communities in boreal forested regions is a growing concern among politicians, social scientists, land managers, and most importantly, among community members themselves. As illustrated earlier, the world is swept by internationalization and neo-liberal economies. The results of this merging of forest companies are lean operations with substantial downsizing and huge layoffs in forestry operations. This often occurs in towns with only one major employer, namely the forest industry. In the boreal zone, many communities will not survive without a solid forest sector and vice versa, the forest sector operations can not be carried out without stable forestry communities (e.g., Lundquist *et al.* 1997; Bradshaw and Lynn 1996). Two different schools of thought with respect to natural resources and regional development exist. One is that regions having an abundance of natural resources may have an advantage over less resource-rich regions in the form of better incomes and in alternative strategies for industrialization (e.g., Gerschenkron 1962; and Bradshaw and Lynn 1996). The other school, the resource-curse theory, highlights the disadvantages that resource-dependent regions face in their attempts to achieve long-term economic growth (the so-called Dutch Disease). Examples of advocates for this second school are Spooner (1981), Auty (1993), and The Economist (1995).

In order to achieve sustainable forestry communities, three aspects have to be fulfilled (Kaufman 1990): land-use aspects, industrial and employment aspects, and the social welfare and organizational aspects.

Current analyses of forest communities (Carroll and Lee 1990) suggest that ecological and economic analyses alone are not sufficient to understand the adaptability of forest communities under stress. There must be increased cooperation among foresters, ecologists, economists and sociologists in order to address the sustainability issue of forestry communities. Muth (1990) suggests that the most productive approach to study the relationship between forest resources and sustainbility of forest communities could be that provided by structural sociology. He also stresses that the issue of sustainable forest communities are best understood in terms of a sustainable community institutional structure, rather than in terms of traditional indicators such as employment, resource supply, ecology, economics, and inhabitants.

Apedaile (1992) points out that understanding of the underlying order of the rural economic system and its place among other systems is a precondition to steering the development of the rural system. He also points out that rural-urban integration changes the system-millieu, but it does not change the underlying features of the resource-based economy, and that over time these underlying features worsen the economic performance of natural resource-based activities of the rural system. Without a solid background of the above, it will result in flawed policies for sustainable rural development. Williamson and Annamraju (1996) underline that an *adequate* of the sustainability approach is necessary to achieve forest community sustainability. We, in the forestry community, have not been very successful in tackling forest community problems in the way I have outlined above. Given the conditions we are facing in the boreal zone, i.e., a stagnating and aging population, the continual, substantial downsizing of the workforce in forest communities (Financial Times 1997), the strong structural changes expected in the boreal forest sector, and the crisis situation in the Russian forest communities (Lundquist *et al.* 1997), I strongly recommend the following to IBFRA:

- *Start a program with the objective to find the soul of the future boreal forest communities (Robson 1996). The keywords in this program should be integrated analyses, structural sociology, and community institutional structures. The program must aim to produce results for sustainable development of boreal forest communities.*

WHERE IS THE LOGIC AND WHERE ARE THE SCIENTISTS?

During the last 10-15 years and especially during the 1990s, demands on the forest sector by society and governments have increased dramatically. Societies and governments require that a new set of principles be implemented to improve social, economic and environmental benefits of the forests. These demands can be illustrated by the conclusions from the 7th American Congress:

- *The forests should sustainably provide a range of goods, services, experiences and values that contribute to community well-being, economic opportunity, social and personal satisfaction, spiritual and cultural fulfillment, and recreational enjoyment,*

- *The forests should be sustainable; support biological diversity; maintain ecological and*

evolutionary processes; and be highly productive, and

- *The forests should be managed with consideration for global implications.*

This will require tremendous resources in the form of people, money and a healthy forest industry to implement these principles and visions. But what are governments doing? They are implementing policies that totally contradict the points made above. This can be illustrated by data kindly supplied to me by Dr. René Samson, Canada, and Dr. Sven Svensson, Sweden. They stress that there are uncertainties in the data, but the trend is assumed to be correct (tables 5 and 6). When comparing the increased demands with public allocation of manpower and money to forestry, it can only be concluded that this equation is not sustainable. Therefore, a logical question is if the different political commitments made, both at national and international levels, to reach sustainable development (all aspects) in the boreal zone is just lip service. With this simple overview the conclusion points in that direction.

And where is the scientific community in this non-sustainable equation? Our usual attitude is that we are scientists and do not get involved in policies and politics. Therefore, the boreal scientific community is silent. But in reality, as stated by the Director General of UNESCO, "Science is a highly political issue and part of the political decision-making process and therefore science must make its voice heard (Nature, 1997)." Hellström and Palo (1995) point out that research and development in forestry should be regarded as one of several instruments in forest policy-setting.

It can be concluded that there is a rather low probability that those visions set in the boreal zone to achieve sustainable development can be reached with current governmental allocations. This has to be made clear to the decision-makers and the scientific community has to act as the messenger.

Table 5.--*Relative Development in Canada.*

Year	Person-year in Canadian Forest Service	Canadian Forest Service Budget (Deflated to 1990 Values)	Public (Provincial and Federal) Funding on Forest Management (Delfated to 1990 Values)
1989/90	100	100	100
1990/91	101	73	104
1991/92	100	81	96
1992/93	96	91	90
1993/94	103	94	85
1994/95	97	87	93
1995/96	89	70	?
1996/97 estimate	71	46	?

Table 6.--*Relative Development in Sweden.*

Year	Number of Service Days by the Regional Forestry Boards	Regional Forestry Boards Budget (Delfated to 1991 Values)	Maximum Amounts of State Subsidies to Forestry (Deflated to 1990 Values)
1989/90	--	--	100
1990/91	--	100	75
1991/92	100	90	48
1992/93	97	89	33
1993/94	66	72	44
1994/95	66	73	24
1995/96	75	76	33

My final recommendation is:

- *To get heavily involved in the policy debate with respect to the future of boreal forests. Your important work and results will only be implemented by an active role in the debate on future policies for sustainable development in the boreal zone.*

SUMMARY

I have attempted to illustrate, at an aggregated level, how vital the boreal forest zone is from an ecological, economic and social point of view. But I also concluded that the boreal forest zone and boreal forest communities face serious structural changes in the form of stagnating (or declining) and aging population, resulting in changed attitudes in many respects. Due to the globalization of the forest industry, the boreal zone faces a challenge with structural changes, of which we do not know the outcome. It can also be concluded that the current allocation of resources to the boreal forest zone by the public will hardly make it possible to achieve those visions set by societies and governments for a sustainable boreal forest zone.

The boreal forest zone and the scientific community in the zone face challenges that have to be approached in a non-traditional way to become successful. To summarize, I make the following recommendations to the players and scientists in the boreal forest zone and IBFRA:

- *To broaden its current concept with a rather basic ecological approach to include social, economic, cultural and political processes otherwise the organization will not be successful in the very important work it is carrying out in order to contribute to the sustainable development of the boreal forests.*

- *To include analyses of the development of the industrial boreal forest sector in the current framework. Without a strong sector I see limited possibilities to achieve sustainable development in the boreal forests.*

- *To expand analyses to include links to and impacts on the sustainable development (both from a socioeconomic and environmental point of view) of the boreal forests, with the development of the resources and forest sector at a global level.*

- *Start a program with the objective to find the soul of the future boreal forest communities (Robson, 1996). The keywords in this program should be integrated analyses, structural sociology, and community institutional structures. The program must aim to produce results for sustainable development of boreal forest communities, and*

- *To get heavily involved in the policy debate with respect to the boreal forests. Your important work and results will only be implemented by an active role in the debate on future policies for sustainable development in the boreal zone.*

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GLOBAL SIGNIFICANCE OF DISTURBANCES IN BOREAL FORESTS

Anatoly Shvidenko[1 2]

ABSTRACT.--Disturbance (D) significantly impacts the state, structure, productivity and dynamics of boreal forests. The unified hierarchical typological classification of D and set of indicators to quantify D are presented for the circumpolar boreal forests. Basic principles of a logically complete description of D are: (1) understanding that D of the forest ecosystem is simultaneously D of the (definite) landscape, and the consistent landscape approach is one reliable method for all estimates; (2) landscape regularities define disturbance regimes (DR); (3) D and DR are strongly dynamic processes, and appropriate quantification of them should be based on succession regularities; (4) the set of indicators that describe D has to be used as a subset of criteria and indicators of sustainable development/sustainable forest management. Based on the above classification and principles, available publications, data and research results, the paper presents a brief overview of: (1) quantitative descriptions of D in the boreal zone; (2) aggregated estimates of impacts of D on current productivity, species distribution and age structure; (3) estimates of the role of D in the interaction of boreal forests with global carbon budget; (4) globally significant features of predicted dynamics of D in boreal forests during the next century under expected global climate change.

INTRODUCTION: BASIC DEFINITIONS, CLASSIFICATION SCHEMES AND INDICATORS

Disturbance (D) can be defined in many different ways. Disturbance is mainly understood as a destructive impact on an ecosystem. "A disturbance is any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment" (Pickett and White 1985). "Disturbance is considered to consist of the mechanisms which limit the plant biomass by causing its partial or complete destruction" (Crime 1979). "Disturbance is the destruction of living biomass or accumulated detritus" (Reiners 1983). Other definitions (Sousa 1984; Forman and Godron 1986) point out different aspects of the phenomenon. "A disturbance is a discrete, punctuated killing, displacement, or damaging of one or more individuals (or colonies) that directly or indirectly creates an opportunity for new individuals (or colonies) to become established" (Sousa 1984). "Disturbance ... is considered as something that causes a community of ecosystem characteristic ... to exceed or drop below its common (homeostatic) range of variation" (Godron and Forman 1983). "Disturbance is an event that causes a significant change from the normal pattern in the ecological system" (Forman and Godron 1986).

We do not believe that terminological discussions are relevant in this paper. It is evident that: (1) there are no boreal forests, both managed and not managed, which would be not impacted by D, and namely D is a crucial factor of the current state, productivity and dynamics of forests; (2) any definitions (and classifications) should be dependent on the goals and objectives being considered; (3) many basic types of Ds cannot be considered as a destructive factor exclusively (Furyaev 1996), and even the term "creative destruction" (Holling 1992) could be used; (4) Ds are spatial and time dependent; (5) although intuitively D seems to be tied with human impact, natural Ds play a crucial role in the structure and function of boreal forest ecosystems; (6) the definition of Ds in forests should consider specifics of forests as a natural phenomenon, as well as the requirements and traditions of forest management. The multi-faceted influence of Ds generates the major patterns of the boreal forests, defines fragmentation and mosaic of boreal landscapes, and eventually boreal vegetation structure, distribution, stability, diversity and productivity. All of these features are comprehensively revealed in successions. Thus, a relevant evaluation system of Ds is their close connection with an interdependence on succession dynamics of forests.

[1]*Professor Anatoly Shvidenko, Senior Research Scholar, Sustainable Boreal Forest Resources Project, International Institute for Applied Systems Analysis (IIASA), A-2361 Laxenburg, Austria.*

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We define *disturbance* as any (exogenic) impact, which changes natural (evolutionary) dynamics of a forest ecosystem or a (forest) landscape. This definition, as all others, has a shortcoming as it requires both the explicit understanding of the notion and the availability of the reference base, which would quantify the *natural* (i.e., undisturbed) state and *natural* dynamics of forests. A comprehensive and formal description of the problem would require much more space than would be relevant here. We only note that the approach is often used in both the theory of forestry and forest management practice, e.g., in the classical theory of *normal* forests, or in some basic principles implemented in yield tables of *normal (fully stocked)* stands, or in the *normal (sustainable)* structure of a landscape.

D usually starts from an initially short impact at the beginning of a new succession. Evidently, the magnitude of this impact would exceed the threshold of the self-organization of a forest ecosystem. Thus, we can say that D is a purely exogenic succession. "Exogenic" is used to some extent as a relative term, e.g., Forman and Godron (1986) suggested not to consider a disturbance as any exogenic factors, which is inherent in natural conditions of sites. This recommendation has a solid ecological background, but it is probably not acceptable from a practical point of view because based on this approach, for example, we cannot consider insect outbreaks as D.

Different approaches could be used to classify Ds; causative factors (genesis of Ds), changes of forest functions or structure, severity of consequences, etc. Ds can be classified by the objects disturbed (e.g., Ds of needles (leaves), individual trees, forest ecosystems, landscapes, forest cover, etc.). They can be human-induced and natural, reversible and irreversible; stand-replacing and non-stand replacing; simple and composite; can occur in "infrequent" and "multiple disturbance" ecosystems (Reiners 1983), etc. We suggest the classification in figure 1 which is based on causative factors that are closely connected with the upper level of a relevant classification of successions (Rojkov *et al.* 1996). Such an approach is very close to the traditional understanding of D in forest management practice and seems to be applicable to all types of disturbances in boreal forests as well as to other types of natural vegetation in the boreal zone.

The set of indicators appropriate to quantify Ds should provide two interconnecting and overlapping goals: (1) to be applicable for forest management practice, and (2) to present sufficient and complete information for different ecological applications. Both are closely tied with criteria and indicators of sustainable development (Status Report 1994; Criteria 1995; Conference 1993).

There are two basic elements in evaluating Ds': a forest ecosystem and a forest landscape. It is often impossible to present a consistent description of D for one without considering the other. The main reasons for this are: (1) any description of D as a succession even for a separate ecosystem includes basic landscape peculiarities; (2) many Ds have a typical landscape extent, genesis and magnitude; (3) any D of a separate ecosystem impacts a landscape's property; (4) specifics of major Ds as well as interactions and interdependence between different types of D have an explicit landscape nature.

These interconnections make it necessary to consider *disturbance regimes* (DR), i.e., stable combinations of types of Ds and their characteristics, as well as specifics of interactions between them. DR can be defined (for a definite landscape) by both a major type of D (e.g., pyrogenic or industrial exploitable types of DR), and the type of interactions expressed in terms of synergetic, neutral, contradictory and exclusive interactions. Generally, all DRs can be combined into two types: digressive DR, i.e., the DR under which the climax state of an indigenous forest ecosystem cannot be reached, and progressive DR where the natural successions are maintained and the climax state is reached. In the framework of the digressive DR, Ds can lead to the degradation of forest ecosystems. Forest degradation could be defined as a process that describes human-induced phenomena, which lowers the current and/or future capacity of the forest to protect the environment and to satisfy human requirements to all benefits from the forests. Similarly to Ds, degradation can be classified by types (e.g., site impoverishment due to water and wind soil erosion, "green" desertification), and be described by quantitative indicators. The project of Global Assessment of Soil Degradation (GLASSOD) recommended identification of *type, degree, extent, rate* and *causative factor* of soil degradation; a clear distinction should be made between *status, rate* and *risk*. The eventual degree of forest degradation can be expressed in terms of the impacts of *forest degradation on productivity*. Such an approach develops the additional link between Ds and degradation. Indicators of sustainable forest management should include an expedient unification

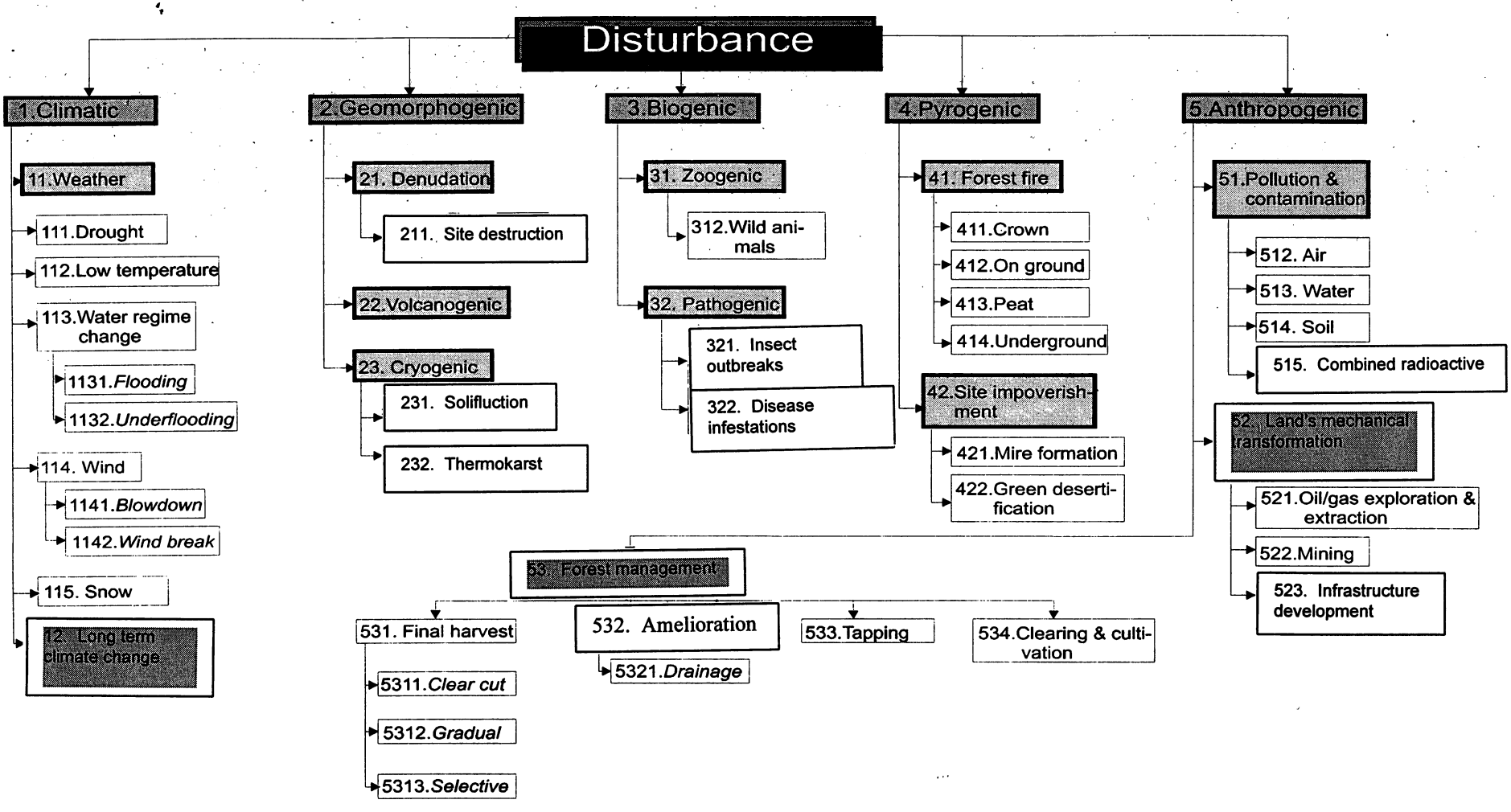

Figure 1. Classification of disturbances

of a subset of D indicators and the degradation mentioned above.

D and DR are a geographical phenomenon. At least three major gradients impact distribution, dominant forest types, succession regularities and DR of the boreal forest: temperature, moisture (or climate continentality) and anthropogenic pressure.

The set of indicators suggested to describe Ds in boreal forests is presented in table 1. Several indicators (e.g., extent, frequency) were widely used before; some of them ("universal" time, asymmetry) are referred to as succession regularities. A definite type of D requires quantitative indicators expressed explicitly. We briefly illustrate the implementation of the indicators' system presented in table 1 to describe forest fires as D. There have been previous attempts made to sufficiently quantify it by Kurbatsky 1970a; Heinselman 1981; Van Wagner 1983; and Furyaev 1996. There are several rather complicated national fire classifications (e.g., for Russia, Kurbatsky 1970; Sheshukov *et al.* 1992); but forest fire protection in boreal countries eventually uses simplified classifications which include crown fires, surface (on-ground) fires, peat (litter, turf or ground) fires and spotting (underground) fires (Van Wagner 1983; Shetinsky 1994). The *extent or size* of a fire has an explicit meaning, and modal area (in year t), and mean area for a definite period are used widely in forest management practice. The *intensity* of a fire is expressed in units or energy released per unit area per unit time. Fire *frequency* is usually defined as the number of fires per unit time in a definite area. Several derivative indicators could be useful. *Fire (return) interval* is the number of years between two successive fires; *mean fire interval* is used as the indicator of the DR. The length of time used for averaging depends upon the goals of the evaluation; the succession phase seems to be relevant for many of them. *Return time* or *fire rotation period* (natural fire rotation period; Heinselman 1973, fire cycle; Van Wagner 1978) is an indicator to describe how complete the scope of a fire is in a given point of an ecosystem. *Predictability* characterises the regularity of fires, and can be estimated as variability of fire interval, or risk (probability) of fire.

From a theoretical point of view, a change of ecological efficiency could serve as an indicator of the *severity* of a fire. However, this indicator is rather complicated for practical purposes, and other derivative indicators, such as the loss of live biomass including direct burning and post-fire die-back, could be recommended. Russian fire protection practices use a very simple and informative indicator of *fire force* which is a function of the average height of fire marks on the stems, and is highly correlated with both forest combustibles consumed and post-fire die-back (Matveev 1992). *Spatial structure* of fire as a D can be characterised by the spatial variability of its severity. *Universal time* and *asymmetry* are theoretical estimates; the first describes the possibilities to reach the climax phase of the succession consequence under the definite DR, and the second indicator indicates the shift, which the current succession cycle provides in the structure and productivity of its climax phase. *Interaction's* indicator is based on the types of D interactions in a forest ecosystem, and is measured by enforcing the ecological effects of different types of Ds, e.g., in southern taiga ecoregions of Northern Eurasia post-fire outbreaks of secondary pests could significantly increase the severity of DR. Finally, fire can significantly impact neighbouring and/or even remote areas of a landscape; *spatial coherence* is suggested to measure it. Quantification of the two last indicators could be made based on a formal description of the landscape (Rojkov *et al.* 1996).

CURRENT IMPACT OF DISTURBANCES ON BOREAL FORESTS

Currently, the three most important types of Ds in boreal forests are: fire, insect outbreaks, and harvest. Some regions are impacted by air pollution and suffer from the unregulated development of industrial infrastructure. Unfortunately, comprehensive and reliable data for all boreal zones do not exist due to different classifications used in different countries, and vast unmonitored areas, specifically in Russia. Nevertheless, the magnitude of Ds can be defined rather accurately. Some examples are presented below. Recurring forest fires are widespread in the boreal world. It is very probable that there are no vegetative areas where fires have not occurred during Holocene, and charcoal in peat deposits is evidence of this. The natural fire cycle depends upon many natural and anthropogenic circumstances.

Table 1.--*System of the Basic Disturbance Indicators.*

No	Indicator	Object	Definition and examples of units of measurement
1	Extent	FE,FL	Physical extent of D, area, size, volume
2	Magnitude or intensity	FE	Physical force of D, units of energy, concentration, etc.
3	Frequency	FE, FL	Recurrence of Ds, expressed in the number of Ds per unit time in a definite area, yr^{-1}
4	Return time	FE, FL	Average time between Ds of the same type at a particular site (in a given point), yr^{-1}
5	Severity	FE, FL	Ecological effect of D on a forest ecosystem
6	Spatial structure	FE, FL	Territorial integrity and specific of D, e.g., inverse value of the number of patches
7	Predictability	FE, FL	Regularity of frequency distribution
8	Universal time	FE	Assumed (model) time required to reach the climax state under a given regime of Ds, year
9	Interaction	FE	Strengthening or weakening of integrated ecological effect of different types of Ds on forest ecosystem
10	Asymmetry	FE	Shift of productivity/biodiversity by the climax phase of the succession consequence
11	Spatial coherence	FL	Impact of D on the landscape mosaic outside of the D area

The average estimates for the North American boreal forests are 50-100 years (Heinselman 1981; Vireck 1983) varying from 26 years in dry deciduous forests in interior Alaska (Yarie 1981) to 500 years in the wet areas of eastern Canada (Foster 1983). Data for Sweden are very similar, ranging from 50 to 270 years and averaging 110-155 years (Zakrisson 1977; Englemark 1984). Data for Russia varies greatly, from 10-15 to 250-300 years depending on the location, forest types and level of anthropogenic pressure (see review in Shvidenko *et al.* 1995).

About one-third of the boreal zone is not under any fire protection (e.g., 40 % of total Forest Fund area in Russia, or about 500 million ha of sparse taiga, forest tundra and tundra zones are not protected). According to official data for the protected areas, 20,000 to 40,000 forest fires occur annually, which cover a total average area of about 4-5 million ha (FAO 1995). From the total amount, stand-replacing fires (crown, peat) comprise about 20 % of these areas. There exist only expert estimates for the total boreal forests. These estimates are about 25 %-30 % higher in comparison with data for protected areas. For example, the annual average fire area between 1988-1993 on protected areas in Russia is 1.12 million ha, of which forested areas were 0.76 million ha. The estimate for all Forest Fund areas is 3.0 respectively 1.5 million ha (Shvidenko *et al.* 1995). The first remote sensing survey which covered all Russia estimated fire areas at 1.5 million ha (Cahoon *et al.* 1995); official statistics for protected areas were 0.9 million ha. Seasonal variability of burnt areas is very high – from 2 million ha to 15 million ha for different years.

Insect and disease damage is the second crucial factor of boreal forest dynamics. There is a rather corresponding opinion that the damage caused by insect and disease outbreaks is as great as forest fire damage in the boreal countries. Annual forest losses due to insect attacks in Canada are estimated at 51 million m^3 per year, or 1.5 times that due to wildfire and about one-third of annually harvested wood (Hall and Moody 1994). Spruce budworm *(Choristoneura fumiferana)* in eastern Canada killed 720 mln m^3 wood between 1919 and 1920 (Blais 1968). In 1992 budworm-related mortality of balsam fire and white spruce was detected on 3.94 million ha in Ontario alone (Howse and Applejohn 1993). Spruce beetle (*Dendroctonus rufipenis*) causes widespread mortality in white spruce (*Picea glauca*) in south-central Alaska (Werner and Holsten 1983). But probably the most impressive examples of severe defoliating insect outbreaks are in Russia. According to incomplete official statistics, the average annual area attacked by insect and disease outbreaks between 1971-1988 was 2.73

million ha (Isaev 1991). The area of forests destroyed by *Dendrolimus sibiricus* in Siberia during the first half of the twentieth century is estimated to be 13 million ha (Katajev 1980), and from the outbreak of 1954-1957 is 3 million ha. The last outbreak of *Dendrolimus sibiricus* in Krasnoyarsk kray between 1994-1996 damaged forests on an area of more than 1 million ha, including 0.5 million ha where more than 51 % were destroyed (Isaev 1997).

Harvest is the largest human-induced disturbance in boreal forests. Between the 1980-1990s, annual removal was between 600 and 700 million m^3, of which almost 80 % were harvested in two countries – Russia and Canada (FAO 1996). By 1995, total harvest decreased to about 500 million m^3 due to decline of harvest in Russia (volume of harvested wood in 1996 was about one-third of the level of 1988).

The effects of air pollution are significant in several regions of Russia. A special survey of forests damaged by pollutants (1989) and covering about 60 % of Russian forests estimated 0.8 million ha of dead and strongly disturbed forests. In 1993 the area of dead forest tundra landscape due to industrial emissions around Norilsk (North Siberia) reached 2.1 million ha including 0.6 million ha forested area; the earliest data from the late 1960s indicated only 5,000 ha were affected (Kharuk 1993). Based on recent estimates, there are 210 million ha of forests at risk from sulphur deposition in Asian Russia. No specific symptoms or specific responses on the forest ecosystems due to pollution impact in Canada were detected (Hall and Addison 1991).

Other types of disturbances are numerous, and their total impact on boreal forests is significant. We limit our review to a few typical examples in Russia. Large areas of forests are transformed in regions where intensive gas and oil exploration and extraction have occurred, e.g., about 0.1 million ha annually alone in Tjumen oblast in 1980s-1990s (West Siberia). Northern deer pastures in Magadan oblast (the total area by 1990s, mainly in Forest Fund areas, is about 60 million ha) decreased over the last 20 years from 5 %-7 % for each 5-year period (Stepanicheva and Nagibina 1988) due to excess grazing and fires. Many other examples could be presented (Shvidenko *et al.* 1997).
The major consequences of Ds are: death of forests due to stand-replacing Ds (e.g., all forests after crown and peat fires die within the year the fire occurred, and forests on roughly the same area die during the first 5 post-fire years as a result of other types of fires); partial mortality of trees (e.g., on-ground fires cause mortality on 10% to 50%-60% of the growing stock); changes of structure, species composition and productivity of forests. Undesirable changes of species are common in many regions. For instance, vast post-fire and post-cutting areas of indigenous coniferous forests in boreal Russian Eurasia are regenerated by softleaves species (birch, aspen); such areas reach about 30 million ha in Russian European North alone by the 1990s, and is predicted to increase to 70 million ha by the year 2015. Ds (basically on-ground fire) are one of the main reasons for the development of uneven-aged stands which are widely distributed in boreal Eurasia (uneven-aged stands of different types comprise 40 % to 90 % of the forests covered by coniferous species in the middle and southern taiga zones of Russia). On the contrary, the general opinion is that a major part of boreal zone forests in Canada, from forest-tundra to the closed-crown forests of the south, is presented by even-aged forests (Black and Bliss 1978; Cogbill 1985; Morneau and Payette 1989; Sirois and Payette 1989).

The consequential impact of Ds on boreal forest is a decrease in productivity for two basic reasons. First, stand-replacing Ds change the distribution of forest land cover, increasing areas of temporary unforested lands (clear cuts, burns, etc.), and secondary forests of low value and low productivity. Second, non-stand replacing Ds decrease productivity of actual stands. In order to quantify this impact we can express changes of productivity of actual forests in relative terms to non-degraded forests. Based on global vegetation models it is possible to estimate *potential* productivity, or productivity of a climax state of a forest ecosystem, although some artificial assumptions remain, e.g., we must assume zero areas of stand-replacing natural Ds due to fire or insect plagues. *Potential* or *climax* production of forest vegetation is only an abstract understanding of the vegetation's state if no human were to exist and nothing occurred on Earth. Such an approach does not include the transformation of lands, and consequently cannot be used as a basis for degradation estimates. Much more interesting is *achievable* productivity, or maximal possible productivity under the historically stipulated transformation of natural landscapes. In other

words, the achievable state of forests (including share and state of temporarily unforested areas) is understood as a state corresponding to the paradigm, criteria and indicators of sustainable forest management. A comparison of forest productivity in boreal Russian Asia revealed that the actual productivity of about 450 million ha is only 55 %-60 % of the achievable one (Shvidenko *et al.* 1997). We could compare it, for example, with Austria; the total aboveground NPP of the actual vegetation was found to be 7 % less than the potential natural vegetation, and an additional 34 % of potential production is harvested (Haberl 1997).

IMPACT OF DISTURBANCES ON GLOBAL CARBON BUDGET

Ds generate significant additional C fluxes to the atmosphere for which the total value can be estimated as:

$$TDF_{\rho,t_1} = \sum_{\rho} DF_{\rho,t_1} + \sum_{\rho,t<t_1} PDF_{\rho,t<t_1} - \sum_{\rho,t<t_1} RG_{\rho,t<t_1}$$

Where TDF_{ρ,t_1} is the total C flux from a forest ecosystem generated by disturbances ρ during the year t_1; $\sum_{\rho} DF_{\rho,t_1}$ is direct emissions of year t_1 (e.g., direct fire emissions), $\sum_{\rho,t<t_1} PDF_{\rho,t<t_1}$ is post disturbance (biogenic) flux caused by disturbances ρ which occur during previous years (e.g., post fire flux generated by decomposition of uncombustible organic and post-fire dieback), and $\sum_{\rho,t<t_1} RG_{\rho,t<t_1}$ is accumulation of C due to regrowth on areas disturbed during previous years $t<t_1$. The first two items in the equation give the "pure" C flux caused by disturbances. Each type of disturbance requires the explicit expression of all items of the equation presented above (Shvidenko *et al.* 1997).

There are no national publications which could comprehensively consider the impact of major types of Ds to the global carbon budget. Usually, calculations are restricted by forest fires, harvest and land-use change. Stocks (1991) estimated the total direct forest fire emission (DF in the equation) of boreal circumpolar countries at 70 Tg C year $^{-1}$. He used the average value of consumed fuel which he defined for Canada – 1.125 kg C/m^2. Based on data, presented by Kashishke *et al.* (1995), the average direct C emission for Alaska can be estimated at 1.70 kg C/m^2. This result is very close to the average data (for all types of fire) reported for Russia – 1.66 kg C/m^2 (Shvidenko *et al.* 1995), and that were used by Seiler and Crutzen (1980) for all boreal forests. Thus, probably the DF estimate for boreal forests, 70 Tg C year $^{-1}$, is underestimated by about 30 %.

Historically, land-use changes decreased the area of boreal forests and carbon content in vegetation and soils. Houghton (1996) has shown that over a period of 130 years (1850-1980) boreal forests have lost 9 Pg C (6 Pg C in forest vegetation and 3 Pg C in soils). Taking into account that circumpolar boreal forests area increased during the last two decades, it could be concluded that current land-use transformation is not a source of C uptake to the atmosphere. Nevertheless, the regional processes are observed.

The equation presented above has been applied to all basic types of Ds in Russian forests (impacts of the regrowth were not considered) (table 2). Annual average C flux generated by Ds ("pure" flux) was estimated at 377 Tg C year^{-1} for 1989-1993. The direct fluxes are estimated to 28 % of the total. This can be explained by the suppression of forest fire and decrease of harvest in Russian forests during the last decade. If we assume the same magnitude of Ds in all circumpolar boreal forests we could conclude that Ds generate a C uptake to the atmosphere of about 500-550 Tg C annually.

DISTURBANCE AND CLIMATE CHANGE

All major GCMs (OSU, GFDL, GISS, UKMO, etc.) give rather consistent predictions of ambient temperature change for the boreal zone (Ganopolsky 1994). The transient response for the boreal zone estimates that the mean annual temperature will increase by up to 1.5° C-2° C by the year 2030 and up to 3° C-5° C by the end of the next century. Significant seasonal differences are predicted. The maximal increase is expected in the northern part of central Eurasia in the winter – about 6° C-8° C (and only 3° C in Scandinavia). On the southern boundary of the boreal zone and during the Scandinavian summer, the temperature will increase by 3° C-4° C, while for the northern boundary of the boreal zone temperature increases

by only 2° C or less (Manabe *et al.* 1991-1992). The reliability of precipitation forecasts is much less than for temperature. Nevertheless, a consistent viewpoint is that precipitation will increase over the major part of the boreal zone for both cool and warm seasons, but as a rule not significantly, except for a few regions, and that increase will be greater in winter than in summer. The precipitation's increase will be not enough to compensate the temperature's increase, and the soil moisture in summer will decrease over the boreal zone by up to 20 mm except in north-eastern Siberia (Houghton 1990; Houghton *et al.* 1990; Manabe *et al.* 1992; Gubash *et al.* 1992). The results of climate change projections based on paleoanalogies (Budiko 1988; Budiko and Groissman 1991) is in the line with the conclusions mentioned above; where an even more stringent increase in temperature is predicted.

The general conclusion is that new climatic conditions will probably support the increase in productivity over all boreal and temperate zones in Russia. Potential shifts of northern species boundary are predicted for *Larix, Pinus, Picea, Abies*, etc.; although some decrease in humidity may negatively influence forest productivity in the permafrost zones. The quantitative estimates of the increased growth rate (IGR) were reported for the boreal zone from about 8 % to 15 % for the northern taiga, 5.5 %-15 % for the middle taiga and about 2 %-6 % for the southern taiga (e.g., Alcamo 1994; Kokorin and Nazarov 1994), and even 28 %-50 % for the total boreal zone (Budiko and Izrael 1987) during the next century. All of them are based on indirect analogies. Generally, direct consequences of climate change seem to be either favorable or neutral for major areas of the boreal zone.

Indirect global change consequences are much more dangerous and uncertain because they are strongly dependent on the climatic peculiarities which cannot be reliably predicted through the climatic scenario available. Moreover, a crucial point is the prediction of extreme events, e.g., the occurrence of very hot and very dry periods, storm winds, etc. Increased temperature and aridity of the major part of the boreal zone, decrease of the water table due to melting of permafrost can generate an increased probability of large-scale fires, and specifically peat and underground fires. Taking into account huge reserves of organic matter stored in boreal wetlands (from 110 to 270 Pg C in different estimations) this factor can catastrophically accelerate fire emissions to the atmosphere.

However, such predictions cannot be validated directly, and only some assumptions can be expressed. Due to expected climate change, and under additional assumptions that (1) the distribution of extreme value of climatic indicators will be similar to the current one, and (2) the level of forest fire protection will be the same, we could expect a dramatic increase of severity and frequency of fire. Some model calculations show that the probability of big fires is close to 1.0 for all major boreal regions of Russian Eurasia during the next century (Shvidenko *et al.* 1997). Evidently, the second assumption cannot be true; at least the suppression of fire during last decade is evident for all boreal zones. In addition, large regional variability is expected (Bergeron and Flannigan 1995).

Table 2.--*C Fluxes Generated by Disturbances in Russian Forests (annual average for 1989-1993, Tg C).*

Type of disturbance	Direct flux (DF)	Post disturbance flux (PDF)	Total flux (TDF)
Fire	58	92	150
Biotic	10	80	90
Abiotic	15	35	50
Harvest	23	64	87
Total	106	271	377

There is a high risk of a significant increase of insect and disease outbreaks. There is evidence that even short warm and dry periods coinciding with the life rhythm of insects can promote large-scale outbreaks of dangerous insects such as *Dendrolimus sibirica, Lymantria monacha, Limantria dispar*, etc. (Il'insky 1981; Konikov 1970; Isaev and Girs 1975; Isaev 1997) in boreal Eurasia or *Choristoneura fumiferana Clem.* and *C. pinus Free.* in Canada (Fleming and Volney 1995; Volney 1996). Warming can promote a geographical shift of dangerous insects from the northern regions to the southern regions due to a decrease in the stability of stands caused by climate change. Climate change will probably increase the widespread distribution of some diseases, e.g., stem and root fungi decay.

In order to avoid huge losses of forests due to fires in the boreal zone and taking into account the irreversible character of catastrophic fires, an anticipatory adaptation forest management system should be developed. One of the major parts of the system is the problem of forest fire protection, which could be solved based on a correct understanding of the ecological role of forest fires, and on current and future economic possibilities of human society. Among others, such a system could include the following strategic options: (1) Fire-preventing arrangement of forest landscapes, e.g., optimal spatial interchanging of forested and unforested areas, coniferous and deciduous stands; (2) Development of relevant infrastructure in the forest fund areas: road networks, forest fire break lines, etc.; (3) Hierarchical zonation of the forest protection priorities. The problem of fire protection in remote and unused boreal territories requires both ecological and economic investigation. The current fire protection paradigm, which is based on the priority of high-value resources, has to be reconsidered.

There are great uncertainties regarding the future fate of boreal wetlands (93 % of world's total wetlands). Northern ecosystems contain up to 455 Tg of C in the active soil layer and the upper levels of permafrost, or up to 60 % of the 750 Pg C, which are currently in the atmosphere as CO_2 (Oechel *et al.*, 1993). Recent measurements gave significant effluxes from permafrost soils in tundra: 35 g-160 g C/m^2 year over 125 summer days (Oechel *et al.* 1993) in Alaska, and 75 g-175 g C/m^2 year during the summer and 89 (83-94) g C/m2 year during the winter in north-eastern Asia (Zimov *et al.* 1993, 1996). These results increased uncertainties in attempting to understand the future C budget of northern territories.

INSTEAD OF CONCLUSION – RESEARCH NEEDS

Disturbances rule over the boreal world. People are able to manage them only to some, relatively modest extent, and the basic reason for that is an incomplete knowledge and insufficient mutual understanding of the problem by all boreal forests stakeholders, forest managers, harvest companies, scientists, politicians, and indigenous people. There are urgent circumpolar research needs in order to improve the multi-faceted benefit of the boreal forests:

1. To investigate a circle of classification problems such as a unified classification of Ds and a relevant set of circumpolar indicators to describe D and DR which would be internationally consistent and correspond to the requirements of national forest management systems. The close connection of Ds with successions and landscape peculiarities makes it necessary to extend the problem to consider the relevancy of a comprehensive system of boreal forest classification including a classification of forest types, forest land-use-land-cover and successions.

2. To develop a unified approach to the principles of spatial partition, quantification and mapping of boreal landscapes as a basis for the description of forest dynamics, succession regularities and disturbances' predictions.

3. To generate a unified DB on boreal forests and corresponding GIS components including consistent information on state and dynamics of forests, disturbances, etc.

4. To improve the national forest monitoring systems, and to learn possibilities of developing an internationally compatible system.

5. To develop a regionally distributed vegetation model based on successions and landscape regularities which would describe the dynamics of actual forests.

6. To develop a circumpolar boreal forest carbon and nitrogen model, based on the above

findings, and to provide unified estimations of the biospheric functions of forests for all boreal countries.

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SUSTAINABILITY OF THE BOREAL FOREST IN CANADA

J. Peter Hall[1]

ABSTRACT.--Sustainable development as promoted by the Brundtland commission is the "economic development that meets the needs of the present generation without compromising the ability of future generations to meet their own needs." This paper describes four science and policy initiatives designed to address forest sustainability. They are: Forest Health Monitoring, Forest Science, Model Forests, and Criteria and Indicators. Conventional approaches to forest management that emphasize maximum sustained yield of fibre may not fulfill the goal of sustainability. Sustainable forest development means recognizing the limits of forests to withstand environmental change and to manage human activities to optimize forest benefits within biological limits.

INTRODUCTION

The concept of sustainable development promoted by the Brundtland Commission is "economic development that meets the needs of the present generation without compromising the ability of future generations to meet their own needs." The concept was developed for biological systems; forestry and agriculture particularly. This concept has captured the imagination of the public as well as of policymakers at local, national and international levels since it is seen as a worthy goal for society and for the global community at large. The term has stimulated much discussion at the conceptual level, however, there has been little elaboration on how to apply it. (Maini 1990).

The role of forests in the biodiversity and climate change conventions negotiated in the UNCED process demonstrates the importance of issues of forests as protectors of soil, water, and ecological cycles; habitats for diverse flora and fauna. These issues are receiving increasing consideration by science and policymaking communities. (Maini 1991). The increasing commitment to environmental stewardship reflects the view that the environment, particularly its natural resources, is not an asset inherited from the past but one held in trust. Consequently, forest practices and use must reflect these values.

This paper describes four science and policy initiatives designed to address forest sustainability. They are Forest Health Monitoring, Forest Science, Model Forests, and Criteria and Indicators. The Boreal Shield Ecozone in Canada is discussed in terms of these policy initiatives, and their relationship to sustainability.

APPROACHES TO SUSTAINABLE DEVELOPMENT OF FORESTS

Forests cover half of Canada and play an important role in local, regional and global cycling of water, carbon, oxygen and nutrients. Forest ecosystems are subjected to many stresses associated with forestry activities including road-building, harvesting, and manipulation of species and cover types through reforestation. The use of forest management techniques for protection against fire, insects, diseases and competing vegetation may also stress forests. Stresses are caused by non forest related human activities such as air and water pollution, and other global changes. Forests are unique in that they are located within the territories of sovereign nations but their environmental role extends well beyond national borders (Maini and Ullsten 1993). Management activities in one country may impact on forests biologically similar but under different jurisdictions. The role of forests in global climatic cycles demonstrates their environmental significance beyond political boundaries.

Conventional approaches to forest management that emphasize maximum sustained yield of fibre may not fulfill the goal of sustainability. To fulfill this goal, an enhanced knowledge base coupled with fundamental changes in forest policies is needed. Policymaking communities are faced with the need to reconcile national interests with international responsibilities and to develop appropriate policy responses, instruments, and institutions. Sustainable forest development means recognizing the limits of forests to withstand environmental change, and managing human activities to produce the maximum

[1] *J. Peter Hall, Canadian Forest Service, 580 Booth Street, Ottawa, CANADA K1A OE4*

level of benefits obtainable within biological limits (Maini 1992).

Several parameters describing productive capacity, renewal capacity and species and ecological diversity are used to assess forest sustainability. Nations with a significant forest sector are accelerating the development of monitoring systems to provide timely information on the state of health of forests (Hall *et al.* 1996). Nations are establishing national ecological reserves and parks of representative and unique forest types to protect biological diversity and to provide baselines against which the environmental consequences of human activities can be determined.

Sustainable development requires systematic socio-economic research to understand and influence decision-making processes and to develop innovative approaches to harmonize social, economic, and environmental goals. Forest managers are developing policies and frameworks for sustainable forest development strategies. An equitable distribution of the benefits of forest ecosystems is a prerequisite for creating the incentives needed to maintain the Earth's biological wealth (Watson *et al.* 1996). Application of these measures have different impacts on various parts of our forests although the goal of sustainability is universal. Canada has several initiatives designed to promote sustainable development. They are discussed below.

Monitoring Forest Health

In 1994, as a result of predictions of damage from pollution, the Canadian Forest Service (CFS) initiated a national program of forest monitoring called the Acid Rain National Early Warning Sytem. The objectives of this program are to detect the early symptoms of damage to forest trees and soils caused by acid rain, to identify the causes of damage to forest ecosystems, and to monitor changes in vegetation and soils (Hall and Addison 1990, Hall 1995a). This is a national program; other similar systems have been established to study conditions on a regional basis (Lachance *et al.* 1995).

Results have shown that although considerable damage occurs in boreal forests, most of it is caused by known factors: insects, diseases or weather events. Boreal forests are generally adapted to these stresses and hence few adverse effects on sustainability are expected. Some forests have been damaged severely by pollution in the past, and some are currently being damaged (Hall 1995a). Forest ecosystems are not well adapted to human-caused stresses such as air pollution or global warming and exposure to these stresses can adversely affect sustainability. These early warning systems enable the forest manager and the policymaker to take preventative and remedial actions in the face of environmental stress and so protect our forests.

Forest Science

An environmental forest research program is essential to understand the basis for a sustainable forest (Maini 1991, Hall *et al.* 1990). Science must be developed in such a manner as to allow its inclusion into the policymaking sphere. It is essential to use the existing knowledge base to support activities contributing to forest sustainability. We need research to predict the response of forest ecosystems to disturbances associated with natural causes and human activities so that development in forest ecosystems occurs within an environmental perspective (Maini 1990, Hall *et al.* 1990).

A series of site classifications for Canadian forests has been developed based on the biological characteristics of the disturbed sites (Sims 1992). Application of the most effective forest management techniques ensures that forest lands can be regenerated adequately following disturbances such as harvesting, insect attack or blowdown. A biological approach enables the natural successional patterns to be encouraged resulting in the growth of a healthy, productive and diverse forest.

Biological control methods are being developed for the control of exotic forest pests through the use of introduced predators, and parasites. These are not normally part of the parasite/predator complex associated with pests in their native habitat. Several exotic pests formerly damaging to Canadian forests have been controlled using classical biological control methods. Successful control means that pest damage is kept at levels that do not adversely affect sustainability, and the use of chemicals which may harm other beneficial organisms is avoided (Hall *et al.* 1990).

For several decades the only available control method of indigenous insect pests was that of chemical insecticides which adversely affected other organisms in the forest and upset natural ecological balances. The encouragement of natural control systems was seen as a better approach. The naturally occurring bacterium *Bacillus thuringiensis* (Bt) which infects defoliating insects, was developed to an operational stage and is now used to manage such pests. Pest

such as the jack pine budworm (*Choristoneura occidentalis*), gypsy moth (*Lymantria dispar*) and forest tent caterpillar (*Malcosoma disstria*), are now managed using Bt. (Hall *et al.* 1990). This natural control agent has minimal effects on the forest environment, other than the target insects, leaving natural insect populations intact.

Large-scale reforestation programs were implemented in Canada in the 1980's. In 1989, approximately 800 million seedlings were raised in nurseries for reforestation of cutover and burned forest sites. Extensive seedling mortality soon became common caused by a rapidly-spreading root-feeding nematode. Research on the ecology of this organism showed that it could be controlled by modifying growing methods. This was as effective as pesticides which had been previously used, but avoided adverse environmental impacts (Hall *et al.* 1990).

These few examples of science addressing forest biology problems by working within the natural system provide the support for sustainable forest management. Research is being done in many fields, and is now expanding into biodiversity and the management of forests on a landscape basis (Hall 1997). These fields are being explored while incorporating the advances made in previous years as newer and more complex problems are addressed.

Model Forests

There is a need to develop and implement innovative on-the-ground solutions to address local and global concerns. The task for forest managers is to try to accommodate diverse expectations from the forest while ensuring the economic well-being of forest dependent communities. The Model Forest Network is an initiative designed to generate ideas and solutions for sustainable management of forests (Anon. 1996). Ten such forests have been established in Canada and interest is growing in the concept elsewhere.

A common feature of the Model Forest program is the building of local partnerships among individuals and organizations with an interest in sustainable forest management. Participants collaborate to determine approaches to sustainable forest management appropriate to their environmental, socio-economic, cultural, and political context. The Model Forest partnership has no jurisdictional authority over the land base, however, all those who do have such authority are participants (Hall 1996). In this fashion, the groundwork is laid for implementation of sustainable policies that can evolve over time.

The Model Forest partners use a variety of approaches to develop effective decision-making structures. The enthusiasm and innovative spirit that emerges is a more powerful force than that of groups working against each other, or independently, for similar objectives. Social stability and protection of peoples' livelihoods are keys to sustainable forests. The Model Forest Network is built upon a broad range of interests; educational institutions, industry, aboriginal groups, various levels of government, and community and public action groups.

Four of the 10 Canadian Model Forests are situtated in the boreal forest ecozone and two others are in boreal-like conditions. They cover extensive areas of 1-2 million hectares each and, contain a variety of partners, and are actively promoting a sustainable approach to forests.

Model Forests go a long way toward enabling forest managers to determine the management planning process required to provide a sustainable supply of goods and services from the forest, without jeopardizing the biodiversity, ecosystem integrity, or productive capacity of the forest. Partnerships create new challenges and increase the complexity of current ones, but at the same time they offer the opportunity to share information and to make choices between conflicting objectives. They key to sustainable forest management lies in reconciling diverse objectives.

Criteria and Indicators

One approach to defining and assessing sustainable development of forests is the integration of science and policy in Criteria and Indicators of sustainable forest management (Anon. 1995a, 1995b). Criteria are a category of conditions or processes by which sustainable forest management is assessed. The Indicators are measures of the conditions or processes of a criterion to explain and demonstrate sustainability. They are scientifically based, nationally and internationally accepted measures providing the basis for assessing and monitoring sustainability.

The Canadian C&I were developed by science and technical groups using a consultative and inclusive process by people from governments, the academic community, industry, non-government organizations,

the aboriginal community, and other groups with an interest in forests. The result of these discussions is the current assessment of forest sustainability representing the best available scientific knowledge of the subject.

The Canadian C&I reflect an approach to forest management based on the need to manage forests as ecosystems in order to maintain their natural biological processes. The C&I recognize that forests simultaneously provide a wide range of environmental, economic, and social benefits, and that an informed, aware, and participatory public is essential for sustainable management.

The C&I provide information on trends in the status of forests and related values over time. Much of the data required for national reporting are provided through current information systems. Some indicators will evolve as new data, techniques, or research results become available. Where there are no reasonable quantitative measures, qualitative or descriptive indicators are used to describe the current status. The C&I are designed to reflect the range of forest conditions and so may be appropriate for reporting at a national or provincial level, or for an ecological zone or region. They are a flexible framework within which to capture the range of ecological, social, economic, and cultural conditions.

The Canadian C&I are designed to be compatible with similar international processes, while providing more detain and precision on our own forest values. National circumstances differ in economic development, patterns of land ownership, population, forms of social and political organization, and expectations of how forests should contribute or relate to society. C&I will facilitate international reporting, and provide the framework for international agreements while reflecting national differences in characteristics and descriptions of forests.

Concepts of forest management evolve based on scientific knowledge of how forest ecosystems function and respond to human interventions, as well as in response to changing public demands for forest products and services. The criteria and indicators are intended to be reviewed and refined on an ongoing basis to reflect new research, advances in technology, enhanced capability to measure indicators, and an improved understanding of what constitutes sustainable forest management.

BOREAL SHIELD ECOZONE

This Ecozone covers the forested part of Canada from northeastern Alberta east to the Atlantic Maritime Ecozone (Wiken 1986). The Boreal Shield Ecozone comprises about 30% of the area of forest in Canada. It is dominated by broadly rolling uplands and lowlands. Precambrian granitic bedrock outcrops are interspersed with glacial moraines, and fluvial deposits. Soils are mainly Podzols in the south and Brunisols in the north. The Boreal Shield Ecozone generally supports closed stands of conifers: white spruce (*Picea glauca*), black spruce (*P. mariana*), balsam fir (*Abies balsamea*), and tamarack (*Larix laricina*). Towards the south, there are more broadleaf trees: trembling aspen (*Populus tremuloides*), balsam poplar (*P. balsamifera*), and white birch (*Betula papyrifera*). Conifers common in the south include white pine (*Pinus strobus*), red pine (*P. resinosa*), lodgepole pine (*P. contorta*), jack pine (*P. banksiana*), and eastern white cedar *(Thuja occidentalis*). Lakes are common and cover about 20% of the area (Lowe *et al.* 1996). The continental climate is modified by maritime conditions in the east and by cold artic air masses from the north.

The Ecozone is characterized by settlements with small, resource-dependent societies based on forestry, mining, trapping, and tourism. Large-scale forest exploitation began in the late nineteenth century with the harvesting of conifers for the pulp and paper industry. The ecozone represents a major economic asset with about 32% of all standing timber in Canada. Disturbances occur on an extensive scale; fires, insects and harvesting are the primary influences on forest succession.

Much of this forest regenerates naturally following disturbance, but natural regeneration is often adversely affected by poor harvesting practices. If these practices leave slash and cones on the site instead of skidding branches and tops to roadside then many sites will regenerate naturally (Hall *et al.* 1990). With the increase in mechanization, however, less emphasis was placed on biology than on economics, and regeneration problems increased. Only when former practices were resumed did the sites regenerate naturally. Harvesting practices although economically attractive, proved to be unsustainable.

To ensure adequate regeneration in boreal forests, a series of biologically-based site specific prescriptions was developed using successional descriptions and empirically derived regeneration techniques. The results of many years of ecological studies were

synthesized to develop descriptions of forest ecosystems (Sims 1992). This provided the framework for sustainable harvesting practices and encouraged the establishment of natural regeneration, both requirements for sustainability.

Results of many years of monitoring by forest health practitioners indicates that damage in boreal forests was caused by insects, disease, weather events, fire etc., all of which are natural occurrences and to which forests are adapted. Other forests in Canada have been and are being affected by pollutants, but no such damage has yet been observed in boreal forests. (Hall 1995b).

Wildfire is a characteristic feature of boreal forests and appears to be increasing while resources for suppression are decreasing. Increased losses of wood, coupled with losses from harvesting, insects, and diseases, are increasing the pressure on forest use. If these large-scale disturbances continue to deplete the forests at current rates, then sustainability may be compromised. (Salmoni *et al.* 1995, Simard 1996). The C&I integrates socioeconomics, science and development to promote sustainability in these forests. The Boreal Shield Ecozone forests are in a relatively early stage of exploitation, and mechanisms are now in place to monitor sustainability.

CONCLUSIONS

The importance of forests to Canada and indirectly to the international community has initiated the development of several measurements of sustainability. This development has both a biological and ecological component and reflects a growing awareness of forests as an essential part of our environment. Forest health monitoring, science, demonstrating and testing sustainability in model forests, the integration of science and policy in Criteria and Indicators were established to promote sustainability.

The science of forest management has evolved along with settlement, political and economic evolution, and the demand for forest products. Society over the same time has evolved in terms of demands and the sophistication with which we view the forest.

This overview does not lead to precise conclusions. Is the boreal forest sustainable? The largest of Canada's forested ecozones yields products and has been exploited for the last century. Now there are concerns that insects, diseases, wildfire and harvesting are together exceeding the annual production of fibre. Is this sustainable, and if not, what management actions must be taken to ensure sustainability? The Boreal Shield Ecozone is utilized and managed extensively and is becoming subject to land use conflicts. The products of these forests have changed as a result of demand; at the same time the value the public places on forests has changed. What part, or parts, of the ecozone are sustainable and how do we keep it so?

What we can conclude is that a number of measures to assess sustainability have been or are being implemented and it is now up to society to use them to ensure sustainable forest management. I remain an optimist that we will do so.

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CHANGING LAND-USE PATTERNS IN BOREAL FORESTS

DEGRADATION OF FORESTLAND IN LAND-USE/COVER PATTERNS OF RUSSIA

Vladimir Stolbovoi[1]

ABSTRACT.--By manifesting the response of land to human activity and impacts, degradation of forestland indicates the locations where society is in conflict with sustaining the forest environment. The analysis of land degradation in various land-use/cover patterns of Russia's vast forest zone (about 1,050 million ha, or 63% of the country) clearly demonstrates two human-induced problems, inappropriate technology and improper management, causing land degradation on 9% of the territory. The study illustrates the high vulnerability of forest soils (in comparison to steppe grassland soils) after conversion to intensive cultivation. Thus a balanced combination of forest and cropland has been found to be the most sustainable land use in the forest zone.

INTRODUCTION

Russia's potentially forested territory, termed the forest zone, covers about 1,050 million ha or 63% of the country. The territorial structure of this huge area comprises an intricate mosaic of various land-uses and land-covers (LU/LC pattern). Land use refers to the purpose for which humans exploit the land and includes the characteristics of the biophysical environment and socio-economic conditions. Land cover refers to characteristics of the land surface, i.e., the external features of land use. Major features of the LU/LC patterns are driven by a combination of physiographic peculiarities of the territory, such as relief, vegetation, soils and a variety of land-uses occurring in the area--for instance, cultivation, wood cutting, settlements, etc. These interrelated elements affect the environment, and changes in one area may have positive or negative reactions in others. Applying various land-use systems and changing them in historical perspective, humans have actively shaped the forest environment, converting forest into agricultural land, transforming natural forest into managed ecosystems, etc. However, these activities do not always enrich natural forest potentials, to allow the increasing and changing human needs to be met in a better way. Very often, particularly in the long run perspective, they create problems in sustaining production or the environment, suggesting a reassessment of the benefits that had been initially achieved.

Land degradation [4,6] affects land quality and/or functioning, often indicating a negative reaction of the land to inappropriate human activity. As a concept, land degradation was primarily developed for agricultural land, referring to the deterioration of land productivity. Expansion of this concept to forestland recognizes that forests play a multi-functional role, for instance, in land protection, hydrological and geochemical cycles, climate impact, etc. This means that productivity decline, which is the main criterion for quantifying degradation of agricultural land, does not encompass all aspects of the negative consequences for forestland. It leads to the conclusion that the concept of land degradation has to be adapted to the complex functions of the forest environment. This concept will be valuable for the assessment of technology and management systems that are applied in the forest sector. For example, industrial harvest by clear-cutting causes replacement of naturally self-maintaining forest by man-managed ecosystems. In some regions [1] due to this technology, vast territories have lost the potential for afforestation, or planted forest can produce only low-quality wood instead of real forest vegetation. The investigation of the dynamics of this development [2] has concluded that in the next century broad-leaved and coniferous broad-leaved forest will practically be replaced by anthropogenic ecosystems. Due to this perspective, to maintain the ecological balances on this huge territory, appropriate land-use regulations have to be implemented that protect the ecological functions of the forest. The alarming fact is that currently nobody can predict what the cost of current forest practices to future generations will be.

Obviously, land degradation has most severely developed on the territories where current land use does not fit environmental conditions. Thus it could be applied as obligatory information to indicate "hot" spots where society has harmfully intruded on the forest environment. It is important to implement preventive measures against land degradation at an early stage in these regions to avoid irreversible damage. Furthermore, the analysis of degradation in

[1]*Vladimir Stolbovoi, Dokuchaev Soil Institute, 109017, Pyzhevskey Line 7, Moscow, Russia*

the context of the negative effects of human impacts might be applied as a helpful tool to find a balance between human activities and nature, optimizing structure and functioning within LU/LC patterns. Some authors have stipulated that maintenance of the productive potential of land resources, and prevention of land degradation, are fundamental elements of sustainable land use [7].

To investigate the LU/LC patterns a multi-sectoral analysis approach has to be used. One of the problems that make the task difficult to fulfill is the lack of a common information base. Traditionally, the inventory and analysis of the LU/LC patterns is carried out as separate compartments (cropland, forest, water, etc.) and by different institutions (agriculture, forestry, etc.). This approach creates a narrow and fragmented view of the territory for holistic understanding. The problem becomes much more complicated when taking into account the fact that various science disciplines and practices use different methods and terminology. Currently it seems that these problems can be partly overcome by applying geographical information systems (GIS) methods to allow the integration of different knowledge on the basis of spatial unity.

OBJECTIVES

The overall goal of the present study is to improve the understanding of the interaction between society and nature. The research uses data on degradation of forestland to indicate main human-induced damage to the forest environment occurring in the Russian forest zone. To achieve this task the following topics have been considered:

1. Overview of degradation of forestland within land-use/cover patterns of Russia;
2. Investigation of major anthropogenic and natural driving forces causing negative developments in the forest zone of the country.

MATERIALS AND APPROACH

The present study is based on a broad geographical analysis of a new digital georeferenced database on the land of Russia. The database has been created jointly by the United Nations Food and Agricultural Organization (Rome, Italy), the International Institute for Applied Systems Analysis (Laxenburg, Austria) and the Dokuchaev Soil Institute (Moscow, Russia).

The database comprises of a set of spatially and temporally consistent digitized georeferenced coverages (relief, vegetation, soil including soil degradation attributes, land category, etc.) handled by GIS Arc/Info. An overlay of the coverages on land categories and soil (with soil degradation attributes) was used in the analysis.

The LU/LC patterns relevant to the research topic were distinguished by aggregation of the legend of the digitized Land Category Map [5]. The aggregation was aimed at differentiating type, degree/extent of human intervention for a number of broad land-use/cover categories. Soil and land degradation attributes [9] for the newly generated LU/LC pattern polygons were established by overlaying with the soil coverage.

DISTRIBUTION OF LAND DEGRADATION BY LAND-COVER PATTERN

Land degradation types, definition and causative factors are shown in table 1. Land degradation types can be grouped with regard to the distortion of major forest functions: the forest protection function (water and wind erosion, terrain deformation, compaction); the hydrological function (erosion, compaction, desertification, partly secondary salinisation and underfloods); the local climate regulation function (desertification, surface corrosion, thermokarst); and the biodiversity and production function (disturbances of soil organic horizons due to wood cutting and fires). It is necessary to note that this grouping is rather simplified because in reality most degradation types may affect several of the land functions. For instance, disturbances of the topsoil organic horizon may aggravate erosion and affect the hydrological function.

Analysis of human-induced factors (table 1) causing increased land degradation shows that they can be combined into two general groups. The first deals basically with inappropriate technologies in harvest processes, type of machinery and wood transportation. The second deals mainly with improper management systems for forest protection against fires. In both cases land degradation gives a clear signal on the negative response of the forest environment to human intervention. Thus it can be concluded that degradation of forests could be used for detecting society-forest environment conflicts. This indicator can be applied as a powerful argument

Table 1.--*Land degradation types, definition and causative interpretation.*

Type of degradation	Definition	Causes
Water erosion	Loss of topsoil by sheet erosion	Inappropriate technology causes unobstructed flow of run-off water, weak soil structure
Water & wind erosion	Combined action of water and wind depending on seasons humidity	Inappropriate technology causes ineffective seasonal protection against water and wind action
Wind erosion	Loss of topsoil by wind action	Inappropriate technology causes insufficient soil protection by vegetation
Terrain deformation	Development of gully or rill erosion or mass movements	The same
Compaction	Deterioration of soil structure by trampling	Inappropriate technology, repeated use of heavy machinery, overgrazing and overstocking.
Underfloods	Human-induced hydromorphism	Inappropriate technology causes rising water table, increased flooding
Secondary salinization	Increase of the salt content of the topsoil	Inappropriate technology causes improper irrigation
Desertification	Decrease of average soil moisture content	Inappropriate technology causes changes in the local hydrology
Disturbances due to wood cuttings	Distortion of the topsoil organic horizons	Inappropriate technology, clear cutting, skidding, etc.
Disturbances due to fires	Distortion of the topsoil organic horizons	Inappropriate management, insufficient forest protection
Surface corrosion	Solifluction, landslides	Inappropriate technology causes distortion of the thermoisolating topsoil
Thermokarst	Development of thermokarst caves very often filled with water	Inappropriate technology causes distortion of thermobalance of subsurface permafrost

Table 2.--*Types and extent of degraded and stable land in land-use/cover patterns of forest zone (FZ) of Russia.*

Land use/cover (LU/LC)	Forestland converted to cropland				Timber cutting (groups III & II)		Use of natural				Unused (Elfin) woodland		Total D&S	
Types of	Totally		Partly				Forest		Forest-tundra					
Degradation (D), stabilization (S)	Million ha	% of LU/LC	Million ha	% of LU/LC	Million ha	% of LU/LC	Million ha	% of LU/LC	Million ha	% of LU/LC	Million ha	% of LU/LC	Million ha	%
Water erosion	5.3	30	0.3	25	2.0	8	1.1	2	0.0	0	0.0	0	8.7	9
Water and wind erosion	0.2	1	0.1	8	0.4	1	0.4	1	0.0	0	0.0	0	1.1	1
Wind erosion	0		0.1	8	0.3	1	0.2	0	0.0	0	0.0	0	0.6	1
Terrain deformation	0		0.0	0	0.1	0	0.2	0	0.0	0	0.0	0	0.3	0
Compaction	12.3	69	0.0	0	0.2	1	0.1	0	0.0	0	0.0	0	12.6	14
Desertification	0		0.3	25	1.5	6	2.1	5	0.0	0	0.0	0	3.9	4
Secondary salinisation	0		0.1	8	0.1	0	0.7	2	0.0	0	0.0	0	0.9	1
Disturbances due to cuttings	0		0.0	0	10.0	41	0.0	0	0.0	0	0.0	0	10	11
Underfloods	0		0.0	0	0.1	0	0.6	1	0.0	0	0.0	0	0.8	1
Disturbancies due to fires	0		0.3	25	4.0	16	9.3	21	0.0	0	0.5	16	14.1	15
Surface corrosion	0		0.0	0	4.7	19	28.2	62	0.6	75	2.2	69	35.7	39
Thermocarst	0		0.0	0	1.0	4	2.3	5	0.2	25	0.5	16	4	4
Total D and % of the LU/LC	17.8	94	1.2	4	24.3	6	45.2	9	0.8	4	3.2	4	92.7	9
Artificially stable	not calculated		7.7	25	26	7	16.9	3	0	0	0.1	0	≅50	5
Stable by vegetation	not calculated		22.4	75	345	90	414.6	96	19.6	94	74.5	95	≅876	92
Stable unvegetated	not calculated		0	0	13	3	4.8	1	1.2	6	3.5	5	≅22	2
Total S and % of the LU/LC	not calculated	≅6	30.1	≅96	383.4	≅94	436.4	≅91	20.7	≅96	78.1	≅96	≅947	≅91
Total LU/LC and % of the FZ	18.9	2	31.3	3	403.8	38	489.1	47	22.4	2	82.4	8	1049.6	

humid climate, where water erosion causing soil detachment and transport is very high. Secondly, traditionally cultivated land in the forest zone occupies externally drained slopes. Flat interfluves are covered by swamps and bogs and are unsuitable for cultivation. Slope cultivation requires land protection measures, which obviously can decrease erosion to some extent but very seldom prevent it completely. Thus, the development of compaction and water erosion in the forest zone is strongly pre-conditioned by physiographic and geographic features.

The nature of forest soils is another important factor influencing erodibility. Soil erodibility is the integrated effect of processes that relate to the resistance of the soil to particle detachment by rainfall and subsequent transport. Figure 1 shows the extent of water erosion and compaction on the cultivated soils formed under forest and steppe vegetation. As can be seen, 61% of cultivated soils (cropland) which formed under forest are affected by compaction, and 27% of the soil is influenced by water erosion. Cultivated soils developed under steppe grassland contain only 30% of compacted and 7% water eroded soils, i.e., they are only half and one-fourth affected compared to soils originating from forest. These figures completely correspond to differences in erodibility of major arable soils of Russia (fig. 2). As can be seen, Podzoluvisols, which primarily formed under forest vegetation, manifest erodibility at the levels between 3 t/ha and 3.5 t/ha for medium textured soils and from 2 t/ha to 2.5 t/ha for fine textured soils. Chernozems, originally developed under steppe vegetation, are characterized by erodibility from 1.5 t/ha to 2 t/ha (medium textured soil) and 1 t/ha to 1.5 t/ha (fine textured soil). These values are about half compared with Podzoluvisols. Greyzems, formed in a forest-steppe environment and having mixed features of forest and steppe soils, demonstrate a medium level of erodibility. Consequently, the erodibility of medium textured Greyzems varies from 2.5 t/ha to 3 t/ha and of fine textured from 1.5 t/ha to 2 t/ha. We conclude that the greater extent of water erosion and compaction of the cultivated soils primarily formed under forest compared to soils developed under steppe vegetation can be explained by differences in their nature. A more detailed explanation of the mechanisms causing this difference can be found from soil physics and chemistry.

It is well known that soil erodibility depends highly on soil characteristics such as particle size distribution, structural stability, organic matter content, nature of clay minerals, and chemical constituents, which determine water stable structure [8]. Water-stable macro-aggregates (>0.25 mm) are formed by processes of adhesion of micro-aggregates (<0.25 mm) and mineral particles when the soil has a high humus content, humic acids prevailing in the soil organic composition, saturation of exchange complex by calcium, etc. Forest soils have low humus content, dominant fulvic acids in soil organic composition, and unsaturated exchange complex by calcium. To improve these conditions, usually large amounts of organic fertilizers, liming, etc. have to be applied. Macro-aggregates in forest soils are formed during the winter season. When freezing, mineral soil particles and micro-aggregates are compacted by growing ice crystals. However, the adhesion of macro-aggregates is provided by water films and becomes unstable when additional water is supplied. During water logging in the spring season, the macro-aggregates are detached and the soil mass becomes dispersed. This leads to a decreasing infiltration rate, and an increase in both runoff and water erosion.

Figure 3 illustrates the composition of water-stable aggregates in forest loamy Podzoluvisols and steppe loamy Chernozems. The diagram shows that Chernozems have much more water-stable macro-aggregates as compared with Podzoluvisols. It is important to note that the amount of micro-aggregates is more than 5 times in Podzoluvisols as compared with Chernozems. Figure 4 shows the dependence of the infiltration rate on the composition of micro-aggregates (silt fraction) and aggregates in soils. As can be seen, the infiltration rate in Chernozems containing less than 10% silt particles can be expected at the level of 60 mm/hour. The infiltration rate of Podzoluvisols containing 50% of silt particles is estimated to be 0.6 mm/hour i.e., 100 times less. Thus it is absolutely clear that soil erodibility of forest soils will be much more than for soils formed under steppe grassland.

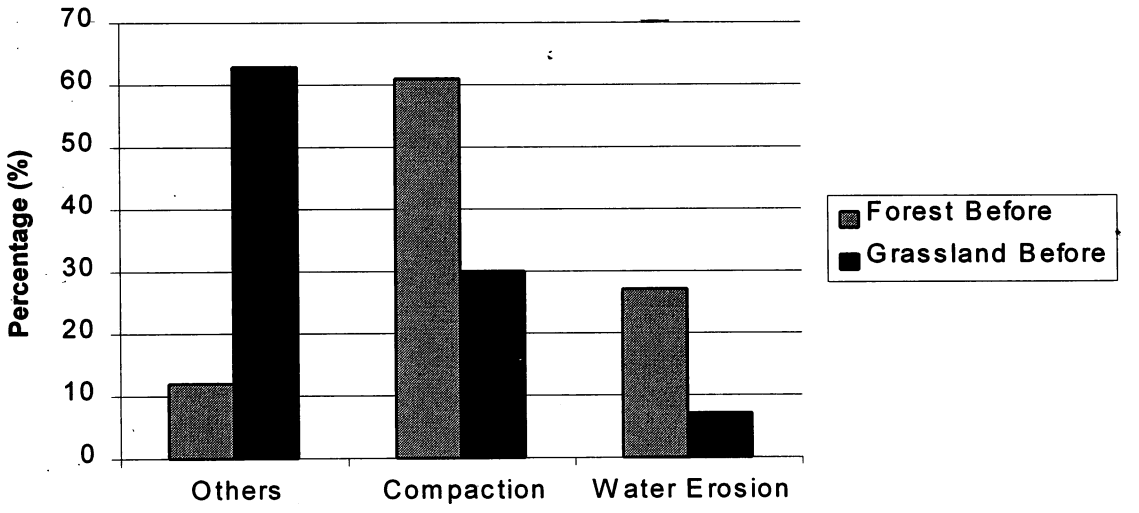

Figure 1.--*Soil compaction and erosion in soils of forest and grassland origin.*

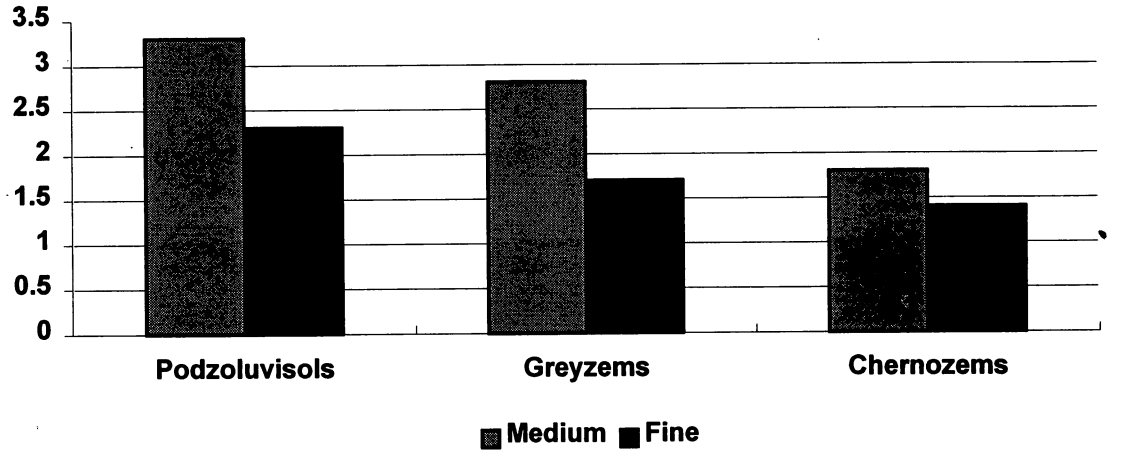

Figure 2.--*Erodibility (t/ha) of some medium and fine textured soils of Russia.* [based on 8]

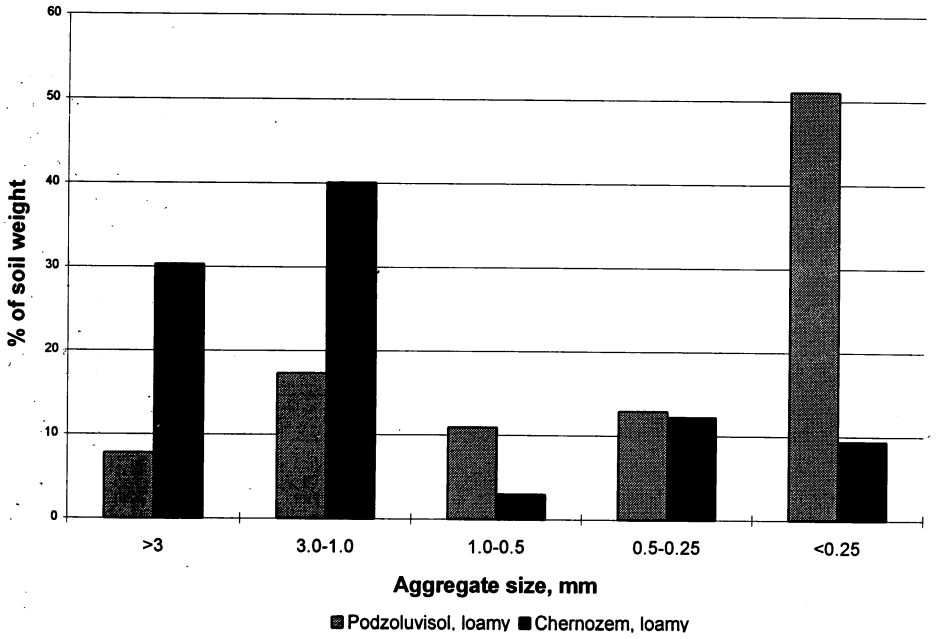

Figure 3.--*Water stable aggregates composition* [based on 8].

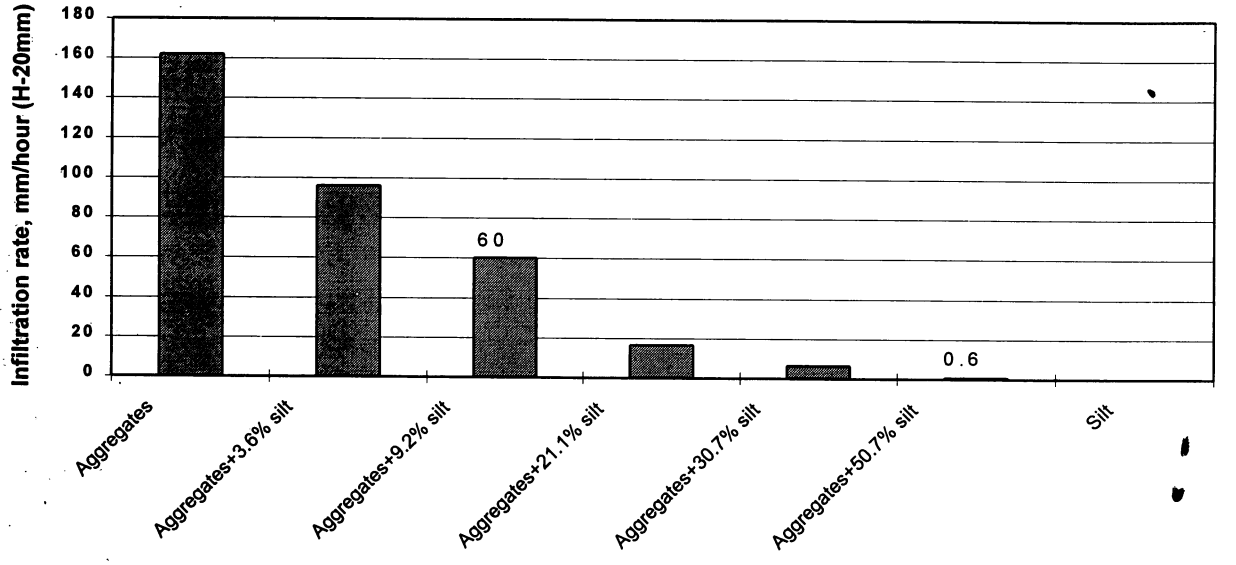

Figure 4.--*Dependence of infiltration rate upon composition of microaggregates and aggregates in soils* [based on 8].

CONCLUSIONS

It is estimated that:

- 93 million ha (about 9%) of the land of Russia's forest zone is affected by various types of degradation. More than 90% of the forest zone is still stable and performing a very important environmental role. Beyond purely economic factors, more attention to the ecological functions of forests is required when assessing the forest resources.

- Land degradation data manifest such geographically explicit data sets the negative impacts of human-induced activities on land. They allow to delineate areas, detect causes and understand mechanisms of society-forest environment conflicts. The information can help to establish suitable and functional land-use structures in various LU/LC patterns.

- Inappropriate technology and improper management are major societal causes of land degradation of the Russian forest zone. Improvements in both aspects could contribute considerably to sustaining development of the forest zone.

- Natural factors (climate, relief, soils, permafrost, etc.) are the main driving regularities of the spatial distribution of the LU/LC patterns within the forest zone, providing differentiation by types and degree of human intervention across the country. Due to low organic content, prevalence of fulvic acids in humus composition, an exchange complex unsaturated by calcium, etc., forest soils are vulnerable to intensive agricultural use. Lacking a water-stable structure, they are strangely affected by compaction and water erosion. This negative inspect of cultivation on forestland could be partly avoided by maintaining forests within cropping areas.

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PRINCIPLES OF NATURE USAGE IN BOREAL FOREST ZONE OF RUSSIA

Andrey F. Tchmyr[1]

ABSTRACT.--The landscape-geographic principle of evaluating nature usage standards is different from the principle for evaluating adopted sanitary-ecological norms. It allows monitoring of natural area complexes with respect to ecology within a landscape area and with consideration of the character of the territory's development, accounting for ecological interests in the context of solving social problems of the citizens. This approach is particularly important in reserve and refuge establishment, national parks planning and for other categories of specially protected areas. The study of natural ecosystems productivity and biodiversity, creation of cultural landscapes and gene fund preserving in taiga ecosystems is only possible at the level of natural area complex.

INTRODUCTION

Russian boreal forests occupy 20% of the territory of global forests. For their structure and organization, such forests are the most stable self-controlling ecosystems with the largest stock of carbon pool in the Northern Hemisphere (Shubin 1994). However, present local and regional influences on boreal ecosystems, as well as technogenic stresses of global importance, cause irreversible phenomena that can affect considerably the global climate and living conditions. The essential influence is observed on most taiga forests of the developed European part of Russia. The fact is that in the European part of Russia for the last 130-150 years forests have not been used according to landscape principles. In this zone, a social-economic principle of nature usage prevailed, which seldom agreed with landscape ecology principles. Considering the present condition of taiga lands it is of vital necessity to develop ecological standards based on the landscape-geographic principle.

This paper addresses three aspects of the nature usage problem in Russian taiga forests:

1. Development of a landscape-geographic approach for development of nature usage standards;
2. Methods of genetic stock and biodiversity preservation in boreal forests;
3. Description of technogenic pressures in regional development taking as an example the Leningrad region.

METHODS

The problems of ecological standards for nature usage gave rise to increasing interest of different specialists in recent years (Isachenko, G.A. 1991, Isachenko, A.G. 1993, Polyakova 1982, Alexandrova *et al.* 1988). A.G. Isachenko and G.A. Isachenko (1993) point out that the present condition of ecological standardization in Russia has been found unsatisfactory and without scientific foundation by practically all specialists.

There is no lack of different administrative regulations limiting the impact of forestry activities on the environment. Even 10 years ago, the USSR System of Standards for nature protection included up to 60 state, 40 all-European (within the system of Economic Assistance Council), and over 100 environmental protection standards of different enterprises. These were generally sanitary-hygiene standards for acceptable concentration of pollutants (ACP), industrial outbursts (AIO) and acceptable level of physical impact on the environment (ALPI). Orientation to these separate standards provides neither an opportunity to account for interaction of different pollutants nor their combined impact to the environment. Furthermore as pointed out by A.G. Isachenko (1993), sanitary-hygiene standards calculated based on the needs of the human organism are not acceptable for the whole biota. As far as biota are concerned, ecology standards should mainly take into account the weakest points of the system,

[1]*Dr. Andrey F. Tchmyr (French transcription) Federal Forestry Service of Russia, Saint-Petersburg Forestry Research Institute, 194021 Russia, Saint Petersburg, Institutski pr. 21. Tel: (812) 552-80-20/Fax: (812) 552-80-42/E-Mail: serg@forest. spb.su*

considering pollutants' direct influence and indirect influence through trophic chains, water migration, soil and air accumulation, etc.

It is difficult to deny that ecology standards should be estimated from human needs, and should protect people from dangerous technogenic influence. Finally, the problem should be solved by estimating impacts of anthropogenic pressure on geosystems, aiming to protect them from dangerous influences that prevent their natural normal functioning and lead to violation of the human life-support system. The geographic landscape can be considered as such a system. Therefore, the problem of ecology standards takes on a geographic character, which suggests new problems, such as what can be considered "standard" for a geosystem, how it can be evaluated, and what dimensions can be used.

If the focus is only on the spatial (territorial) aspect of calculating standards, the necessary data will not be possible to collect. For every natural land system (NLS) the conventionally-radical condition can be considered normal, providing a stable dynamic balance between variable landscape components (biota, soil cover) and site factors (substrate, relief, hydrothermal conditions). In this case, without deteriorating influences on the geosystem, the NLS can remain in conventionally-radical condition forever. However, with the present condition of the environment, NLS can hardly be found because of the permanent or periodic influence of factors such as climate change, tectonic phenomena, and strong anthropogenic stresses (i.e., clear felling, following use as storage ponds). Under the influence of such factors, the condition of geosystems will definitely deviate from conventionally-radical for different periods. The period of these temporal deviations depends on spatial influence (it should be considered which components of the landscape were influenced and have been changed: at the level of facies, stand or site). The higher the altitude of the landscape component being influenced in the hierarchy, the longer is the period of its original state reconstruction. Therefore, clear felling at biogeocenosis (facies) and stand levels with subsequent species change will take different times.

RESULTS

Among many dynamic deviations, there is a deviation that helps to restore the initial conventionally-radical condition of the geosystem and a deviation that completely changes its appearance and condition. The first one can be called "normal", the second, "regressionally-abnormal". Considering evaluation of normal pressure on a geosystem it is necessary, in any case, first of all to take into account the stability of its present condition that to some degree has been changed by anthropogenic influence. Further it is important to decide whether it is necessary to preserve this condition or to restore "original" conditions or otherwise transform the system to a new condition with given parameters. The last decision can be fully realized by forming a cultural landscape with given characteristics and stable conventionally-radical condition. It is also important that every ecology standard should be developed separately for geosystems of different hierarchical levels (the scale of ranks). For practical realization of such approach, it is necessary to study the landscape structure of taiga forests and map the landscapes.

In developing ecology standards for nature usage, a problem of particular importance would be preserving the north taiga genetic stock and biodiversity. General methodological principles of database formation for annual agricultural cultures are not possible to use for forest ecosystems. Therefore, the only way to preserve boreal forest genetic stocks is to organize reserves, natural parks, refuges and other types of specially protected areas. The history of reserves in our country was not consistent and has had dramatic events.

From the beginning, the network of forest reserves has been formed spontaneously. Many of them were established to preserve some rare or valuable animals and later plants. In the initial period, this purpose was justified, and even at present it is still important for realization of the goals of reserve forestry. However, presently the reserves have started to be used for many purposes. First of all, as mentioned above, they are regarded as natural stores of genetic stock and laboratories for: the study of biodiversity of particular geosystems, inventory and development of bioresource regeneration, protection of waters, etc. One of the first-rank important functions of reserve territories is to be a standard model of natural complexes, not only in the manner of museum exhibits of undisturbed nature, but also as natural laboratories for scientific research. We can note with satisfaction

that such work has started in Russia in spite of the difficulties of researchers' life and work in our country at present. This work has been carried out by a group of scientists at the Karelian Research Center of the Russian Academy of Science (Volkov, Gromtsev, Erukov, Karavaev *et al.* 1995).

If a proper quantity of standard natural models (examples) of geosystems could not be preserved, we would miss forever a chance to deeply investigate natural connections, processes and laws in their multiple appearances on the Earth. Therefore, the necessity of a landscape approach has become evident. Furthermore, the first and essential condition to be observed should be that main sites to be protected would be geosystems existing in a complete natural unity. The second, but no less important, problem in establishing landscape reserves is to make them representative. Reserve representativeness is a multi-meaning notion that includes definition of the typical character of a particular reserve representing one landscape type. It is also important to evaluate optimum size and configuration of reserve area. It is natural to expect that a larger area would be more representative and would provide more reliable conditions for maintaining a reserve in a natural condition.

Two main criteria would be basic in defining optimum reserve size. The first criterion is that the area of the reserve should encompass a definite landscape system. In other words, it should include all morphological components: facies, stands, and sites in their regular conjugated series typical for a given landscape. A full landscape profile embracing single and subordinate facies should be an essential requirement for establishing a reserve. The second consideration is that best variant of reserve establishment is to unite a group of conjugated landscapes. The biggest number (19) of reserves is in east-European sub-taiga landscapes. However their total area is hardly more than 0.5% of the whole territory, with an average area of 24,000 ha per reserve.

Landscape study has revealed that hill area reserves in the deserts and northern subtropics of Russia are in a more favorable condition, although it is evident that there are not enough landscapes embraced by reserves. Even fewer reserves have been established in the Arctic, forest tundra, north and south taiga, and West Siberia. In some sense, this drawback can be compensated for by national and natural parks. Their establishment, however, has not been consistent with the landscape-ecology principle. Of 28 national parks under the Federal Forestry Service of Russia, only half of them are well-organized and even they were established according to the methods generally used for state forest farms, which is not in accordance with landscape-geographic principle.

In developing ecology standards of nature usage for European Russia, our biggest worry is two components of the environment - "developed" and "man-made". The outstanding USA ecologist E. Odum (1986), analyzing nature usage models in the USA, followed a traditional way of dividing landscapes--accepted in landscape architecture--into three environmental components: "natural", "developed" and "man-made". In this classification "natural" is forests, meadows, bogs, and waters of lakes and rivers, "developed" - agricultural landscapes and agricultural lands, "man-made" - urbanized environments dependent on the functioning of the other two components of the life-support system.

Irrespective of the categories of environment usage described by the American ecologist, we can consider the Leningrad region area system represented by landscapes of Russian south boreal forest zone. Nowadays more than half of the Leningrad region territory is covered with forests, 20% of the area is occupied by agricultural lands, and 7% by industrial enterprises, roads and urban territories (table 1, fig.1). Comparing the nature use structure of Leningrad region with the model suggested by E.Odum it seems like there is nothing to worry about. However, areas occupied by urban and industrial systems make up about 8% of the region--without the St. Petersburg area, which is 240,000 ha and causes a severe stress on the region within a radius of 40 km - 60 km. Without considering this technical "monster", no ecology standard can be effective. The region has only 3.02% of specially protected area, which is two times less than the all-Russia standards. At present, the only regional reserve is "Vepsky Forest", which was established without respect to geographic landscape. The region desperately needs to develop ecology standards for nature usage as well as criteria and indices for sustainable forest management. At the same time, it is necessary to lessen the quantity of factories and industrial enterprises in St. Petersburg.

Table 1.--*Leningrad Region Land Fund Categories of different uses (without the Saint-Petersburg area).*

Land category	Area, (10^3 ha)	Percentage of regional area, (%)
1. Lands of agricultural enterprises	1,679.80	20.00
2. Urban areas (towns, villages and settlements)	231.70	2.80
3. Lands under industrial enterprises, roads, communications	406.90	4.80
4. Forest lands	4,716.30	56.20
5. Waters	1,078.80	12.90
6. Stock lands	230.00	2.70
7. Lands used by other regions, etc.	6.50	0.10
8. Other lands	40.80	0.50
TOTAL	8,390.80	100.00

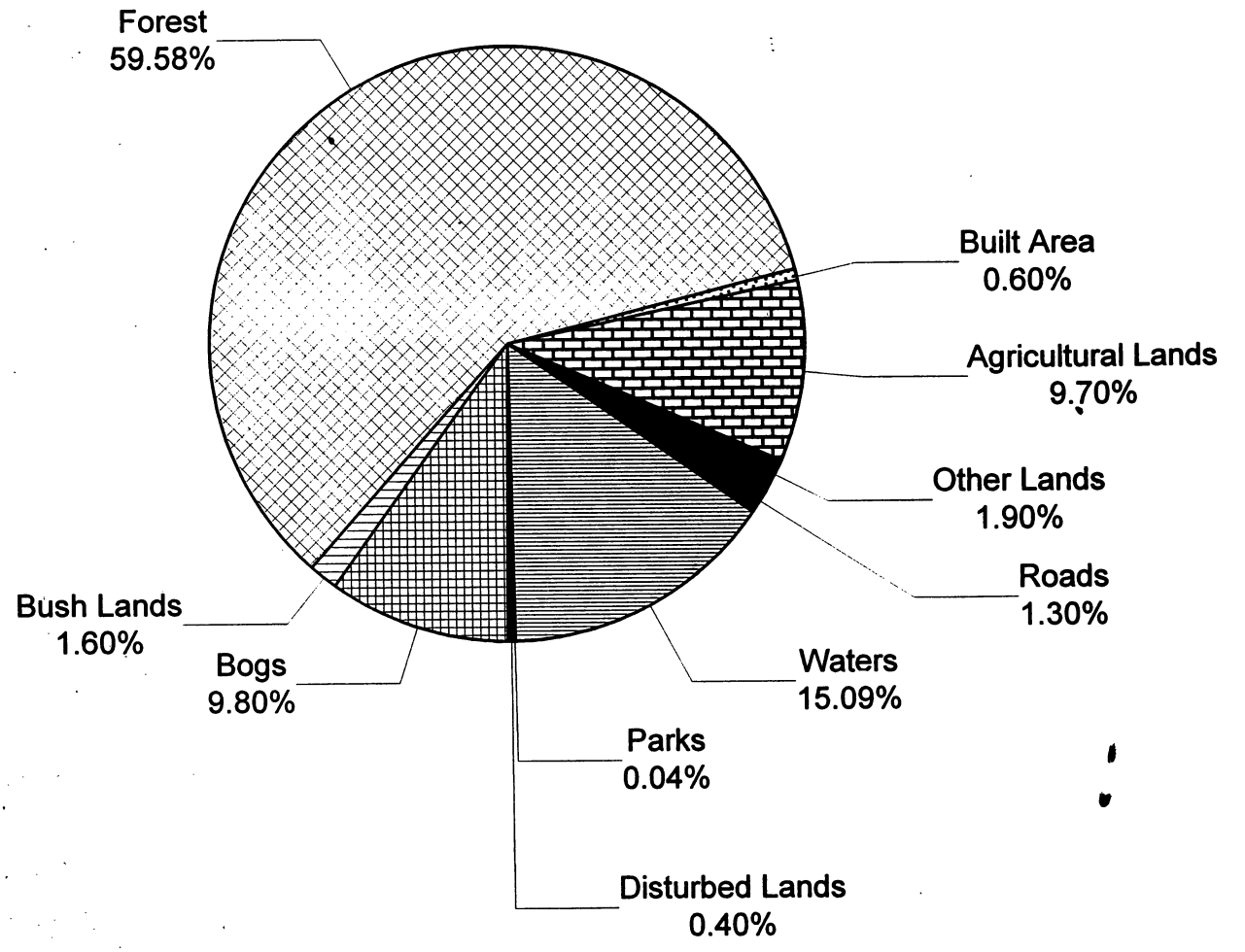

Figure 1.--*Structure of Leningrad Region Land Fund (Note: Numbers do not add to 100% due to rounding errors.)*

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SOIL AND FOREST FORMING PROCESSES AND LAND USE IN BOREAL ZONE

A. Sapozhnikov[1]

ABSTRACT.-- Determination of actuality and insufficiently studied aspects of forest utilazation organization in the boreal zone. In this context the term land use is understood as ecologically or economically determined expediency (priority) of concrete land site attribution to this or that type of utilization (for example, forest, agrarian, built-up, etc), which is determined in the first turn by ecological conditions.

The global role of boreal zone forests, thanks to its extensive territory, lies in producing a volume of phytomass that is able to influence substantially the carbon stock of the planet. This global importance a priori produces a negative attitude to any change of production potential connected with disturbances of the natural progress of forest and soil formation processes. Even well-timed (for example, plantation) forest regeneration is not able to compensate the lost carbon storage ability of forested lands for many decades.

Without commenting upon the theoretical aspects of soil and forest forming process links, independent of the rate (level) of forest exploitation or its susceptability to other types of destruction (for example, pyrogenic), the forest cover is constantly under the effect of dynamic processes - successional, age, geomorphogenic, climatogenic, hydrogenic, pedocryogenic and so on. Correspondingly, the transformation of the active biogenic part of the soil cover is observed. All this is integrated in naturally or anthropogenically changed structures of forest lands and, correspondingly, demands the introduction of corrections into the structure and system of forest (forest resource) land use.

Besides the global significance of the boreal forests, there is always utilitarian need for them, reflecting the regional and local (regions, inhabited localities) needs not only for the forest production, but also for land, including the necessity of the increase of land suitable for agriculture. The possibility and permissibility of this in rich forests regions earlier were introduced by L.V.Popov. In other words, the necessity of forest conservation constantly collides with the consumptive interests of people, thus invasion of human demands into the environment appears to be inevitable. Consequently, the challenge is in directing the nature use system to overcome and soften inevitable contradictions. This is possible with the realization of strict regulation of nature use and full elimination of the irreversible decomposition of forests and soils.

The extensive territory, occupied by the boreal belt, the diversity of forest vegetational conditions, and the variety of assimilation levels and nature use traditions suggest the potential complexity of approaches to solving this problem in different boreal zone countries. None the less, there are common problems that I would like to comment upon.

The soil forming process is the aggregate of the elementary processes of soil formation, spontaneously connected with forest forming processes. Irreversible transformations of fundamental trends occur with the break of even one link in the complex chain of interconnections. Anthropogenic impact on these relations, as a rule, even with spot (one moment) impact always appears to have multi-faceted spatial and temporal impacts. Especially sensitive in this aspect are the forest soils because their formation (soil forming process) is closely connected with forest forming process. That is practically any impact on the forest leads to disturbance of the natural procession of soil formation, which is observed during the change or elimination of vegetational cover. It can occur with or without the mechanical impacts on soils. The forest ecotope destruction process and its consequent development are fully simulated by the multiple impacts of forest fires.

In the absence of essential mechanical destruction of upper soil levels, post-anthropogenic soil formation in cut and burned areas is closely connected with forest cover successions. The following types of changes in soil forming processes are possible with this: (1) ecotope hydroregime change due to redistribution of sediments and its elements - evaporation, transpiration, inner and surface flow redistribution; the result is either bogging or increasing of drainage regime and as consequence podzol formation; (2) vegetation change is followed by turnover structure change both in composition and mass of annual top humus as well as in intensity of its further decomposition, which in the long run is reflected on postpyrogenic (postanthropogenic) formation of humus formation type; (3) under conditions of long-term seasonal permafrost usually as a result of

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[1]*A.Sapozhnikov, Far East Forestry Research Institute, 71, Volochaevskaya Street, 680020, Khabarovsk, Russia.*

cryoregime violation, stabilized by native vegetation, the ecotopic position of edaphotope changes and connected with it direction of soil formation goes to bogging and peat accumulation; (4) as a result of grass-bush layer conversion the processes of soddiness, meadows appearing, peating, arborization, ambustization and so on are likely to develop. Other types of changes and prevailing of elementary processes of soil forming are possible, but in all cases the traumatism of soils indirected through vegetation is accompanied by change of dominating (and correspondingly, initiating this soil) elementary soil process. It must be taken into account that in the limits of one soil forming process several variants of forming process are possible.

Forest soils development as a result of their exclusively mechanical damage or even destruction is also connected with the process of forest regeneration in damaged sites. But, besides this, a significant restructuring of soil cover occurs with the formation of more contrast boundaries between contours of postanthropogenic (pyrogenic) origin and not damaged ones. In the temporal aspect the dominants change in soil forming process, connected with a more dynamic forest forming process, is catastrophic only in the case of its irreversibility. Otherwise, these are inevitable fluctuations, connected with the successional transformations of forest cover and generally with the types of vegetational cover. The period of fading of the stress changes in organogenic soil horizons after clear cutting in cases of successful intensive regeneration of cut areas depending on forest formation and concrete conditions of habitat are fluctuate from 5 to 10 years.

It must be ascertained, that there is no general theory of post anthropogenic forest soil formation, and a formulation of its principles is impossible without taking into account the theory of successions and syngenesis of forest vegetation. In today's stage, characterized by increased, actually uncontrolled, anthropogenic invasion into forest, the development of such a theory becomes highly actual.

Main types of impact on the forest and soil forming process under conditions of the Russian Far East boreal zone are: forest fires, commercial forest exploitation, open mining, agriculture. The forest fires increase outside natural nature level as a result of economic activity of man is attributed by us to anthropogenic impact. Intended time limitless lands expropriation for construction, communications and so on is not considered here. It is also possible not to take into account mining because they are wittingly catastrophic and for a long period destruct the usual process of forest formation and convert soils into technozems. All types of the impact lead to the carbon emission increase and decrease of depositing functions of forest ecosystems.

Besides anthropogenic transformations it is necessary to take into account and global processes as well. So, Plochl and Cramer (1995) on the basis of the analysis of simulation of the global warming consequences showed that: (1) primary netto-productivity (PNP) of the northern biomes can decrease; (2) but common PNP of this or that region can increase due to the shift to the north of taiga biome.

Thus, under conditions of boreal zone we encounter the determined rigidity of the environment, influencing the organization and planning of rational land use. It is necessary to establish the limit below which it is inadmissible such an intervention in the boreal forests environment, which is followed by the loss of land suitability for afforestation for a long period. It is possible to propose two evident signs, reflecting the ecosystem condition under watershed assessment of the territory: (1) the appearance in the land structure the lands which are non-forest-producing and not suitable for afforestation; (2) watershed afforestation decrease up to the level of less than 60%.

In connection with high sensitivity of the forest cover the problem land use organization is actual. It is known, that the agrarian significance of lands here is extremely low. Thus, it is extremely important here the multifunctional land use, including agroforest use. That is we are talking about realization in the boreal zone of agroforestry as it is understood by Kovda (1984) and King (1990). In the boreal zone the only reserve of the agro fund lands increment is ploughing up of the forest lands. Now the similar direction must take into account the danger of erosion and mountaneous rivers pollution.

Nevertheless, the receipt of additional production without ploughing up of new lands and, moreover, without their serious transformation is fully possible: (1) by the way of more rational land use, including forested lands; (2) by creation of protective forest stripes system on farming lands; (3) by maximum full use of forest food, technical and medicinal raw materials; (4) by organization of regulated under canopy of the cattle grazing; (5) by creation of openings for the grazing; (6) by the use of openings, open stands, clearings for getting fodder; (7) by use of between-strip space in the forest plantings for getting agroproduction; (8) by attraction of branch feed for animal husbandry; (9) by regulated extraction of the part of forest top humus and forest litter for utilization as compost material; (10) by introduction of fruit underbrush under the stand canopy; (11) by development of beekeeping; (12) by organization of cell fur farming; (13) by hunting trade and so on.

The concrete choice of economic activity is connected with the necessity of forseeing (prognosis) of possible consequences of land assimilation. In relation with this the classification of lands is proposed according to their stability (resistivity) to any external impacts:
STABLE - destruction is possible only as a result of intended mechanical impact (for example, mountainous mining); uncontrolled destructions (erosion, windfall) are unlikely;
RELATIVELY STABLE - extraction of vegetational resources or vegetational cover destruction is not followed by irreversible consequences and is restored up to original condition in a comparatively short period (not more than 5 years);
MID-STABLE - lands are rapidly restored under any types of nature use, but subjected to the impact of calamities - windfall, entomologic pests, influence of channel flows, floods and so on;
INSUFFICIENTLY STABLE - they are easily subjected to postfire destructions or anthropogenic vegetation changes; however, if the destruction of soils do not occur, the restoration is observed not later than 20 years;
LOW STABLE - together with destruction of vegetational cover the local (mosaic) soil destruction is registered due to which the return to original condition is difficult to forecast;
UNSTABLE - any impact leads to irreversible changes of the landscape; tundra permafrost landscapes are the manifestations of this in a greater extent.

The decrease of stability level is evident in the course of movement to the north - if in the forest zone all classes of stability are possible, then in tundra-forest and tundra zones there are only last three classes, which testifies their high sensitivity. It is observed analogously in high zonality.

There are different approaches to the classification of landscape stability depending on the taken signs for the diagnostics. There is a way, which is more suitable for creating the Classificator of Landscape Stability, necessary for rational selection of land use type. The evaluation of landscape stability (sensitivity) according to soil signs are widely known. This is justified by sufficiently high level of soil conservatism in comparison with the other active components of landscape. Really, if as a result of any impact the vegetation suffers, but soil is maintained, then the primary landscape ecotopic signs are maintained; in case of soil destruction the irreversible consequences are inevitable for vegetation with the loss by biome of initial ecotopic signs. Correspondingly, the change of land use type is inevitable. Thus, The Classificator, actuality of which is mentioned above, must be based on thorough registration of soil and forest forming processes interconnection, that is, it must answer the questions: what are the consequences of vegetation destruction for soils, and on the contrary, the role of soil destruction in vegetation changes or destruction. It is especially important on the sites, attributed to ecological skeleton of the territory. Only in this case the rational land use system organization is possible.

The situation analysis along the Far East transect (135 meridian) showed, that in boreal belt there is not very large choice of land use systems. Mainly these are the combinations of forest resources and limited agrarian types. Promising is the development of agroforestry, permitting to optimize the use of production potential of boreal zone with maintaining the global ecological role of the forests.

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SPECIES COMPOSITION AND RESTORATION OF FORESTS WITH DIFFERENT HISTORIES OF ECONOMIC USE

V. N. Korotkov[1]

ABSTRACT.--History of forest tracts of the Natural Historical Reserve "Gorki" (Moscow region, Russia) is revealed in the maps from the 18th and 19th centuries with use of a geographic information systems. The analysis has shown significant distinctions of floristic diversity of forest communities formed on sites with different histories of human impact. Long-term economic use of forests has resulted in significant loss of taxonomic and structural diversity. In those forests that have lost the capability for natural restoration, experiments on reconstruction of gap-mosaics are conducted by combining gap felling with regeneration cultures of species that occupied dominant positions in preagricultural forests.

The purpose of this work is to describe and evaluate the biodiversity and the successional state of forest ecosystems that have different histories of economic use but occur in similar edaphic conditions. The current state of forest communities and secondary succession processes were studied in "island" anthropogenic forest tracts of the Natural Historical Reserve "Gorki". The territory is located on the Moscow-Oka plain (in the south of the Moscow Region, Russia, geographical coordinates: 55°34' - 55°29' North and 37°45' - 37°53' East) within the limits of a polydominant spruce-broadleaved forest zone. The cover loam soils are on a carbonate moraine, which acts as the substrate for soil formation. The forests, are typical for the middle part of European Russia.

At the moment, the "island" forest tracts are predominantly composed of stands dominated by birch (62% of the forest area). Broad-leaved forests with prevalence of oak and lime cover about 14% of the forest area. Each of the forest tracts of the reserve has a unique combination of various tree species in communities that is related to the peculiarities of past economic use of the land (table 1).

HISTORY AND TREE SPECIES COMPOSITION OF FORESTS

The history of forest tract formation was investigated based on contemporary maps from the 18th and 19th centuries. The analysis of the history of economic use enabled us to divide the variety of current forest communities of the reserve "Gorki" into two groups: (1) forests formed as a result of repeated clearcutting on the lands and not disturbed by ploughing at least during the last 200-300 years; (2) forests arising as a result of regrowth on ploughed fields since the middle of the 19th century and then disturbed several times by clear cutting.

The forestry characteristics of timber stands with various histories of land use were analysed using a complex forest management information system (DBMS+GIS), created based on stand descriptions made in 1991 (an attribute database) and on the forest plans at a scale of 1:10,000 (a geographic information system).

In the forests that were cut many times on persistently non-arable areas, mixed timber stands prevail, with a dominance of broad-leaved species (fig. 1) while the area of pure timber stands is insignificant. The stands with populations of oak (*Quercus robur* L.) from 2 up to 6 units in the timber stand formula (20% to 60% of total stem density), occupy the greatest area. Lime (*Tilia cordata* Mill.) is rather common, and lime stands of moderate relative density (up to 4 units—40% of stems) with admixtures of other species occupy the greatest area. The low relative density of lime is accounted for by selective use of this species for large-scale harvesting of bast over a long time in past centuries (Kurnaev 1980). At the moment, lime is actively restoring its dominance owing to abundant seed renewal. A characteristic feature of these forests is also admixtures of maple (*Acer platanoides* L.), which is completely absent in the forests on former arable lands. Birch (*Betula pendula* Roth. & B. *pubescens* Ehrh.) is also widely distributed. However, the area under pure birch stands is insignificant (8.3%), and typically the share of birch reaches only two units (20%). Aspen (*Populus*

[1]*Vladimir N. Korotkov, All-Russian Research and Information Centre for Forest Resources, Novocheremuschkinskaya, 69A, Moscow, 117418, Russia, e-mail: Korot.v@g23.relcom.ru.*

Table 1.--*Spatial distribution of dominant trees in forests of the Natural Historical Reserve "Gorki".*

Dominant trees	Forest tracts									
	S'janovsky		Korobovsky		Bogdanovsky		Kazansky		Total	
	area (ha)	%	area (ha)	%	area (ha)	%	area (ha)	%	area (ha)	%
Betula pendula	342	56.2	321.4	48.5	638.3	74.6	189.3	67.8	1,491	62
Populus tremula	23.2	3.8	11.7	1.8	55.8	6.5	6.3	2.3	97	4
Alnus incana	11.7	1.9	0.2	0	23.3	2.7	0.5	0.2	35.7	1.5
Quercus robur	61.6	10.1	126	19	23.5	2.7	-	-	211.1	8.8
Tilia cordata	58.9	9.7	54.2	8.2	3.3[1]	0.4	3.3[1]	1.2	119.7	5
Pinus sylvestris[1]	59.9	9.8	82.7	12.5	46.1	5.4	50.7	18.2	239.4	10
Picea abies[1]	4	0.7	0.4	0.1	5.2	0.6	15.6	5.6	25.2	1
Larix europea[1]	0.6	0.1	5.6	0.8	0.6	0.1	2.3	0.8	9.1	0.4
Other species[1]	-	-	2.4	0.4	7	0.8	-	-	9.4	0.4
Stocked area	561.9	92.3	604.6	91.3	803.1	93.8	268	96.1	2,237.6	93
Unstocked area	47.1	7.7	57.4	8.7	52.9	6.2	11	3.9	168.4	7
Total area	609	100	662	100	856	100	279	100	2,406	100

[1] - forest plantation

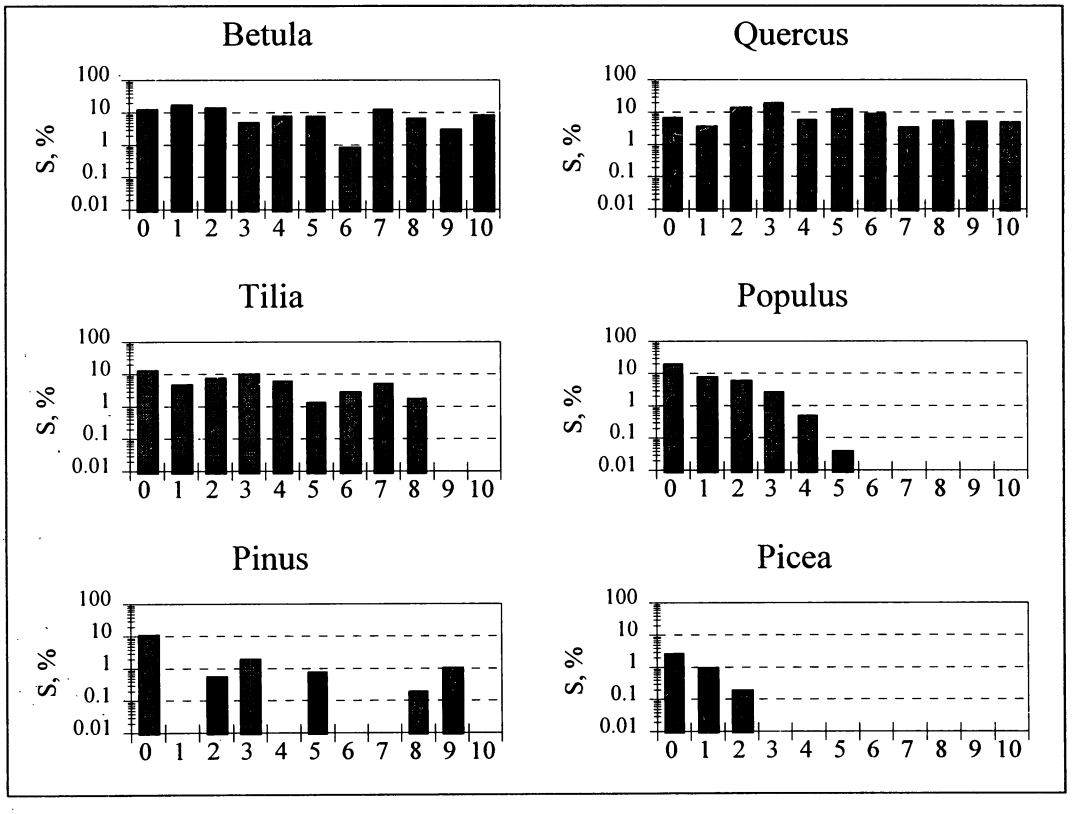

Fig. 1.--*Spatial and abundance distribution of tree species in timber stands over forests, which were cut many times on durably not arable areas. X-axis - share of tree species in a woody layer (10=100% relative density); Y-axis - area,%.*

tremula L.) and pine (*Pinus sylvestris* L.) occur rarely in the stands; thus, the greatest number of combinations is observed in stands with a share of these species varying between one and two units.

The analysis of species composition for the first group of stands has shown that repeated clearcutting resulted in the maximum prevalence of two species: birch and oak. The greatest variety of combinations of relative species densities, as represented by the timber stand formula, is formed only by these species. Thus, most of the area is in mixed stands of a variety of compositions, rather than in monocultures.

In the forests growing on the former arable lands, more than 80% of the area is occupied by birch stands, with the share of this species being equal to 7-10 units (fig. 2). Pure birch stands (species share of 10 units), occupy the greatest area (56.5%). The most usual admixture in birch stands is aspen. Stands with an aspen share of less than 2 units cover 50.9% of this area. On more than 30% of the area, birch stands include pine as an admixture.

For the forests on former arable land the minor presence of broad-leaved species other than birch and aspen is peculiar. The greatest share in timber stand composition corresponds to stands with an impurity of oak trees in the canopy layer (no more than 1 unit). Forests with an admixture of lime occupy an insignificant area; and among them stands with a solitary admixture of lime prevail (11.6% of the area). It is necessary to note that the stands dominated by oak or lime in the canopy layer occupy a very small area, and all of them are forest cultures.

The forests growing on former arable land contain about twice as much area of stands with coniferous species, i.e., pine and spruce (*Picea abies* (L.) Karst.) as forests on the areas that had not been used as arable land. This is because forest cultures of coniferous species were established mainly on arable land.

Analysis of the spatial distribution of stands with broad-leaved species in the canopy layer, in relation to the history of forest tract formation (fig. 3-4), confirms the inferences stated above and is of decisive importance when forecasting development of forest tracts in the near future.

SUCCESSIONAL STATE AND BIODIVERSITY

The geobotanical description and demographic analysis of tree and shrub species populations on 110 permanent and temporary sample quadrats of 0.04 to 1 ha, were used for the analysis of successional status of forest communities.

To estimate successional state and biodiversity of stands, the following characteristics (Zaugolnova *et al.* 1995) were used:

1. Degree of floristic diversity - a ratio between the number of species in the existing flora and the number of species of potential flora, expressed as a percentage. The potential flora includes all vascular plant species within the Russian non-chernozem zone, of which the ecological amplitudes of major environment factors are related to the ranges of estimated biotopes on ecological scales (D.N.Tsyganov 1983). This parameter was calculated separately for each sinusium.

2. Number of species in a 100 m^2 sample.

3. Share of participation of *Betula* sp. and *Populus tremula* (R-species) in the tree layer: the ratio between the projected cover of R-species and the total projected cover of all species within the tree canopy, expressed in percentage.

4. Degree of dominance: the ratio between projected cover of the most abundant species and total projected cover of species within a layer or a sinusium, expressed in percentage. This parameter was calculated separately for each sinusium.

5. Share of populations with complete ontogenetic spectra (all stages of development represented): the percentage of species with complete ontogenetic spectra relative to the total number of species in the sinusium. This parameter was calculated separately for sinusia of trees and shrubs.

6. Share of nemoral species (shade-tolerant understory species of broadleaf forests): the ratio between total projected cover of nemoral species and total projected cover of herbaceous species, expressed in percentage.

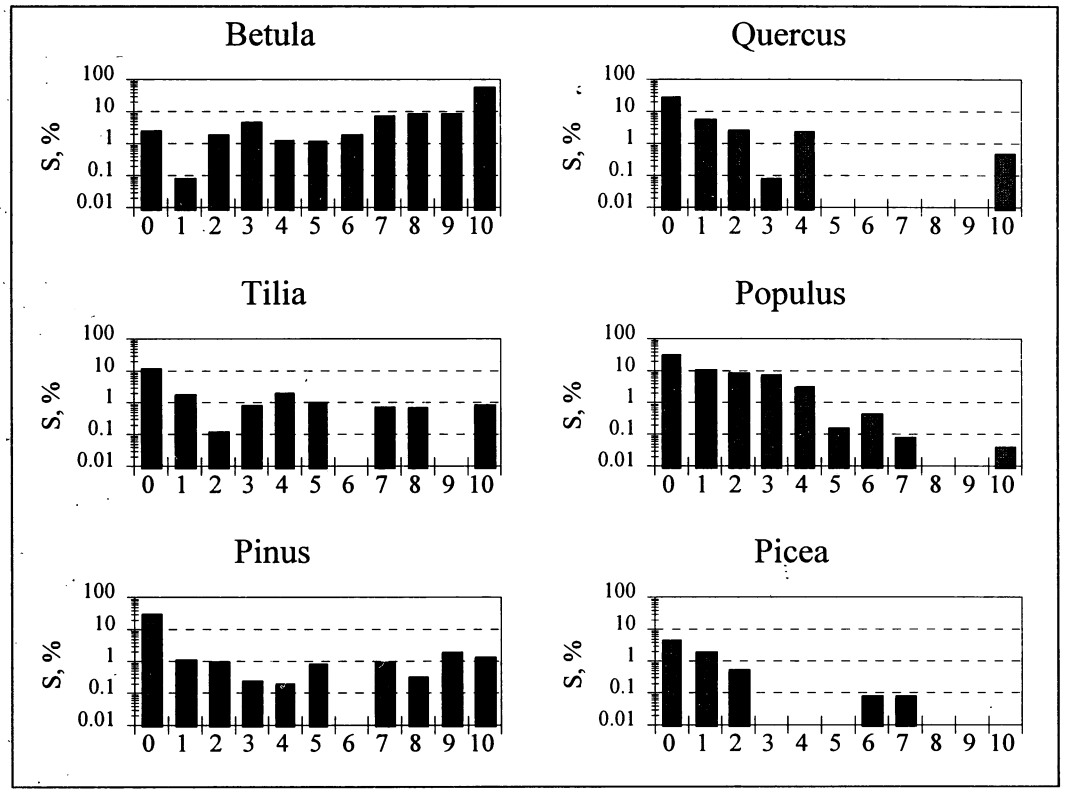

Fig. 2.--*Spatial and abundance distribution of tree species in timber stands over former arable land. X-axis - share of tree species in a woody layer(10=100% relative density); Y-axis - area,%.*

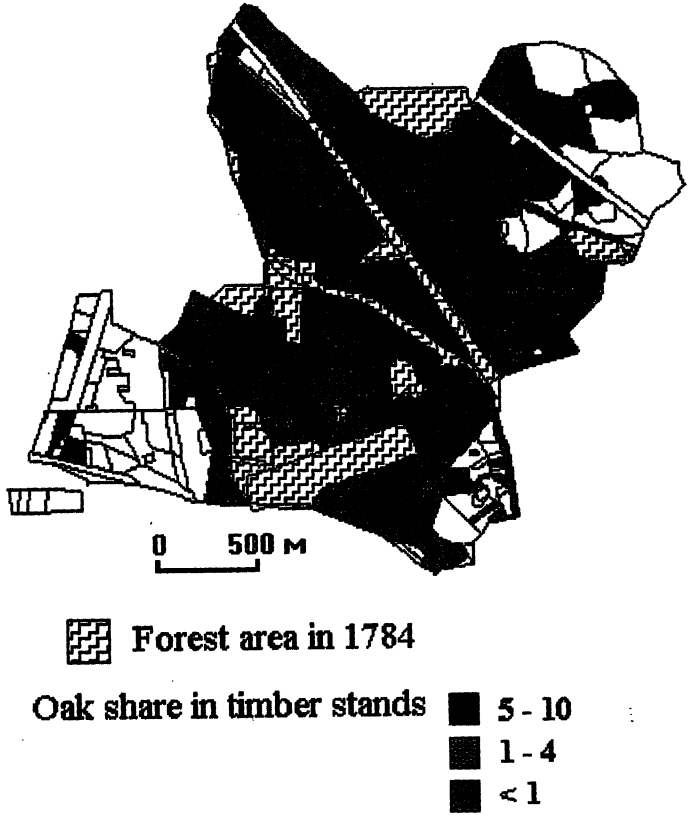

Fig. 3.--*Spatial distribution of forest contours with different abundance of Quercus robur in a woody layer in Korobovsky forest tract.*

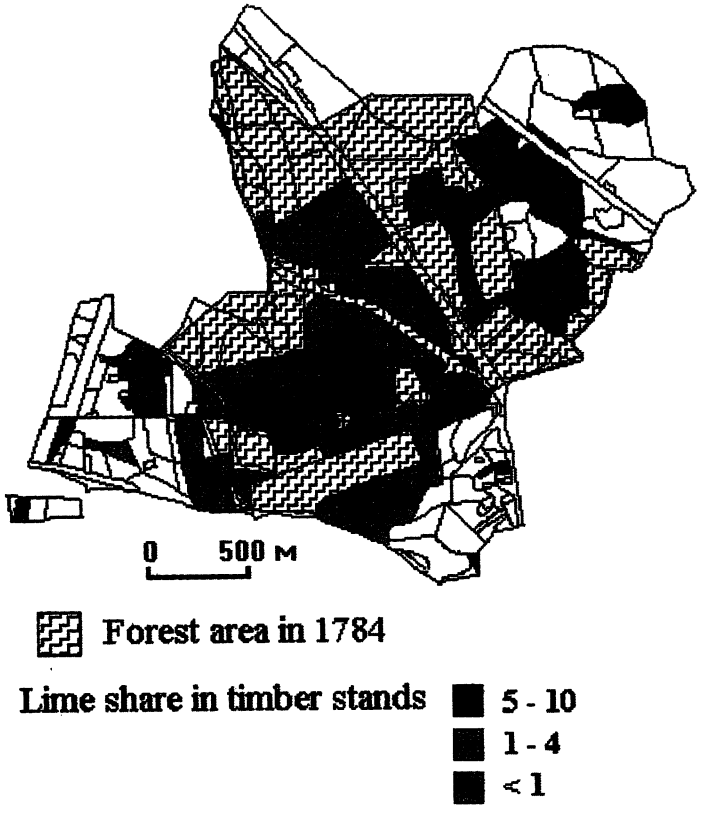

Fig. 4.--*Spatial distribution of forest contours with different abundance of Tilia cordata in a woody layer in Korobovsky forest.*

The estimation of ecological regimes on Tsyganov's (1983) ecological scales has not shown essential distinctions between the investigated forest ecotopes.

All variety of forest communities in the reserve "Gorki" occupying watershed sites, represent demutation variants of one floristic association - *Tilio-Carpinetum geranietosum* (Korotkov and Morozova 1988). The following demutation variants are distinguished: (1) those occurring on forest areas of long standing (broad-leaved forests with domination of lime and/or oak; nemoral birch forests); (2) those occurring on former arable land (birch forests rich in herbs - natural overgrowth, nemoral pine & spruce forests - forest cultures).

Researchers have shown that the present-day distinctions of successional states of forests (tree species composition, floristic diversity, ontogenic structure of tree species population) under similar ecotopic conditions are explained by different histories of land use.

The largest changes in species composition occur in the forests growing on arable areas. Mainly pure birch stands are formed there with a scanty share of nemoral species (table 3). In the birch forests a renewal of typical forest species of trees and shrubs takes place. The participation of nemoral species in the herbaceous cover depends also on the distance to broad-leaved stands, which serve as seed sources.

The woody and shrub species, which are capable of vegetative regeneration, are retained after clearcutting in forest communities on the lands not disturbed by ploughing (table 2). The herbaceous cover is retained, with dominance by nemoral species.

Predictions of development of the "island" forest tracts are made based on demographic analysis of woody and shrub species. It has been shown that the direction of forest secondary succession is determined by opportunities of seed dispersal and seedling establishment in those "island" forest tracts which have been shielded from active economic use.

The longterm economic use of forests has resulted in significant loss of taxonomic and structural diversity (table 2 and table 3). Therefore, it was necessary to develop special methods directed at restoration of forest ecosystems.

PRELIMINARY RESULTS OF EXPERIMENTAL WORK ON RESTORATION OF POLYDOMINANT UNEVEN-AGED BROAD-LEAVED FORESTS

One of the ways to find a compromise between economic needs and nature protection requirements is to apply group-selective cutting aimed at imitating the natural mosaic structure of uneven-aged forests (Methodical recommendations... 1989). Using group-selective cutting with subsequent planting of oak, the best results were obtained in secondary even-aged birch forests (Bogdanovsky forest tract). In birch forests, the density of tree and shrub regrowth has increased more than 10 times in 5 years after cutting of parent stands on plots of 1,200 m^2 to 2,000 m^2. At present, oak dominates the young growth (9,800 individuals per ha); the height of 8 year-old oak trees varies from 200cm to 320cm, the increase in height being from 120cm to 140cm.

Oak planting was done by separate biogroups, with each biogroup covering an area from 40 m^2 to 750 m^2. After 5 years, the tree crowns in the biogroups were closed, and it was necessary to perform annual weeding. The best results were observed when oak was established from seed.

Work on restoration of populations of other broad-leaved tree species *(Tilia cordata, Acer platanoides, Fraxinus excelsior)* is under way.

Application of the proposed cutting method does not reduce the expense of the subsequent tending of regrowth. At the same time, however, it creates favorable conditions for the restoration of plant populations in all sinusia. The number of species sharply increases in the depleted even-aged stands (in birch forests, from 35 - 40 to 50 - 60 species per 100 m^2.), and the growth of species diversity is occurring mainly due to natural species of light habitats.

CONCLUSION

The analysis has shown significant distinctions in floristic diversity of forest communities formed on territories with different histories of human impact. Longterm economic use of forests has resulted in significant loss of taxonomic and structural diversity. In those forests that have lost the capacity for natural restoration, experiments on reconstruction of gap-mosaics were conducted by combining gap felling

Table 2.--*Successional state of the biotopes formed on the historically non-arable forest areas as a result of repeated clearcutting.*

Biotopes	**Oak-forests**		**Nemoral Birch Forests**		
Forest tracts	**Koro-bovsky**	**S'janov-sky**	**Koro-bovsky**	**S'janov-sky**	**Bogdano-vsky**
Number of geobotanical. Descriptions	37	16	27	4	8
Total number of vascular plant species	89	84	78	69	57
TREE SYNUSIUM					
Share of populations with complete ontogenetic spectra,%	33.3	25	30	18.2	10
Share of R-species in tree layer,%	24.2	28.3	82.5	92.7	73.8
Degree of floristic diversity,%	80	53.3	66.7	68.8	66.7
Total number of species of all geobotanical. descriptions	12	8	10	11	10
Degree of dominance in tree layer,%	46.5	47.5	64.9	43.9	55.3
Average (maximum) number of species per 100 m^2	2 (5)	2.1 (4)	2.1 (4)	2.5 (4)	2.5 (4)
SHRUB SYNUSIUM					
Degree of floristic diversity,%	50	50	41.2	46.7	46.7
Total number of species of all geobotanical. descriptions	8	8	7	7	7
Average (maximum) number of species on 100 m^2	2.8 (7)	2.9 (7)	3 (5)	5.9 (7)	6.8 (7)
Share of population with complete ontogenetic spectra,%	50	50	57.1	71.4	71.4
Degree of dominance in shrub sinusium,%	82	38.4	84.9	42.5	30.3
HERB SINUSIUM					
Degree of floristic diversity,%	44.3	46.8	41.3	39.5	34.5
Share of nemoral species,%	47.8	47.4	48.3	30	44.6
Total number of species of all geobot. descriptions	62	65	57	47	40
Average (maximum) number of species per 100 m^2	14.6 (24)	23.4 (33)	15.4 (25)	19.4 (24)	26.3 (31)
Degree of dominance in herb sinusium,%	14.5	38.4	12.4	42.5	11.8

Table 3.--*Successional state of the biotopes formed on arable land.*

Biotopes	**Herbage Birch Forest**			**Pine Forests (plantation)**		
Forest tracts	**Koro-bovsky**	**S'janov-sky**	**Bogda-novsky**	**Koro-bovsky**	**S'janov -sky**	**Bogda-novsky**
Number of geobotanic descriptions	35	5	9	6	2	2
Total number of species	112	70	85	66	49	66
TREE SYNUSIUM						
Share of population with complete ontogenetic spectra,%	16,7	33,3	20	10	10	0
Share of R-species in tree layer,%	100	95.7	100	84.8	100	100
Degree of floristic diversity,%	80	60	66.7	62.5	62.5	68.8
Total number of species of all geobotanic descriptions	12	9	10	10	10	11
Degree of dominance in tree layer,%	66.4	53.1	67.7	67.3	68.7	56.2
SHRUB SYNUSIUM						
Degree of floristic diversity,%	56.3	50	41.2	56.3	62.5	56.3
Total number of species of all geobot. descriptions	9	8	7	9	10	9
Share of population with complete ontogenetic spectra,%	66.7	62.5	42.9	66.7	70	55.6
Degree of dominance in shrub sinusium,%	36.1	45.2	87.5	40.5	28	37.5
HERB SINUSIUM						
Degree of floristic diversity,%	55.1	43.3	41.5	31.1	24.1	28.4
Share of nemoral species,%	28.8	25.7	21.8	19.4	24.6	19.1
Total number of species of all geobotanic descriptions	81	52	63	42	27	40
Degree of dominance in herb sinusium,%	8.42	45.2	7.55	10.2	28	10.3

with regeneration cultures of species that occupied dominant positions in preagricultural forests.

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FOREST DYNAMICS, STRUCTURE, AND LANDSCAPE PATTERNS

A DIGITAL LANDCOVER MAP OF THE FORMER SOVIET UNION BASED UPON A TIME SERIES OF 15 KM RESOLUTION NOAA AVHRR DATA

T. A. Stone, R. A. Houghton, and P. Schlesinger[1]

ABSTRACT.--We have created a 15 km resolution digital landcover map of the region of the former Soviet Union based on a 10 year (1984-1993) time series of NOAA AVHRR Global Vegetation Index (GVI) data. We defined 60 phenology classes and then labeled them using the Olson World Ecosystems Map. We established 14 primary landcover types including water. The results from this research were compared with other, primarily Soviet or Russian, landcover classifications of the FSU. We also examined the classes in terms of the distribution of average monthly surface temperatures.

INTRODUCTION

The current landcover of the region of the former Soviet Union (FSU) is not well known by the international scientific community. Estimates of the area of forests and other landcovers vary widely. The reasons for this include not only the "cold war," but also the sheer size and geographical complexity of the region. Understanding the landcover and its changes are fundamental to understanding the global carbon cycle as the Former Soviet Union contains more forest (approximately 8 million km^2 in Russia alone) than any other country, and more forest soil carbon than any other country (Dixon *et al.* 1994). Also, the largest peatlands in the world are in Western Siberia, and probably more forest fires occur in Russia each year than in any other country in the world (Stocks 1991). Finally, the extraordinary amount of land at high latitudes also makes the FSU the region where the first effects of expected global warming are likely to occur, with some of the greatest effects on forests (IPCC 1995).

Our objective was to create a 15-km resolution landcover map of the region of the former Soviet Union. The new map was based on a 10 year time series of National Oceanographic and Atmospheric Administration (NOAA) Advanced Very High Resolution Radiometer (AVHRR) Global Vegetation Index (GVI) data.

DATA AND METHODS

The GVI data are a low-resolution equivalent to the Normalized Difference Vegetation Index or NDVI (Tucker *et al.* 1980, Holben *et al.* 1980) which is created using satellite data from the visible red and near-infrared portions of the spectrum. The visible red part of the spectrum (where plants absorb and use light for photosynthesis, where plants reflect light due to leaf cell structure and water content), has proven useful to understanding plant growth rates, estimating net primary productivity (NPP), and estimating green leaf biomass (Tucker *et al.* 1985). There is an extensive literature on the meaning and use of the NDVI (Tucker *et al.* 1980, Goward *et al.* 1991, Prince and Goward 1995, Holben *et al.* 1980).

The signal in GVI data is dominated by the response of vegetation to sunlight, temperature, and precipitation. Essentially, a time series of the data represents vegetation phenology unless clouds, low sun angle, or other atmospheric effects contaminate the data. Phenology, the study of the timing of recurring biological events, in our case was meant to include spring vegetation greenup, leaf out and maturation, and fall senescence. Reed *et al.* (1994) have provided a good overview of the cautions necessary in this type of work. Several other authors have reported using GVI data for creating large area landcover type maps (Goward *et al.* 1985, Townshend *et al.* 1987, Houghton *et al.* 1993, Stone *et al.* 1994). Bradley *et al.* (1994) and Gaston *et al.* (1997) have used similar data for their analysis of the former Soviet Union.

We assembled a 10-year GVI time series data set covering 1984 to 1993 for the region of former Soviet Union (FSU). Data from the NOAA First Generation Weekly Composite products were re-mapped from a Polar Stereographic to a Platte Carree map projection using the algorithms in Kidwell (1990). These data, with data from the NOAA second-generation weekly composite product, were read into a matrix of 25000 X 904 cells using both

[1]*Woods Hole Research Center, PO Box 296, Woods Hole, MA 02543, USA.*

IDRISI and ERDAS software formats. A total of 1.02 GB of data were used.

The inverse of the standard GVI equation (p. 15, Kidwell 1990) was used to convert the scaled NDVI data values to the bright-land byte range. For instance, the transformation converted an NDVI value of 0.0 to a new value of 100, creating a minimum and maximum range from 90 to 163. Thus, a pixel with a value of 163 is equivalent to an NDVI of 0.63.

The resulting data set had 48 to 53 weeks for each of the 10 years from 1984 to 1993. The data in these 510 weeks were then averaged into 120 months (12 months^{yr} for 10 years). The 120 monthly data were used to create a continuous FSU land surface, which crossed the international dateline (E. and W. longitude 180°) and had a size of 1,183 columns and 277 rows.

A 12 month dataset was created from the 10 year, 120 month dataset by averaging together, for example, all of the Januarys to create a 10-year January average. Then, Februarys and the following months were averaged to create an image with 12 bands. Averaging with maximum monthly GVI values reduces the effects of inter-satellite mis-calibration, sensor drift, orbital slippage, and noise in the data. These degrading effects upon AVHRR data have been thoroughly documented and are well known (Goward *et al.* 1991, 1993; Tateishi and Kajiwara 1991; Eastman and Fulk 1993). In general, using a long time series and a long averaging period reduces the amount of noise in the data.

Water pixels were deleted with a mask created from a re-projected hydrographic dataset digitized from a 1:8,000,000 scale Russian base map (Main Geodesy and Cartography Organization for Ministries of the USSR 1990). This hydrographic dataset, containing coastlines, islands, and major inland water bodies, was selected because of its size and ease of use. A comparison of coastlines between the Digital Chart of the World (DCW) by ESRI (1993) and the 1:8,000,000 scale map yielded an areal difference of less than 0.1 (10%) of a GVI pixel.

As most of the degrading effects in the data were related to low winter sun angles for this boreal region, we chose to eliminate from the analysis December, January, and February. These months are the times of extreme cold for the vast majority of the region of the FSU, during which vegetation is inactive.

Map Data

The following ancillary datasets were used either in the evaluation of the map produced or as a guide during its creation.

The 1990 Soviet Landcover and Potential Agriculture Map

The original version of this map (Institute of Geography 1990) was at 1:4,000,000 scale, with 374 classes of vegetation. The four panels of the map were digitized, merged, re-coded, and re-projected to a latitude and longitude map projection. The vegetation types were combined into 74 simpler classes.

The 1990 Forest Cover Map of the Soviet Union

This map (Garsia 1990) of dominant or economic forest types at 1:2,500,000 scale was partially digitized here. The remaining portions of the map were obtained from the World Conservation Monitoring Center (WCMC). Much of the work in digitizing was under the direction of Dr. Igor Lysenko at the Russian Research Institute for Nature Conservation, Moscow (Billington, pers. comm. 1996) and Moscow State University. We converted the map to a latitude - longitude projection. It consists of about 45 landcover classifications of which 36 are forest types. The remaining cover types include outcrops, tundra, open land, sea, water, and unclassified non-forest lands. The latter category is the largest single class, covering about 56% of the entire map. See table 1.

The 1973 Forest Cover Map of the Soviet Union

This map is from the 1973 Forest Atlas of the Soviet Union (State Committee on Forests 1973) and is at 1:15,000,000 scale. It describes 17 forest cover types. The cover types and areas of cover type are in table 2. It has been used as the basis for a forest carbon map of Russia (Schlesinger *et al.* 1997) described elsewhere in this volume (Stone *et al.* 1998).

Table 1.--*Areas of the landcover types according to the USSR Forest Cover Map (Garsia 1990) are in the 2nd column. As Arctic tundra is not distinguished in the map, lands above 60° N have been listed separately in the 3rd column. This helps in categorizing much of the "Open Lands" (Class 42) as northern open lands that are likely to be tundra. Also, we can see that the majority of the spruce, burned forest, and larch are above 60° N and that many species do not occur at higher latitudes.*

Class #	Area, km^2	Area N. of 60° N Lat.	% Total Area	% in N. Area	Map Name
1	1,287,810	533,130	5.9	41.4	Scots Pine (Pinus syl.)
2	5,300	4,770	<0.1	90.0	Sparse Pine
3	972,970	603,020	4.5	62.0	Spruce (Picea sib.)
4	2,410	600	<0.1	25.0	Sparse Spruce
5	121,390	9,180	0.6	7.6	Fir (Abies sib.)
6	1,000	-	-	-	Sparse Fir
7	119,680	32,220	0.6	26.9	Spruce and Fir
8	140	-	-	-	Sparse Spruce and Fir
9	3,456,400	1,960,400	16.0	56.7	Larch (Larix sp.)
10	88,050	55,260	0.4	62.8	Sparse Larch
11	404,960	133,870	1.9	33.1	Kedra (Pinus sib. & kor.)
12	1,310	990	<0.1	75.7	Sparse Kedra (Pine)
13	10,280	-	0.1	-	Artcha (Mtn. Cedar, Juniper)
14	133,100	-	0.6	-	Oak (Quercus sp.)
15	980	-	<0.1	-	Sparse Oak
16	40,130	-	0.2	-	Beech (Fagus sp.)
18	11,880	-	0.1	-	Hornbeam (Caprinus sp.)
19	82,550	110	0.4	0.1	"Stony" birch (Betula ermanii)
20	4,100	-	<0.1	-	Sparse Stony Birch
22	1,081,500	318,610	5.0	29.5	Birch
23	1,420	50	<0.1	3.3	Sparse Birch
25	141,000	13,680	0.7	9.7	Aspen (populus trem.)
26	120	-	<0.1	-	Sparse Aspen
27	15,490	-	0.1	-	Linden (tilia)
28	62,620	10,910	0.3	17.4	Other
29	4,650	-	<0.1	-	Other w/dom. Maple (Acer)
30	500	-	<0.1	-	Other w/dom. Walnut (Juglans)
31	1,770	-	<0.1	-	Other w/dom. Pistachio (Pistace.)
32	22,750	10,500	0.1	46.1	"Ernik" - Dwarf Arctic Birch
33	322,400	205,320	1.5	63.7	Cedar Elfin Wood (krumholz)
34	149,270	144,590	0.7	96.9	Dispersed Larch Wood
35	218,320	156,060	1.0	71.5	Burned Forest
36	27,080	17,190	0.1	63.5	Glades (cleared lands)
37	190,010	138,560	0.9	72.9	Tundra
38	267,400	50,650	1.2	18.9	Outcrops and Rocks
40	96,900	40,100	0.5	41.4	Tundra and Rocks
42	12,127,180	3,850,380	55.9	31.8	Open Lands
44	100	100	<0.1	100.0	Sea
45	213,130	54,660	1.0	25.7	Water
Total	21,688,070	8,344,890	100.0	-	
Forest	8,547,950				(Classes 1 to 34)
Non-Sparse Forest	8,295,830				

Table 2.--*Forest Cover based on the Atlas of Forests USSR (State Comm. On Forests 1973). Two methods of examining the data are shown in this table. First, the forest types and percentages in the left half of the table are from the summary page and pie chart in the Atlas. Second, the right half of the table shows the area estimates and percentage of forest cover based on our digitization of the map. The percentages of forest types shown by the two methods are in close agreement.*

Class	From original map graphic		From our digital version of the 1973 map				
#		% of forests		#	km^2	% of area	% of forests
1	Scots Pine	16.30	Scots Pine	1	1,371,040	14.70	15.20
2	Spruce	12.20	Spruce	2	974,840	10.50	10.80
3	Fir	1.80	Fir	3	134,070	1.40	1.50
4	Spruce Fir	--	Spruce Fir	4	117,420	1.30	1.30
5	Larch	38.10	Larch	5	3,345,150	36.00	37.00
6	Kedra (Pine spp.)	5.60	Kedra	6	357,820	3.80	4.00
7	Creeping Cedar	0.10	Creeping Cedar	7	23,130	0.30	0.30
8	Krumholz	3.80	Krumholz	8	296,090	3.20	3.30
9	Oak	1.40	Oak	9	181,540	2.00	2.00
10	Beech	0.50	Beech	10	64,940	0.70	0.70
12	Dwarf Birch	0.80	Dwarf Birch	11	96,840	1.00	1.10
16	Saxaul	--	Saxaul	12	681,560	7.30	7.50
11	Birch	13.30	Birch	13	1,054,390	11.30	11.70
13	Aspen	2.70	Aspen	14	115,590	1.20	1.30
			Water	16	259,100	2.80	-
			Sparse Scots. Pine	17	10,240	0.10	0.10
			Sparse Spruce	18	6,680	0.10	0.10
14	Lime	0.40	Lime	31	15,820	0.10	0.20
15	Other Softwoods	2.30	Other softwoods	33	102,720	1.10	1.10
17	Sep. & sparse trees	0.70	Sparse Larch	39	102,430	1.10	1.10
	Total	100.00	**Total Area**		9,311,330		100.00
			Total Forest		9,052,230		
			Non-Sparse Forest		8,932,880		

Olson et al. World Ecosystems

From the global data set of Olson *et al.* (1983, 1984), we extracted the region of the former Soviet Union that contains about 40 classes of land cover. See tables 3 and 4.

Thirty Year Average Temperature Data

A continuous FSU temperature surface was extracted from the Plate Carree projection matrix of Leemans and Cramer (1991). The temperature data were rounded to whole degrees Celsius. The original data then were organized using geographic coordinates in a global Plate Carree projection of 1/2-degree cells yielding an array of 720 columns by 360 rows. Of these data, an array of 720 columns by 259 rows was extracted from the region of 55° to 75° North. These extracted windows were resampled via a nearest-neighbor algorithm to an array consisting of 2,500 columns by 904 rows using IDRISI software to match the original array structure of NOAA GVI data. A Boolean mask was used to extract all former USSR territories. The mask was created by digitizing administrative boundaries from a 1:8 million-scale map of USSR Physical Geography. This raster mask and the corresponding temperature did not match exactly on the northern border, and left residual flag values of -999 from the original data.

Table 3.--*Landcover types of the former Soviet Union from Olson et al. (1983,1985). The ten largest categories are in bold face.*

Class	Cover Type	km^2
20	Snowy, Rainy, Coastal Conifers (with alder etc.)	11,550
21	**Main Boreal Conifers**	**3,281,160**
22	**Snowy, Non-Boreal Conifer Forest**	**758,550**
23	**Conifer/Deciduous, snow in winter**	**737,890**
24	Semi-Evergreen Deciduous, little or no snow	10,030
25	Temperate Deciduous Forest, snow in winter	154,150
26	Temperate Deciduous Forest, little or no snow	80,180
30	**Cool Farmland / Settlements (snowy)**	**1,270,630**
31	Mild Hot Farmlands / Settlements	541,900
37	Warm Irrigated crops	109,530
38	Cool Irrigated crops	74,740
39	Cold Crops and pasture	82,960
40	**Cool Snowy Grass Shrub**	**1,333,670**
41	**Mild/Warm/Hot Grass Shrub**	**1,295,210**
42	Cold steppe Meadow	322,280
44	Mire (acid bog &/or groundwater-fed fen) cold peatland/sphag.	568,510
45	Marsh / Swamp (warm-hot) salt & fresh water, thicket, flooded	9,350
46	Mediterranean Evergreen Tree/Shrub (winter rain)	540
47	Other Dry or Highland Scrub/Tree	62,130
48	Eucalyptus or Acacia, Quebracho, Saxaul	6,310
50	Sand Desert, partly blowing	523,590
51	Other Desert and Semi-desert	447,740
52	Cool/cold Shrub Semidesert/Steppe (sagebrush.)	496,200
53	**Tundra (polar, alpine)**	**3,083,370**
55	Grass/Crop +<40% woods: cool-cold, persistent snows	531,270
56	Regrowing Woods + Crop/Grass	12,440
57	**Regrowing Woods+Crop/Grass:cool-cold, persistent snows**	**967,770**
58	Grass/Crop+<40% woods:warm,hot	78,340
60	Southern Dry Taiga (and other aspen/birch, etc.)	533,330
61	Siberian Larch Taiga, with other taiga-Main Boreal Conifers	308,830
62	**Northern or Maritime taiga/tundra**	**2,586,910**
63	**Wooded tundra/margin (or mtn. Scrub, meadow)**	**703,010**
65	NW quadrant near most land (mainland, large island)	16,630
66	NE quadrant near most land (peninsula, small islands)	12,980
67	SE quadrant near most land (or isthmus)	2,820
68	SW quadrant near most land	12,0200
69	Polar Desert (rock lichens)	103,530
70	Glaciers (other polar and alpine)	11,020
71	Salt/soda flats (playas, lake flats, rarely wet)	24,740
	Total	21,167,790

Table 4.--*Recombination of Table 3 (from Olson et al. 1983,1985) into simpler categories to allow comparison with Table 6.*

Generalized Landcover Types	% of Area		Area (km^2)	
Mixed coniferous & deciduous forest	4.6		982,240	
Main boreal coniferous forests	15.5		3,281,160	
Northern or maritime taiga forests	13.7		2,907,290	
Snowy non-boreal coniferous forest	3.6		758,550	
Southern dry taiga forests	2.5		533,330	
Forested Subtotal		**40.0**		**8,462,560**
Cool to cold shrublands	5.0		1,064,710	
Cool snowy grasslands & crops	17.1		3,615,550	
Mild to hot grasslands, farmlands, & shrublands	14.2		3,021,030	
Regrowing woodlands & croplands	5.0		1,058,550	
Shrublnds, Grasslnds & Farmlnds Subtotal		**41.4**		**8,759,840**
Tundra	15.3		3,242,380	
Wooded tundra margin	3.3		703,010	
Tundra Subtotal		**18.6**		**3,945,390**
Total Area		**100.0**		**21,167,790**

We eliminated the temperature files for December, January, and February to conform to the 9 months used in the 60-class phenology product. We summed all of the months and divided by 9 to create a data set with a 9-month temporal average for each pixel location. We reclassified all no-data, and sea locations to 0. Then, we extracted the minimum, modal, and maximum temperatures for each of the 60 phenological land cover classes. Nine monthly images yielding 18 files included a minimum value file and a maximum value file by class for each month.

The Creation of GVI Data Phenology Classes

Using an ISODATA algorithm (ERDAS 1994), we clustered the 9-month GVI dataset into 60 classes using a maximum of 100 iterations and a 95% convergence threshold. From the clustered data, we then extracted the mean NDVI for each month for each of the 60 classes, and graphed the data against time (fig. 1). The distribution of the 9 month means were examined for the sum, standard deviation, coefficient of variation, minimum, median, and maximum values.

Clustering and classification of the GVI data did not create unique landcover labels, but rather created a series of classes based on phenology curves. The next step, labeling of the curves was more challenging than the numerical clustering of the data and required the use of ancillary data sets described above, expert opinion, and the development of a set of decision rules. The need to use very large ancillary data sets is common in work like this because of differences among the classifications (Nemani and Running 1997).

We labeled the 60-class map based on the Olson *et al.* (1983, 1985) World Ecosystems Map. The area covered by the 60-class map and the Olson *et al.* map were compared within the GIS. The label of the Olson *et al.* class that covered the majority of the phenology-based class was assigned to the phenology class. For instance, if the phenology class was composed of 60% spruce, 20% fir, and 10% tundra, the class would be called spruce. The second and third most important or sub-dominant Olson *et al.* labels (fir and tundra, in this case) were defined and used later in re-combinations of the classes.

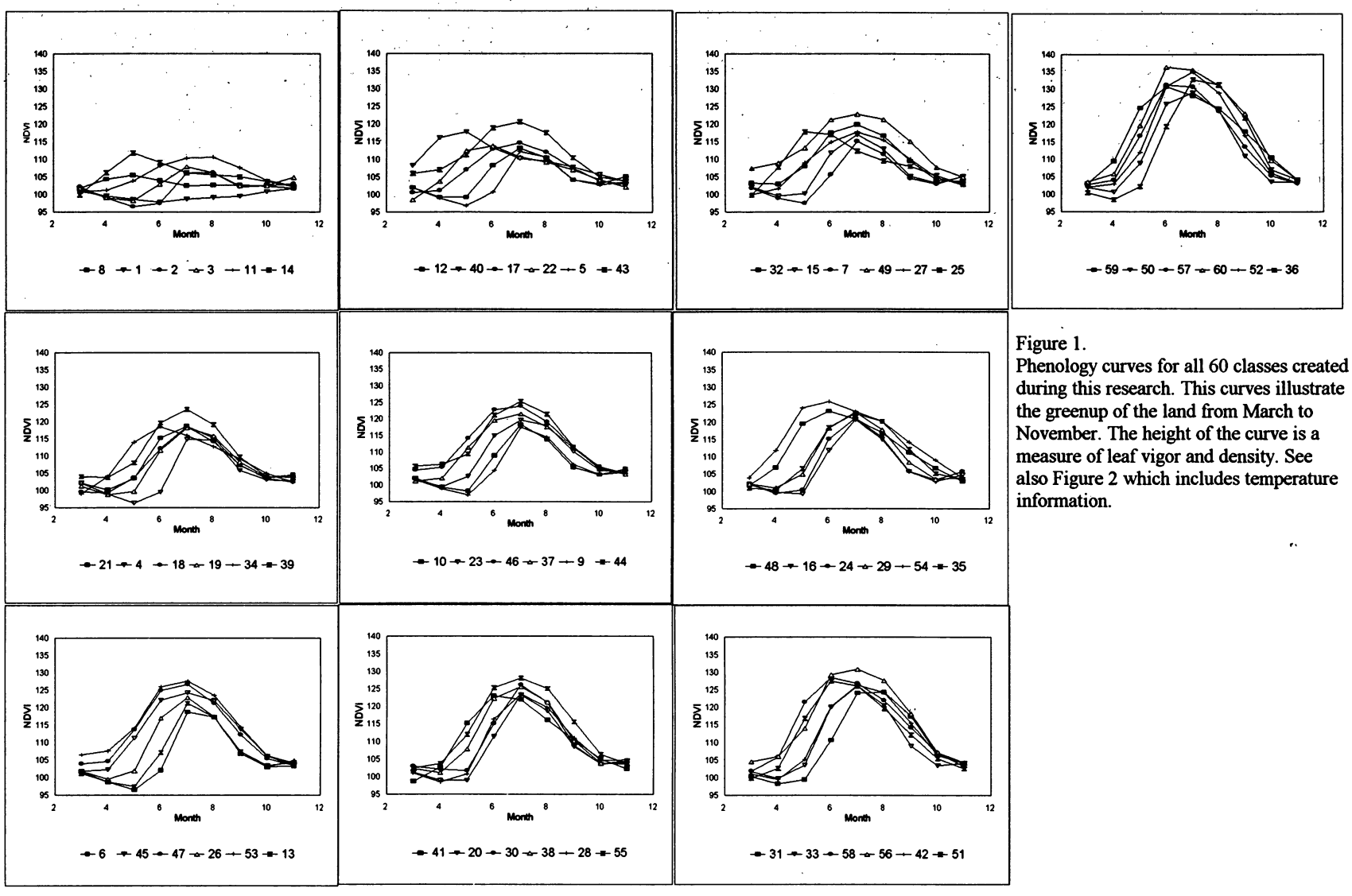

Figure 1.
Phenology curves for all 60 classes created during this research. This curves illustrate the greenup of the land from March to November. The height of the curve is a measure of leaf vigor and density. See also Figure 2 which includes temperature information.

Temperature and GVI

Nemani and Running (1997) used seasonal NDVI data from the NOAA AVHRR and compared it with temperature data to develop a general model for landcover vegetation. They hoped to develop classification descriptions that allowed labeling of classes independently of the large volumes of ancillary data required in earlier research (Loveland *et al.* 1991, Zhu 1994, Stone *et al.* 1994). Nemani and Running (1997) divided landcover into four broad types including (1) water limited (e.g., deserts and shrublands), (2) energy limited (e.g., high latitude and high altitude vegetation), (3) atmospherically coupled (e.g., rough canopied evergreen forests), and (4) atmospherically decoupled (e.g., crops that are fertilized, irrigated, and have smooth canopies). They also showed the seasonal distribution of various landcover types in NDVI and temperature space using data of various landcover types of the USA.

We used the concepts of Nemani and Running (1997) with the GVI data and the temperature data of the USSR from Leemans and Cramer (1991). We defined the July modal temperature value for each class and have plotted it versus mean class NDVI. Each of the 9 months of temperature data was imported into ERDAS Imagine NT. A 20 class Isodata unsupervised clustering operation with a maximum of 20 iterations and a 95% convergence threshold was run to create the 30-year temperature file. The resulting file contained 21 temperature zones. These were compared to the phenological land cover classes using ERDAS Imagine NT to derive a count by input class, where the largest value in each listing is the majority class. As July is the hottest month, generally, we defined the modal temperature value for each class and plotted it versus mean class NDVI (fig. 2).

RESULTS

We have made a 15 km resolution map the landcover of the region of the former Soviet Union based on a 10 year time series of GVI data (fig. 3). In general, we found 6.8 million km^2 of forested lands, 4.3 million km^2 of mixed forests and steppes, 6.5 million km^2 of shrublands, grasslands and farmlands and some 3.7 million km^2 of tundra. The names of the 60 classes have been assigned according to the World Ecosystems Map of Olson *et al.* (1983, 1985). Secondary and tertiary labels have been assigned also as more than one Olson-type was usually included in a phenology type. A simplification of the map based on the assigned primary Olson-type label reduced the number of classes to 12. Areal estimates of all landcovers of the FSU are shown in tables 5 and 6.

The addition of the temperature data to the GVI data allowed comparisons with the conceptual model of Nemani and Running (1997). Our effort has applied their concepts (but not methodology) to the largely boreal and forested realm of former USSR. The Nemani and Running (1997) conceptual model fit well with our landcover classification labels. Classes with the lowest mean NDVI and highest July temperatures appear in the water-limited region according to the Nemani and Running (1997) conceptual model. The geographic location of these classes (nos. 1, 9, & 11) were in Uzbekistan, Turkmentestan and Kazahkstan. Classes with very low mean NDVI and low July temperature should plot in the energy limited portion of the graph. Three sample classes here (nos. 2, 3 and 5) are very high latitude tundra or montane sites and include the Taymyr Peninsula. Classes closest to the atmospherically coupled portion of the graph include those that cover Kamchatka and the extreme Russian Far East near Magadan (nos. 31,42,36). These sites are moist, forested and are analogous to the US Pacific Northwest. Sites that are closest to the atmospherically decoupled extremes include the wheat belts of Ukraine, the northern steppes and Samara. Although this was not meant to be a rigorous evaluation of the Nemani Running work, their conceptual model appears to work and be useful in the largely boreal realm of the region of the FSU.

DISCUSSION

It is tempting but perhaps unwise to use the areal estimates of landcover type from this research and to compare these results with other estimates of the landcover or forest cover of the FSU. This result and the Gaston *et al.* (1997) effort are phenology-based maps. All other data sets were produced from different starting points, with different data sets, and with different goals. And, given the low resolution of the GVI data, all pixels will be mixed pixels. Also, very different landcovers may exhibit the same phenological characteristics in the same region and, thus, with this method would be labeled the same. Finally, the size of the FSU can also mean that regions several thousand miles apart with the same

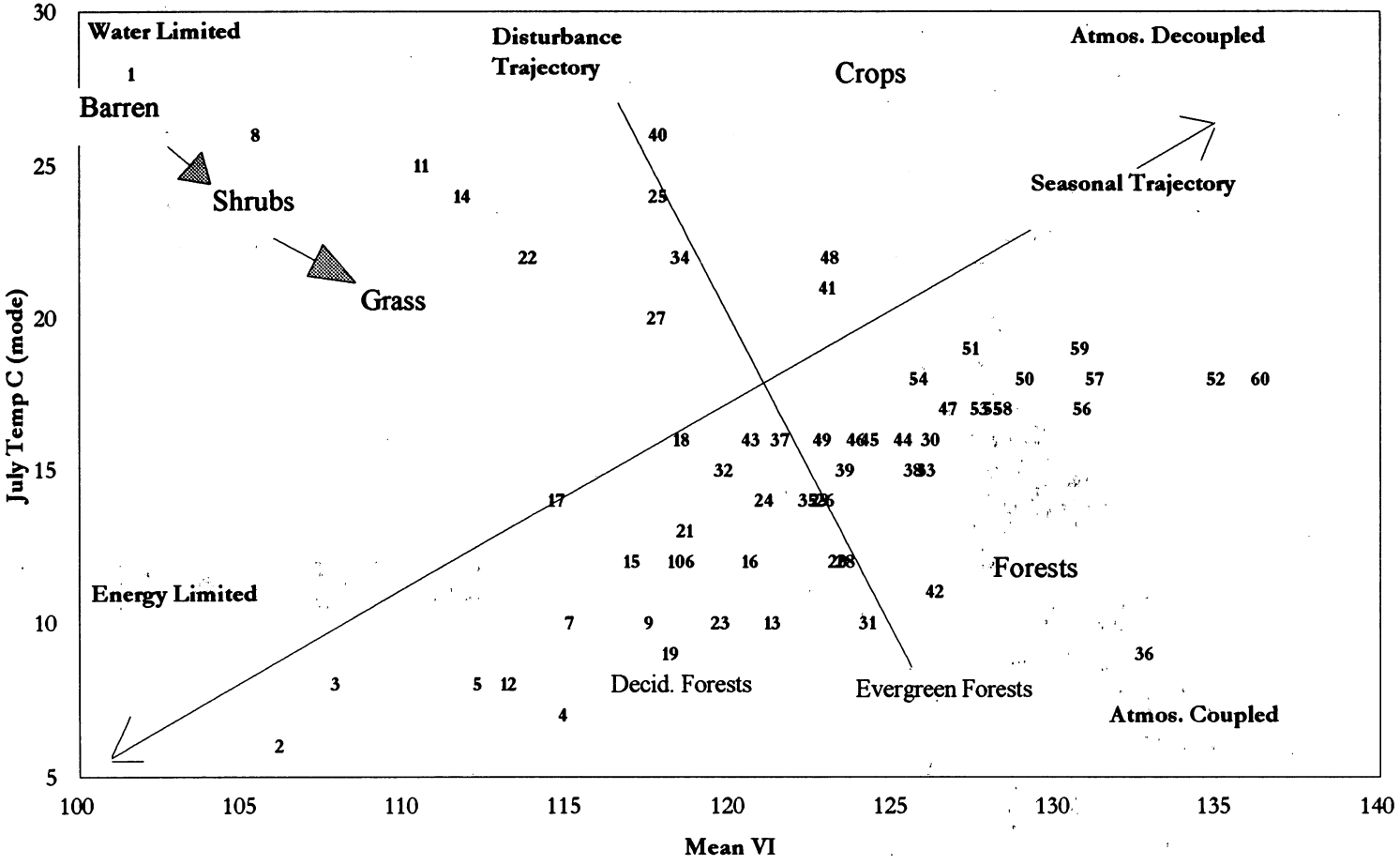

Figure 2. A plot of mean NDVI for the 60 phenology classes of the FSU vs. July temperature using the model of Nemani and Running (1997) which describes four broad land cover types of Water Limited, Energy Limited, and Atmospherically Coupled or Decoupled. See Table 5 for the composition of classes 1 to 60.

Figure 3. *Landcover classes of the former Soviet Union (FSU)*

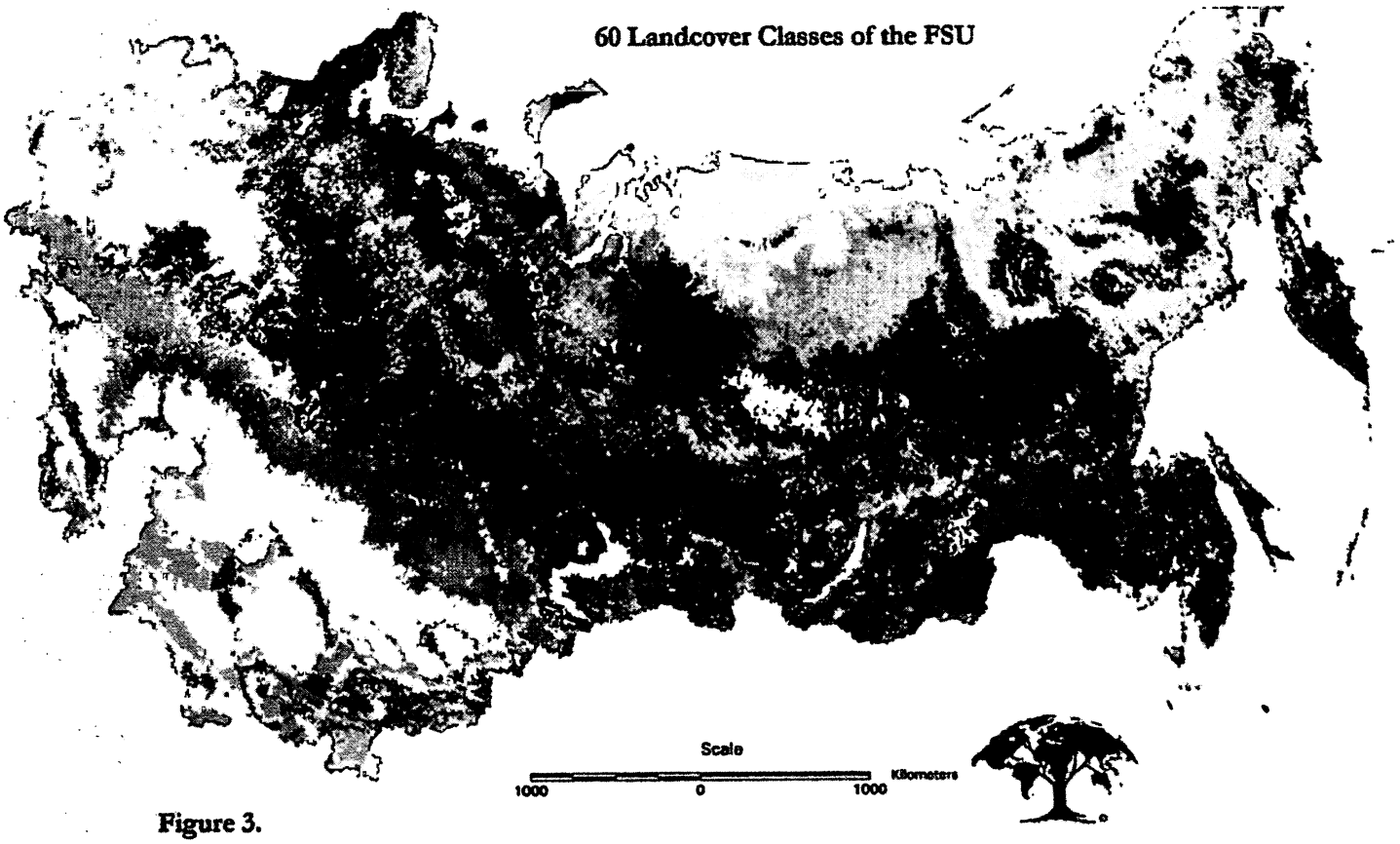

Figure 3.

Classes 1 to 20

- tundra, othor desert, sand dose
- tundra, polar desert,
- tundra, no/mar. taiga,
- tundra, polr desert, wood tund
- tundra, wood tundra, no or mar
- tundra, wood tundra,
- tundra, no/mar taiga, wood tund
- Mild/hot gr/shr, sand des,
- tundra, wood tundra, no/mar tai
- tundra, no/mar taiga, main bor
- Mild/hot gra/shr,tundra,sand do
- no/mar taiga,tundra,
- tundra, wooded tundra, no/mar
- mild/hot wrm gr/shr,cool/cold
- no/mar taiga,tundra, main bor
- no/mar taiga,tundra, main bor
- main bor con,cool snwy gr/sh,
- no/mar taiga,cool farms,cool
- no/mar taiga,tundra,wood tundra
- tundra, no/mar taiga, wood tund

Classes 21 to 40

- no/mar taiga,main bor for,tun
- cool/cold shr,cool snwy gr,mild
- no/mar taiga,tundra,main bor
- no/mar taiga,tundra,main bor
- cool/cold shr,mild/hot shr,col
- no/mar taiga,main bor conif,tun
- cool farms,cool snwy gr, main
- no/mar taiga,tundra,wood tundra
- main bor conif,no/mar taiga,tun
- no/mar taiga,main bor conif,tun
- wood tundra,tundra,main bor con
- main bor conif,tundra-taiga
- no/mar tai,main bor con,cold
- cool snwy gr, cool/cold shr,coo
- main bor conif,no/mar taiga,tun
- wood tundra,cold steppe,tundra
- main bor con,cool snwy gr,cool
- main bor con,cold step,no/mar
- main bor con, no/mar taig, tund
- mil/ht wrm gr/sh,mil/ht far,

Classes 41 to 60

- cool farms,cool snwy gr, gr/cr
- no/mr taiga,tundra,main bor con
- main bor con,snwy non-bor con,
- main bor con, so dry taig, tun
- main bor con,cool frms,sib l
- main bor con, conl/deci, mire
- main bor conf,conf/deci,mire
- mild/hot farms,regrw wods/cr,
- Snwy nonbor conf, so dry taig,
- main bor con, cold step,no/mar
- cool farms,gr/crp,col snwy gr
- regr wood/crp, so dry taig,cool
- so dry taig, snwy nonbor con,mn
- Mild/ht farms,coofarms-regr woo
- main bor con,sib lar taig,snw
- Snwy nonbor con,main bor con,
- con/deci,regr wod/crps,col sny
- regrw wood/crps,cool farms,gr
- cool farm,regrw wods/crp,gr/crp
- snow nonbor conf,regr wods/cr

Table 5.--*Map from this work with labels based on Olson et al. (1983,1985). Values rounded to nearest 100 km².*

Class #	(km²)	Primary label	Secondary	Tertiary
1	510,800	Tundra, confused with desert	Other desert	Sand desert
2	225,200	Tundra	Polar desert	Not present
3	289,500	Tundra	Northern/maritime taiga	Not present
4	312,800	Tundra	Polar desert	Wooded tundra
5	314,700	Tundra	Wooded tundra	Northern / maritime taiga
6	278,800	Tundra	Wooded tundra	-
7	208,700	Tundra	Northern/maritime taiga	Wooded tundra
8	1,063,100	Mild/hot warm grass/shrub	Sand desert	Other
9	314,300	Tundra	Wooded tundra	Northern / maritime taiga
10	341,600	Tundra	Northern/maritime taiga	Main boreal forest
11	133,600	Mild/hot grass/shrublands	Tundra	Sand desert
12	239,900	Northern/mar. taiga	Tundra	Not present
13	326,800	Tundra	Wooded tundra	Northern / maritime taiga
14	645,100	Mild/hot warm grass/shrub	Cool/cold shrublands	Sand desert
15	343,300	Northern/maritime taiga	Tundra	Main boreal conifers
16	384,400	Northern/maritime taiga	Tundra	Main boreal conifers
17	151,900	Main boreal conifers	Cool snowy grass/shrubs	Conifers
18	144,600	Northern/maritime taiga	Cool farmlands	Cool snowy grasslands
19	259,000	Northern/maritime taiga	Tundra	Wooded tundra
20	323,200	Tundra	Northern/maritime taiga	Wooded tundra
21	270,400	Northern/maritime taiga	Main boreal forest	Tundra
22	421,900	Cool/cold shrublands	Cool snowy grasslands	Mild hot warm grass/ shrub
23	270,400	Northern/maritime taiga	Tundra	Main boreal conifers
24	389,400	Northern/maritime taiga	Tundra	Main boreal forest
25	248,200	Cool/cold shrub	Mild/hot shrublands	Cool snowy grasslands
26	385,800	Northern/maritime taiga	Main boreal conifers	Tundra
27	277,100	Cool farmlands	Cool snowy grasslands	Main boreal forest
28	295,300	Northern/maritime taiga	Tundra	Wooded tundra
29	378,400	Main boreal conifers	Northern/maritime taiga	Tundra
30	216,100	Northern/maritime taiga	Main boreal conifers	Tundra
31	132,100	Wooded tundra	Tundra	Main boreal conifers.
32	200,200	Main boreal conifers	Tundra-taiga	Tie
33	290,700	Northern/maritime taiga	Main boreal conifers	Cold steppe
34	323,200	Cool snowy grasslands	Cool/cold shrublands	Cool
35	367,600	Main boreal conifers	Northern/maritime taiga	Tundra
36	143,600	Wooded tundra	Cold steppe	Tundra
37	285,100	Main boreal conifers	Cool snowy grasslands	Cool farmlands
38	438,200	Main boreal conifers	Cold steppe	Northern/maritime taiga
39	330,800	Main boreal conifers	Northern/maritime taiga	Tundra
40	184,900	Mild/hot warm grass/shrub	Mild/hot farmlands	Cool croplands
41	211,600	Cool farmlands	Cool snowy grasslands	Grasslands /croplands
42	197,900	Northern/mar. taiga	Tundra	Main boreal conifers
43	298,500	Main boreal conifers	Snowy non-boreal con	So dry taiga
44	324,100	Main boreal conifers	So dry taiga	Tundra
45	464,000	Main boreal conifers	Cool farms	Siberian boreal taiga
46	374,800	Main boreal conifers	Conif./decid. Forests	Mires
47	437,300	Main boreal conifers	Conif./decid. Forests	Mires
48	449,700	Mild/hot farmlands	Regrowing woods/crops	Conifers
49	409,700	Snowy nonboreal conifers	So dry taiga	Tundra
50	481,100	Main boreal conifers	Cold steppe	Northern/maritime taiga
51	520,900	Cool farmlands	Grassland / croplands	Cool snowy grasslands
52	470,400	Regrow. Woodlands/crops	So. dry taiga	Cool snowy grasslands
53	497,200	So dry taiga	Snowy nonboreal conif.	Main boreal forest
54	298,400	Mild/hot farmlands	Cool farms-regr wodlnds	Tie
55	581,200	Main boreal conifers	Sib. Larch taiga	Snowy non-boreal conif. forest
56	605,800	Snowy nonboreal conif	Main boreal conifer	Southern dry taiga
57	590,600	Conif/decid forest	Regrow wood/crops	Cool snowy grasslands
58	849,200	Regrow woodlands/crops	Cool farms	Grass/croplands
59	461,800	Cool farm	Regrw woods/crop	Conif./decid. forest
60	284,900	Snowy nonboreal conifers	Regrw woods/crop	Conif./decid. forest
	21,469,900	**Total**		

Table 6.--*Simplified landcover types defined for the FSU based on this research. The 12 types listed here are the major groupings of the 60 landcover types in the original classification (Table 5).*

Landcover Types	% Area	Area (km^2)
Mixed coniferous & deciduous forests	2.8	590,600
Main boreal coniferous forests	15.3	3,292,910
Northern or maritime taiga forests	5.4	1,162,940
Snowy non-boreal coniferous forests	6.1	1,300,400
Southern dry taiga forests	2.3	497,200
Forested Subtotal	**31.9**	**6,844,050**
Mixed main boreal forests & steppe/grasslands	8.5	1,820,300
Mixed northern or maritime taiga forests & steppe/grassland	11.8	2,524,130
Mixed Forest/Steppe Subtotal	20.3	4,344,430
Cool to cold shrublands	3.1	670,200
Cool snowy grasslands	1.5	323,200
Mild to hot grasslands, farmlands, & shrublands,	12.9	2,774,900
Regrowing woodlands & croplands	6.1	1,319,600
Cool farmlands & settlements	6.9	1,471,400
Shrublands, Grasslands, & Farmlands Subtotal	**30.6**	**6,559,300**
Tundra	16.1	3,446,500
Wooded tundra margin	1.3	275,700
Tundra Subtotal	**17.4**	**3,722,200**
Total Area		**21,469,900**

phenological characteristics might be labeled the same. The tremendous geographic extent of Russia and the FSU is not considered in the classification technique used. With this method, the spatial proximity of pixels was not considered. Instead, the month by month GVI values controlled the clustering of data. This can lead to classification of sites widely separated as the same landcover type simply because their phenological curves are the same. Having the same phenology does not mean having the same landcover. Also, small regions whose phenological cycle are determined by irrigation or by river waters may be out of phase with surrounding regions and are more likely to be mislabeled. The simple addition of temperature data assists in sorting out some of the potential difficulties. This new map is suitable for stratifying the region of the FSU using either the 12 or 60 classes version into regions which then could be sub-sampled by higher resolution satellite data such as that from NOAA AVHRR LAC, LANDSAT MSS, TM or SPOT.

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EFFECTS OF WITHIN-STAND HABITAT AND LANDSCAPE PATTERNS ON AVIAN DISTRIBUTION AND ABUNDANCE IN NORTHERN MINNESOTA

Carol W. Pearson[1] and Gerald R. Niemi[2]

ABSTRACT. -- We studied 26 mature aspen (*populus tremuloides*) stands to test whether both within-stand habitat characteristics and landscape patterns influenced breeding bird abundance in contiguous forests of the northern U.S. Breeding birds and habitats were sampled in 1991 and 1992. Proportions of different forest types surrounding each stand were calculated for three buffers and two bands of different radii from the sample stand boundaries (100 m, 250 m, 500 m). Models based on multiple regression analyses were calculated to predict the relative abundance of seven bird species. Five species, American Redstart (*Setophaga ruticilla*), Black throated Green Warbler (*Dendroica virens*), Magnolia Warbler (*D. magnolia*), Nashville Warbler (*Vermivora ruficapilla),* and Blackburnian Warbler (*D. fusca),* were correlated to varying degrees with at least one of the landscape-level variables. Two species, Veery (*Catharus fuscescens)* and Ovenbird (*Seirus aurocapillus)*, were correlated only with habitat variables measured within the sample stands. Species with the most general habitat associations within these forests were least influenced by landscapes, while species that were more specific (e.g., the highly conifer dependent Blackburnian Warbler) were most influenced by the surrounding landscape.

INTRODUCTION

Over the past 20 years, a great deal of research effort has been directed toward studying the effects of forest fragmentation on avian communities. Because of the similarities between true oceanic islands and isolated forest fragments or "habitual islands," many researchers have applied the theory of island biogeography (MacArthur and Wilson 1967) to the study of fragmented terrestrial habitats. Most of these studies have focused on patch size and degree of isolation to explain variations in the composition of avian communities within forest fragments (Galli *et al.* 1976, Whitcomb *et al.* 1977, Whitcomb 1977, Robbins 1979, Lynch and Whigham 1984, Blake and Karr 1984, Haila 1986). The importance of landscapes surrounding a forest stand have only recently been considered (Ambuel and Temple 1983, Wilcove 1985, Verner 1986, Addicott *et al.* 1987). Two habitat islands of the same size, shape, quality, and degree of isolation may be subject to different conditions depending on the landscape mosaic surrounding them.

Current research suggests that vegetation patterns in forest-dominated landscapes can effect the composition of avian communities within individual forest stands (Rosenberg and Raphael 1986, Urban *et al.* 1987, Hansen *et al.* 1991, Hejl 1992, Pearson 1994). The objective of this study was to determine the relative importance of vegetation patterns in the landscape surrounding a forest stand to avian abundance within the stand. To accomplish this objective, the study focused on two questions: (1) do correlations exist between the distribution or the abundance of selected species and habitat variables within the surrounding landscape; or (2) are the number of birds recorded within a forest stand correlated only with habitat variables within that stand?

STUDY AREAS AND BIRD SPECIES SELECTION

The study areas were 26 mature quaking aspen stands located on the Superior and Chippewa National Forests in northern Minnesota. The sample stands were a subset of forest stands selected as part of a National Forest Bird Monitoring Project (Hanowski and Niemi 1994). Stands of only one forest type were selected to reduce within-stand habitat variability and, hence, to increase the possible variation that could be attributed to differences in the landscape surrounding the stands. For more details concerning study area selection and locations see Pearson (1994).

Seven bird species were selected *a priori* for analysis using the following criteria: (1) species that were relatively abundant on the study sites, (2) species with habitat associations along a deciduous to

[1]*Carol W. Pearson, Natural Resources Research Institute, 5013 Miller Trunk Highway, Duluth, MN 55811; Present address: Minnesota Department of Natural Resources, 500 Lafayette Road, St. Paul, MN 55155-4025*

[2]*Gerald J. Niemi, Natural Resources Research Institute, 5013 Miller Trunk Highway, Duluth, MN 55811*

coniferous gradient, and (3) species classified as neotropical migrants. Habitat associations for birds of the region were identified using bird census data collected for a National Forest Bird Monitoring Project (Hanowski and Niemi 1994) from 1991 through 1993. Based on these criteria the following species were selected: Veery and American Redstart (deciduous forest association); Black throated Green Warbler, Ovenbird, and Magnolia Warbler (mixed forest association); and Nashville Warbler and Blackburnian Warbler (coniferous forest association).

DATA COLLECTION

Three replicate points (each 10 min in duration) were censused within each stand during a 2 week period in June of 1991 and 1992 (Reynolds *et al.* 1980, Hanowski and Niemi 1994). Five trained observers (Pearson 1994) conducted the census which were completed between 0.5 hours before and 4 hours after sunrise. Censuses were conducted only during good weather (i.e., wind<15 mph and no precipitation). All birds heard or seen within 100 m of each census point were recorded. The observers collected information on habitat structure and plant species composition at each census point after completing the bird census. The within-stand habitat data were averaged for the three points in each sample stand over all observers who had sampled them (Pearson 1994).

Aerial photo interpretation was used to analyze and classify the forest types within a 500 m radius of each sample stand. Types were classified to tree species where possible, and four size (age) classes were identified (Pearson 1994). The smallest area delineated by the aerial photo interpretation was about 0.1 ha. A Bausch & Lomb zoom transfer scope was used to delineate the results of the aerial photo interpretation on United States Geological Survey (USGS) 7.5 min topographic maps (scale - 1:24,000) for purposes of georeferencing, and to eliminate distortion in the arial photos caused by varying topography (Alberts 1992).

GIS AND DATA ANALYSIS

The photo interpretation results were digitized from the USGS topographic maps using PC ARC/INFO [Version 3.4 (ESRI 1991)] to create a digital map of the landscape surrounding (and including) each sample stand. The boundaries of the sample stands were determined by locating the three census points on the larger landscape maps and using GIS techniques to delineate the area within 100 m of the census points (the radius within which birds were recorded). Buffers of three different radii (100 m, 250 m, and 500 m) from the sample stand boundaries were then delineated. The areas, perimeters, and frequencies of forest types were calculated for each sample stand and for each of the three buffers for each stand.

The landscape matrix data derived from the GIS analysis were imported into Paradox (Version 3.0), and combined with the within-stand habitat data. Two new data sets were created by subtracting the areas of the 250 m buffer from the areas of the 500 m buffer, and by subtracting the areas of the 100 m buffer from the areas of the 250 m buffer. These areas, designated as the "250 m band" and the "500 m band," represented the matrix habitat in a band between 100 m and 250 m away from the sample stand, and between 250 m and 500 m away from the sample stand, respectively. Proportions of different habitat types within the sample stands, buffers, and bands were calculated by dividing by total area of each.

All subsequent data analyses were completed using SAS/PC, Release 6.04 (SAS 1988). The data set described above contained more than 100 independent variables measured for each sample stand. To reduce the number of variables, similar age classes and habitat types were combined into broader categories. For example, two age classes representing regenerating forests<20 year old were combined. Similarly, all mature upland deciduous types were combined into one category. In this manner, the landscape-level forest types were reduced to 10 variables representing broader habitat categories (Pearson 1994). At this point, with landscape cover types and within-stand habitat variables, 32 independent variables remained in the data set. Two approaches were used to further reduce the number of independent variables. First, simple correlations between variables were examined, and one of each pair correlated (r>0.50) variables was eliminated by flipping a coin. Secondly, the proportion of total habitat represented by each of the matrix habitat variables within the sample stands and in each of the bands and buffers we examined. Because some habitat types were present in small proportions, a habitat variable was eliminated if fewer than 10 stands had 10% of that habitat type. After these procedures were completed 10 independent variables were retained for the multiple regression analysis (table 1).

Table 1.--*Description of habitat variables retained as independent variables for multiple regression analyses.*

Within-stand Variables	
cht	= canopy height (in meters)
hcc	= percent high canopy cover code[1]
uc	= percent understory (1m to 4m) cover code[1]
gc	= percent ground (up to 1m) cover code[1]
bals	= frequency of balsam recorded at the census points (summed for three points and averaged over two years)
bir	= frequency of birch recorded at the census points (summed for three points and averaged over two years)
mixin	= percent mature upland mixed forest (25 - 75% deciduous) within sample stands
Matrix Variables	
mupdec	= percent mature upland deciduous forest - includes pole-sized and saw-sized classes of all upland deciduous forest types classified as > 75% deciduous
mupmix	= percent mature upland mixed forest - includes pole-sized and saw-sized classes of all upland mixed forest types classified as 25 - 75% deciduous
mupcon	= percent mature upland conifer forest - includes pole-sized and saw-sized classes of all upland conifer forest types classified as < 25% deciduous

[1] % cover codes: 0 = <10%, 1 = 10-40%, 2 = 40-70%, 3 = >70%

Multiple regression analyses were completed separately for three buffers and two bands for each of the seven species. Dependent variables consisted of the census data collected for each species on the sample stands. The census data were summed for the three census points in each stand and averaged over the two years of censusing. The independent variables consisted of seven within-stand habitat variables and three matrix habitat variables (table 1). Matrix habitat variables (which were proportions) were arcsine transformed, and bird counts were logarithmically transformed to stabilize the variance. The SAS/PC "maxR" regression method (SAS 1988) was used to select the best subsets of variables to explain variation in numbers of birds recorded on the sample stands. Residuals were tested for normality and residual plots were examined to identify outliers and non-linear trends. For a more detailed description of methods, see Pearson (1994).

RESULTS

A total of 1,605 individuals of 66 species were recorded on the 26 sample stands over the 2 years of censusing (Pearson 1994). The most abundant species recorded on the study sites was the Ovenbird, which was present on all of the sites in both years. The Veery and the Nashville Warbler were also very common and occurred on most sites in both years. The black-throated Breen Warbler, Magnolia Warbler, Blackburnian Warbler and American Redstart were the least abundant of the species chosen for analysis.

Table 2 lists the proportions of mature upland mixed forest within the sample stands and the mean values for habitat structure and tree species recorded at the census point. Since both balsam (*Abies balsamea*) and birch (*Betula papyrifera*) appear frequently in the regression models presented below, an explanation of the forest associations of these two tree species is important in interpreting the results. The frequencies of both balsam and birch were negatively correlated (r=-.061 and r=-.36, respectively) with the mean percent deciduous code recorded at the census points. This suggests that both of these tree species were assigned with an increasing conifer component within the stands selected for this study. Mature upland forest types represented the largest proportion of the landscape matrix surrounding the study sites. Lowland habitat types (e.g., marsh, water, lowland forest), agricultural, and urban areas were poorly represented.

Because the results were very similar in all bands and buffers, only the "best" regression model for each species is presented (table 3). The "best" model was defined as a statistically significant model (P,0.015), with the highest adjusted R^2 (adjusted for the number of variables included) of the models tested, in which all of the variables included in the model were also significant ($P<0.05$). Comparisons of the results for the different bands and buffers are shown in figure 1.

Two of the seven species, Veery and Ovenbird, were not correlated with any of the matrix habitat variables. Because these species apparently responded only to within-stand habitat variables, the results were identical for all the buffers and bands (fig. 1 a, d). Significant negative correlations were found between the abundance of the Veery and percent high canopy cover within the forest stand. Veery abundance was also significantly negatively correlated with the frequency of balsam recorded with the study sites. The model explained 52% of the variation in Veery numbers recorded on the sample stands (table 3). Ovenbirds were significantly correlated with three within-stand habitat variables. The species was positively correlated with percent high canopy cover and percent ground cover, and negatively correlated with percent understory cover. The model including these three variables explained 46% of the total variation in Ovenbird abundance (table 3).

Five species (American Redstart, Black-throated Green Warbler, Magnolia Warbler, Nashville Warbler, and Blackburnian Warbler) were correlated wih matrix habitat variables as well as within-stand variables. The proportion of variation explained by the matrix variables (partial correlation coefficient) varied among the five species, and between the different bands and buffers for individual species (fig. 1).

In the 250 m band, the 250 m buffer, and the 500 m buffer, the abundance of American Redstarts was negatively correlated with the frequency of birch (recorded within the stand), and with percent mature upland mixed forest in the landscape surrounding the stand. In the 100 m buffer and the 500 m band, balsam was the only variable significantly correlated (negatively) with Redstart abundance. Model R^2 values ranged from 0.19 to 0.28, and the partial correlation coefficients for percent mature upland mixed forest ranged from 0.17 to 0.19 (fig. 1 b).

The Black-throated Green Warbler showed significant positive correlations with canopy height in the 100 m buffer, the 250 m band, and the 250 m

Table 2.-- *Means for habitat variables, frequencies (freq) of tree species, and proportions of mature upland mixed forest within sample stands.*

Site Name[1]	Canopy Height (m)	High Canopy Cover[2]	Understory Cover[2]	Ground Cover[2]	Freq of Balsam[3]	Freq of Birch[3]	Mature Upland Mixed Forest[4]
BlkDuck 47_8	20.8	2.00	3.00	3.00	0.50	2.00	0.05
CassLk 110_33	23.8	2.50	2.83	2.83	0.50	1.50	0.00
CassLk 16_21	21.5	2.83	2.83	2.67	0.00	0.50	0.13
DeerRiv 4_36	17.1	2.50	1.83	2.67	2.00	1.50	0.00
Kawish 103_8	15.7	1.00	2.44	2.89	1.50	1.50	0.22
Kawish 144_4	23.2	1.56	3.00	2.67	0.00	3.00	0.01
Kawish 146_14	22.3	2.33	2.78	2.78	0.00	2.50	0.17
LaCroix 109_98	25.1	2.33	2.11	2.56	3.00	1.00	0.27
LaCroix 110_57	17.3	1.83	3.00	3.00	1.00	3.00	0.02
LaCroix 110_78	16.8	2.67	2.00	3.00	3.00	2.50	0.21
LaCroix 130_17	22.3	2.11	1.89	2.67	2.50	1.00	0.26
LaCroix 130_9	19.2	2.00	2.33	3.00	3.00	2.50	0.13
LaCroix 135_7	19.0	1.67	1.67	3.00	1.50	1.50	0.20
LaCroix 203_94	24.7	1.83	2.83	3.00	0.00	1.00	0.25
LaCroix 204_30	20.7	2.33	2.00	2.78	2.50	3.00	0.13
LaCroix 42_30	19.8	1.56	2.44	2.89	0.50	2.00	0.10
LaCroix 92_108	14.0	2.20	2.60	2.80	2.00	2.00	0.34
LaCroix 97_46	15.4	2.00	2.44	2.89	1.00	3.00	0.20
Tofte 79_10	11.7	1.44	1.56	2.33	3.00	2.50	0.02
Tofte 86_26	17.3	1.00	2.67	2.44	2.50	2.50	0.04
Tofte 86_27	20.8	1.44	2.67	2.22	3.00	3.00	0.00
Walker 103_32	20.1	2.22	2.67	3.00	0.00	1.00	0.00
Walker 10_30	14.2	1.83	2.67	2.67	0.00	2.00	0.00
Walker 21_21	23.1	2.20	2.60	2.07	0.00	0.50	0.00
Walker 8_17	17.4	2.00	2.17	1.67	0.00	2.50	0.01
Walker 93_11	20.8	2.67	2.50	2.83	0.50	1.50	0.10

[1] Site Name = District, Compartment_Stand

[2] % Cover values: 0 = <10%, 1 = 10-40%, 2 = 40-70%, 3 = >70%

[3] Frequency summed for three points and averaged over two years

[4] Proportion of mature upland mixed forest identified by aerial photo interpretation

Table 3.--*"Best" multiple regression models for individual species. Variables included in the models are significant at P < 0.05. "Matrix R^2" is the proportion of total variation explained by landscape-level variables. Variables: bals = balsam, bir = birch, gc = percent ground cover, hcc = percent high canopy cover, uc = percent understory cover, mupmix = mature upland mixed forest, and mupcon = mature upland conifer forest.*

Species	Regression Model	Buffer or Band	F	P	Model R^2	Adj R^2	Matrix R^2
DECIDUOUS-ASSOCIATED SPECIES							
Veery =	1.91 - 0.33 hcc - 0.27 bals	All	14.55	0.0001	0.56	0.52	0.00
American Redstart =	1.29 - 0.30 bir - 2.71 mupmix	250 m Band	5.97	0.0082	0.34	0.28	0.19
MIXED-ASSOCIATED SPECIES							
Black-throated Green Warbler =	- 1.31 + 1.84 mupcon + 0.32 bir + 0.40 hcc	500 m Band	4.44	0.0138	0.38	0.29	0.18
Ovenbird =	+ 1.04 + 0.37 hcc - 0.46 uc + 0.42 gc	All	8.20	0.0008	0.53	0.46	0.00
Magnolia Warbler =	- 0.47 - 0.36 hcc + 0.15 bals + 0.46 uc + 1.11 mupcon	500 m Buffer	8.61	0.0003	0.62	0.55	0.08
CONIFER-ASSOCIATED SPECIES							
Nashville Warbler =	- 0.50 + 0.45 bir + 3.20 mupmix	500 m Buffer	8.28	0.0020	0.42	0.37	0.24
Blackburnian Warbler =	+ 0.04 + 0.14 bals + 3.01 mupcon	500 m Buffer	13.66	0.0001	0.54	0.50	0.40

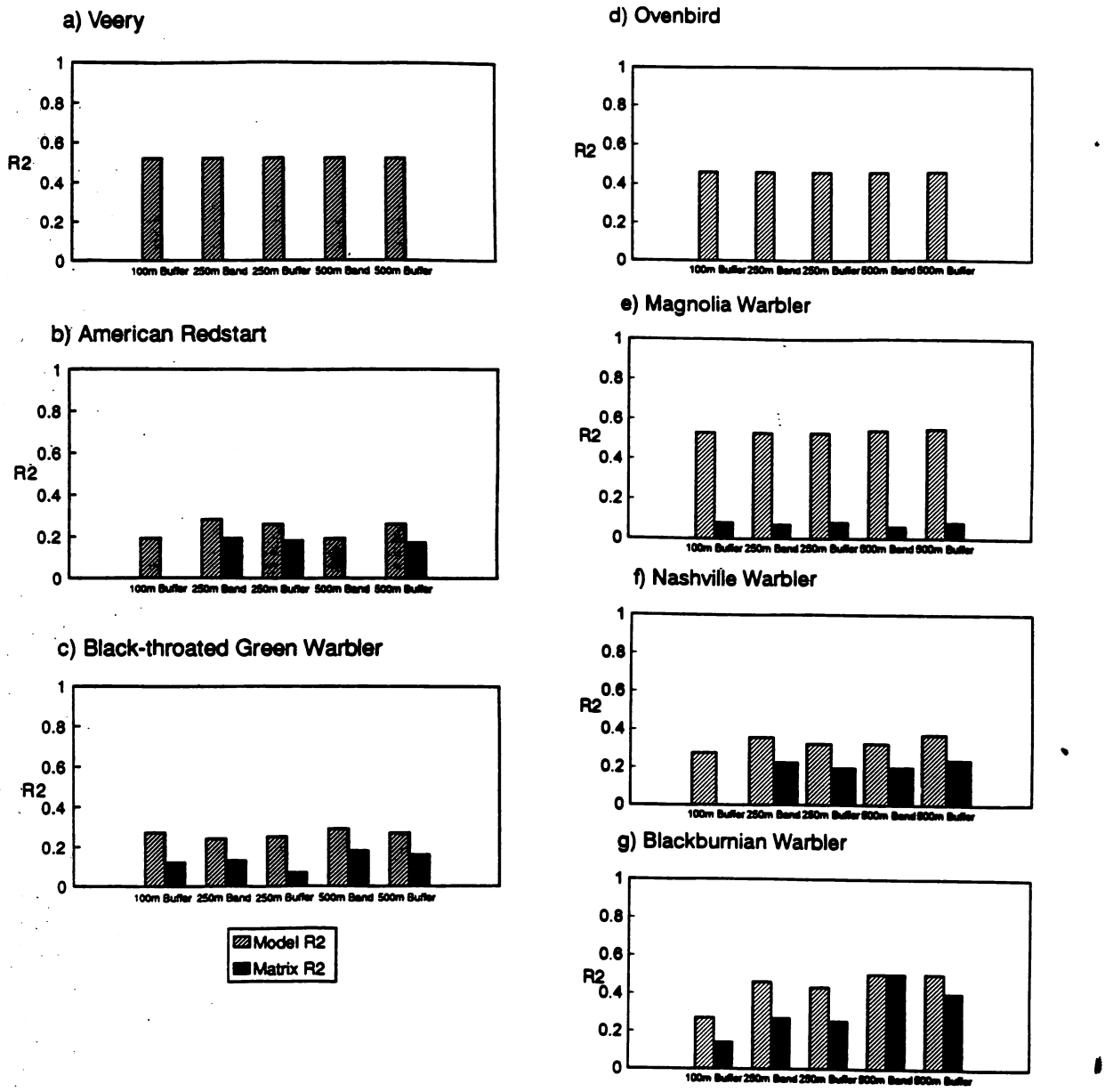

Figure 1.--*Comparison of results between the different bands and buffers for seven individual species: a) Veery, b) American Redstart, c) Black-throated Green Warbler, d) Ovenbird, e) Magnolia Warbler, f) Nashville Warbler, and g) Blackburnian Warbler. "Model R^2" is the adjusted R^2 value for the complete model. "Matrix R^2" is the proportion of total variation explained by landscape-level variables alone.*

buffer. It was significantly positively correlated with birch in all bands and buffers. There were significant negative correlations with percent understory cover in the 100 m and 250 m buffers, and in the 250 m band. In the 500 m band and the 500 m buffer, there were significant positive correlations with percent high canopy cover. Percent mature upland deciduous forest in the surrounding landscape was negatively correlated with the abundance of this species in the 100 m and 250 m buffers. In the remaining buffer and bands, there were positive correlations with percent mature upland conifer in the landscape matrix. Model R^2 values ranged from 0.24 to 0.29, and partial correlation coefficients for the matrix variables ranged from 0.07 to 0.18 (fig. 1c).

The Magnolia Warbler was negatively correlated with percent high canopy cover in all the bands and buffers. It was positively correlated with percent understory cover and frequency of balsam recorded within the stands, and with percent mature upland conifer forest in the landscape matrix in all bands and buffers. Model R^2 values ranged from 0.53 to 0.55, and partial correlation coefficients for mature upland conifer forest in the surrounding landscape ranged from 0.06 to 0.08 (fig 1e).

In the 100 m buffer, the abundance of Nashville Warblers was positively correlated with the frequency of balsam and birch recorded within the stand. In the other buffers and in both bands, there was a positive correlation with birch and with the proportion of mature upland mixed forest in the landscape matrix. Model R^2 values ranged from 0.33 to 0.42 and the landscape-level variable explained from 0%-33% of the total variation in abundance for this species (fig. 1f).

Significant positive correlations were found in all bands and buffers for the Blackburnian Warbler with balsam (within the stand) and percent mature upland conifer forest in the surrounding landscape. The models including these two variables explained form 27%-50% of the variation in abundance of this species within the sample stands (fig. 1g.). Partial R^2 values for mature upland conifer forest in the surrounding landscape ranged from 0.14-0.40 (fig. 1g.) In the 500 m band, a model which explained 50% of the variation included two landscape-level variables, percent mature upland conifer, and percent mature upland mixed forest. No within-stand habitat variables were significant.

DISCUSSION

Studies conducted in the eastern United States have documented the detrimental effects of fragmentation on area sensitive species, many of which are neotropical migrants (Galli *et al.* 1976; Robbins 1979; Whitcomb *et al.* 1981; Blake and Karr 1984; Robbins *et al.* 1989). In northeastern Minnesota, however, isolated forest patches are rare, and the concept of fragmentation is not as clear. In this region, most contemporary forest changes are created by logging. Timber harvest has produced a regional landscape mosaic of different forest types and ages. The effects of these landscape patterns on avian distribution and abundance have not been studied previously. The results of the study presented here offer some preliminary answers to the question of how different habitat types in the landscape matrix may affect avian populations within individual forest stands. Two of the species tested in this study, the Ovenbird and the Veery, responded only to within-stand habitat variables. The remaining five species- American Redstart, Black-throated Green Warbler, Magnolia Warbler, Nashville Warbler, and Blackburnian Warbler - were correlated to varying degrees with landscape-level variables. These are all species that appear to exhibit more specific forest type associations in this region that the Veery and the Ovenbird.

Although the Veery is generally classified as a deciduous-associated species, a review of the literature suggests that it is found in a wide variety of both deciduous and mixed forests. In the Michigan Breeding Bird Atlas Habitat Survey, most Veeries were found in moist forest types, generally northern hardwoods, but often with a conifer component (Brewer *et al.* 1991). On the Chippewa National Forest in northern Minnesota, Veeries were most abundant in mixed stands of mature balsam fir, aspen, and paper birch. They were also recorded in mature oak stands, jack pine saplings, and aspen saplings (Hanowski and Niemi 1991). Probst *et al.* (1992) found the Veery to be common in both intermediate and mature stages of aspen regeneration. These results suggest that the Veery occurs in a wide variety of forest types.

In the present study, Veery abundance was negatively correlated with percent high canopy cover and with the frequency of balsam recorded within the sample stands. Percent high canopy cover explained 12% of the total variation in Veery abundance in the sample stands. Reduced high canopy cover suggests a relatively open forest, and is usually correlated with an increase in the density of understory cover.

Robbins *et al.* (1989) found a significant positive correlation between Veery abundance and foliage density from 0.3 m - 1 m tall. Balsam alone explained 44% of the total variation in Veery abundance. This result seems to suggest that the Veery prefers deciduous habitats since the frequency of balsam was negatively correlated with the percent deciduous component in the sample stands (r=-0.61). The Veery may be more abundant in deciduous stands not because of a specific preference for deciduous trees, but because understory cover is denser (in general) in deciduous dominated stands. With its ground-nesting and ground-foraging habits, dense understory cover may be among the most important habitat features for the Veery.

Another possible factor influencing Veery distribution is competition with other thrushes. Harrison (1975) suggests that competition between the Hermit Thrush and the Veery is reduced by their differing habitat preferences. According to Ehrlich *et al.* (1988), the Veery expands its range of breeding habitats where Hermit and Swainson's Thrushes are absent. Since the Hermit Thrush and Swainson's Thrush are found primarily in coniferous forests in this region and the Wood Thrush is relatively rare, the Veery is the primary thrush in deciduous forest stands.

The Ovenbird is associated with mature forests throughout its range in eastern North America. Data from the National Forest Bird Monitoring Program (Hanowski and Niemi 1994) and the results of studies in other regions, suggest that the Ovenbird, like the Veery, is found in a wide range of forest types. In northern Minnesota, the Ovenbird was found in virtually all habitat types and ages (Hanowski and Niemi 1991, Person *et al.* 1993). On the Chippewa National Forest, this species was recorded in stands of jack pine red pine, white pine, maple and birch. In general, Ovenbirds were more abundant in mature forest stands (Hanowski and Niemi 1991).

In the present study, Ovenbird abundance was positively correlated with percent high canopy cover and percent ground cover. It was negatively correlated with percent understory cover (1-4 m). Several other studies have found positive correlations between Ovenbird abundance and percent canopy closure (Shugart and James 1975, Webb *et al.* 1977, Kahl *et al.* 1985). Titterington *et al.* (1977), Lynch and Whigham (1984) and Robbins *et al.* (1989) found positive correlations between Ovenbird abundance and tree density. In contrast, Anderson and Shugart (1974) characterized Ovenbird habitat in Tennessee as having open canopy and subcanopy layers, and dense understory. Martin (1960) also found Venbirds in stands with dense understories in Ontario. Hann (1937) recorded Ovenbirds in forests with open understories in Michigan, and Titterington *et al.* (1977) found a negative correlation between Ovenbird abundance and shrub density on study plots in Maine. Lynch and Whigham (1984), and Kahl *et al.* (1985) reported positive correlations with percent ground cover. The correlations found in the present study agree with the results reported in most previous studies, but contradict the results of others. Because of the lack of standardized methodology in relating birds to their habitats and the likelihood of geographic variation in habitat preferences, simple explanations are difficult. In this region, however, the Ovenbird is found in relatively mature forests with a high degree of canopy closure, and the surrounding landscape seems to make little difference in influencing its relative abundance.

Although the Veery was classified in this study as a deciduous-associated species and the Ovenbird as a mixed-associated species, both species are found in a wide variety of deciduous and mixed forests. In this region, the Veery and the Ovenbird might be described as "forest habitat generalists". This lack of specific forest habitat associations may explain why neither of these species was correlated with the proportions of different forest types in the landscape surrounding the sample stands.

A further explanation for the lack of correlations between these species and the landscape-level variables may be related to the extent of the forests found in northern Minnesota. Several studies have provided evidence suggesting that both the Veery and the Ovenbird are sensitive species (Blake and Karr 1984, Robbins *et al.* 1989, Brewer *et al.* 1991). In a study conducted in east-central Illinois, Blake and Karr (1984) did not detect Ovenbirds in forest "islands" of less than 24 ha, or Veeries in patches less than 28 ha. According to Brewer *et al.* (1991), the Veery has declined in areas in Michigan where urbanization and agriculture have reduced previously extensive forests to isolated patches. Robbins *et al.* (1989) found significant correlations between the area of forest tracts censused, and the occurrence of both the Veery and the Ovenbird. These data suggest that the area of a forest patch may be more important to these species than the forest type or the habitat structure within the stand. In the present study, however, habitat structure within the stand was the best predictor of abundance for both the Veery and the Ovenbird. In contrast to the forest patches censused in the studies cited above, none of the sample stands in the present study was isolated from

other mature forest stands. The minimum forest area requirement of the Veery and the ovenbird was probably satisfied in the sample stands chosen for the present study, and consequently, the habitat structure within the stands was more important in influencing their distribution and abundance. Lynch and Whigham (1984) found similar results for a number of species in their study of relatively non-isolated forest patches in Maryland. They found that significantly more species responded to floristic qualities within the stands, than to the area of the patches. They concluded "above some minimum patch size, correlations between local point density and area of individual forest patches will tend to be outweighed by the effects of habitat quality" (Lynch and Whigham 1984).

The American Redstart has been reported to breed in both deciduous and mixed forest types, as well as in alder swamps and various early succession habitats. In northeastern Minnesota, Redstarts were most abundant in stands of mature paper birch and mature oak (Hanowski and Niemi 1991). They were also recorded in pole-sized jack pine, mature aspen, mature elm-ash-red maple, and regenerating aspen (Hanowski and Niemi 1991). Probst *et al.* (1992) found the American Redstart most abundant in the intermediate stages of aspen regeneration on the Chippewa National Forest. In the present study, all of the variables to which the American Redstart responded represented an increasing conifer component within the sample stands (balsam and birch) and in the landscape surrounding the stands (mature upland mixed forest). Moreover, all of the correlations were negative, suggesting that this species prefer deciduous forest.

The regressional models calculated for this species were significant in all the bands and buffers (P,0.05), however, they explained a relatively small proportion of total variation (19%-28%). This suggests that one or more important variables may be missing from the models. For example, according to Sherry (1979), the Redstart prefers habitats with open areas at mid-height within the forest (i.e., open subcanopy) which permit aerial pursuit of insects. In the present study, percent subcanopy cover was not included in the final regression analyses, and it may have been an important variable in explaining the distribution of redstarts. Bond (1957) found the highest density of Redstarts in stands characterized by a diffuse canopy with 70%-80% closure. There were fewer saplings in these stands than in more mesic stands, but greater diversity of sapling species, and more sapling-like shrubs. Bond (1957) speculated that the more evenly branched and foliaged saplings in these stands "would probably provide more of the upright crotches generally required as nest sites by this species" which is a sapling nester. In the present study, sapling structure was not measured, and may have been another important variable in explaining the abundance of this species in the sample stands.

The Black-throated Green Warbler was positively correlated with birch in all the bands and buffers, and with mature upland conifer in all but the 100 m buffer. In the 100 m buffer, it was negatively correlated with mature upland deciduous forest. These results suggest that the Black-throated Green Warbler is relatively strongly associated with conifers since all of these variables represent an increasing conifer component both within the sample stands and in the landscape surrounding them. Descriptions of Black-throated Green Warbler habitat from the literature support this conclusion. According to Collins (1983), the Black-throated Green Warbler is found primarily in mixed forests including spruce-fir-deciduous, beach-maple-birch, pine, spruce, cedar, and balsam fir. Robbins (1991) identifies the breeding habitat in northern Wisconsin as maple-hemlock-pine forest and conifer forest. Green (1991) lists this species as preferring contiguous coniferous forests in Minnesota.

Other correlations found for this species in the present study were a negative correlation with percent understory cover, and positive correlations with both canopy height and percent high canopy cover. These variables, together with those mentioned above, describe mature forest with a relatively dense canopy, and an increasing conifer component. The Black-throated Green Warbler seems to respond to specific within-stand habitat variables, but is partly influenced by the extent of mature coniferous forest surrounding the stand.

The Magnolia Warbler is strongly associated with the intermediate stages of coniferous forest regeneration. Although it is found in some black spruce bogs and white cedar lowlands (Green and Niemi 1978, Niemi and Pfannmuller 1979), young conifer transition communities with conifer trees from 5 m-10 m tall are its primary breeding habitat (Green and Niemi 1978, Brewer *et al.* 1991). In the study presented here, the Magnolia Warbler was negatively correlated with percent high canopy cover, and positively correlated with percent understory cover, balsam, and percent mature upland conifer in the landscape surrounding the sample stands. Correlations with mature upland conifer were comparatively weak ($0.05<P<0.10$) and this variable explained a relatively small proportion of total variation (6% to 8%). This

is not surprising since young conifers rather than mature conifers are the primary breeding habitat of the species in this region.

It is likely that the Magnolia Warbler was responding to the understory conifer component within the sample stands which was not measured in this study, but is probably correlated with the percentage of mature upland conifer forest in the landscape surrounding the stands. In a landscape where there is more mature upland conifer forest there will probably also be more young conifers. Percent high canopy cover (negative correlation) and percent understory cover (positive correlation) explained the largest proportion of total variation in abundance of this species (34% and 14%, respectively). These variables describe open forests with relatively dense understory development (mostly young balsam). The Magnolia Warbler was positively correlated with balsam in all the bands and buffers. Balsam alone explained from 4% to 15% of total variation in abundance of this species on the study sites.

All of the variables to which the Nasheville Warbler responded represented an increasing conifer component within the sample stands (balsam and birch) and in the landscape surrounding the stands (mature upland mixed forest). The Nashville Warbler is found in a variety of forested habitats from black spruce/tamarack bogs and mature deciduous forests to recently cut areas (Green and Niemi 1978, Niemi and Pfannmuller 1979, Hanowski and Niemi 1991). It is most abundant in conifer-dominated habitat with suitable shrub cover (Green and Niemi 1978). In the present study, the proportion of mature upland mixed forest in the landscape surrounding the stand explained the largest proportion of total variation in abundance of this species within the sample stands (up to 33%). For these stands, it appears that the regional conifer component (represented by mixed forest in the landscape matrix) influenced the presence of this species in aspen-dominated deciduous stands.

Of the species analyzed in this study, the Blackburnian Warbler responded most strongly to the presence of mature upland conifer forest in the landscape matrix. The relative abundance of Blackburnian Warblers was positively correlated in all the buffers and bands with balsam recorded within the stands, and with percentage of mature upland conifer forest in the landscape surrounding the sample stands (partial correlation coefficients 0.14-0.42). These results suggest a strong association between the Blackburnian Warbler and mature upland conifer habitats. Descriptions of Blackburnian Warbler habitat in the literature confirm this conclusion. According to de Kiriline Lawrence (1953), its habitat requirements in Ontario include mature stands of white pine, white and black spruce, hemlock, and red pine in that order of importance. In the Appalachians, the species will inhabit pine-oak vegetation (Harrison 1975), however, when it is found in mostly deciduous forests, it is associated primarily with mature coniferous trees within these areas (Morse 1989). At the Hubbard Brook forest in New Hampshire, Holmes and Robinson (1981) report that Blackburnian Warblers "only occur where there is at least one large conifer or preferably a clump of conifers (specifically spruce) which is used for nesting and some feeding. In a study conducted in Maine, Titterington *et al.* (1977) found the Blackburnian Warbler most abundant on sites characterized by a dense softwood overstory and open forest floor (in other words, mature coniferous forest). They found a significant positive correlation between Blackburnian Warblers and softwood (coniferous) trees with dbh greater than 20 cm (Titterington *et al.* 1977). In New York, Webb *et al.* (1977) found that the numbers of Blackburnian Warblers recorded on logged plots were significantly below the numbers recorded on unlogged control plots. The Blackburnian Warbler responded equally negatively to all levels of crown disturbance. Webb *et al.* (1977) concluded that the Blackburnian Warbler "must be considered an indicator of undisturbed forest conditions, because it gave a full negative response even to light levels of logging."

These results suggest that the proportions of mixed and conifer forests in the landscape surrounding a forest stand are important in explaining the presence of some conifer-associated species within primarily deciduous forest stands. Bird populations often occur in several habitat types on the same region (Pulliam and Danielson 1991), however, there is evidence to suggest that reproductive success can vary in different habitats (Gates and Gysel 1978, Ambuel and Temple 1983, Brittingham and Temple 1983). According to Pulliam and Danielson (1991), "population size and growth rate may vary as functions of the relative proportions of different habitat types available." Pulliam (1988) suggests that surplus individuals form highly productive "source" habitats (i.e., habitats that produce a net increase in population through successful reproduction) may disperse into less productive "sink" habitats (i.e., habitats in which reproductive success is insufficient to maintain a population). In such cases, he concludes that "alterations in the availability of source habitat may greatly affect the size of

populations in other nearby habitats" (Pulliam and Danielson 1991).

The result of the present study may be an indication that the conifer forests in the landscape surrounding the sample stands represent source habitats from which surplus individuals are immigrating into the nearby deciduous stands. It is possible that deciduous forest stands represent sink habitats for conifer-dependent species, although this suggestion is speculative, and requires further study. If conifer-dependent species occur in deciduous stands as a result of immigration from conifer forests in the surrounding landscape, then the diversity of forest birds within a stand is partly influenced by landscape-level management. Diverse forest bird populations would require the development or maintenance of diverse forest mosaics at a landscape level.

Individual tree species appeared frequently in the regression models calculated for this study. The frequency of balsam recorded in the sample stands was correlated with the relative abundance of six of the seven species analyzed. Birch appeared in the regression models for the American Redstart, the Black-throated Green Warbler, and the Nashville Warbler. In a study of tree species preferences of foraging insectivorous birds in New Hampshire, Holmes and Robinson (1981) found that the most abundant bird species showed the least preference for specific trees, while those birds with the strongest preferences for particular tree species were the least abundant in the forest. Common species were relatively evenly dispersed throughout the forest, and used different tree species roughly in proportion to their availability (Holmes and Robinson 1981). In contrast, rarer species were widely scattered, and often very localized in their distribution. Holmes and Robinson (1981) suggest that for the rarer bird species, "their distribution and perhaps even their abundances may be linked to the presence and distribution of particular tree species, or at least to trees with similar physical characteristics." Because the presence of specific kinds of trees appeared to be of major importance at least to some bird species, Holmes and Robinson (1981) concluded that the implications for forest management were significant. They cited the Blackburnian Warbler and the solitary Vireo as examples and suggested that "the removal of coniferous trees from the Hubbard Brook forest would probably result in the elimination of [these species] from the community" (Holmes and Robinson 1981). The results of the present study support Holmes and Robinson's conclusions concerning the importance of specific tree species to certain bird species.

CONCLUSIONS

The results of the present study suggest that the proportions of different forest types within a region are relatively unimportant to species that have generalized habitat needs (e.g., Veery and Ovenbird) provided that some minimum requirement of forested area is satisfied. When sufficient forest of (almost) any type is present, habitat structure within the individual forest stand is the best predictor of abundance for these "forest habitat generalists."

In contrast to the above conclusion, the results of this study provide strong evidence to indicate that the proportions of different forest types in the regional landscape mosaic are important in explaining the distribution of some species that exhibit more specific habitat associations with a particular forest type, especially conifer-associated species. This conclusion is supported by the presence of conifer-dependent species like the Blackburnian Warbler in primarily deciduous forest stands, and by their highly significant correlation with mixed and conifer forests in the landscape surrounding the stands.

Given the increasing demand for wood products, timber harvest in Minnesota is unlikely to decrease in the foreseeable future. How Minnesota's forests are managed will have important consequences for the future of avian biodiversity in the state (Pfannmuller *et al.* 1993). A relatively large proportion of the original conifer component of Minnesota's forests was lost during the late 1800's and early 1900's as a result of logging and the fires which often followed logging (Mladenoff and Pastor 1993). The results of the study presented here demonstrate the need to develop silvicultural methods to maintain and possibly increase the conifer component of forests in northern Minnesota. Balsam recorded within the sample stands was positively correlated with the relative abundance of both Magnolia and Blackburnian Warblers. Thus the maintenance of balsam or other conifer trees as components within deciduous stands appears to be an important factor in enhancing avian diversity within these stands. Natural aspen stands are often of mixed composition, including other hardwoods and conifers (Graham *et al.* 1963). In many cases, species such as balsam have been eliminated in clear-cuts with short rotations (Mladenoff and Pastor 1993). Even-aged aspen stands often differ most from natural stands by a lack of dominant conifers (Mladenoff and Pastor

1993). The use of partial cutting with an extended rotation and a two-phase cutting cycle would encourage the retention of a conifer component within these stands (Mladenoff and Pastor 1993). Moreover, the retention of conifer stands within the landscape would enhance the diversity of wildlife populations that depend on conifer habitats and also influence species diversity within deciduous stands in these landscapes.

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SURVEY OF WOODLAND KEY HABITATS

Mikael Norén[1]

WOODLAND KEY HABITATS IDENTIFIED IN THE SWEDISH FORESTS

A survey is being made of the entire Swedish forest landscape in searching for areas with high nature values. These areas are called Woodland Key Habitats and yield new knowledge on the status of the Swedish forest. The results of the survey will provide the basis for ensuring that Sweden achieves the goal of environmentally adapted forestry, where conservation of biodiversity is of great importance. The survey is financed mainly by the state. In addition to locating tens of thousands of key habitats - many of which were totally unknown earlier - the work has led to a significant raising of competence levels both of employees and forest owners within Swedish forestry, particularly in the areas of conservation biology, landscape ecology and recognition of species.

FORESTRY POLICY

Swedish forestry policy changed markedly in 1993. The new policy approved by parliament consists of two forestry policy goals; a production goal and an environmental goal. Both are of equal importance and should be given equal weight in forestry. No contradiction in the two goals is implied and they are, in fact, compatible with one another. The environmental goal, for example, states that biodiversity shall be secured and that endangered species and ecosystems must be conserved. The forestry policy is aligned towards maintaining multiple-use aspects of the Swedish forest. Consideration must always be paid to nature in all stands irrespective of the silvicultural measures being taken. Silviculture and consideration to nature should, however, be adapted to circumstances found in the different stands, meaning that nature conservation is significantly more far-reaching in some cases than in others.

The ambition is that Swedish forestry policy will be achieved chiefly on a voluntary basis by the forestry sector itself. This assumes that forest owners and the whole forest industry will raise their levels of competence, implement ecologically-adapted forestry methods, and that they are prepared to accept responsibility, e.g., by voluntarily setting aside stands for free growth. Society's main task is to disseminate knowledge to the forest owners and to ensure that they comply with legislation.

Achievement of forestry policy goals requires knowledge of conditions prevailing in the forests, e.g., the nature values of the forest landscapes. Knowledge of valuable forest landscape areas has earlier been inadequate. The survey of Woodland Key Habitats has provided an unexpectedly rich knowledge of the nature values of the forest.

WHAT IS A WOODLAND KEY HABITAT?

The term Woodland Key Habitat indicates a quality in the forest landscape in terms of the high natural values of an area. The definition of a Woodland Key Habitat is an area where one or more red-listed species occur, or the very nature of the forest itself should indicate a strong likelihood of finding red-listed species. Species are grouped on the red list as either being Critically Endangered, Endangered, Vulnerable, at Lower Risk. The Swedish red list is compiled by the Swedish Threatened Species Unit at the Swedish University of Agricultural Sciences and is finally approved by the Swedish Environmental Protection Agency. It is not economically possible to search for species whilst making the survey. The emphasis is rather on defining and describing the valuable habitats.

A Key Habitat can theoretically be of any size, with quality being the decisive factor in setting the limits. A Key Habitat may be anything from a single ancient oak tree in the south, to a larger area of several hundred hectares of old coniferous forest in the north. The term 'Key Habitat', as used in the survey, lacks legal status and consequently, there is no automatic legal protection against felling in Woodland Key Habitats.

[1] *Swedish National Board of Forestry, 551 83 Jönköping, Sweden*

THE SURVEY

Surveys are being carried out throughout the whole of the forestry sector. The Swedish Forestry Administration has been commissioned by the government to conduct inventories of Woodland Key Habitats on small-scale forestry areas, which correspond to about half the forest area. All of the County Forestry Boards throughout the country are working on systematic surveys. Surveys commissioned by the state were started in 1993 and should be completed by 1998, at a cost of about US$ 13 million. The remaining areas are surveyed and paid for by the responsibility of the companies themselves. This division of responsibility has been achieved on a voluntary basis. As a result, there is no legal obligation that requires the forestry sector to carry out inventories of Key Habitats. The Swedish Forestry Administration is responsible for training, education and support as well as follow-up of this part of the survey work. Surveys of Key Habitats in areas with large-scale forestry are expected to be completed by 2003.

The compilers often have higher education in forestry and are especially trained in knowledge of different species and the recognition of habitats for this assignment.

The survey is conducted in two stages. First, the compilers search source material in different archives for possible Key Habitats. These are then assessed in the field and sites with sufficiently high nature values are defined and described as Woodland Key Habitats. Main emphasis is placed on the survey work done on-site. Important components in the preparation stage include, forestry classification material, forestry plans, maps, infrared air photographs, and knowledge that can be obtained from landowners or voluntary nature conservation societies. After the preparation work has been carefully completed, about 5% of the whole forest land area is visited on-site.

Searching for Key Habitats is very much a question of detective work. Most of the Key Habitats are unknown. They are often relatively small and off the beaten track in the forest. Fallen tree trunks, high stumps, stumps of broad-leaved trees, old trees, rocks and rock faces, and water courses, are all examples of structures that are decisive for assessing whether or not red-listed species can be expected or not in the habitat. Several species that indicate high nature values are used in the project. The existence of indicator species tells us a lot about the habitat. Presence of these species indicates that the environment may be suitable for red-listed species.

Other important grounds for assessment of possible habitats are the history of the forest, its age, and the degree of influence, plus the presence of certain types of tree. An important question is whether or not there has been unbroken forest continuity on the site, if it has been forestland since time immemorial or not. In general, forests of great age and with long forest continuity have higher nature values.

During the survey, information is registered in a database with details of the location of the Key Habitat, the soil type, the habitat's tree varieties, the forest structure in detail, indicator species and red-listed species.

WHAT ARE THE WOODLAND KEY HABITATS?

The method chosen in Sweden is principally adapted to the managed forest landscape. Key Habitats are often found as fragments or 'island' habitats, that often differ greatly from the surrounding landscape. They may be remains of the natural landscape or of old forest stands, or forest stands with long forest continuity. Many environments with long forest continuity are to be found in agricultural landscapes created by man in the transitional areas between forests and farmland. Physical factors such as air humidity conditions, may also be fundamental in Key Habitat quality.

THE RESULTS

So far, the results show that the Woodland Key Habitats account for less than 1% of the total area of productive forestland. The final result of the survey may show a total number of 60,000-80,000 Key Habitats spread throughout the country's 23 million hectares of productive forestland. They appear unevenly in the landscape. In certain districts large areas of forest are of Key Habitat quality, while other sections of the landscape more or less completely lack Key Habitats. Generally speaking, Key Habitats are larger in the north of Sweden than in the south. Habitats in the north also increase in size from east to west. The number of key habitats per 1,000 ha varies significantly and is greatest in parts of southern Sweden.

THE ROLE OF THE LANDOWNERS

The landowner is informed as quickly as possible when a Woodland Key Habitat is discovered, and given advice on how the nature values of the Key Habitat may be conserved. Naturally, the intention is that nature values of the key habitat shall remain in the future. The interest and wishes of the landowner to conserve the natural values are of fundamental importance.

Voluntary efforts are important for the conservation of Key Habitats. A number of Key Habitats may fall within the scope of legislation on nature conservation according to the Forestry Act. In some cases, decisions in accordance with the Nature Conservation Act may be applicable. If habitat protection involves a significant deterioration in the on-going use of the land, the landowner has the right to compensation.

Many Key Habitats have to be left totally untouched in order to keep their values. In some cases, buffer zones may be needed, e.g., felling in an adjacent stand. In some Key Habitats, it may be possible to extract timber. Occasionally, appropriate management inputs may be necessary in order to retain certain habitat characteristics or species on-site.

KEY HABITATS AND THE FUTURE

The Woodland Key Habitats can be seen as core areas for biodiversity. They account for the major share of biodiversity, in relation to their comparatively small areas. The results of the survey will play a major role in many ways, not least in forestry planning of both real estate and in a landscape perspective. The new knowledge provided by the survey will be an important tool in our efforts to achieve environmentally adapted forestry for many years to come.

FOREST ECOSYSTEM RESEARCH NETWORK OF SITES (FERNS)

D.A. MacIsaac[1] and J. Wood[2]

ABSTRACT.--The Canadian Forest Service (CFS) has organized a national network of research sites, named the Forest Ecosystem Research Network of Sites (FERNS). These sites are focused on the study of sustainable forest management practices and ecosystem processes at the stand level. This network promotes this research nationally and internationally, improves linkages among sites, preserves the long-term research investments already made on these sites and provides a forum for information exchange and data sharing. While the CFS coordinates and promotes FERNS, the network consists of local autonomous partners nationwide who benefit from the FERNS affiliation through increased visibility for their sites.

INTRODUCTION

Forests cover almost one-half of the land mass of Canada and 10% of the global forest cover is located within our borders. As a result, Canada finds itself at the center of the growing international debate on the environment. Science has a critical and increasing role to play in this debate. As a nation, Canada is being evaluated internationally against complex criteria such as effects of forestry practices on ecosystem productivity, ecosystem processes and long term biological diversity. Because of the complexity of these issues, as well as the inherent complexity of forest ecosystems themselves, multidisciplinary and multi-partner approaches to forest ecosystem management research have become the norm in the 1990's.

Across Canada, long-term multi-disciplinary forest management research sites and studies are already in place, but the linkages between the individual sites has not yet been made. The network of sites, named the Forest Ecosystem Research network of Sites (FERNS), is the framework for a national suite of research sites that are focused on the development of innovative forest management practices and understanding of ecosystem processes at the stand level.

The overall goal or mission of FERNS is to provide and promote a network of sites, representative of major Canadian forest ecosystems, for long-term multi-disciplinary research on forest management practices. FERNS has three specific objectives: promote forest management practices research nationally and internationally, improve linkages among sites, and preserve the long term research investments already made on these sites.

FERNS promotes Canadian forest management practices research as well as the individual research sites both nationally and internationally, using traditional print media as well as the latest electronic technology. Individual FERN Sites are examples of high quality forestry practices research.

FERNS provides researchers and students with a set of well-characterized and liked sites across Canada on which to work. By providing opportunities for comparative and collaborative research, FERNS also fosters partnerships between governments, universities and industry, and serves as a national communication tool between researchers and with forest managers. The network also links researchers working at the various sites as well as providing a forum for information exchange and data sharing. In this regard, FERNS is fostering a sense of community among forest ecosystem management researchers in Canada. In addition, at the national and international levels, the network is linking forest managers to researchers.

FERNS is helping to protect the investment made in the installation of long-term stand

[1]*D.A. MacIsaac, Natural Resources Canada, Canadian Forest Service, Northern Forestry Centre, 5320-122 Street, Edmonton, AB, Canada T6H 3S5 Phone: (780) 435-7332 FAX (780) 435-7359. E-Mail: dmacisaa@nrcan.gc.ca.*

[2]*J. Wood, Natural Resources Canada, Canadian Forest Service, Pacific Forestry Centre, 506 West Burnside Road, Victoria, B.C., Canada V8Z 1M5 Phone: (250) 363-6008 FAX (250) 363-0775. E-Mail: jwood@pfc.forestry.ca.*

manipulation studies, the gathering of extensive biophysical databases, and the conduct of site-specific studies. Although FERNS does not provide funds, the association of each site to the network will enhance its visibility and its ability to attract funding from agencies or from industrial partners.

A new global scientific culture is emerging in which archiving and inter-connectivity of data sets is becoming increasingly important. Researchers are now being encouraged to make their data available for wider distribution once the scientific paper has been published. In the future, FERNS may be able to play an important data management role by providing a means to synthesize and integrate the scientific information across a number of sites.

CHARACTERISTICS AND CRITERIA

Each FERN Site possesses a number of characteristic features. For example, the research conducted on FERN Sites:

- Addresses forest harvesting options and ecosystem functioning aimed at sustainable forest management;
- Is multidisciplinary and multi-partnered; and
- Is sound ongoing research.

FERN Sites must be:

- Well documented with good historical stand and site information;
- Accessible;
- Located in a forest that is representative of one of the major forested ecozones in Canada; and
- Protected for long-term research values.

In addition, FERN Sites must include areas of mature forest where natural successional stages can be observed and/or investigated.

CANADIAN FOREST SERVICE ROLE

The Canadian Forest Service (CFS) is the largest national forest science organization in Canada and has a mandate to address both national and international issues affecting forests and forestry practices. There is a need to bring together key research activities and to provide a framework for a more integrated Canada-wide approach to forest ecosystem management research among CFS researchers and partners in industry, provinces and universities.

The CFS will contribute to the coordination and promotion of FERNS, but it must be recognized that the network will consist of a collection of local partners across the country that have come together freely to achieve things which exceed what any one of them, individually, could achieve. As such, each site will continue to operate autonomously. Where a benefit can be recognized, however, sites may act together as a network. Partnerships and links between FERNS and national and international programs will be encouraged.

ESTABLISHED FERN SITES

The attached map (fig. 1) indicated established FERN Sites across Canada as of August 1997, with the numbers below corresponding to the numbers on the map.

Pacific Maritime Ecozone

1. Montane Alternative Silvicultural Systems (Vancouver Island, British Columbia)
Contact:
Dr. Al Mitchell, Pacific Forestry Centre, Canadian Forest Service, Victoria, B.C.
Phone: (250) 363-0600
E-mail: amitchell@cfs.nrcan.gc.ca.

Salal Cedar Hemlock Integrated Research Program (SCHIRP) (Vancouver Island, British Columbia)
Contact:
Dr. Cindy E. Prescott, Project Leader
Faculty of Forestry, University of British Columbia, Vancouver, B.C.
Phone: (604) 822-4700
E-mail: cpres@unixg.ubc.ca
Website:
http://www.forestry.ubc.ca/schirp/homepage.html

Bill Beese, MacMillan Bloedel Ltd., Nanaimo, B.C.
Phone (250) 755-3422
E-mail: w.j.beese@mbltd.com.
Website:
http://www.pfc.cfs.nrcan.gc.ca/practices.mass

Montane Cordillera Ecozone

2. Sicamous Creek Silvicultrual Systems Project (southern interior, British Columbia)

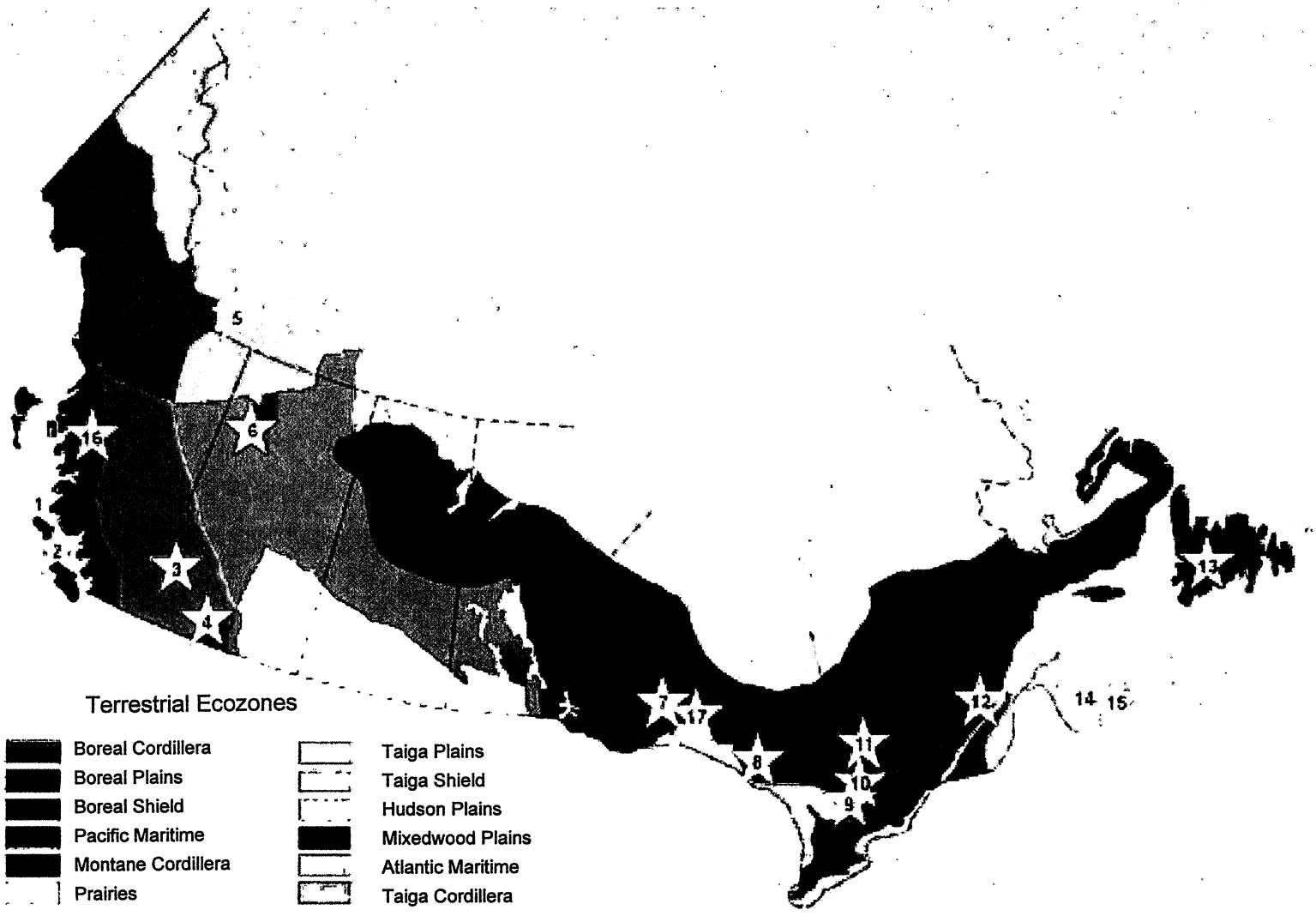

Figure 1.--*Established FERNS (Forest Ecosystem Network of Sites) sites across Canada as of August, 1997.*

Contact:
Alan Vyse, BC Ministry of Forests, Kamloops, B.C.
Phone (250) 828-4183
E-mail: avyse@gemsl.gov.bc.ca
Website:
http://www.hre.for.gov.bc.ca/grops/kamloops/sicrule.html

3. Lodgepole Pine Management Study (southern interior, British Columbia)
Contact:
Roger Whitehead, Pacific Forestry Centre, Canadian Forest Service, Victoria, B.C.
Phone: (250) 363-0600
E-mail: rwhitehead@pfc.cfs.nrcan.gc.ca

Dr. Les Safranyik, Pacific Forestry Centre
Canadian Forest Service, Victoria, B.C.
Phone (250) 363-0600
E-Mail: lsafranyik@pfc.cfs.nrcan.gc.ca
Website: http://nrcan.gc.ca/cfs.proj/sci-tech/arena/styr_e.html

Taiga Plains Ecozone
4. Muskeg River Silvicultural Study (Liard Valley, Northwest Territories)
Contact:
Derek Sidders, Northern Forestry Centre, Canadian Forest Service, Edmonton, AB
Phone: (403) 435-7355
E-mail: dsidders@nrcan.gc.ca.

Boreal Plains Ecozone
5. Hotchkiss River Mixedwood Timber Harvesting Study (northwestern Alberta)
Contact:
Dan MacIsaac, Northern Forestry Centre, Canadian Forest Service, Edmonton, AB
Phone: (403) 435-7332
E-mail: dmacisaac@nrcan.gc.ca.

Boreal Shield Ecozone
6. Black Sturgeon Boreal Mixedwood Project (north-central Ontario)
Contact:
Al Cameron, Great Lakes Forestry Centre, Canadian Forest Service, Sault Ste. Marie, ON
Phone: (705) 949-9461
E-mail: acam@nrcan.gc.ca
Website:
pfc.cfs.ncrcan.gc.ca/practices/ferns/hotchkiss.htm

7. Turkey Lakes Silvicultural Study (northeastern Ontario)
Contact: Dr. Al Cameron, Great Lakes Forestry Centre, Canadian Forest Service, Sault Ste. Marie, ON
Phone: (705) 949-9461
E-mail: alcam@.nrcan.gc.ca

8. Scott and Swan Lake Forest Research Reserve
(central Ontario)
Contact:
Dr. Bill Cole, Ontario Forest Research Institute, Ontario Ministry of Natural Resources, Sault Ste. Marie, ON
Phone: (705) 946-2981 Ext. 113
E-mail: coleb@gov.on.ca.

9. White Pine Silvicultural Study (central Ontario)
Contact:
Steve D'Eon, Petawawa Research Forest, Canadian Forest Service, Chalk River, ON
Phone: (613) 589-3009
E-mail: sdeon@intranet.ca.

10. Lac Duparquet Research and Teaching Forest (northwestern Quebec)
Contact: Dr. Yves Bergeron, Université du Québec à Montreal, PQ
Phone: (514) 987-3000 Ext. 4872
E-mail: bergeron.yves@uquam.ca
Website: www.pfc.cfs.ncrcan.gc.ca/practices

11. Montmorency Experimental Forest (St. Lawrence Region, Quebec)
Contact:
Dr. Louis Belanger, Laval Univeristy, St. Foy PQ
Phone: (418) 656-2131 Ext. 6110
E-mail: louis.belanger@sbf.ulaval.ca.

Dr. Brian Harvey, Université du Québec en Abitibi-Témiscamingue, PQ
Phone (819) 762-0971 ext. 2361
E-Mail: brian.harvey@uquat.uquebec.ca
Web site: http://web2.uqat.uquebec.ca/ferld/

12. Glide Lake (western Newfoundland)
Contact: Dr. Doyle Wells, Atlantic Region, Canadian Forest Service, Corner Brook, NF
Phone: (709) 686-5038
E-mail: dwells@nrcan.gc.ca.

Atlantic Maritime Ecozone
13. Acadia Experimental forest (south-central New Brunswick)
Contact: Edwin Swift, Atlantic Forestry Centre,

Canadian Forest Service, Fredericton, NB
Phone: (506) 452-3175
E-mail: eswift@nrcan.gc.ca

14. Fallingsnow Ecosystem Project (north-central Ontario)
Contact:
F.W. Bell, Project Co-ordinator
Phone: (705) 946-2981 x225
E-mail: bellw@gov.on.ca

R.A. Lautenschlager, Co-ordinator
Phone: (705) 946-2981 x228
E-mail: lautenr@gov.on.ca

16. The Hayward Brook Project (southeastern New Brunswick)
Contact: Dr. Gerry Parker, Canadian Wildlife Service, Sackville. NB
Phone: (506) 364-5045
Website:
http://www.pfc.cfs.nrcan.gc.ca/practices/ferns/acadia/acadia.htm

16. The Date Creek Silvicultural Systems Research Project (north-central B.C.)
Contacts: Dave Coates, Project Leader
B.C. Forest Service, Smithers, B.C.
Phone: (250) 847-7436
E-Mail: Dave.Coates@gems7.gov.bc.ca
http://www.pfc.cfs.nrcan.gc.ca/practices/FERNS/date/date.htm

Phil LePage, Research Silviculturalist,
B.C. Forest Service, Smithers, B.C.
Phone (250) 847-7437
E-Mail: Phil.LePage@gems8.gov.bc.ca
Website:
http://www.pfc.cfs.nrcan.gc.ca/practices/FERNS/date/date.htm

17. Fallingsnow Ecosystem Project (north central Ontario)
Contacts: F.W. Bell, Project Coordinator
Phone: (705) 946-2981- x225
E-mail:
R.A. Lautenschlager, Co-ordinator
Phone: (705) 946-2981 x228
E-mail: lautenr@gov.on.ca

CURRENT STATUS

There is solid and growing support for FERNS from research partners both inside and outside of the CFS. Since an initial set of FERNS sites was announced early in 1997, there have been several new additions to the Network.

In August 1997, an Internet web site was launched to profile FERNS. This web site is accessible through the Canadian Forest Service, Pacific Forestry Centre home page at http://www.pfc.forestry.ca or http://www.pfc.cfs.nrcan.gc.ca.. This internet site includes information on individual FERN sites, as well as linkages to collaborators.

TO OBTAIN MORE INFORMATION

For further information about FERNS contact Chris Lee, FERNS Coordinator, Canadian Forest Service, Pacific Forestry Center, Victoria, British Columbia, Canada. Phone: (250) 363-0600. E-mail: clee@pfc.forestry.ca. To obtain information on an individual site, the specific contact person listed in this document can be contacted.

For assistance in accessing the FERNS web site contact Chris Lee (contact information listed above) or Rod Maides, Communications, Canadian Forest Service, Pacific Forestry Centre, Victoria, British Columbia, Canada. Phone (250) 363-0737. E-Mail: rmaides@pfc.forestry.ca.

SIMULATION OF HETEROGENEOUS UNEVEN-AGED STANDS DYNAMICS UNDER HUMAN-INDUCED DISTURBANCE

S.I. Chumachenko[1] V.V. Syssouev, M.M. Palyonova, M.A. Bredikhin, V.N.Korotkov[2]

ABSTRACT. -- In the framework of a complex information system (DBMS, GIS, MODEL) for a forest management unit (FMU) a simulation model was developed, which describes the growth dynamics of heterogeneous uneven-aged stands as a function of their spatial and ontogenetic structure. It can be used for forecasting stand growth both in natural and artificial forests in stable and disturbed habitats. The input and output data are represented in the form of standard inventory descriptions and forest plans. A series of simulation calculations under human-induced disturbance was carried out using as examples forests in a FMU in the Moscow Region.

Complex investigations carried out in recent years (Smirnova 1994, Kirikov 1979), showed that forests in the central part of European Russia developed essentially under human-induced disturbance and at present there are no undisturbed woods in the zone of coniferous-broad-leaved and deciduous forests. Now among the old forests one can see the results of various anthropogenic impacts such as burning, felling, and uprooting. Better understanding of the processes that brought contemporary forests to such a state, facilitates simulation of different patterns of their transformation. The methods of mathematical modeling allow us to solve the problems of stand transformation taking into account the historic land exploitation of the territory. The quality of these methods enables us to make long-term forecasts (one cutting cycle and more), and, therefore, to elucidate the results of anthropogenic impacts that occurred far in the past when compared to the duration of development of tree edaphicators.

This paper describes simulation of the impact of historical land exploitation on the growth of forest coenoses under different silvicultural treatments. The problem was solved within the framework of the program complex DBMS-GIS-MODEL, in which DBMS and GIS accumulate and process the attribute and spatial information for the forest object, and MODEL develops the forest dynamics forecasting. The most important feature of this model is the of strata position and influences between strata.

MODEL DESCRIPTION

The per-stratum data bank and cartographic bank of plans and cartographic materials of the forest planning and inventory works are used as initial information for the forecast module of the model. For individual forest management units these data exist in the form of information systems (DBMS or GIS). As a result of the model processes similar information is obtained from the different databases. To take into account bioecological processes in real forest communities, a large number of reference data bases is used within the simulation. The main ones are listed below.

1. **Bioecological characteristics of species development.** The information, compiled in these data bases, was assembled as a result of literature analysis and long-term natural research studies (Smirnova *et al.* 1990, Smirnova 1994). Also the accumulated experience of mixed uneven-aged stands simulation is taken into account (Chumachenko 1993). For the model development reference bases are created for species main bioecological parameters separately for each stage of tree development of each species. These reference bases were compiled proceeding from the concept of discrete description of ontogenesis (Cenopopulation... 1988). The application of this concept for the simulation of complicated stands gives more authentic results than with fixed biological parameters for the whole period of ontogenesis. These reference bases are listed below:

[1] *Sergey I. Chumachenko. Moscow State Forest University, 1 Institutskaya, 1, Mytischi, 141001, Russia, e-mail: chumachn@cc.mgul.ac.ru.*

[2] *Vladislav V. Syssouev, Maria M. Palyonova, Mikhail A. Bredikhin, Vladimir N.Korotkov All-Russian Research and Information Centre for Forest Resources, Novocheremushkinskaya, 69, Moscow, 117418, Russia, e-mail: romanova@sci.lpi.msk.su.*

2. **Crown biometrics characteristics**. The crown form is approximated by a combination of elementary bodies of rotation: cone and cylinder. One model peculiarity is that it takes into account the internal crown space (shadow cone), in which there are no photosynthetic elements (Nosova *et al.* 1995). The comparison of stand development at different densities has shown, that growth in height depends on light availability at the active growth zone and not at the top of a tree. The active growth zone is a part of the tree crown, where the large share of its green biomass is concentrated. In the reference base this zone is given by the position of the shadow cone in the tree crown. Besides these ones are used the following reference bases:

3. **Demand for light (shade tolerance)** (Alekseev 1975, Tsel'niker 1978).

4. **Crown transmittance of light**. (Alekseev 1975).

5. **Distance of seed dispersal** (Udra 1990).

6. **Sprouting ability** (Udra 1990, Smirnova *et al.* 1990).

7. **Yield tables** (YT) (Kozlovsky and Pavlov 1967); local yield tables or those obtained on the bases of inventory descriptions are preferrable (Bredikhin 1985).

The first stage of initial data transformation - the construction of a planar uniform grid - is carried out by means of GIS (fig. 1). The square dimension is determined by the geographical latitude of the district and by the height of the top stand story. So, for example, for Moscow latitude and stand height 30 m the area is about 270 m^2. As a result the complicated configuration of each stratum in the plan is approximated by a set of elements, having properties of the stratum, to which they belong. Further the element is divided vertically into cells 2 - 2.5 m tall. The cell (from the point of view of simulation) is an indivisible minimum unit of the three-dimension space. Such representation of the simulated space permits us to take into account both self shadowing, and shadowing from adjacent elements during the daily solar movement.

To calculate light availability astronomical data about solar radiation at given latitudes of the district are used, the light intensity during daylight being taken into account. The differentiation of light flow is determined not only by azimuth, but also by height of the sun above the horizon and quantitative relation between direct and scattered radiation. Data about average long-term quantity of sunny (cloudy) days in the region during the vegetative period; about conditions of clouds and dust in the daytime; and about weakening of natural light by general air pollution were also used.

There are two different approaches to calculate light regime under the canopy layer. At the basis of the first one is the fact that in practically every forest there are patches of sunlight. The frequency of their appearance and duration vary, depending upon stand density and species composition (Tsel'niker 1978). Thus, at the basis of the approach proposed by T. Nilson *et al.* (1977) to the description of radiation weakening under heterogeneous plant cover there is a calculation made by means of random value that defines whether the direction of a light beam through the crown.

The second approach is based on the idea of a "muddy" layer, i. e. solar radiation passing through an absorbing layer is partially weakened. In numerous works (Botkin *et al.* 1972, Shugart *et al.* 1974, Karmanova and Abramov 1990), in which the problems of light intensity influence on tree growth are considered, different modifications of Monsi-Saeki's relation are used to connect the light flow with leaf surface area (Monsi and Saeki 1953). When calculating shade density, notwithstanding the chosen methods, the following approaches to modeling stand canopy spatial heterogeneity are used: accounting for only horizontal and vertical heterogeneity of stand canopy (Botkin *et al.* 1972, Shugart *et al.* 1974, Leemans and Prentice 1989) or partial accounting for volume heterogeneity (Korzukhin and Ter-Mikhaelyan 1982, Pacala S.W. *et al.* 1995).

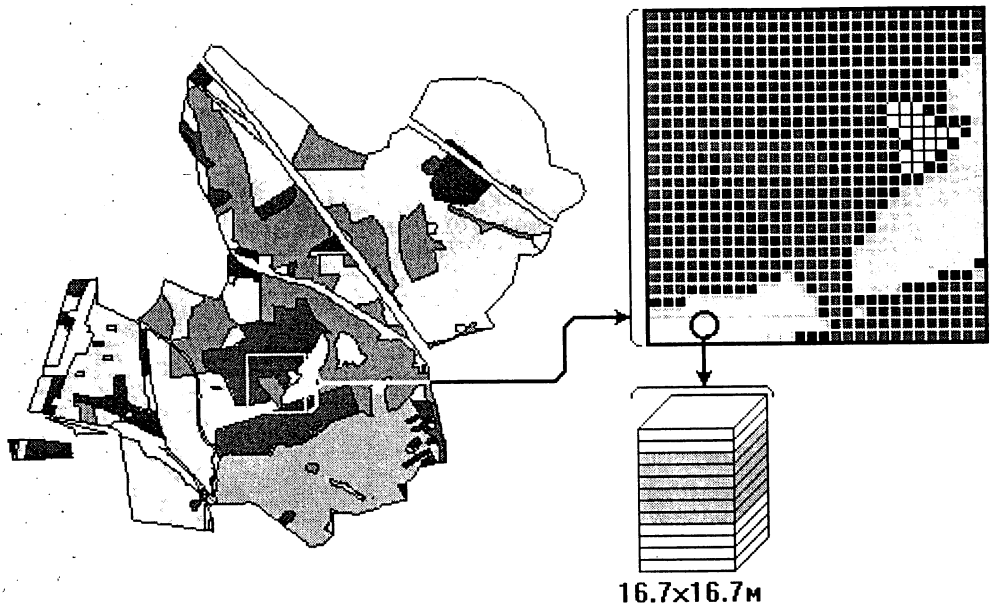

Figure 1. -- *Spatial data preparation.*

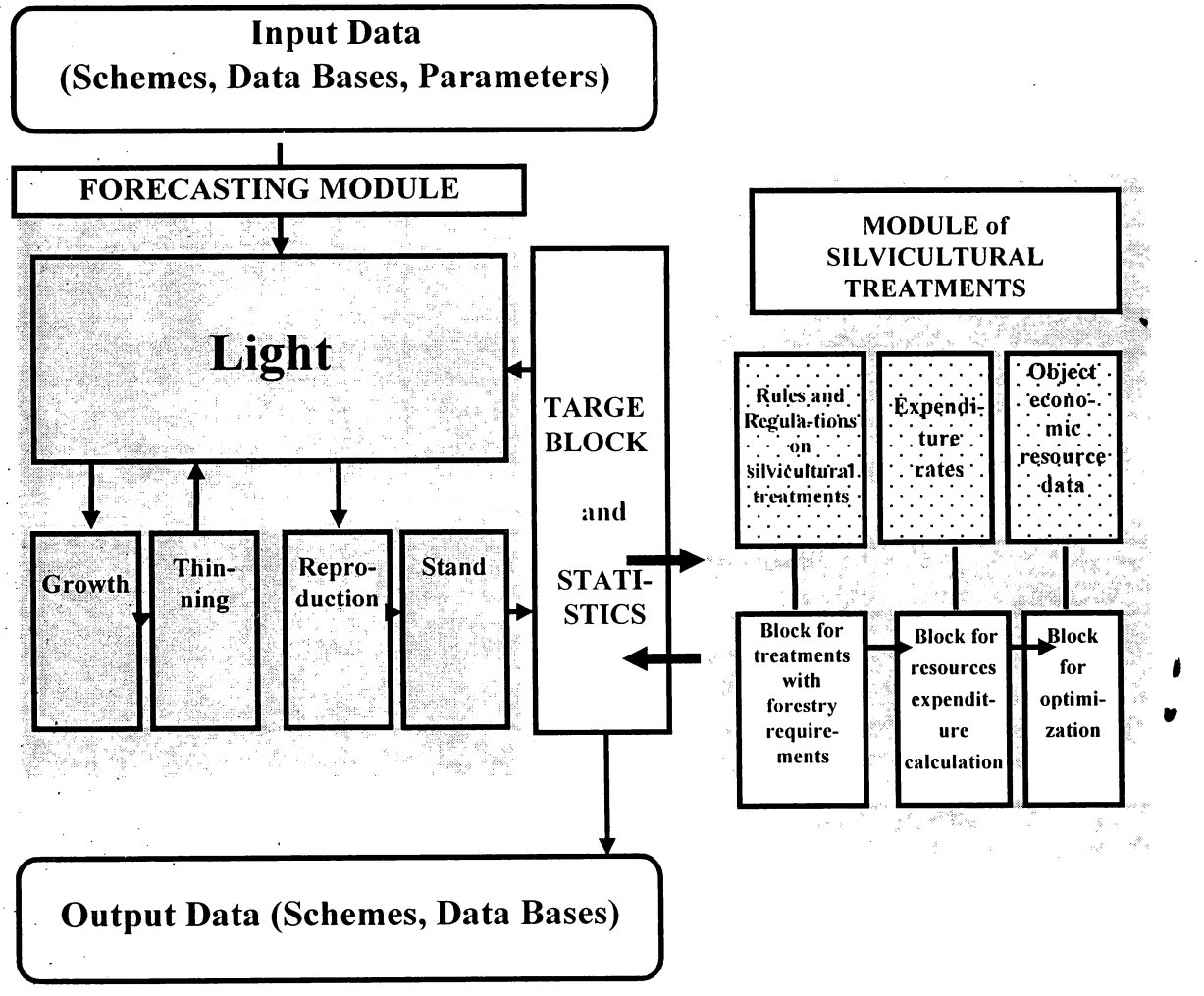

Figure 2. -- *Model scheme.*

In the proposed model each cell is represented by a semiopaque volume, therefore it is better to use the idea of a "muddy" layer (volume), where a part of the light is absorbed passing through it, according to Monsi-Saeki's relation.

From the point of view of block "Light" work the state of a cell is defined by the following parameters for each species: their quantity, average height, crown length, its form and penetrability. For each cell the shares of species and tree age present are calculated, thus are formed the set of "muddy" layers (fig. 2).

In the model sets of point light sources are situated at fixed places of the firmament, which have precalculated values of solar radiation. When calculating the cell light availability an account is taken not only of the adjacent cells, but also of other near-by ones through which the light is passing at a definite angle at a certain moment in the day-time. Thus, to calculate the light regime for the lower canopy 14 elements are used, and 15 if self-shadowing is to be taken into account, for a total of more than 160 cells.

The simulation model of multispecies uneven-aged stand dynamics consists, of several target blocks (fig. 2). The "Light" block begins each step of the simulation and determines the light availability in each cell of the modeled space (fig. 2). The calculating algorithm of light availability is described thoroughly by Chumachenko (1993).

Using data about growth of individual species in the stand, the "Growth" block calculates for each element the current increment of the species by diameter and height, taking into account the element position and its light maintenance. It is assumed in the model, that the main factor affecting the tree growth under the forest canopy is the light availability. Other important factors (such as available moisture and soil fertility) are modeled as rather stable integrated (average for 10 year) characteristics of the element and are taken into account when species biological quality are determined for each stratum. The methods of defining biological site classes for stands, typical in various growing conditions, are still developed insufficiently. In the present version of the model the table of greatest possible site classes for the prevailing species is built by an expert in interactive mode on the basis of initial inventory data and relief characteristics, by taking into account available moisture and soil active nutrients, with the use of the following main assumptions.

The dependence of bioproductive process on habitat conditions may be presented by a bell-form distribution -- schematically this dependence can be described by a system of second-power equations of three variables, namely moisture amount in air drainage zone; sum of temperatures; and a hydrothermic factor that describes the relation between heat and moisture (Pusatchenko 1987). For each kind of plant these functions have approximately the same form, but differ by appropriate constants.

The "Thinning" block includes two main options of behavior of the model: endogenic thinning, as a consequence of intra- and interspecific competitive relations, and thinning as a result of exogenous factors. The endogenic thinning is simulated by three consecutive operations. First, the model searches and excludes from the element those species which do not reach the habitus typical for a given species at a given age (for the lowest of site classes available). The model also excludes species that have reached the maximum age, according to YT data. Second, because of light being the main limiting factor in the model, for each species in the given stratum element a factor of increment loss during one step is calculated. The factor in the model is defined as a ratio of the real increment to that from YT. The dying off criterion is the calculated factor being reduced below the given level (in the present version of the model -- 10 %).

Third, to maintain tree density below critical values (according to biological parameters), the "Thinning" block executes the recalculation of tree number within an element. In each story the zone of best competition for space -- a cell with maximum crown cover, and accessible area for crown growth -- is determined. The redistribution of area in story is executed at the expense of intercrown clearance and begins with shade-tolerant species. In such way, supersede more intolerant, changing the proportion of areas occupied by each species.

Thinning as a result of exogenic factors includes anthropogenic, technogenic, and other external effects on the simulated species. The conditions of additional thinning can be incorporated at any step, and they are selective ones. In the last version of the model they are taken out of the model into the block "Target blocks and statistics". After simulation of

thinning, light availability of each cell is recalculated with simultaneous changes in crown height for some species in the element.

In the block "Reproduction" the quantitative valuation of young growth appearing is made. The intensity of germ motion occurrence depends on the degree of remoteness of the element from seed trees of the given species and the light conditions of their development. The opportunity of coppice young growth in the element is also simulated. In view of various speed of growth and ontogenesis duration, and different YTs for coppice and seedling stands, the calculations for them are carried out separately.

After processing of each step of the model, the element contains characteristics for species and age structure of the stand and its biometrics parameters: species, stem quantity, age and age condition, average height of tree and average crown height, average stem diameter, crown canopy and crown form, achieved (calculated) site class. Besides these are calculated stock and stand density. These data are basic for the work of various applied blocks, external to the main model. By means of the rules given in the model the block "Target blocks and statistics" carries out samples, performs statistical processing, executes the assembly of elements into inventory strata and prepares the information for representation to DBMS and GIS.

The submitted imitation model is an open one, i.e., it permits to add various target blocks. Such blocks (e.g., stimulating final felling of various configurations, cleaning cutting, etc.) can be attached at any step of the model work. The model permits simulation of the effect of various catastrophic factors (fires, blowdowns, etc.) and forest management influence (final felling, cleaning cutting, forest regeneration measures, etc.).

SAMPLE OBJECT

For checking the model work the forest range of Korobovsky forest district was chosen This is part of the Natural-historic reserve forest-park "Gorky". The territory is located on the Moscow-Oka plain (in the south of the Moscow Region) within the spruce-broad-leaved forest zone. The landscape structure of the territory is characteristic of the moraine-fluvioglacial plain: flat interfluves, covered with loam soils, divided by river and stream valleys, built by deluvial and alluvial deposits. Thus, there are two main groups of habitats: flat and gently sloping hilly plains and those of ravine and fluvial valley slopes. The morphology of the landscape is complicated by long economic use that causes the original plant structure, including the woody one.

The sample object area is about 600 ha. Forests, typical for the middle part of European Russia, are growing there. On the basis of ancient maps (XVIII - XIX cc) the history of forest range formation has been followed. The central part of the forest range (55% of its area) is covered by broad-leaved forests with prevalence of lime and oak of 70-100 years old, that have appeared at durable forest lands as a result of repeated felling. These lands have not been ploughed during at least the last 200-300 years. At the periphery of the forest range, there are birch forests of 60-80 years old (45% of the area), that have arisen on arable land since the middle of the XIX century and undergone several harvest cycles.

SIMULATION RESULTS

In this examle stand dynamics were simulated under two kinds of external forestry effects: (1) final felling only, when the stand reaches its maturity age and (2) combined cleaning cutting and final felling. As a result of the model process per-stratum data bases are obtained of the sample forest range for 10 steps of simulation (100 years) according to the two options of external effects.

Stand Stock Dynamics for the Territory as a Whole

In the first case of simulation (simulation of clear final felling), the dynamics of standing crop changes was closely connected with the dynamics of birch stand timber volume. In consequence of this felling, at the 8th step of the simulation a sharp increase of birch stock occurred. The share of valuable species -- spruce and seedling oak -- under such a regime of management remains at a very low level.

The second simulation differs, because in this scenario the simulation of cleaning cutting is added. The main difference of this variant from the previous one is the smooth increase in the share of valuable species (spruce and seedling oak) in the stand structure. It should be noted, that when executing cleaning cutting the birch and aspen stocks are at a lower level, than in the first case. Besides that, the

regular execution of cleaning cutting ensures a constant volume of cut wood of not less than 50 thousand cubic m for each step of the simulation.

Stand Stock Dynamics for Territories with Different Exploitation History

In connection with the different history of present forest territory use we have already a different composition (by stock) at the zero step. So, on durable forest lands the share of valuable and other species is 59% by stock, while on arable lands it is only 3%. The stocks of soft-wooded broad-leaved species on these lands are correspondingly 41% and 97% (fig. 3).

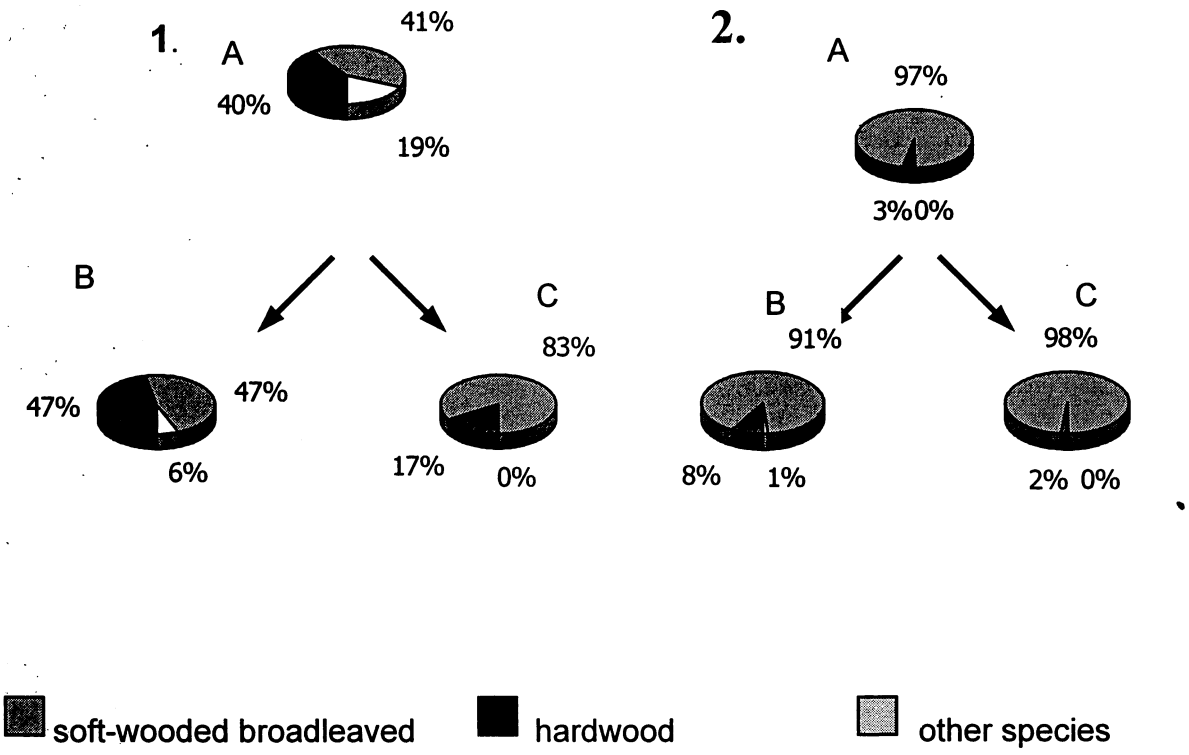

Figure. 3. -- *Stand species composition (by stock) on durable forest land (1), on arable land (2):*
A - zero step of simulation
B - 8-th step (with cleaning cutting and final felling)
C - 8-th step (only final felling)

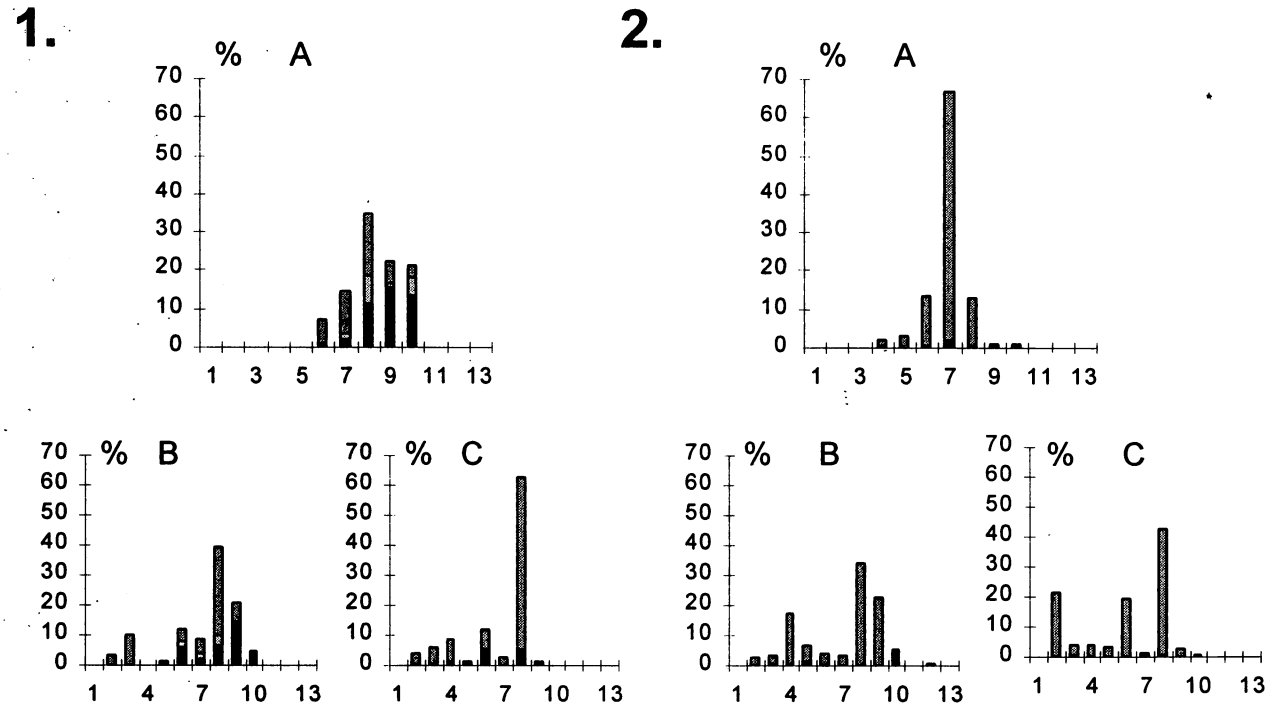

Figure 4. -- *Stand age structure (by area) on durable forest land (1), on arable land (2), the axes of x - age class, the axes of y - percent of area*

A - zero step of simulation

B - 8-th step (with cleaning cutting and final felling)

C - 8-th step (only final felling)

Such a distinction in initial simulation conditions influences the final results. Under equal scenarios of forest management, different results have been obtained for territories with different pre-history. So, on durable forest lands the share of valuable and other species make up 48% under final felling and cleaning cutting, and only 9% when cleaning cutting are absent. Just as on arable lands those shares make up 17% and 2%.

The simulation confirmed the observations from forestry data, that to obtain a greater share of valuable species, where there are none, it is necessary to plant forest cultures, especially on arable lands.

Forest Age Structure

For the present stand age structure (fig. 4) a sharp domination of 70-80 year old stands is typical that is due to the final cutting at the beginning of the XX century. For regular forest the area distribution by age classes is even. From the figures it is seen that the model well describes the goal-oriented human impact by means of cleaning cutting that leads to the improvement of the age structure of the valuable species. However, at first such treatments must be applied on arable lands (fig. 4).

CONCLUSION

The exploitation history of modern forest lands is essential for understanding the species composition of contemporary forest ranges and their age structure. Therefore, for rational forestry management all silvicultural treatment plans have to take into account the land exploitation history. The suggested mathematical model of heterogeneous uneven-aged stand dynamics may be used for elaborating impacts of human disturbance (in the framework of silvicultural treatments) and for forecasting stand growth both in nature and artificial forests in stable and disturbed habitats.

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Bredikhin, M.A. 1985. Dynamics model of forest growth. In: Mathematical modelling in biogeocenology. Petrozavodsk: Karelian branch Academy of Sciences of the USSR: 36-37. (in Russian).

Coenopopulation of plants (essays on population biology). 1988. Moscow: Nauka. 183 p. (in Russian).

Chumachenko, S.I. 1993. Basis model of dynamics of multispecies uneven-aged forest cenosis. Transactions of Moscow Forestry Institute. 248: 147-180. (in Russian).

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ROLE OF DISTURBANCE IN BOREAL FORESTS

Disturbance by Insects

THE INFLUENCE OF INSECTS ON BOREAL ECOSYSTEMS UNDER GLOBAL CLIMATE CHANGE

A.S.Isaev[1], T.M.Ovchinnikova[2] , E.N.Pal'nikova[3], V.G.Soukhovolsky[4], O.V.Tarasova[5]

ABSTRACT.--To model climate change impact on the population dynamics of forest insects it is proposed to consider the forest/insect system on different time scales. For every subsystem (forest and pest separately) it is possible to consider two characteristic time scales: large and small. Three possible models with different time scales were used for analysis of climate change impacts on the population of pine looper (*Bupalus piniarius* L.) that is a dangerous pest for the boreal forest of North Eurasia.

INTRODUCTION

The predicted global climate changes may results not only in changes in forest distribution over the Earth's surface but in changes of the character of forest-pest interactions. Forest-pest interactions and not the direct impact of climate may determine the future vegetation if pest outbreaks cause strong damage and elimination of forest stands. So it is important to study the possible climate change impact on the population dynamics of forest pests.

We want to analyze the different time scales in the forest-pest system. For every subsystem (forest and pest separately) it is possible to consider two characteristic time scales: large $t^{(L)}$ and small $t^{(S)}$. The characteristic time is correlated with the period of subsystem life that is 1-2 year for insects and several hundreds of years for forest stand. For insects $t_i^{(L)}$ corresponds to 10-100 generations, $t_i^{(s)}$ corresponds to one generation. For forest stands time $t_f^{(L)}$ and tree time $t_f^{(S)}$ may be determined analogously. The four possible variants of interactions of t_f and t_i are given in table 1.

In this work the proposed approach is used to analyze climate change impacts on the population of pine looper (*Bupalus piniarius L.*) that is dangerous pest for the boreal forest of North Eurasia. Models representing the four variants illustrated in table 1 are discussed below.

SIMULATION MODEL OF PINE LOOPER (*BUPALUS PINIARIUS* L.) DYNAMICS WITH ACCOUNT FOR CLIMATE CHANGE. $t_i^{(S)}$ - $t_f^{(L)}$

For every stage of insect population the abundance of the current and the next stages are connected by the equation:

$$N(i,j) = \Pi f(x(i,j,k,))*N(i,j-1)$$

where N(i,j) is the number of j- th stage in the i-th year. The value of $0 < f(x(i,j,k)) < 1$ is the proportion of survival of insects of the j-th stage in the i-th year after the action of k-th ecological factor. We assume that the mortality factors occur one by one (Pal'nikova *et al.* 1995).

One of the most important factors for decreasing the pine looper population is the weather. The key parameters are: the temperatures of May, July, September, October and December; the precipitation of May, June, July, September and December. We used also the hydrothermal coefficient (HTC) for each month, determined as a ratio of the monthly average precipitation to the monthly average temperature.

A Monte-Carlo procedure was used for weather simulations. A set of climate parameters for every year with given average and dispersion is a Monte-Carlo output. 100 different scenarios SC(i,j) with i-th average and j-th dispersion are presented in table 2.

[1] *A.S.Isaev. International Forestry Institute (Moscow), postmaster@spepl.msk.su*

[2] *T.M.Ovchinnikova and V.G.Soukhovolsky. 660036, Krasnoyarsk. V.N.Sukachev Institute of Forest ifor@krsk.infotel.ru*

[3] *E.N.Pal'nikova . Krasnoyarsk State Technological Academy*

[4] *T.M.Ovchinnikova and V.G.Soukhovolsky. 660036, Krasnoyarsk. V.N.Sukachev Institute of Forest ifor@krsk.infotel.ru*

[5] *O.V.Tarasova Krasnoyarsk State University*

Table 1.--*Different possible models of the climate change impact on forest-pest system.*

Pest	Forest: Small t_f	Forest: Large t_f
Small t_i	$t_i^{(S)} - t_f^{(S)}$ Models of tree response to leaf damage by insects	$t_i^{(S)} - t_f^{(L)}$ Simulation models of population dynamics, the search for analogues data in the past
Large t_i	$t_i^{(L)} - t_f^{(S)}$ Unreal	$t_i^{(L)} - t_f^{(L)}$ Biogeographical analysis

The values i,j correlate with different climatic zones in the Krasnoyarsk region: North (1,2), Low Angara (3,4), Central part near Krasnoyarsk (5,6), South (7- 10). Two locations have been used in our model: Krasnoturansk (South of Krasnoyarsk region) and Motygino (Low Angara). In Krasnoturansk the pine looper has in regular outbreaks in the pine forests. In Motygino there have been no outbreaks in over a large forest area according to long time investigations.

Typical outputs of the model for Krasnoturansk and Motygino are shown in fig.1. In Krasnoturansk quite regular peaks in the insect density occur with an average period of about 10 years; this model output with experimental observations of cyclic outbreaks of pine looper in the South of Siberia. For Motygino the average and maximum values of the insect density over a large time period are rather low and the cyclic oscillations are absent.

To reveal the periodicity in the population dynamics we have used a Fourier spectral analysis of the time series. In fig. 2 the spectral density of the scenario SC(10,10) with maximal temperature and dispersion is given. The clearly defined peak shows a periodic increase of population density with a period of 10 years, the large amplitude of the peak confirms that it describes the outbreaks. In fig. 3 the spectral density of the scenario SC(1,1) with minimal temperature and dispersion is given. The low value of spectral density in comparison with SC(10,10) and the smearing of the peak indicate the absence of regular oscillations of population density with small deviations from average number. It is explained by the colder and wetter climate.

A very important characteristic of the population dynamics and the insect effect on forests is the ratio of the number of years above the critical density of population in the outbreak regime to the total time interval under investigation. This ratio is not large for the populations with rare occasional outbreaks. On the contrary, this ratio is large for the populations with regular outbreaks. It is clear from fig. 4 that the warmer climates with higher scenario numbers experience more frequent outbreaks.

The difference of climate in the south of the Krasnoyarsk region and the region of the city of Krasnoyarsk is clearly shown by the HTC for May (2.8 for south and 4.5 for Krasnoyarsk). Under the warming in May of 2° C -3° C and the 20%-30% decrease in precipitation that is proposed by some scenarios of global climatic change, the central part of Krasnoyarsk region may become more susceptible to outbreaks of pests of the Central Siberia forest and the outbreak areas will increase. For the pine looper the border of this area may reach 56°-57° north latitude.

The dispersion dependence is more complicated. For SC(7-10,j) the increasing dispersion of climate parameters decreases the number of outbreak years while for SC(2-4,j) increasing increases the number of outbreak years. Scenarios SC(5-6,j) correspond to the intermediate dynamics without dependence on the dispersion. We conclude that for warmer climates the increasing dispersion results in stabilization of population dynamics and for colder climates it has a destabilization role.

BIOGEOGRAPHICAL ANALYSIS OF THE PINE LOOPER POPULATION $t_i^{(L)} - t_f^{(L)}$

An increasing area of insect habitat is one of the effects of the climate change on the insect population. So it is important to analyze the connection of local environment and the population numbers. The data on the population dynamics of philophagous insects Krasnoturansk (South of Krasnoyarsk region) in the period 1979-1994 have been analyzed by a multidimensional statistical method.

Table 2.--*Climatic data for model scenarios, average (a), dispersion (b).*

N	Temperature (C^0)			Precipitation (mm)			HTC		
scenario	July	October	December	June	July	December	May	July	September
				a) average parameters					
10	20	3	-12.2	20.1	33.1	7.4	2.7	2.7	2.7
9	19.6	2.4	-13.5	22.2	36.6	9	3	3	3
8	19.2	1.8	-14.9	24.5	40.4	11	3.3	3.3	3.3
7	18.8	1.2	-16.4	27.1	44.7	13.5	3.7	3.7	3.7
6	18.4	0.6	-18.2	30	49.4	16.4	4.1	4.1	4.1
5	18	0	-20.1	33.1	54.6	20.1	4.5	4.5	4.5
4	17.6	-0.6	-22.2	36.6	60.3	24.5	5	5	5
3	17.2	-1.2	-24.5	40.4	66.7	30	5.5	5.5	5.5
2	16.8	-1.8	-27.1	44.7	73.7	36.6	6	6	6
1	16.4	-2.4	-30	49.4	81.5	44.7	6.7	6.7	6.7
11[1]	19.4	2.7	-16.4	47	49.4	10	2.8	2.9	3.6
12[2]	18.2	-1.4	-20.9	32.1	49.4	32.1	5.5	5	6.1
				b) dispersion of logorithms of parameters					
1	1.5	3	0.01	0.2	0.6	0.1	0.2	0.2	0.2
2	1.6	3.4	0.06	0.25	0.55	0.15	0.25	0.25	0.25
3	1.7	3.8	0.11	0.3	0.5	0.2	0.3	0.3	0.3
4	1.8	4.2	0.16	0.35	0.45	0.25	0.35	0.35	0.35
5	1.9	4.6	0.21	0.4	0.4	0.3	0.4	0.4	0.4
6	2	5	0.26	0.45	0.35	0.35	0.45	0.45	0.45
7	2.1	5.4	0.31	0.5	0.3	0.4	0.5	0.5	0.5
8	2.2	5.8	0.36	0.55	0.25	0.45	0.55	0.55	0.55
9	2.3	6.2	0.41	0.6	0.2	0.5	0.6	0.6	0.6
10	2.4	6.6	0.46	0.65	0.15	0.55	0.65	0.65	0.65
11[1]	2	3.4	0.27	0.46	0.56	0.54	0.6	0.55	0.59
12[2]	1.7	5.3	0.06	0.21	0.26	0.16	0.54	0.25	0.27

[1] - Krasnoturansk
[2] - Motygino

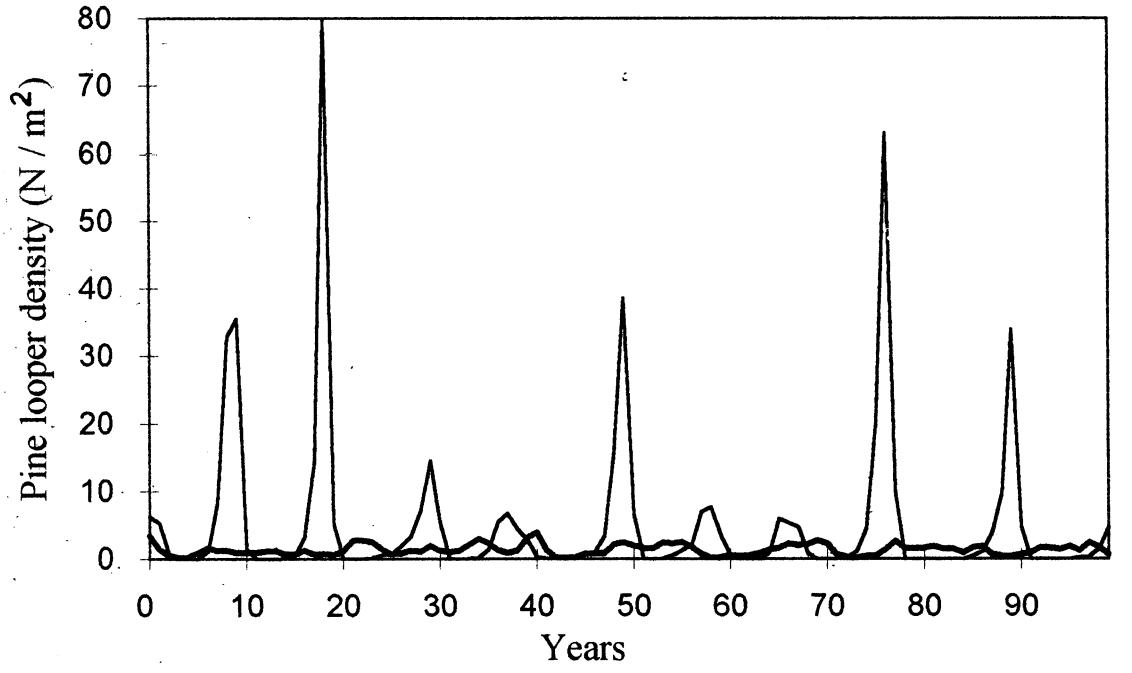

Figure 1.--*Typical outputs of the pine looper model for two climatic scenarios: Krasnoturansk (thin line), and Motygino (thick line).*

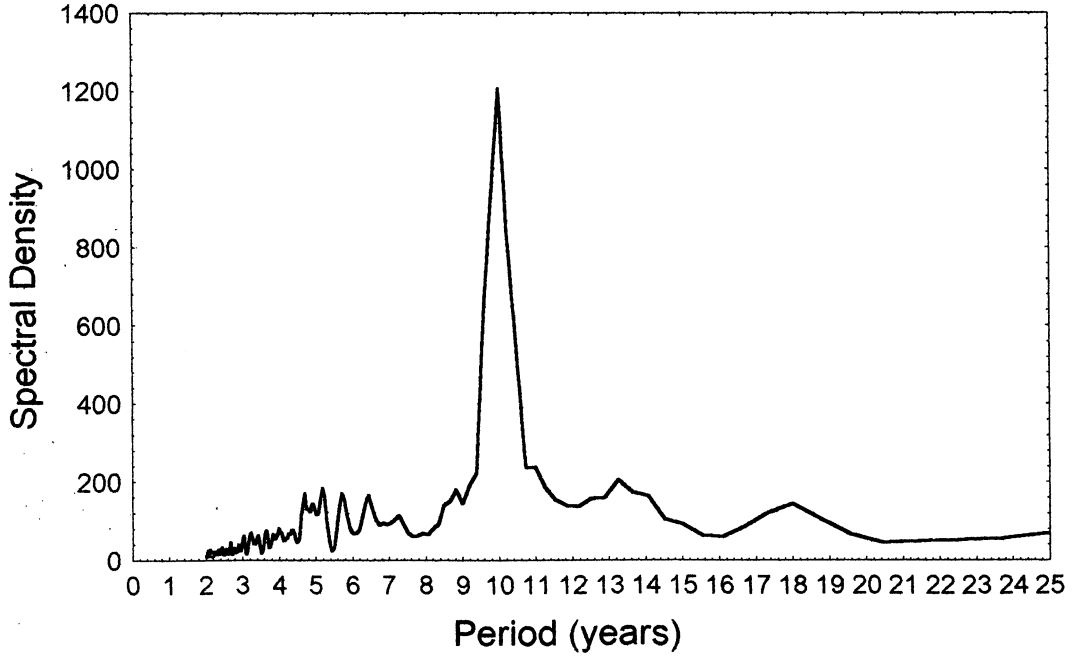

Figure 2.--*Spectral anlysis: SC (10, 10)*

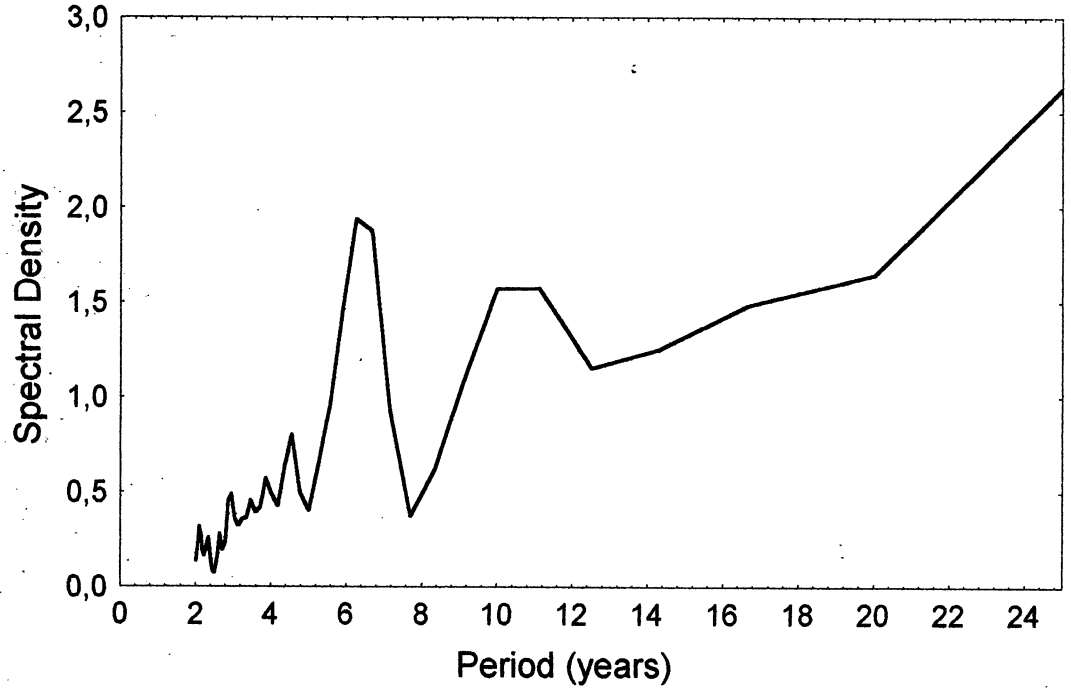

Figure 3.--*Spectral analysis: SC (1,1)*

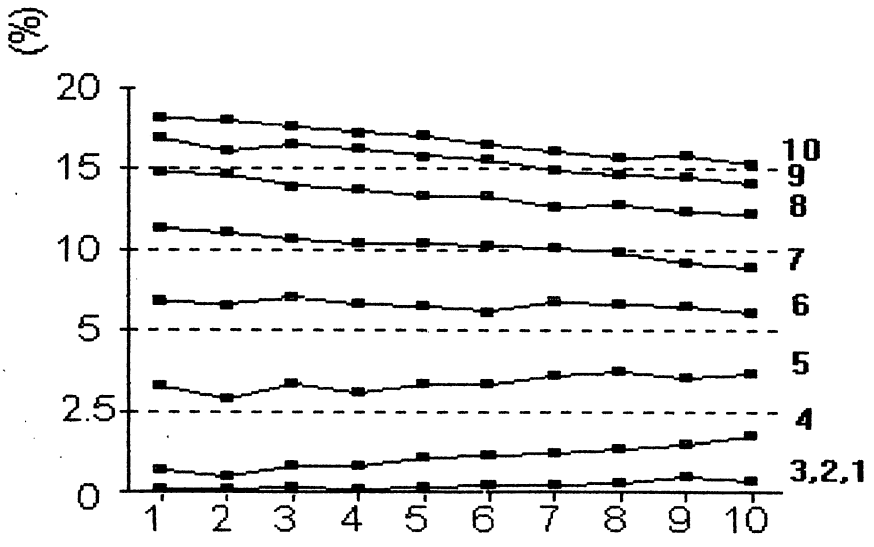

Figure 4.--*The ratio of number of years with population of pine looper pupae above the critical density 15 p/m^2 as a fraction of scenario dispersion number. Parameters at the right indicate scenario average number.*

Table 3.--*Parameters of habitat in Krasnoturansk pine forest.*

N	Parameter of habitat	Number of habitat I	II	III	IV	V
1	Maximum temperature of June	19	34	35	40	57
2	Maximum temperature of July	25	30	42	44	52
3	Maximum temperature of August	20	29	34	50	55
4	Maximum temperature of September	16	18	22	26	45
5	Relative humidity, %	84	79	68	66	66
6	Absolute humidity of soil, mm	124	13	5	4	3
7	Maximum daily difference of temperatures[1]	17	17	25	27	28
8	Altitude above the sea level, m	330	380	450	350	430
9	Exposition[2]	2	1	2	3	3
10	Steepness of slope[3]	1	2	1	2	3
	Stand parameters:					
11	Average diameter, cm	21	25	18	22	23
12	Height, m	20	20	17	18	19
13	Stocking	0.8	0.8	0.9	0.7	0.8

[1]Taken on soil surface.
[2]1- north slope, 2 - flat surface, 3 - south slope.
[3]1- slope <5°, 2 - slope 5°-10°, 3 - slope 30°-40°

The experimental areas in the Krasnoturansk forest have been chosen in 5 different landscapes (I - bottom, II - gently sloping, III - narrow watersheds, IV - dunes, V - tops and steep slopes of hills).

The group of pine pests include: pine looper (*Bupalus piniarius L).*, *Semiothisa liturata* Cl., *Dendrolimus pini* L., *Gilpinia virens* Kl. *and Hyloicus morio Rotsch.* et Gord. A number of landscapes, climatic and inventory parameters have been studied (table 3) by the principle component method (Isaev *et al.* 1997).

The first principal component accounts for 59.3% of the total dispersion of the habitats, the second component accounts for 22.7%. So the two first principle components take more than 80% of total dispersion. The correlation of the first principal components for every landscape with the multiyear values of the abundance of *Bupalus piniarius* L., *Dendrolimus pini* L., is shown in fig. 5.

The analysis of the contributions of different climatic parameters from table 3 to the 1st and 2nd principal components has revealed that in response to some microclimatic changes in different landscapes due to global climate change the redistribution of the pine looper to more uniform population density across various landscapes is possible. The increase of summer-autumn average temperature and decrease of precipitation is expected to result in increasing area of pine looper outbreaks. For example, a warming in summer-autumn months of 2^0 on average will increase the present optimal area for the pine looper from 660 ha up to 1,200 ha.

CLIMATIC CHANGE IN THE SOUTH OF KRASNOYARSK REGION AND ITS EFFECT ON PINE LOOPER OUTBREAK. $t_i^{(S)}$ - $t_f^{(L)}$

Anthropogeneous factors have induced some local climate changes that influence insect populations. The study of such territories can give information on climate change impact on population dynamics (Pal'nikova *et al.* 1996). One such territory is the Minusinsk valley in the South of Krasnoyarsk region where a large water reservoir of the 300 km length was established in 1964. The climate change due to the reservoir is the following: the average temperature and precipitation in period 1965-1988 is the same as in 1937-1965 years while the characteristics of associations between temperature and precipitation have changed greatly (fig.6). The percent of years with both dry and warm Augusts decreased from 44% in 1937-1964 years down to 13% after 1965, the percent of dry and warm Septembers decreased from 37% to 21% in the same time.

A dry and warm August-September period is one of the key factors associated with the pine looper outbreaks. After 1937 four large outbreaks of the

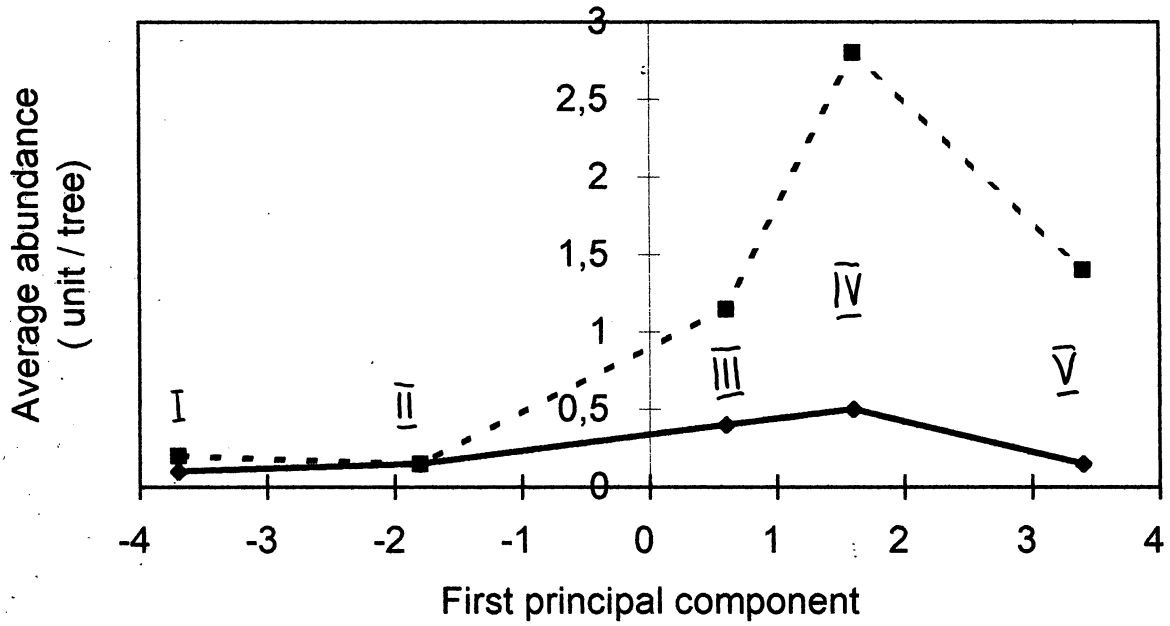

Figure 5.--*The average abundance of the* Bupalus piniarius L., *(dashed line) and* Dendrolimus pini L. *(solid line) as a function of the first principle component.*

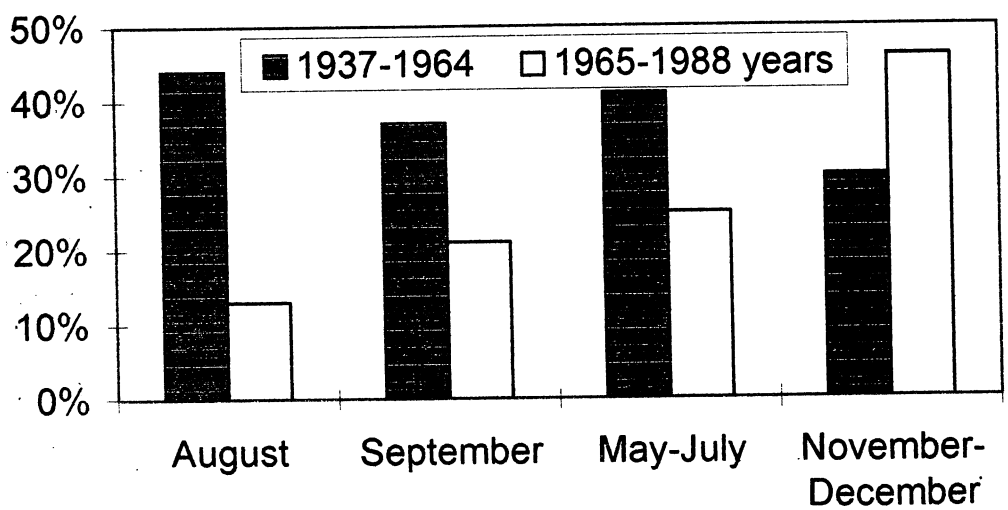

Figure 6.--*The changes in the ratio of dry and warm months before and after the water reservoir construction in 1964.*

pine looper have been observed in that area: in 1944, 1954, 1962 and 1974. Since 1974, pine looper outbreaks have not been observed. It is quite reasonable to associate this fact with the local climate changes that reduced the number of years, which were appropriate for the outbreaks.

THE MODEL OF LEAF RESPONSE ON THE DAMAGE BY INSECTS $t_i^{(S)}$ - $t_f^{(S)}$

It is known that fast changes in chemical composition of leaves occur in the process of interactions between trees and insects. These fast changes in leaves affect the insects' activity and decrease insect survival probability. Mathematically it is the case, when characteristic time periods of leaf state and insect state changes are equal and both small.

A model has been proposed to describe "leaf-insect" interactions, where the effect of leaf antibiosis is taken into account (Tarasova & Soukhovolsky 1996). It is assumed in the model, that the leaf may have two states: state s1, when its possible for insects to consume the leaf, and state s2, when, after increases in the levels of secondary compounds in the leaf, insects do not consume the leaf.

The flow of insect, which attacks the leaf, has the intensity λ and we assume that two or more insects do not attack one leaf simultaneously. We assume that the probability of insect attack at time moment t does not depend on past history of interaction. In this condition the distribution of attack number during time intervals (0, t) is described by a Poisson equation.

After insects successful attack and the same quantity of leaf biomass withdrawal the process of reparation in the leaf begins and for a certain time τ the insect does not attack the leaf. The value of the mean reparation intensity $\mu = 1/\tau$ distributed according to the equation:

$$(1)\quad f(\mu) = \mu * \exp(-\mu T)$$

Let p1(t) = the probability of leaf state s1 in time moment t, and p2(t) = the probability of leaf state s2 in time moment t. If we study a steady state "leaf-insect" system, it is possible to find the values of p1 and p2 using the Kolmogorov balance equation:

$$(2)\quad p1 = \mu/(\lambda+\mu); \qquad p2 = \lambda/(\lambda+\mu)$$

Really p1 and p2 are the parts of times, then the leaf have every of the states s1 and s2.

It's possible to calculate the number of leaves damage during time t:

$$(3)\quad N = \lambda\mu t/(\lambda + \mu)$$

If $\mu \to 0$, and the reparation time $\tau \to \infty$, the leaf after first insect attack has the state s2, when it is impossible for insects to consume the leaf tissue. If the process of leaf reparation is very fast, then $\mu \to \infty$ and $N = \lambda t$, i.e., the leaf damage number is proportional to the number of attacks during time period t. In this case the leaf is not resistant to insect attacks.

In this model it is possible to simulate the influence of climate changes on the character of "leaf-insect" interactions by changing the model parameters λ and μ in dependence of environment temperature T changes. Various trajectories of leaf damage number N(t) will exist depending on values of $d\lambda/dT$ and the sign of $d\mu/dT$ (fig.7).

I. $d\lambda/dT > 0$, $d\mu/dT > 0$. In this case according (3) $dN/dT > 0$ and with the temperature increasing the leaf damage number will be increase (fig.7, curve 1).
II. $d\lambda/dT < 0$, $d\mu/dT < 0$. In this case according (3) $dN/dT < 0$ and with the temperature increasing the leaf damage number will be decrease (fig.7, curve 2).
III. $d\mu/dT < 0$, $d\lambda/dT > 0$.
IV. $d\mu/dT > 0$, $d\lambda/dT < 0$.

In these cases in dependence of values λ, μ, $d\lambda/dT$ and $d\mu/dT$ various behaviour of N(T) curve is possible, for example, N(T) curve will have maximum or minimum (fig.7, curves 3, 4).

The analysis of "leaf-insect" interaction reparation model shows that in dependence of values of model parameters, $d\lambda/dT$ and $d\mu/dT$ sign it's possible both the increasing of leaf damage by insect and the decreasing of insect attacks intensity.

More detailed information on physiological processes in leaves and insects during temperature changes is necessary for precise estimation of the "tree-insect" interaction. But the model shows that increasing tree damage level and insect outbreaks are potentially possible during global climate changes. This work is supported by Russian Foundation for Basic Research,

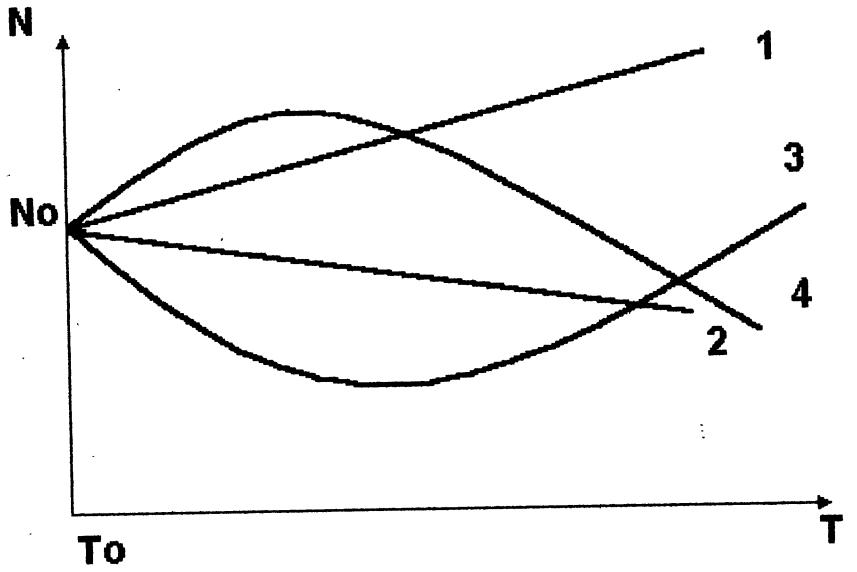

Figure 7.--*Dependence on temperature of intensity of the leaf damage by insects. See text for description of individual curves.*

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NOAA/AVHRR DATA IN DETECTION OF INSECTS INDUCED CATASTROPHIC CHANGES IN SIBERIAN TAIGA

V. I. Kharuk[1], A.S. Isaev[2], Yu P. Kondakov[1], A. G. Kozhuhovskaya[1], and E.V. Fedotova[1]

ABSTRACT.--In 1992-1996 ~0.7 million ha of *Abies sibirica*, *Pinus sibirica*, and *Picea obovata* stands along the Yenisey meridian were affected by *Dendrolimus sibiricus*. Historical aspect and causes of pest outbreaks were analyzed. Temporal (1995-1996) scenes of NOAA/AVHRR images were used for analysis of outbreak dynamics. Damaged stands were detected, and heavy (50%-75%) and very heavy (>75%) levels of defoliation were classified. Summer/winter images were used for delineation of the northern border of the outbreak region. Application of AVHRR images for GIS generation, insect "food base" estimation, and mapping potential risk of outbreaks is discussed.

INTRODUCTION

The Yenisey river meridian (~90° E) forests are the habitat of many insect species, periodic outbreaks of which caused decrease of the growth increment, forest decline, and mortality over vast areas. Pests are among the primary factors determining forest succession in Siberia. Table 1 lists the main *Lepidoptera* species which produce outbreaks in Siberian forests. The reported damage varies from 5,000 to more than 1 million ha. In table 1 the values of pest danger were categorized as high (I), medium (II) and low (III). Pest danger depends on the size of affected territories, level of defoliation, and tree species. Pests that damage evergreen conifers are considered very harmful (grade I), as the latter are highly sensitive to defoliation and are the source of valuable wood. Low-harm pests feed on broad leaf species and larch; which can survive after 2-3 consequent years of complete defoliation. Exceptions are the field protection forest strips because of their importance and because they are often growing in extreme conditions.

The primary enemy of Siberian forests is *Dendrolimus sibiricus*, or Siberian silkworm (table 1). This pest affects stands of fir *(Abies sibirica)*, Siberian pine *(Pinus sibirica)* (the preferred species) and larch *(Larix sibirica)*.

At the peak of an outbreak, spruce *(Picea obovata)* also could be damaged. A large-scale outbreak of Siberian silkworm was observed in the south of the Yenisey meridian (Niznee Priangar'e) in 1992-1997. The total outbreak area was about 700,000 ha and includes about 300,000 ha of 50%-100% defoliated stands. The mosaic of damaged stands is of geographic scale and lies between 57° and 59° parallels and the 93° and 98° meridians. The catastrophic scale of this outbreak needs adequate methods of detection and control. The purpose of this paper is the analysis of potential applications of broad-view satellite images for pest monitoring.

STUDY SITE AND METHODS

Historical data (1920-1995) of Siberian silkworm outbreaks were used for classifying the potential level of damage for the Yenisey meridian forests. During the latest *Dendrolimus sibiricus* outbreak the main damaged stands were concentrated in the region of Niznee Priangar'e, between the Yenisey and the Angara rivers (fig. 1). Relief of the territory is a plato with low hills. Soils are mainly podzol. Climate is continental. Annual precipitation is 400 mm-450 mm. Mean annual temperature is +2.6° Celsius with absolute minimum -54° (December) and maximum +36° (July). Vegetative period is about 100 days. Forest covers 95% of the territory. Dominant species is Siberian fir (*Abies sibirica*); other species are represented by Siberian pine (*Pinus sibirica*), spruce (*Picea obovata*), pine (*Pinus silvestris*), larch (*Larix sibirica*), aspen (*Populus tremula*), birch (*Betula verrucosa*). Stands are of 3-4 site index with wood stock 200 m^3-230m^3 and mean age 135 years. Time

[1] *V.I. Kharuk, Yu P. Kondakov, A. G. Kozhuhovskay, and E. v. Fedotova, Forest Institute, Academgorodok, Krasnoyarsk, Russia, 660036. E-mail: ifor@krsk.infotel.ru.*

[2] *A.S. Isaev ,International Forestry Institute (Moscow). Email: postmaster@spepl.msk.su*

series (1995-1996) of NOAA/AVHRR images on this area were used for analysis. On-the-ground resolution (pixel) of the images is about 1.1 km. Image processing was done by IDRISI software. Data from all AVHRR channels (0.58-0.68; 0.725-1.1; 3.55-3.93 μm, and thermal IR channel) and their combinations were analyzed. Supervised classification was based on Russian Federal Forest Service inventory maps. Ground truth data were obtained by standard technique and represent the following grades of defoliation: (A) two grades (50%-75%, and >75%) for the whole territory; (B) classification were obtained using NDVI (difference between second and first channels divided by its sum). The precision of classification was estimated by two methods. (1) Test sites were occasionally four grades (0%-25%, 25%-50%, 50%-75%, and >75%) for the selected part of it. The best results of distributed over inventory map and classified image. The percentage of omitted damaged sites and incorrectly included undamaged sites on the image were calculated. (2) Kappa statistic was calculated (Monserud and Leemans 1992).

Table 1.--*Lepidoptera species which cause outbreaks in the Yenisey meridian forests.*

	Insect species	Main food species	Max. outbreak territory, thousand Hectares	Danger level in Forest zones			
				South taiga	Mid taiga	Mountain forest	Forest/ steppe
1.	Dendrolimus sibiricus*	fir, pine, larch, spruce	> 1,000	I		I	I
2.	Lymantria dispar*	larch, broadleaf	300			II	II
3.	Orgyia antiqua	larch, birch	40			III	
4.	Dasychira abietis	spruce, fir, pine, larch	1,000	III		III	III
5.	Leucoma salicis*	aspen, willow	100	III	III	III	II
6.	Lymantria monacha*	pine, spruce	5				II
7.	Ectropis bistortata*	spruce	400	II		I	
8.	Bupalus piniarius*	pine	50	III			I
9.	Semiothisa signaria*	spruce, fir	10	III		II	
10.	Simiothisa continuaria	larch	5			II	III
11.	Erranis jacobsoni*	larch	50			II	II
12.	Biston betularius	birch	50				III
13.	Phalera bucephala	birch	50	III		III	II
14.	Clostera anastomosis*	aspen, willow	10	III			II
15.	Zeiraphera rufimitrana	fir	100	III		III	
16.	Coleophera dahurica	larch	100			III	II
17.	Zeiraphera griseana	larch	>1,000	III	III	III	

* Species which cause forest mortality.

RESULTS

Along the Yenisey meridian *Dendrolimus sibiricus* outbreaks were observed from the south border of the Krasnoyarsk region (~52°) till ~57° on the East side of the Yenisey meridian and to 62° on the west side. The analysis of ecology, landscapes, and history of outbreaks allows to delineate nine main regions of *Dendrolimus sibiricus* outbreaks (table 2, figure 1). The regions were ranged according to the following criteria:

1. The frequency of pest outbreaks. It varies within range from 2 to 10 per century. Frequency range 10-8 is grade I, range 7-5 is grade II, and range 4-2 is grade III.
2. The size of the territory (S). It was classified as small (grade III, S <10 thousand ha), big (grade II, 100>S>10 ha), and very big (grade I, S>100 thousand ha).
3. The "food species." Value of fir for *Dendrolimus sibiricus* was assumed as I, Siberian pine as II, and larch as III.

The total sum of these criteria was calculated and used for the ranking the regions. Values 3-4 correspond to the highest level of danger (I), 5-6 to the medium (II), and 7-9 to the lowest (III). Table 2 was used for generation of sketch-map of the level of pest outbreak danger in the Yenisey meridian forests (fig. 2).

The latest (1992-1997) *Dendrolimus sibiricus* outbreak falls mainly into region 2 (table 2). AVHRR images were applied for analysis of this phenomena. It has shown that (fig. 1):

1. The damaged territories could be detected.
2. At least two gradations of damage could be classified. The dynamics of outbreak during 1995 (the year of the most intensive damage) is presented on figure 1. Within the window on this figure the statistic of damage size was calculated (table 3). This table reflects dramatical changes in the size of the affected area. For example, very heavily damaged area (level of defoliation is >75%) increased from 884 pixels at the beginning of June 1995 to 3,053 pixels in the middle of August. Omission, inclusion, and summary errors were 7%, 10%, and 17%, respectively.

The classification procedure for figure 1 was made on the basis of a forest inventory map with two grades of defoliation: heavy (50%-75%) and very heavy (>75%). As was shown above, two gradations could be satisfactorily detected using AVHRR data. Next task was the following: if less defoliated stands could be detected? This problem was analyzed using a map (fig. 3A) with four levels of defoliation: slight (0%-25%), moderate (25%-50%), heavy (50%-75%) and very heavy (>75%). Processed image (fig. 3B) represents all levels of defoliation. There is some visual similarity between on-ground (fig. 3A) and remotely obtained data (fig. 3B). On the other hand only very heavy levels of damage have satisfactory kappa statistics (kappa = 0.35). Poor statistics for heavy level may be explained by time shift between satellite (16 August 1995) and on-ground (1 August 1995) data obtained: during this period the level of defoliation could increase. Moreover, the date of the map generation evidently does not correspond to the dates of on-ground measurements, since the latter was the durable process. All this led us to conclude that the lowest detectable level of defoliation is about 25%-50%, although it must be proved in future investigations.

Regular application of remote sensing for pest monitoring is based on summer images only, though winter images have some potential advantages (e.g., lack of errors caused by broad-leaf species and grass communities). To estimate the value of winter images, the February 1997 image was used for analysis; it includes the same territory as on figure 1. Figure 4 represents the result of its classification. Damaged stands are clearly detectable on the snow background, and three levels of the damage are classified. The precision of classification was not evaluated. Visual comparison with a summer image (fig. 1C) shows a similarity between them.

Table 2.--Dendrolimus sibiricum *outbreaks in the forests of the Yenisey meridian.*

Regions	Forest types	Heights above sea level, m	Main species	Territory of outbreaks	Frequency of outbreaks	Level of danger
	West Siberian plain region					
1. Chulim-Ket	south taiga	150-200	fir, Siberian pine	I	10	I
	Mid Siberian Plato region					
2. Priangar	south taiga	170-350	Fir	I	9	I
3. Priyenisey	mountain taiga	200-450	Fir	I	7	I
4. Kan-Birusa	subtaiga-foreststeppe	200-350	Larch	III	6	III
	Altai-Sayan mountain region					
5. Kan-Agul	subtaiga-foreststeppe	350-500	Larch	III	7	III
	black taiga	450-800	fir, pine	II	9	II
6. Kuznetz-Alatau	subtaiga-foreststeppe	500-900	Larch	I	10	II
7. Sisim-Tuba	black taiga	500-800	Fir	II	4	II
8. West Sayan	black taiga	350-900	Siberian pine Fir	III	3	III
9. Usa	subtaiga-foreststeppe	700-1,200	Larch	III	7	III

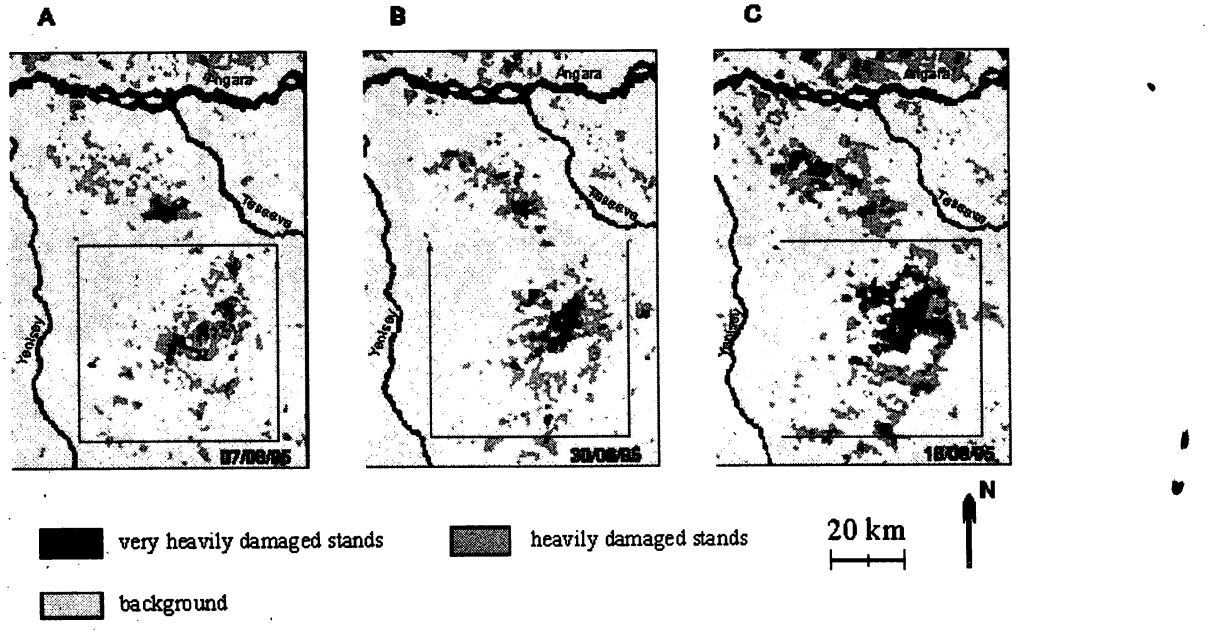

Figure 1. *The dynamics of the* Dendrolimus sibiricus *outbreak.*

Table 3.--*The dynamics of the damage within the window on figure 1.*

	Area in pixels		
Date	7 June 1995	30 June 1995	16 August 1995
Very heavily damage	884	1,261	3,053
Heavily damage	3,255	3,255	3,963
Background	23,734	23,180	20,362

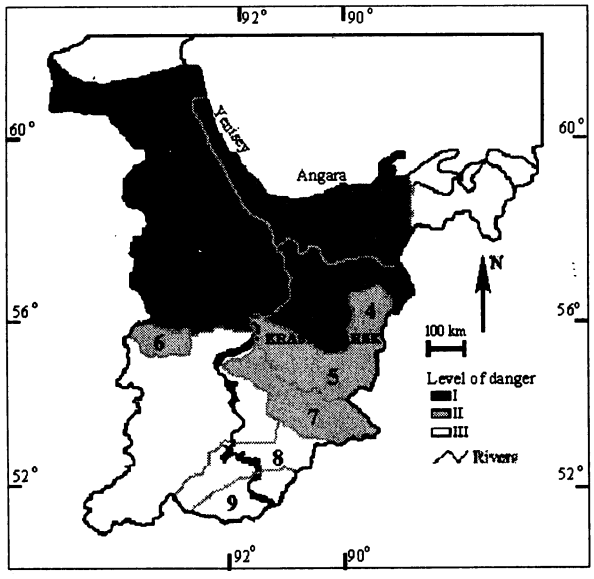

Figure 2.--*Sketch-map of* Dendrolimus sibiricus *outbreak danger in the regions along the Yenisey meridian (see also table 2).*

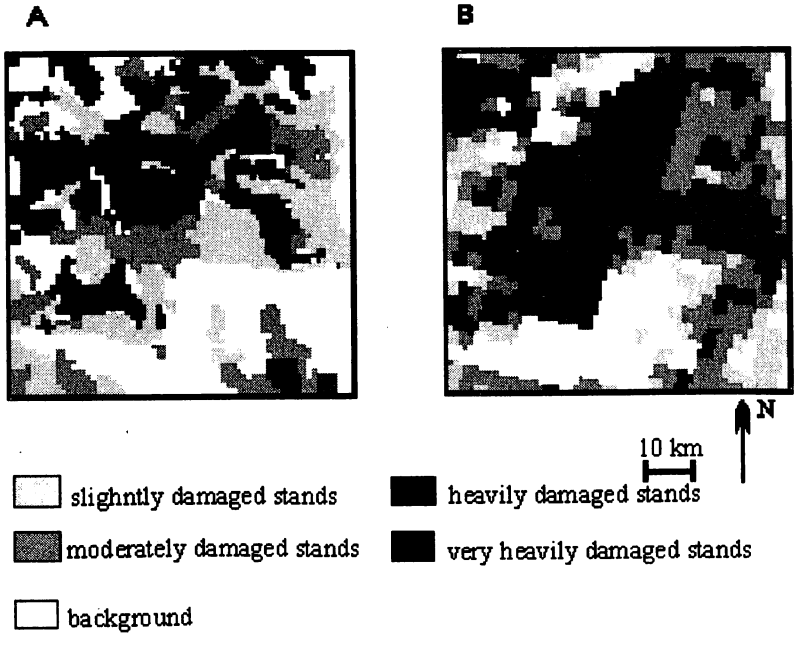

Figure 3.--*Fragments of inventory map (A) and classified image (B).*

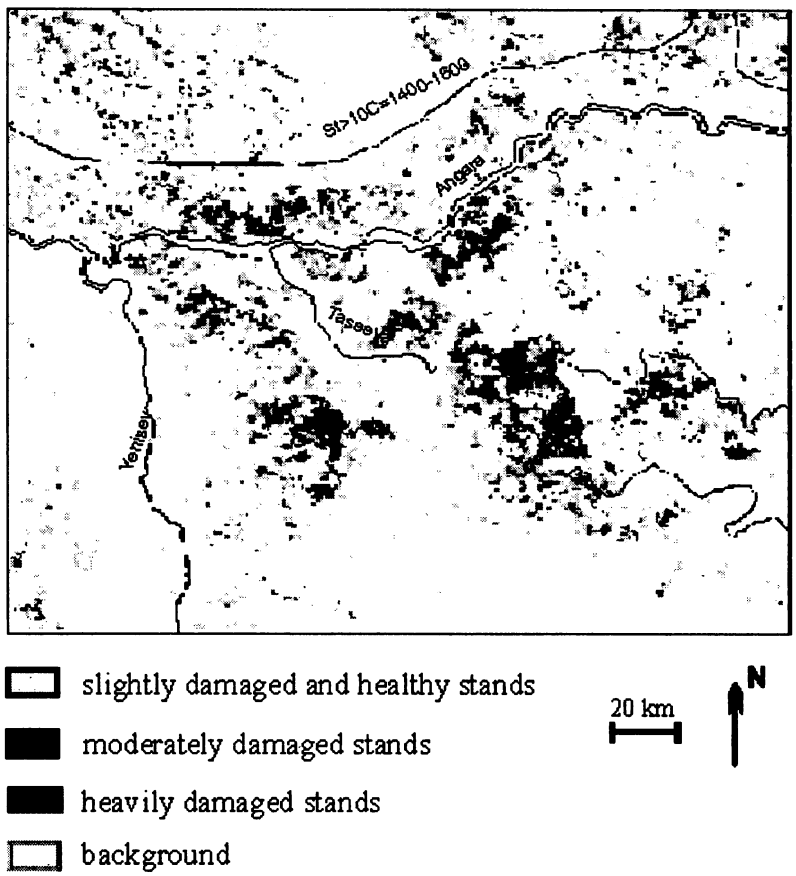

Figure 4.--*Damaged areas on the winter AVHRR image. Upper line is the warmness index isotherm (1,400C° - 1,600C°.*

The investigated outbreak covers the territory near the northern border of a Siberian silkworm outbreak area. Weather conditions (temperature and humidity) were optimal for maximum spread of outbreak, and forest protection measures were unsatisfactory. We consider that during this outbreak the pest "reveals" its northern border of potential outbreaks. This border is clearly detectable on both summer and winter images (fig. 1, 4). The area of pest outbreaks is controlled by climatic factors, such as an index of warmness and amount of precipitation. The northern border could be approximated by a warmness index, the sum of the positive (>+10 C°) temperatures with values 1400C° -1600 C° (fig. 4); this line actually corresponds to the historical data of *Dendrolimus sibiricus* outbreaks.

DISCUSSION

The sketch-map of *Dendrolimus sibiricus* outbreak danger in the Yenisey meridian forests (fig. 2) is based mainly on historical data. It should be useful as general direction for pest outbreak danger. For modifying this map for operative purposes it is necessary (1) to know real food base for the insect, and (2) to include data of actual weather. Food base could be relatively easily estimated since "black needle" taiga (fir + Siberian pine) is detectable on the AVHRR images. Analysis of insect food base using GIS technologies (which includes cutting of the territories with not appropriate climatic conditions, e.g., high altitude zones), will be useful for generation of the operative maps of potential outbreaks.

Climatic data (indexes of warmness and dryness) could be taken from mid (3.55-3.93 mkm) and thermal (10.3-11.3, 11.5-12.5 mkm) IR channels. Generally speaking, potential AVHRR data application is the analysis of ecological and climatic conditions of the given territory, and classification of the territory from the point of view of its optimality for the pests.

The analysis has shown that NOAA/AVHRR channels 1 and 2 are the most informative in detection of damaged stands. The first one (0.58 mkm-0.68 mkm) corresponds to the absorption band of chlorophyll. The signal in the second one (0.725 mkm-1.1 mkm) is controlled by reflectance/scattering on leaf microstructures. Therefore, defoliation causes decreasing of the signal in this band. Defoliation and yellowing of the needles are equivalent to the loss of chlorophyll; this process is indicated by the signal level in channel 1.
The sources of the errors in the AVHRR analysis are the following:

1. Forests are non-homogeneous in comparison of AVHRR on-ground resolution (pixel) (about 1 km). Each pixel could include different kinds of the objects.
2. There are some principal differences between on-ground criteria and remotely obtained data:
   2.1. Forest inventory is essentially a subjective process. The levels of defoliation or discoloration were estimated visually. Some important parameters of outbreak could be missed. For example, initial phases of defoliation; it's known that this process starts from the tops of the trees. On the contrary, very heavy damage could be precisely evaluated by on-ground investigations; that's why the kappa statistics is the best for that kind of the damage (fig. 3B).
   2.2. Even completely defoliated stands have NDVI typical of chlorophyll containing objects because of:
      2.2.1. A mixture of broadleaf species and pine;
      2.2.2. On-ground vegetative cover;
      2.2.3. Presence of chlorophyll in the bark tissues. It should be noted that bark photosynthesis could be significant for a tree's survival after defoliation (Kharuk *et al.* 1996).
3. The analysis of the temporal row of the images could be complicated by fenology. For example, figure 1A corresponds to the very beginning of the vegetative period, when leaves and needles start to flash and grass communities start to grow. Consequently, input of spectral signatures are considerably higher for images taken later (B, C).
4. On-ground data collection takes time. Consequently, the dates of map generation regularly do not coincide with the dates of remotely obtained data. This could create dramatic discrepancies between on-ground and satellite data. For example, at the density of 500-600 caterpillars per tree the stand could be completely defoliated during 5-10 days. This density was typical of the outbreak in 1995. Maximal density reached 12,000 caterpillar/tree; for comparison, in the primary reservation of Siberian silkworm during period between outbreak this density is only 3 caterpillars per 10 trees.

All this supports an idea that actual precision of satellite generated maps is higher than it was shown by their comparison with on-ground data.

AVHRR information on post *Dendrolimus sibiricus* outbreak situation is very important for estimating the danger of secondary damage of the weakened forests by bark beetles. One of the most dangerous species is *Monochamus urussovi*, fir bark beetle. Outbreaks of this pest spreads over the territory up to 1,000,000 ha. Estimation of the "food base" for that species is the priority task of future work.

Finally it should be noted that there is similarity in AVHRR image application to monitoring of pest outbreaks, from one side, and pollution induced forest decline, from the other side. It was shown (Kharuk *et al.* 1995) that AVHRR images were a valuable tool in analysis of large scale forest decline caused by Norilsk smelters. Since in both cases the general symptoms of decline are similar - loss of chlorophyll and biomass - similar approaches could be used for AVHRR data analysis.

In the system of pest monitoring NOAA/AVHRR data are "at the top" of the hierarchy. They give foresters generalized information on the scale of pest outbreak, food resources for the insects, and ecological-climatic parameters of the inspected territory. That these data are an adequate tool for detectable size of outbreak depends on the level of defoliation and homogeneity of the damaged stands and could be estimated as 10,000 ha-20,000 ha. Potentially, outbreaks of many species listed in table 1

could be detected by AVHRR technique. First phase of large scale outbreaks of *Dendrolimus sibiricus* cover ~20,000 ha. Naturally AVHRR data should be accompanied with information of higher resolution, such as aiborne, SPOT or Landsat data, and on-ground observations. AVHRR data are especially necessary for monitoring Siberian forests which are of large geographic scale, and the former system of monitoring which has collapsed.

CONCLUSIONS

1. NOAA/AVHRR images could be used for analysis of large scale *Dendrolimus sibiricus* outbreaks. At least two levels of damage (50%-75% and >75% of defoliation) could be detected. Since the pattern of the damage is similar for different pests, this method could be applied for monitoring of the other species.
2. A sketch-map of *Dendrolimus sibiricus* outbreaks danger in the Yenisey meridian forests was generated. It is based on the frequency of outbreaks, size of the damaged territory, and tree "food species".
3. Catastrophic pest outbreaks delineated the northern border of the pest outbreaks area. This border could be approximated by the isotherm of index of warmness equal to 1400 C^{o} -1600 C^{o}.
4. Winter images could be used for detection and analysis of the damaged stands.
5. Since general phenomena of forest decline caused by pest invasions and air pollution are similar (loss of chlorophyll and biomass), similar algorithms of image processing/analysis could be used in the both cases.

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CHARACTERIZING SPRUCE BUDWORM CAUSED DISTURBANCE REGIMES IN ONTARIO

Jean-Noël Candau[1], Richard A. Fleming[1], Anthony Hopkin[1] and Ajith H. Perera[2]

ABSTRACT.--It has long been recognized that disturbances such as insect outbreaks and fires play an important role in wild forest ecosystems. These natural disturbances instill resilience and sustainability, and provide habitats and niches for rarer species. For these reasons, a central concern in managed forests is that management practices do not inadvertently exceed some unrecognized natural threshold and thus undo or eliminate much of the positive contribution that natural disturbances make to forest health. Hence, current policy aims at developing a thorough understanding of natural disturbance dynamics and their patterns in both time and space.

The research reported here represents a first step towards developing such understanding for natural spruce budworm (*Choristoneura fumiferana* Clem. [Lepidoptera: Tortricidae]) disturbances in Ontario. The simplist approach to this goal might be a broad statistical analysis of current disturbance patterns, but because fire suppression and pest control activities have been part of forest management in Ontario for decades, current disturbance patterns, like current forest structure, should not necessarily be considered 'natural'. Nonetheless, the processes of insect outbreak (or fire) growth and decline, spread and dissolution, are thought to be essentially unchanged (except when and where intervention has occurred).

We discuss the early stages of an approach to deriving spatially explicit simulation models of landscape dynamics which predict the effects of these processes for insect disturbance systems. The focus is on the spruce budworm, the dominant insect disturbance factor in Ontario's boreal forest landscape. The spruce budworm's disturbance regime is described and then analyzed so as to reveal characteristics of the underlying dynamic processes. It is shown how these characteristics can be used to provide a skeleton for model development.

INTRODUCTION

The spruce budworm, *Choristoneura fumiferana* Clem. (Lepidoptera: Tortricidae) is the most damaging defoliator in the boreal North American forest. This insect is characterized by widespread outbreaks which can cause considerable tree mortality in balsam fir and white spruce stands and sometimes cause significant damage to black spruce, red spruce, and subalpine fir (Brown 1970). A variety of approaches have been used to analyze the historical records of spruce budworm outbreaks (Fleming *et al.* 1984). Opposing theories suggest that spruce budworm outbreaks spread contagiously (Holling 1986, Jones 1979, Stedinger 1984) or are the result of simultaneous cycles in population density over vast areas with the amplitude of the cycle depending on local conditions (Royama 1992). We are in the process of developing a spatially explicit model for simulating the damage (i.e., defoliation) patterns caused by outbreaks at the landscape level. As a first step, we have constructed a comprehensive spatial data set of spruce budworm defoliation. Here we present the results of an initial statistical analysis of these data.

METHODS

It is anticipated that much of the raw data for building the planned model will be extracted from maps of aerial surveys of spruce budworm defoliation. These maps were digitized and saved in a common geographical database to ensure consistency. Other kinds of information were derived from the original maps using various computational and geographical operations.

Aerial Surveys

Since 1941, FIDS (Forest Insects and Diseases Survey group of the Canadian Forest Service) has been conducting aerial reconnaissance of large-scale defoliation events in all of Ontario's productive forest in the exploitable forest area. Ideally, the reconnaissance flights begin as soon as the current season's defoliation is completed. Depending on the

[1] *Canadian Forest Service. Great Lakes Forest Research Centre. 1219 Queen Street East, P.O. Box 490. Sault Ste. Marie, Ontario P6A 5M7. Canada.*

[2] *Ontario Forest Research Institute. 1235 Queen Street East, Sault Ste. Marie, Ontario P6A 5N5. Canada.*

weather, this is usually in mid- to late-July. Survey planes usually fly at about 170 km/h with their height varying between 360m-600m. Survey lines are typically 6-10 km apart. In the aircraft, defoliation is delineated on 1:125,000 or 1:250,000 maps and this information is later transferred to smaller scale base maps (e.g., 1:500,000). Allen *et al.* (1984), Dorais and Kettela (1982), and Sanders (1980) provide additional technical details about aerial sketch mapping and describe some of the uncertainties associated with this data collection technique.

Database Construction

Historical maps of defoliation were digitized and saved in a geographical information system (GIS). The complete database is organized in 56 layers of annual defoliation covering the years 1941-1996. During the process of digitizing, each defoliated area was considered as a polygon. The coordinates and degree of defoliation of current foliage associated with this polygon were recorded. Three degrees of defoliation of the foliage produced in the current year were commonly adopted: light, moderate and severe. Light defoliation corresponds to 11% to 25% loss of current foliage, moderate to 26% to 70%, and severe to more than 70%. Light defoliation has been inconsistently recorded in the past, due mainly to difficulties in surveying this type of damage, and hence, usually has the highest degree of associated error (MacLean 1996). For this reason, we consider only areas of moderate and severe defoliation in what follows.

The GIS database was implemented under Arc/Info on Unix workstations. Other layers added to the database include the provincial boundaries, lakes, and river contours.

Derived Maps

New maps were built from the original dataset using spatial operations such as those calculating the intersection and union of several layers. The map of defoliation time series was created by building the union of the 56 layers of annual damage from 1941 to 1996. In this map, each polygon is associated with a series of 56 numbers corresponding to the defoliation status of the polygon each year.

RESULTS

Most of the results presented here were obtained from the map of defoliation time series. The advantage of this map resides in its polygon topology which ensures computations that preserve all information. Depending on the scale and other factors, grid or point topologies often involve some loss of information. The main inconvenience of polygon topology resides in the variability in polygon areas, and consequently, in their relative representation. One way to overcome this difficulty is to weight each polygon's frequency of recorded defoliation by its area. This operation leads to distributions for which frequencies are not expressed as numbers of polygons but as total areas of these polygons. These distributions are referred to as "weighted distributions".

Cumulative Frequency of Defoliation

For each polygon of the map of defoliation time series, the cumulative frequency of defoliation was calculated as the number of years when the polygon was defoliated. The "weighted distribution" of the cumulative frequency of defoliation is plotted as figure 1.

The cumulative frequencies of defoliation range from 1 to 25 years of defoliation over the 56 year period covered by the database. The "weighted distribution" (fig. 1) appears complex. Even if the primary event (defoliation/no-defoliation) is binary, the frequency distribution is obviously not binomial. This is not unexpected because outbreaks often last 5-15 years. Consequently, the defoliation state of a polygon in year t is not independent from its state in the previous year, t-1, or from the state of neighboring polygons in years t and t-1.

The distribution (fig. 1) may represent the sum of two binomial distributions with different parameter values. For instance, a binomial distribution with a low probability of defoliation (approximately 2%) may dominate in polygons defoliated less than 3 years. Polygons more frequently defoliated seem to follow another binomial distribution with a higher probability of defoliation (approximately 20%). The precise value of 2% may be an artifact of the data in the sense that only polygons which were defoliated at least once in the 56 years of observation (or at least 2% of the time) were considered in the analysis. The 20% applies to most of the area in our analysis, the area in which the outbreak cycle of the spruce budworm presumably runs its course. In New Brunswick, this cycle is approximately 35 years long and high population densities typically last 5-10 years (Royama 1992). The midpoint of this interval of high populations (i.e., 7.5 years) is about 20% of the length of this cycle. It is uncertain whether these different probability values (2% and 20%) reflect

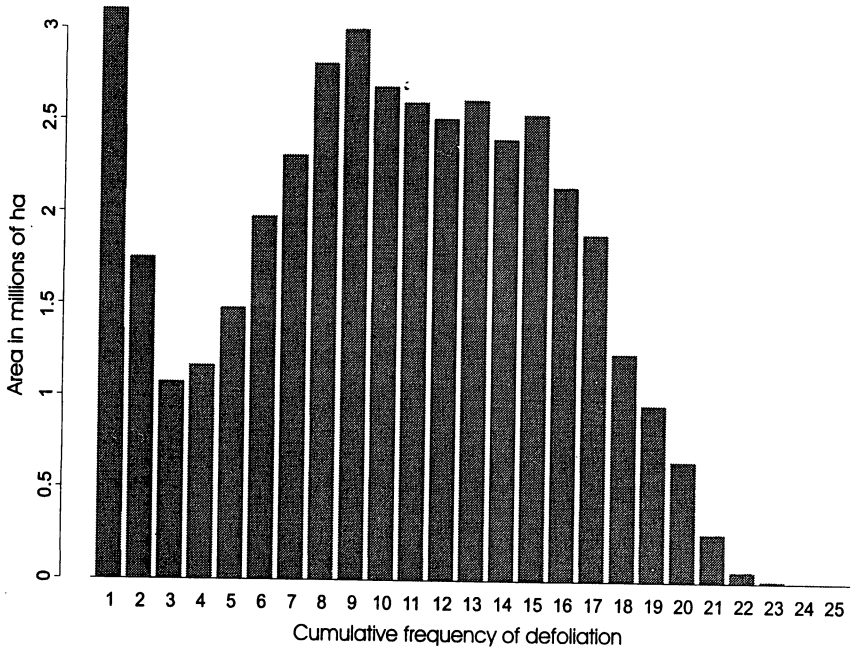

Figure 1.—*The "weighted distribution" of the cumulative frequency of defoliation*

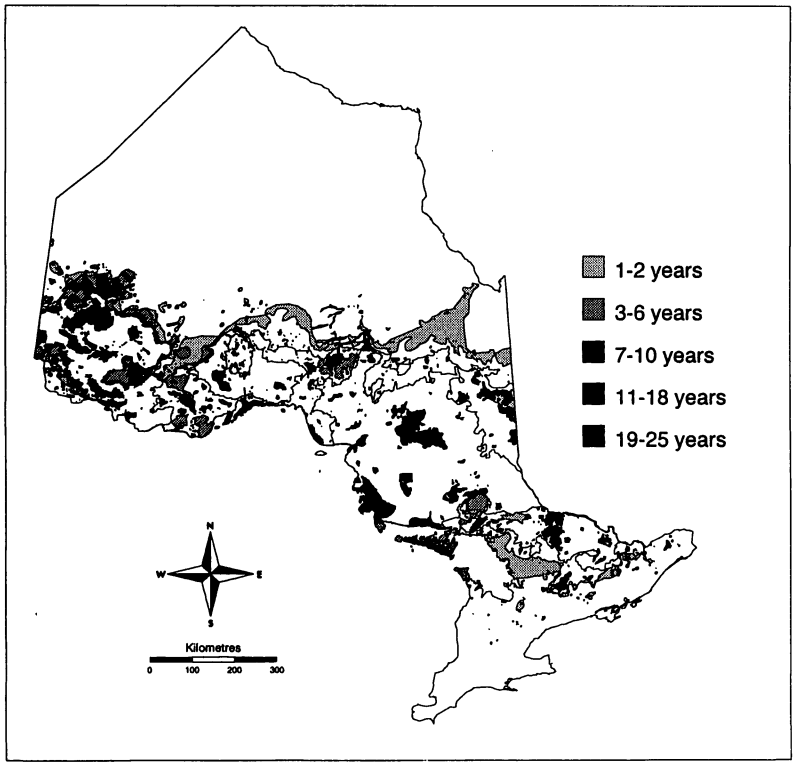

Figure 2.—*A map of cumulative frequencies of defoliation.*

only differences in the ecology of the areas concerned or whether differences in sampling intensity are involved as well.

For further analysis, the cumulative defoliation frequencies were grouped into the five classes: 1-2, 3-6, 7-10, 11-18 and 19-25 years of moderate or severe defoliation. The choice of class widths was based on the binomial distribution. If, for each polygon, the event "defoliation/no-defoliation" was independent from one year to the next, then the cumulative frequency would follow a binomial distribution and the class widths defined above would ensure that the frequencies of any particular class are significantly different (a=0.05) from those of its neighboring classes once removed. For instance, the class in which defoliation was most common (19-25 years) would not be significantly different from its immediate neighbor (11-18 years), but would be significantly different from the three most seldom defoliated classes. Since the distribution of cumulative defoliation frequency is not strictly binomial, our classification scheme must be viewed as a first approximation.

A map of cumulative frequencies of defoliation (fig. 2) was established using the classification scheme discussed above. The defoliation follows a belt located between the 45th and the 52nd parallel and there appear to be spatial patterns in the cumulative defoliation frequencies within this belt. Relatively small areas of very frequent defoliation (19-25 years) are surrounded by regions where frequencies decline approximately with distance from these 'foci'. The northern and southern fringes of the belt are rarely defoliated. In the north-east, defoliation was recorded along the banks of the Moose river just once, in 1979. This helps explain the unusual spatial extent of the class with 1-2 years defoliation in the north-east (fig. 2) as well as the large area defoliated just one year (fig. 1).

Fig. 2 provides additional information about the results in fig. 1. The map of cumulative frequencies (fig. 2) shows that areas rarely defoliated (e.g., 2% of the time) tend to located at the periphery of the defoliation belt whereas virtually all the area of the main body of the belt is defoliated much more often. This observation supports the idea that the dynamics of the defoliation process may vary from one part of the defoliation belt to another. As a result, a credible model should allow for spatial heterogeneity in the outbreak dynamics.

Analysis of Defoliation Sequences

The duration and frequency of defoliation and no-defoliation periods were calculated for each polygon of the map of defoliation time series. A defoliation period is defined by one year or several consecutive years of defoliation, preceded and followed by at least one year without defoliation. A no-defoliation period is defined conversely. The "weighted distributions" of these variables are plotted in figure 3.

The defoliation periods ranged from 1 to 17 years. These periods can be compared to the 4-5 consecutive years of 100% current defoliation that it takes to kill balsam fir. White and black spruce are less vulnerable (Morris 1963) so the longer periods of defoliation may be concentrated in stands of these species.

The "weighted distribution" of periods of no defoliation appears to be multi-modal. The shortest durations (from 1 to 6 years) can be attributed to variations in the extent of defoliation during the course of an outbreak, whereas longer durations (from 23 to 44 years) probably correspond to the period of time separating outbreaks. These long periods without defoliation appear to be separated into groups with peaks at 26, 36 and 42 years. Each peak might correspond to a different outbreak period; further investigations are necessary to establish this. For example, the polygons corresponding to each peak could be mapped to see whether each peak corresponds to a different area in the province.

On a province-wide scale, there have been two broad time-intervals in which spruce budworm outbreaks have occurred since 1941, with peaks of defoliation in 1945 and 1980 (Kettela 1983). The "weighted distribution" of the number of defoliation periods (fig. 4) is consistent with this observation since the highest "weighted frequency" was found for two periods. One and three periods of defoliation are also found frequently. These results are preliminary, a statistical analysis of the time series of defoliation is needed to draw more precise conclusions.

SUMMARY AND CONCLUSIONS

The map of defoliation time series proved to be a useful starting point for the analysis. The polygon topology of this map permitted computations that preserve most information. (By contrast, grid or

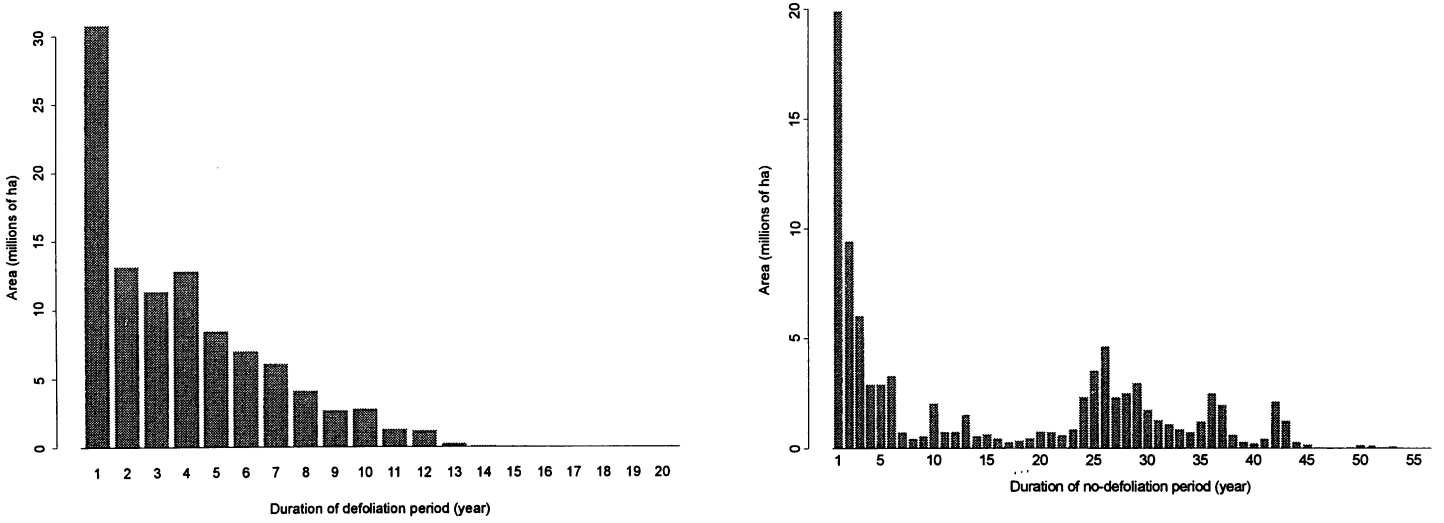

Figure 3.--*Distribution of defoliation and no-defoliation periods in the polygons of the map of cumulative defoliation. The frequency is weighted by the area of the polygon to take into account the differences in sizes.*

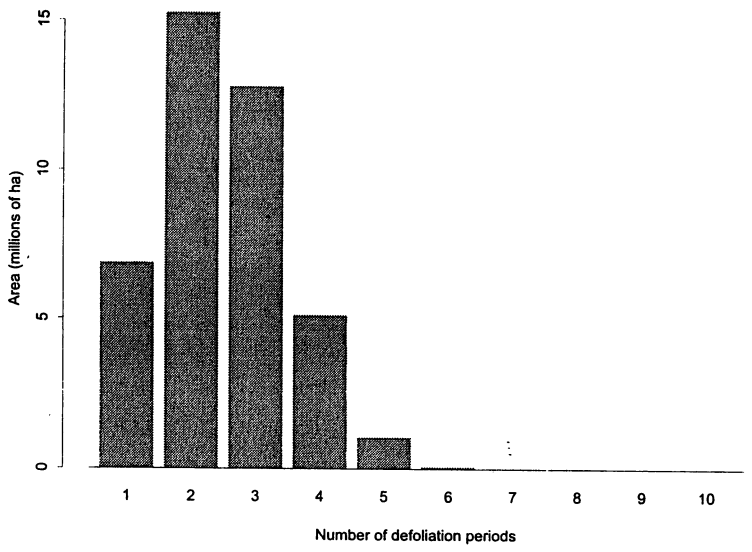

Figure 4.—*The "weighted distribution" of the number of defoliation periods.*

point topologies necessarily involve much greater information loss). To determine how large an area was defoliated how often, each polygon's defoliation frequency was weighted by its area. This showed that some areas were defoliated as often as 25 of the 56 years covered by the database.

The relationship between total area and frequency of defoliation is complex. It suggests the sum of two binomial distributions with different parameter values. For instance, a binomial distribution with a low probability of defoliation (about 2%) may dominate in polygons defoliated less than 3 years. More frequently defoliated polygons seem to follow another binomial distribution with a higher probability of defoliation (about 20%). These probabilities are approximately equal to the reciprocal of the 56 year length of the observation period (2%) and the proportion of the spruce budworm cycle when its populations are high.

A map of cumulative frequencies of defoliation showed that the defoliation follows a belt running across Ontario between the 45th and the 52nd parallels. Within this belt, relatively small areas of very frequent defoliation (19-25 years) are surrounded by regions where frequencies decline approximately with distance from these 'foci'. The northern and southern fringes of the belt are rarely defoliated. These observations suggest that the dynamics of the defoliation process may vary from one part of the defoliation belt to another. Consequently, a credible model should allow for spatial heterogeneity in the outbreak dynamics. In some polygons defoliation occurred up to 17 consecutive years. The distribution of the durations of intervals when no defoliation was observed appears to be multi-modal. The shortest durations (from 1 to 6 years) can be attributed to variations in the extent of defoliation during the course of an outbreak, whereas longer durations (from 23 to 44 years) probably correspond to the time between outbreaks.

These results are part of a preliminary study to analyze spruce budworm defoliation patterns at the landscape scale. Further statistical analysis is needed to draw more precise conclusions. Ultimately, the information gathered will be used to model the defoliation regime.

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SPRUCE BARK BEETLE DAMAGE WITHIN AN UNMANAGED SPRUCE FOREST DURING SEVEN YEARS FOLLOWING WINDFELLING

Jan Weslien [1] and Hansjochen Schröter [2].

ABSTRACT.--The loss of spruce trees due to attacks by spruce bark beetles, *Ips typographus*, in a nature reserve at high altitude in Southern Germany was estimated. After a storm in 1990 tree mortality rose constantly. Up to 1996 ca 4,000 dominant spruce trees or 16% of the spruce trees in the overstory have been killed. The bark beetle attacks have formed gaps of various size in the forest. The increasing number of gaps may lead to higher risk for additional storm and insect damage. Since the slopes are very steep an increased the risk for soil erosion cannot be excluded. This combined with a very poor natural regeneration of the gaps might be a serious threat to the forest reserve.

INTRODUCTION

Widespread outbreaks of the spruce bark beetle *Ips typographus* (L.) have occurred frequently in Europe. The main host of this bark beetle species is Norway spruce. The beetles can breed in standing spruce trees only if the number of attacking beetles is high enough to overcome the defensive system of the trees. Otherwise they are restricted to breed in windthrown, broken, or cut stems, usually a limited and scattered resource. Outbreaks, during which large numbers of standing trees are killed, typically occur after extensive stormfellings, when beetles have reproduced in a surplus of non-resistant breeding substrate. Due to counteractive measures by man, there are few studies that describe the natural spreading of attacks after a stormfelling. In recent years there has been a growing debate about the scarcity of dead wood in managed forests. A direct conflict between nature conservation and "clean management" of forests has arisen (Späth 1992, Dengler 1995).

In February 1990 central Europe especially Southern Germany, Austria, and Switzerland was hit by a heavy storm. Spruce bark beetles reproduced in the windthrown trees during 1990 and in 1991 great amounts of spruce trees were killed. The outbreak reached its peak in the dry summer of 1992 (Schröter *et al.* 1994 a, b; Reindl *et al.* 1994). This pattern was true for the lowlands. At high altitudes, as in the study area "Napf" in the present study, the pattern seemed to be different. Few attacks were observed in 1990 and 1991. Thereafter an increasing trend was observed in 1992 and 1993. Thus it was not obvious that the attacks were induced by the stormfelling. This study describes temporal, and to some extent also the spatial attack pattern by the spruce bark beetle within the nature reserve "Napf" during seven summers following a stormfelling.

MATERIALS AND METHODS

Study Area

The study area, "Napf" has been described thoroughly by Lopez (1985). Briefly it is an old forest consisting mainly of Norway spruce, 105 ha large, located on the west facing slopes of Feldberg, which mounts the highest point of Schwarzwald in southern Germany (1,493 m). The steep slope, ranging from 1,000 to 1,360 m in altitude surrounds a basin formed during the last glacial period.

Mapping

The forest was inventoried during May and June in 1994 and in August 1996 with regards to spruce trees killed by *Ips typographus* during the preceding years. The slope was first surveyed with binoculars from the opposite slopes. Each detected spot with killed trees was mapped and thereafter visited. The number of trees was counted and the year of death was

[1] *The Forestry Research Institute of Sweden (SkogForsk), S-756 46, Uppsala, Sweden.*
[2] *Forstliche Versuchs- und Forschungsanstalt Baden-Württemberg,, D-79007 Freiburg, Germany.*

estimated (for detailed method see below). In 1994 the diameter at breast height d.b.h. was measured. While cruising through the forest, killed trees, undetected from distance, where mapped and recorded as above. Since practically all of the forest was cruised there is very little chance of killed standing trees being left undetected. These could easily be spotted from quite a distance (at least 50 m). In 1994 recently wind-felled trees (still having twigs and some bark) were registered.

Determination of Year for Attack

In 1994 the dating was based on the following variables:

1. presence of living (hibernating) *Ips typographus* adults in bark
2. presence of ongoing attacks (fresh boring dust) by the ambrosia beetle *Trypodendron lineatum*
3. presence of brown/green needles
4. presence of fresh phloem in root legs
5. presence of fine twigs.
6. amount of bark still on tree
7. color of wood
8. presence and amounts of fungi
9. amount of litterfall on bark coming from tree

Any one of variables 1 or 2 was considered as direct evidence of a 1993 attack. It was found that variables 3 and 4 were strongly correlated to the first two variables. Thus after some time also the color presence of needles and/or fresh of phloem was considered as sufficient evidence for a 1993 attack.

No single variable could accurately indicate a 1992 attack. The 1992 attacks can be described still having fine twigs, some bark, and brownish wood. This was not sufficient to clearly distinguish from 1993 attacks. However, this in combination with the absence of variables 1-4 was regarded as sufficient.

Older attacks were not possible to date to the exact year. Based on variables 5-9 the killed trees were separated in two classes 1990/1991 and 1989 and older. These trees fell in two quite distinct categories, one recent and one older. According to the local foresters the recent attacks were probably from 1990 and 1991 and the older ones probably emanating from snow-breaks before 1985.

In 1996, attacks from the previous years (1994 and 1995) were dated using the same method as in 1994. Attacks from 1996 were easily recognised by presence of green needles and callow *Ips typographus* adults under bark.

Dating of *Ips typographus* attacks is usually difficult. Killed trees develop differently depending a variety of factors. The method used here may thus be less reliable elsewhere or under other weather conditions. In this study it was easy to distinguish previous years attacks from older ones, probably due to the relatively slow development of bark beetle brood and slow aging of trees at this high altitude. In the lowlands where *Ips typographus* has at least two generations per year in Southern Europe, this separation would be harder.

Transect Inventory

After having mapped and dated all *Ips typographus* attacks an inventory was made to determine the spatial relationship between windthrown trees and trees killed by *Ips typographus*. This was considered essential since the temporal attack pattern in "Napf" could not strengthen the hypothesis that the storm in February 1990 had induced the outbreak. A ca 3,000 m transect was subjectively drawn to cover a variety of storm gaps and killed trees. Along this transect all windthrown trees within ca 20 m on both sides were inspected to see if they contained galleries of *Ips typographus*. Also all trees killed by *Ips typographus* during 1990-1992 along this transect were registered.

RESULTS AND DISCUSSION

Almost 4,000 spruce trees were killed by *Ips typographus* during 1990-1996 (fig. 1). Windthrown trees, of which most were estimated to have fallen recently (probably in February 1990), amounted to ca 16% of total loss. The inventory of killed spruce trees can be regarded as total. Few trees can have escaped detectionThe mean diameter of the trees killed during 1990-1993 was 46 cm at breast height, as was the mean diameter of the windfelled trees along the transect. The diameter of killed trees was not measured in 1996 but it was obvious that most killed trees were from the overstory. Lopez (1985) estimated the total number of spruce trees in the overstory in "Napf" to 24,500 in 1982 (mean d.b.h. 46 cm). Thus the trees killed by spruce bark beetles since then amounts to ca 16% of the total stem number in the overstory.

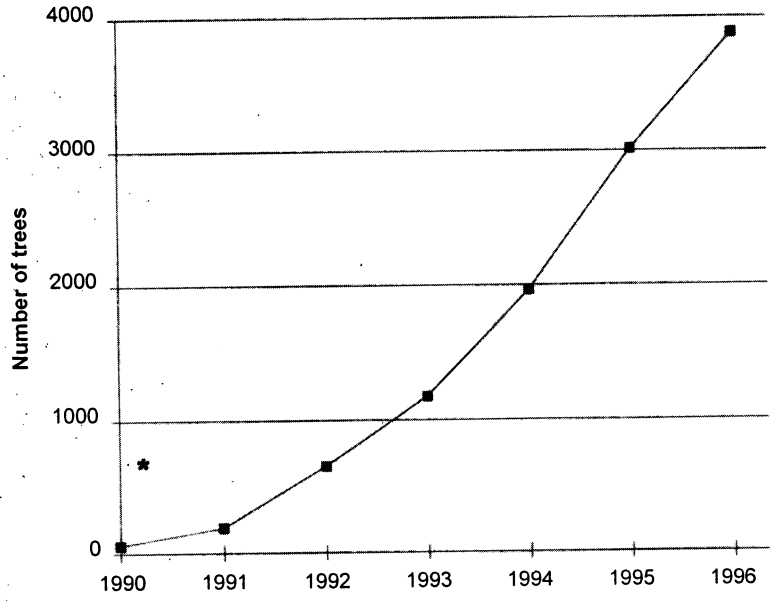

Figure 1.--Cumulative number of trees killed by Ips typographus in the forest reserve. "Napf" * marks ca 700 spruce trees felled in the storm of February 1990.

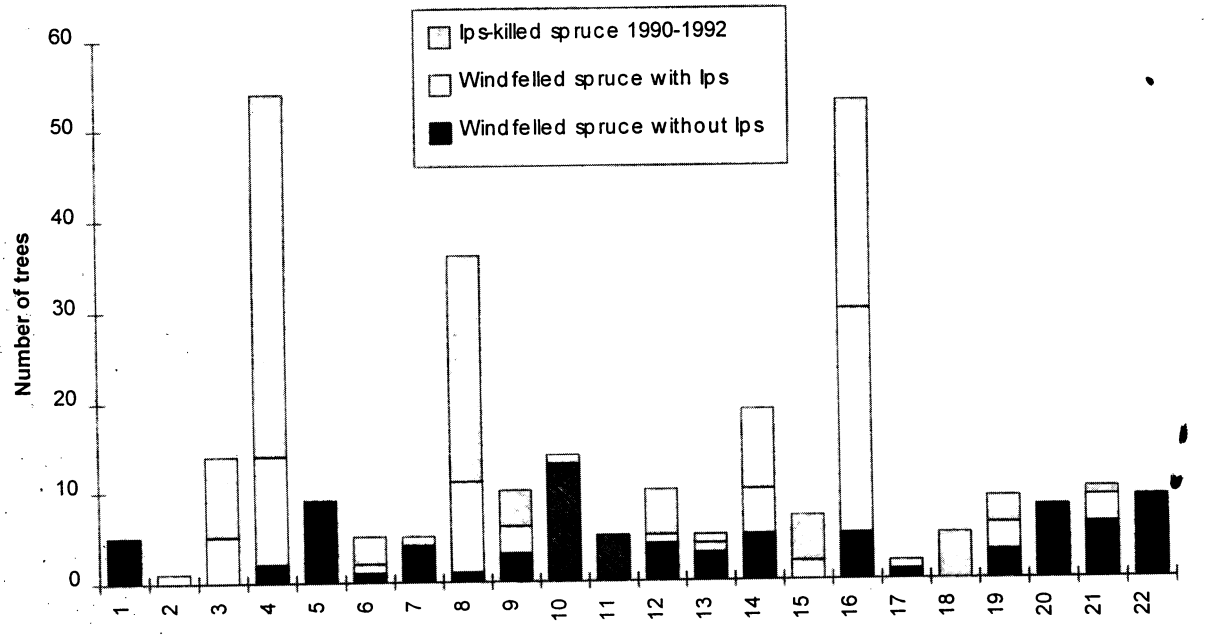

Figure 2.-- *Number of windfelled trees and* Ips typographus-*killed trees in 22 gaps along a 3,000 m transect. Each bar represents trees from the same gap.*

The transect inventory included 162 windthrown trees and 118 trees killed by *Ips typographus*. Windthrown trees attacked by *Ips typographus* had almost always been followed by attacks on the standing trees next to them. Windthrown trees without *Ips typographus* did not show this pattern (fig. 2). Of all 162 windthrown trees 73 (45%) had been colonised by *Ips typographus* The spatial correlation between windthrown trees and killed trees strongly indicates that the outbreak was induced by the storm-felling. Windthrown trees from 1990 were probably used by a relatively sparse *Ips typographus* population during both 1990 and 1991. This was possible due to the slow aging of the windthrown trees. Reproduction by *Ips typographus* during 2 years in trees downed at one occasion has earlier been documented by Butovitsch (1971) in Sweden. Apart from the cold climate in Napf, another factor, the steep slopes, probably affected ageing significantly. In cases where trees had fallen uphill, root contact was not entirely broken and the fallen trees could probably stay green quite long.

At lower altitudes in Germany, much higher rates of tree mortality has been observed than in this study. Niemeyer *et al.* 1995 found that 56% of the spruce stems were killed by *Ips typographus* during 3 years within a 63 ha forest. Schröter and Schelshorn (1994) estimated that the area with killed trees in a 10 ha forest stand increased from 0.2 ha to 4,5 ha from 1991 to 1994. It might be argued that annual tree mortality during outbreaks is lower whereas the outbreaks last longer in areas with cold climate than in areas with warm climate. In Sweden Lekander (1972) found that 16% of the spruce trees (31% of volume) within a 300-ha forest were killed during 7 years following a stormfelling. That outbreak collapsed due to high within-tree mortality of *Ips typographus*, without any counteractive measures taken by man. Within-tree mortality due to natural enemies under bark may be important for the collapse of *Ips typographus* outbreaks (Weslien and Regnander 1990, 1992; Weslien 1994). So far, however, tree mortality due to spruce bark beetle attacks has not shown any signs to decrease in "Napf".

The gaps formed in "Napf" should be regarded as a part of the natural dynamics that should occur there. However, according to Lopez and Bücking 1994) the natural regeneration of gaps is poor due browsing by the very high populations of chamois (*Rupicapra rupicapra*) and roedeer (*Capreolus capreolus*). Storms and spruce bark beetles will surely continue to "harvest from above" and this in combination with the poor regeneration could be a serious long-term threat to the forest. Due to the steep slopes there may also be an increased risk for soil erosion. Napf has probably increased in "value" since the storm in 1990 due to the large amounts of dead, large diameter spruce wood, a very scarce resource that can harbour many threatened species. Therefore clean management is not regarded as an option. Regarding this a more efficient game management is recommended in order to allow the natural gaps dynamics to occur. The study shows how difficult it may be to combine conservation and natural disturbance regimes in small protected areas.

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Effects of Fire

CHARCOAL IN DATED LAKE SEDIMENTS AS A PROXY FOR FIRE HISTORY IN JASPER NATIONAL PARK, CANADA

Jacqueline K. Huvane[1]

ABSTRACT.--Parks Canada is developing a fire management plan for Jasper National Park (JNP) to restore the natural fire regime that occurred prior to forest fire suppression in the 20th century. While fire history for both the 19th and 20th centuries is well documented, less is known about the earlier fire history of JNP. This project utilizes paleoecological methods to analyze forest disturbance regimes over the past ca. 300 years in the Athabasca River valley of JNP. Freeze cores (ca. 50 cm in length) were obtained from Mina Lake in 1995 and 1996. The recent sediments were dated by ^{210}Pb analyses, and the examination of charcoal particles was used to infer the fire history of the Park. I tested the validity of these methods by focusing on recent lake sediments (ca. the past 200 years) and comparing the charcoal inferred fire history to the known fire history derived from tree ring and fire scar data. The results from the Mina Lake cores indicate that macroscopic charcoal particle analyses can detect the major documented fires in the Park. In particular, the last major fire year in JNP occurred in 1889, and is represented by a broad peak in the macroscopic charcoal record. After 1889, there was little fire in JNP, and this is reflected by low macroscopic charcoal concentrations in the most recent sediments. Thus, charcoal particles can effectively serve as a proxy for fire history in JNP.

INTRODUCTION

During the past few decades, the issue of fire management has been under debate, and fire suppression policies have been questioned (Knight and Wallace 1989, Schullery 1989). The Canadian National Parks Act "requires that ecological integrity be a first priority when considering a park management plan" (Woodley 1993), and since 1986, Parks Canada has used prescribed burns in a few parks to restore areas to their natural fire regimes (Woodley 1995). One obvious question is how will natural fire regimes be defined? The past few centuries encompass several periods with different climates and human activities that may have affected fire regimes, including the recent period of fire suppression, European arrival, and the several hundred year cold period known as the "Little Ice Age". Therefore, the dendrochronological fire regime of only the past few hundred years (the temporal limit of tree-ring analyses in Jasper National Park (JNP)) may not be an accurate representation of long-term natural fire patterns.

In the past century, the area burned by fires in JNP has been low, and many have attributed this type of phenomenon to fire suppression policies (Banff-Bow Valley Study 1996, Woodley 1995, Kay *et al.* 1994). However, it has also been suggested that the lack of fire may be due to a variety of factors, including climate and availability of fuel (Tande 1977, 1979). One way to evaluate these hypotheses is to examine the fire history of JNP for a time interval that includes periods with different climatic regimes and vegetation types. This would determine whether the fire regime during the era of fire suppression is unusual in the context of the longer fire history in JNP. In this paper, I attempt to deduce fire histories from temporal patterns of charcoal deposition in lake sediments. If possible, this type of long-term ecological data will provide useful information to managers interested in developing fire management plans, and to those concerned with the effects of global warming on fire regimes.

This paper presents preliminary findings based on analyses from one lake, Mina Lake (fig. 1). The recent charcoal deposits are compared to the dendrochronological fire history in order to verify the charcoal technique.

[1]*Jacqueline K.Huvane, Department of Biological Sciences, University of Alberta, Edmonton, AB T6G 2E9, Canada. Present address: Duke University, Wetland Center, Box 90333 Nicholas School of the Environment, Durham, NC 27708-0333 USA.*

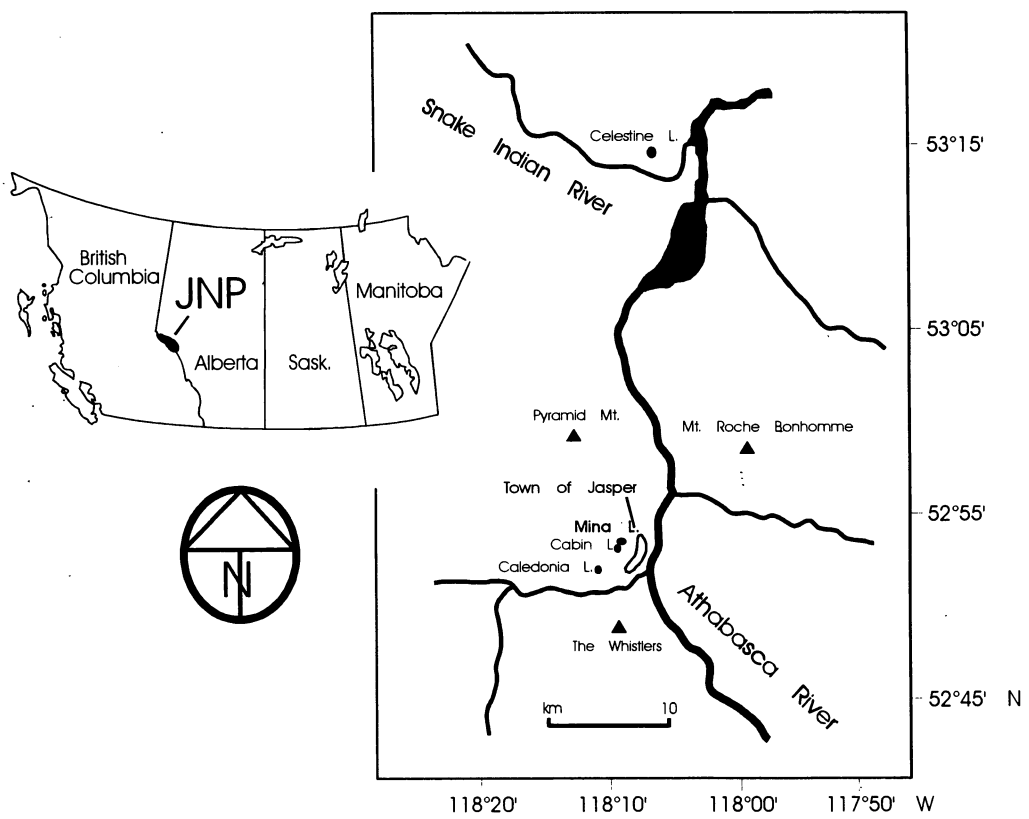

Figure 1.-- *Map of Jasper National Park, Canada showing the locations of the townsite and Mina Lake.*

RATIONALE

Charcoal produced as a result of fire is dispersed mainly by wind or water, and is commonly preserved in lake sediments. Clark (1988) suggests that surface runoff is not an important source of charcoal to lake sediments, and that eolian transport is more important. Charcoal may be transported to lakes via stream inflows, however, sites may be chosen that do not receive inflow from streams, such as Mina Lake. Accordingly, the discussion of charcoal transport here relates to windblown particles. Although much is unknown about charcoal dispersal, it is generally assumed that larger particles are deposited close to the source of fire, while the smaller particles are deposited from both local and regional sources (Tolonen 1986, Patterson 1987, Clark 1988, MacDonald *et al.* 1991, Clark and Royall 1995). Clark (1988) theorized that smaller particles are preferentially transported away from a fire due to prolonged suspension in the atmosphere as compared to larger particles that tend to settle more quickly. A study of the fire history of Wood Buffalo National Park, Canada showed that microscopic charcoal particles deposited in lake sediments were indicative of the regional fire history, while macroscopic charcoal particles were more closely associated with local fires, although not all local fires were represented by macroscopic charcoal peaks (MacDonald *et al.* 1991). In another study, Clark and Royall (1995) examined the correspondence between charcoal particle size and the source area of a fire in a

suite of eastern North American lakes. They found that smaller particles derive from regional sources, while larger particles derive from local wildfires. However, again, not all local fires were represented by increases in the macroscopic charcoal record. Similarly, Millspaugh and Whitlock (1995) examined the macroscopic charcoal record from lake sediments in Yellowstone National Park, U.S.A., and found a general correspondence between the charcoal record and known fire history. However, at one site not all known fires were represented by charcoal peaks. The lack of correspondence between known fire history and the charcoal record suggests that many factors, including the amount and type of charcoal, and dispersal patterns are important in determining whether a fire results in a sedimentary charcoal peak. Additionally, a single fire may be represented by a broad charcoal peak (Patterson 1987), the result of a variety of processes including charcoal deposition, post deposition mixing (Davis 1968, Davis and Ford 1982) and sediment sampling protocols (Clark 1988).

These studies suggest several hypotheses concerning charcoal deposition. The one discussed in this paper, that the larger macroscopic charcoal particles will be more indicative of local fire patterns than smaller, microscopic particles has already been examined to some extent in the literature (Clark and Royall 1995, MacDonald *et al.* 1991). Here I examine these ideas in the context of the montane environment in JNP. I am also concerned with distinguishing single fires from charcoal peaks in the sediment record. Because a single fire may produce a broad charcoal peak, periods when fires occur infrequently will probably result in obvious peaks in the stratigraphy. However, if fires are very frequent, then there may be a relatively steady delivery of charcoal to lake sediments, making it difficult to discern individual fires.

METHODS

Study Site

Mina lake is a relatively small lake, with a surface area of 8.3 ha., and a watershed area of 41 ha. It is located in the Athabasca Valley, near the Jasper, Alberta townsite, at an elevation of 1214 masl (fig. 1). Mina Lake has a maximum depth of 13.2 m, an average depth of 5.7 m, and an outflow, but no inflow (Anderson and Donald 1980). The vegetation in the area consists of Douglas-fir and lodgepole pine forests, with black spruce often occurring around lake margins (Tande 1979). The climate around the townsite tends to be warmer and drier than in other areas (Tande 1979). Data from the Jasper weather station (52°53'N 118°04'W at 1061 masl) for the years 1926 to 1990 show a mean daily air temperature of 3.1° C, a mean maximum of 9.0° C, and a mean daily minimum of -2.8° C. The mean annual precipitation for the same time period was 393.7 mm with a mean of 281.6 mm as rainfall and 143.9 cm as snowfall (Environment Canada).

Dendrochronological Fire History

A study of tree ring and fire scar data by Tande (1977) provides an excellent record of fire history in the Athabasca River valley of JNP over the last two to three centuries. The last major fire in JNP occurred in 1889, when about 20% of the park burned (Van Wagner 1995).

The fire history can be examined at two different scales. The smaller, local scale concerns fires in and around the Mina Lake watershed, and the larger Park-wide scale concerns fires that burned large portions of the entire Park.

Large Park Fires:

The last large fire in JNP occurred in 1889, when about 20% of the Park, including the Mina Lake area, burned (fig. 2) (Van Wagner 1995). The previous

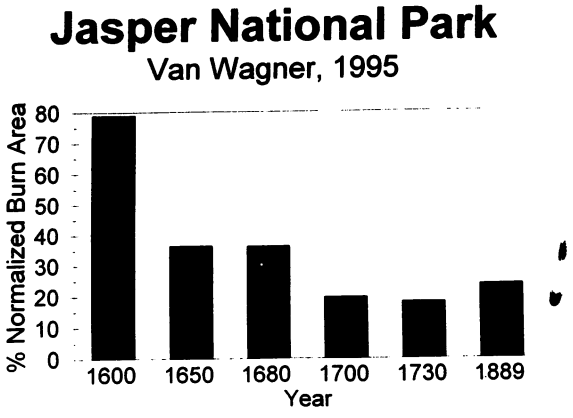

Figure 2.--*A histogram of the five largest park fires as determined by Van Wagner (1995). They Y axis shows the normalized burn area as a percentage of the park area. The X axis shows the year of the burn. These data assume a 150-year fire frequency from 1915 back to 1735, and a 55-year cycle prior to 1735.*

large fire occurred in 1730, when about 18% of the Park burned. Fires that occurred in the 17th century were probably larger, burning more than 30% of JNP. However, the dendrochrono-logical data for the 17th century is sparse.

Van Wagner (1995) reported a change from a 55-year fire frequency to a 120-fire frequency around 1735. Johnson and Larsen (1991) reported a similar shift to a longer fire frequency after 1730 in the southern Canadian Rockies, in the Kananaskis Watershed. They suggested that the shift was the result of changing climatic patterns, specifically to cooler, wetter conditions associated with glacial readvance. Luckman (1988) identified two periods of glacial advances for the Athabasca and Dome glaciers in the Canadian Rockies in Alberta: the "Little Ice Age" advance (ca. 1844) and an earlier advance in 1714. Thus a climatic explanation as evidenced by glacial advance could also explain the change in fire frequency proposed by Van Wagner for JNP.

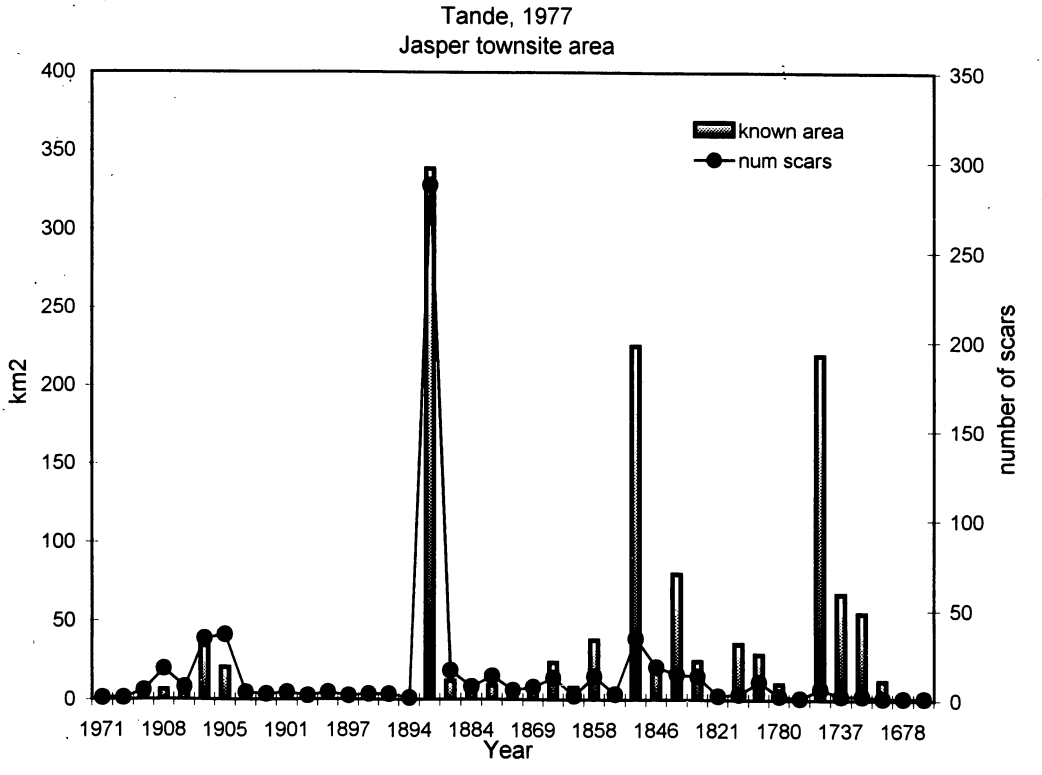

Figure 3.--*This graph is adapted from Tande (1977) and displays the relationship between the known area of a burn and the number of fire scars found from a particular burn year. On the first Y axis is the known burn area in square kilometers and on the second Y axis, the number of fire scars for a particular burn. The X axis displays the fire years.*

Small Local Fires:

In the Jasper townsite area, the largest fires occurred in 1889, 1847, and 1758 (Tande 1977). The 1889 fires were the most extensive and were of medium to high intensity. Tande (1977) states that the 1847 and 1758 fires behaved similarly to the 1889 fires. The Mina Lake watershed burned in 1889, 1847, 1837, 1834, 1797, 1758 and 1727. The majority of these historical fires in JNP were of low intensity, and would probably not be evident in the charcoal record. There is considerably less evidence available for the older fires (Tande 1977). For instance, 287 fire scars provided evidence for the 1889 fire, 34 for the 1847 fire, and 6 for the 1758 fire (fig. 3). Evidence for fires prior to 1758 were based on only one or two fire scars. As expected, there is less information concerning fires prior to the 19th century.

Core Collection

Sediment cores were collected during March 1995 and March 1996. In 1995, a cylindrical freeze core was used (Miskimmin *et al.* 1996). Briefly, a hollow tube is filled with dry ice and alcohol, lowered into the lake at a site of maximum lake depth, and allowed to free fall about one meter away from the sediment water interface. The corer will sink into the sediments, which then freeze to the outside of the tube. Retrieved from the lake bottom, the corer with the frozen sediments can be taken back to the lab for analyses. This procedure allows for the collection of an undisturbed sediment core. With this method we obtained a 51 cm long core from Mina Lake. In 1996, a frozen core, 42 cm in length, was obtained using a rectangular corer. This different shape provided additional material and allowed us to do more analyses (macroscopic charcoal) for this site.

Core Chronology

In order to determine the fire history based on lacustrine sediment cores, it is necessary to date the sediments. This can be achieved by a technique based on ^{210}Pb decay. Binford (1990) provides an extensive review of these methods. If a constant rate of supply of ^{210}Pb to the sediments is assumed, then the uppermost (youngest) sediments will have the greatest ^{210}Pb activity (Appleby and Oldfield 1978). By measuring the activity of ^{210}Pb at the sediment water interface, and at several levels throughout the core, it is possible to derive a model for dating the sediments. This method allows for the dating of sediments up to about 150 years old. For older sediments, other techniques such as AMS ^{14}C dating of terrestrial macrofossils may be employed (Baer 1989, Olsson 1986). However, in this study we will be focusing on the recent sediments.

A half section of the 1995 freeze core was subsampled into 1 cm intervals. Approximately 5cm^3 from each sample was dried and sent to Paul Wilkenson at the Department of Fisheries and Oceans in Winnipeg, Manitoba for ^{210}Pb analyses. Similar methods were used for the 1996 core, however, those results are not yet available.

Charcoal Analyses

Two categories of charcoal were analyzed: microscopic charcoal counted from pollen slides, and macroscopic charcoal counted from sieved sediment samples.

Microscopic charcoal analyses

A quarter longitudinal section from the 1995 core was fine sectioned into 2.5 to 3 mm thick transverse sections. These samples were digested using a standard procedure for pollen preparations (Faegri and Iverson 1975) to concentrate the charcoal and pollen for enumeration. Briefly, samples were first treated with 10% sodium hydroxide to break apart the material and remove humic substances. Then samples were treated with hydrochloric acid to remove any carbonates. This was followed by an hydrofluoric acid treatment to dissolve silicates. Finally, the samples were subjected to an acetolysis with sulfuric acid to digest cellulose.

Charcoal particles were counted from select samples along with pollen grains. Microscopic charcoal particles ranged in size from less than 450 μm^2 to over 10,000 μm^2. Particles smaller than 450 μm^2 were not included. This smaller sized fraction probably represents long distance transport, and often these smaller particles are difficult to positively identify as charcoal. The charcoal particles counted were separated into three size classes. Small particles ranged from 450-1,600 μm^2, and medium ranged from 1,601-2,500 μm^2. Particles greater than 2,501 μm^2 were initially divided into three subclasses, however the scarcity of large particles in the samples made it necessary to pool these subclasses. For each

sample counted, the number of charcoal particles as a percentage of the total pollen sum was calculated (Swain 1973). These data are displayed as the difference from the core median value.

Macroscopic charcoal analyses

The 1996 freeze core was subsampled into 1.0 cm thick transverse sections for macroscopic charcoal determinations. A finer subsampling scheme was not possible because these determinations required larger amounts of sediment than for microscopic charcoal analyses. For each sample, a measured amount of sediment between 3 and 5 cm^3 (usually 5 cm^3), was rinsed through a #100 sieve (149 µm aperture). All particles retained on the sieve were examined with the aid of a dissecting microscope at 150X magnification. Particles were enumerated and a length and width measurement was recorded. For each sample, the total charcoal area was calculated and expressed as a concentration (charcoal area/cm^3 sediment).

Statistics

A simple Runs Test above and below the median was performed on the stratigraphic charcoal data. These tests determine whether the sequence of events is random, or whether a given event is a function of the outcome of a previous event (Sokal and Rohlf 1981). Each sample was assigned either a plus or a minus depending on whether it was above (+) or below (-) the core median. The sequence of pluses and minuses was then subjected to the Runs Test. One obvious drawback of this method is that it only takes into account the direction of the change and not the magnitude of change. However, it provides a quick method to determine whether a sequence of peaks and troughs is non-random.

RESULTS

^{210}Pb Dating

The results from the 1996 freeze core suggest that the Mina Lake 1995 core sediments were not appreciably mixed. Figure 4 shows the sediment depth versus the ^{210}Pb date. A constant rate of supply model gave a date of 1892 at 15.5 centimeters and an average sedimentation rate of 1.5 mm per year for the top 16 cm of sediment.

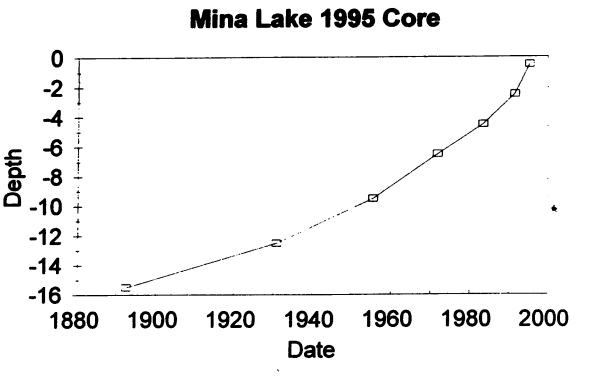

Figure 4.--*Sediment depth versus ^{210}Pb date for the 1995 core.*

Microscopic Charcoal

The microscopic charcoal data are displayed in figure 5 as the difference in the percentage from the core median. The Runs Test of the microscopic charcoal data did not produce statistically significant results, indicating that the sequence of peaks was random. However, several peaks were still evident in the stratigraphy. Small charcoal particles displayed a peak between 5 and 10 cm, ca. 1955- 1980, and another peak at 16 cm, ca. 1889. Prior to ca. 1889 there were no obvious major peaks, only slight peaks at 26, 35, and 40 cm. The medium size class showed a similar distribution. There was a recent peak between ca. 1955 and 1980, and a peak ca. 1889. There were no peaks between ca. 1889 (16 cm), and 40 cm. Based on the sediment accumulation rate for the top portion of the core and sediment bulk densities, the 40 cm level probably dates to the 17th century.

The largest microscopic charcoal size class also showed a recent peak between 8 and 10 cm ca. 1955-1960, and a major peak ca. 1889. Between 20-27 cm, there was a distinct trough in the large charcoal particle percentages. This probably corresponds to the period between ca. 1845- 1800. There were peaks at 28, 34, and 40 cm. The peak at 28 cm probably corresponds to the early 19th century and the later peaks to periods in the 17th century.

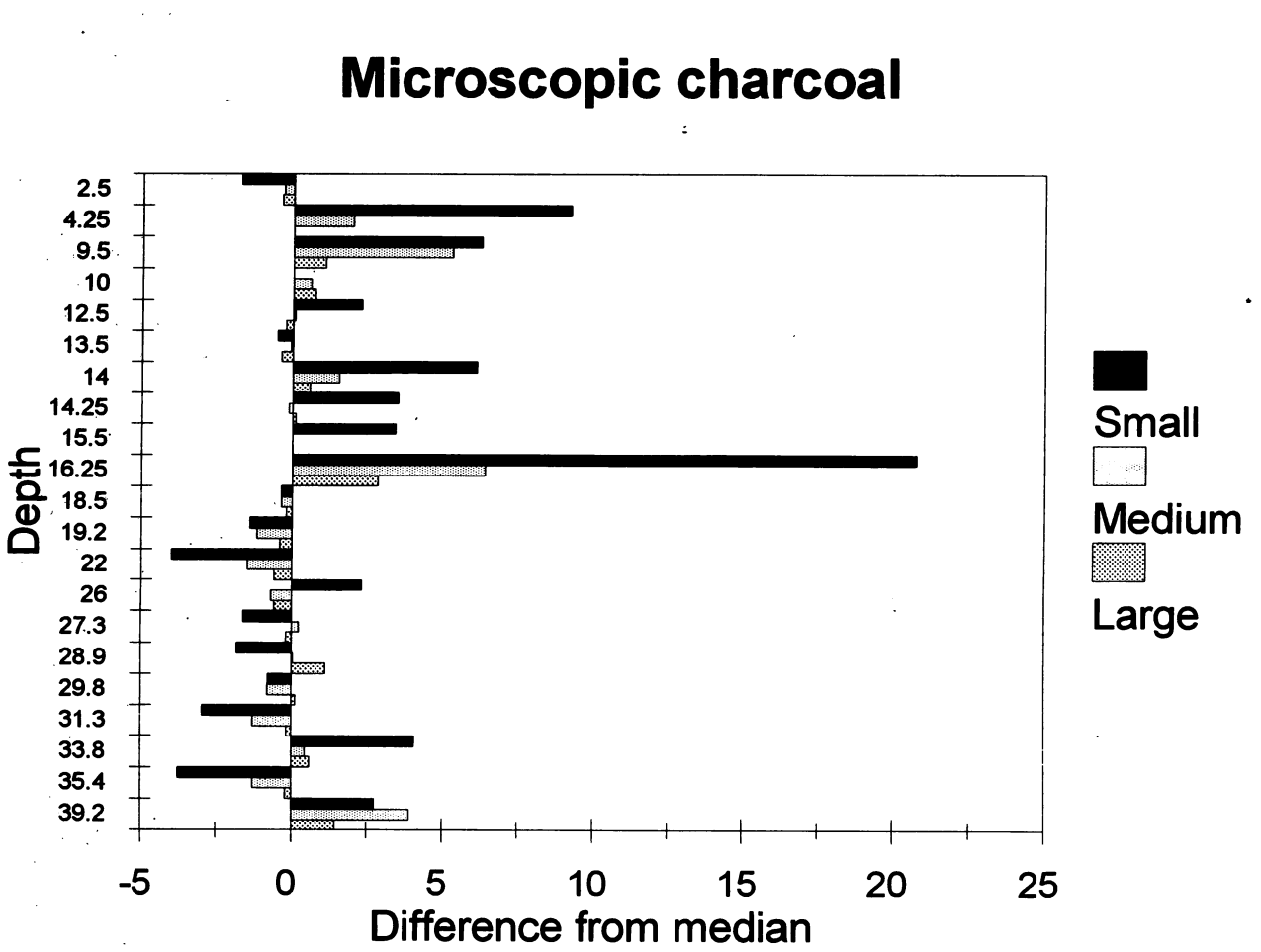

Figure 5.--*Three size classes of microscopic charcoal versus sediment depth. The data are expressed as a difference in the percentage from the core median percentage (charcoal/pollen*100). The zero line represents the median percentage of charcoal.*

Macroscopic Charcoal

The Runs Test of the macroscopic data indicated that the series of charcoal peaks and troughs were non-random (fig. 6). There were no major peaks before 13 cm, where there was a broad peak until about 18 cm. Although we do not have ^{210}Pb dates for this core yet, this interval likely included 1889, based on the dates from the 1995 core. Between 18 and 26 cm there was period of low charcoal concentrations, and then another peak between 26 to 30 cm. This peak probably occurred during the mid-to-late 18th century. Prior to this second peak there was more noise in the record, with charcoal concentrations oscillating about the core median.

DISCUSSION

Although the patterns in the microscopic charcoal data were not statistically significant, all size classes showed a major peak ca. 1889, when the last major fire occurred in the park. The non-significant result may be attributed to the fact that the test does not account for the magnitude of change associated with the charcoal peaks.

All size classes also showed recent peaks, although there was very little fire in JNP during this period. This suggests that, as in other studies, the microscopic charcoal is responsive to regional fire patterns and not necessarily local patterns. For instance, Clark and Royall (1995) found that microscopic charcoal particles were common in the recent sediments of midwestern lakes, despite effective fire suppression in that region.

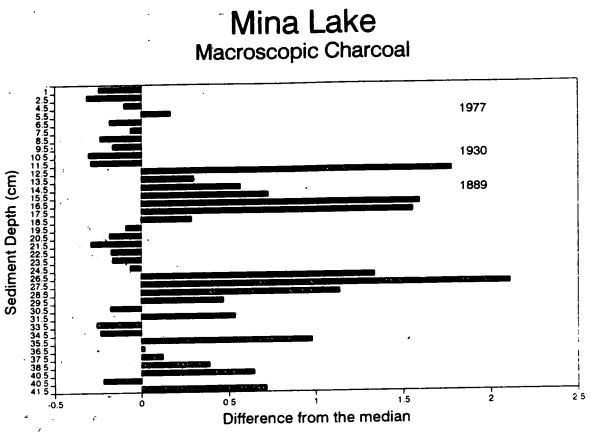

Figure 6.--*The macroscopic charcoal concentration (area of charcoal/cc sediment) as a difference from the core median. The zero line represents the core median of .312 μm^2 charcoal/cm^3 sediment. Estimates of core dates are indicated on the right.*

There was no indication of a mid 19th century fire, although fires occurred in the Mina Lake watershed during this period. There was some evidence for fires in the 17th and 18th centuries as illustrated by peaks in the bottom portion of the core. However, these peaks were small, suggesting that microscopic charcoal does not distinguish regional from localized fire events.

The macroscopic charcoal data corresponded more closely to the Park's known fire history. There were no pronounced recent peaks, and the 1889 fire was clearly indicated. However, there were also no mid 19th century peaks associated with the moderate to high intensity fires that occurred around the Jasper townsite and the Mina Lake watershed in 1847. This suggests that the macroscopic charcoal was not recording smaller fires in and around the watershed. This is not unusual in charcoal studies of fire history (Swain 1978, MacDonald *et al.* 1991, Millspaugh and Whitlock 1995). Although an extensive portion of the townsite area burned during the 19th century, the 1847 fire was not included in Van Wagner's list of the major fires within the Park. The macroscopic charcoal in the Mina Lake sediments appear to provide a signal for large scale fire events that occurred in the Park, but not the smaller fires that occurred locally around the Mina Lake watershed.

The next major peak probably occurred in the mid-to-late 18th century, when there was both a major Park fire (1730) and a major watershed fire (1758). It will be impossible to distinguish these two events until the ^{210}Pb dates are established for this core and other indications of fire are obtained such as pollen or magnetic susceptibility data. For example, Millspaugh and Whitlock (1995) established the occurrence of local, watershed fires in Yellowstone National Park when there were coincident peaks in macroscopic charcoal and magnetic susceptibility. Fire in the watershed produces secondary magnetic minerals that are delivered to the lake sediments via erosion (Thompson and Oldfield 1986). Therefore, increases in magnetism may be used as another indicator of local fire.

The two broad peaks of increased macroscopic charcoal concentrations associated with JNP fires are in contrast to the single sample peaks in the microscopic charcoal. This may be the result of a variety of factors. One possibility is that as Clark suggests, "following fire, larger fragments are moved about locally to a greater extent than the pollen sized charcoal... because these particles are relatively easy to lift at low wind velocities" (Clark 1988). Millspaugh and Whitlock (1995) also note broad peaks in macroscopic charcoal records and attribute them to lags in deposition time, sediment mixing, and dating errors. Another important consideration is the sampling scheme employed. The microscopic charcoal samples in this study were 2-3 mm thick, representing approximately 2-5 years, whereas the macroscopic samples were 10 mm thick, representing approximately 10 to 15 years. The net effect is to average or smooth the macroscopic charcoal record.

Van Wagner (1995) identified several large JNP fires in the 17th century, but there was little evidence for these fires in the macroscopic charcoal stratigraphic record. If there was a change to a more frequent fire regime in this area as suggested by Van Wagner, then these fires may not show up as distinct charcoal peaks in the stratigraphic record. The pattern of charcoal concentrations oscillating about the core median is consistent with the notion that the fire interval became shorter at this stage. Clark (1988) suggests that at a sediment accumulation rate of 0.1 cm per year and a sampling scheme of 1.0 cm intervals, the fire interval must be greater than 50 years to discern all fires. The fire interval may be resolved in the Mina Lake sediments by examining a longer record at finer intervals (e.g., 0.5 cm intervals as opposed to 1.0 cm),

and by obtaining a more detailed core chronology. While there were earlier charcoal peaks found in the microscopic record that likely correspond to this time period, these peaks may only reflect regional processes and not necessarily local or extra-local events.

CONCLUSIONS

The macroscopic charcoal particles at Mina Lake provide a proxy for the large fires in JNP, but not for smaller watershed fires. The microscopic particles likely represent a regional signal and are of less interest to Parks managers. The data suggest that it is possible to derive a long-term record of major fire frequency in JNP using macroscopic charcoal. Periods that contain a "noisy" record may signal a shift to a more frequent fire interval. However, these conclusions must remain tentative until we have obtained the ^{210}Pb chronology for the 1996 core.

Because there is always uncertainty involved in ecological studies, it is desirable to use multiple lines of evidence to infer fire history. In the larger study of the fire history of JNP mentioned earlier, several other variables are being considered. These include an examination of pollen remains to infer vegetation change in the area. After a major fire, successional vegetational changes are recorded in the lake sediments, such as an increase in non-arboreal pollen and early successional species. Magnetic susceptibility is another variable being considered to discern local fire events.

This study clearly shows the importance of having knowledge of the recent fire history in order to gain information concerning the relationship between stratigraphic charcoal and the known fire history at a given site. This will allow for a more accurate interpretation of older stratigraphic data. This information used in conjunction with a variety of indicators should provide the best reflection of the long-term fire history of the area.

ACKNOWLEDGEMENTS

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POST FIRE CHANGES OF LIVING GROUND COVER IN THE FORESTS OF CENTRAL SIBERIA

Galina A. Ivanova and V.L. Perevoznikova[1]

ABSTRACT.--Post fire observations showed that as moss cover degrades, the number of vegetation micro-groups decreases, and both the ratio among dominant species and their coenotic role change in light and dark coniferous forests with green moss, which are remarkable for a pronounced mosaic of living ground cover. Against the background of highly dynamic succession processes, communities tend to lose species and that can eventually lead to a decrease in ground vegetation diversity.

INTRODUCTION

Forest fires, being a major ecological factor, influence biodiversity on both a site and an ecotope scale and determine vegetation. Vegetation cover diversity implies abundance of species and their positions in the vegetation hierarchy.

Fire has the most profound influence on living ground cover and soil properties. The role of fire in the formation of living ground cover of forest coenoses is very complex. Low intensity fires, having little influence on forest soil properties, result in only partial mineralization of dead organic matter, reduce its depth, and thereby promote certain grasses and shrubs.

With increasing combustion intensity, fire influence on soils becomes more considerable to result in changing living ground cover regeneration patterns, i.e., fire causes changes of species composition of the latter, as well as coverage and abundance of each species and population (Popova 1975, Popov 1982, Komarova 1992, Furyaev 1996). Under repeated fires, individual species, their groups and even communities develop various mechanisms of adaptation. For example, fire stimulates *Chamerion angustifolium* over other coenotic populations.

The trend of post fire forest regeneration is largely dependent on its first stage, which in turn depends on the level of ecosystem disturbance by fire, dynamics of burned site colonization, and invading species composition and structure. At the initial stage of post fire forest regeneration, living ground cover composition is considered to be most indicative of both site conditions and succession. The rate of vegetation succession in burned areas can vary significantly not only within stands, but also on an ecotope scale. The aim of this study was to compare the initial stage of post-fire succession in different stands.

STUDY AREA AND METHODS

The study area was situated in low East Sayan Mountains, central Angara region. For comparative investigation of succession processes, we selected pine and dark-needled stands that experienced surface fires. The vegetation inventory is presented in table 1.

In each site, rate of burning was determined from the level of dead organic matter consumption (Khlebnikova *et al.* 1982): heavily burned sites (all dead matter consumed), moderately burned sites (only a part of dead organic layer was burned and to a small depth), and lightly burned sites (fire consumed only the surface layer).

Post fire living ground cover dynamics was studied in 10 m x 10 m plots laid out in sites differing in the rate of burning. The plots were divided into 2 m x 2 m squares in order to make detailed mapping and description of grass cover feasible. Collected data were analyzed using mathematical methods (Vasilevich 1969, Schmidt 1984, Titlanova *et al.* 1993). Coefficient of species similarity was calculated as: $K = (2c/(a+b)) \times 100$, where a is the number of species in the first species list, b is the number of species in the second list, and c is the total number of species in the two lists. Coefficient of species difference was determined by: $Prs = (X+Y-Z)/(X+Y+Z)$, where X is the number of species of the first list that are absent from the second list, Y is the number of species of the second list that are absent from the first list, and Z is the number of species occuring in both lists. This coefficient is a

[1] *V.N. Sukachev Forest Institute, Siberian Branch, Russian Academy of Sciences, Krasnoyarsk, Russia. E-mail: ifor@krsk.infotel.ru.*

scale of -1 to +1. A specific thing about this coefficient is that when it ranges from -1 to 0, it indicates similarity in species composition between communities, and its variations from 0 to +1 show difference.

In order to quantify succession dynamics, we used dynamics indicators (D) calculated as: D general: $Dg = (n_2 + n_3 + n_4)/ n_1$, where n_1 is species observed in a community before and after fire (persistent or steady species); n_2 is the number of newly occurred species; n_3 is the number of species lost during some period of time; n_4 is the number of fluctuated species, and D succession: $Ds = (n_2 + n_3)/ n_1$. From the succession dynamics indicator, one can tell which way succession occurs; if $D > 1$, new species are greater in number than pre-fire species, while at $D < 1$ the situation is opposite. $A = n_2/n_3$ shows an increase ($A>1$) or a decrease ($A<1$) in the number of species of a community (Titlanova *et al.* 1993).

RESULTS AND DISCUSSION

Post fire regeneration of living ground cover of pine stands in central Angara region varies with site conditions. The rate of recovery of burned sites depends on available fuel load, because non-uniform fuel distribution results in different fire intensity even within a relatively small site.

In pine stands, tree mortality is usually negligible after surface fires. Most profound post fire changes are typical for living ground cover. For example, in a pine stand with *Vaccinium vitis-idaea* and mixed grasses, where similarity between pre- and post fire ground cover compositions was pretty high (71%) and the overstory survived 100%, succession occurred at a much slower rate compared to all other forest types under study (table 2). The number of species tended to increase, from which we can expect some increase in species diversity. However, the succession appeared to be most dynamic in a pine stand with *V. vitis-idaea* and green mosses and a larch stand with *Carex* and *Spiraea* ground cover, although their coefficients of similarity with unburned forest were the lowest. The indicator of species number decrease and increase (A) that reflects species diversity dynamics, shows that new species are possible to appear in the larch stand, whereas the species composition grows poorer in the pine *V.vitis-idaea*-green moss stand (table 2).

Research studies we conducted in West Sayan mountain forests show that in dark coniferous stands, unlike in pine stands with green moss ground cover, trees also suffer from fire. Tree mortality amounts to 80%-90% in the year following a fire. Practically complete degradation of mosses is observed. Dead organic matter decreases more than 50%. Mineralization of organic matter is, however, incomplete, with underburn accounting for 73% of the total dead matter and living ground cover.

According to the obtained data, living ground cover species composition and structure change most vividly during the first year after fire in stands with green moss ground cover, independently of what the dominant tree species is. Coenotic role of many herbacious species also changes. Mosses fully degrade, and other species, such as *Chamerion angustifolium, Calamagrostis obtusata, Millium effisium, Brachypodium pinnantum, Iris rhutenica, Stellaria bungeana* and *Lathyrus humilis,* become dominant ground cover species. Noticeable differences occur in ground cover species composition between burned and unburned sites. The coefficient of similarity of the grass-small shrub layer composition between burned and unburned sites is 33%-43%. Index of dynamics (Titlanova *et al.* 1993) ranges from 2.40 to 4.33, which indicates the processes occuring in the ground cover are highly dynamic. At the same time, a trend is observed for the community to loose species and this suggets a potential decrease in species diversity (table 2).

Table 1.-- *Inventory characteristics of studied stands.*

Forest type	Stand		Site quality class	Average		Load (m^3)
	Age (yr.)	Composition		Diameter (cm)	Height (m)	
P.sylvestris/V.vitis-idaea/mixed herbs;	100	8P1L1B	III	**21**	**22**	**351**
P.sylvestris/mixed herbs	120	9P1L	III	**23**	**21**	**330**
P. sylvestris/V.vitis-idaea/green mosses	100	7P2L1S,B	III	**18**	**19**	**308**
P.sylvestris/mixed herbs/green mosses	160	8P2L	III	**27**	**22**	**435**
Larch with spruce/*Spiraea-Carex*	L180 S120	5L4S1P	III	**32** **18**	**26** **20**	**250**
Spruce-small grass/green moss	160	6S2F2SP	III	**20**	**21**	**210**

- L (larch); - B (birch); - P (pine); - S (spruce) - SP (Siberian pine) - F (fir)

Table 2.-- *Succession dynamics in forest types after surface fire.*

Forest type	Number of species			Coefficient of similarity with unburned sites, %	Index	
	Total	Lost	New		Succession dynamics (D)	Species number (A)
P. sylvestris/V.vitis-idaea-green mosses	23	4	1	71	0.23	4.0
P.sylvestris/mixed herbs/green mosses	23	6	11	60	1.31	0.55
P. sylvestris-V.vitis-idaea-green mosses	29	3	23	33	4.33	0.13
Larch with spruce-*Spiraea-Carex*	6	3	3	36	3.00	1.0
Spruce-small grass-green moss	13	4	8	43	2.40	0.5

D>1 - number of new post fire species is greater than that of the pre-fire species
D<1 - Surviving species are greater in number than new post fire species
A>1 - number of species of a community tends to increase
A<1 - a tendency for a community to loose species

Vegetation is known to re-establish in two major ways in sites subject to fire: (a) from remained pre-fire phytocoenoses whose population structure has changed as a result of fire, and (b) from seeds and separate vegetative organs that were present in the soil before a fire. Along with these, diaspores can be carried from the adjacent unburned forest. Our investigation shows that reproduction from seeds prevails over other types of reproduction at the earliest stage of ground cover recovery in burned pine sites. This type of reproduction was established to account for 80% of ground cover species in a pine *V.vitis-idaea* mixed herb stand, with vegetative reproduction accounting for only 10%.

The amount of viable seeds in soil is thus critical in the formation of post fire living ground cover structure. Seeds in soil, being an emergency reserve, become active after ground cover and tree layer disturbances. Our studies suggest that 2,300 to 4,200 of buried seeds per hectare can survive fire in a pine stand with *V. vitis-idaea* and mixed herbs (Ivanova and Perevoznikova 1996).

In burned sites, ground cover can also re-establish by intensive resprouting. For example, individuals of non-seed origin can prevail in pine stands with *V.vitis-idaea* and mixed herbs in the second year following a fire. Reproduction by seeds slows down in time, although it was observed to prevail for certain herbacious species (*Calamagrostis arundinacea*) during early stages of post fire regeneration.

Typological structure of pine stands after surface fires is to a large extent determined by pre-fire living ground cover structure. According to Ivanova's and Perevoznikova's data (1996), *Calamagrostis arundinacea*-dominated vegetation groups are promoted after fire in mosaic mixed herb-green moss ground cover in pine stands (fig. 1). Ground cover mosaic elements change and grow bigger in stands with green mosses. As a result, the number of mosaic elements decreases considerably. Mosses die and are replaced by tall grass-dominated microgroups (*Fabaceae* and *Onagraceae*), which can form various mosaic combinations.

In pine stands with *V.vitis-idaea* and mixed herbs, where living ground cover mosaic is much less clear, boundaries of vegetation groups are usually indistinct. Burned site recovery depends on the level of disturbance of dead organic layer and living ground cover in pine stands of this type.

For example, in pine *V.vitis-idaea* and mixed herb stands, where dead organic matter and soil surface are severely disturbed by fire, a distinctly mosaic pattern of re-establishing ground cover is manifested through formation of mono-dominant microgroups (*Chamerion angustifolium, Maianthemum bifolium, Brachypodium pinnantum, Calamagrostis arundinacea,* or *Carex macroura*), which interchange with bare patches burned down to mineral soil.

Succession dynamics appears to be much higher in heavily burned sites than in moderately and lightly burned sites, with the coefficient of species composition similarity between heavily burned and unburned sites being pretty low. Although the coefficient of similarity tends to increase in moderately burned sites, the coefficient of discrimination shows a significant difference in species composition. Lightly burned sites appear to have the highest similarity with unburned sites in terms of ground cover composition and structure. Coefficient of succession dynamics, being lower than in moderately and heavily burned sites, shows that the number of pre-fire species is greater than that of new species in lightly burned sites. But even with light burning, species loss still occurs (at $A<1$), though inconsiderable (table 3).

Coenotic role of individual species and groups of species also changes significantly in stands burned by surface fire. Some species demonstrate a positive response to low-intensity surface fires: *Brachypodium pinnatum, Iris ruthenica, Galium boreale, Lathyrus humilis,* and *Trollius asiaticus.* Depending on intensity of burning, abundance of *Chamerion angustifolium* increases from sol to cop2-3, with its coverage being 50%-70% (100% in heavily burned patches). *Carex macroura* also responds positively to light and moderate burning. In general, intensive grass growing can have noticeable influence on forest regeneration in burned sites.

When comparing living ground cover regeneration after surface fires in pine and spruce stands, the following things become obvious: a post fire decrease in the number of ground species is small in pine stands, while their number decreases almost 50% in spruce stands. Coefficient of species similarity with unburned forest sites is relatively high in both cases, although Stugren-Radulesku's coefficient (coefficient of species difference or discrimination) indicates noticeable differences in living ground cover composition between burned and unburned spruce sites. The most considerable

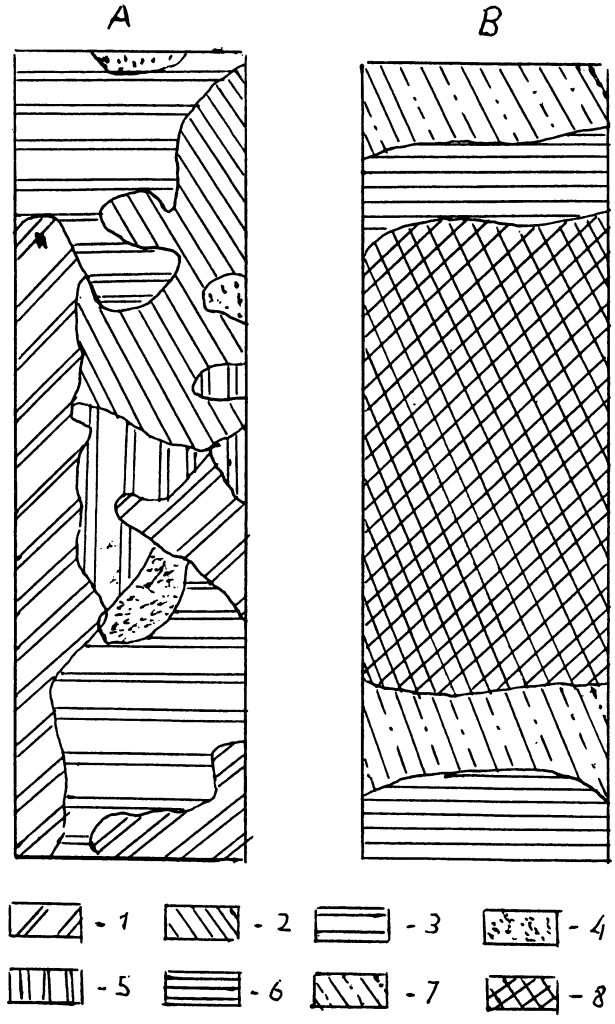

Figure 1.--*Elementary plant communities (Snusia) of* V.vitis-idaea *mixed herb living ground cover in pine stands before (A) and after (B) fires. Elementary plant communities: 1. Mixed herb-green moss; 2.* V.vitis-idaea *mixed herb; 3.* Ledum-*green moss; 4.* Ledum*; 5.* V.vitis-idaea-*green moss; 6.* Fabaceae *(mixed herbs); 7. Mixed herbs of multi-strata*

Table 3.-- *Comparison of succession in differently burned sites in a pine stand with V.vitis idaea and mixed herb ground cover.*

Indicator	Burned sites		
	Heavily burned	Moderately burned	Lightly burned
Coefficient of species similarity with unburned sites	0.55	0.67	0.90
Stugren's coefficient of difference in species composition	+0.20	+0.10	-0.68
Succession dynamics	1.60	1.10	0.28
Increase or loss of species in a coenosis.	0.13	0.18	0.20

Table 4.-- *Dynamics of post fire succession in living ground cover.*

Indicator	Pine stands	Spruce stands
Number of species under the canopy of forest	25	17
Number of species under the canopy in burned sites	21	9
Coefficient of similarity with unburned sites	0.70	0.61
Coefficient of difference	-0.60	+0.11
General dynamics	0.78	1.62
Succession dynamics	0.66	1.37

changes are observed in succession dynamics, which are manifested through quantitative characteristics of succession process. According to our data, the number of new post fire species in living ground cover composition is much bigger than that of pre-fire species in spruce stands, where succession dynamics is higher (table 4).

CONCLUSION

Changes of living ground cover after low intensity fires are short-term in pine/*V.vitis-idaea*/mixed herb stands. They do not influence significantly the structure of the cover, while they promote spread of forest grasses, especially Carex, which can in turn have some influence on forest regeneration. In sites where fire severely disturbed dead organic layer and soil, regeneration is noticeably slower, which is often reflected in decreases in grass species abundance and coverage.

Post fire observations showed that as moss cover degrades, the number of vegetation micro-groups decreases, and both the proportions of dominant species and their coenotic role change in light and dark conifer-green moss forests, which are notable for a pronounced mosaic of living ground cover. On the background of highly dynamic succession processes, the community tends to lose species, a process which can result in decreasing ground cover diversity.

When considering general dynamics of succession occuring during the initial stage of forest regeneration after surface fire, it becomes clear that early post fire succession dynamics is much higher in dark coniferous forest (a case study in a spruce - small grass - green moss stand) than in light conifers (a case study in pine stands with *V. vitis-idaea* and mixed herbs). The high succession dynamics often leads to the replacement of certain pre-fire dominant species of living ground cover by different species.

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GASEOUS AND PARTICULATE EMISSIONS FROM BURNING IN THE BOREAL FOREST

Joel S. Levine[1], Wesley R. Cofer III, Donald R. Cahoon, Jr., Edward L. Winstead[2], Brian J. Stocks[3], Veronica A. Krasovic[4] and Lawrence Mtetwa[5]

ABSTRACT.--Biomass burning is a significant regional and global source of gaseous and particulate emissions to the atmosphere. These emissions impact the composition and chemistry of the troposphere and stratosphere and affect the transfer of both incoming solar radiation and outgoing infrared radiation, and hence, impact global climate. Due to the unique characteristics of both the boreal forest and the composition and chemistry of the overlying boreal atmosphere, the gaseous and particulate missions resulting from boreal forest fires are important for both the regional and global atmosphere.

INTRODUCTION

Biomass burning is the burning of the world's vegetation, including forests, savannas, and agricultural lands after the harvest, for land-clearing and land-use change. Estimates of the annual amount of biomass matter consumed by burning in these ecosystems are given in table 1. To attest to the growing importance and interest in this subject, biomass burning has been the subject of no less than six volumes over the last 6 years detailing burning in the diverse ecosystems of our planet (Levine 1991; Crutzen and Goldammer 1993; Goldammer and Furyaev 1996; Levine 1996a, and 1996b; van Wilgen *et al.* 1997).

The immediate effect of burning is the production and release into the atmosphere of gases and particulates resulting from the combustion of biomass matter. The instantaneous combustion products of vegetation burning include carbon dioxide (CO_2), carbon monoxide (CO), methane (CH_4), nonmethane hydrocarbons (NMHCs), nitric oxide (NO), methyl chloride (CH_3Cl), methyl; bromide (CH_3Br) and various combustion particulates. If the burned vegetation is not regrown, such as most of the world's burned forests, then the liberated carbon dioxide remains in the atmosphere. If the burned ecosystem undergoes regrowth, as do the savannas, the carbon dioxide is eventually removed from the atmosphere via photosynthesis and is incorporated in the new vegetation growth. However, the gaseous emissions other than carbon dioxide remain in the atmosphere and are not reincorporated in the biosphere. Longer term effects of biomass burning include enhanced biogenic soil emissions of nitric oxide and nitrous oxide following burning. Gases produced by biomass burning are environmentally significant, including carbon dioxide and methane, which are greenhouse gases that impact global climate. particulates impact the global radiation budget, and hence, also impact climate. Carbon monoxide, methane, non-methane hydrocarbons, and nitric oxide are all chemically active gases and lead to the photochemical production of ozone (O_3) in the troposphere. Methyl chloride is also produced during biomass burning. Methyl chloride is a source of chlorine (Cl) to the atmosphere, which leads to the chemical destruction of stratospheric ozone. A single chlorine atom can destroy 100,000 molecules of ozone in the stratosphere. Very recently it was discovered that biomass burning is also an important global source of atmospheric bromine (Br) in the form of methyl bromine (Mano and Andreae 1994). Bromine leads to the chemical destruction of ozone in the stratosphere and is about 40 times more efficient in the destruction of stratospheric than is than chlorine on a molecule-for-molecule basis.

[1]*Joel S. Levine, Wesley R. Cofer III, and Donald R. Cahoon, Jr., Atmospheric Sciences Division, NASA Langley Research Center, Hampton, Virginia 23681-0001.*

[2]*Edward L. Winstead, SAIC/GATS, 28 Research Drive, Hampton, Virginia 23666*

[3]*Brian J. Stocks, Forestry Canada, Ontario Region, Sault Ste Marie, Ontario P6A 5M7 Canada*

[4]*Veronica A. Krasovic, Atmospheric Sciences Program, Applied Science Department, College of William and Mary, Williamsburg, Virginia 23187*

[5]*Lawrence Mtetwa, Atmospheric Sciences Program, Applied Science Department, College of William and Mary, Williamsburg, Virginia 23187*

Bromine leads to the chemical destruction of ozone in the stratosphere and is about 40 times more efficient in the destruction of stratospheric than is than chlorine on a molecule-for-molecule basis.

BIOMASS BURNING IN THE WORLD'S BOREAL FORESTS

It has generally been believed that biomass burning is a uniquely tropical phenomenon. This is because most of the information that we have on the geographical and temporal distribution of biomass burning is based on tropical burning. Due to poor satellite coverage, among other things, there is very little information available on the geographical and temporal distribution on biomass burning in the world's boreal forests, which cover about 25% of the world's forests. One of the largest fires ever measured occurred in the boreal forests of the Heilongjiang Province of Northeastern China in May 1987. In less than 4 weeks, more than 1.3 million hectares (1 hectare = 2.47 acres) of boreal forest were burned (Levine *et al.* 1991; Cahoon *et al.* 1994). At the same time, extensive fire activity occurred across the border in Russia, particularly in the area east of Lake Baikal between the Amur and Lena rivers (Cahoon *et al.* 1994). Estimates based on NOAA AVHRR imagery indicate that 14.446 million hectares (35.697 million acres) in China and Siberia were burned in 1987 (Cahoon *et al.* 1994).

To illustrate how are knowledge of the geographical extent of burning in the world's boreal forests has increased in recent years, consider the following: Early estimates based on surface fire records and statistics suggested that as much as 1.5 million hectares of boreal forests burn annual (Seiler and Crutzen 1980). Follow-on studies based on more comprehensive surface fire records and statistics indicated that earlier estimates underestimated burning in the world's boreal forests and that an average of 8 million hectares burned annually during the 80's, with great year to year fluctuations (Stocks 1991). The most recent study, this time based solely on satellite measurements, indicates that in 1987 alone, approximately 14 million hectares burned in the boreal forest of eastern Asia (Cahoon *et al.* 1994). While 1987 was an extreme fire year in eastern Asia, the sparse data base may suggest may suggest a fire trend. Is burning in the boreal forests increasing with time, or are satellite measurements providing more accurate data? Our feeling is that satellite measurements are providing a more accurate assessment of the extent and frequency of burning in the world's boreal forests. However, it is important to point out that as global warming becomes a reality, predicted warmer and dryer conditions in the world's boreal forests will result in more frequent and larger fires. Increased burning will have an amplifying effect on global warming!

Calculations using the satellite derived burn area and measured emission ratios of gases for boreal forest fires (Cofer *et al.* 1990 and equations 1, 2, and 3 in this paper) indicate that the Chinese and Siberian fires of the 1987 contributed about 20% of the total carbon dioxide produced by savanna burning, 36% of the total carbon monoxide produced by savanna burning and 69% of the total methane produced by savanna burning (Cahoon *et al.* 1994). Since savanna burning represents the largest component of tropical burning in terms of the vegetation consumed by fire (table 1), it is apparent that the atmospheric emissions from boreal forest burning must be included in global species budgets.

There are several reasons that burning in the world's boreal forests are very important:

1. The boreal forests are very susceptible to global warming. Small changes in the surface temperature can significantly influence the ice/snow/albedo feedback. Thus, infrared absorption processes by fire-produced greenhouse gases, as well as fire-induced changes in surface albedo and infrared emissivity are more environmentally significant than in the tropics.

2. Computer calculations of climate change indicate that in the world's boreal forests, global warming will result in warmer and drier conditions. This in turn will result in an enhanced frequency of fire and the accompanying enhanced production of combustion greenhouse gases, such as carbon

Table 1. *Global estimates of annual amounts of biomass burning and of the resulting release of carbon into the atmosphere (Andreae 1991).*

Source of burning	Biomass burned *(Tg dry matter/yr)*	Carbon released
Tropical forests	1,260	570
Temperate and boreal forests	280	130
Savannas	3,690	1,660
Agricultural waste	2,020	910
Fuel wood	1,430	640
Charcoal	21	30
World total	8,700	3,940

dioxide and methane, that will further amplify the greenhouse effect.

3. Fires in the boreal forests are perhaps the most energetic in nature. The average fuel consumption per unit area in the boreal forest is on the order of 25,000 kg/ha, which is about an order of magnitude greater than in the tropics. Large boreal forest fires typically spread very quickly, most often as crown fires. Large boreal forest fires release enough energy to generate convective smoke columns that routinely reach well into the troposphere, and on occasion may directly penetrate across the tropopause. The height of the tropopause varies from a high of about 18 km in the tropics to a low of about 8 km near the poles. Hence, the tropopause and hence, the stratosphere is at a minimum height over the world's boreal forests. As an example, a 1986 forest fire in northwestern Ontario (Red Lake) at times generated a convective smoke column 12 km to 13 km high, which penetrated the tropopause (Stocks and Flannigan 1987).

4. The cold temperature of the troposphere over the world's boreal forests results in very low levels of tropospheric water vapor. The deficiency of tropospheric water vapor and the deficiency of incoming solar radiation over most of the boreal year results in very low photochemical production of the hydroxyl radical (OH) over the boreal forests. The OH radical is the overwhelming chemical scavenger in the troposphere and controls the atmospheric lifetime of most tropospheric gases. The very low concentration of the OH radical over the boreal forests will result in enhanced atmospheric lifetimes for most tropospheric gases, including the gases produced by fire combustion. Hence, gases produced by burning, such as CO, CH_4, CH_3Cl, CH_3Br and the oxides of nitrogen will have enhanced lifetimes over the boreal forest.

GASEOUS AND PARTICULATE EMISSIONS FROM BIOMASS BURNING

Knowledge of the geographical and temporal distribution of burning is critical to assess the emissions of gases and particulates from biomass burning in to the atmosphere. More than a decade ago, Seiler and Crutzen (1980) showed that to a first approximation, the total amount of biomass (M) burned in a particular ecosystem may be given by the following equation:

(1) $M = A \times B \times a \times b$ [grams dry matter per year (g dm/yr)]

where A = total land area burned annually [m^2/yr], B = the average organic matter per unit area in the individual ecosystem [g dm/m^2], a = fraction of the above-ground biomass relative to the total average biomass B, and b = the burning efficiency of the above ground biomass. Parameters B, a, and b are determined during biomass burn field measurements in diverse ecosystems. Determination of these parameters require measurements before, during, and after burning.

Once the total amount of biomass M burned is known, the total mass of carbon M(C) (gm) released to the atmosphere during burning may be estimated using the following equation:

(2) $M(C) = 0.45\ M$

since carbon comprises about 45% of biomass by weight. The carbon released during burning takes the form of several gaseous and particulate compounds, including carbon dioxide, carbon monoxide, methane, nonmethane hydrocarbons, and particulate or elemental carbon. The ratio of any carbon compound to CO_2 produced in burning, or the ratio of a nitrogen compound to CO_2 produced in burning, may be determined by knowledge of the "emission ratio" or ER. The emission ratio is the amount of any compound X produced during burning normalized with respect to the amount of CO_2 produced during burning. The ER is usually normalized with respect to CO_2 since CO_2 is the overwhelming carbon species produced during biomass burning and it is a relatively easy gas to measure. The emission ratio, ER is defined as:

(3) $ER = \text{delta } X/\text{delta } CO_2$

where delta X is the concentration of the species X produced by biomass burning, delta X = X* - X, where X* is the measured concentration of X in the biomass burn smoke, X is the background (out-of-plume) atmospheric concentration of the species, and delta CO_2 is the concentration of CO_2 produced by biomass burning, delta $CO_2 = CO_2^* - CO_2$, where CO_2^* is the measured concentration in the biomass burn plume and CO_2 is the background (out-of-plume) atmospheric concentration of CO_2.

One of the important discoveries in biomass burning research over the last 5 years is that fires in diverse ecosystems are very different in the production of gaseous and particulate emissions. Gases and particulate emissions depend on the type of ecosystem burning, i.e., the nature and distribution of the vegetation (i.e., forests or grass), the moisture content of the vegetation as well as on the nature, behavior, and characteristics of the fire (i.e., flaming or smoldering phase, fast or slow spreading), etc. It is no longer correct to assume constant emission ratios for all fires and for the entire lifetime of a particular fire (Cofer *et al.* 1990, 1991).

The gaseous and particulate emissions produced during biomass burning are dependent on the nature of the biomass matter, which is a function of the ecosystem and the temperature of the fire, which is also ecosystem dependent. In general, biomass is composed mostly of carbon (about 45% by weight) and hydrogen and oxygen (about 55% by weight), with trace amounts of nitrogen (0.3 to 3.8% by mass), sulfur (0.1 to 0.9%), phosphorous (0.01 to 0.3%), potassium (0.5 to 3.4%) and still smaller amounts of chlorine, and bromine. During complete combustion, the burning of biomass matter produces carbon dioxide (CO_2) and water vapor as the primary products, according to the reaction

$$CH_2O + O_2 \text{ -----> } CO_2 + H_2O$$

where CH_2O represents the average composition of biomass matter. Biomass burning can be considered as an abiotic equivalent of respiration/decay. Estimates for the release of carbon (in units of Teragrams of carbon, TgC/yr, where 1 Teragram = 10^{12} grams = 10^6 metric tons) into the atmosphere from biomass burning for different ecosystems are summarized in table 1.

In the case of incomplete combustion in cooler and/or oxygen-deficient fires, i.e., the smoldering phase of burning, carbon is released in the forms of carbon monoxide, methane, nonmethane hydrocarbons, and various partially oxidized organic compounds, including aldehydes, alcohols, ketones, and organic acids and particulate black (soot) carbon. Nitrogen is present in biomass mostly as amino groups ($R\text{-}NH_2$) in the amino acids of proteins. During combustion the nitrogen is released by pyrolytic decomposition of the organic matter and partially or completely oxidized to various volatile nitrogen compounds, including molecular nitrogen (N_2), nitric oxide (NO), nitrous oxide (N_2O), ammonia (NH_3), hydrogen cyanide (HCN), cyanogen (NCCN), organic nitriles, i.e., acetonitrile (CH_3CN), acrylonitrile (CH_2CHCN), and propionitrile (CH_3CH_2CN), and nitrates. The sulfur in biomass is organically bound in the form of sulfur-containing amino acids in proteins. During burning the sulfur is released mostly in the form of sulfur dioxide (SO_2) and smaller amounts of carbonyl sulfide (COS) and nonvolatile sulfate (SO_4^-). About one half of the sulfur in the biomass matter is left in the burn ash, whereas, very little of the fuel nitrogen is left in the ash.

EMISSIONS FROM BURNING IN THE BOREAL FOREST: THE BOR FOREST ISLAND FIRE EXPERIMENT

An experimental high-intensity stand-replacement fire was conducted in Siberia on July 6, 1993. Science experiments designed around this fire were

the first joint East-West fire research activities organized under the Fire Research Campaign Asia North (FIRESCAN) and sponsored by the Siberian Branch of the Russian Academy of Sciences, the Russian Aerial Forest Fire Protection Service, the International Boreal Forest Research Association (IBFRA), the International Global Atmospheric Chemistry (IGAC) Project, part of the International Geosphere-Biosphere Program (IGBP), and the Max Planck Institute for Chemistry. A research team from the Atmospheric Sciences Division of the NASA Langley Research Center participated in this experiment and obtained samples of gaseous and particulate emissions from the fire (Cofer *et al.* 1996). The Bor Forest Island site (60° 45'N, 89° 25'E) is about 600 km north of Krasnoyarsk, Siberia, and consisted of about 50 ha of 130-year old, 20-m high, live, standing Scots Pine (*Pinus silvestris*), with a forest floor covering composed essentially of lichen (*Cladonia* sp.) residing on 5 cm to 7 cm of partially decomposed organic matter. Fuel loading was determined to be 3.4 kg/m^2. A high intensity experimental fire was possible because the burn site was a small island of live trees surrounded by relatively wet marshlands--preventing fire control problems. The fire burned from about 1430 to 1700 local time, during which time there were no significant natural winds. Due to the low wind conditions, the fire was ignited by hand torch along the perimeter, creating a convection fire. Once ignited, the fire generated its own wind field and produced a smoke column rising high into the upper troposphere. The Bor fire can be divided into three distinct phases. First, a flaming surface fire that consisted primarily of the combustion of litter, lichen, and duff on the ground. This phase transitioned into an intense flaming fire, involving explosive migrations of fire into needled tree crowns. The final phase was a smoldering stage.

Grab samples were collected using a Russian MI-8 helicopter fitted with the NASA Langley Research Center smoke sampling equipment. Particle-filtered samples were drawn (via high-volume pump) through a nose-mounted probe, coupled to the high volume pump with flexible hose. Smoke samples were fed into 10 liter Tedlar gas sampling bags. Each bag constitutes one sample, which was composed of one helicopter pass through the smoke column. After sample collection, the gas was transferred into stainless steel grab sampling bottles, which were returned to our laboratory and analyzed for CO_2, CO, H_2, and CH_4 within 2 weeks of collection. Smoke sampling was conducted at altitudes as low as safety would permit, determined most often by fire intensity and smoke turbulence. Flight paths chosen during smoke plume/column sampling were based on visual keys such as smoke color, flame characteristics, apparent turbulence, and combustion stage. It is this ability to select and sample smoke from specific parts of a fire that is the real strength of helicopter sampling. For example, when the crowns of a section of trees were observed to explode into flames, we were able to chase and sample the resulting smoke. The helicopter was always moving forward during sampling at no less than 40 knots. At 40 knots, rotor downwash is well behind the sampling probe, thereby eliminating collection of smoke other than that targeted. Mean emission ratios (ERs) normalized with respect to CO_2 for CO, H_2, and CH_4 determined for the Bor Forest Island fire are summarized in table 2. Another way to look out at the combustion emissions is the emission factor (EF), which is defined as the ratio of the mass of the combustion gas, i.e., CO_2, CO, CH_4 and H_2 (in grams) to the mass of the biomass dry fuel (in kilograms). The emission factors determined for the Bor Forest Island fire are summarized in table 3.

GLOBAL EMISSIONS OF GASES AND PARTICULATES FROM BIOMASS BURNING

By combining estimates for the global annual amounts of biomass burning given in table 1, with information on the emission ratios for various compounds produced during burning, we can estimate the global emissions of various combustion products (both gaseous and particulate) and compare the source strength of these emissions to the production of these compounds by all sources, including biomass burning. Estimates for the global emissions of gases and particulates from biomass burning and from all sources are given in table 4.

Table 2.--*Emission ratios (ERs) (in %) normalized with respect to carbon dioxide for Bor Island forest fire (Cofer* et al. *1996) The Flaming 1 measurements represent flaming combustion before the ground fire erupted into the tree crowns, and consisted of emissions produced by burning lichen, surface litter, and duff. The Flaming 2 measurements represent measurements of flaming combustion on both the ground and in the tree crowns.*

	CO/CO_2	CH_4/CO_2	H_2/CO_2
Flaming 1 (4 samples)	8.8+/-2.7	0.5+/-0.1	1.2+/-0.2
Flaming 2 (5 samples)	11.3+/-2.7	0.4+/-0.1	1.6+/-0.1
Smoldering	33.5+/-4.5	1.3+/-0.2	2.2+/-0.2

Inspection of table 4 indicates that biomass burning is a significant global source of many environmentally significant radiatively-active and chemically-active species.

It is generally believed that the vast majority of global burning is human initiated and that it has significantly increased over the last few decades (Levine 1991). During the burning of a forest, carbon dioxide that was sequestered in the forest over time scales ranging from decades and centuries is suddenly released and returned to the atmosphere in a matter of only a few short hours. The burning of the forests also destroys an important sink for atmospheric carbon dioxide. Hence, burning has both a short-term and long-term impact on the global carbon dioxide budget. The world's tropical rain forests, in addition to being an important global sink for atmospheric carbon dioxide, is also an important source of atmospheric oxygen, which is produced as a byproduct during photosynthesis, the carbon dioxide sequestering process. Yet this important and unique ecosystem is being destroyed via burning at an estimated rate of about 30 million acres per year.

Table 3.--*Emission factors (EFs): the mass of combustion gas to the mass of biomass fuel (in g/kg) for the Bor Island forest fire* (*Cofer* et al, 1996)

CO_2	1475+/-40
CO	180+/-40
CH_4	4.2+/-0.8
H_2	0.8+/-0.3

CHEMISTRY OF THE HYDROXYL RADICAL (OH) IN THE TROPOSPHERE

The hydroxyl radical (OH) is the major chemical scavenger in the troposphere and it controls the atmospheric lifetime of most gases in the troposphere. The atmospheric lifetime, t, of any gas, x_i, that reacts with the OH radical is given by the following expression:

$$t = 1/k\,[OH]$$

where k is the kinetic reaction rate for the reaction between OH and x_i and [OH] is the concentration of the OH radical (molecules cm^{-3}).

The concentration of the OH radical is controlled by the balance between its chemical production and destruction. The hydroxyl radical (OH) is formed by the reaction of excited atomic oxygen ($O(^1D)$) with water vapor (H_2O):

$$O(^1D) + H_2O \text{ -----> } 2OH,\ k_1$$

Tropospheric excited atomic oxygen ($O(^1D)$) is produced by the photolysis of ozone (O_3). In the troposphere, this photolysis reaction occurs over a very narrow spectral interval, between 290 nm and 310 nm. The production of excited atomic oxygen decreases as the latitude increases, i.e., less incoming solar radiation for photolysis is available. The bulk of the water vapor in the atmosphere resides in the troposphere. The amount of H_2O in the atmosphere is controlled by the saturation vapor pressure, which decreases with decreasing atmospheric temperature, i.e., as altitude or latitude increases.

Table 4.--*Comparison of global emissions from biomass burning with emissions from all sources (including biomass burning) (Andreae 1991).*

Species	Biomass Burning Tg element/yr	All sources Tg element/yr	Biomass burning
Carbon dioxide (Gross)	3.500	8,700	40
Carbon dioxide (Net)	1,800	7,000	26
Carbon monoxide	350	1,100	32
Methane	38	380	10
Non-methane hydrocarbons	24	100	24
Nitric oxide	8.5	40	21
Ammonia	5.3	44	12
Sulfur gases	2.8	150	2
Methyl chloride	0.51	2.3	22
Hydrogen	19	75	25
Tropospheric ozone	420	1,100	38
Total particulate matter	104	1,530	7
Particulate organic carbon	69	180	39
Elemental carbon (black soot)	19	<22	>86

The OH radical is destroyed via its reactions with CO and CH_4, both important products of biomass burning:

$OH + CO \text{ -----> } H + CO_2$, k_2, and

$OH + CH_4 \text{ -----> } CH_3 + H_2O$, k_3

The concentration of the OH radical (molecules cm^{-3}) is determined by dividing the OH production term by its destruction terms:

$$[OH] = 2 k_1 [H_2O]/k_2 [CO] + k_3 [CH_4]$$

PRODUCTION OF OZONE (O_3) IN THE TROPOSPHERE

In addition to controlling the chemical destruction of OH, the oxidation of CO and CH_4 by OH outlined above, initiates the carbon monoxide and methane oxidation schemes, which lead to the photochemical production of O_3 in the troposphere

Carbon Monoxide Oxidation Chain

$CO + OH \text{ -----> } CO_2 + H$

$H + O_2 + M \text{ -----> } HO_2 + M$ (H = atomic hydrogen, HO_2 = hydroperoxide radical)

$HO_2 + NO \text{ -----> } NO_2 + OH$ (NO = nitric oxide, NO_2 = nitrogen dioxide)

$NO_2 + hv \text{ -----> } NO + O$ (O = atomic oxygen)

$O + O_2 + M \text{ -----> } O_3 + M$ (O_2 = molecular oxygen)

Net Reaction: $CO + 2O_2 \text{ -----> } CO_2 + O_3$

Note that the key gases in the carbon monoxide oxidation chain leading to the photochemical production of tropospheric ozone, CO, and NO, are both produced by biomass burning.

Methane Oxidation Chain

$CH_4 + OH \text{ -----> } CH_3 + H_2O$ (CH_3 = methyl radical)

$CH_3 + O_2 + M \text{ -----> } CH_3O_2 + M$ (CH_3O_2 = methylperoxyl radical)

$CH_3O_2 + NO \text{ -----> } CH_3O + NO_2$ (CH_3O = methoxy radical)

$CH_3O + O_2 \text{ -----> } CH_2O + HO_2$ (CH_2O = formaldehyde)

$HO_2 + NO \text{ -----> } NO_2 + OH$

$2 * (NO_2 + hv \text{ -----> } NO + O)$
$2 * (O + O_2 + M \text{ -----> } O_3 + M)$

Net Reaction: $CH_4 + 4O_2 \text{ -----> } CH_2O + H_2O + 2O_3$

Note that the key gases in the methane oxidation chain leading to the photochemical production of tropospheric ozone, CH_4, and NO, are both produced by biomass burning.

CHEMISTRY OF NITROGEN OXIDES IN THE TROPOSPHERE

In addition to being a key player in the carbon monoxide and methane oxidation chains leading to the chemical production of ozone in the troposphere, NO also leads to the chemical production of nitric acid (HNO_3), the fastest growing component of acidic precipitation. NO is chemically transformed to nitrogen dioxide (NO_2) and then to nitric oxide via the following reactions:

$NO + O_3 \text{ -----> } NO_2 + O_2$

$NO + HO_2 \text{ -----> } NO_2 + OH$

$NO + CH_3O_2 \text{ -----> } NO_2 + CH_3O$

$NO_2 + OH + M \text{ -----> } HNO_3 + M$ (HNO_3 = nitric acid)

CHEMISTRY OF THE STRATOSPHERE

Approximately 90% of the ozone in the atmosphere is found in the stratosphere (15 - 50 km), with about 10% in the troposphere (0 - 15 km). Stratospheric ozone is very important because it absorbs ultraviolet radiation (200 nm - 300 nm) from the Sun and shields the surface from this biologically lethal radiation. Stratospheric ozone is destroyed via a series of chemical reactions involving chlorine and bromine. Chlorine destroys stratospheric ozone through the following catalytic cycle (Wayne 1991).

$Cl + O_3 \text{ -----> } ClO + O_2$

$ClO + O \text{ -----> } Cl + O_2$

The net reaction is $O + O_3 \text{ -----> } 2O_2$

Bromine destroys stratospheric ozone via the following three reaction cycles (World Meteorological Organization 1995)

Cycle 1

$BrO + ClO + hv \text{ -----> } Br + Cl + O_2$ (BrO = bromine monoxide, ClO = chlorine monoxide)

$Br + O_3 \text{ -----> } BrO + O_2$ (Br = atomic bromine)

$Cl + O_3 \text{ -----> } ClO + O_2$ (Cl = atomic chlorine)

The net reaction is $2O_3 \text{ -----> } 3O_2$

Cycle 2

$BrO + HO_2 \text{ -----> } HOBr + O_2$
$HOBr + hv \text{ -----> } OH + Br$

$Br + O_3 \text{ -----> } BrO + O_2$

$OH + O_3 \text{ -----> } HO_2 + O_2$

Cycle 3

$BrO + NO_2 + M \text{ -----> } BrONO_2 + M$ ($BrONO_2$ = bromine nitrate)

$BrONO_2 + M \text{ -----> } Br + NO_3$ (NO_3 = nitrate)

$NO_3 + hv \text{ -----> } NO + O_2$

$Br + O_3 \text{ -----> } BrO + O_2$

$NO + O_3 \text{ -----> } NO2 + O_2$

There are two other channels for the reaction between BrO + ClO (Wayne 1991)

$BrO + ClO \text{ -----> } Br + OClO$

$BrO + ClO \text{ -----> } BrCl + O_2$

There is also a relevant reaction between BrO and itself (Wayne 1991)
$BrO + BrO \text{ -----> } Br + Br + O_2$

ATMOSPHERIC NITROGEN AND OXYGEN

Biomass burning is both an instantaneous source (combustion of biomass) and a long-term source (enhanced biogenic soil emissions via increased

nitrification and denitrification in soil) of gases to the atmosphere. These gases impact the chemistry of the troposphere and stratosphere, as outlined in this paper. In addition, biomass burning may also impact atmospheric concentrations and the biogeochemical cycling of nitrogen and oxygen, the two major constituents of the atmosphere. Through the process of nitrogen fixation, molecular nitrogen (N_2) is transformed to the surface in the form of "fixed" nitrogen, i.e., ammonium (NH_4^+) and nitrate (NO_3^-). Nitrogen fixation results from both natural processes (biological fixation in root modules in certain agricultural crops and atmospheric lightning) and human processes (the production of nitrogen fertilizer and high temperature combustion). The world's use of industrially fixed nitrogen fertilizer has increased from about 3 Teragrams (Tg) of N/year in 1940 to about 75 Tg N/yr in 1990 (Levine *et al.* 1996b). The "fixed" nitrogen in the forms of NH_4^+ and NO_3^- is returned to the atmosphere mainly in the form of N_2, with smaller amounts of N_2O, and still smaller amounts of NO by denitrification and in the form of NO by nitrification. Burning or "pyro-denitrification" may also be an important source of nitrogen, mostly in the form of N_2, from the biosphere to the atmosphere (Lobert *et al.* 1991, Levine *et al.* 1996a). The problem is that it is difficult to quantify the amount of N_2 released during burning (Lobert *et al.* 1991), however, biomass burning or pyrodenitrification may prove to be an important process in the recycling of nitrogen compounds from the biosphere to the atmosphere.

Burning impacts the concentration of atmospheric oxygen in two ways. Carbon released during the burning of biomass combines with atmospheric oxygen to form carbon dioxide. Hence, burning is a sink for atmospheric oxygen. In addition, biomass burning destroys the very source of atmospheric oxygen--its production in the biosphere via the process of photosynthesis in the world's forests.

Due to its impact on the chemical composition and chemistry of the regional and global atmosphere and on global climate, biomass burning is an important driver for global change on our planet (Levine *et al.* 1995). A major uncertainty in our understanding of the impact of burning on the atmosphere and climate is the geographical and temporal distribution of fires. Space offers a unique vantage point for these measurements. A dedicated mission, called FireSat, has been developed to measure/monitor the geographical and temporal distribution of global fires (Levine *et al.* 1996a). The FireSat instrument is a seven-channel radiometer to monitor/measure active fires, fire scars, and combustion aerosols in the atmosphere. FireSat can record fires with a ground resolution of 250 m and monitor about 95% of the Earth's surface in a single day from its 830 km orbit.

CONCLUSIONS

In addition to being a significant instantaneous regional and global source of gases and particulates to the atmosphere, burning also enhances the biogenic emissions of nitrogen and carbon gases from the world's soils to the atmosphere (Levine *et al.* 1997). In addition, biomass burning impacts the reflectivity and emissivity of the Earth's surface. Burning also affects the hydrological cycle by changing rates of land evaporation and water runoff. In December 1997, representatives from the countries of the world will meet in Kyoto, Japan, to discuss global warming and ways to reduce the build-up of greenhouse in the atmosphere. The production of atmospheric carbon dioxide by global burning will be an important point of discussion at this international conference.

For the reasons discussed in this paper, it appears that biomass burning in general, and burning in the boreal forests, in particular, is a significant driver for global change. It seems appropriate to conclude this summary with a quotation from fire historian, Stephen J. Pyne (1991):

We are uniquely fire creatures on a uniquely fire planet, and through fire the destiny of humans has bound itself to the destiny of the planet.

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Impacts of Severe Weather

ECOLOGICAL APPROACH TO ASSESSING PINE FOREST DISTURBANCE IN CENTRAL SIBERIA

V. A. Ryzkova, F. I. Pleshikov, V. Y. Kaplunov, and J.V. Usoltzeva[1]

ABSTRACT. -- Estimates are given of levels of pine stand disturbance in south Central Siberia by the November 1994 hurricane. Major factors accounting for decrease in stand resistance to wind are determined. These include: specific environmental conditions (complex relief patterns, ground structure, and light soil structure), infestation by fungi (*Heterobasidion annosum* (Fr.) Bref.), and disturbance of natural conditions by human economic activity. Measures are suggested to optimize wind resistant pine stand formation.

INTRODUCTION

In Siberia, pine forests are widespread on big river terraces and along glacial water runoffs. They are unique natural objects, which grow along the southern boundary of the forest vegetation area and cover some 65,000 ha. These pine stands are considered to be particularly valuable and fall in Group I of the Forest Value Classification System. They protect soil and water bodies, and perform sanitary and recreational functions.

Over recent decades, the tendency for worsening of the ecological state of pine stands has become stronger. They are periodically damaged by dendrophilous insects, wild animals, cutting, and hurricanes. Windfalls are a very remarkable feature of the present appearance of the pine stands. Due to the light mechanical composition of the soils, tree infestation by fungi (*Heterobasidion annosum* (Fr.) Bref.) the dune/hilly relief, and the high density of woody vegetation at some sites, the pine stands are regularly disturbed by strong winds, which results in considerable economic and environmental damage. In November 1994, strong winds (more than 30m/s) made about 300,000 cubic m of wood fall in an area exceeding 2.5 million ha. Wind-caused tree root disturbance had led to weakening of the pine stands over large areas. This makes the understanding of pine stand resistance to wind a very serious and important research problem.

An ecological approach is described for assessment of pine forest disturbance caused by anthropogenic factors and the influence of its effects on stand wind resistance. The solution of this problem should be based on full accounting for all environmental factors, as well as thorough study of forest soils, site conditions, stand composition, structure and productivity specific to geomorphological situations, and analyses of forest planting methods (Rozkov and Kozak 1989; Mitchell 1995).

Our investigation encompassed the following interrelated aims:

- determine the scale of pine stand disturbance caused by the November 1994 hurricane;
- estimate factors that influence forest resistance to wind in relation to local environmental conditions;
- make recommendations on how to improve methods of wind resistant plantation establishment.

METHODS

Over many years, scientists of the Institute of Forest, Siberian Branch, Russian Academy of Sciences, have performed complex studies in pine stands in the south of Central Siberia dealing with soils, site conditions, forest productivity, and the current ecosystem state (Soil Factors Controlling Pine Stand Productivity, 1976). The work involves representative sites, transects, landscape profiles, permanent sample sites, and the use of 1:12,000 - 1:200,000 aerial images.

As for the methods used, we collected information from a variety of sources, ranging from space imagery to ground observations, to result in different scales of geographic detail output. This involved use of available forest inventory data, ecological maps, and information on the current forest state. When working on the representative sites and transects, we plotted the boundaries of disturbed pine stands on maps and aerial images. In addition, we used data from observation routes covering the whole of the area of interest. Wind-induced snapping of tree trunks and windfall structure were studied at permanent sample sites.

[1]*Institute of Forest, Siberian Branch, Russian Academy of Science, 660036, Akademgorodok, Krasnoyarsk, Russia Phone: (7)(3912)49-46-67 / Fax:(7)(3912)43-36-86 / e-mail: dndr@ifor.krasnoyarsk.su*

RESULTS

The results of our investigation show that, in the south of Central Siberia, the level of wind-caused pine stand disturbance is determined by topographic (relief, soil-ground thickness structure), biological (soil richness and stand age, density, productivity, and horizontal structure), and economic activity factors. In figure 1, an ecological information-based map is presented of one of the studied pine stands. Each zone in the map represents a certain type of site conditions, which are relatively uniform with respect to the complex of environmental factors, such as mesorelief type, general slope aspect, lithologic parameters, soil, and vegetation pattern.

Below is a brief description of the zone:

1. Situation of soft slopes and bald hills with northwest aspect underlain by sandy and loamy deposits and colonized by pine stands on well-developed loamy sand soils; site-specific stand productivity (SSSP) of classes II and III.

2. Areas of relatively flat and undulating relief mostly with northwest aspect underlain by three-layer deposits (loam/sand/loam) and colonized by pine forest on loamy soils; SSSP classes Ia, I, and II.

3. Areas of small hills and bald mountains with northwest aspect underlain by two-layer deposits (loam and sand) and colonized by pine and birch stands on loamy soils; SSSP classes I and II.

4. Areas characterized by soft bald hills with southeast aspect and by pine and mixed birch/pine stands on light loams; SSSP classes I and II.

5. Softly sloping erose surfaces of the southeastern aspect underlain by two-layer deposits and colonized by pine forest on loamy sand; SSSP classes II and III.

6. Heavily broken areas of dune-like relief underlain by sandy deposits and colonized by pine forest on immature sandy and primitive soils; SSSP classes IV and V.
7. Flat and gently sloping surfaces characterized by stratified loamy sand soils supporting pine stands; SSSP classes II and III.

Each zone on the map has its own specific combination of the above mentioned environmental factors and is characterized by a certain level of forest disturbance due to wind. The highest disturbance is observed in the sites of mostly northwest aspect with relatively flat, slightly undulating or gently-sloping/small bald hilly relief (zones 1, 2, and 3), which were directly affected by the hurricane. In these areas, relatively rich soil and site conditions of classes I and II promote highly productive mature pine stands. Wind-fallen trees account for 30%-50% of their total number. The sites that suffered most (wind felled up to 90% of trees) are shown in the map as areas of continuous windthrow.

Zones 4, 5, and 7 represent a macroslope of the southeast aspect. Forest disturbance is considerably less here (disturbance of 10%-20% on the average). However, individual wind-thrown trees are found everywhere. The factors accounting for a lower disturbance level might include not only an opposite aspect, but also lower pine density and productivity, as well as the presence of mixed birch-pine stands with grass ground cover and young pine stands.

The areas largely represented by 50-60 year-old and young pine stands (6), where site conditions are poor and relief is heavily broken (dunes and hills), appeared to be the most resistant to wind.

To sum up, the highest disturbance occurs in mature pine stands (SSSP classes I and II) covering northwest slopes (more than 15°) easily affected by wind, and with relatively flat, softly sloping/hilly surfaces.

Based on an ecological map previously created with help of our original technique (Ryzkova V.A., Pleshikov F.I., Tcherkashin V.P. 1992) we could perform a more detailed analysis of areas with continuous windthrow. Fragments of a large-scale ecological map and a vegetation map were entered into a computer, overlaid, and analyzed using a forest inventory database. Integral indicators of the quality of site conditions are as follows: actual and potential productivity, and the percentage of realization of potential site condition by vegetation.

Figure 2 shows fragments of large-scale maps of forest sites disturbed by wind and represented by different types of site conditions. Site A (site condition type 2; fig. 1) is characterized by rich soils and stands of high productivity. Sites where windthrow is the highest have been subject to partial logging over the last 10-15 years; this has decreased the wood biomass amount and disturbed the natural stand structure. For these areas, average realization of site potential by vegetation is 45%-65%. Man-caused changes of the natural structure of these pine stands resulted in their low resistance to negative environmental factors including wind.

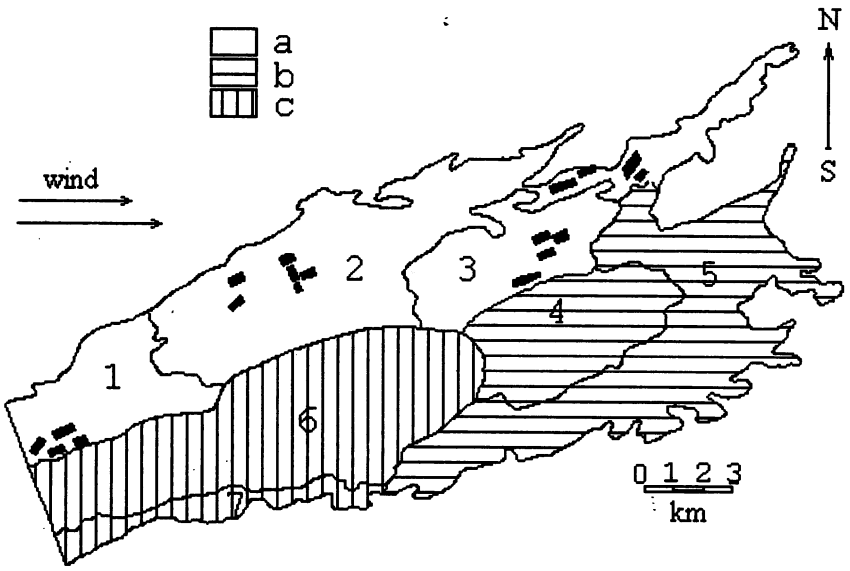

Fig.1.--A map of disturbances of a forest management area (pine forest belts in the south of Central Siberia) caused by a hurricane in November 1994.
Disturbance levels:
a - heavy, b - medium, c - light.
1-7 - types of site conditions (see text for explanation).
- boundaries of site types.
- places of continuous windfall (wind-fallen trees are 80- 90 %).

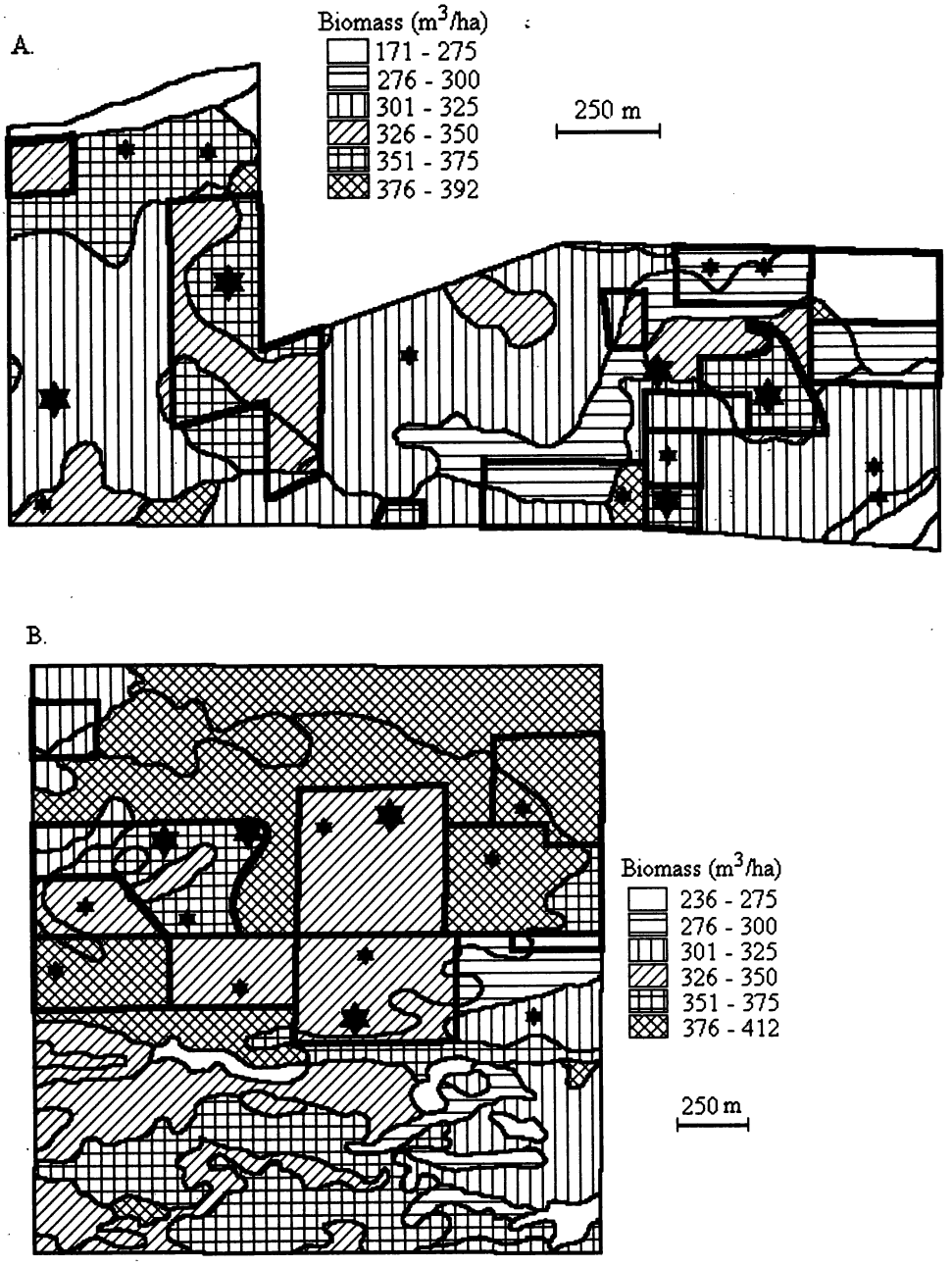

Fig.2.--Disturbance of forest sites (A and B) by 1981-1989 partial cuts and 1994 windfall.

- forest site completely killed by wind (80-90% of wind-fallen trees).
- places where wind-fallen trees account for 30 to 50 %.
- contours on the ecological map.
- boundaries of sites subject to partial cutting.
- permanent sample sites.

Site B is situated at the boundary between site conditions of types 1 and 6. In the upper part of the site, mature stands prevail (SSSP classes II and II), which have also experienced selective logging. The coefficient of site potential realization by vegetation is 40%-50%. This is where the windthrow was extremely high.

The lower part of the site is very different from the upper part in terms of type of site conditions (shallow sandy soil, dune/small hill relief), SSSP class (IV and V), and stand age structure. Here, a considerable area is covered by natural young pine stands. The coefficient of realization of site potential by vegetation is 90%-100%. That means that the natural forest cover structure is only slightly disturbed here. Single wind-thrown trees can be found in saddles between dunes (where site SSSP class increases to III). In general, this part of the pine forest was left practically unaffected by the windthrow. One of the factors that contributed to the high wind resistance of this area is that it had not been disturbed by logging because of low productivity of the stands.

Analysis of the vegetation inventory data obtained from the permanent sample sites allowed us to determine the reasons and identify the mechanisms of decreasing stand resistance to wind after partial logging. Since there are ecological constraints for forest thinning operations in the forest-steppe zone, these stands remain very dense up to the age of 50-60 years. As a rule, pine stands are intensively thinned during partial logging operations. According to our data, some 11%-38% of the total stand wood biomass amount is removed by partial logging. Since the remaining mature trees grow slower, they adapt themselves to the altered ecological situation slowly and become very sensitive to wind. Investigation of the last windthrow in the pine stands under consideration showed that some 90% of disturbed stands had been subject to partial thinning of different extents in the preceding years.

We failed to identify any relationship between the space available for a tree to grow and its resistance to wind. We can only mention one thing: the biggest number of wind-thrown trees occurred in less dense stands, with other parameters being the same.

CONCLUSIONS

Pine stands of the southern part of Central Siberia are remarkable for their generally low resistance to strong wind influence. The two major factors accounting for this are the specific characteristics of the environmental conditions and disturbance of natural vegetation communities by human economic activities. Least wind resistant are the mature, high-productivity pine stands with discontinuous horizontal canopy closure growing on pure loamy sand deposits, which result in only surface root system development. Underlying loamy sand which is reachable by tree roots improves tree resistance to wind significantly due to the better development of main roots. Infestation of the pine stands, which grow densely up to the age of 50-60 years, by fungi (*Heterobasidion annosum* (Fr.) Bref.), following intensive thinning of the stands, during partial logging operations, also contributes to pine wind resistance decrease.

Pine stand resistance to wind can be improved through optimizing forest maintenance based on well-planned thinning activities. The latter should be aimed at promotion of strong root systems and long crowns of trees (Stoyko 1965). When planning forestry activities dealing with increasing stand resistance to wind, one should take into account site conditions, ground thickness structure, topography, forest land structure, and the inventory/morphological characteristics of the future vegetation communities that will in time replace disturbed forests.

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DISTURBANCES IN THE SCOTS PINE FORESTS OF SOUTH SIBERIA CAUSED BY OCCASIONAL STORM WINDS

A. I. Buzykin, L. S. Pshenichnikova, N. M. Tchebakova[1]

ABSTRACT. -- Scots pine forests stretching along the sandy bands of ancient alluvial sediments and granite outcrops in the south of Siberia are called band-like pine forests. These forests are of special value for protecting adjacent soils, agricultural fields, and rivers. The stands have become too dense because of limited forestry activities. A significant part of them has become weakly resistant to wind. As a result of the strong cyclonic storm of 1994 (more than 30 m/s windspeeds), dense stands of 90 and more years old were significantly damaged or killed. Now, these areas are under threat of spreading forest fires and insect outbreaks. The possible causes of the powerful windthrow in these forests are: (1) In dense stands, the tree root systems do not develop properly to support trees under wind impacts; and (2) Dense stands are also subject to *Fomitoptis annosa* invasions, which damage the roots. Thinning band-like pine forests combined with introduction of deciduous tree species like birch is an effective measure both to form thick-set stems with vigorous root systems and to inhibit *Fomitopsis annosa* invasions.

INTRODUCTION

This study addresses the results of our investigation on possible causes of a catastrophic windfall of 1994 in Scots pine forests in the south of the Krasnoyarsk Territory, Siberia. The pine (*Pinus sylvestris*) forests called band-like pine forests are scattered in small separated groves in steppe areas or in band-shaped stands stretching along ancient alluvial sediments and granite outcrops on sandy soils in the south of Siberia. These band-like pine forests are ecologically valuable because they perform important protection and recreation functions. Only intermediate cuttings of different intensities for protection against forest fire are being performed in these forests. Because of limited forestry operations most band-like pine forests appeared to be weakly-resistant to wind. A dramatic windthrow resulted from a cyclonic storm in 1994. Pine forests were damaged over an area of 28,000 ha, and the volume of the fallen and broken trees was as much as 200,000 m^3.

It is a well known fact that windthrows depend on a number of factors such as: topography, soil properties, tree species, stand age and structure, tree diseases (Belov 1983, Skvortsova *et al.* 1983, Kenstavichyus 1984, Burenina 1988, Mitchell 1995, Ruel 1995).

Our study showed that the main cause of weak resistance to wind was high density of stands. This is primarily due to two factors: (1) tree root systems do not develop properly in over-stocked stands; and (2) roots are affected by *Fomitopsis annosa* (in Russian "root sponge") to a greater extent in dense stands than in thinned stands. Moist, rich, and light by mechanical composition soils also condition too weak root systems to support vigorous above-ground parts of trees.

One can rightfully conclude that to grow wind-resistant and productive stands for strong protective objectives, it is necessary to reduce stand density by regular intermediate cuttings as stands develop and to replace mature and overmature stands by young generations through clear cuts.

METHODS AND RESULTS

Region Characteristics

The study was conducted in the Minusinsk forestry management area located in the forest-steppe zone. The climate of the region is characterized by warm summer (July temperature 19.6°C) and cold winter (January temperature -21.2°C) with little snow, and low annual precipitation (250-350 mm). Soils are forest gray, sandy and loamy sandy. Average age of the band-like pine forests is 78 years. Productivity expressed through site quality class is II.2. According to Orlov's (1929) classification used in Russian inventory, class I is good, III is intermediate, and V is poor. Forest types with green moss in the ground cover dominate (75%); forest types with forbs in the ground cover amount to 12%. The main peculiarity of the band-like pine forests is their high density: according to the

[1] *Institute of Forest, Russian Academy of Sciences (Siberian Division), Academgorodok, 660036 Krasnoyarsk, Russia. Fax: +7-3912-433686, E-mail: ifor@krsk.infotel.ru.*

standard silviculture norms for the region, dense stands with density 0.8 and more cover 81% of the forested land.

Methods

About 30,000 ha of the band-like pine forests have been examined to evaluate the type and degree of damage caused by the storm wind of 1994 as well as their current state. Forty temporary sample plots were established to evaluate real densities of the band-like pine stands and correctly identify intermediate cutting regimes (intensity and frequency) in them. Plot size varied from 0.02 to 0.5 ha and the number of trees varied from 200 per plot in mature stands to 400 per plot in young stands. About 200 sample trees were felled and 100 stumps and 200 cores were additionally used to characterize stand structures. Healthy, damaged, and killed trees were enumerated. Standard inventory characteristics like diameter, height, number of trees, volume, actual stand basal area, and normalized stand basal area (NSBA, a ratio of actual stand basal area to that of an ideal stand with maximum crown cover) were used. For comparison, standard inventory data on wood volume taken out of stands by intermediate cuttings and stand states after intermediate cuttings were additionally used in the analysis.

Analysis of the Field Data

The pine forests most damaged by the powerful windthrow of 1994 were examined in 1996. All fallen trees were found to be infected by *Fomitopsis annosa*. The decay was detected mainly in roots and sometimes above the root neck. Causing destruction of the root system followed by its death, the root sponge is as a rule a cause of a massive windfall or stand drying off and mortality. Our examination showed that on average each hectare of the pine forest of premature age (60 years and older) contained 20-25 m^3 of wind-fallen wood. This phenomenon can be conditionally called a specific "wind" method of intermediate thinning. In order to prevent spontaneous damage of stands by occasional winds, it is necessary to replace "wind" thinning by controlled purposeful decrease of a stand density by regular intermediate cuttings while forming a stand; as well to replace dense mature stands (100 and over years old) by younger generations as early as possible.

How occasional, unique, or frequent are windthrows in the band-like pine forests? It is difficult to settle this question. Obviously, there is a real constant threat of windthrows. Evidence of wind catastrophes reflected by such specific nano-relief elements as small bumps and depressions left after wind-fallen trees witness this phenomenon. Our plot data showed that high density stands dominate in fact in all age groups (table 1).

Table 1. -- *Characteristics of* Pinus sylvestris *stands examined in the field.*

Stand age (yr)	Average diameter (cm)	Average height (cm)	Number of trees per ha	Actual basal area (M^2/ha)	Normalized basal area, portion of "1"	Volume, (M^3/ha)
18	2.4	3.4	37538	18.8	1.84	48.8
30	4.3	6.5	18613	27.9	1.40	141.5
34	8.4	10.9	5750	31.6	1.16	200.1
45	8.7	12.2	3850	42.6	1.46	148.2
45	6.7	10.0	5425	35.0	1.29	116.1
45	8.7	12.2	3824	36.8	1.26	126.1
51	10.7	14.6	3593	32.3	1.13	300.0
67	14.9	15.9	1781	32.0	0.97	250.0
90	23.0	21.6	1018	42.4	1.21	376.7
91	19.6	20.0	1274	38.5	1.12	382.2
96	22.3	20.4	1181	46.2	1.36	416.5
103	27.5	25.1	846	50.3	1.38	507.6

Analysis of the Inventory Data

According to the previous inventory of 1989-90, an average NSBA of pine stands in the Minusinsk leskhoz is 0.63 with low density stands 4.2%, medium density stands 85.3%, and high density stands 10.1%. Based on the inventory data, these 10% high density stands are highest priority for intermediate cuttings. This number appeared to be underestimated due to groundlessness of a local density standard. In fact, the real average density measured by NSBA from our field data was 0.80 with dense stands (NSBA is 0.80 and greater) growing on 80% of the pine forested area.

For the period of 1989-99, the area subject to intermediate cuttings in the leskhoz was only 4,000 ha or 7.9% of the forested area (table 2). Too small area and not intensive intermediate cuttings attract attention. Of course, low intensity cuttings may be compensated by higher frequency. However, frequent or moderately frequent cuttings can be hardly justified from forestry-ecological considerations. Cuttings and especially taking off the wood by machines significantly damage stems and canopies if done in winter; when done in a warm period, tree root systems are additionally damaged. In any event, sanitation states of stands are being worsened under frequent cuttings.

The analysis of industrial intermediate cuttings made in the Lugavsk forestry district showed that a system of low intensity cuttings dominated. They provide minimum interference into a natural growth and development process of stands (table 3). Stands are supported in dense and productive states. From the forestry view point, such a system of intermediate cuttings based on natural tree mortality and favoring growth of high productive stands is quite justified if high density does not inhibit tree growth and decrease weak-resistance to wind. Even in this case, the real intensity of the intermediate cuttings in the young stands (10-11% of volume in the Minusinsk leskhoz) and especially the intensity of increment cuttings (5.5% in the Lugavsk district) seem to be extremely insufficient. More frequent intermediate cuttings in these stands

Table 2. -- *Predicted volume of intermediate cuttings in the Minusinsk area (in 1990-2000 yrs.).*

Type of intermediate cuttings	Characteristics of stands assigned for intermediate cuttings			
	Stand age (yr)	Total area of stands aged for intermediate cuts	Stand area in need of intermediate cuts (ha/%)	Intensity of intermediate cuts by mass (%)
After young growth	before 40	2,325	880/37.9	14.3
Thinning	41-60	6,555	1,538/23.5	13.8
Increment	after 61	41,546	1,578/3.8	11.8
TOTAL		50,426	3,996/7.9	12.9

Table 3. -- *Average data on intermediate cuttings performed in 1970-1995 yrs. in the Lugavsk district.*

Intermediate cutting type	Characteristics of stands assigned for intermediate cuttings			
	Age, yr.	Normalized basal area, portion of "1"	Site quality class	Volume, m^3
After young growth	20-30	0.79	II.6	105
Thinning	60	0.82	II.2	225
Increment	70	0.81	II.2	256

are required to grow wind-resistant trees, although the cuttings are less profitable from both technical and economical perspectives compared to high intensity cuttings.

The Lugavsk forestry district represents a good model of growing highly-productive (dense and closed) stands based on a full use of environmental resources but without taking into account a resistance to wind. Because the band-like forests are occasionally subject to catastrophic storms and windfalls are a real threat, improving a stand resistance to wind is the main purpose of forming stable stands. Purposeful growing up such stands is possible only under a system of intermediate cuttings of high intensity. According to our data, intensity of intermediate and thinning cuttings in the young stands should be doubled and be about 30% of the primary stand volume.

DISCUSSION

- There are no general patterns of how intensive intermediate cuttings should be in the band-like pine stands. Stand norms for intermediate cuttings should be differentiated regarding potential resistance to wind based on following:
- presence/absence of previous or current
- evidence of windfalls;
- stand site topography regarding prevailing wind directions with emphasis to the warm period, e.g., windward slopes of hills and dunes or leeward slopes of hills, inter-dune depressions, and other topographic elements protected from wind;
- a sanitation state of a stand;
- protective functions of the band-like stands on the borders with non-forested areas, e.g., along permanent and temporary streams and other geomorphological elements risky from erosion view point.

In dense stands (NSBA more than 1.0) under age 40, high intensity cuttings are recommended to decrease NSBA down to 0.6-0.7. In older stands, under thinning and especially increment cuttings, taking out a big number of trees is not desirable. NSBA in such stands should remain about the same 0.7-0.8. If stands thinned suddenly, at one go, there may arise a danger of wind-fallen trees because of their weak root systems and of broken trunks because trunks are not enough set-up. In this case, selection should be done in two phases. The repeated thinning should be made 5-7 years later.

Trees in the thinned stands respond to continuous strong wind impacts by changes of trunk shapes, by development of stronger root systems, and by forming tractive wood of the main roots. The result is that trees become more resistant to wind. Doing this, we inhibit a development and expansion of *Fomitopsis annosa.*

Intermediate cuttings should be carry out in winter when there is no threat of affecting stumps with *Fomitopsis annosa* spores. To improve stand sanitation conditions, healthy birch and aspen trees are desirable to leave uncut. In mature and overmature stands infected by *Fomitopsis annosa*, selective sanitation cuttings are expedient. Under massive tree damage by the "root sponge", complete sanitation cuttings are required. If cuttings are not carried out in winter then it is necessary to treat stumps with antiseptics or to excavate them and burn them.

If an initial high intensity thinning in young stands is conducted in two phases, then the later need for high intensity thinning no longer arises and increment cuttings may not be required at all. The entire cycle of forest-growing is controlled then by several consistent ways of intermediate cuttings regarding an individual stand state.

CONCLUSION

To conclude, it is necessary to replace low intensity intermediate cuttings (thinnings) with medium and high intensity intermediate cuttings to form and grow stands with increased resistance to wind. All stands that are over-stocked to different extents should undergo high intensity intermediate cuttings. In fact, this means that about 80% of the band-like pine forests should undergo repeated intermediate cuttings for the forest-growing period. We speak here about a forestry-environmental necessity of significant (several times) increase of the area which is to undergo all types of intermediate cuttings and wood volume taken out in the process of cuttings. Cuttings in premature and mature stands where clear cuts are not allowed are included as well. All these cuttings are renovation, transformation, and sanitation-reconstruction ones.

Our results show that decreases of the current stand volume on an area basis that follow apparently after intermediate cuttings to improve wind-resistance in band-like pine forests do not much alter total stand productivity, if the volume of wood taken out by intermediate cuttings is included.

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ECOPHYSIOLOGICAL RESPONSE TO INTERACTING EFFECTS OF DROUGHT AND NITROGEN, AND REVERSIBILITY OF DROUGHT EFFECT IN PEATLAND AND UPLAND BOREAL SPRUCE*

Q.L. Dang[1], T.B. Patterson[2] and R.D. Guy[3]

ABSTRACT.--Seeds of peatland black spruce (*Picea mariana* (Mill.) B.S.P.) and upland white spruce (*Picea glauca* (Moench) Voss) were obtained from five forest stands in the boreal forest near Fort Nelson, British Columbia, Canada. Seedlings were grown in a large walk-in growth chamber for two weeks and then subjected to factorial combinations of two water regimes (drought and non-drought) and two nitrogen regimes (high and low) for sixteen weeks (28 drought cycles). Ecophysiological responses to the treatments and their interactions were examined. It was found that white spruce had significantly higher rates of photosynthesis, mesophyll conductance, stomatal conductance and photosynthetic nitrogen-use efficiency than did black spruce, but the two species had similar photosynthetic water-use efficiency. White spruce was better able to improve photosynthetic performance under improved water and nitrogen regimes than black spruce. The day after rewatering at the end of the last drought cycle, the effects of drought on most ecophysiological variables were partially reversed in both species, but they were more reversible in black spruce than in white spruce. The high nitrogen treatment increased the stomatal sensitivity to drought but decreased the sensitivity of the mesophyll. Drought stress increased the sensitivity of ecophysiological variables to the nitrogen treatment.

INTRODUCTION

Water and nitrogen are two common limiting factors for terrestrial plants. The effects of water and nitrogen regimes on the growth, development and physiology of plants have been studied extensively. While a trade-off between the efficiencies of water- and nitrogen-utilization has long been recognized (e.g., Field *et al.* 1983), relatively few studies have addressed the interacting effects of water and nitrogen on plant physiological processes (e.g., Ghashagaie and Saugier 1989, Walters and Reich 1989, Nátr 1992, Green and Mitchell 1992, Stark 1992). Of those that did, plants were first grown under different nitrogen regimes and then exposed to drought (e.g., Reich *et al.* 1989, Walters and Reich 1989, Green and Mitchell 1992), with only few exceptions (e.g., Squire *et al.* 1987, Stark 1992). In the natural environment, however, water and nitrogen interact continuously. They not only affect plant physiological processes directly (Hirose 1984, Hirose and Werger 1987, Seemann *et al.* 1987, Walters and Reich 1987, Seiler and Cazell 1990, Dang *et al.* 1991 and 1997) but also indirectly through their impact on the structure and development of plant tissues (Radin and Ackerson 1981, Hirose 1984, Gebauer *et al.* 1987, Sage *et al.* 1987, MacAdam *et al.* 1989, Walters and Reich 1989, Nátr 1992, Nunes *et al.* 1993, Penuelas *et al.* 1993, Evans and von Caemmerer 1996). The effects of drought and nitrogen are similar in some ecophysiological traits (e.g., photosynthesis and growth) but are opposite in others (e.g., water-use efficiency). We hypothesize that similar effects will be additive but opposite effects will cancel each other when drought and nitrogen stress are imposed simultaneously.

Soil water and nitrogen conditions in the field vary seasonally and fluctuate over shorter periods of time. For example, droughts are interspersed with temporary relief by rainfall. The conditions are more dynamic on hummocks in wetland sites than on upland sites. Although boreal wetlands are characterized by high ground water tables, tree roots generally grow on hummocks above the seasonal average ground water table (Lieffers and Rothwell 1986). Hummocks consist of poorly decomposed and/or undecomposed organic matter with low water holding capacity and low capillary surface for lifting ground water (Paivanen 1984, Mannerkoski 1985). The moisture conditions of the hummock substrate, therefore, fluctuate with precipitation and changes in the ground water table and thus are less stable. A

* This experiment was conducted at the University of British Columbia

[1] *Faculty of Forestry, Lakehead University, 955 Oliver Road, Thunder Bay, Ontario, Canada P7B 5E1*
[2] *Botany Department, University of Wisconsin, Madison 53706, USA*
[3] *Faculty of Forestry, University of British Columbia, Vancouver, British Columbia, Canada V6T 1Z4*

corresponding dynamic response in ecophysiological processes in response to fluctuations in water regime would not only allow the plant to survive the stressful conditions (e.g., flooding and drought) but also allow them to take advantage of the transiently favorable conditions.

Black spruce (*Picea mariana* (Mill.) B.S.P.) and white spruce (*Picea glauca* (Moench) Voss) both grow in the boreal forest region, yet they generally dominate on sites with distinctly different water and nutrient regimes. Black spruce often forms pure stands or is interspersed with tamarack (*Larix laricina*) on water-logged, nutrient deficient (particularly nitrogen) wetland sites. White spruce, on the other hand, mainly occupies well-drained upland sites (Hosie 1979). Although the two species have similar shade tolerance and their stands often grow adjacent to each other, they rarely form mixed stands (Vincent 1965, DeLong *et al.* 1991). Being congeneric and having similar growth form, significant differences in phenology and morphology are unlikely (Dudley 1996). Ecophysiological adaptation, therefore, is likely a major mechanism for the different ecological distributions of the two species. We hypothesize (1) that white spruce is physiologically more competitive than black spruce under favorable soil water and nitrogen conditions and (2) that black spruce is more responsive to changes in water regimes and more resistant to nitrogen stress than white spruce.

In an earlier paper (Patterson *et al.* 1997), we reported the long-term, integrated effects of drought and nitrogen on the growth and whole-plant nitrogen- and water-relations of wetland black spruce and upland white spruce. In this report, we discuss the interacting effects of drought and nitrogen on gas exchange and the reversibility of drought effects under different nitrogen regimes in the same two species.

MATERIALS AND METHODS

Materials and Treatments

Seeds of black spruce (*Picea mariana* (Mill.) B.S.P.) and white spruce (*Picea glauca* (Moench) Voss) were obtained from the southern edge of the boreal forest near Fort Nelson (59^0 latitude, 670 m elevation), British Columbia, Canada. The two species are dominant conifers in the region (DeLong *et al.* 1991). The landscape of the region is composed of a mosaic of level lowlands (peatlands) and sloping uplands (Valentine 1971, DeLong *et al.* 1991). Black spruce dominates peatlands while white spruce mainly grows on uplands, but they do not form mixed stands. The monthly average temperature remains below 0 ^{0}C for 5-7 months of the year, and above 10 ^{0}C for 2-4 months. The annual rainfall is between 200 and 290 mm and total precipitation about 446 mm (DeLong *et al.* 1991).

Five pairs of adjacent white spruce and black spruce populations were selected. Each pair was approximately 10 km away from others. Black spruce seeds were collected from 35 randomly selected trees from each population. White spruce seeds were obtained from the B.C. Ministry of Forests Tree Seed Centre.

Seeds from different trees of the same population were mixed and randomly sampled for planting in Ray Leach Single Cell containers (Stuewe and Sons, Corvallis, Oregon, U.S.A.). White spruce seeds were soaked for 12 hours and then stratified at 2 ^{0}C for three weeks before planting. Seedlings were grown under the following environmental conditions: 20-hour photoperiod, 35% day time RH, 24/14 ^{0}C day/night temperature, and 400 μmol m^{-2} s^{-1} PAR flux density.

The seedlings were watered four times and fertilized twice per week for the first two weeks after germination. Starting from the third week, the seedlings were exposed to factorial combinations of two moisture (droughted and non-droughted) and two nitrogen regimes (low and high). Each treatment had 10 replicates. The seedlings of both species were about 2 cm tall when the treatments were imposed.

The non-droughted seedlings were watered daily. The seedlings were weighed before and after watering. The amount of water added to each container was estimated by the amount of water loss over the last 24-hour period. The droughted seedlings were not watered until the soil water potential in ten randomly selected containers of each species dropped down about -1.0 MPa, at which time seedlings in all treatments were watered to drip point and fertilized. The treatments lasted for sixteen weeks (28 drought cycles). The average predawn water potential (Ψ) at the end of the last drought cycle was -0.6 MPa in the non-droughted seedlings and -1.1 MPa in the droughted ones. There was no significant difference in Ψ between the two species or

between the two nitrogen treatments (Patterson *et al.* 1997). More details about the experiment are provided in Patterson *et al.* (1997).

Seedlings in all treatments received identical concentrations of phosphorus (625 mg L^{-1} P), potassium (94 mg $^{L-1}$ K), sulfur (10 mg L^{-1} S), and STEM micro-nutrients (1 mg L^{-1}). The nitrogen concentrations were 25 and 225 mg L^{-1} for the low and high nitrogen treatments, respectively.

Gas Exchange Measurement and Determination of Nitrogen Content

Foliar CO_2 and H_2O gas exchange was measured using an ADC LCA-3 open system and PLC C leaf cuvette (Hoddesdon, England) at 600 µmol m^{-2} s^{-1} PAR flux density, 23-24 ^{0}C temperature, 345 ± 15 µmol mol^{-1} CO_2, and 50 ± 10 % RH. Droughted seedlings were measured twice: the first measurement was taken at the end of the last drought cycle (referred to as the end of the drought cycle hereafter), then the seedlings were watered to saturation, and the same foliage was measured again on the next day (referred to as the beginning of the drought cycle hereafter).

The projected leaf area was determined on a leaf area meter (Li-3000, Li-Cor, Inc., Lincoln, Nebraska). Leaf nitrogen content was determined using the micro-Kjeldahl method and expressed on a leaf area basis (N_a) and a leaf dry mass basis (N_m).

Net photosynthetic rate (A), transpiration rate (E), stomatal conductance to H_2O (g_{sw}) and intercellular CO_2 concentration (C_i) were calculated according to von Caemmerer and Farquhar (1981) and expressed on a leaf area basis. Stomatal conductance to CO_2 was calculated as $g_{sc} = g_{sw} / 1.6$ (Coombs *et al.* 1987). Photosynthetic water-use efficiency (WUE) was calculated as the molar ratio of A/E. Mesophyll conductance (g_m) was determined as A/C_i (Fites and Teskey 1988). The photosynthetic nitrogen-use efficiency (NUE) was calculated as A/N_a.

Data Analysis and Presentation

The effect of species, population, drought, nitrogen, drought cycle (i.e., the beginning vs. the end of the drought cycle), and their interactions were tested using ANOVA. Populations were considered as nested within species and had a random effect. Each ecophysiological variable was tested by three separate ANOVAs: (1) Data from droughted seedlings at the end of the drought cycle were tested against data from non-droughted seedlings to examine all main effects and interactions except the effect of drought cycle; (2) Data from droughted seedlings at the end of the drought cycle were tested against data from the same seedlings but collected the next day after they were watered, to examine the significance of recovery in ecophysiological variables; (3) Data from droughted seedlings measured the next day after watering at the end of the last drought cycle were tested against data from the non-droughted ones, to determine the completeness of the recovery and interactions associated with rewatering. Since Analysis 1 showed that population did not have a significant effect on any of the variables examined, population was not considered in subsequent analyses.

Treatment effects are quantified as follows: The net effect of drought (%) was calculated within each nitrogen treatment (i.e., high or low nitrogen) as:

$$[(W^- - W^+) / W^+] \times 100 \qquad [1]$$

where W^- and W^+ are droughted and non-droughted, respectively. The net effect of nitrogen (%) was determined within each drought treatment in the same fashion,

$$[(N^- - N^+) / N^+] \times 100 \qquad [2]$$

where N^- and N^+ are low and high nitrogen treatment, respectively.

The effect so calculated reflects the net impact of a treatment without the interference of other main effects and is comparable between treatments, species, and between different variables. For example, a value of -50 indicates a 50% reduction (as compared to control). The percentage reversal of drought effect the next day after rewatering was calculated as:

$$[(W^{-b} - W^{-e}) / (W^+ - W^{-e})] \times 100 \qquad [3]$$

where W^+, W^b and W^e indicate non-droughted control, and droughted at the beginning and the end of the drought cycle, respectively.

RESULTS

Species Effect

Rate of net photosynthesis (A), mesophyll conductance (g_m), stomatal conductance (g_{sc}), photosynthetic nitrogen-use efficiency (NUE) and nitrogen content per unit leaf area (N_a) were generally greater in white spruce than in black spruce (fig. 1 and fig. 2, table 1). The differences were generally more prominent in non-droughted than in droughted seedlings (fig. 1 and fig. 2). The C_i/C_a ratio and photosynthetic water-use efficiency (WUE) were similar in the two species (figs. 1 and 2, table 1). Nitrogen content per unit foliar dry mass (N_m), on the other hand, was generally greater in black spruce than in white spruce (fig. 2 and table 1). The difference in N_m between the two species was generally greater in the high nitrogen than in the low nitrogen treatment and greater in droughted than in non-droughted seedlings (fig. 2, W×S and N×S effects in table 1).

Drought Effect and Its Reversibility

Drought significantly reduced A, g_m, g_{sc}, C_i/C_a and NUE, but increased WUE, N_a and N_m, in both white spruce and black spruce (figs. 1, 2 and 3, tables 1 and 2). The effect ranged from -86 % (g_{sc}) to +122 % (WUE) (fig. 3). The effect on A and g_{sc} was significantly greater and the effect on N_a and N_m was significantly smaller in white spruce than in black spruce, but there were no significant species-drought interactions in other variables (fig. 3, table 1 and table 2). Drought effect on A, g_m and NUE were significantly greater, while the effect on g_{sc}, C_i/C_a and WUE were significantly smaller in the low nitrogen treatment than in the high nitrogen treatment (fig. 3, table 1). The effect of drought on g_m was almost zero in the high nitrogen treatment (fig. 3).

There was a significant three-way interaction between drought, nitrogen and species (table 1): N_a was more sensitive to drought in the high nitrogen treatment than in low nitrogen treatment in white spruce, but the trend was the opposite in black spruce. In contrast N_m was more sensitive in the low nitrogen treatment than in the high nitrogen treatment in white spruce, and again the trend for black spruce was the opposite (table 2).

Drought effects on A, g_{sc}, C_i/C_a, WUE and NUE were reversed by 54%, 52%, 34%, 93%,100% and 34%, respectively, the next day after rewatering (fig. 4, C effect in table 3 and W effect in table 4). The drought effect on A was significantly more reversible in black spruce than in white spruce (C × S effect in table 3). Drought effect on other variables was also more reversible in black spruce than in white spruce (but not significantly so, $P > 0.06$, fig. 3, C × S in table 3). Because there was no significant drought effect on g_m in the high nitrogen treatment, the significant reversal of the drought effect on g_m in the low nitrogen treatment was reflected as a significant nitrogen by drought-cycle interaction (fig. 3, N × C in table 3). Drought effect on g_m in the low nitrogen treatment was reversed by 42 % in response to watering. Drought effects on A, WUE and NUE were more reversible and the effect on g_{sc} was less reversible in the high nitrogen treatment than in the low nitrogen treatment, but the difference was not significant for A and g_{sc} (fig. 3, N × C in table 3). In fact, WUE in the droughted, high nitrogen treatment was significantly smaller than in the corresponding control the next day after rewatering (fig. 3). The significant interactions between species and drought (in A and g_{sc}) at the end of the drought cycle remained significant after rewatering, but the significant interactions between drought and nitrogen treatment (in g_{sc} and WUE) became statistically insignificant after rewatering (W × S and N × W in table 4).

Nitrogen Effect

A, g_m, WUE, N_a and N_m were significantly smaller, and NUE and C_i/C_a were significantly greater in the low nitrogen than in the high nitrogen treatment (fig. 1, fig. 2 and fig. 5, table 1 and table 2). The net effects ranged from -51 % (g_m) to +107 % (NUE). There were also significant interactions between nitrogen, species, drought and drought cycle (table 1 and table 3). g_{sc} was significantly greater in the low nitrogen than in the high nitrogen treatment for the droughted seedlings (fig. 2 and fig. 5). The nitrogen effects on A, g_m, g_{sc}, C_i/C_a and WUE were significantly greater and the effect on NUE was significantly smaller, in droughted than in non-droughted seedlings (fig. 2 and fig. 5). In fact, the nitrogen effects on A, g_m, g_{sc} and C_i/C_a were negligible in the non-droughted seedlings (fig. 5). Nitrogen effect on N_m was greater in non-droughted

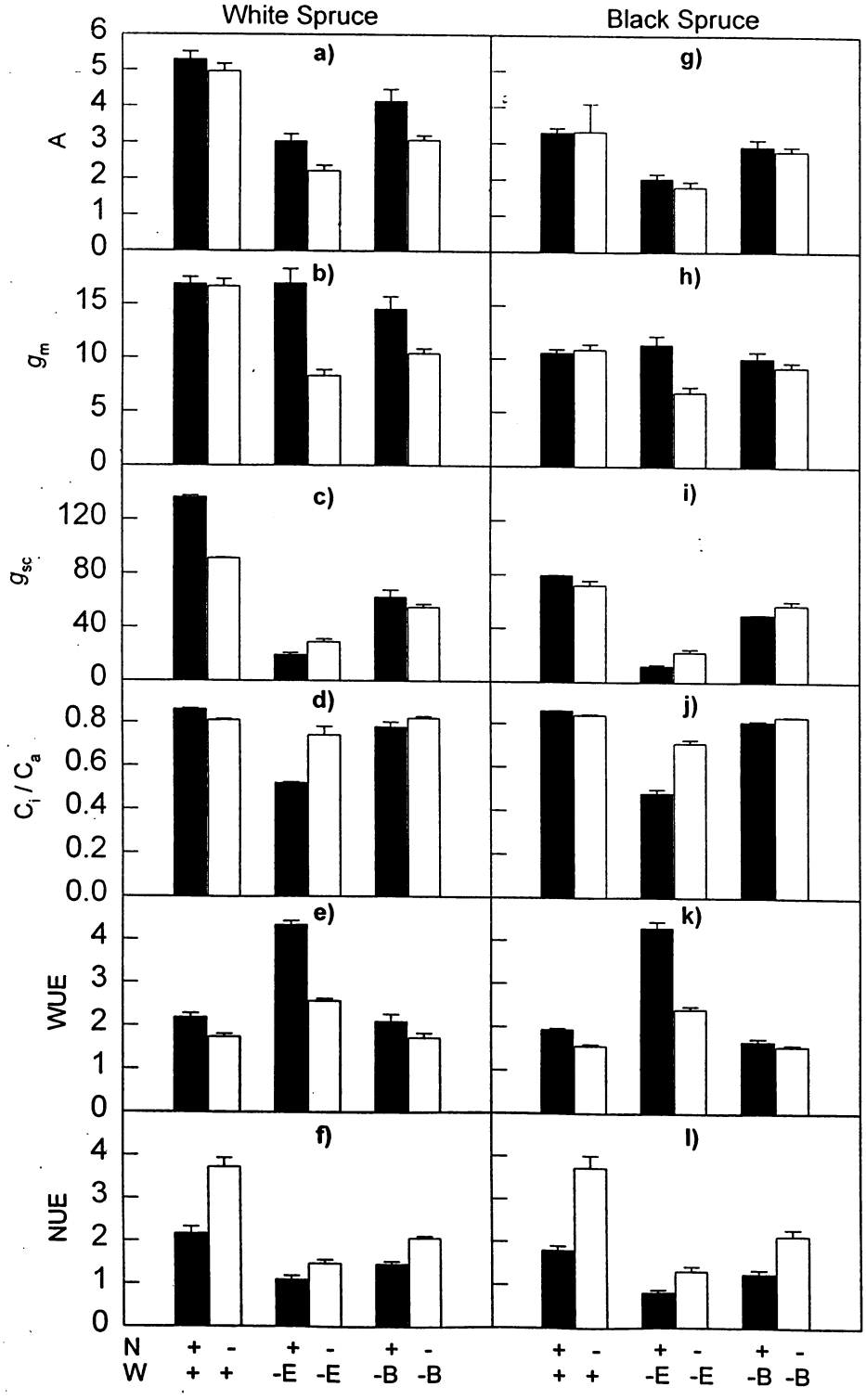

Figure 1.--Net photosynthesis (A), mesophyll conductance (g_m), stomatal conductance (g_{sc}), C_i/C_a ratio, photosynthetic water-use efficiency (WUE) and nitrogen-use efficiency (NUE) (± s.e., n = 25) in black and white spruce grown under different water (W) and nitrogen (N) regimes. N: '-' indicates low nitrogen treatment and '+' high nitrogen treatment. W: '+' indicates non-droughted; '-E' and '-B' indicate droughted at the end of the last drought cycle and the next day after rewatering, respectively. The units are: μmol CO_2 m^{-2} s^{-1} for A; mmol CO_2 m^{-2} s^{-1} for g_m and g_{sc}; μmol CO_2 $mmol^{-1}$ H_2O for WUE; μmol CO_2 s^{-1} g^{-1} N for NUE.

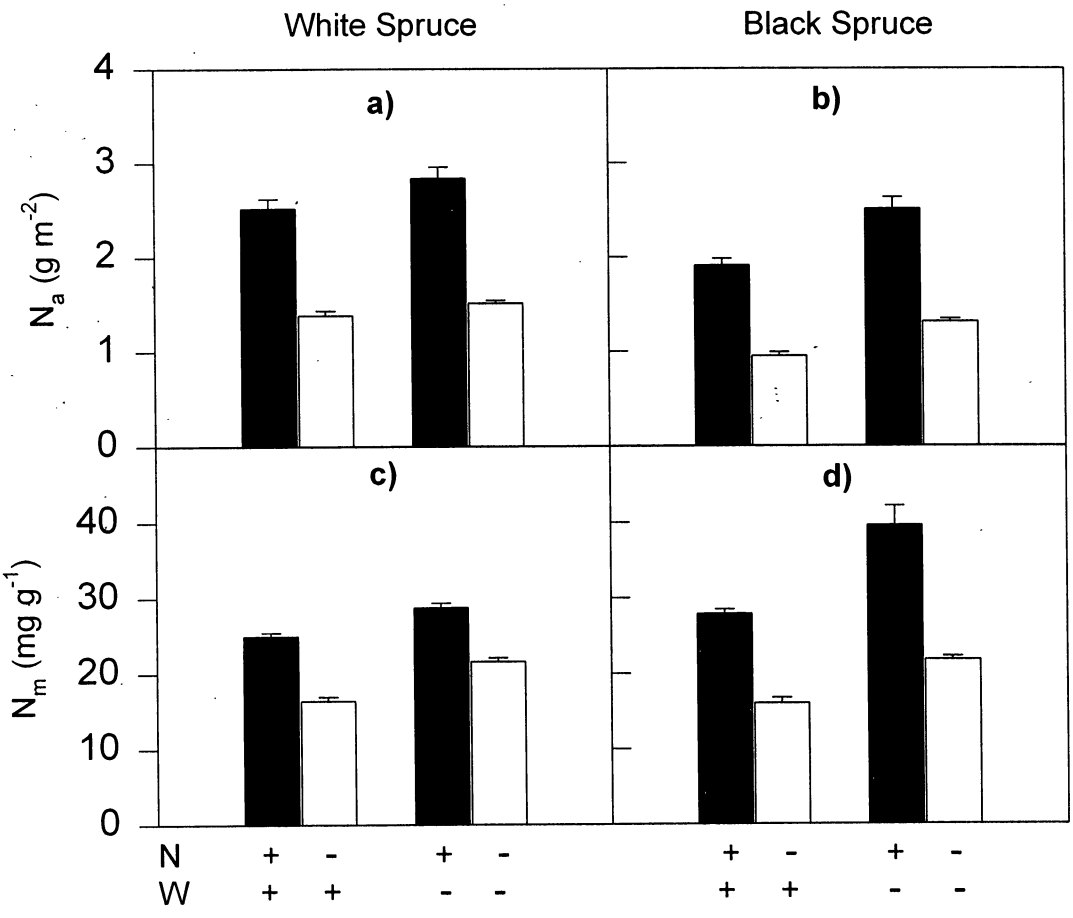

Figure 2.--Foliar nitrogen content per unit leaf area (N_a) and per unit dry mass (N_m) in black spruce and white spruce grown under different water and nitrogen regimes (see Figure 1 for details).

Table 1.--*Analysis of variance on the effects of species (S), population (P), drought (D), nitrogen stress (N), and their interactions on net photosynthesis (A), mesophyll conductance (g_m), stomatal conductance (g_{sc}), photosynthetic water-use efficiency (WUE), nitrogen-use efficiency (NUE), C_i/C_a ratio, foliar nitrogen content per unit leaf area (N_a), and per unit foliar dry mass (N_m) in white and black spruce. Data used in this test were collected from droughted seedlings at the end of the drought cycle and from non-droughted seedlings. Values in the table are probabilities for greater F values.*

Effect	**Ecophysiological Variable**							
	A	g_m	g_{sc}	WUE	NUE	C_i/C_a	N_a	N_m
S	**0.0001**	**0.0006**	**0.0009**	0.2469	**0.0329**	0.3162	**0.0011**	**0.0013**
P(S)	0.7972	0.5107	0.7605	0.7000	0.8464	0.6297	0.6111	0.6872
D	**0.0001**	**0.0181**	**0.0001**	**0.0001**	**0.0001**	**0.0001**	**0.0002**	**0.0001**
D × S	**0.0164**	0.1490	**0.0156**	0.5354	0.9813	0.1226	**0.0500**	**0.0333**
N	**0.0455**	**0.0002**	0.1087	**0.0001**	**0.0001**	**0.0001**	**0.0001**	**0.0001**
N × S	0.1473	0.1022	0.0805	0.9519	**0.0405**	0.4538	0.1780	**0.0001**
N × D	**0.0217**	**0.0020**	**0.0027**	**0.0003**	**0.0018**	**0.0001**	0.1700	**0.0280**
N × D × S	0.6949	0.3859	0.0521	0.7000	0.6868	0.4042	0.0550	**0.0019**

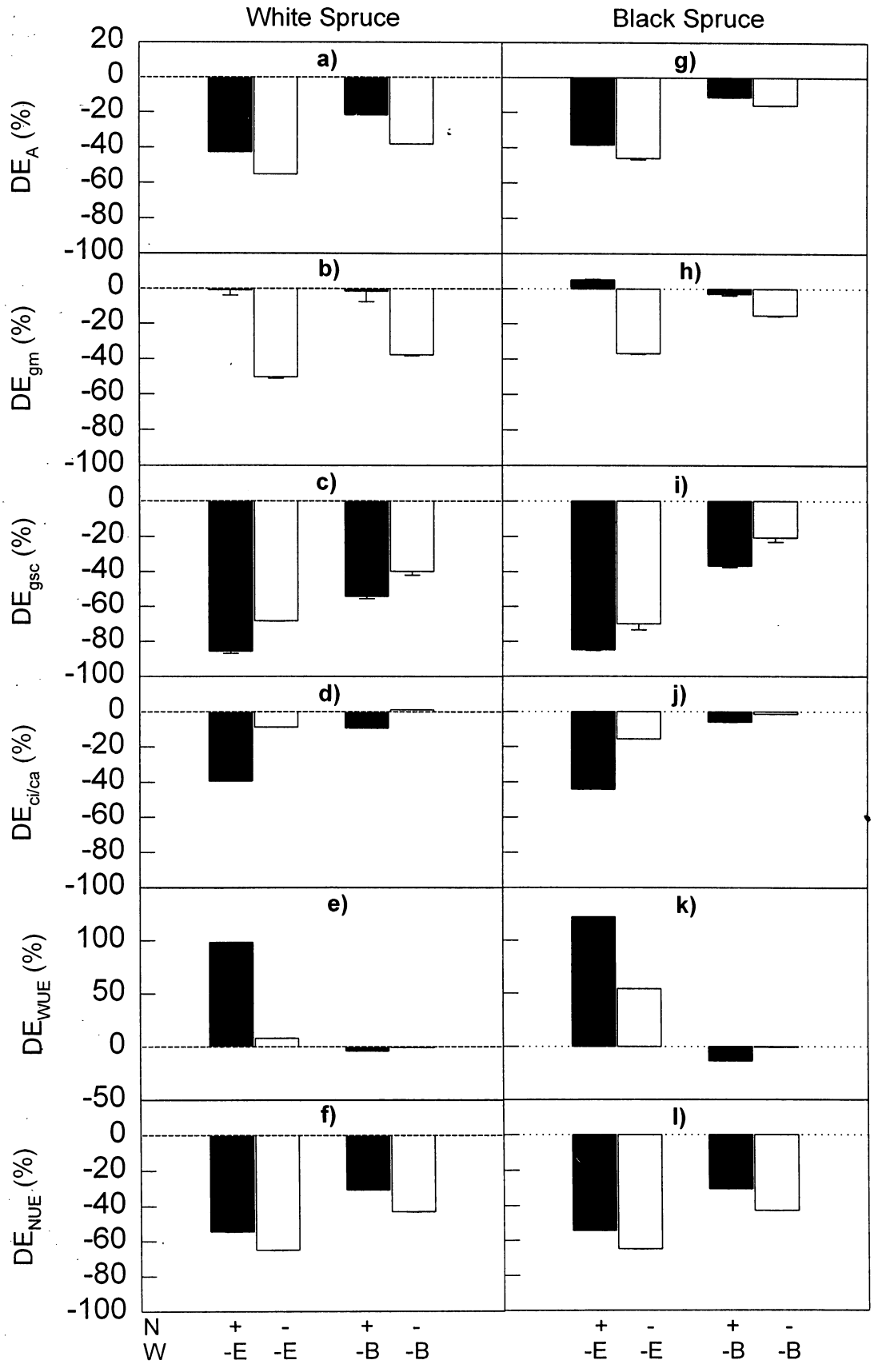

Figure 3.--Drought effect (DE) on A, g_m, g_{sc}, C_i/C_a ratio, WUE and NUE in black spruce and white spruce. Values in the graph are the net effect as calculated by Equation 1 (see text for details). Other explanations are as in Figure 1.

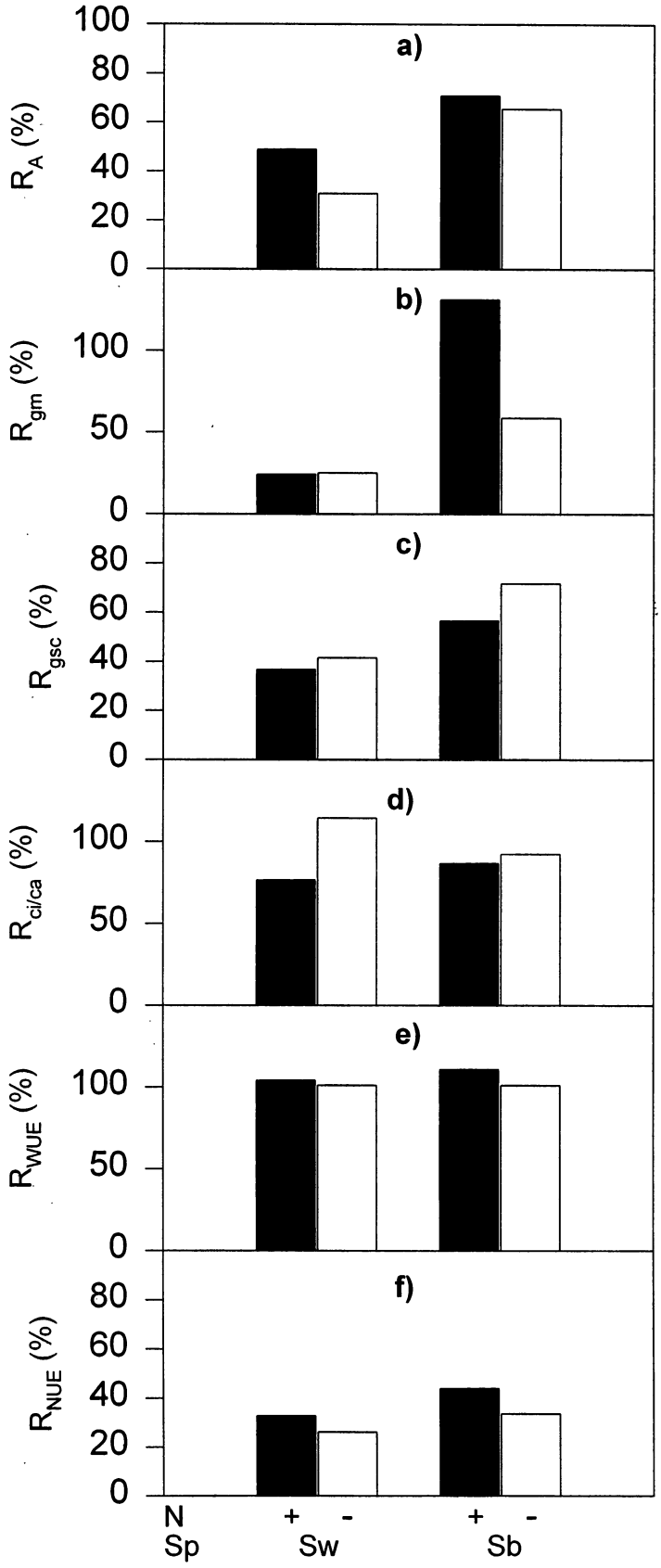

Figure 4.--Percentage of drought effect that was reversed upon re-watering (R) in black spruce (Sb) and white spruce (Sw) grown under low (-N) and high (+N) nitrogen regimes. Values are calculated by Equation 3 (see text for details). Other explanations are as in Figure 1.

Table 2.--*Effects of drought and nitrogen on leaf nitrogen content per unit area (N_a) and per unit dry mass (N_m) in white spruce and black spruce. Values in the table are net treatment effects, expressed as a percent change, as calculated by Equation 2 (see text for details).*

Variable	Species	Drought Effect (% change)		Nitrogen Effect (% change)	
		High Nitrogen	Low Nitrogen	Non-Droughted	Droughted
N_a	White Spruce	12.7	9.4	-45.2	-46.8
	Black Spruce	31.4	37.9	-50.3	-47.8
N_m	White Spruce	15.2	31.7	-34.4	-25.0
	Black Spruce	41.9	36.3	-42.7	-44.9

Table 3.--*Analysis of variance on the effect of drought cycle (C, end of drought cycle vs. next day after rewatering) and its interactions with species (S) and nitrogen (N) on A, g_m, g_{sc}, WUE, NUE and C_i/C_a ratio in white spruce and black spruce. Data used in this test were from droughted seedlings only, but collected at the end of the last drought cycle and the next day after rewatering. Values in the table are probabilities for greater F values.*

Effect	Ecophysiological Variable					
	A	g_m	g_{sc}	WUE	NUE	C_i/C_a
C	**0.0001**	0.9983	**0.0001**	**0.0001**	**0.0001**	**0.0001**
C × S	**0.0090**	0.5242	0.5867	0.3479	0.0880	0.0654
N × C	0.7593	**0.0001**	**0.0478**	**0.0001**	**0.0167**	**0.0001**
N × C × S	0.6482	0.6644	0.2826	0.4032	0.6628	0.8567

Table 4.--*Analysis of variance on the effects of drought (D) the next day after rewatering as compared to the control and its interactions with species (S) and nitrogen treatment (N) on A, g_m, g_{sc}, WUE, NUE and C_i/C_a ratio in white and black spruce. Values in the table are probabilities for greater F values.*

Effect	Ecophysiological Variable					
	A	g_m	g_{sc}	WUE	NUE	C_i/C_a
D	**0.0025**	**0.0070**	**0.0004**	0.0797	**0.0001**	**0.0001**
D × S	**0.0411**	0.0557	**0.0350**	0.4562	0.6051	0.1042
N × D	**0.0360**	**0.0109**	0.0699	0.2787	**0.0139**	**0.0003**
N × D × S	0.6490	0.4308	0.3096	0.6141	0.8225	0.1060

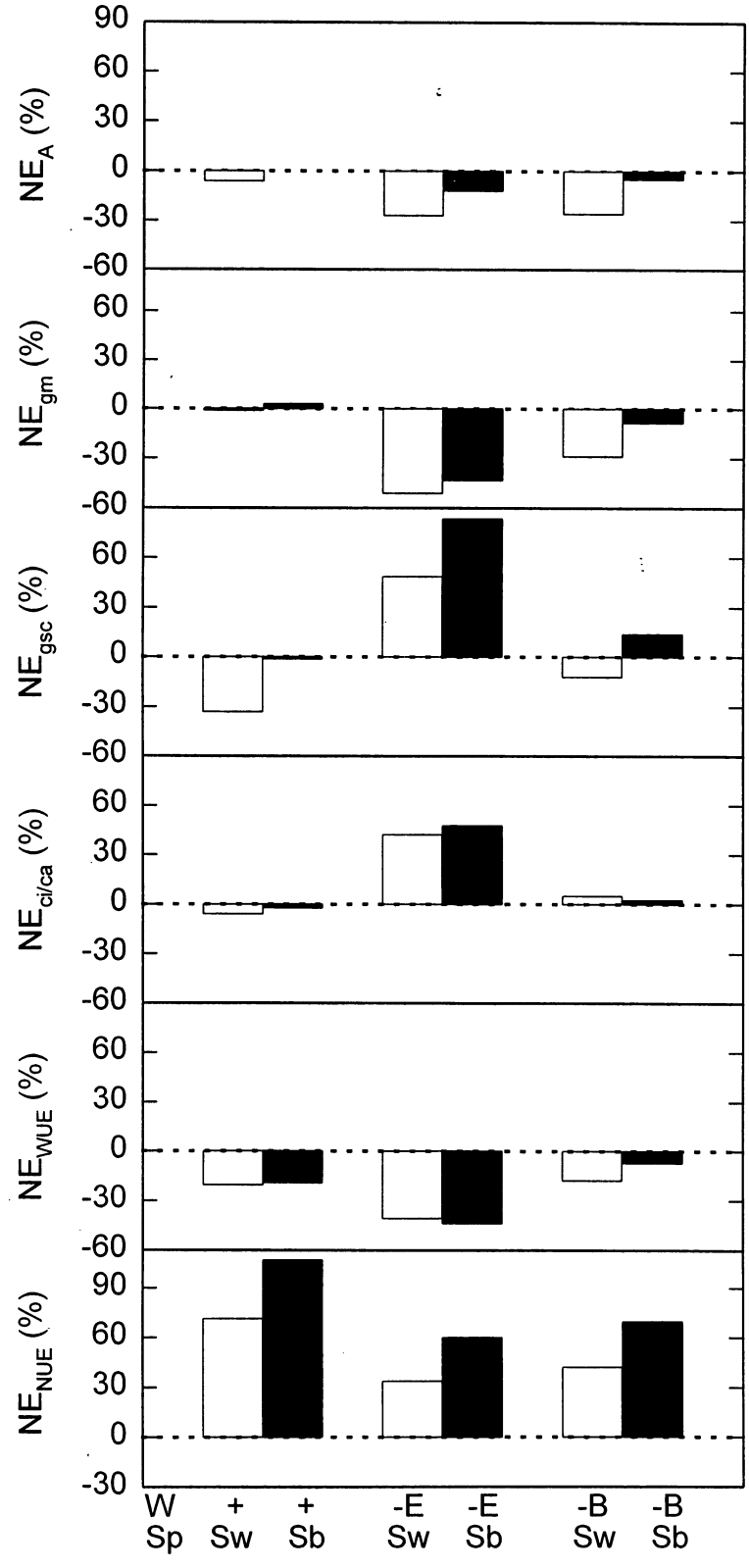

Figure 5.--Nitrogen effect (NE) on A, g_m, g_{sc}, C_i/C_a ratio, WUE and NUE in black spruce (Sb) and white spruce (Sw) grown under drought and non-drought conditions. Values in the graph are the net nitrogen effect as calculated by Equation 2 (see text for details). Other details as in Figure 1.

than droughted white spruce, but the trend was the opposite for black spruce (fig. 2, table 1 and table 2). The nitrogen effects on NUE and N_m were significantly greater in black spruce than in white spruce (fig. 5).

For droughted seedlings, nitrogen effects on g_m, g_{sc}, C_i/C_a and WUE were significantly smaller and the effect on NUE was significantly greater at the beginning than at the end of the drought cycle (fig. 5, N×C effect in table 3).

Discussion

The high nitrogen treatment significantly increased the sensitivity of stomatal conductance to drought and decreased the sensitivity of mesophyll conductance, but decreased stomatal conductance was the primary limiting factor to photosynthesis in both high and low nitrogen treated seedlings of both species (fig. 3). Stomatal conductance is also found to be more sensitive to drought in high nitrogen leaves than low high nitrogen leaves in several other species (Shimshi 1970, Morgan 1984 and 1986, Walters and Reich 1989). Ogren (1988) found that nitrogen status of willow leaves affected the sensitivity of photosynthesis to long-term but not short-term drought.

Decreases in root/shoot ratio associated with increasing nitrogen supply (Shimshi 1970, Morgan 1984 and 1986, Walters and Reich 1989, Penuelas *et al.* 1993, Patterson *et al.* 1997) may be an explanation for the increase in stomatal sensitivity to drought in high nitrogen plants. High nitrogen can also decrease the threshold water potential for an increase in endogenous ABA content and thus increase stomatal sensitivity to drought (Radin 1981). Hunt *et al.* (1985) also found that the stomatal conductance of high-nitrogen leaves were more sensitive to leaf-to-air vapor pressure difference. These changes can improve the stomatal control on transpirational water loss and thus water relations (Brix and Mitchell 1986, Hillerdal-Hagstromer *et al.* 1982).

The decreased mesophyll sensitivity to drought with increasing nitrogen was probably more related to increases in nitrogen allocation to photosynthetic apparatus (Sheriff *et al.* 1986, Evans 1989) and increases in mesophyll cell area per unit of leaf area (Nobel *et al.* 1975, Rovenska and Nátr 1981). Evans and von Caemmerer (1996) found that increased mesophyll surface decreased mesophyll resistance to CO2 diffusion. Increased nitrogen supply increases chlorophyll content, and the number and size of chloroplasts per unit leaf area (Tesarová and Nátr 1986), the content of soluble proteins and Rubisco activity (Evans and Terashima 1978 and 1988, Seemann *et al.* 1987). Additionally, osmotic adjustment associated with nitrogen supply (Tan and Hogan 1995) may also alter the cell environment for photosynthesis.

The result that stomatal conductance was the primary limiting factor to photosynthesis in seedlings subjected to drought stress seems to contradict recent findings that drought decreases photosynthesis primarily by depressing mesophyll conductance (e.g., Osonubi and Davies 1980, Farquhar and Sharkey 1982, Melzack *et al.* 1985, Teskey *et al.* 1986, Dang *et al.* 1991). Since the sensitivity of mesophyll to drought increased and the sensitivity of stomatal conductance decreased with decreasing nitrogen supply, it is possible that the plants used in the above studies were subjected to even lower nitrogen supply than the plants in the low nitrogen treatment in this study. Indeed, the trees used by Dang *et al.* (1991) were from a nitrogen deficient peatland. MacDonald and Lieffers (1990) have shown that the foliage nitrogen concentration of black spruce from the study site of Dang *et al.* (1991) was 8 mg N/g foliage, only about 36% of the nitrogen concentration of the low-nitrogen black spruce (22 mg N/g foliage) in this study. Our results support Hirose's Nitrogen/Carbon Limitation Model (Hirose 1984) which suggests that mesophyll limitation to photosynthesis increases with decreasing nitrogen concentration.

Water regimes affected the response of both species to nitrogen treatment. Nitrogen treatment had little effect on photosynthesis and mesophyll conductance in well-watered seedlings of both species, particularly in black spruce, but the high nitrogen treatment significantly increased photosynthesis and mesophyll conductance in droughted seedlings (fig. 1 and fig. 5). This suggests that the low nitrogen treated plants received sufficient nitrogen to support a high rate of photosynthesis. The extra nitrogen in the high nitrogen treatment was probably put in storage (luxury consumption) and thus resulted in much lower photosynthetic nitrogen use efficiency (fig. 1 and fig. 5). However, to increase the drought resistance of photosynthetic machinery required addition nitrogen resource. Hence, the high nitrogen treatment increased photosynthesis and mesophyll conductance of droughted seedlings of both species (fig. 1 and fig. 5). This is in contrast to the finding by Walters and

Reich (1989) that well-watered *Ulmus americana* was more sensitive to nitrogen treatment than water-stressed ones. The conflicting results suggest that we probably examined a different portion of the photosynthesis-nitrogen curve than Walters and Reich (1989) did. Over the whole range of leaf nitrogen concentration, photosynthesis is curvilinearly correlated with nitrogen (Hirose 1984, Hirose and Werger 1987, Evans 1989). In the portion of curve where photosynthesis is limited by nitrogen which is presumed to be the case in the study of Walters and Reich (1989), well-watered plants should be better able to use nitrogen for photosynthesis than droughted ones because the nitrogen used to increase drought resistance in droughted plants can be used to increase the photosynthetic capacity in well-watered ones. Water stress is found to affect the photosynthesis-nitrogen relationship in several other species (e.g., Farquhar and von Caemmerer 1984, Wolfe *et al.* 1988, Gollan *et al.* 1986).

White spruce was more responsive to the nitrogen treatments than black spruce. Although the high nitrogen treatment increased mesophyll conductance and photosynthesis in both species, the increase in white spruce was almost twice as high as that in black spruce (fig. 1 and fig. 5). This probably reflects the inability of black spruce utilize the improved nitrogen supply in the high nitrogen treatment. There was little difference in the rate of photosynthesis of black spruce between high and low nitrogen treatment (figures 1 and 5). White spruce, on the other hand, increased photosynthesis in response to the high nitrogen treatment. A similar phenomenon was found at the whole plant level (Patterson *et al.* 1997). This was related to the natural habitat to which each species was adapted. Black spruce seeds were collected from peatland sites where nitrogen supply was very poor (Payandeh 1973, Makitalo 1985, Mannerkoski 1985). White spruce seeds were from upland sites where the nitrogen supply was comparatively better. The inability of black spruce to use high nitrogen supply may reflect its adaptation to low nitrogen habitats and this adaptation probably puts this species at a competition disadvantage in occupying nitrogen-rich upland sites. Van der Werf *et al.* (1993) observed that species from high productivity sites had significantly higher A than species from low productivity sites under high N treatment, but the both groups had similar A under low soil N conditions.

White spruce was also more responsive to water regimes. Although photosynthesis in well-watered seedlings was significantly higher than in the droughted seedlings in both species, the difference was much more prominent in white spruce than in black spruce (fig. 1 and fig. 3). This reflects that white spruce was better able to improve photosynthetic performance under improved soil water conditions than black spruce. This may give white spruce another competitive advantage over black spruce in occupying sites with good soil moisture conditions.

Black spruce appeared to be better adapted to dynamic soil water regimes than white spruce. The two species were subjected to similar levels of drought stress, the same number of drought cycles, and identical aboveground environmental conditions. However, the drought effects on black spruce were more reversible than those on white spruce the second day after rewatering at the end of the last drought cycle. For example, the drought effect on photosynthesis was almost twice as reversible in black spruce as in white spruce (fig. 4). This may be related to the different water regimes of their natural sites. Black spruce primarily grows on peatlands in the boreal forest region (DeLong *et al.* 1991). While peatlands are characterised by high ground water table, trees generally grow on hummocks which are above the average surface of the peatland (Lieffers and Rothwell 1986). The substrate on the hummocks consists of poorly decomposed or/and undecomposed organic matter and has very poor water holding capacity and low total capillary surface (Paivanen 1984, Mannerkoski 1985). Thus, the water regime of the hummocks fluctuates with precipitation and ground water table (Dai *et al.* 1972, Mannerkoski 1985) and the trees growing on them experience periodic flooding and drought (Boggie and Miller 1976, Mannerkoski 1985, Dang *et al.* 1991). A faster reversal of drought effects plus the greater capacity of anaerobic respiration of the root system under cold, flooded conditions (Conlin and Lieffers 1993) should be an adaptive advantage for peatland black spruce. White spruce, in contrast, mainly grows in well developed soils that can provide a more stable water condition. In addition, the root system of white spruce is generally better developed than that in black spruce (Lieffers and Rothwell 1986). White spruce, therefore, may experience less fluctuations in soil water conditions at their natural sites than black spruce.

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MODELS FOR SNOW AND WIND DAMAGE ASSESSMENTS IN SWEDISH PINE, SPRUCE, AND BIRCH FORESTS

Erik Valinger[1] and Jonas Fridman[2]

ABSTRACT.--Damage to forests caused by snow and wind accounts for high economic losses annually. In Europe, the scale of the problem amounts to hundreds of millions of US$ each year. In Sweden alone, approximately 4 million m^3 are damaged annually by snow and wind[1], roughly corresponding to a value of 150 million US$. To minimize these high losses, a tool for risk assessment within forest management was developed. Predictions of damage risk from snow and wind using tree, stand, and site characteristics of Scots pine *(Pinus sylvestris* L.), Norway spruce (*Picea abies* (L.) Karst.) and Birch (*Betula spp.* L.) were made using data from permanent sample plots within the Swedish National Forest Inventory (NFI). The plots were inventoried twice at five year intervals between 1983 and 1992. Measurements at first inventory of tree characteristics for the largest sample tree, site, and stand data were combined with snow and wind damage registrations from the second inventory, as input data for model development. For pine and spruce stands, models were developed for three different regions, while one model for the whole country was developed for birch stands. The models developed can be used to detect sites with high probability of damage from snow and wind, and reduce future damage and costs in forestry.

INTRODUCTION

Forest damage caused by snow and wind accounts for approximately 4 million m^3 annually in Sweden (Valinger and Fridman 1995), corresponding roughly to a value of 150 million US$. In Europe, hundreds of millions US$ are lost annually because of snow and wind damage.

Damage to stands leads not only to losses of high value timber, detrimental insect attacks on the remaining stand (Schroeder and Eidmann 1993), and reduced seed production among older trees (Nixon and Cameron 1994), it also necessitates unscheduled and costly thinnings.

To reduce economic losses, effective methods to minimise damage caused by snow and wind are needed. High risk sites must be identified, and silvicultural strategies for such sites must be tailored (Hirvela and Hynynen 1990, Laiho 1997).

Different approaches to identify stand susceptibility and high risk sites are already in use, e.g., the hazard classification systems in UK (Booth 1997, Miller 1985). Meteorological studies have clarified the circumstances under which the combination of snow and wind load is most likely to cause damage to a tree (Solantie 1994). Another approach has been to identify susceptible sites using GIS with a terrain model (Wright and Quine 1993). Other studies have contributed to the knowledge from a biomechanical point of view (Fraser 1964, Mattheck *et al.* 1993, Peltola and Kellomäki 1993, Gardiner 1995).

All the above mentioned studies show that the processes of snow and wind damage in forests are very complex. To develop a risk classification system, information from a wide range of levels is needed;

(i) single tree data, e.g., species, biomechanical resistance, tree form, and root structure,
(ii) stand data, e.g., species composition, stand age, mean height, and stand density, and
(iii) site data, e.g., climatic conditions, geographical location, soil type, moisture condition, slope and aspect.

As stem form is affected by stress factors during both growth (Jacobs 1954) and dormancy (Lundquist and Valenger (1996), single trees can be used as measures of probability of abiotic site risk (Valinger and Fridman 1997). The largest tree on a plot is, most

[1] *Swedish University of Agricultural Sciences, Faculty of Forestry, Department of Silviculture, S-901 83 Umeå, Sweden. E-mail: Erik.Valinger@ssko.slu.se*

[2] *Swedish University of Agricultural Sciences, Faculty of Forestry, Department of Forest Resource Management and Geomatics, S-901 83 Umeå, Sweden.E-mail: Jonas.Fridman@resgeom.slu.se*

likely, the tree that has been most exposed, and therefore the tree best adapted. According to this view, single tree characteristics would be good indicators of susceptibility to snow and wind damage.

The objective of this study was to develop nation-wide risk assessment models for sites dominated by Scots pine (*Pinus sylvestris* L.), Norway spruce (*Picea abies* L. Karst), and Birch (*Betula* spp), using easily obtained tree, stand, and site characteristics as independent variables. The study is part of a European Community research project (AIR 3-CT94-2392).

MATERIALS AND METHODS

Data Used

Data from the Swedish National Forest Inventory (NFI) were used. The NFI is a stratified systematic sampling including both permanent and temporary plots clustered into permanent and temporary tracts (Ranneby *et al.* 1987). Data from the period 1983 to 1992 were used.

All sample trees from the first inventory of permanent plots were remeasured at the second inventory. Type and cause of damage, if present, were recorded for each sample tree. This study was restricted to damage caused by snow or wind, i.e., uprooted, broken, or leaning trees. For description of tree form on each plot, the sample tree of the dominating species (at least 70% of the total basal area) with the largest dbh was used.

Models were developed for stands dominated by Scots pine (*Sp*), Norway spruce (*Ns*) and Birch (*B*). *Sp* and *Ns* models were developed for three different regions; North, Mid, and South Sweden. A model for the whole country was developed for *B*.

Data comprised 6,756 plots, of which 3.2% were damaged. Description of tree form, stand, and site characteristics, was based on data from the first inventory, while observations of damage were based on the second inventory, i.e., the remeasurement.

Statistical Method

To model the probability of damage (p_1) by snow or wind for an arbitrary tree, characteristics for the largest sample tree, the stand, and the site were used as independent variables. The following model for the logit was fitted to the data using logistic regression (Agresti 1984, Anon. 1990);

$$\text{logit}(\hat{p_1}) = \ln(p_1/p_0) = \beta_0 + \beta_1 x_1 + \beta_2 x_2 + ... + \beta_n x_n \quad (1)$$

where p_0 is the probability that an arbitrary tree is not damaged by snow or wind, $x_1, x_2, .. , x_n$ are independent variables, and $\beta_{0\text{-}n}$ are parameters.

The parameter estimates from (1) were used to estimate the probability of damage caused by snow or wind for an arbitrary tree.

In this study, sample plots with different number of sample trees were used, as well as sample trees with different probability of inclusion. To take this into consideration, the events/trials model syntax in the the SAS© PROC Logistic (Monserud 1976) was used.

Results from the logistic analyses were applied to the same data as for model development. Mean values of estimated probabilities for the plots were calculated for squares including at least five plots, and projected on maps.

Evaluation of the Models

To test goodness of fit of the models, a c-table was used (cf. table 2). The observed damage on a plot, *I*, which was set to 1 if at least one of the sample trees on the plot was damaged by wind or snow, and to 0 otherwise, was compared with an estimated status variable, $\hat{I}$. $\hat{I}$ was set to 1 if $\hat{p_1}$ exceeded the cut value *C*, otherwise $\hat{I}$ was set to 0 (Monserud 1976). This method allows comparisons between observed and estimated binomial values.

Results

Independent variables included in the models are tree, stand, and site characteristics (table 1).

Table 1.--*Variables included in the models for the different regions (I-III) and species (Ns = Norway spruce, Sp = Scots pine, B = Birch). Signs in cells (+, -) refers to contribution of the variable to the estimated probability.*

VARIABLE	REGION						
	I (N. Sweden)		II (M. Sweden)		III (S. Sweden)		I-III (Sweden)
	Species		Species		Species		Species
	Ns	Sp	Ns	Sp	Ns	Sp	B
Tree							
Diameter					-		
Tree height			-				
Bole height				-			-
Crown length			+				
Volume on bark						-	
Height/diameter						-	+
Relative crown length						-	
Upper taper	+						
Stand							
Stand age		-				+	
Stems ha^{-1}				-			
Mean diameter	+	+			+		
Basal area					+		+
Mean height				+	+		
Total volume			+		-		
Proportion of spruce		+					
Site quality	-	-					
Moisture							+
Sediment					-		+
Peat	+						
Site							
Altitude		+	+	+	+	+	+
Latitude	-	-	-	-		-	-
Longitude	+	+	+		+	-	+
Inclination	+						

Table 2.--*Frequency (n) and proportion (%) of predicted damage ($\hat{I}$) and observed damage (I) for Scots pine, Norway spruce, and Birch. Probabilities predicted for regions I-III using the same data as for model development.*

Scots pine

		$\hat{I}$ (I)			$\hat{I}$ (II)			$\hat{I}$ (III)		
I		**0**	**1**	**Σ**	**0**	**1**	**Σ**	**0**	**1**	**Σ**
0	n	710	353	1063	430	268	698	669	430	1099
	%	66.8	33.2	95.4	61.6	38.4	93.7	60.9	39.1	92.6
1	n	21	30	51	20	27	47	36	52	88
	%	41.2	58.8	4.6	42.6	57.4	6.3	40.9	59.1	7.4
Σ	n	731	383	1114	450	295	745	705	482	1187
	%	65.6	34.4	100	60.4	39.6	100	59.4	40.6	100

Norway spruce

		$\hat{I}$ (I)			$\hat{I}$ (II)			$\hat{I}$ (III)		
I		**0**	**1**	**Σ**	**0**	**1**	**Σ**	**0**	**1**	**Σ**
0	n	516	300	816	403	107	510	869	688	1557
	%	63.2	36.8	88.2	79.0	21.0	88.1	55.8	44.2	91.2
1	n	39	70	109	26	43	69	62	89	151
	%	35.8	64.2	11.8	37.7	62.3	11.9	41.1	58.9	8.8
Σ	n	555	370	925	429	150	579	931	777	1708
	%	60.0	40.0	100	74.1	25.9	100	54.5	45.5	100

Birch

		$\hat{I}$ (I-III)		
I		**0**	**1**	**Σ**
0	n	361	120	481
	%	75.1	24.9	96.6
1	n	8	9	17
	%	47.1	52.1	3.4
Σ	n	369	129	498
	%	74.1	25.9	100

The probability of damage is highest in Norway spruce sites in mid Sweden (figure 1). When probabilities calculated for all three species are mapped together the increased risk of that area is also evident.

The joint significance of the covariates was tested using a Score statistic (Monserud 1976), the resulting p-value was low (p=0.0001). The estimated proportion of damaged plots was overestimated (25.9% - 45.5%), compared to the observed proportion of 3.4% - 11.9% (table 2).

DISCUSSION

The use of sample trees and plots with different numbers of sample trees, makes it necessary to use the model syntax of *events/trials*. This results in under-estimations of the p-values both for the models and the independent variables. Therefore, a significance level of 0.05 for a variable to enter and remain in the model, was applied.

The most important factor in deciding overall susceptibility for a site, is presence of snow and wind. One way of detecting susceptible sites, is thus to use detailed climatic measurements. Such measurements are, however, hard to obtain as climatic stations are seldom situated in forested areas. As the trees themselves register all external forces acting on them, and respond to them during growth (Telewski 1995), we have used data from the largest sample tree on each plot, together with stand and site characteristics, as measure.

When the susceptibility for a site is known, the most common method of minimizing future damage from snow and wind is to avoid treatments that increase the risk for damage. Earlier studies have highlighted silvicultural treatments that should be avoided, e.g., heavy thinning and fertilization in combination and repeated fertilization (Hirvela and Hynynen 1990, Laiho 1987). In the UK, the use of risk assessment models (Booth 1997, Miller 1985) has been used to adapt silvicultural regimes to sites of different susceptibilities. As the system has been found to overestimate risks, work within the EC-project will contribute to the amendment of it.

This study shows that it is possible to classify overall susceptibility of sites to damage from snow and wind using a logistic analysis technique. However, models developed are unable to predict catastrophic events of snow and wind damage. The models developed can be used as a means to detect high risk sites, and by taking that into account, minimize future damage and costs in forestry.

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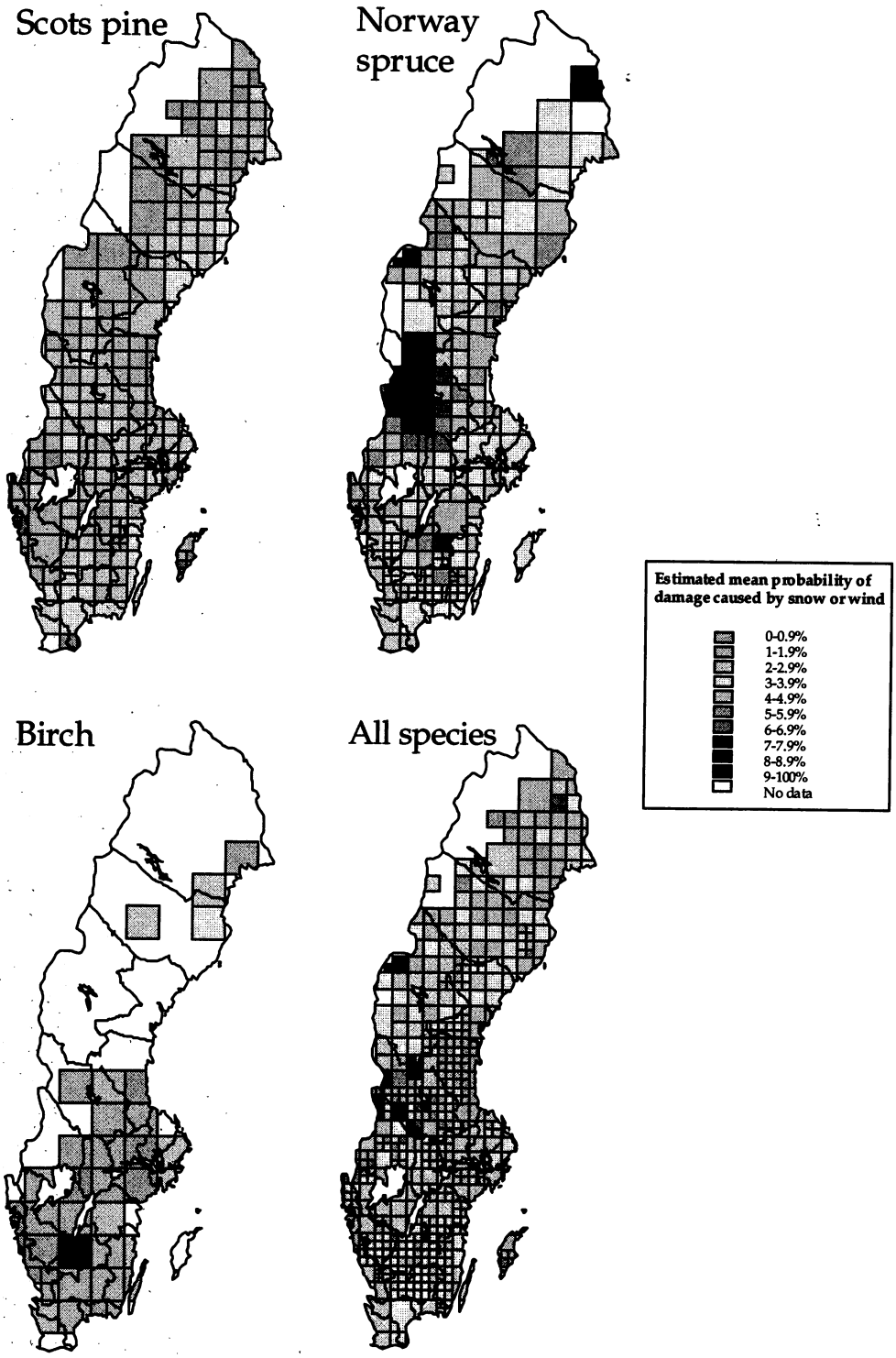

Figure 1.--Mean value of probability of damage from snow and wind for squares including at least five sample plots. Scots pine, Norway spruce, Birch and all species mapped together.

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A MATHEMATICAL MODEL OF TREE ROOT FORMS AND ESTIMATION OF EFFECTS ON DECREASING WINDFALL RISK

Y. V. Zakharov[1] and V.G. Soukhovolsky[2]

ABSTRACT.--The tree root form is considered from the viewpoint of optimization of mineral salts supply and safeguard against windfall. To describe tree stability under wind impact two models have been considered: a rigid column on elastic substrate and two types of sailing yachts. The stability of trees with horizontal and vertical root systems is compared. The effects of root form change under their growing on hard substrate have been considered.

WIND AS AN ENVIRONMENTAL FACTOR INFLUENCING THE TREE FORM

A growing tree has to solve a number of survival problems. First, the tree has to obtain a reasonable quantity of light energy. In most cases (except for creepers) this problem is resolved by lifting the photosynthetic surface upward. Second, the tree has to supply a necessary quantity of water and mineral salts from the soil. This objective is accomplished through a branched root system.

However elevating the tree crown, while solving the light trapping problem, simultaneously puts forward two new problems. First, water and salt translocation to a high crown calls for a large quantity of energy, so that the energy consumption for matter translocation into the crown diminishes the gain in light energy. Second, for a high tree with a high dense crown, the problem of the tree as stability against wind impact arises. This problem is resolved by increased density and depth of the root system. This solution is related to increased energy consumption for matter translocation from the roofs to the stem base and further into the crown.

Thus, for the purpose of survival, it is essential that a tree's geometric proportions satisfy a lot of contradictory requirements. On the one hand, the tree has to be not too tall, or the windfall probability increases. On the other hand, a low tree with a scanty crown is incapable of trapping the required light energy. To maintain the functions of a tree foundation, the root system has to be extensive, but it ought not to be too branched and deep because in this case the energy consumption for matter translocation increases.

Nevertheless, trees exist and successfully stand up to wind impact. This points to the fact that a compromise has been attained between all the contradictory requirements through evolution. And when solving the problem of tree stability under wind impact it is essential not to consider the tree a simple mechanical system but to take into account environmental restrictions.

Optimization of tree form leads to some restrictions on tree morphological parameters. As a tree gains height, the probabilities of perishing due to windfall $[p_1(H) = H/(A+H)]$ and energy consumption for matter translocation into the crown $[p_2(H) = H/(C+H)]$ increase, and the probability of perishing due to the lack of necessary light $[p_3(H) = B/(B+H)]$ decreases. Tree height will be optimal (H_{opt}), when the product of probabilities $p(H) = p_1(H)\, p_2(H)\, p_3(H)$ is minimized. Differentiating the function $p(H)$ with respect to H, and taking the first derivative equal to zero, we obtain the equation for optimal tree height H_{opt}:

$$H^3 - kH - 2ABC = 0 \qquad (1)$$

Where $k=AC+AB+BC$ and A, B, and C are constants.

For trees of sufficient height (more than 10 m) the cubic term of this equation proves to be much greater than the linear one and therefore the optimal height is $H_{opt} \approx (ABC)^{1/3}$ when $H > 10$ m.

Similarly after introducing the probabilities of perishing in consequence of energy consumption for matter translocation $[q_1(L) = L/(W+L]$ and due to small root length $[q_2 = S/(S+L)$ where L is the root length], the optimal root dimension $L_{opt} \approx (SW)^{1/2}$

[1]*Y. V. Zakharov, Siberian State Technological University and Institute of Biophysics SB RAS; address: Institute of Biophysics, Siberian Branch Russian Academy of Science (SB RAS). 660036 Krasnoyarsk, Russia; e-mail: dir@iph.krasnoyarsk.su.*

[2]*V.G. Soukhovolsky, Sukachev Institute of Forest, SB RAS, 660036 Krasnoyarsk, Russia; e-mail: ifor@krsk.infotel.ru.*

is obtained. The overall probability of tree survival will be maximum for the tree of optimal height H_{opt} and root length L_{opt}. Thus the wind impacts on a tree of specific height and root length depend on constants (fig. 1).

STABILITY OF TREES UNDER WIND IMPACT

The comparison of influence of two distinct types of root systems on tree stability under wind impact is made. Consider two types of root systems: the vertical system of the tree which grows up from a soil layer of sufficient depth, and the horizontal one on a thin soil layer with a hard rock substrate. The stability with respect to windfall will be different for trees with such types of root systems. There is a need to introduce a tree model for the quantitative assessment of this stability.

Two models describing tree response to wind impact can be introduced. Both models show that a stable equilibrium of a tree is possible up to a critical wind force. This critical force is the instability threshold and is connected with the Euler load value.

The first model treats tree stability as the stability of a rigid column on an elastic substrate under the impact of a horizontal directed force P (Kramer 1936; Panovko and Gubanova 1979). The physical properties of the substrate are described by the well-known Winkler model, according to which the substrate reaction at a point is proportional to the foundation base settling at the same point ($r=-cy$, where c is the substrate modulus).

The column deflection angle θ (fig. 2) is defined by the equation of moments:

$$Ph + Ql\theta - c\theta J = 0 \qquad (2)$$

In this equation the first term is the moment of the horizontal force P, the second term is the moment of the column weight Q, the approximation $\theta = \sin\theta = \tan\theta$ is made, and the third term is the moment of the distributed forces of the upgrade reaction with the modulus c. Here J is the moment of inertia for the foundation area with respect to the axis passing over the foundation base gravity center normal to the drawing plane.

From the equation follows the dependence of P as a function of θ:

$$P = (cJ - Ql)\theta / h \qquad (3)$$

When P=0 we obtain the Euler critical load $Q_c = cJ/l$.

The dependence according to Eq. (3) is correct until the moment when the left part of the foundation base begins to detach from the substrate as the angle θ increases. This is shown in the substrate reaction diagram in fig. 2. The critical detachment angle for the rectangular base of sides a and b and the moment $J=ba^3/12$ is equal to $\theta_1=2Q/ca^2b$, and the critical force is $P_1=Qa(1-Q/Q_c)/6h$.

On further increasing of the force P the relationship between the deflection angle and the acting force becomes nonlinear:

$$P = (Q/h)\left[a/2 - (1/3)(2Q/\theta bc)^{1/2} - l\theta\right] \qquad (4)$$

And the dependence $P(\theta)$ has the maximum (fig. 3):

$$P_c = (Qa/2h)\left[1 - (Q/Q_c)^{1/3}\right] \qquad (5)$$

The corresponding angle is $\theta_c=(Q/18\, l^2bc)^{1/3}$. When $P > P_c$, there are no equilibrium states and the column falls down.

In the approach above, simplifying assumptions about the small angle ($\theta \approx \sin\theta \approx \tan\theta$), the absolute rigidness of the column, and settling of the small base were made. Considering these circumstances properly, the curves of equilibrium states will be situated below the displayed ones (fig. 3) but this improvement is of a quantitative rather than a qualitative character.

In comparing a vertical root system to a horizontal one, analysis of modification of the curve $P(\theta)$ is of interest for our purposes. In this model such a transition from a vertical root system to a horizontal one implies an increase in the horizontal foundation sizes a and b and thus an increase in the moment of inertia J. The moment J is proportional to L^4 (or to the square of a root system area S^2). The changes of the distances l and h are proportional to the characteristic measure of the root system L. However, the fractional changes of these distances are much less than the fractional changes in the linear foundation sizes. Thus it may be considered that the declination angle of the straight line portion of the curve $P(\theta)$ [see Eq. (2)], in changing from a vertical tree root system to a horizontal one, will increase proportionately to L^3 (that is proportionally to $S^{3/2}$), and the critical force P_c will increase proportionally to L (that is proportionally to $S^{1/2}$, where S is the

In general defoliation of the tree crowns and their distribution over damage classes may have causes of two different types: first, direct impact of air pollution or other damaging factors of regional scale and, second, a large number of local ecological factors with various effects on tree health, including random ones. So, damage classes can be represented as follows:

$$E_i = iA,$$

where i = 0, 1, 2, 3 - denotes the base damage classes: 0-no damage, 1-slightly, 2-moderately, and 3-severely damaged and dead trees. E is the damage class corrected for human impact, A - is a dimensionless number to characterize the intensity of direct human impact on forest health, if A=1 then direct impact is absent. The health status of the forest is the mean corrected damage (E) class of the sampled trees (Alekseyev 1990):

$$E = \sum_{i=o}^{3} E_i p_I,$$

if so, then variation of the forest health status may be naturally presented as the sum of two components corresponding to two kinds of reasons which determinate such a variation:

$$dE = \sum_{i=0}^{3} p_i dE_i + \sum_{i=0}^{3} E_i dp_i = dW + dQ.$$

In this formula first component (dW) describes variation in forest health status due to direct human impact which is common for the trees of all damage classes, second component (dQ) - is variation in the forest health as a result of random migrations of the trees between fixed damage classes.

Consider in more detail the process of random migrations of the trees between damage classes. Such migrations will occur both to the better damage classes (with low numbers) and to the worse (with high numbers) according to following scheme:

$$E_0 \underset{a_{10}}{\overset{a_{01}}{\Leftrightarrow}} E_1 \underset{a_{21}}{\overset{a_{12}}{\Leftrightarrow}} E_2 \underset{a_{32}}{\overset{a_{23}}{\Leftrightarrow}} E_3,$$

here E_i- damage classes; a_{ij} - the intensity of migrations of the trees from damage class i to j (dimension - 1/time). The dynamics of vital structure of the forest in this case must follow the below system of differential equations:

(1)

$$dp_i/dt = \sum_j (p_j a_{ji} - p_i a_{ij}),$$

and for stationary conditions the principle of detailed equilibrium must be fulfilled (Rubin 1984):

(2)

$$p_j a_{ij} = p_i a_{ij}.$$

It follows from equation (2) that any two probabilities of health state of trees in the forest are connected by the relationship below:

(3)

$$p_j = p_i a_{ij} / a_{ji} = p_i K_{ij},$$

where $K_{ij} = a_{ij} / a_{ji} = p_j / p_i$ - is the constant of equilibrium for migrations of trees from damage class i to j. For any chain of migrations we have following relationship:

(4)

$$p_k = p_l K_{li} K_{ij} K_{jk},$$

or in logarithmic form:

(5)

$$\ln p_k = \ln p_l + \ln K_{li} + \ln K_{ij} + \ln K_{jk}.$$

For the migration process of trees between fixed damage classes, due to its stochastic nature the principal is a cyclic process. For such a process from equation (4) follows:

(6)

$$1 = K_{ki} K_{ij} K_{jk}$$

Each damage class has its own characteristics:

$$t_i = (E_i - E) / B,$$

where B - multiple with dimension of damage classes, so t_i is dimensionless.
For the cyclic migration process $k \Rightarrow i \Rightarrow j \Rightarrow k$, the sum of the damage class characteristics must be equal to zero:

(7)

$$0 = (t_k - t_i) + (t_i - t_j) + (t_j - t_k).$$

There is only one shape of the functional relationship of the constant of equilibrium K_{ij} from t_i and t_j which simultaneously satisfies both conditions of the cyclic migration process (6) and (7), this is the exponential function:

(8)

$$K_{ij} = \exp(t_i - t_j).$$

After substitution of (8) to (5) from last follows equation which are right for all pairs k and l:
(9)

$$B\ln p_k + E_k = B\ln p_l + E_l.$$

If we denote as F the number equal to the left and right sides of (9), then we have obtained the distribution of the trees over damage classes in canonical Gibbs form (Gibbs 1982):
(10)

$$F = B\ln p_i + E_i$$

$$p_i = \exp((F - E_i) / B).$$

From condition of standardization and (10) follows:

$$F = - B\ln Z,$$

where $Z = \sum_{i=0}^{3} \exp(-E_i / B)$ - is the statistical sum of the distribution of the trees over damage classes. Finally the distribution of the trees to different damage classes can be more conveniently calculated as follows:
(11)

$$p_i = (1/Z)\exp(- E_i/B) = (1/Z)\exp(-iA/B).$$

This distribution which we will call the distribution of the first type is very often found in forest monitoring data (table 1) and describes fully healthy or slightly damaged forests.

To clarify what the number F means remember that in statistical physics F is the free energy of the system whose elements are distributed over energy levels in accordance with (11). Notice that from (11) follows:
(12)

$$p_0 = 1/Z, \quad Z = 1/p_0$$

$$\text{and} \quad F = B\ln p_0,$$

so, F under constant B is determined by the proportion of the fully healthy (undamaged) trees in the forest and therefore is characteristic of its vitality. The meaning of parameter B may be clarified with equation (8) from which it follows that:

$$B = (E_i - E_j) / \ln K_{ij}$$

If B is maximum (infinity) then $K_{ij} = 1$, in other words, if B is maximum then the intensity of migrations of the trees due to random reasons are equal to and from each damage class. So, B may be considered as characteristic of random transitions of the trees over damage classes. In statistical physics the analog of B is the temperature of the system which describes chaotic movements of the system elements. This fact allows us to consider the meaning of parameters F and B to be similar in forest ecology and statistical physics.

There are some useful relationships between distribution parameters which can be easily derived from equations (10):
(13)

$$F = E - BH$$

where $H = - \sum_{i=0}^{3} p_i \ln p_i$ - is entropy of distribution of the trees over damage classes. From last equation accounting for (12) it follows that:
(14)

$$B = E / (H + \ln p_0) \text{ or } B / A = i_m / (H + \ln p_0)$$

Here i_m is the mean damage class of the forest, calculated using base damage classes.

The second type of distribution of trees over the damage classes arises if in (7) we use for damage class characteristics the numbers t_i^2 instead of t_i. The same analysis presented above gives the following distribution of the trees over damage classes:
(15)

$$p_i = (1/Z^*)\exp(-(E_i - E)/B)^2 = (1/Z^*)\exp(-(i - i_m)A/B)^2,$$

Table 1.--*Analysis of vitality structure of the coniferous forests of Europe.*

Country	Area of coniferous forests (1000 ha)	Number of sample trees	Distribution of the trees over damage classes (%) 0	1	2	3	Type of distribution	Distribution parameters A	B	R^2 (%)
France	5040	3488	77.9	13.9	7.6	0.6	1	0.61	0.38	99.4
Portugal	1340	1545	71.9	21.0	6.2	0.9	1	0.71	0.57	99.9
Luxemburg	30	386	63.0	28.0	7.0	2.0	1	0.94	1.01	99.1
Russia[1)]	25518	1656	56.6	38.9	3.6	0.9	1	0.96	1.29	88.6
Sweden	19729	15657	62.4	27.0	8.9	1.7	1	0.98	1.07	99.5
Austria	2922	5723	56.5	35.3	7.6	0.6	1	1.03	1.38	93.6
Finland	18484	3754	60.7	23.7	14.2	1.4	1	1.10	1.26	98.7
Romania	1929	47288	53.0	30.4	15.0	1.6	1	1.28	1.91	97.1
Spain	5637	5510	50.0	35.4	11.6	3.0	1	1.32	2.18	92.6
Greece	954	1015	44.3	41.8	11.6	2.3	1	1.41	1.43	88.8
Hungary	267	3591	50.5	29.4	15.4	4.7	1	1.46	2.36	98.7
Englend	1550	5376	40.9	42.3	15.7	1.1	2	1.51	1.98	92.9
Norway	7000	6904	44.6	34.5	16.8	4.1	1	1.58	3.23	90.9
Estonia	1135	2065	41.3	37.5	19.6	1.6	1	1.60	3.78	78.7
Netherlands	165	20675	53.1	16.3	25.8	4.8	1	1.61	2.37	79.2
Belgium	302	1205	38.0	43.7	16.2	2.1	2	1.62	2.10	93.7
Germany	6946	55950	39.4	39.2	19.8	1.6	2	1.64	2.39	89.8
Croatia	321	395	55.0	11.1	28.1	5.8	1	1.66	2.85	57.2
Bulgaria	1172	4492	36.2	36.9	23.1	3.8	2	1.85	2.98	86.7
Switzerland	818	1212	27.6	51.7	18.4	2.3	2	1.87	2.07	98.7
Ukraine	2931	1348	25.4	53.3	20.9	0.4	2	1.89	2.05	99.8
Ireland	334	462	31.4	39.0	28.6	1.0	2	1.95	2.91	91.9
Denmark	308	975	42.6	20.4	25.6	11.4	1	2.07	5.03	78.7
Slovenia	500	380	24.0	49.0	23.0	4.0	2	2.10	2.48	98.3
Moldova	6	104	35.6	19.2	45.2	0.0	1	2.15	1.80	53.8
Lithuania	1073	3787	15.5	55.3	25.7	3.5	2	2.30	2.27	99.3
Belarus	4757	7185	15.9	50.3	32.4	1.4	2	2.34	2.57	99.4
Latvia	1633	6854	17.0	42.0	39.0	2.0	2	2.47	3.07	93.8
Czech Republic	2051	11545	13.1	34.1	47.6	5.2	2	2.84	3.40	86.5
Slovak Republ.	816	1822	7.8	42.3	46.3	3.6	2	2.86	2.90	97.7
Poland	6895	25260	5.8	43.4	47.9	2.9	2	2.90	2.72	98.5

[1)] *Only St.-Petersburg region.*

Table 2.--*Estimations of needed numbers of sample trees for statistical reliability of the forest monitoring data .*

Distribution type	Variance Damage class 4	5	Needed number of sample trees Damage class 4	5
Standard (lower limit)	0.61	0.73	1220	1455
Uniform (upper limit)	1.25	2.00	2500	4000

here $Z^* = \sum_{i=0}^{3} \exp(-(Ei - E)/B)^2$.

For the second type of distribution of the trees over damage classes there are equations analogous to (13) and (14):
(16)

$$F^* = \sigma^2 - B^2H, \quad B^2 = (\sigma^2 - E^2) / (H + \ln p_0), \quad (B/A)^2 = (\sigma^2_1 - i^2_m) / (H + \ln p_0),$$

here σ^2_1 is the variance of the distribution of trees over base damage classes.

The second type of distribution of trees is usual for moderately and severely damaged forests, in this case the maximum number of trees are not in the full health (no damaged) class as under distribution of the first type but in the mean damage class for whole forest. Such distribution also often occurs in forest monitoring data (table 1).

The third type of distribution of the trees over damage classes occurs if damaging factors are very strong and as a result the forest decline is unreversible. In this case immigration of trees between damage classes follows the scheme:

$$E_0 \xrightarrow{a_{01}} E_1 \xrightarrow{a_{12}} E_2 \xrightarrow{a_{23}} E_3.$$

Under such dynamics of vitality groups of trees there is no equilibrium state, all trees become dead in time. The decline dynamics can be described with the balance differential equations:

$$dp_0/dt = - a_{01}p_0, \quad dp_1/dt = a_{01}p_0 - a_{12}p_1, \quad dp_2/dt = a_{12}p_1 - a_{23}p_2, \quad dp_3/dt = a_{23}p_2.$$

The analitical solution of these equations has the following form:

$$p_i(t) = \sum_i A_i \exp(-b_i t),$$

where, t - is the time; A_i, b_i - parameters which depend on a_{ij} and vitality structure of the forest at initial moment of time - p_i^0, for example:

$$p_0(t) = p_0^0 \exp(- a_{01} t).$$

If the coefficients of intensity of the trees transitions - a_{ij} are known then it is possible to determine the average life span of the forest - T and the time needed for a 50% decrease in the number of fully healthy trees as - $T_{1/2}$:

$$T = \Sigma(1/a_{ij}), \quad T_{1/2} = \ln 2/a_{01}.$$

The third type of forest vitality structure does not occur in European forest monitoring data and is only found near strong pollution sources such as large chemical or smelter plants (Alekseev 1990, Norin and Yarmishko 1990).

The published forest monitoring data for European countries can be very well described by these theoretically derived distributions of trees over damage classes of the first and second type. All European Boreal countries (Norway, Sweden, Finland, and Russia data only for Saint-Petersburg region) have the first type of distribution of trees over damage classes (figs. 1 - 6). Figures 5 and 6 illustrate the second type of distribution of trees over damage classes for England and Poland. The forest health in England is the best among countries with the second type of distribution, while forest health in Poland is the worst.

MEASUREMENT OF DIRECT HUMAN IMPACT ON FOREST HEALTH BY INTRODUCING OF STANDARD (NORMAL) FOREST

The published forest monitoring data describes vitality structure by the distribution of the sample trees over base damage classes. However this data is not enough for determination of all vitality structure parameters, first of all the intensity of direct human impact on forest health. In order to solve this problem remember that all direct measurement is the comparison of the object with some standard (Pfanzagl 1971). The intensity of direct human impact on forest health status can be measured only by comparison with a specially introduced standard forest vitality structure represented by a standard (normal) distribution of the trees over damage classes. Such a standard introduction is more or less voluntary but it must satisfy some obvious conditions. First, the standard distribution of trees over damage classes must be of the first type, which is usual for fully healthy forests; second, the mean damage class for this distribution--calculated using base damage classes--must be equal to approximately 0.5 as it is for non- damaged forests; and, third, the standard distribution must be simple. There is only one simple distribution which satisfies all three conditions:

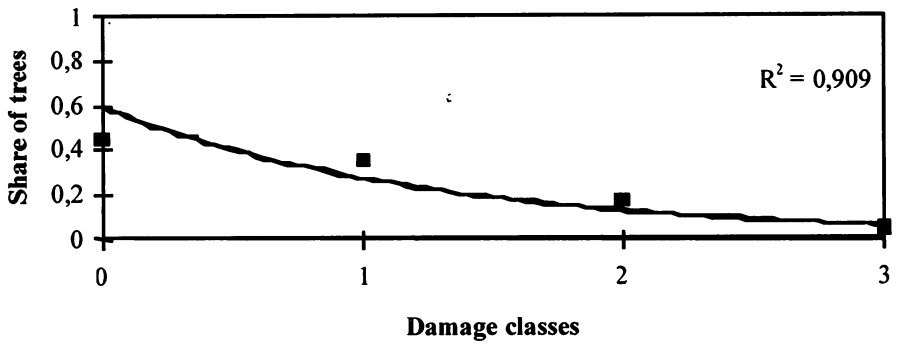

Figure 1. *Distribution of the trees over damage classes for Norway (Type 1).*

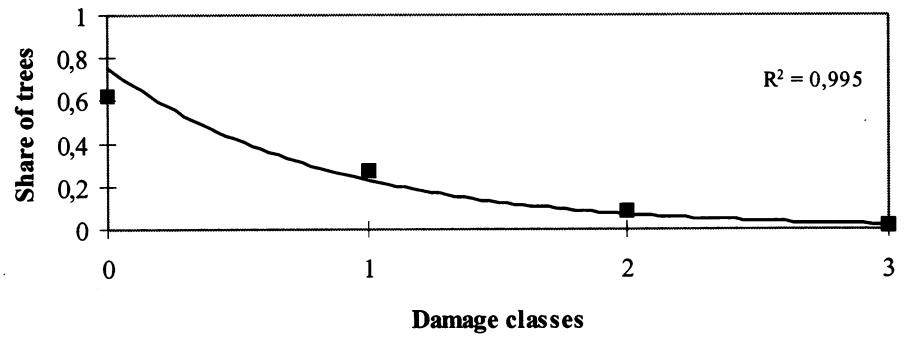

Figure 2. *Distribution of the trees over damage classes for Sweden (Type 1).*

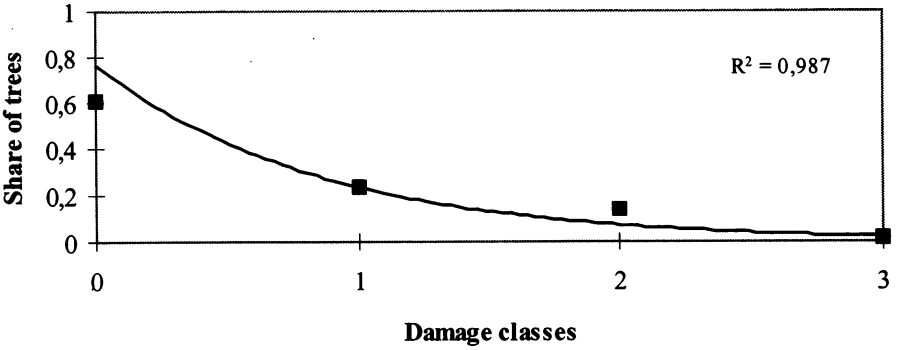

Figure 3. *Distribution of the trees over damage classes for Finland (Type 1).*

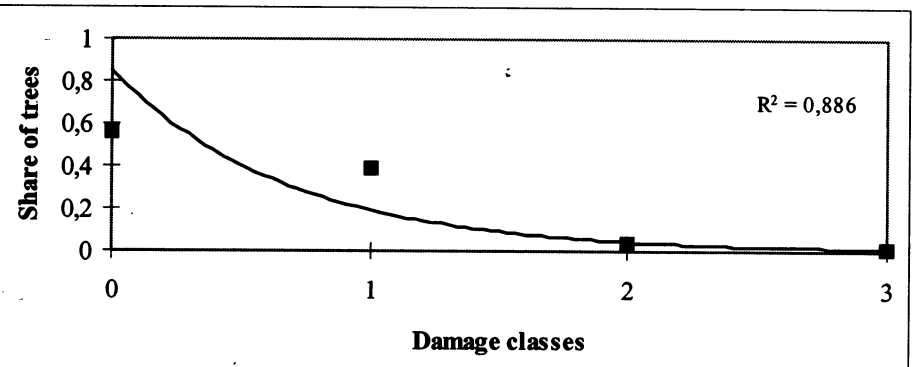

Figure 4. *Distribution of the trees over damage classes for Russia (Sankt-Petersburg region) (Type 1).*

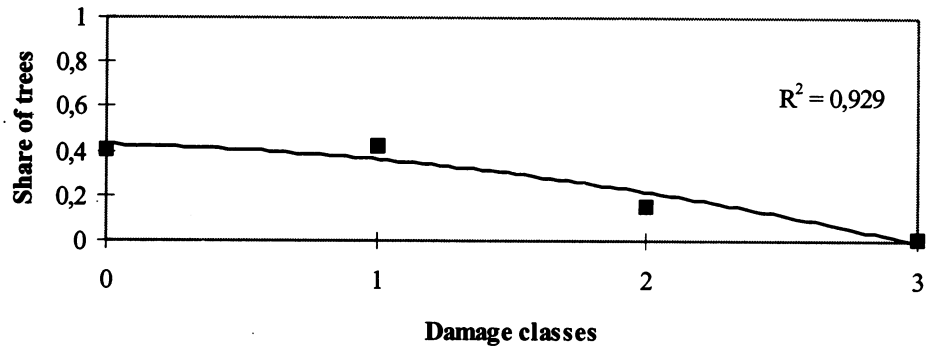

Figure 5. *Distribution of the trees over damage classes for England (Type 2).*

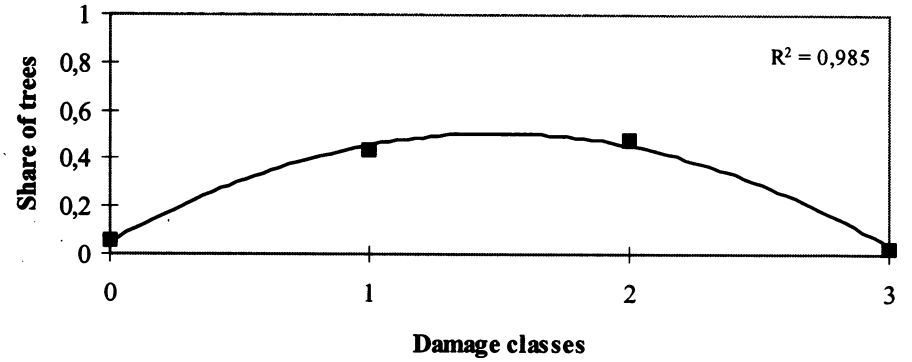

Figure 6. *Distribution of the trees over damage classes for Poland (Type 2).*

(17).

$$p_i = \exp(-i) / \sum_{i=0}^{3} \exp(-i) = 0.64 \exp(-i).$$

The standard (normal) distribution of trees over damage classes is presented in figure 7. For this distribution, A=B=1. In other words direct human impact is absent and the standard (normal) intensity of random transitions of the trees between damage classes (normal "temperature" of the forests) is equal to one. The mean damage class (i_m) calculated using base damage classes is 0.51. In order to distinguish this mean damage class for standard distribution from others it will be marked i_{st}, so $i_{st} = 0.51$.

Direct human impact changes the health status of the whole forest at once. This is described by mean damage class, therefore relative intensity of impact (A) can be calculated as follows:

$$A = i_m / i_{st}.$$

When A is known using formulas (14) or (16), along with data on distribution of the trees over damage classes, then parameter B and all other needed characteristics of the distribution can be easily calculated.

The results of measurement of direct human impacts on forest health status and the intensity of random transitions of the trees between damage classes (forest "temperature") using European forest monitoring data (Forest condition in Europe... 1994) are presented in table 1.

In table 1 countries are ordered in accordance with increase of intensity of direct human impact on the forests (A). The European forest monitoring data are described by theoretically derived distributions with a high level of accuracy. Coefficients of determination are high (greater than 78%) for all countries, except Croatia and Moldova where determination coefficients are 57.2% and 53.8%, respectively. This can be explained by, first, low number of sample trees, second, methodological mistakes in forest monitoring process, and third, that forests in these countries are not in stationary equilibrium state. Data from table 1 shows that as a rule countries with the first type of tree distribution over damage classes have a low level of direct human impact on forest health status.

APPLICATION OF RESULTS

The results obtained on principal types of distributions of trees over damage classes, an intensity of the direct human impact on the forest health status of European countries, and on intensity of random transitions of the trees between damage classes ("temperature" of the forests) will be used in two ways: first, for comparative analysis of the forest health between three groups of countries: boreal, industrially developed, and other, and second, for analysis of statistical reliability of the data provided by the forest monitoring system.

Comparative Analysis of the Forest Health Status of European Countries

Comparative analysis will be done for European coniferous forests as they are more sensitive to air pollution and some other damaging factors than broadleaf forests. All European contries are divided into three groups: first--countries of the boreal zone: Norway, Sweden, Finland, and Russia; second, industrially developed countries: France, England, and Germany; and third--other countries (see table 1). Intensity of direct human impact and random transitions of the trees between damage classes for these groups of countries were calculated as weighted averages, the area of coniferous forests in each country was used as a weighting factor.

Coniferous forests of Europe are mainly located in the countries of the boreal zone. Figure 8 shows that nearly 60% of the total area of coniferous forests is in this group of countries, with 10% in industrially developed countries and 30% in the others.

Within the group of boreal countries only Norway has an intensity of direct human impact on forest health (A) an intensity of random transitions (B) significantly more than standard or normal. Forest health status of the other boreal zone countries is approximately the same and close to normal (figs. 9 and 10). It can be proposed that the coniferous forests of Norway, more than in neighboring boreal countries, are subject to long-range transboundary air pollution.

Comparison of the forest health status between the groups of countries shows that the lowest intensities of direct human impact and random transitions of the trees between damage classes ("temperature" of the forest) are in the countries of boreal zone, highest in other countries, and middle in the

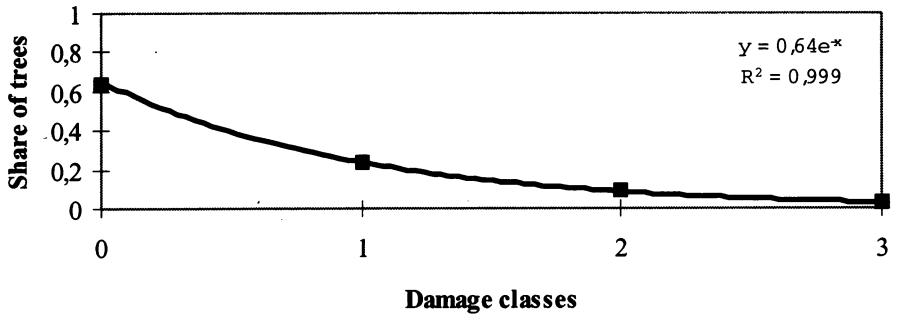

Figure 7. *Standard (normal) distribution of the trees over damage classes (Type 1), as a base for forests health status measurements.*

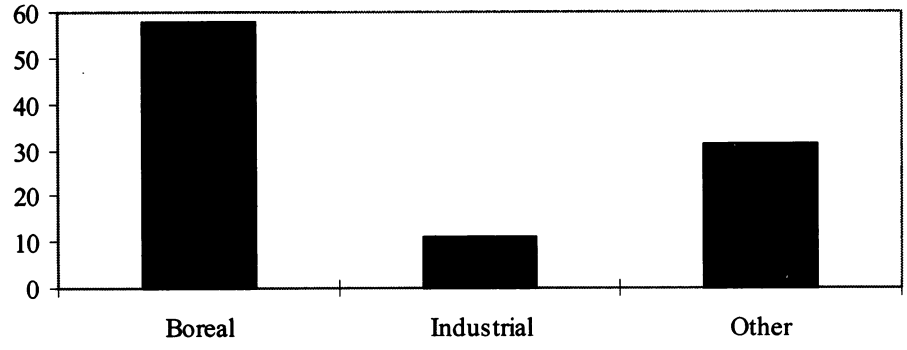

Figure 8. *Shares of the groups of European countries in total coniferous forests area, %*

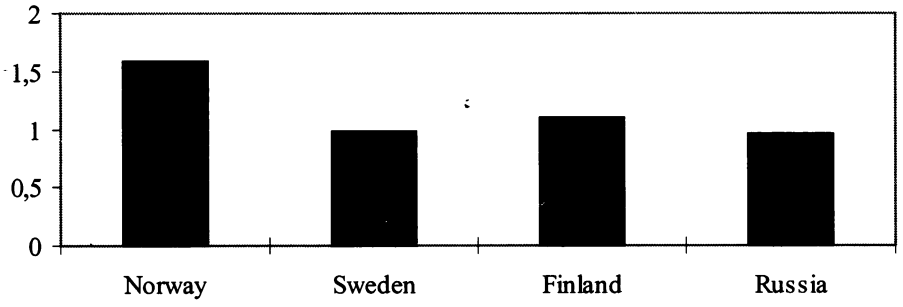

Figure 9. *Direct human impact on the forest health of Boreal zone countries (relative units, 1 - no impact)*

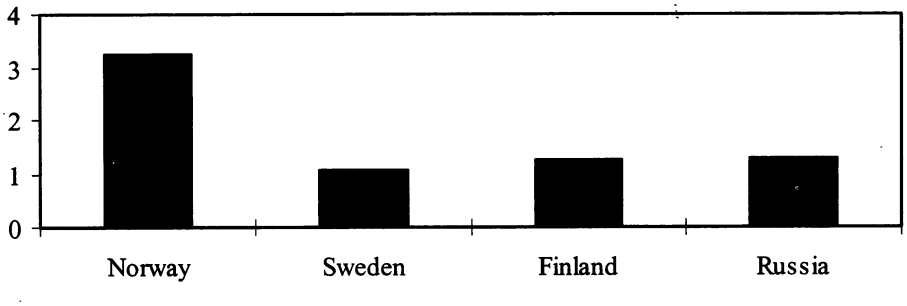

Figure 10. *Intensity of random transitions of the trees over damage classes ("tempereture" of the forests) for Boreal zone countries (relative units, 1 - normal intensity - "tempereture")*

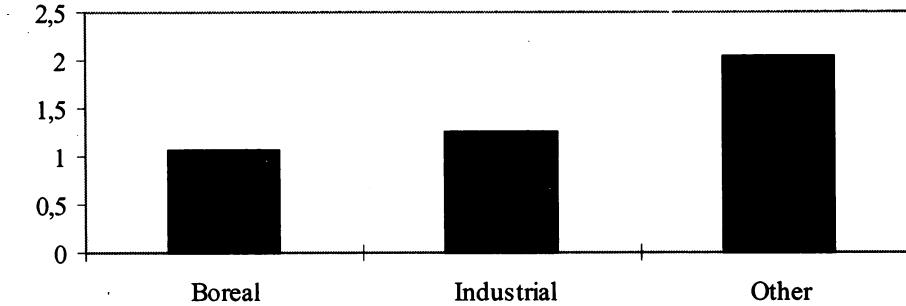

Figure 11. *Direct human impact on forest health for the groups of European countries (relative units, 1 - no impact).*

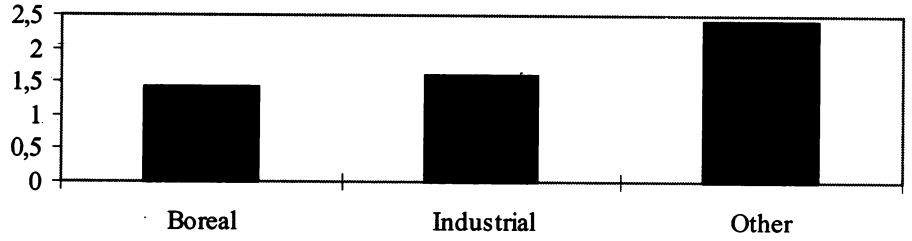

Figure 12. *Intensity of random transitions of the trees over damage classes ("tempereture" of the forests) for the groups of European countries (relative units, 1 - normal intensity-"tempereture").*

industrially developed (figs. 11 and 12). The relatively bad forest health status of the group of other countries is due to such members of this group as Poland, Czech Republic, and some others which have relatively large areas of coniferous forest and high levels of air pollution due to using coal with high content of sulfur. In industrially developed countries, owing to modern technologies and strong environmental policy, human impacts on forest health are moderate. We can conclude that the health status of European coniferous forests depends primarily on the health of the forests in boreal zone countries, which are now in relatively good condition.

Notes on Statistical Reliability of the Forest Monitoring Data

If the forest health status of any country is described with mean damage class of the sample trees from a regular bioindication grid then the number of sample trees must be determined to give the data statistical reliability. As was shown above the distribution of trees over damage classes may be of three types, two of which are usual for all European countries. Because of such a variety of distributions and statistical interdependence between health status of sample trees, we cannot base estimation of needed number of sample trees on the normal (Gauss) law usually used for this purpose. In this case we must use more common form of Large number low in form of Chebyshev unequality:

$$P (|x - x^*| > a) < s^2 / a^2 N,$$

here, x - mean damage class determined using N samples; x^* - true mean damage class for some territory or country; a - allowable deviation from true mean damage class; s^2 - variance of the distribution of the trees over damage classes; P - probability of deviation of mean damage class calculated using N samples from true mean more than on a. From the above unequality the number of sample trees needed for statistical reliability is:

$$N = s^2 / a^2 P.$$

For calculations with this formula we must determine the components of its right side. Let allowable deviation of calculated mean damage class from true mean - a = 0.1, and probability of more than a deviation - P = 0.05. In order to determine s^2 we must know the distribution of trees over damage classes, which is unknown before the monitoring programme will be done. There is one special type of distribution of trees over damage classes which is common for distributions of types 1 and 2--uniform or regular. Such a distribution has the highest variance, and estimations of the need number of sample trees with it can be considered as an upper limit for real number. Meanwhile a lower limit may be based on estimation with standard (normal) distribution of the trees over damage classes. Estimations of the needed for statistical reliability of the forest monitoring data numbers of sample trees are presented in table 2. Because damage to trees is specific for different species, estimations shown in table 2 are valid for only individual species or close groups. Table 2 demonstrates that the number of sample trees needed depends strongly on the number of damage classes. Published data (Forest condition ... 1994) are grouped into four damage classes but usually dead trees are accounted for separately. Furthermore, sometimes dead trees are separated into new and old mortality. Comparison of the estimated numbers of sample trees needed for statistical reliability with real numbers of sample trees (table 2) shows that in some countries real numbers are less than lower limit.

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DEGRADATION OF MONTANE FIR FORESTS IN THE KEMEROVO OBLAST: SEARCH FOR CAUSES

Vladislav A. Alexeyev[1]

ABSTRACT.--Siberian fir forests in mountains of the Kemerovo Oblast (southern part of Western Siberia, Russia) are in decline. A field expedition of scientists conducted complex research studies in 1996-1997 to determine reasons for this forest decline. This work clarified that the current condition of the weakened stands that were studied was caused by several fungi, including the rust fungus *Melampsorella caryophyllacearum* Schroet., which is distributed on a pandemic scale Most of the fir trees (from 30 to 90% in the studied stands) showed symptoms of fungus infection: yellow witches brooms, bole and branch cankers, and dieback of some needled twigs. The first and second signs occurred in trees of all ages; the third was more common in mature trees. Direct mortality from cankers occurs rather rarely, but stands weakened by cankers are more easily damaged by insects and other destructive agents; they can become stands at risk. The area of infected fir forests exceeds 2 million ha.

INTRODUCTION

More than 400,000 ha out of 1,300,000 ha of Siberian fir (*Abies sibirica*) forests observed by forest pathologists are in decline in the Kuznetsk Alatau Mountains of the Kemerovo Oblast in Middle Siberia. These forests are distributed about 70 km to 110 km from industrial sites. The area includes 170,000 ha of moderate damage, 150,000 ha of heavy damage, and 80,000 ha of dying stands (Kovalev 1994). A 1994 survey disclosed a worsening of the forest condition (Kovalev 1994). Forest pathologists and ecologists supposed that the development of forest decline in mountain ecosystems would increase soil erosion, threaten watersheds and streams, and produce an ecological catastrophe in the region (Gritsko *et al.* 1996). In spite of several studies, the causes of the decline have remained unclear. Hypotheses for the reasons of forest decline have included two major factors: (1) regional air pollution impact, and (2) prevalence of overmature stands in virgin mountain fir forests. However no reliable data was available. For this reason the Forest Administration of the Kemerovo Oblast suggested that I organize a project, "The revelation of reasons of fir forest decline in mountains of Kemerovo Oblast and development of management for damaged forests." In 1996 a research team consisting of an ecologist, a forest inventory specialist, a soil scientist, a forest pathologist-mycologist, a geobotanist, an entomologist, a lichenologist, and several chemists initiated a 3 year study on this topic. To our surprise, the current condition of the degraded stands was caused by several fungi, including the rust fungus *Melampsorella caryophyllacearum* Schroet., which is distributed on a pandemic scale. In 1997 this research was continued also in the Krasnoyarsk Kray.

The goal of this paper is to present the first-year results of this study and to attract the attention of the international community to this phenomenon.

METHODOLOGY

To determine reason(s) for forest decline, we used a complex of ecosystem research methods. Standards for arranging sample plots and methods to estimate sanitary conditions were published by the Forest Service of Russia (1992). Methods for estimation of forest health are reported by Alexeyev (1989, 1990); and descriptions of forest vegetation were made by Sukachev and Zonn (1961). Samples of epiphytic and ground lichens, needles, twigs, litter, and soils were taken from healthy and damaged forest ecosystems for chemical analysis, which were conducted by standard methods (Arinushkina 1970, Tsyplenkov *et al.* 1981, Yagodin 1987, Mochalova 1975).

STUDY AREAS

Primary research measurements were made in the region of the most heavily damaged stands in the Kuznetsk Alatau Mountains (the watersheds of the upper reaches of the Usa and the Black Usa rivers). Plots were selected in highlands (on plateau, slopes, and valleys) and included middle-aged, maturing, and mature stands of different health. Then measurements were continued in the foot-hills of the mountains. A brief description of several plots is given in table 1.

[1]*V.A. Alexeyev, St.-Petersburg Forestry Research Institute, 21 Institutskiy Ave, Saint-Petersburg 194021, Russia.*

Sample plots 1 and 2 are on a plateau. They characterize widely distributed maturing and mature stands with closed canopies. Plot 3 represents middle-aged closed fir stands in valleys of high-mountain creeks. Plot 4 characterizes extremely dense middle-aged fir stands on the lower parts of highland slopes. Plot 5 represents a natural middle-aged stand of fir regeneration after a clearcut in the 1940's in the foot-hills of the Kuznetsk Alatau Mountains.

Apart from these permanent sample plots, we sampled temporary plots and several inspection routes in Kemerovo Oblast with total length more than 200 km. We could not find dying fir stands in this region.

RESULTS AND DISCUSSION

Health of Fir Forests

In accordance with the project objectives life state (damage) and health condition of fir trees and stands of sample plots were estimated. Indices of health of stands varied from damaged to heavily damaged and depend on whether estimates were based on the number of trees or tree volume (table 2). Mature (plot 1) and maturing (plot 2) stands contained many dead stems of different size and age which were killed 10-20 years ago. Other stands had many small, supressed dead trees. The appearance of new dead trees was negligible; the proportion of dying trees was also insignificant (table 3). These data indicate that at the present time intensive processes of forest degradation are not occurring. This important conclusion was checked and confirmed on all inspection routes. Thus we cannot confirm the thesis of rapid worsening of forest health. Nevertheless large portions of trees in stands are weakened (damaged) and heavily weakened (heavily damaged) and the average health status of stands in most cases was weakened. Therefore a search for the reasons for degraded stands is valid.

Air Pollution Impact on Degradation of Fir Stands

Investigation of fir trees on sample plots and routes did not show symptoms of air pollution impact on fir trees. Maximum life span of fir needles in most damaged mountain forests was 14 years and the average was 9 years. Necroses and chloroses were not observed. Only in the vicinity of Mezhdurechensk city, needles older than one year had chloroses (less 10% of needle area) and small spots of necrosis (1%-3% of needle area).

Table 1.--*Brief forest inventory description of permanent sample plots.*

Plot	Composition of stands[1]	Average age	Average height	Average diameter	Relative basal area	Site quality class	Type of forest	Growing stock
		years	*m*	*cm*				*m^3/ha*
1	100% A	95	15	23	0.79	IV	A. mh[2]	160
2	100% A	85	13	18	0.77	IV	A. mh[2]	130
3	10 0%A	55	12	14	1.24	II	A. mh[2]	190
4	100% A	50	13	14	2.03	II	A. nudum	350
5	80%A 20%B	50	15	20	0.77	I	A. mh[2]	180

[1]A -- *Abies sibirica*; B -- *Betula pendula.*
[2]*Abietum magniherbetum.*

Table 2.--*Disturbance and sanitary conditions of fir stands on sample plots.*

N plot	Disturbance of stands estimated by:				Sanitary condition of stands estimated by:			
	Number method		Volume method		Number method		Volume method	
	Index	Life state	Index	Life state	Index	Condition	Index	Condition
	%		%					
1	28	damaged	30	damaged	2.9	heavily damaged	3.0	heavily damaged
2	29	damaged	38	damaged	3.2	heavily damaged	2.8	heavily damaged
3	40	damaged	33	damaged	3.2	heavily damaged	2.4	damaged
4	14	healthy	6	healthy	2.8	heavily damaged	1.6	damaged
5	21	healthy	12	healthy	2.4	damaged	1.8	damaged

Table 3.-- *Distribution of trees (percent) of various health status and sanitary condition with and without acounting of their rank.*

N plot	Total	Category of health status of trees (%)					
		Healthy	Weakened	Heavily weakened	Dying off	This year dead trees	Many years ago dead trees
	%						
		Estimated by volume method					
1	100	24.5	27.9	18.8	2.4	0.0	26.4
2	100	11.8	38.3	34.4	0.9	0.0	14.6
3	100	20.3	46.3	23.7	1.5	0.7	7.5
4	100	77.0	12.6	1.5	0.8	0.0	8.1
5	100	62.9	20.6	6.0	0.7	0.0	9.8
		Estimated by number method					
1	100	30.0	25.9	17.1	3.5	0.0	23.5
2	100	22.6	32.0	10.7	4.0	0.0	30.7
3	100	15.0	28.7	24.9	5.8	1.5	24.1
4	100	49.4	14.7	3.5	1.8	0.0	30.6
5	100	39.2	31.1	10.8	1.3	0.0	17.6

Numerous analyses of fir needles for the territory of the Mezhdurechensk forest farm and six other forest farms of the Kemerovo Oblast were carried out. Sulfur content in the needles of damaged and healthy trees was the same and did not exceed 0.12% of needle dry weight. However, close to Mezhdurechensk city sulfur contents were 0.17%. It is interesting to note that in the Lake Baikal region sulfur content in fir needles is 0.17%-0.19% in clean locations (Rozhkov and Kozak 1989).

As is well known, many species of lichens are very sensitive to sulfur pollutants. Therefore special attention was paid to the condition of lichens. The region of most heavily damaged fir stands was characterized by abundant distribution of epiphytic lichens (*Usnea* ssp., *Hypogimnea physodes*, etc.), which were observed on 100% of trees. Open sites had a lot of well-developed ground lichens: *Alectoria ochroleuca, Cetraria islandica, C. laevigata, Cladina rangiferina, C. stellaris, Cladonia amaurocrala, C. gracilis, Stereocanlon sp.*, etc. According to T.N. Otnyukova[2] the morphological state of *Cladina rangiferina* (test lichen) in the studied region has no divergence from normal. Sulfur content in lichens of different species varied from 0.12%-0.42%. The lowest average sulfur content was observed in lichens from open sites. Studies of forest soils and vegetation of the lower layers of open forest communities also showed no anomalies.

On the basis of these data, we conclude that the level of present (and recent) regional air pollution cannot be the reason for modern degradation of fir forests in the territory studied.

Impacts of Age of Fir Stands on Their Mass Degradation

Siberian fir is a primary species of Kemerovo Oblast. In the Mezhdurechensk forest farm which is the main site for this project, fir occupies 82% of the forested area (377,800 ha). According to the last forest inventory of forest farms young fir stands were distributed on 8.5% of area, middle-aged on 12.8%, maturing on 9.9%, mature on 68.4%, and overmature stands on 0.4% (1,400 ha) of the fir area. So overmature stands cannot be the reason for mass degradation of fir stands. In addition, fir communities as a rule form uneven-aged stands (Falaleyev 1985), as can be seen in table 4.

Distributions of damaged trees by age class (table 4) and diameter group (table 5) also show that age structure of fir forests is not a source of their mass degradation: in a mature stand (plot 1) heavily weakened and dying trees occurred in different levels of the canopy, whereas in middle-aged (plots 3-5) and maturing (plot 2) stands, they were formed mainly as result of competition. Statistical analysis did not find significant correlation between the health and age of tree. At the same time in sampled stands there are too many weakened and heavily weakened trees; this confirms the data of previous research about the worrisome situation in these mountain fir forests.

Reasons for Disturbance of Fir Stands

At the time of estimation of health of trees, we described evident reasons for their weakening, and noted infected trees that were starting to lose their health. To our surprise we observed mass distribution of cankers from the rust fungus *Melampsorella caryophyllacearum*. This perennial rust fungus is described by many authors (Transhel 1939; Vanin 1955; Peterson 1963, 1965; Scharpf 1993; Reed and Farr 1992). Trees of all ages are susceptible to the parasite. Infected trees have stem or branch yellow witches-brooms, trunk burls, branch swellings, and sometimes spike tops. Indeed, many trees in the Kuznetsk Alatau Mountains have intensive development of the disease and different symptoms of canker can be developed on one tree. About 30 percent of the fir seedlings are infected by *Melampsorella caryophyllacearum*; some of the young trees have systemic canker. One-year fir twigs with microstrobiles often have dry orange needles before more advanced disease symptoms appear. Injured twigs concentrated in the subtop of the crown can make the tree look heavily damaged, especially from the air.

Sixty years ago the portion of trees with rust canker in this region amounted to 2%-7% (Dravert and Popov 1938). Now infection of *Melampsorella caryophyllacearum* in Kemerovo Oblast has reached epidemic proportion: 50%-95% of the trees in each mountain and foothill fir stand had symptoms of cankers. Some examples of the canker's occurrence are shown in table 6. Canker of trees was the dominant reason for tree weakening; doubtful cases of the disease are not included, and the real situation may be more severe than our data indicate. Dense middle-aged stands have many small trees weakened by direct or indirect damage from adjacent trees as a result of competition; we believe these are the primary

[2] *T.N. Otnyukova, V.N. Sukachev Institute for Forest Research, Siberian Branch of Russian Ac. Sci., Academgorodok, Krasnoyarsk, 660036, Russia, personal communication, 1996.*

Table 4.--*Distribution of trees (percent) by age classes.*

Age class years	Plot 1 %	Plot 2 %	Plot 3 %	Plot 4 %	Plot 5 %	Plot 6 %
< 20	--[1]	--	--	--	5.6	--
21-40	2.7		22.7	13.8	16.7	27.3
41-60	10.8	25.0	56.8	79.3	44.4	45.5
61-80	35.1	32.1	18.2	6.9	27.8	18.2
81-100	32.4	25.0	2.3	--	5.6	9.1
101-120	8.1	7.1	--	--	--	--
121-140	5.4	10.7	--	--	--	--
141-160	2.7	--	--	--	--	--
161-180	2.7	--	--	--	--	--

[1]no trees.

Table 5.--*Index of damage (percent) of fir trees of various diameter on sample plots.*

Diameter group	Plot 1	Plot 2	Plot 5	Diameter group	Plot 3	Plot 4
cm	*index, %*	*index, %*	*index, %*	*cm*	*index, %*	*index, %*
8	46	29	43	4	65	63
12	23	33	30	6	55	60
16	21	26	20	8	52	37
20	16.5	22	13	10	40	17
24	29	10	11	12	31	6
28	33	30	22.5	14	26	6
32	26	0	--[1]	16	27	0
36	22	60	--	18	22	0
40	10	--	--	20	27	12
44	30	--	--	22	30	4
60	60	--	--	24	22	0
--	--	--	--	26	0	0
--	--	--	--	28	42	--
--	--	--	--	32	30	--
--	--	--	--	38	0	--
--	--	--	--	52	60	--

[1]no trees.

reasons for supressed trees and seedlings. The group of "other" reasons includes: cankers, damage from insects (*Monochamus urussovi*, *Mindarus abietinus*, *Aphrastasia pectinatae*), wood and root rots (*Phellinus hartigii*, *Heterobasidion annosus*, etc.), cracks from lightning, frost damage, etc. Each individual stand had its own set of damaging factors but in all stands, cankers were distributed in epidemic proportion. This disease was the single generally acting damaging factor. Direct mortality from cankers occurs rather rarely. But stands weakened by cankers are more easily damaged by insects and other destructive agents.

SUMMARY

1. The level of present and recent regional air pollution cannot be the reason for widespread fir forest degradation in Kemerovo Oblast.

2. The present age of fir stands in Kemerovo Oblast cannot be the reason for widespread fir forest degradation.

3. The main agents of forest decline in mountain fir forests of the Kemerovo Oblast are fungi including *Melampsorella*. Weakening of trees by rust fungus, *Melampsorella caryophyllacearum*, makes them less tolerant to other factors. Where weakened stands are distributed across a huge territory, such forests are areas of risk and potential sources of mass outbreaks of insects and other damaging factors.

4. In contrast to Kovalev (1993, 1994) we observed no large areas of dead or dying fir stands in Kemerovo Oblast. It is possible that Kovalev's use of helicopters for visual estimation of forest health led to overestimates of forest degradation because observations were mainly of the upper parts of tree crowns.

5. Ecological catastrophe does not threaten Kemerovo Oblast in the near future. However it could happen in 10-20 years as result of outbreaks of insects in huge areas of risk formed by initial activity of *Melampsorela caryophyllacearum* and other fungi that damage twigs.

Table 6.--*Reasons for weakening of Abies sibirica.*

Reason for weakening	**Distribution of different reasons for tree weakening, %**				
	plot 1	**plot 2**	**plot 3**	**plot 4**	**plot 5**
	%	%	%	%	%
competition	24	43	41	12	--[1]
trees with cankers	7	11	17	--	--
other agents	76	57	59	89	100
trees with cankers	51	40	54	83	86
Total	100	100	100	100	100
trees with cankers	59	51	71	83	86

[1]no trees.

6. It is necessary to continue this research. Our survey of fir stands in Krasnoyarsk Kray and adjacent territories shown a great distribution of rust cankers. It is possible that this phenomenon is more important than was supposed in the beginning.

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EVALUATION OF HEALTH OF *PINUS SYLVESTRIS* TREES NEAR MAIN LOCAL POLLUTION SOURCES OF LITHUANIA

E. Kupčinskienė, J. Nemaniūtė, D. Dailidienė, A. Krasauskas[1]

ABSTRACT.--Recent economic changes at the main factories in Lithuania require proper evaluation of the impact of industrial pollution on conifer stands. The objective of the current research was to estimate pollutant effects on aboveground parts of *Pinus sylvestris* over a 3-year period. NO_2 concentration was estimated as the marker of air quality at the Scots pinė sampling locations. *Pinus sylvestris* trees were examined in autumn 1994-1996 near an oil refinery, a nitrogen fertilizer plant and a cement factory. Comparison of needle surface quality by SEM, wax amount, gravimetric characteristics and assessment of hydrophobic features of needles in some cases revealed significant differences between the sites. Data obtained show the possibility for a low level of pollutants to induce detectable damage in one-year-old needles.

INTRODUCTION

The potential threat of acidic pollutants to natural vegetation in East and Central Europe has been the subject of much speculation during the previous decade. Realistic assessments of the threat posed by these pollutants depend on a thorough understanding of the sources and quantities emitted, their subsequent distribution and, most important, the typical concentrations to which plants are exposed in field environments.

Sulfur dioxide is the most commonly occurring acidic pollutant gas found in Europe. The second most common acidic pollutant gas is nitrogen dioxide (NO_2). NO_x emissions have increased dramatically over the last 50 years and have not decreased over the last 10 years in the same way as SO_2 emissions (Campbell 1988).

Few estimates have been made of NO_2 concentrations except in city centers. In the United Kingdom, rural NO_2 measurements have shown annual mean values in the range of 3-12 $\mu g\ Nm^{-3}$ (Ashenden and Edge 1995). Concentrations in the range of 4.8-9.7 nl $liter^{-1}$ are considered typical for rural regions in Germany (Simonis and Weidner 1988). In a larger survey of about 20 sites covering 9 European countries annual mean concentrations in the range 0.5-14.4 nl/liter have been recorded for 1983-1985 (Schaug *et al.* 1987).

Data concerning rural NO_2 concentrations in Lithuania are quite scarce. Average amount of NO_2 in the atmosphere of the main industrial cities of Lithuania (1984-1988) was 0.01-0.09 mg/m^3 (Kairiukstis *et al.* 1992). Although visual forest tree observations can show the extent of current injury, they give little indication of potential future visible symptoms. They are also to some extent subjective, and there is need to establish objective methods of quantifying the extent of forest damage. Plants may differ widely in the rate of production of their total dry weight as well as separate parts. The dry weight/fresh weight ratio difference between current and 2-year-old needles was found to show a significant relationship to tree damage in areas affected by forest decline (Mehlhorn *et al.* 1988). The largest aerial interface between forest trees and surrounding atmosphere is formed by the plant cuticle, which covers all primary plant organs. Gaseous pollutants have been reported to increase the rate of water loss from excised leaves of trees in controlled conditions (Mansfield and Freer-Smith 1984) and in the field (Huttunen *et al.* 1981). Cuticular waxes have been shown to be solely responsible for the barrier properties of plant cuticles since an extraction of leaf surface waxes with organic solvents increases cuticular permeability by a factor of about 100 to 1,000. Erosion of waxes of the needle surface was observed as a consequence of gaseous pollutants, simulated acid rain, acid mist or acid fog and basic dust (Turunen and Huttunen 1990). The overall erosion rate has usually been related to the level of air pollution and based on the comparison of areas with different emission loads (Tuomisto 1988). Monitoring the fusion of wax tubes provides information about the direct effects of air pollutants on the needles. Air pollution-induced wax erosion resembles accelerated natural weathering of needle surfaces, which differs from mechanical, fungal, and insect damage. In heavily polluted areas wax tubes

[1] *Kupčinskienė E., Assoc.-prof., Department of Botany, Lithuanian University of Agriculture, Kaunas-Noreikiškės, Lithuania 4324;* Nemaniūtė *J., M.Sc. Department of Biology, Vytautas Magnus University, Kaunas, Daukanto 28, Lithuania 3000;* Dailidienė *D., B.Sc., Department of Biology, Vytautas Magnus University, Daukanto 28; Krasauskas A., Prof.-assist., Department of Botany, Lithuanian University of Agriculture, Kaunas-Noreikiškės, Lithuania 4324.*

may start fusing only a few weeks after needle flushing, the rate being 2 to 5 times faster than in clean areas, where it usually starts during the second year of the needles existence. The accelerated erosion of needle surfaces can be detected in all needle age classes and it is the most rapid during winter (Huttunen and Laine 1983). The use of the epicuticular wax structure as an indicator of air pollution has been mainly based on direct observations of changes in physical form as seen with the scanning electron microscope (SEM) or on indirect methods, such as determining the wettability of needles by measuring the contact angles of water droplets on needle surfaces (Cape 1983; Haines *et al.* 1985).

Air pollutants may retard the development of the cuticle and alter the wax synthesis, resulting in a reduced concentration or altered composition of the waxes, both of which lead to a changed physical structure of the waxes (Riding and Percy 1985). Thus, a variety of tests are proposed to evaluate latent injury of trees under air pollution, although, there are many uncertainties left in early diagnosis, and field experiment data are needed to ascertain validity of proposed tools for identification of invisible damage. The major objectives in setting up the survey were to determine concentrations of NO_2 in selected sites situated at different distances from main local pollution sources in Lithuania, as well as to evaluate *Pinus sylvestris* trees under the influence of ambient pollution according to morphophysiological tests proposed to detect latent injury.

MATERIAL AND METHODS

Study Area

Forest sites near an oil refinery, a cement factory, and a nitrogen fertilizer plant – the main local pollution sources in Lithuania--were investigated. During the study period (1994-1996) total emissions from the oil refinery comprised 28.9-31.4 thousand t/year, from the cement factory - 5.8-18.1 thousand t/year, and from the nitrogen fertilizer plant - 4.9-5.8 thousand t/year (data provided by Environmental Protection Ministry of Lithuania). Forest sites on podzols (in few cases on gleyic podzols) mainly in north-east direction from each pollution source were selected. Adjacent opened areas in front of woods were places for aerial NO_2 estimations. Near the oil refinery 4 sites (M1-1, M1-2, M4-1, M4-2) with 43-54 year old trees (further named as middle-aged stands of *Oxalido-Pinetum* type) in 5.5 km transects and 3 sites (M1-2, M2-1, M3) with 15-20 year old trees (further named as young stands of *Myrtillo-oxalido-Pinetum* or *Vaccinio-myrtillo-Pinetum* type) on a 10.5 km line were selected. Related to the cement factory 4 sites (A1-A4) with 40-67 year old trees (stands belonging to *Carico-sphagno-Pinetum* or *Vaccinio-myrtillo-Pinetum* type) on a 10 km transect were examined. Beside the nitrogen fertilizer plant 10 sites (J1-J10) with 15-25 year old trees (stands of *Oxalido-Pinetum* or *Vaccinio-myrtillo-Pinetum* type) on a 21 km transect were chosen to evaluate pollutant effects on pine stands.

Determination of Concentrations of Nitrogen Dioxide

In order to ensure the true monitoring of NO_2 pollution levels caused by the nitrogen fertilizer plant, the cement factory or the oil refinery, all monitors were located well away from other sources of pollution. As far as possible, that meant points at least 1 km from major trunk roads, towns and 100 m from minor roads and buildings. The precise location of sites depended on the availability of secure sites. Care was taken to place monitors away from obvious obstructions which could shield air streams from any direction, or trees and hedgerows which could filter pollutants from the air being sampled. All air samplers were placed on wooden stakes at about 1.5 m above ground level. There were 17 (JJ1-JJ17) points in sites near the nitrogen fertilizer plant, 18 (MM1-MM18) points in sites near the oil refinery and 9 (AA1-AA9) points in sites near the cement factory; those are listed in Table 1. The exposure of tubes was performed during February-March 1994. The diffusion tubes comprised 7.2 x 1.12 cm diameter acrylic tubes fitted with airtight polythene end-caps and were supplied by Gradco International Limited of Winchester, UK. Two acid-cleaned stainless steel mesh discs were soaked in 50% v/v triethanolamine (TEA)/acetone solution and, after allowing the acetone to evaporate, fitted inside one end of the tube. These discs acted as collectors of NO_2 during exposure period. TEA is highly efficient in the absorption of NO_2. The resultant TEA-NO_2 complex is stable so that trapped NO_2 can be stored for several months prior to analysis (Ashenden and Bell 1989). During exposure, the polythene end-cap at the opposite end of the tube to the mesh collector was removed and the tubes mounted vertically, with the open end pointing downwards to prevent the entry of rain and dust particles. At all times except during exposure, tubes were kept in sealed polythene bags. Duplicate tubes were exposed at each point for a period of 30 days, after which the open ends of the tubes were re-sealed and the tubes stored for analysis. Nitrogen dioxide absorbed by the collectors was determined colorimetrically as NO_2^-. The nitrite ion diazotized

Table. *NO_2 concentration (in ppb) near the main local pollution sources of Lithuania (August-September, 1994)*

Characteristic of point	Number of point	Tree site*	Latitude	Longitude	Distance (km)**	NO_2 ppb
North direction	MM 01	M1-1	56°24'	22°10'	2.0	3.8
North direction	MM 02	M1-1	56°24'	22°10'	2.7	3.9
North direction	MM 03	M1-4	56°25'	22°11'	3.7	3.9
North-East direction	MM 04		56°24'	22°12'	2.2	4.1
North-East direction	MM 05		56°23'	22°12'	2.1	3.7
North-East direction	MM 06		56°24'	22°12'	3.7	4.0
East direction	MM 07	M1-2	56°22'	22°07'	2.1	3.8
West direction	MM 08		56°24'	22°05'	3.7	3.6
West direction	MM 09		56°24'	22°07'	6.6	3.6
West direction	MM10		56°22'	22°09'	3.0	3.6
South direction	MM 11		56°22'	22°14'	1.5	3.7
East direction	MM 12	M2-1	56°23'	22°14'	3.9	3.0
East direction	MM 13	M2-2	56°23'	22°14'	3.9	2.9
East direction	MM 14		56°23'	22°15'	3.9	2.9
East-South direction	MM 15		56°22'	22°15'	5.4	2.7
East-South direction	MM 16		56°21'	22°12'	5.8	2.8
South-East direction	MM 17		56°20'	22°17'	6.3	2.9
South-East direction	MM 18	M3	56°19'	24°21'	8.8	2.8
West direction	JJ 01	J1	55°05'	24°21'	0.5	4.1
North-West direction	JJ 02	J2	55°05'	24°21'	0.5	4.0
North-East direction	JJ 03	J3	55°05'	24°21'	0.5	4.1
East direction	JJ 04		55°05'	24°22'	0.5	4.3
North direction	JJ 05	J4	55°07'	24°28'	2.5	3.0
East direction	JJ 06	J6	55°06'	24°30'	5.0	2.4
East direction	JJ 07	J5	55°07'	24°32'	6.0	2.3
East direction	JJ 08		55°07'	24°32'	8.5	2.5
North-East direction	JJ 09	J9	55°09'	24°30'	15.0	2.1
North-East direction	JJ 10	J8	55°12'	24°33'	19	2.5
North-East direction	JJ 11		55°12'	24°33'	18	2.3
North-East direction	JJ 12		55°11'	24°34'	17	2.1
North-East direction	JJ 13		55°10'	24°37'	17	2.5
North-East direction	JJ 14		55°11'	24°39'	21	2.2
North-East direction	JJ 15	J10	55°10'	24°39'	22	2.2
East direction	JJ 16		55°04'	24°30'	8	2.3
East direction	JJ 17		55°04'	24°32'	12	2.2
North direction	AA 1		56°11'	20°54'	0.5	3.3
East-South direction	AA 2	A1	56°19'	23°55'	0.5	2.9
East-South direction	AA 3	A2	56°19'	23°55'	0.5	2.7
East direction	AA 4	A3	56°19'	23°56'	1.5	2.3
East direction	AA 5		56°20'	23°56'	3.8	2.0
East direction	AA 6		56°18'	23°00'	7.8	1.5
East direction	AA 7		56°18'	23°05'	13.0	1.6
North direction	AA 8		56°23'	22°59'	8.5	2.0
North direction	AA 9	A4	56°24'	22°59'	11.5	1.9

* *Pinus sylvestris* site located in the immediate vicinity of points for exposure of NO_2 tubes

** Point distance from the pollution source

sulfanilamide and the resulting salt were coupled with *N*-1-naphthylethylene-diamine-dihydrochloride (NEDA) to give a purple-red azo dye. The absorbance of the samples was measured at 520 nm (Ashenden and Edge 1995).

Sampling of Trees

Scots pine *(Pinus sylvestris* L.), the most common forest tree species in Lithuania, was selected for study. At each site eight apparently healthy trees were chosen for investigations. The selected trees were assessed visually according to defoliation, dechromation and needle retention (Nemaniute *et al.* 1997). Sampling was done in September 1994-1996. Material was collected from the middle part of the crown of young age trees or from the part of the crown 6m-8m above ground. An extending "squirrel pruner" was used to cut branches. Needles mainly from one-year-old (c+1) shoots were tested and for some tests current year (c) shoots were used. At each time all type estimations were taken from the same branch. In all, 168 trees at 21 sites were sampled each year. Sites were always sampled in the morning. All specimens from the nitrogen fertilizer region were taken in one day. One week later test material was collected from both the cement factory and oil refinery zones.

Epidermal Water Loss and Gravimetric Characteristics

Leaf drying rate measurements were made for excised needles. After transportation to the laboratory, twigs were cut from the branch and cut ends placed in water to allow saturation during night. Great care was taken when removing needles from the shoot that the leaves would not be damaged, as cut or broken needles completely ruin the result. Needles were removed using a small scalpel blade, cutting the bark at the needle base. Subsequently the bases of cut needles were coated in petroleum jelly to prevent water loss from the open wounds and then 20 needles from each shoot were placed into labeled open aluminum plates on a laboratory bench. Each batch of needles was weighed immediately after being excised from the shoots and then at intervals over that day and the following days. The temperature and humidity could not be controlled during the period of these measurements, but the needles from different sites were all exposed to the same conditions. In addition, several other measurements were made of foliar properties likely to have responded to pollutants.

The ratios of dry/fresh weight for needles of different age classes have been proposed as indicators of 'decline' (Mehlhorn *et al.* 1987). Dry weight/fresh weight ratios (D/F) were obtained by weighing out 20 fresh needles, and dry weights were estimated after drying for 3 hours at 105° C. Dry weight/fresh weight ratio difference (R1-R0) was calculated by subtracting the ratio obtained for current-year needles from that for one-year-old needles.

Contact Angles

A test of water-droplet contact angles with the leaf surface was carried out using a method described by Cape (1983). Contact angle measurements were made at room temperature, using distilled water. 15-20 needles of every age class from each of 8 trees per site (totally 120-180 measurements per site) were analyzed. A 0.2µl droplet was placed on the center of the abaxial surface of needles by means of a syringe and viewed using a binocular microscope (*50) with protractor graticule. As far as it was possible the water surface interface was viewed tangentially. Contact angle was assessed immediately after the droplet was placed on the needle. Measurements were carried out using repeating cycle of analyzing one branch per each site to avoid artifacts due to surface quality changes per time. Needles were tested within 5 days of collection.

Chloroform - Extractable Wax

To determine extractable wax, 10 needle pairs were shaken with 40-ml chloroform ($CHCl_3$) for 15 s as described by Cape *et al.* (1988). Chloroform extracts were decanted through a small Buchner funnel under gravity into a sealable bottle. The solution was transferred to a pre-weighted aluminum foil dish, and the remaining solvent was allowed to evaporate in a fume cupboard. Samples were weighed until a stable weight was reached to ensure the complete evaporation of solvent. The washed needles were placed on aluminum dishes for drying and later used for dry weight determination. Amounts of wax were expressed in terms of percentage of dry weight (d.w.).

Epicuticular Wax Morphology

To prevent possible mechanical damage at each site needles were excised from the middle part of the one-year-old shoots. Only green undamaged needles were selected. After transportation, needles were kept in a freezer until the assessment time. Surface moisture was allowed to evaporate at room temperature for several days. The middle part of undamaged needles was secured with the abaxial surface uppermost. The samples were coated with

silver using sputter equipment (VUP-4K) and investigated with a JEOL-JSM-IC25S scanning electron microscope (17 kV accelerating voltage). Representative areas of each needle were viewed at magnifications of 100X to 10,000X. Areas investigated were the epistomatal chamber, slopes of the epistomatal chamber and the surface around stomata, called 'interstomatal area'. Needle surfaces were rated according to the following classes (Turunen *et al.* 1992): class I - distribution of crystalloid waxes 100%; class II - distribution of crystalloid waxes 71%-99%; class III - distribution of crystalloid waxes 31%-70%; class IV - distribution of crystalloid waxes 1%-30%; class V - distribution of crystalloid waxes 0%.

Statistical Analysis

ANOVA analysis was used to evaluate significance of differences between data sets in sites.

RESULTS AND DISCUSSIONS

Emissions of Nitrogen Dioxide

NO_2 as a common pollutant for all 3 regions near the factories was selected despite the fact that it is not of greatest importance according to emission amounts. The diffusion tubes were used because they are a cheap and reliable method of collecting data on average concentrations of NO_2 (Ashenden and Bell 1989). Results are shown in table 1. NO_2 values estimated were 2.1-4.3 ppb in the area effected by the nitrogen fertilizer plant, 2.7-4.1 ppb in points near the oil refinery and 1.5-3.3 ppb in the region of the cement factory. Data from the present survey show that NO_2 is found as a pollutant in all points investigated. In our study concentrations of NO_2 were found to vary across transects and to some extent were greater at the sample points near the pollution sources. The levels found in 1994 are comparable with those documented in a 1993 survey (Kupcinskiene *et al.* 1996). Our results show similarities with data collected by other researchers in Lithuania in the previous decade (Kairiukstis *et al.* 1992).

The concentrations defined in our study were low in comparison with the rural parts of other countries: the lowest concentrations of NO_2 throughout Wales had a background level below 4 nl litre^{-1}, annual mean concentrations of 12.2 and 11.6 nl litre^{-1} were registered at 2 sites in England and 1.7 nl litre^{-1} NO_2 at a site in Northern Ireland, while concentrations in the range 4.8-9.7 nl litre^{-1} are considered typical for rural regions in Germany (Simonis and Weidner 1988; Ashenden and Bell 1989).
Lower NO_2 concentrations of background values and ones near the factories found in our study could be due to the nearly twofold decreased productivity of industrial units of Lithuania compared to the previous decade and the overall lower density of traffic in Lithuania compared to the countries in West Europe. In our experiment the points were specially chosen to avoid the influence of local sources of pollution and thus the concentrations measured are considered to be minimum levels. Furthermore, all concentrations quoted are for NO_2 and no account has been taken of the other oxides of nitrogen, in particular NO. In terms of total NO_x pollution, it seems possible that the concentrations may be up to 50% greater than that shown in the table. In addition, it should be stressed that the concentrations described are 'mean concentrations of NO_2'. In a field situation, pollutant levels fluctuate substantially and concentrations of 10-20 times the monthly mean may be expected to occur for short periods. There are some difficulties when predicting the effects of industrial emissions of NO_2 on forest stands and other types of vegetation. On a broad scale, NO_2 will contribute to soil acidification if nitrogen is in excess of vegetation requirements. Because near the industrial units NO_x always exists together with the other pollutants, it is the responses of plants to the pollutants mixture should be considered. Combination of NO_2 and SO_2 produces a highly phytotoxic atmosphere since they often interact synergistically in inducing plant responses. Free-Smith and Mansfield (1987) have shown exposures to 30 nl litre^{-1} SO_2 and NO_2 to decrease the resistance of Sitka spruce to frost injury. Similarly, 10 nl litre^{-1} of each NO_2 and SO_2 has been found to reduce the capacity for stomata to close under conditions of severe water stress (Mansfield *et al.* 1988). In 1988 critical level for vegetation of 10 ppb NO_2 and in 1992 of 30 μgm^{-3} NO_x (about 15 ppb) as an annual mean were proposed for adverse ecophysiological effects (Ashenden and Edge 1995).

Water Loss from Excised Needles

The experiments reported here were designed to investigate the ability of detached one-year old needles to conserve water after prior exposure to pollution of different level. Results from sites near the nitrogen fertilizer plant are shown in figure 1.

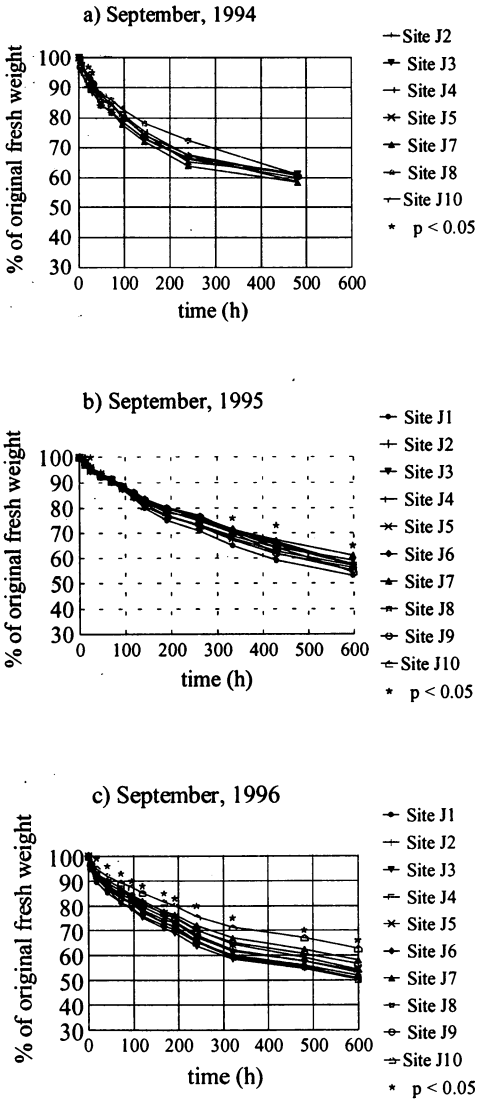

Figure 1.--*Annual changes in fresh weight of one-year-old needles over time after excision from trees growing at different distances from the nitrogen fertilizer plant.*

Estimates of drying of excised leaves were taken at 480 h-600 h intervals. Despite the uncontrolled temperature conditions in which needles were observed, the extent of water loss during the whole recording time was similar in all 3 years of experiments and in each year ranged between 38%-50%. Generally, quicker water loss was documented during the first 30 h, and later drying occurred much slower. This could be explained by intensive stomatal closure at the beginning of the observation period. According to 1994 data significantly quicker water loss was characteristic for J2 (0.5 km distance from the plant) site needles compared to J4, J7 or J10 sites after 6 h, 22 h and 30 h period of being excised, while at other time intervals differences were negligible. Few bigger differences between stands were detected in 1995: at the beginning of exposure of needles the most rapid evaporation was common for site J8 ($p<0.05$ for 12h, 24h) and later for site J1, which was close to the factory ($p<0.05$ for 334h, 430h and 598h). Determination of water loss of detached needles in 1996 year showed more significant differences between sites. During the 43 h-598 h interval site J3 (closely related to the factory) needles dried up at significantly higher speed compared to J10. Comparison of 3 years data for the sites near the nitrogen fertilizer plant did not reveal any site with a permanent tendency for significantly increased or decreased water loss from excised needles. No relationships were found between site distance from the factory and water state of examined needles.

Results from drying of one-year old needles from the oil refinery sites are shown in figures 2 and 3. Overall characteristics of curves obtained in this region resemble those from the nitrogen fertilizer plant. According to our data, at the sites closest to the factory there was a noticeable decline in the rate of water loss, suggesting similar stomatal closure compared to the needles from the other sites. Comparison of middle-aged stands (fig. 2) located at different distances showed only a few transitional effects in the 3-year study period ($p<0.05$ after 384 h in 1995 and after 17h, 25h in 1996). No one site showed a clear pattern of permanently increased or decreased water loss from detached needles. A different picture was obtained from analyzing young stands. The amount of needle water lost during the interval of observation was equal to 50%-55% in 1994, 42%-50% in 1995, and 32%-37% in 1996. In the 5 h to 360 h period significant differences were found among sites, with the quickest water removal from the needles belonging to site M1-2 (closest to the oil refinery stand). Similar results were obtained in the 1995 assessment, where at all time points (6 h-598 h) needles from site M1-2 dried at the highest speed ($p<0.05$) compared to sites M3 or M2-1. In the last year of observation, no significant differences among young stands were documented, although site 1-2 showed results in the same direction as in previous years. Behavior of the needles from site 3M had an inconsistent pattern throughout the whole 3-year experiment, while M2-1 samples were in an intermediate position or lost water at the slowest speed. Results obtained from the young stands near the oil refinery could be explained by either higher uptake of water by roots or lower water retention in the needles including increased cuticle permeability of the closest to the factory M1-2 site trees.

Estimates of water loss from excised needles were done near the cement as described above (fig.4).

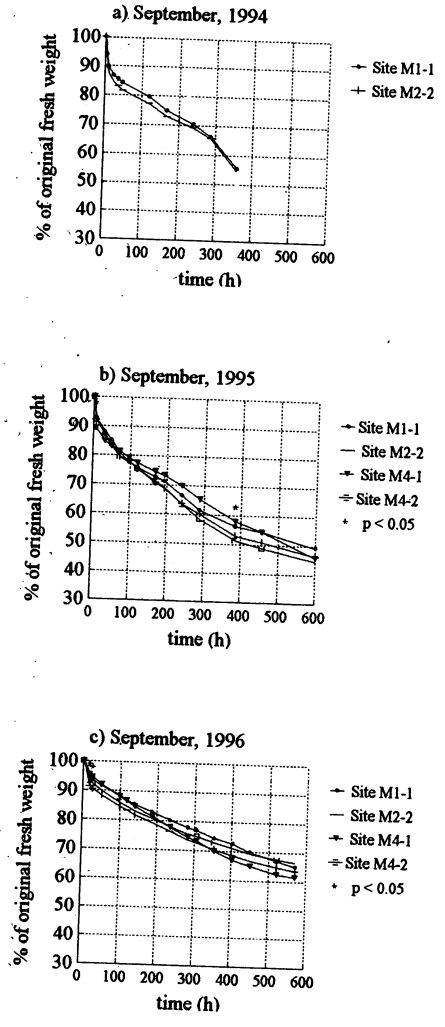

Figure 2. *Annual variation of alternation in one-year-old needles fresh weight over time after excision from middle-aged trees growing at different distance from the oil refinery*

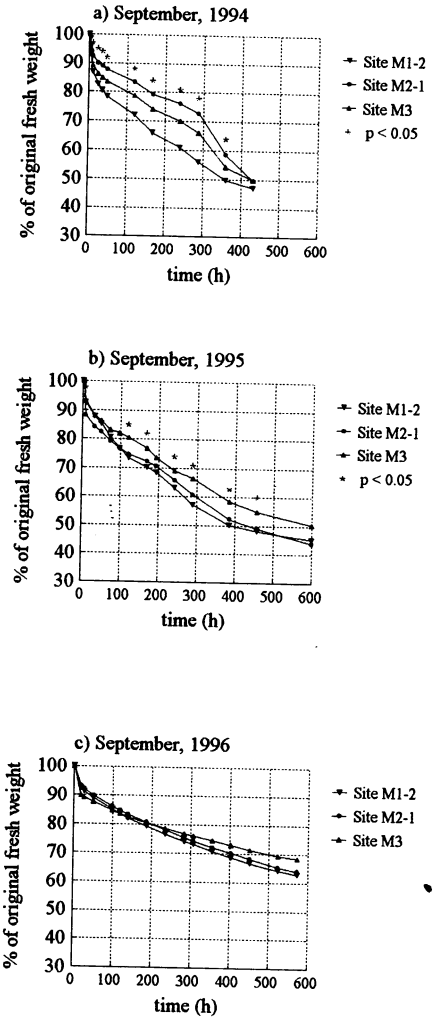

Figure 3. *Annual variation of alternation in one-year-old needles fresh weight over time after excision from young tress growing at different distance from the oil refinery*

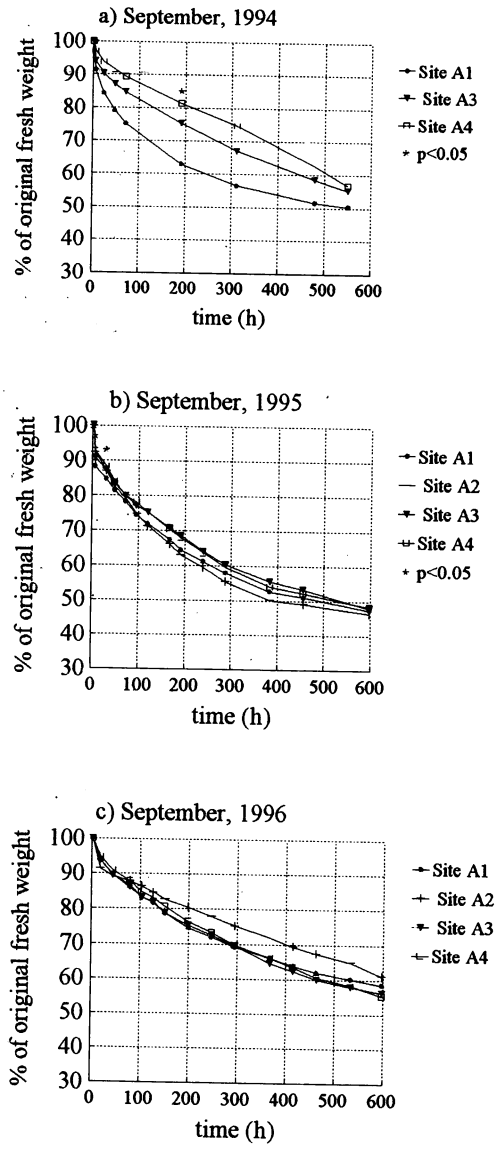

Figure 4. *Annual variation of alternation in one-year-old needles fresh weight over time after excision from trees growing at different distance from the cement factory*

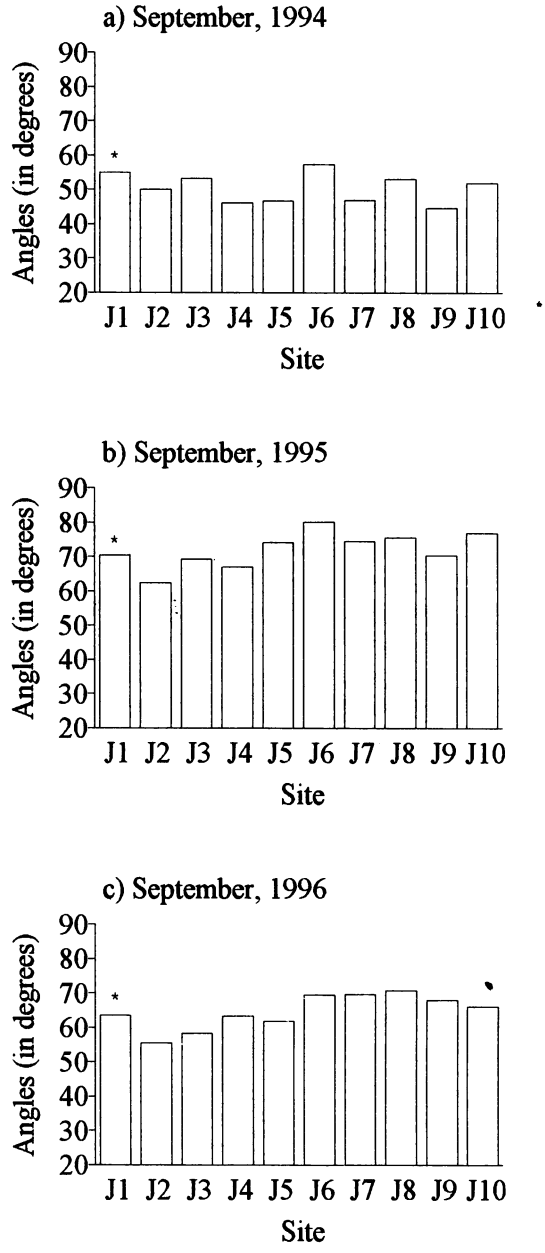

Figure 5. *Contact angles (degrees) of water droplets placed on one-year-old needles from trees growing at different distance from the nitrogen fertilizer plant. * - significant difference ($p<0.05$) among sites*

Extent of water loss was alike in the sites near the oil refinery and nitrogen fertilizer plant. After a 550 h interval the weight of A1-A4 detached needles decreased 42%-50% (1994), 52%-54% (1995) and 39%-45% (1996). At most sampling times differences among sites were not significant although showed the same tendency: the highest speed of drying was common for site A1 (1994), sites A1, A2 compared to A3, A4 (1995) and sites A1, A3 compared to A2, A4 (1996). Significantly quickest drying was documented for A1 needles after 6h and 192h intervals (1994), 6 h and 30 h intervals (1995). A tendency for quicker water loss from A1 needles in the 1994-1996 autumn period coincides with results obtained in July-August 1994 by our laboratory (unpublished data). Severe drought in summer 1994 could cause significantly faster drying of needles from the A1 site, while later in the period which was the subject of the present paper there were no climatic extremes. It is important to mention that in the selected regions all differences among the sites obtained for one-year-old needles were more expressed for current year needles.

Dry Weight-Fresh Weight Ratios

Dry weight-fresh weight ratios (D/F) are good indicators for needle water balance. Under equal water supply from the soil differences in this parameter could be related to drying or wetting through the needles and subsequently higher defoliation. According to the literature both effects can be evoked by air pollutants (Mehlhorn *et al.* 1988). In the sites near the nitrogen fertilizer plant D/F ranged between 0.42-0.45 (1994), 0.36-0.39 (1995), 0.39-0.45 (1996) with significant differences ($p<0.05$) among J1-J10 sites during all 3 years. Among middle-aged tree sites (M1-1, M2-2, M4-1, M4-2) near the oil refinery D/F values were as follows: 0.46 (1994), 0.37-0.39 (1995), 0.34-0.42 (1996). D/F estimated for older trees were similar to those registered for young (M1-2, M2-1, M3) stands: 0.45-0.47 (1994), 0.37-0.38 (1995) and 0.33-0.40 (1996). Detected differences in water loss among young trees near the oil refinery can not be explained by negligible D/F deviations and suggest the possibility of surface damage rather than disturbances in needle supply of water. The scope of D/F (0.46-0.47 in 1994, 0.38-0.40 in 1995, and 0.42-0.44 in 1996) for A1-A4 sites resembled the other regions. In all cases gravimetric parameters did not differ significantly among the oil refinery sites as well as between the cement factory sites. D/F obtained from various regions showed similarities in each year, while annual variations were greater: higher values were common for 1994 and 1996 and lower ones in 1995. Generally dry weight-fresh weight ratios were not related to the speed of water loss from excised needles.

Calculations of dry weight/fresh weight ratio differences between one-year-old and current year needles (R1-R0) revealed significant variations (0.029-0.055; $p<0.05$) among J1-J10 sites only in 1994 autumn after severe drought, while in subsequent years deviations were negligible (0.022-0.039 in 1995, 0.029-0.059 in 1996). R1-R0 varied insignificantly among sites of young trees (0.04-0.05 in 1994, 0.02-0.03 in 1995, 0.03-0.04 in 1996) as well as among middle-aged stands (0.04-0.05 in 1994-96) near the oil refinery. Sites near the cement factory (A1-A4) differed according to R1-R0 only in 1994 (0.032-0.047; $p<0.05$). Insignificant variation was registered in subsequent years (0.038-0.044 in 1995, 0.027-0.050 in 1996) similar to stands near nitrogen fertilizer plant. Annual and regional variations of R1-R0 were less expressed compared to D/F values. Both D/F and R1-R0 values did not have relation to water loss data a well as to defoliation or aerial NO_2 concentrations. Small differences among sites in D/F and R1-R0 indicated at least equal needle supply with water hereby differences obtained examining water loss could be related to some type of surface damage including disturbances in stomatal closure.

Polluted plants may have reduced tolerance to water stress. The variables of interest are stomatal behavior and cuticular transpiration. Some kind of stomatal disruption might be expected to occur in polluted conditions since guard cells and their subsidiary cells are in an exposed position. Removal of leaves of *Aesculus hippocastanum* and weighing them at intervals after excision has shown that leaves from a polluted area dried faster than those taken from an unpolluted area, suggesting either a loss of capacity for stomatal closure, or increased cuticular transpiration (Godzik and Piskornik 1966). Interactions between the pollutants SO_2 and NO_2 and water stress have been found in *Betula pendula* and *B. pubescens*. Following exposure for 40 days to 20 ppb SO_2 + 20 ppb NO_2, the ability of excised leaves to conserve water under conditions of limited availability was considerably reduced (Wright *et al.* 1986). A clear dose-response relationship was demonstrated for leaves previously exposed to a number of SO_2 and NO_2 concentrations. The authors suggested that exposure to SO_2 and NO_2 had increased the permeability of the cuticle to water so that under conditions of water-stress the maintenance of turgor was made more difficult. Various gaseous and aqueous pollutants have also been reported to increase the rate of water loss from trees in controlled conditions (Mansfield and Freer-Smith 1984, Mengel *et al.* 1989, Barnes *et al.* 1990) and in the field (Huttunen *et al.* 1981). Several other

papers indicate either a reverse effect or no effect of pollution on stomata (Neighbour *et al.* 1988).

Multicomponent pollution gradients to which different trees are exposed in uncontrolled conditions can cause synergistic effects. Lack of clearly expressed permanent effects of higher pollutant concentrations on water loss from excised needles in our experiment could be due to quite a low level of pollution: There was evidence that the threshold concentration for damage after the period of exposure was at or below 40 nl l^{-1} $SO_2 + NO_2$, while the lowest pollution treatment of 20 nl l^{-1} of both gases hardly affected the transpiration rate of the plants (Neighour *et al.* 1988). Under water stress conditions, the trees that had been exposed to acidic fog showed significantly higher transpiration rates than the control trees (Mengel *et al.* 1989). The other possible reason could be moderate climatic conditions without expressed drought periods at sampling times.

Needle Wettability

Data for nitrogen fertilizer plant sites are shown in figure 5. Contact angles on one-year-old needle surfaces significantly varied ($p<0.001$) from 44.7 to 57.4° (1994), from 62.3° to 80.1 ° (1995) and from 55.5° to 70.7° degrees (1996). In 2 of 3 years the J2 site showed the lowest value and in all 3 years the J6 site had the highest estimate. Site J1, located near the factory against prevailing wind direction, had higher values compared to the sites J2 and J3, which were adjacent to the plant but situated in the opposite direction. There was no tight relationship between values of contact angles on the needles and NO_2 concentration in the sites or site distance from the nitrogen fertilizer plant.

Figure 6 represents data obtained estimating wettability of needles taken from sites near the oil refinery. In 1994 contact angles ranged from 55.6° to 63.4 ° ($p<0.001$) for young stands and from 45.1° to 61.5° ($p<0.001$) for middle-aged stands. The next year wettability was lower for all sites and varied from 66.0° to 70.5° ($p<0.001$) for young stands and from 65.9° to 68.2° degrees (non-significant differences) for middle-aged stands. In 1996 contact angles in all sites were the highest compared to the other year results. Estimated degrees ranged from 72.2° to 78.2° ($p<0.001$) for young stands and from 64.8° to 73.7° ($p<0.001$) for middle-aged stands. Comparison of mean values obtained from all young tree and from all middle-aged tree needles shows significantly lower contact angles of water droplets on the one-year-old needles of older trees.

For young stands wettability varied with each year and did not have a relationship to site distance from the pollution source, while among middle-aged stands the lowest values were always documented for M1-1, the site closest to the oil refinery where the highest concentration of NO_2 (table) and the highest defoliation were detected.

Similar evaluations of contact angles on the needles from the cement factory sites (fig. 7) revealed significant differences ($P<0.001$) among stands in all 3 years, with variations in the interval of 27.3 ° - 42.0 ° in 1994, 45.6 ° -56.7 ° degrees in 1995 and 47.3 ° -58.5 ° in 1996. During the whole observation period site A1 had the lowest values of contact angles of water droplets on the one-year-old needles compared to the other sites.

Different years' data from the nitrogen fertilizer, oil refinery, or cement factory sites showed the lowest values of contact angles on the needle surface in 1994. As expected, the contact angles on older needles were significantly smaller compared to the current year's growth. Low values of contact angles on one-year-old needles are in agreement with short retention of needles (1.8-3.0 years) on the sites of sampled trees (Nemaniute *et al.* 1997). Comparison of data collected in 3 different regions near the pollution sources (see figs. 5, 6, 7) revealed the lowest values of contact angles for the site A1 located near the cement factory. Needle wettability had relationships to the site distance, aerial concentrations of NO_2 and defoliation only for older stands: indicative differences were detected for middle-aged stands near the oil refinery and the cement factory. In these cases the analysis of variance showed a significant effect of industrial emissions, with smaller contact angles from the trees exposed to the higher pollutants' concentrations. There was no detectable effect of the nitrogen fertilizer plant on the water holding capacity of the needles. Inconsistent values of water droplet angles among various young tree sites over the 3 year period may be caused by climate fluctuations which can effect the presence or absence of leaf surface microorganisms. On the other hand, emissions can produce an indirect effect through its control of leaf surface microorganisms; the results could be interpreted in terms of pollution induced inhibiting or stimulating the growth of either hydrophilic or hydrophobic fungi or algae (Cape *et al.* 1995). Greater vitality of young tree needles could cause more intensive exudation and consequently denser population of epiphylic microorganisms. Thus, higher variation in needle wettability may be caused by interaction of 3 variables: climate fluctuations, pollutants and surface microorganisms.

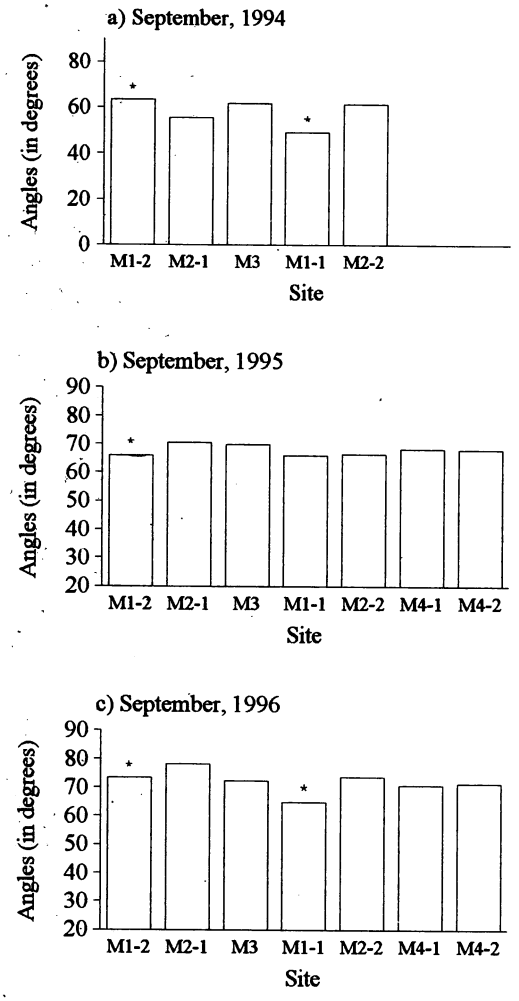

Figure 6. *Contact angles (degrees) of water droplets placed on one-year-old needles from trees growing at different distance from the oil refinery.*
** M1-2 - significant difference ($p < 0.05$) among young stands. * M2-1 - significant difference ($p < 0.05$) among middle-aged stands*

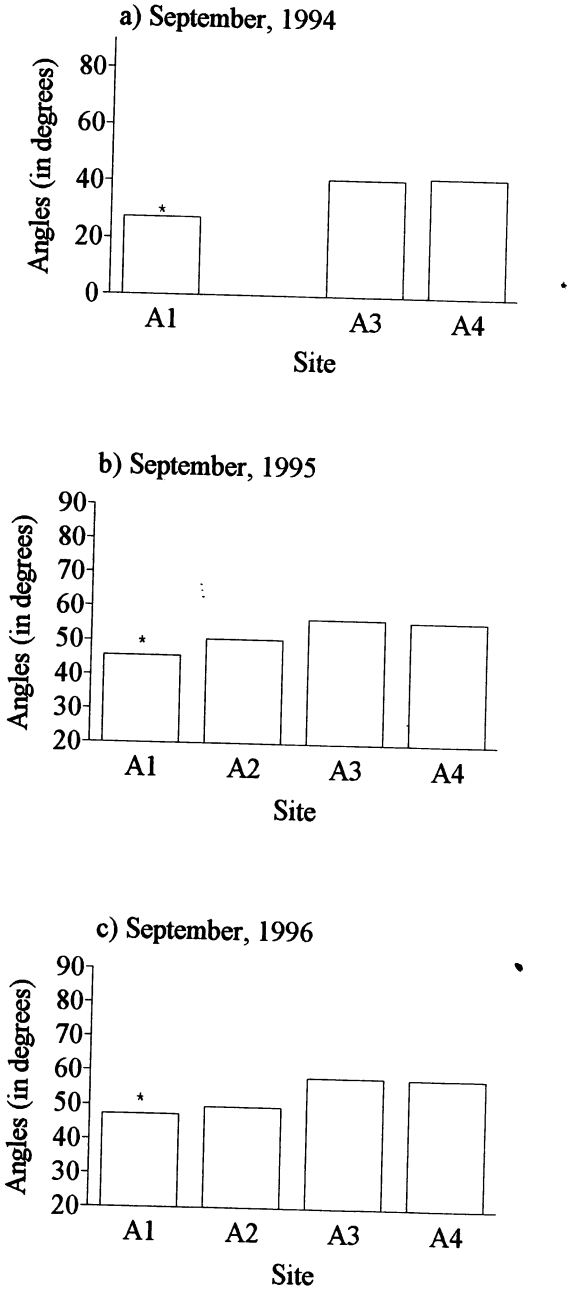

Figure 7. *Contact angles (degrees) of water droplets placed on one-year-old needles from trees growing at different distance from the cement factory. * - significant difference ($p<0.05$) among sites*

Increases in the wettability of needle surfaces could have far reaching physiological consequences for the whole tree (Barnes *et al.* 1990). Changes in wettability may affect the way in which individual needles and the whole canopy intercept and retain precipitation. Increased wettability of the needle surface induced by gaseous and aqueous pollutants can also exacerbate the physical effects of frost on the needle surfaces in winter and the uptake of pollutants from the atmosphere and provide a favorable microclimate for the development of fungal pathogens.

Microscopic Evaluation of the Needle Surface

In September 1995 the quality of the surface of one-year-old needles was evaluated under SEM (fig. 8). The mean distribution of crystalloid waxes in stomatal areas of needles from sites near the nitrogen fertilizer plant ranged from 3.5 to 3.9 with the significantly ($p<0.001$) highest value (i.e., the highest structural degradation of wax) for site J2 closely related to the factory. In our previous SEM observations, where more precise classification of wax structures was used, J2 showed significantly worse surface conditions compared to the J8 site or the relatively 'clean' Kacergine region (Kupcinskiene 1991, 1992; Kupcinskiene *et al.* 1996). For young tree sites near the oil refinery the average surface class varied from 3.6 to 4.2 ($p<0.005$) while for middle-aged stands from 3.6 to 4.6 ($p<0.001$). Surfaces of one-year-old needles in the immediate vicinity of the cement factory were more degraded than needle surfaces from adjacent sites. Mean surface class values significantly varied from 3.5 to 3.9 with the biggest area of eroded wax among needles from site A1 ($p<0.005$). Thus, in all regions with the local pollution sources needle surface observations revealed significantly the worst coverage in sites (A1, J2, M1-1, M1-2) closest to the factories. Results concerning structural degradation of wax in sites near the factories in most cases agreed with the lowest values of contact angles as well as with the greatest defoliation of sampled trees and elevated to a small extent concentrations of aerial NO_2. Among the worst condition sites from various regions the highest structural degradation of wax was detected at site M1-1 adjacent to the oil refinery which is the biggest local polluter in Lithuania. It should be mentioned that site A1, located near the cement factory, had the lowest values of needle surface damage according to SEM and the highest values of surface injury according to very low contact angles of water droplets. Such discrepancy could be explained by factors influencing wettability of A1 needles. Presumably both dust particles as well as erosion of wax contributed to the increased moisture on the surfaces. SEM data obtained for 1994 or 1996 year in most cases resembled 1995 year results discussed above.

Chloroform-soluble Wax Amount

Results on wax amount on the surface of needles (August 1995) are shown in figure 9. There were significant differences among nitrogen fertilizer sites, although no effects were registered of site location relative to the factory on the amount of wax relative to the needle dry weight. No oil refinery effect could be detected on the total amounts of surface wax. For sites near the cement factory amount of wax and dust significantly decreased (correspondingly $p<0.001$; $p=0.001$) with increasing site distance from the pollution source. Quantity of wax extracted from the needle surface ranged between 1.7% d.w. and 3.1% d.w. for sites in the nitrogen fertilizer plant region, between 1.5% d.w. and 1.7% d.w. for sites in the oil refinery region and between 1.5% d.w. and 2.7% d.w. for cement factory sites. These results agree with data from literature, where quantity of waxes on pine needles varies from 0.5% to over 2% dry weight. Data concerning changes in wax amount under conditions of simulated or field pollution are very contradictory (Cape *et al.* 1988, Huttunen 1994) and our results obtained are in support of it. Pollutant or pH treatment showed a tendency to reduce the concentration of waxes: acid rain-treated Scots pine needles had 50% less epicuticular waxes than in the water controls; the wax quantity of needles of *Picea abies* was considerably reduced by fumigation with 300 $\mu g\ m^{-3}$ ozone while UV radiation has been observed to increase epicuticular wax production in plants (Huttunen 1994). According to our results wax synthesis of the needles may have been changed so that the pines growing in the vicinity of the cement factory protect the needles by producing bigger amount of waxes.

CONCLUSIONS

1. Some gradients of aerial NO_2 concentrations were found between sites located in the immediate vicinity of pollution sources and those situated farther from the factories.

2. According to water loss from excised needles or according to dry weight/fresh weight ratio differences between one-year-old and current year needles in most cases no relationships were found between site distance from the factory and water state of examined needles.

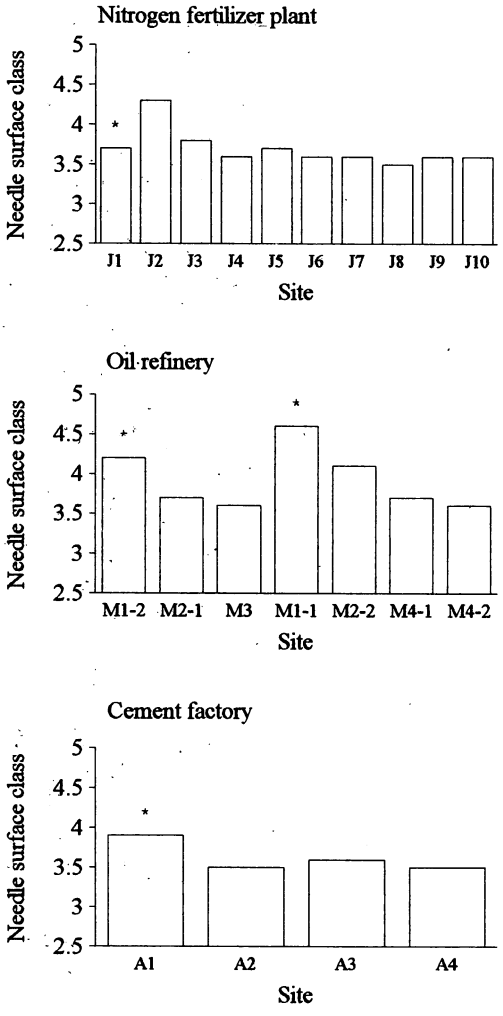

Figure 8. *Mean values of surface class of needles from trees growing at different distance from the pollution source (September, 1995). * - significant difference ($p<0.05$) among sites*

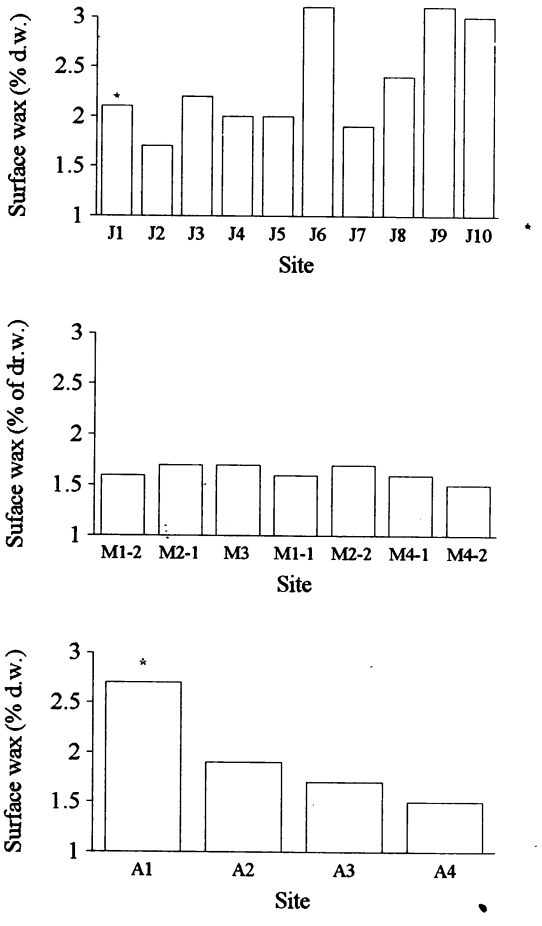

Figure 9. *Wax amount (% of dry weight) on the needles from trees growing at different distance from the pollution source (August, 1995). * - significant difference ($p<0.05$) among sites*

3. Wax amount on the needle surface was significantly increased in trees growing in the neighborhood of the cement factory.

4. Needle wettability had relationships to the site distance, aerial concentrations of NO_2 and defoliation only for older stands: indicative differences were detected for middle-aged pine stands near the oil refinery and the cement factory.

5. Needle surface structure investigated by scanning electron microscopy was the most sensitive parameter among all characteristics tested.

6. According to various parameters investigated, the worst tree condition among sites near the oil refinery was distinguished at site M1-1, which site A1 was the poorest among the cement factory stands selected for study.

ACKNOWLEDGEMENTS

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TEMPORAL MODELING OF FOREST DAMAGE EXPANSION AROUND THE 'SEVERONICKEL' SMELTER ON THE KOLA PENINSULA, NORTHERN RUSSIA

Olga Rigina and Olle Hagner[1]

ABSTRACT.--The central Kola Peninsula, Northern Russia, is one of a few places in the world with large-scale forest decline attributable to a single source of pollution. The study objectives were to estimate expansion of the forest damage area around the 'Severonickel' smelter. Landsat-MSS summer images from 1978, 1986 and 1992 were used in the change-detection analysis. The method applied was iterative histogram matching and subsequent subtraction. A dynamic model for forest damage under long-term SO_2 exposure was applied for the 1960-1996 period. The long-term annual expansion of the damaged area in three classes: visible-, severe-damage and forest-death area was estimated to be 60.3 km^2, 17.8 km^2 and 8.8 km^2, respectively.

INTRODUCTION

Environmental Problems

The central Kola Peninsula in Northern Russia is one of a few places in the world with large-scale forest damage attributable to a single pollution source. The 'Severonickel' smelter is one of the largest in the world and has been in operation since 1938. Due to high-sulphur ores and primitive emission filters, the operation has caused severe environmental destruction in the region. Within a 100-km gradient from the smelter there is a transition from technogenic barren areas to undisturbed ecosystems. The major pollutants in the region are sulphurous gas, exerting direct foliar damage, and heavy metals, which affect mainly via soil toxification. Among them, sulphur dioxide appears to be the dominating toxicant for the forest ecosystems in this area (Lukina and Nikonov 1993).

In 1969 the smelter started to process the Norilsk sulphide copper-nickel ore (with 30% S), which resulted in a dramatic increase of the emission level. In recent years the emission levels have decreased to half of the peak years because of reduced production volume.

Almost 60 years of the smelter's operation have made the surroundings unrecognizable. Studies show that the area of dead forest is steadily expanding at an annual rate of about 0.5 km in the wind direction (Tikkanen 1995). The damaged area is expanding by 5000 ha-7000 ha per year (Tsvetkov 1993). The affected zone also is expanding at a rate of 2,600 ha per annum (Pozniakov 1993).

Methods to Detect Forest Damage

Impacts of airborne sulphur pollution on the northern forest ecosystems on Kola have been given much attention during the last decades. However, using different symptoms of forest decline and different classifications of forest damage makes their results incomparable. Besides, a botanical approach has natural limitations in this area due to the large scale of damage and the mountainous terrain. From this point of view, remote sensing seems to be very promising, allowing for temporal comparison of results, and analysis of trends produced in different investigations and in temporal dynamics.

Remote sensing was successfully applied for detection of forest damage areas by Pitblado and Amiro (1982); for Sudbary, Canada (Ardö *et al.* 1997); for the Czech Republic (Tømmervik *et al.* 1991-1992 and Karpuz and Roberts 1992); for the 'Pechenganickel' smelter on the Kola; Mikkola and Ritari 1992; Kravtsova *et al.* 1995; Rees and Williams 1997, Hagner and Rigina 1997, for the 'Severonickel' smelter on the Kola).

Some attempts were made to identify a correlation between air pollution and forest health in an emission impact zone (Johansen *et al.* 1994; Mikkola 1996; Litinsky 1996).

In summary, several investigations have shown that multi-temporal satellite data can reveal forest decline, caused by the metallurgical industry emission. However, these comparisons with air pollution were of a qualitative character. The temporal dynamics of air pollution impact still remains largely unexplored. Some attempts were also made to analyze the interrelation between air pollution and forest decline by means of mathematical modeling (Mäkelä 1985; Mäkelä *et al.* 1987; Kryuchkov *et al.* 1995; Boltneva

[1]*O. Rigina, O. Hagner, Remote Sensing Lab, Faculty of Forestry, SLU, S-90183 Sweden fax: + 46 90 141915*

et al. 1982; Tarko *et al.* 1988, Alexeev 1995). Hagner and Rigina (1997) found that decreases in near infrared and increases in visible green and red radiances correspond to increasing pollution load predicted (fig. 1).

Sensitivity and Critical Levels

In a polluted tundra/taiga environment the first sign of pollution impact is the deterioration of the lichen cover, and secondly, mosses, trees, shrubbery and grasses. Broad-leaved woody plants show more tolerance to the influence of pollution than coniferous species.

According to ECE (1988), for lichens, mosses and some trees an annual average SO_2 concentration exceeding 20 $\mu g/m^3$ will cause damage. However, damage effects of SO_2 can also occur under lower concentration conditions, if there are other stress factors, for example, low temperature. A critical level of 15 $\mu g/m^3$ is suggested for forests in Northern areas (Convention, 1985). In our study we used three different critical levels for SO_2. The choice corresponded to critical levels for conifers: spruce, pine (15$\mu g/m^3$); deciduous: birch (30$\mu g/m^3$); and shrubs: willow and grasses (70-100$\mu g/m^3$). According to Tikkanen (1995), an average annual SO_2 concentration of 15 $\mu g/m^3$ outlines the visible-damage zone regarding conifers. Mikkola (1996) found a correlation between the zone of severe damage and a mean annual SO_2 concentration of 40-60 $\mu g/m^3$.

The objective of this study was to estimate expansion of the forest damage area around the 'Severonickel' smelter by means of the mathematical modeling in temporal retrospective.

MATERIALS AND METHODS

Investigation Area

The area under study was a 112 km by 76 km region in the central Kola Peninsula (Lat. 67°26′-67°27′ Long. 32°-34°) (fig. 2). The smelter 'Severonickel' is located 5 km south of Monchegorsk at the center of the study area close to the northern tree limit. The topography is mountainous with elevations ranging from 127 m to 1200 m a.s.l. About 14.5% of the area is covered by water. The study area (fig. 2) belongs to the northern boreal zone, mostly covered by coniferous forests, 12 m-16 m height with prevailing Siberian spruce (*Picea obovata)* and Scots pine (*Pinus sylvestris)* and constant presence (up to 50%) of birch *(Betula pubescens).*

Change Detection Data

The results from a previous study (Hagner and Rigina 1997) were used as a reference to evaluate the modeling results. Landsat-MSS images from mid-summer of 1978, 1986 and 1992 were used in the change detection. The satellite scenes were georeferenced and subsequently resampled to 70x70-m cell size by bilinear interpolation. Multispectral change images were derived for periods of 1978-1986 and 1986-1992 by iterative histogram matching and subsequent subtraction (Hagner & Rigina 1997) (fig. 3).

Forest Impact Model

The annual average sulphur dioxide concentration is a relevant description of the pollutant environment (Mäkäla *et al.* 1987). The analysis of the relationship between forest damage and sulphur concentration was guided by the dose-response approach. A simple dynamic model (Mäkelä *et al.* 1987) for damage development in forests under long-term exposure to airborne sulphur was applied for calculation of damage accumulation since 1960 (fig. 4). The input to the model is the mean annual SO_2 concentration, predicted by a Gaussian plume sector model (Baklanov & Rodyushkina 1993) for each year, based on the corresponding meteorology and emission levels (fig. 5).

The model accounts for the regional differences in tolerance of trees due to climatic conditions by making the critical dose a function of Effective Temperature Sum (ETS). In the study area the 20-year-average ETS was 677.6 degree-days. The cumulative doses and damage areas were calculated for the period of 1961-1996 for three critical levels SO_2 (15 $\mu g/m^3$, 50 $\mu g/m^3$ and 80 $\mu g/m^3$), corresponding to different degrees of impact.

RESULTS

Dynamic Area of Degraded Vegetation

The areas of damaged forest as predicted by the mathematical model, along with the annual emission levels, are presented in figure 6. These data correspond rather well with reported figures on forest

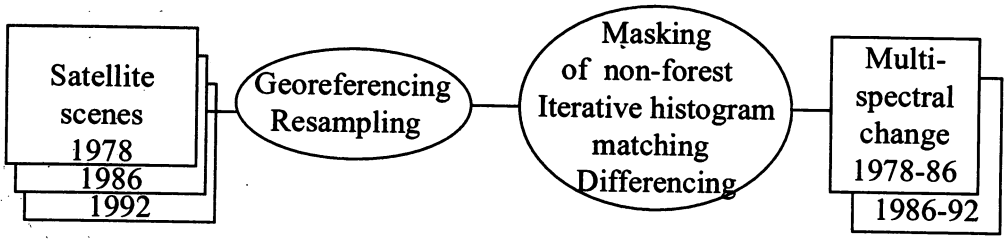

Figure 3. *Remote sensing analysis.*

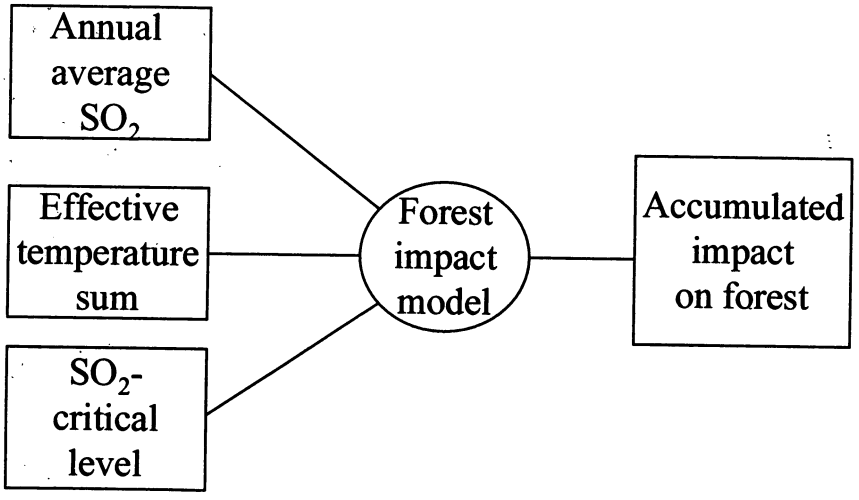

Figure 4. *Forest impact model.*

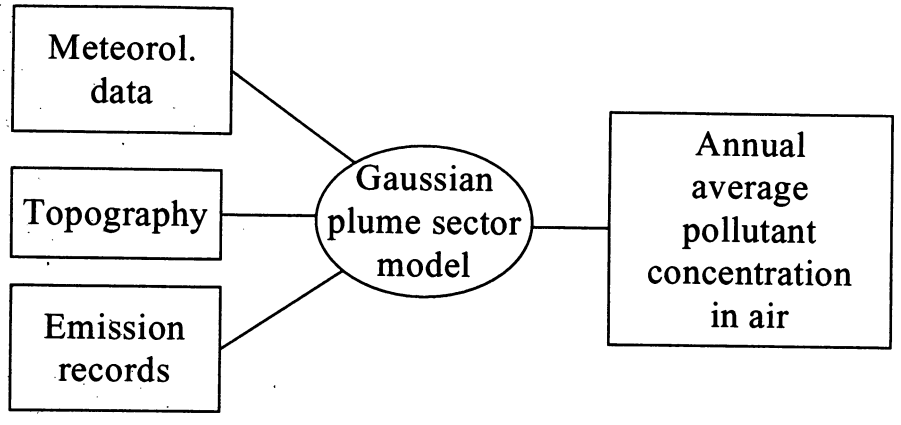

Figure 5. *Gaussian plume model*

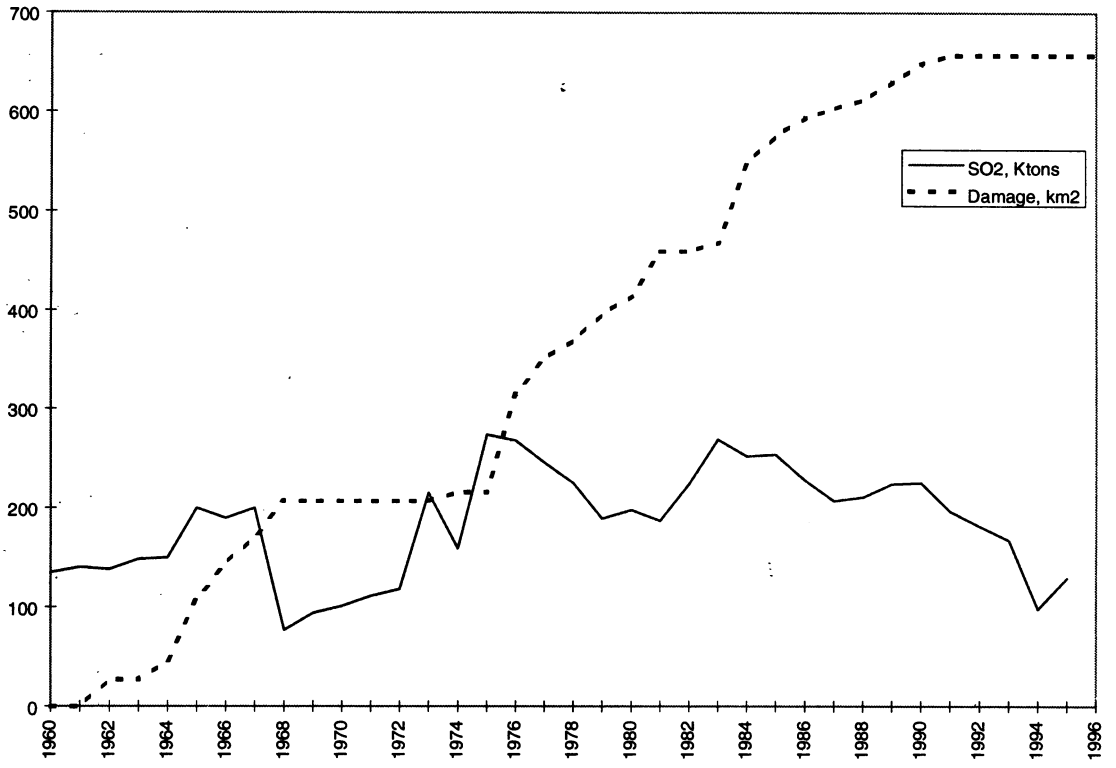

Figure 6. *Calculated damage area corresponding to SO_2 critical level of 50μg/m^3 with the SO_2 emission records in temporal retrospective.*

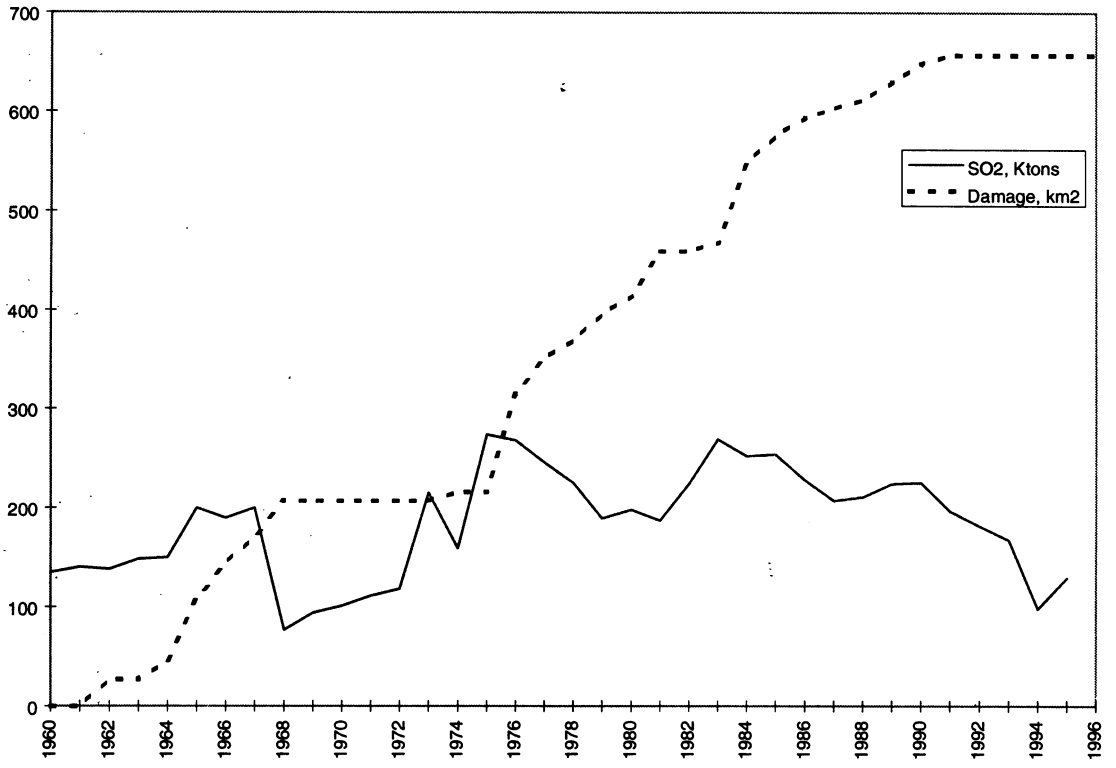

Figure 6. *Calculated damage area corresponding to SO_2 critical level of 50μg/m^3 with the SO_2 emission records in temporal retrospective.*

damage of respective years. The expansion of the damage area is outlined in three classes (fig. 7): visible-, severe damage and forest-death area, roughly corresponding to SO_2 critical levels of $15\mu g/m^3$, $50\mu g/m^3$ and $80\mu g/m^3$. The mean annual increases for these classes are estimated to 60.3 km^2, 17.8 km^2 and 8.8 km^2, correspondingly. The shape of the damage area modeled (fig. 7) corresponds with that from the satellite data (fig. 8) that supports the models used.

DISCUSSION

The results obtained are preliminary. Despite a good correspondence of the modeled damage expansion with the spectral changes for corresponding year, the statistical correlation with the Remote sensing data should be estimated. In future we will continue to integrate the two methods, involving other types of satellite data: spy satellite images from the 1960's and the recent Landsat-TM data.

Acknowledgements

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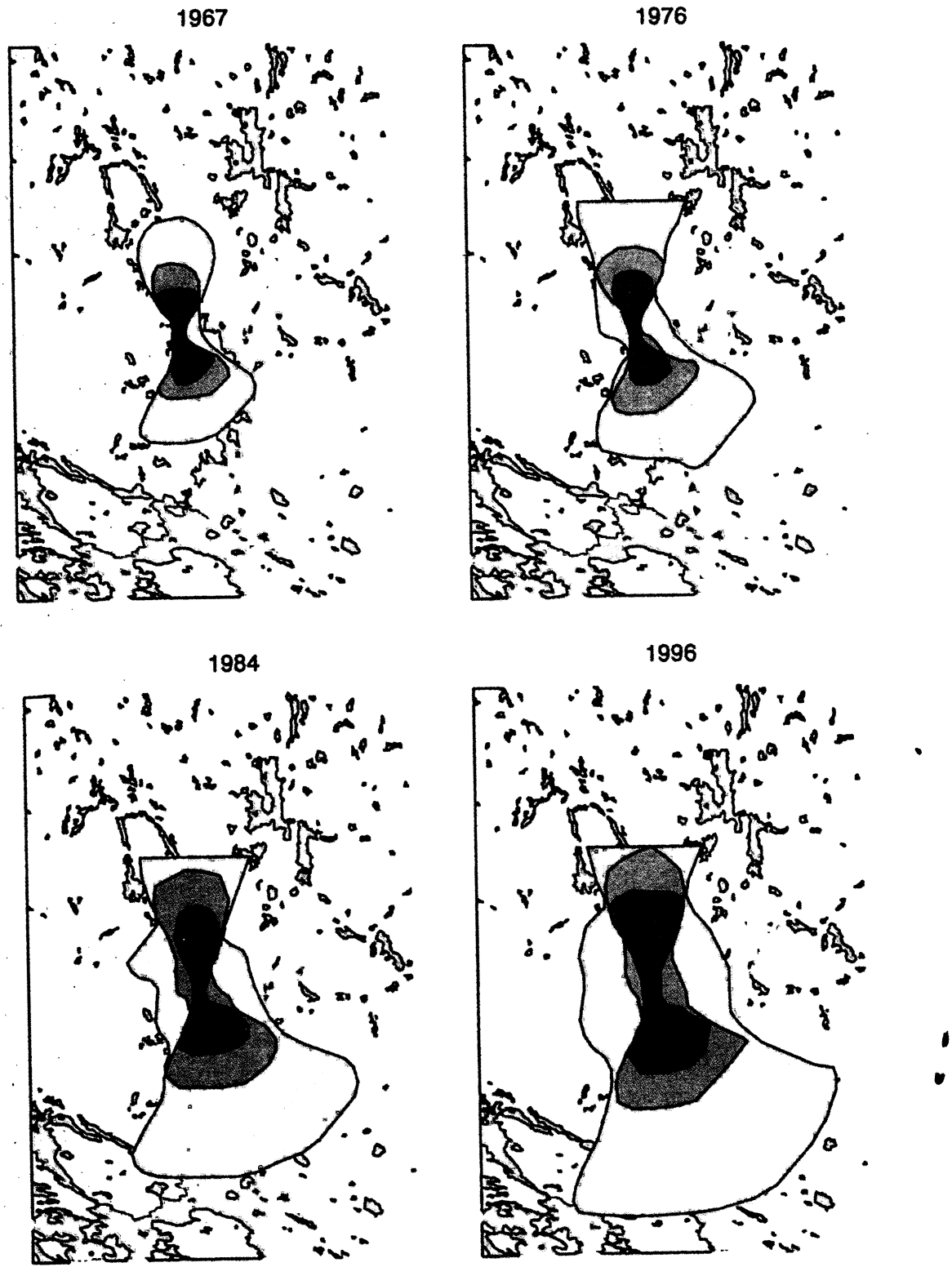

Figure 7. *Expansion of damage area in 1968, 1976, 1984 (peaks of emission) and 1996 corresponding to SO_2 critical levels of 15 (light grey), 50 (medium grey) and 80 (dark grey) $\mu g/m^3$. The grid size is 3x3 km.*

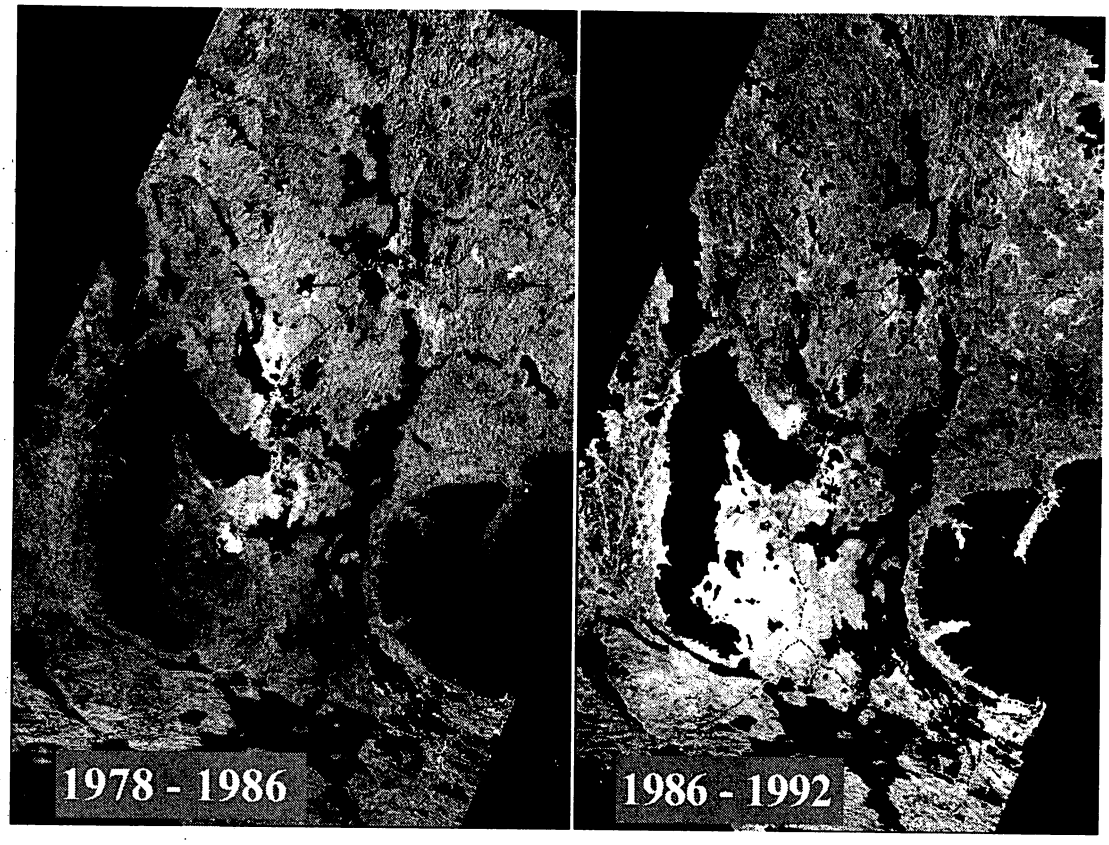

Figure 8. *Multispectral change detection based on Landsat-MSS image data for the periods of 1978-1986 and 1986-1992. Vegetation decline is seen as red hues corresponding to decreased IR- and increased visible wavelenght reflectance.*

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GLOBAL CLIMATE CHANGE AND THE CARBON CYCLE

Climate Change

A METHOD FOR MODEL-BASED RECONSTRUCTION OF THE PALEOCLIMATE OF SIBERIA

Robert A. Monserud[1] and Nadja M. Tchebakova[2]

ABSTRACT.--A bioclimatic vegetation model coupled with the paleovegetation map of Khotinsky and the current vegetation map of Isachenko is used: (1) to reconstruct the paleoclimate of Siberia during the mid-Holocene and (2) to compare it with the current climate. Climatic changes (January and July mean temperatures, and annual precipitation) since the mid-Holocene are then displayed as anomalies by latitude and longitude. The resulting pattern of anomalies is complex. On average, Siberian winters in the mid- Holocene were 3.7°C warmer than now, with greater warming in higher latitudes. The major winter warming was concentrated in the Taiga zone on the plains and tablelands of East Siberia. Siberian summers in the mid-Holocene were 0.7°C warmer than today. July temperature anomalies are distributed mostly latitudinally and increase above 65°N, with nearly no change below 65°N. Annual precipitation in Siberia was predicted to be 95 mm greater in the mid-Holocene than now. Most of the increase was concentrated in East Siberia (154 mm average increase). This increase favored the deep penetration of moisture-demanding dark-needled species (*Pinus sibirica, Abies sibirica, Picea obovata*) into East Siberia in the mid-Holocene, where currently only drought-resistant light-needled species (*Larix* spp.) are found. In combination with 2-5°C warmer summers, moister climates in the Taimyr Peninsula allowed forests to advance far northward into what is now the Tundra zone.

INTRODUCTION

Paleoclimatic reconstructions are a useful complement to atmospheric general circulation models (GCM's) and can be used to validate the GCM's ability to simulate past climates. A wide variety of quantitative methods based on multivariate statistical procedures have been applied to reconstructing paleoclimate directly from pollen data (Howe and Webb 1977, Bartlein and Webb 1985, Bartlein *et al.* 1986, Webb and Bryson 1972, Klimanov 1984, Kay 1979, Andrews *et al.* 1980, Bukreeva 1992, Guiot *et al.* 1989, Huntley 1991, Prentice *et al.* 1991). These methods are based on modern pollen-climate analogues to reconstruct a past climate.

Savina and Khotinsky (1984) introduced the zonal method. The correspondence between climatic indices and vegetation zones (equivalent to biomes or biome subdivisions) has also been used to reconstruct paleoclimate. First, vegetation zones are reconstructed from pollen data. Then, using modern vegetation and climate correlations, climatic indices are calculated that uniquely determine the borders between vegetation zones. These indices are applied to reconstructed paleovegetation to derive information on paleoclimate. The same method was applied to deduce climatic parameters directly from the reconstructed biomes using the global biome model of Prentice *et al.* (1992) for calculating vegetation and soil carbon storage changes since 13,000 years before present (Peng *et al.* (1995). Theirs is basically a model-based approach, as is ours.

We concentrate on the mid-Holocene (Holocene climatic optimum, Late Atlantic period), 4600-6000 yr B.P., when postglacial warming in Siberia reached its peak. The mid-Holocene climatic period is the most relevant for a comparison with the present because the boundary conditions of ice-sheet extent and sea level were similar, and vegetation was close to equilibrium with the climate (Houghton *et al.* 1990). During the mid-Holocene, broad-leaved temperate forests expanded to the north in Europe, shade-tolerant dark-needled Taiga expanded to the north and east in Siberia, and thermophilic taxa were common (Khotinsky 1984a). At middle to high latitudes in the Northern Hemisphere, temperatures were 1°C-4°C higher than today (Lamb 1982, Budyko 1986, COHMAP Members 1988, Frenzel *et al.* 1992). The Holocene climatic optimum, like nearly all the optima of the Pleistocene interglacials, was a period of high humidity in Siberia; conditions were ideal for

[1]*Pacific NW Research Station and Rocky Mountain Research Station, USDA Forest Service, 1221 SW Yamhill #200, Portland, OR 97205 USA. Fax: +1-503-808-2020, e-mail: monserud@forest.moscowfsl.wsu.edu*

[2]*Sukachev Forest Institute, Russian Academy of Sciences (Siberian Branch), Akademgorodok, 660036 Krasnoyarsk, Russia. Fax: +7-3912-433-686, e-mail: ifor@krsk.infotel.ru*

forest growth, especially for shade-tolerant and moisture-demanding dark-needled species. Evapotranspiration and precipitation were balanced over vast areas, with the ratio of precipitation to evaporation near 1.0; under such conditions, biological productivity was high (Khotinsky 1984b).

Objectives

Our goal is to reconstruct the paleoclimate of the Siberian subcontinent during the last warming period, the mid-Holocene thermal optimum. We propose a model-based approach to reconstruct the paleoclimate of Siberia. We rely on three previous developments: Khotinsky's revised paleovegetation map of the mid-Holocene (1984a, p. 194-195), Isachenko's landscape map of the former USSR for modern vegetation (Isachenko *et al.* 1988), and the climatically-driven Siberian vegetation model of Tchebakova *et al.* (1994).

METHODS

Paleoclimatic Reconstruction

Determining the climatic limits bounding a given vegetation class is straightforward (table 1). For each grid cell (0.5° longitude by 0.5° latitude) across Siberia, we determine the climatic limits corresponding to the paleovegetation classes of Khotinsky (1984a) and the modern vegetation classes of Isachenko *et al.* (1988) from the Siberian vegetation model (Tchebakova *et al.* 1994). To apply the modern climatic indices to paleovegetation, we used the aggregation listed by Monserud *et al.* (1993). They discuss these paleovegetation classes in detail. The correspondence is based on dominant genera, species, and physiognomy. The modern vegetation classification is more detailed; therefore, several of its classes have been aggregated to properly match paleovegetation classes.

We convert these climatic limits to January and July temperatures and annual precipitation by using equations 1-5 (see below). The difference between the mean values of a given climatic parameter is the climatic anomaly (Paleo minus Modern) for a given grid cell. Results are displayed as anomalies from the current climate vs. latitude and longitude (figs. 1-3). Because we work at the climatic midpoints of each vegetation class, our method registers no climate change for grid cells with no vegetation change.

Siberian Vegetation Model

The Siberian vegetation model (Tchebakova *et al.* 1994) is an equilibrium model based on bioclimatological considerations. It predicts the major vegetation classes (biome subdivisions) of Siberia from three climatic indices: growing degree-days (base 5°C), dryness index, and continentality index. Because these indices reflect drought resistance, cold tolerance, and plants' requirements for warmth, they define the main features of vegetation zones (zonobiomes sensu Walter 1979) and their subdivisions (table 1). Climatic inputs (monthly mean temperature, precipitation, vapor pressure, cloudiness, and albedo) were obtained from a global climatic database (Leemans and Cramer 1991), supplemented by additional data from weather stations in Siberia and northern Kazakhstan; resolution is 0.5° longitude by 0.5° latitude.

Tchebakova *et al.* (1994) compared the model's predictions with the Landscape Map of the USSR (Isachenko *et al.* 1988), which was not used to develop the model. Predicted vegetation generally matched mapped vegetation well. The general locations of all vegetation classes are predicted correctly. Kappa statistics (Monserud and Leemans 1992) show good agreement at all scales of comparison (Tchebakova *et al.* 1994).

We made two major modifications of the Siberian vegetation model (Tchebakova *et al.* 1994). The first modification improved our procedure for determining albedo, an important component in the calculation of radiation balance and the resulting dryness index. In our earlier work (Tchebakova *et al.* 1994), we used Matthews' (1985) snow-free albedo files, which were limited to the growing season. Instead, we determined monthly albedo based on the reflectance of different generalized surfaces during periods of stable, unstable, and no snow cover (Budyko 1974).

The second modification of the Siberian vegetation model was the refinement of the climatic borders used to predict vegetation classes in our previous work (Tchebakova *et al.* 1994). When we reconstruct paleoclimate from the climatic limits in the Siberian vegetation model, the accuracy of our predictions

TABLE 1.--*The limits defining the modified Siberian vegetation model (Tchebakova* et al.. *1994) for the aggregated vegetation classes common to Khotinsky (1984a) and Isachenko* et al. *(1988). GDD5 is Growing Degree-Days, (°C),* ***DI*** *is Budyko's Dryness Index, and* ***CI*** *is Conrad's Continentality Index. Lower bounds are greater than or equal to the value listed, and upper bounds are less than the value listed. All zones are rectangles in (**DI**, GDD_5) space.*

	Siberian Vegetation Model					
	GDD_5		*DI*		*CI*	
Vegetation Class	Lower	Upper	Lower	Upper	Lower	Upper
Zone	bound	bound	bound	bound	bound	bound
Tundra (moist)	0	300	0.5	1.0	45	60
Tundra (dry)	0	300	1.0	2.6	60	80
Dark-needled Forest-Tundra	300	600	0.5	1.0	45	60
Light-needled Forest-Tundra	300	450	1.0	2.6	60	100
Dark-needled Northern Taiga	600	900	0.5	1.0	45	60
Light-needled Northern Taiga	450	800	1.0	2.6	60	100
Dark-needled Middle Taiga	900	1050	0.5	1.0	45	60
Light-needled Middle Taiga	800	1150	1.0	2.6	60	100
Mixed-conifer Southern Taiga	1050	1200	0.5	2.0	45	60
Birch Subtaiga	1200	1300	0.7	1.0	45	60
Light-needled Subtaiga	1150	1300	1.0	2.0	60	80
Birch Forest-Steppe	1300	1400	0.7	1.0	45	60
Light-needled Forest-Steppe	1300	1400	1.0	2.0	60	80
Steppe	1400	1650	1.0	2.0	45	60
Temperate Birch-Broadleaf Forest	1550	1900	0.7	1.2	40	50
Semidesert/Desert	1650	2000	2.0	3.5	45	60

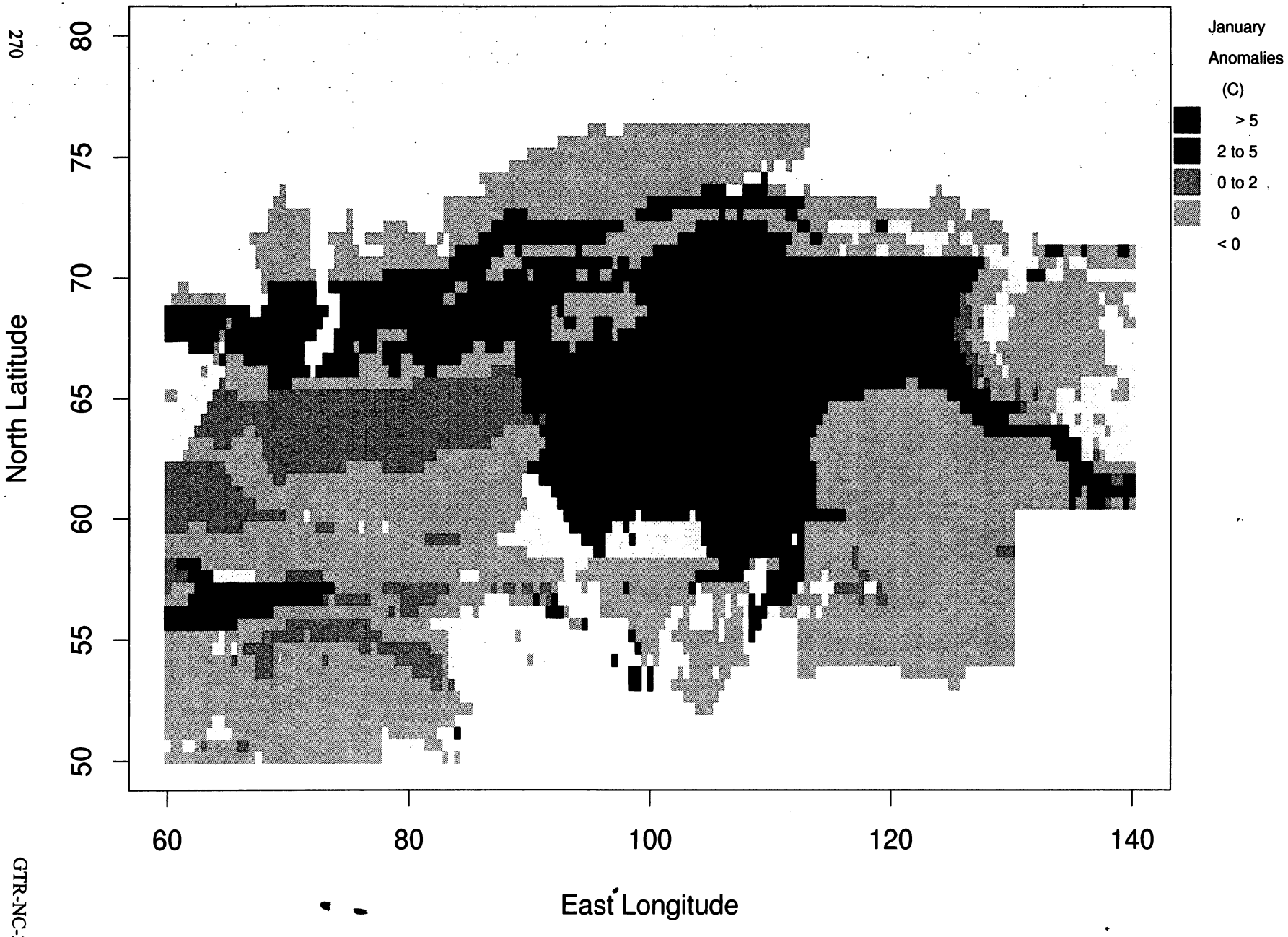

Fig. 1.--*January mean temperature anomalies (palaeoclimate minus modern climate) in the mid-Holocene. Anomalies equal to 0 indicate no change in the vegetation class between the palaeomap of Khotinsky (1984a) and the modern map of Isachenko et al. (1988).*

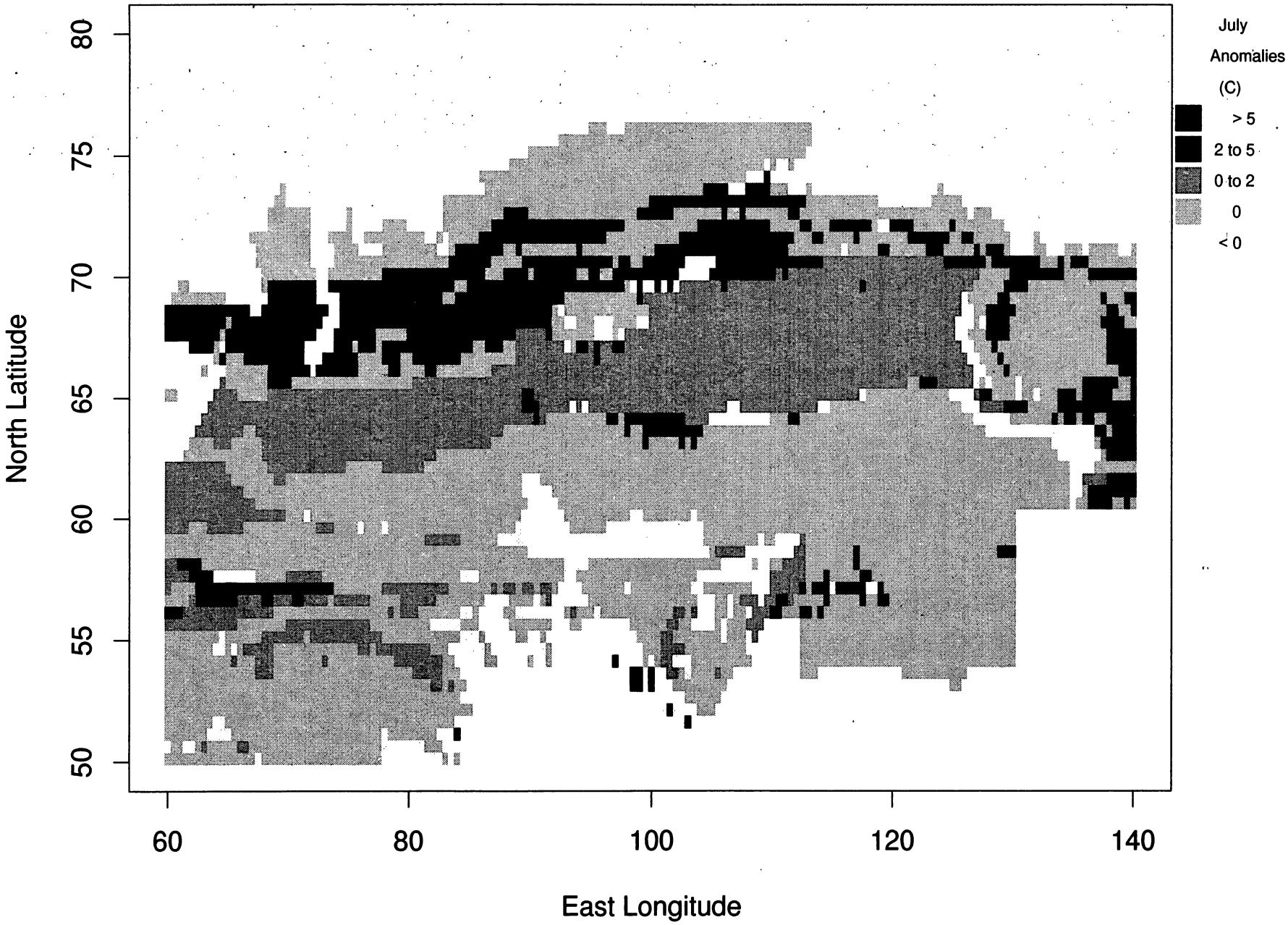

Fig. 2.--*July mean temperature anomalies (palaeoclimate minus modern climate) in the mid-Holocene. Anomalies equal to 0 indicate no change in the vegetation class between the palaeomap of Khotinsky (1984a) and the modern map of Isachenko et al. (1988).*

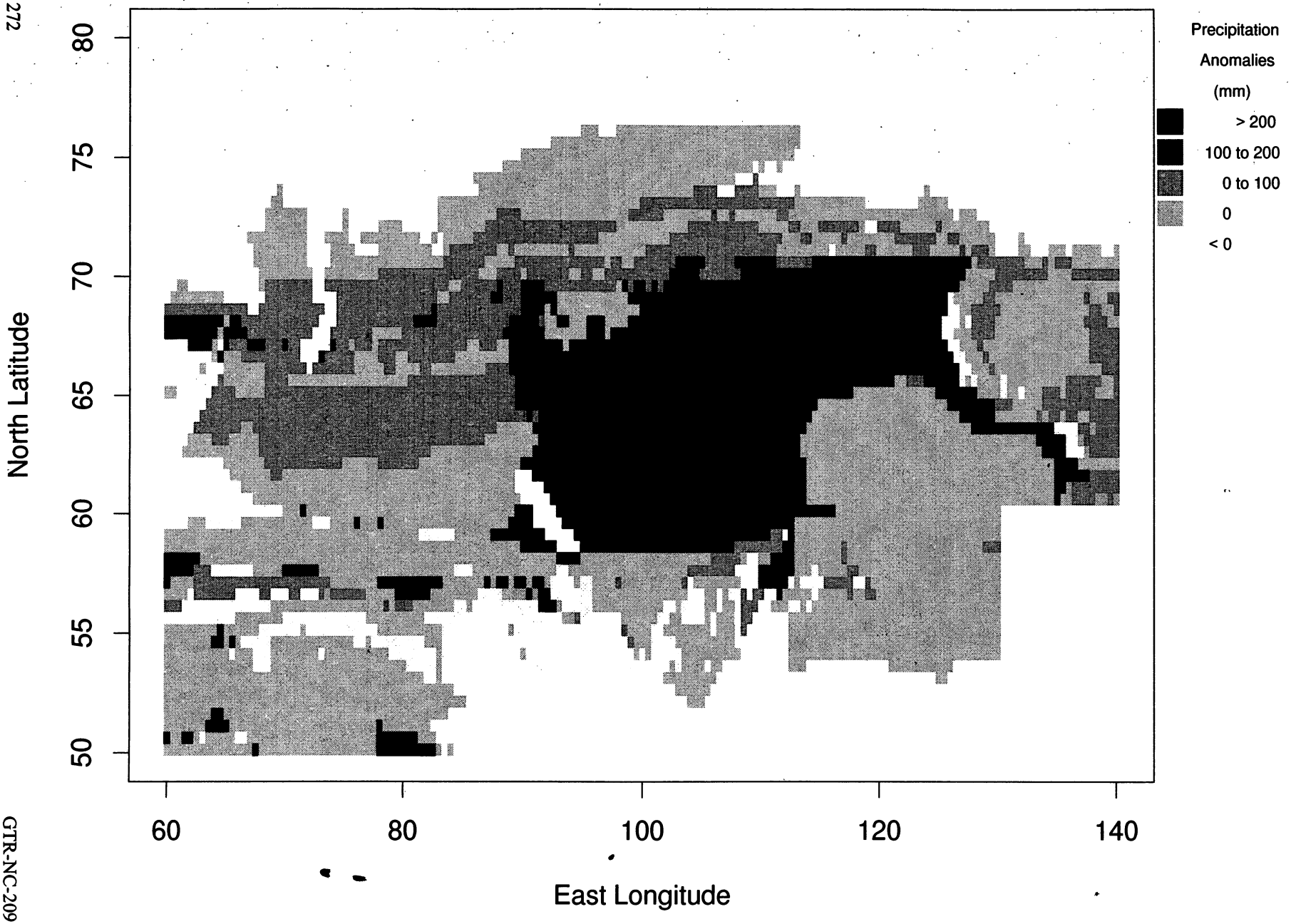

Fig. 3.--*Annual precipitation anomalies (palaeoclimate minus modern climate) in the mid-Holocene. Anomalies equal to 0 indicate no change in the vegetation class between the palaeomap of Khotinsky (1984a) and the modern map of Isachenko et al. (1988).*

improves if the climatic borders defining a class have as narrow a range as possible. When classifying vegetation (Tchebakova *et al.* 1994) rather than reconstructing climate, our only concern was to accurately separate adjacent vegetation classes in climatic space; as a result, several classes had unbounded borders. The resulting modified model is shown in table 1, after aggregating vegetation classes to be compatible with the paleovegetation classes of Khotinsky (1984a).

Converting Climatic Indices to Paleoclimatic Parameters

Paleoclimatic studies routinely reconstruct the mean temperature for January and July and annual precipitation (Bartlein *et al.* 1986). The choice of these climatic variables is justified by their availability from the meteorological record and their close correlation with functional variables more closely related to determining plant (and pollen) distribution and abundance patterns (Huntley 1991), such as growing season temperature sums, exposure to growing season drought, and winter minimum temperature. In the Siberian vegetation model, our analogues for these variables are growing degree-days, dryness index, and continentality index, respectively. To compare our reconstructed climates with results from other paleoclimatic studies, we need to convert our climatic indices to the standard temperature and precipitation parameters. We make the strong assumption that the following climatic conversion factors have not changed much in the past 6,000 years.

We begin with July temperature (t_7, °C), which is strongly correlated to growing degree days. A power function based on data from 151 weather stations in Siberia and northern Kazakhstan (Gidrometeoizdat 1964-70), explains 95% of the variation:

$$T_7=5+0.4042\ GDD_5^{0.4904}$$

where GDD_5 was calculated by daily integration of an annual temperature curve approximated by a cubic Lagrange interpolating function (Greenspan 1971).

January temperature (t_1, °C) can then be derived directly from Conrad's formula for continentality index (CI), when July temperature is known:

$$t_1=t_7-(CI+14)\bullet \sin(\varphi+10)/1.7$$

where φ is latitude.

Precipitation (p, cm) can be derived from Budyko's formula for dryness index (DI):

$$p=B/(L*DI)$$

where B is radiation balance and L is latent heat of vaporization. Using predicted radiation balance on the same 151 weather stations, growing degree days explains 77% of the variation:

$$B=25.068+0.0094GDD_5$$

The final expression to derive precipitation from dryness index and growing-degree days is then:

$$P=(25.068+0.0094GDD_5)/(L*DI)$$

RESULTS

Paleoclimate vs. Current Climate

January Temperature

The pattern of January temperature anomalies (fig. 1) shows that the main changes happened at the higher latitudes. In West Siberia, temperature differences from 0°C -5°C are primarily distributed latitudinally, with anomalies generally increasing to the north; an exception is the large anomalies resulting from the disappearance of Temperate Birch-Broadleaf Forests in southwestern Siberia. In East Siberia, two major results are found: a large area in southeastern Siberia with no changes (light-needled Middle Taiga in paleo and modern times), and a large area in central and northeastern Siberia with large changes greater than 5°C (dark-needled Taiga replaced by light-needled Taiga).

TABLE 2.--*Mean anomalies (mid-Holocene minus modern) of January and July mean temperature and annual precipitation for Siberia. Western and Eastern Siberia are usually separated by 90°E longitude (approximately the Yenisey River).*

	Anomalies (mid-Holocene - modern)		
Location	January temperature (°C)	July temperature (°C)	Annual precipitation (mm)
Latitude (°N):			
70-76	1.8	1.0	44
65-70	7.7	1.3	197
60-65	5.3	0.4	140
55-60	1.1	0.3	22
50-55	-0.4	0.1	-16
60-70	6.5	0.9	168
50-60	0.6	0.2	9
Longitude (°E):			
60-90	0.9	0.8	4
90-140	5.6	0.6	154
Mean	3.7	0.7	95

On average, Siberian winters in the mid-Holocene were 3.7°C warmer than now. Between 60 °N -65°N the paleoclimate was 5.3°C warmer, and between 65 °N -70°N it was 7.7°C warmer (table 2). The major winter warming was concentrated in the Taiga zone on the plains and tablelands of East Siberia (fig. 1). The mean temperature anomaly for East Siberia was 5.6°C compared to 0.9°C for West Siberia (table 2). Tundra vegetation did not disappear completely in the mid-Holocene and was found near the Arctic coast (Khotinsky 1984a). This results in no changes in all three climatic parameters in the far north. In the Taiga zone of West Siberia, between of 63 °N -70°N there were major changes in the dark-needled vegetation classes, indicating winter temperatures that were much warmer. South of 63°N, generally there were only minor changes of January temperature indicated by the analysis (fig. 1), a result of stable zonal vegetation borders there since the mid-Holocene. However, scattered spots of positive anomalies in the South do occur, indicating locations where 6,000 yrs ago Steppe existed rather than Forest-Steppe, or Semidesert rather than Steppe in southwestern Siberia. A noticeable concentration of large positive temperature anomalies occurs between 55 °N and 58°N in western Siberia. This is the location of the Temperate Birch-Broadleaf Forest in the mid-Holocene, which has now been completely replaced by Birch Subtaiga and Birch Forest-Steppe.

Negative anomalies are found at the foothills of the Altai-Sayan mountains and in Central Siberia (about 60°N). In the first case, more continental light-needled Forest-Steppe was replaced by moderately continental birch Forest-Steppe. In the other case, Middle Taiga was replaced by Southern Taiga.

July Temperature

The pattern of July temperature anomalies (fig. 2) is also distributed latitudinally, with greater changes at higher latitudes. Siberian summers in the mid-Holocene were 0.7°C warmer than now, only one-fifth of the corresponding temperature increase reconstructed for winter. There was not much difference in July temperatures between western and eastern Siberia (0.8 °C vs. 0.7°C, respectively), which

contrasts with the large longitudinal differences found for January (0.9 °C vs. 5.6°C; table 2). Summer was 2 °C -5°C warmer than today between 63 °N and 73°N, embracing much of Northern Taiga, Forest-Tundra, and Tundra zones. A band of moderate summer temperature anomalies (0 °C -2°C) is centered at 65°N, and a second band of greater anomalies (2 °C -5°C) is centered at 70°N. In this northern band, Northern Taiga has been replaced by Forest-Tundra, and Forest-Tundra and even Northern Taiga in the mid- Holocene has been replaced by Tundra today. Below approximately 63°N, summer was nearly the same as today across the southern half of Siberia. In southwest Siberia, two narrow strips of positive anomalies indicate that more warmth-requiring vegetation was replaced by less warmth-requiring vegetation (e.g., Steppe by Forest-Steppe, and Temperate Birch-Broadleaf Forest replaced by Birch Subtaiga and Forest-Steppe).

There are several scattered patches of negative anomalies in Central Siberia where Middle Taiga was replaced by Southern Taiga. Two patches are found in the southwest, in Kazakhstan, where Steppe was replaced by Semidesert.

Precipitation

On average, annual precipitation in Siberia was 95 mm higher than now. Most of the increase was concentrated in East Siberia (154 mm average). Latitudinally, the precipitation anomalies are very small in the south (9 mm below 60°N), large between 60 and 70°N (168 mm), and moderate in the far north above 70°N (44 mm) (table 2).

The precipitation anomalies are dominated by a large increase in moisture in the mid-Holocene in central and northeastern Siberia (Krasnoyarsk, Yakutia, and northern Irkutsk Territories of Russia). This is the same location of the largest January temperature anomalies. This area was dominated by dark-needled Northern and Middle Taiga in the mid-Holocene. Annual precipitation in this region in the mid-Holocene was more than 197 mm higher than present (fig. 3, table 2). This increase in precipitation allowed moisture-demanding dark-needled species (*Pinus sibirica, Abies sibirica, Picea obovata*) to penetrate deep into East Siberia (Khotinsky 1984a). Only drought-resistant light-needled species (*Larix* spp.) are found there now (Isachenko *et al.* 1988).

A second, but less dramatic area of increased precipitation (mostly 0 mm-100 mm increase), starts in West Siberia between 63 °N and 70°N and stretches along the North Siberian Depression (Severo Sibirskaya Nizmennost) at the base of the Taimyr Peninsula, and on across northern Yakutia to the mountains in the northeast. In combination with 2 °C-5°C warmer summers, moister climates in the mid-Holocene allowed trees to advance far northwards into the current Tundra zone (Khotinsky 1984a, Avenarius *et al.* 1987). In the mountain ranges of northeastern Yakutia, Northern Taiga was replaced by Tundra.

Vegetation has remained stable over large areas of Siberia since the mid-Holocene. No precipitation anomalies are found in Tundra along the Arctic coast and the northern mountains of Yakutia, in the light-needled Middle Taiga of southeastern Yakutia and Transbaikalia, in Southern Taiga across western and central Siberia, and the Steppes of Kazakhstan.

Several locations with lower annual precipitation are noted: the western foothills of the Altai-Sayan mountains where less moist light-needled Forest-Steppe was replaced by moister Birch Forest-Steppe; northern Kazakhstan, where Steppe was replaced by moister Forest-Steppe; and West Siberia near the Ural Mountains, where colder Middle Taiga was replaced by warmer Southern Taiga.

DISCUSSION

We used a model-based approach to reconstruct the climate of the mid-Holocene for Siberia. Our procedure for climatic reconstruction differs from traditional methods in several ways. First, instead of focusing on a palynological analysis, we begin with the end product of such a thorough analysis: Khotinsky's (1984a) paleovegetation map covering most of Siberia. We refined this map slightly with additional information from the Russian literature. Second, we rely on a bioclimatic vegetation model to specify the relationship between climate and vegetation (Tchebakova *et al.* 1994). Thus we reconstruct climate directly from vegetation, with anomalies determined from climatic differences corresponding to different vegetation classes in the modern and paleomaps. Savina and Khotinsky (1984) used a similar model-based approach to relate climate to vegetation, but on a coarser grid; only positive and

negative anomalies were distinguished.

An advantage of a model-based approach to climatic reconstruction is that grid cells can be modeled independently, without the need for interpolation to create smoothed temperature and precipitation contours. The resulting anomaly maps are generally complex in nature, for no attempt to reduce information through smoothing is made. Because we work at midpoints of vegetation classes, we reconstruct only one value of a climatic parameter for each zone, with no variation. Variation of course exists, but of necessity must be ignored with the current procedure. Thus, our zero anomalies in figures 1-3 indicate only that the climate has not changed enough to support an adjacent vegetation class in table 1.

With such a model-based approach to paleoclimatic reconstruction, it is important that paleo and current vegetation classes should represent the same general vegetation complex, with the same climatic borders. The maps of Khotinsky (1984a) and Isachenko *et al.* (1988), as well as the Siberian vegetation model (Tchebakova *et al.* (1994), are all based on a similar zonal vegetation classification, and employing longitudinal sectors separating light-needled from dark-needled classes in the same zone.

Our analysis showed that paleoclimatic reconstructions are rather sensitive to the precise specification of the climatic borders. The discrete nature of vegetation classes as represented in both the paleovegetation map and the Siberian vegetation model creates an associated problem. Class borders are sharp and distinct, producing extremely large and rapid climatic gradients at the borders between vegetation classes. Such large differences in climate over short distances are unrealistic, but the solution to this problem is far from obvious as long as the ecotones represented by the borders are neither mapped nor continuously modeled. A ramp function (e.g., linear interpolation) could smooth this discrete transition near the class borders, but would not alter the main features of the anomalies displayed in figures 1-3.

We compared the maps of January and July temperatures and annual precipitation anomalies for the mid-Holocene (figs. 1-3) to corresponding maps from the literature, primarily those found in the paleoatlas of Frenzel *et al.* (1992, p. 65, 69, 77).

These paleoclimatic reconstructions are derived from the paleorecord using present associations between pollen and climatic variables. There are both similarities and differences between our reconstructions and those found in literature for the mid-Holocene in Siberia. Our summer anomalies are quite similar in both magnitude and pattern to those from the paleoatlas of Frenzel *et al.* (1992). We also predict winter anomalies greater than summer ones. Examining the Last Interglacial Climatic Optimum (120,000 yr B.P.), Velichko *et al.* (1992) found that winter temperature anomalies were greater than summer anomalies in Russia, with the greatest anomalies in eastern Siberia. This is essentially the same result we found for the mid-Holocene Climatic Optimum.

Our reconstructions differ in several aspects. We predict a different pattern of January temperature anomalies that are stratified longitudinally rather latitudinally as shown by Frenzel *et al.* (1992). The largest January anomalies are predicted for East Siberia (>5°C), where the greatest anomalies of annual precipitation are also found. This corresponds to the large expanse of light-needled larch Taiga in central and eastern Siberia known to have supported moisture-demanding dark-needled Taiga species in the mid-Holocene (Khotinsky 1984a).

The Siberian vegetation model (Tchebakova *et al.* 1994) is driven by climatic parameters, but not by the distribution of permafrost. Velichko *et al.* (1995) concluded that warming during the Holocene thawed the permafrost surface layers over much of northern and eastern Siberia by the mid-Holocene. This thawing was partially responsible for the advance of dark-needle species to the east and north into the current permafrost zone (Velichko *et al.* 1995). Because we base our paleoclimatic reconstruction on the Siberian vegetation model, we associate this advance by the dark-needled species with an increase in temperature and precipitation. By ignoring permafrost, we likely overestimate precipitation in the portion of the permafrost zone that thawed to a sufficient depth in the mid-Holocene to accommodate the advance of dark-needled tree species, for melting of surface layers of permafrost makes additional moisture available for plants. Ignoring permafrost should also result in an underestimate of temperature, for enormous heat resources are required to convert water from the solid to the liquid phase.

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CIRCUMPOLAR WARMING EFFECTS ON TREE-LINE REPRODUCTION IN CANADA AND SIBERIA

Harvey Nichols[1]

ABSTRACT.--The arctic tree-line is sensitive to climatic changes as indicated by paleo-ecological studies, and it is predicted by global circulation models to respond strongly to greenhouse warming. Northern Canadian studies of tree-line reproduction spanning two decades demonstrate a widespread switch from infertility due to cold summers (1960s-1970s) to pollen and cone production (1990s), in line with climatic warming predictions. Ecotonal cone formation is usually sporadic and localized, but this largescale reproductive shift, along a 1,500 km transect, suggests widespread climatic warming since the 1970s across much of the Northwest Territories. Labrador, not included in the original study, has experienced recent cooling.

In 1995 and 1996 I tested the hypothesis by examining arctic tree-line in western Siberia and northern Yakutia, northeast Siberia. At all these sites ecotonal larch trees were reproducing sexually, and greenhouse studies confirm that enough seeds are viable to allow seedling colonization of the tundra. Siberian colleagues noted that the age structure of these "tree-islands" based on tree-ring studies suggested that a recent warming response was identifiable.

These Siberian studies (at 27 sites) represented only a modest fraction of the Eurasian tree-line, but the widespread fertility at so many locations, plus the extensive Canadian evidence, and Fenno-Scandinavian findings, suggest that the predicted polar warming may be responsible. Whether this is due to natural or anthropogenic climatic change, or some combination, and whether it will be short or long-term, is unclear, and merits further study and long-term monitoring.

INTRODUCTION

The boreal forest is now recognized as being one of the earth's climatically most responsive biomes in the late Quaternary period, including the present. Paleo-ecological studies have shown that both the southern and northern boundaries of the forest have responded to environmental changes with rapid movements across long distances (Nichols 1967a, 1969, 1975, 1976; Pielou 1991; Ritchie 1984). The arctic tree-line in Canada and probably its counterpart in Siberia can lay claim to be the most climatically sensitive ecotones of any major biome. Palynological studies demonstrated that post-glacial migration of spruce trees in Keewatin (Northwest Territories, Canada) during mid-holocene Hypsithermal warming equalled or exceeded 240 m per year (Nichols 1967b, p. 185). Similar figures have been suggested for the Northwest Territories by Macdonald *et al.* (1993) and for New England (Davis 1983). If global warming occurs as predicted by global circulation models, and if temperatures eventually equal those of the Hypsithermal, then these figures (above) for forest migration rates may be relevant to future management of the boreal forest. A related extrapolation from paleo-ecology in northern Canada is that each 1° C of mean July warming in the past eventually produced a northward movement of up to 100 km in the arctic tree-line in Keewatin (Nichols 1975).

Early recognition of environmental changes can be expected at climatically controlled ecotones such as the arctic tree-line, which is limited by July 10° C mean temperatures, (Nordensköld and Mecking 1928, Köeppen 1936), the Arctic Front (Bryson 1966, Krebs and Barry 1970), and the 800° day limit above 10° C (Arno 1984, Sirén 1997 p.c.). Thus presence or absence of biospheric responses to environmental changes can be seen as a test of GCM model predictions of polar warming (IPCC 1990, Nichols 1993).

A TWO-DECADE CANADIAN EXPERIMENT

In 1972 and 1973, I placed pollen traps in Keewatin along a transect from the tree-line northwards into the high arctic, (fig. 1) to provide data from modern vegetation and associated present-day pollen deposition. These were analogs for fossil pollen data that allowed interpretation of past vegetational and climatic events (Nichols 1975). As the pollen transect moved north from the tree-line into the tundra I designed it to pass through small clumps of dwarf black spruce (*Picea mariana*), where I placed pollen

[1]*Department EPO Biology, University of Colorado, Boulder, CO 80309-0334.*

traps to test for localized tree pollen release. The pollen collectors were placed close to the dwarf trees, while others were sited at 50 m and 100 m away in open tundra to record the regional pollen deposition from the forest to the south. This proved that in 1972 and 1973 these "tree islands" in the low arctic tundra were not releasing pollen and they were not producing cones. Palynological examination of moss accumulation (polsters) nearby showed that absence of pollen release from these dwarf spruce extended back into the past, at least throughout the 1960s. This agreed with Larsen's (1965) observations of infertility and absence of seedlings along this part of the Keewatin tree-line. Elliott (1979) also reported infertility of spruce at one of these same sites. Thus the infertility of the arctic tree-line in Keewatin, northern Canada, was noted at more than a dozen sites over two decades (1960s - 1970s) by three investigators.

Cone production at long intervals in sub-arctic marginal trees has been recorded elsewhere (Larsen 1980, 1988, Black and Bliss 1980, Khomentovsky 1990). My photographic records show that a few of the dwarf trees visited on three transects in 1972 and 1973 exhibited small numbers of old cones which remain in place for a number of years, but there were no developing or very recent cones at all. These Canadian data suggest that the infertility in the "tree islands" related to prolonged limitation of sexual reproduction by cold summers in the 1960s and 1970s. These observations form the baseline for the current studies that show widespread changes in tree reproduction along the rim of the Canadian arctic. Comparative studies were made in two regions of Siberia to test the original hypothesis of a warming effect on tree-line (see below).

In June and July 1993 I was able to re-visit sites I studied in 1972 and 1973 at tree-line in northern Canada and in the arctic tundra up to 250 km beyond tree-line (fig. 1). The objective of the most recent journey was to detect any alteration in the reproduction of the spruce trees caused by climatic change in the previous 20 years. We know that colder conditions cause the trees to reproduce slowly, by vegetative means as clones, with rare or occasional localized episodes of cone formation, and that warmer climates result in pollen and cone production and eventually seedlings. Additional sites to those of 1972-1973 were examined along a northwest-southeast transect along and north of the tree-line in Keewatin and Mackenzie, Northwest Territories (fig. 1). Eight sampling locations were chosen along an L-shaped 1,500 km transect from west of Hudson Bay, beginning with Nueltin Lake and Ennadai Lake (part of the 1972-1973 transect), past Great Slave Lake, to north of the Great Bear Lake close to the Artic Ocean. Pollen traps were placed in groves of trees (primarily black spruce, *Picea mariana)* in the eastern section, plus white spruce (*P. glauca)* in the western section of the transect) beginning in late June 1993 in order to collect evidence for presence or absence of pollen release by groups of spruce trees isolated in the tundra, north of the woodland edge (details below). The transect was repeated through mid-July, to collect the pollen samples and examine cone formation and rates of annual growth.

At every one of the five re-visited sites (1973/1993) there was strong evidence of change in conifer reproduction compared to 20 years ago, and the three additional sites along the east-west transect were all sexually reproducing. In early to mid-July 1993 pollen release was noted at most of the sites visited, and by mid-July at every one of the eight sites numerous new cones were being formed. Cones retained from the 2 or 3 previous years were observed on all the black spruce (*Picea mariana*), which is characteristic of this species, and cone formation was recognizable back to at least 1991 or 1990 (from the degree of differential weathering of the cones still attached on the trees). Small spruce seedlings were observed at several of the locations nearest to tree-line and thus presumably warmer: site 1, Point Lake, 12 miles (19 km) north of tree-line; site 3, Artillery Lake, at tree-line; and site 5, mid-section of Nueltin Lake, 12 miles (19 km) south of tree-line. They were associated with groves of white spruce (*Picea glauca*), with seedling heights of 15 cm or so, and estimated establishment dating back to early to mid-1980s. The more harsh and northerly tundra environments (sites 2, 4, 6, 7, and 8) had no seedlings. These were areas of delayed phenology as judged by ice on the lakes, in the eastern section of the transect north of Nueltin and Ennadai Lakes. These sites had a mean distance north of tree-line of almost 60 miles (96 km).

The most northerly and exposed sites: 6, 7, and 8 (fig. 1) - had not expanded the clumps of dwarf black spruce (*Picea mariana*) at all, according to my photographic records from 1972-1973; they had merely grown a little taller (< 1 m additional height). By contrast, toward the southern end of Nueltin Lake, site 5, about 40 miles (~ 65 km) south of site 6, and ~ 12 miles (20 km) south of tree-line, the sheltered Indian Camp island exhibited clear evidence of new

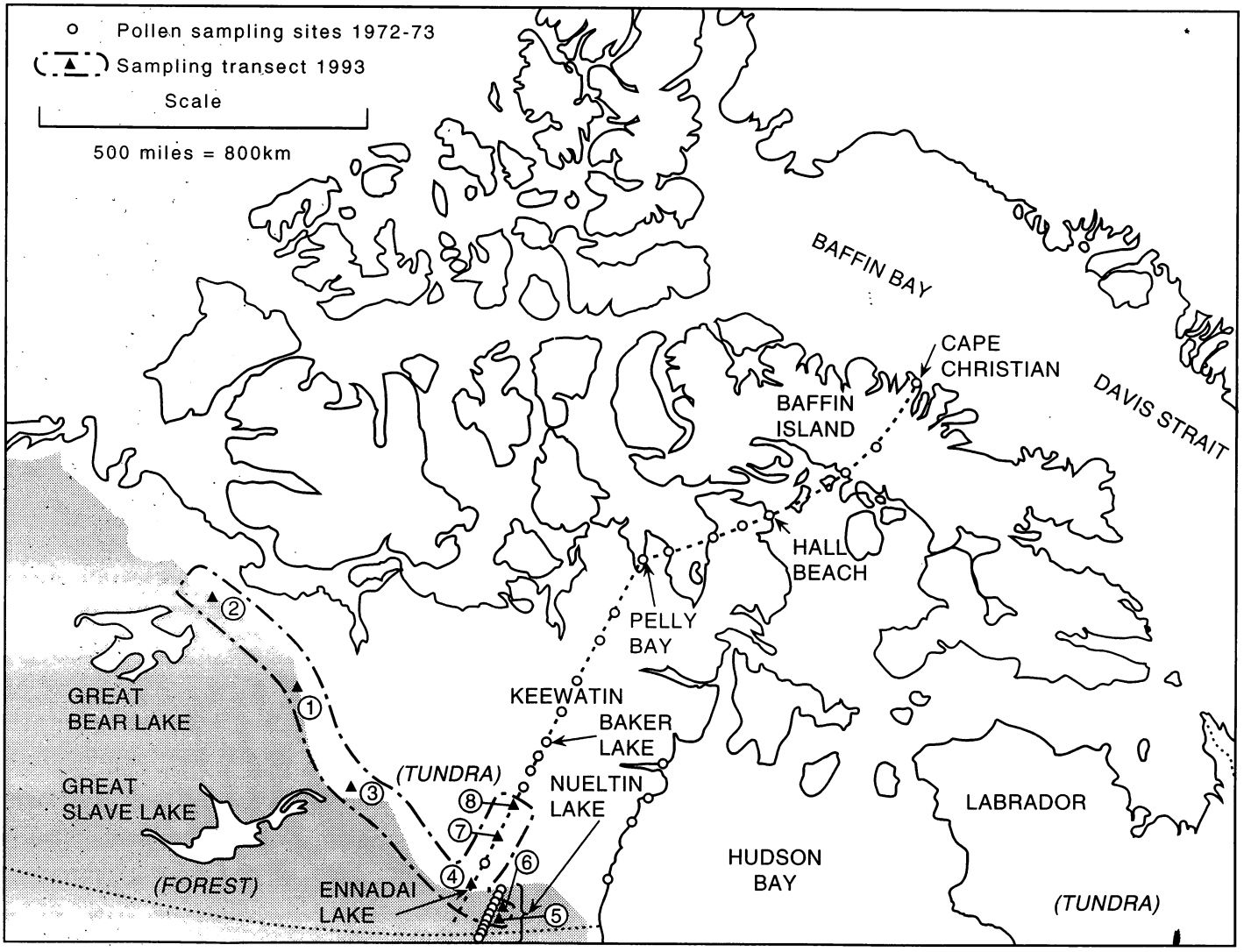

Figure 1.-- Location of pollen sampling sites in 1972-1973 and 1993.

seedling establishment and substantial growth of existing black spruce and larch (*Larix laricina*).

Cone formation such as that recorded above does not guarantee seed viability, because summer temperatures higher than for pollen production and cone initiation are needed for this stage of development, with mean summer temperatures in excess of 15° C required (Black and Bliss 1980). A study at one of the sites here re-investigated (Ennadai Lake, site 4) recorded 100% non-viability of spruce seeds from old cones and soil samples in the late 1970s (Elliott 1979).

Seeds from the 1993 crop of cones were planted in different types of soil to test for germination in Colorado, and several sites exhibited seedling growth in autumn and early winter 1993-1994. As reported for comparable studies elsewhere (Fraser 1976, Alexander *et al.* 1982) only a minority of seeds from these stressed sites proved viable, and not all sites produced seedlings, but these germination data may further support the concept of climatic amelioration. The seeds were planted in plastic cones using three soil types (peat, peat plus sand, potting soil), with refrigeration but without cold stratification at sub-freezing temperatures. The test plantings were most successful in the peat and sand mixture, as compared to peat alone or potting soil. The peat came from my previous fossil peat collections in Keewatin, to most closely approximate the site characteristics in the Northwest Territories. Viability was not equally distributed across the length of the transect. The sites numbered 1, 2, 3, and 5 exhibited zero germination. These sites were mostly in the western arm of the L-shaped transect, and sites 1, 3, and 5 were closest to tree-line, and recorded all the seedlings recorded in the field in this study. The locations are:

Site 1 - (Point Lake, 65° 17' .10 N lat., 113° 41 .65 W long.)

Site 2 - (Dismal Lake, 67° 14' .10 N lat., 116° 36' .80 W long.) and

Site 3 - (Artillery Lake, 62° 57' .92 N lat., 108° 15 .23 W long.).

Site 5 - (Indian Camp Island on Nueltin Lake at 60° 02' N lat., 99° 51' W long.) which also produced no viable seeds, despite there being seedling establishment on thick peat soil and healthy growth of spruce and larch.

By contrast, site 4 - (Ennadai Lake), site 6 - (north end of Nueltin Lake) and sites 7 and 8 (north of Hicks Lake) all produced viable seeds from dwarf black spruce, all from tundra locations generally farther from tree-line. These site locations are:

Site 4 Ennadai Lake, Aeradio Station 61° 08' N lat., 100° 53' W long.

Site 6 Nueltin Lake (north end) 60° 39' .94 N lat., 99° 22' .35 W long.

Site 7 north of Hicks Lake 61° 36' .94 N lat., 100° 08' . 04 W long.

Site 8 north of Hicks Lake 62° 02' N lat., 99° 45' .70 W long.

All locations quoted were determined using GPS equipment on the airplane, except for site 4 where ice prevented float plane landing on July 1, 1993, necessitating walking to the (de-activated) Ennadai Aeradio Station.

Site 4 had a seed germination rate of 2.1%, site 6 = 2.1%, site 7 = 5.2%, site 8 had 8.3% viability. There appears to be an approximate trend in these data, with an increase in viability with greater distance from the arctic tree-line. One would expect the sites with the tallest trees and most advanced phenology (sites 1, 2, 3, and 5) to produce more viable seed than those with dwarfed trees farther out in the tundra (sites 4, 6, 7, and 8), but the reverse has proved true. Sites 4, 6, 7, and 8 were characterized by small groups of diminutive spruce trees, ranging in height from about 2 m (sites 4, 5, 6, 7) to a meter or less (site 8), proceeding from south to north.

CANADIAN DATA IMPLICATIONS

These phenomena of sexual activity in spruce are consonant with the theoretical predictions of exaggerated polar warming due to the anthropogenic greenhouse effect (IPCC 1990), but they do not prove it. A natural "flickering" effect due to variability in the summer position of the Arctic Front

Table 1.--*Distance from tree-line was derived from the map of General Land Cover Classification produced from 1 km imagery by the Manitoba Remote Sensing Centre and plotted at the NWT Centre for Remote Sensing, Yellowknife. Distances were measured from the contact of transitional forest with tundra, aided by personal photographic records.*

Site	Distance from tree line	% seed viability
Site 1	12 miles (20 km) N of tree-line	zero
Site 2	50 miles (80 km) N of tree-line	zero
Site 3	at tree-line	zero
Site 4	31 miles (50 km) north of tree-line	2.1%
Site 5	12 miles (20 km) south of tree-line	zero
Site 6	37 miles (60 km) north of tree-line	2.1%
Site 7	71 miles (115 km) north of tree-line	5.2%
Site 8	105 miles (170 km) north of tree-line	8.3%

episodically affects cone production at tree-line, as does nutrient and energy build-up. Overall, my impression was that the effects of change in climate (if that was the cause) were modest and/or recent, as registered by vegetational response over the last 2 decades. These observations will need to be repeated, and seedling establishment monitored for some time at the most northern sites, for a trend to be confirmed.

The changes, now apparently beginning, may eventually prove to resemble: (1) the minor warming of the late 19th-early 20th century, or (2) the more substantial medieval warming (roughly 600-1,600 years ago), both with some degree of written record and cultural response; or possibly (3) the much longer and more dramatic warming of the middle of our current interglacial period, from 3,000-4,000 years ago back to 8,000-9,000+ years ago, known as the Hypsithermal episode (Lamb 1977). The former changes in the arctic tree-line in Canada caused by these three different warming episodes provide us with a long perspective from which to view the current effects (Nichols 1975). In episode (1), dwarf spruce trees developed upright leaders for several decades before being forced back down to a partially prostrate growth form by colder climate (Jacoby p.c., d'Arrigo and Jacoby 1993). During episode (2), the arctic tree-line extended some tens of kilometers northwards (Bryson *et al.* 1965; Nichols 1967b, p. 185, 1975, p. 66), then retreated during the late medieval Little Ice Age. The greatest biotic effect was registered during episode (3) the Hypsithermal, when summer temperatures 3° -4° higher than the mean pushed the central Canadian tree-line 300 km-400 km north of present limit, at a rate of 100 m-300 m per year (Nichols 1967b, 1975; Macdonald *et al.* 1993). Similar forest advances were registered in Siberia (Levskovskaya 1976). In Canada, an area of about 500,000 km^2 of former tundra was afforested during this episode, over several millenia (Nichols 1976).

The recognition of these phenomena along a substantial portion of a continental ecotone might reasonably be regarded as indicative of more widespread climatic change. It is possible that this is the reflection of a sub-continental shift in the upper atmospheric Rossby wave (the "jetstream"), producing warming due to an atmospheric ridge. Other arctic regions "downstream" in an atmospheric ridge and trough effect might then be expected to exhibit coolings, as indeed is occurring in Labrador currently.

SIBERIAN COMPARISON

I arranged collaborative research expeditions into two areas of western Siberia and eastern Siberia for two months in 1995 and 1996 (fig. 2), making observations at the arctic tree-line to compare with the 1993 Canadian data. We found similar effects as in Canada, with consistent sexual reproduction of Siberian larches (*Larix sibirica* in the Yamal and Taz Peninsulas and *L. kajendera* in northeastern Yakutia) in "tree islands" in the tundra. These studies lack the two-decadal time perspective of the Canadian work, but they indicate contemporary arctic summers warm enough to allow fertility in the "tree islands" located over 60 miles (100 km) north of the open woodland, along with some modest seedling colonization of the "southern arctic" (Russian equivalent of "low arctic" in N. America). The western Siberian larch trees yielded large numbers of mature cones representing the growing season of 1994; the 1995 cones were still forming in July of that year. Approximately 20,000

cones were collected and approximately 5,000 seeds were planted in the greenhouse at the University of Colorado (CU).

In September 1995 these cones were placed in plastic bags in refrigerators at ~ 5°C for 4 weeks and then transferred to a freezer at -20°C for 3 months to simulate autumn and winter conditions. The seeds were initially removed by manual extraction from the cones, but we found that simply shaking the large plastic cone bags released ample seed quantities. The seeds were sown in a standard potting soil mixture used at the CU Greenhouse, and were then divided into two populations. One group was returned to the refrigerator for 2 months to encourage germination. The second group was immediately placed in a room at the CU Greeenhouse designed to mimic a relatively cool moist montane tropical environment, ranging from 15°c-20°c with humidity of 80%. This was not exactly a northern boreal forest or arctic tree-line environment, especially in terms of the high humidity, but was the nearest equivalent then available at the CU Greenhouse. The first group of seeds was removed from the refrigerator and subjected to the same temperature and humidity regime. The two different approaches to cold stratification we employed did not produce markedly different rates of germination; both sets of larch seeds demonstrated ~ 11% viability. Lighting conditions were those of natural daylight here at 40° N latitude with roof shades to control excessive sunlight, with day length typical of spring, summer and autumn at this latitude, compared to the collecting sites at ~ 67° N lat.

The cones from the 1996 expedition were gathered from "tree islands" of *Larix kajendera* isolated in the tundra, along the Indigirka River and westwards towards the Lena River, about 70° N latitude (fig. 2). We recorded sexual reproduction in the form of widespread cone formation at 14 locations. The cones' contents will be tested for viability in the new alpine cold room at the university greenhouse.

We know that seed viability and seedling establishment are limited by temperature and growing season length at the arctic tree-line, and that seed viability in black spruce cones (*Picea mariana*) can range down to zero under severely limiting conditions (Larsen 1980, 1988). The finding of ~ 11% seed viability reported here for Siberian larch (*Larix sibirica*) accompanies the field observations of limited and recent larch seedling establishment in the southern Siberian tundra, and supports the hypothesis of warming at the arctic tree-line. It is not possible to say from this data whether this reflects higher temperatures in July or a longer growing season and milder winters expressed as increased degree days, and whether this results from a natural climatic warming and/or an anthropogenic effect on climate.

The three study regions I have chosen around the circumpolar tree-line are said to be climatologically the most likely sites around the arctic perimeter to register significant warming (Bradley p.c. 1995). Dr. G. Sirén, representing sub-arctic woodland studies in Fenno-Scandinavia, agreed that this paper's hypothesis could include his area of study, which has responded to regional warming (p.c. 1997). These findings from north central Canada, western Siberia and Yakutia, along with Dr. Sirén's Finnish experience, therefore indicates that much of the circumpolar arctic tree-line is currently responding to widespread intercontinental warming.

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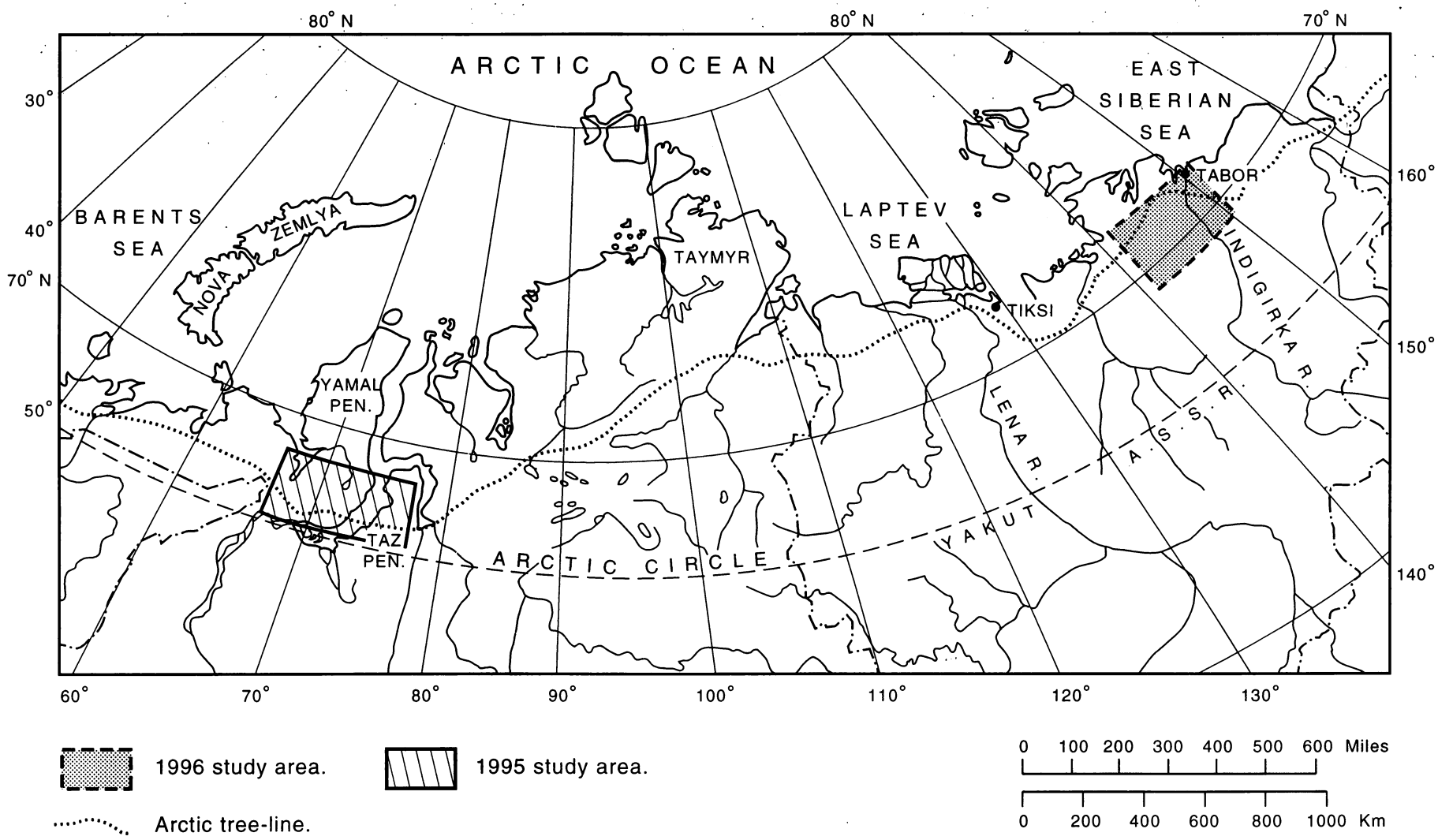

Figure 2.-- The two areas of artic tree-line study in western Siberia (1995) and Yakutia (1996).

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CONSEQUENCES OF A POSSIBLE FUTURE CLIMATE CHANGE FOR THE DISTRIBUTION OF TREE SPECIES IN SCANDINAVIA

Oddvar Skre[1]

ABSTRACT.--The respiration rates in growing shoots of Norway spruce (*Picea abies* (L.) Karst.) have been found to correlate well with the daily growth rates. Similar relationships have also been found in leaf discs of white birch (*Betula pubescens* Ehrh) and elm (*Ulmus glabra* var. *scabra* Huds). Long-term accumulated respiration rates correlate well with the distribution limits of these species. Predictions have been made on the consequences of the expected climate warming. Over time, the tree lines of the investigated species are expected to rise up to 400 m. Experiments with white birch indicate that this species has a high potential for temperature adaptation by different compensation mechanisms.

TREE LINE FORMATION

The objectives of the present review article are:

- to investigate how photosynthesis and dark respiration may be limiting growth in some tree species in Scandinavia;
- to see how summer and winter temperatures may influence the growth and carbon balance in trees;
- to predict from experiments how the expected global warming may influence tree-lines, growth and species composition.

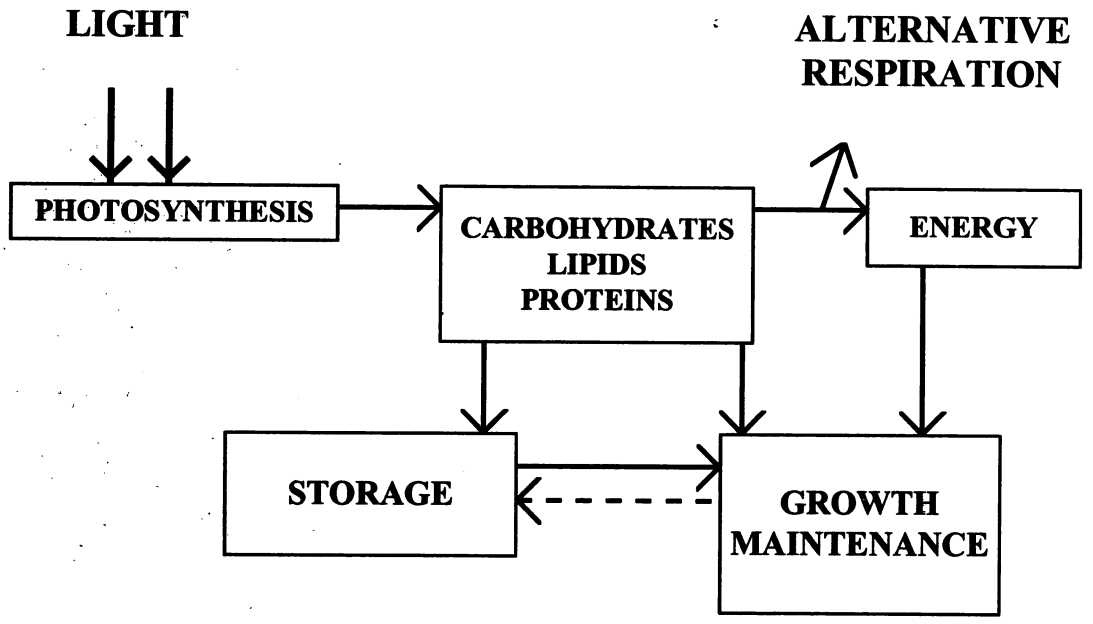
In alpine and arctic areas where climate is constantly changing between a cold and a warm season, most plants will have evolved different methods to survive the unfavorable season, and the selection pressure would tend to be determined by abiotic factors rather than by competition (Kallio 1984). Plants whose winter buds, bark and needles have developed a high tolerance against freezing and drying stress will, therefore, have an advantage. During the year, plants are subjected to many kinds of damage. To repair this and protect the living tissue against new damage, and for reproduction, growth and active uptake in roots, more energy is needed and is obtained by the production of ATP through photosynthesis and respiration (see fig. 1). Plants growing close to their distribution limits would therefore usually be restricted by demands for high temperatures to complete their life cycle and to produce viable seeds and/or winter buds (e.g., Heikinheimo 1932, Langlet 1960).

In the present study I will present some results from my own and other studies, that may indicate how the predicted global warming, caused by a doubling of the CO_2 concentration, would affect treelines and the distribution of plants. One of the most important distribution limits in Fennoscandia is the arctic and alpine tree line. The most important tree line forming species are mountain birch (*Betula pubescens*), Norway spruce (*Picea abies*) and Scots pine (*Pinus sylvestris*). According to the above-mentioned relationships, the development of flower buds (Hagem 1917) and vegetative buds (Romell 1925, Hustich 1944, Junttila & Heide 1981) in spruce and pine is mainly determined by the climate of the preceding summer. After flowering, the immature seeds start to develop embryo and storage tissue. Hagem (1917) and Heikinheimo (1921) found that the temperature during seed maturation was an important restricting factor and that embryos in the seeds from tree line areas were often weak and incompletely developed. According to these authors, the alpine tree lines of spruce and pine in Scandinavia are mainly seed maturation limits. In good seed-bearing years, however, viable seed may be dispersed more than 100 m above the position of their mother trees, giving rise to individuals that are only able to reproduce vegetatively. Scots pine lacks this ability, however, but compensates by a high seed production.

According to some authors (e.g., Michaelis 1934, Tranquillini 1979) reduced photosynthesis as a result of too short a growing season may cause increased winter desiccation in needles because of a reduced cuticle layer. Another explanation, however, is that low temperatures during the bud differentiation period in fall may produce immature buds and needle primordia, that are not able to survive the winter (Sarvas 1972; Dormling *et al.* 1968).

To account for the observed requirements for high temperature, various empirical parameters have been introduced (cf. Skre 1972). The most common is the

[1]*Oddvar Skre, Norwegian Forest Research Institute, N-5047 Fana, Norway*



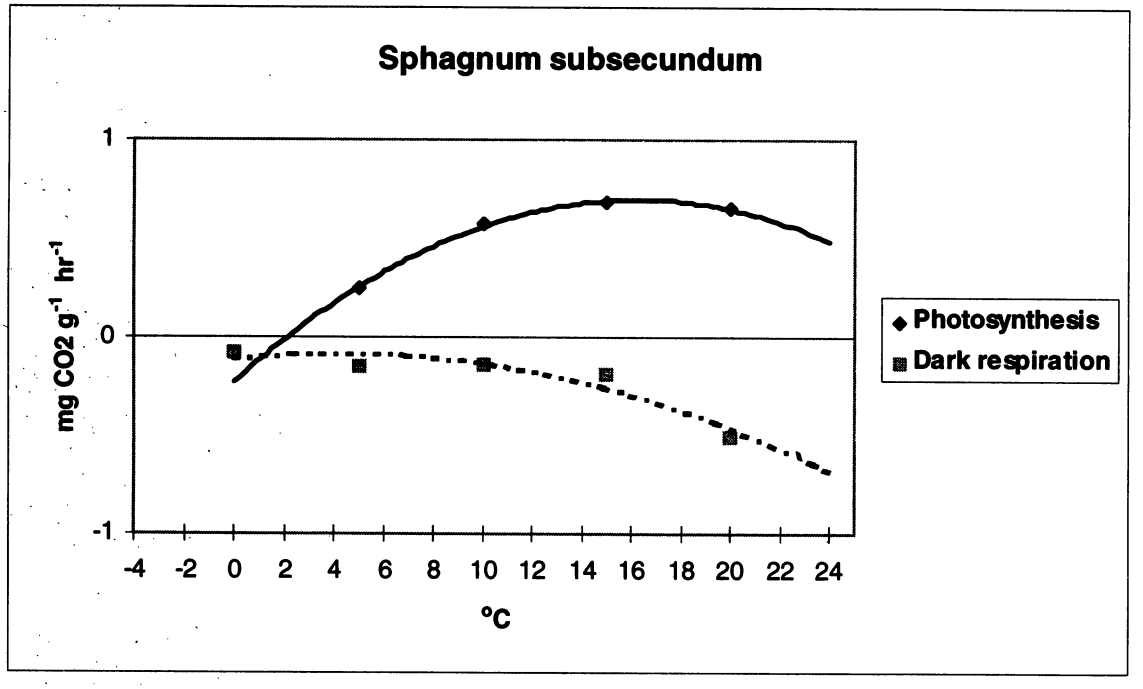


Figure 1.--*Diagram showing the flow of assimilates and energy in plants (top) and rates of net photosynthesis and dark respiration in a sub-arctic moss* (Sphagnum subsecundum) *(Skre & Oechel 1979) as a function of temperature.*

heat sum $W = (t-t_0)$, i.e., the accumulated temperature above a certain threshold value. In the calculation of heat sums, growth is assumed to be a linear function of temperature. However, no specified physiological process has yet been suggested to explain the distributions of plants restricted to areas with warm summers. One possible mechanism for the temperature effect on plant distribution has been suggested by Dahl & Mork (1959) and Skre (1972) based on data from Norway spruce (*Picea abies*). Through photosynthesis, solar energy is transformed to chemical energy and stored in different organic compounds. Differentiation and growth involve the synthesis of new plant tissue. This is an energy-consuming process depending on the supply of organic matter from photosynthesis as well as chemical energy released through the dark respiration process (fig. 1). The energy is released in small steps through the production of adenosine triphosphate (ATP) and other energy-rich compounds, which are translocated to sites where growth takes place. This part of the respiration process, which is called **growth respiration** (Penning de Vries 1972; Douce & Day 1985), is located in the mitochondria and is related to ATP production, while **the light-dependent photorespiration** (Zelitch 1966) and the alternative or **cyanide-resistant respiration** (e.g., Lambers 1980) do not involve such a formation.

Soil temperature is essential for the active uptake of nitrate and phosphate in roots (Karlsson & Nordell 1987). Also maintenance processes and protection against freezing and drying stress are strongly dependent on the supply of energy from ATP and other energy-rich compounds, through photosynthesis and respiration processes (fig. 1). Since there is an exponential decrease in respiration rates with decreasing temperatures, while photosynthesis rates are still high, respiration is most likely to be the limiting process at low temperatures. On the other hand, however, if plants are grown for some time at temperatures close to their compensation temperature t_k, growth may be limited by the supply of photosynthetic products. In the field this situation may occur during exceptionally mild winters, when net photosynthesis and t_k are lowered due to light limitation (e.g., Printz 1933, Hagem 1947). Experiments with birch and spruce seedlings, grown at ambient and at winter temperatures 4°C above ambient, indicated that although spruce was able to carry out some photosynthesis in needles even in a dormant state, the shoot and biomass growth the following season was lower in plants grown at high winter temperatures than in plants grown in ambient temperatures (Skre & Nes 1996). In white birch, on the other hand, high winter temperatures seem to enhance growth in fast-growing populations (fig. 2).

The daily growth in leaf biomass was measured in birch seedlings, grown at 24 hours daylength, and different constant temperatures (fig. 3). There was an approximately exponential increase up to 15°C -20°C, in accordance with respiration rates, while at higher temperatures growth followed a typical optimum curve, similar to the photosynthesis response curve. The figure indicates that dark respiration may be limiting growth at low temperatures and photosynthesis at high temperatures. In this case, a linear relationship would be expected between daily respiration and growth at low temperatures. Such a relationship was found by Dahl & Mork (1959) in growing spruce shoots from southeastern Norway. Generally, the exponential increase in dark respiration rates follows Arrhenius' law (1) for thermochemical reactions (Moore 1963):

(1) $Re = Ce^{-\mu/RT}$, $\ln Re = -\mu/RT \ln C$

The coefficient μ/R (cal/mol) is called the **activation energy**, and is a measure of the steepness of the temperature response curve. Instead of μ/R the ratio Q_{10} between the respiration rate at a given temperature and at 10°C lower temperature, is used. In many plant species with normal growth respiration, Q_{10} varies around 2.5, according to temperature, species and tissue type. Dark respiration in growing leaves of elm (fig. 4) showed a significantly lower temperature coefficient ($Q_{10} = 2.0$) than the corresponding respiration rates in growing spruce shoots ($Q_{10} = 2.65$).

RESPIRATION AND TREE LINES

Most Scandinavian plants are more or less restricted by their frost sensitivity as well as their high temperature requirement, as shown by Iversen (1944) on *Hedera helix* and *Ilex aquifolium* and Faegri (1960) on a number of oceanic plants. To simplify the problem, Iversen (1944) introduced a new method. Plant distributions were shown on a diagram where the mean July temperatures (t_7) were plotted against the mean January temperatures (t_1) of localities, and where the species were present or absent. The author claimed that frost itself, as well as a too short growth season, might induce frost damage on the trees. The method of Iversen (1944) has proved particularly

useful in studies of temperature responses in Scandinavian plants.

When the meteorological stations were plotted on such diagrams and the direction of the axis was reversed, the alpine and arctic distribution limits of a number of species appeared as straight lines with falling slopes. The reason for the falling slope is because plants need a higher summer temperature to complete their growth in a continental climate with cold winters and a short growing season than in an oceanic climate with mild winters and a long growing season. It may be shown (Skre 1979) that actual distribution limits of spruce and elm plotted on such **hermosphere diagrams** coincide with isolines representing certain constant annual respiration equivalents above the maintenance respiration in the two mentioned species (fig. 5). Because of its lower temperature coefficient for respiration, elm seems to be a species that is more adapted to a coastal climate than spruce. The good correlation between respiration values and actual distribution limits supports the main hypothesis about dark respiration as a limiting factor.

If the winter temperature is raised by 4°C and summer temperature by 2°C, as hypothesized in the climate scenario (Pedersen 1993), the distribution limits will move towards more arctic and alpine areas, in part due to higher respiration rates in summer and a longer growing season, represented by a higher January temperature. The effect of a higher winter temperature will tend to be stronger in oceanic than in continental areas in terms of longer growing season (H. Skre 1990). It will also be stronger in a plant with an oceanic distribution and a low temperature coefficient for growth like elm (*Ulmus glabra*) than in plants like Norway spruce (*Picea abies*) or mountain birch (*Betula pubescens*) with a high temperature coefficient 9 (see table 1.).

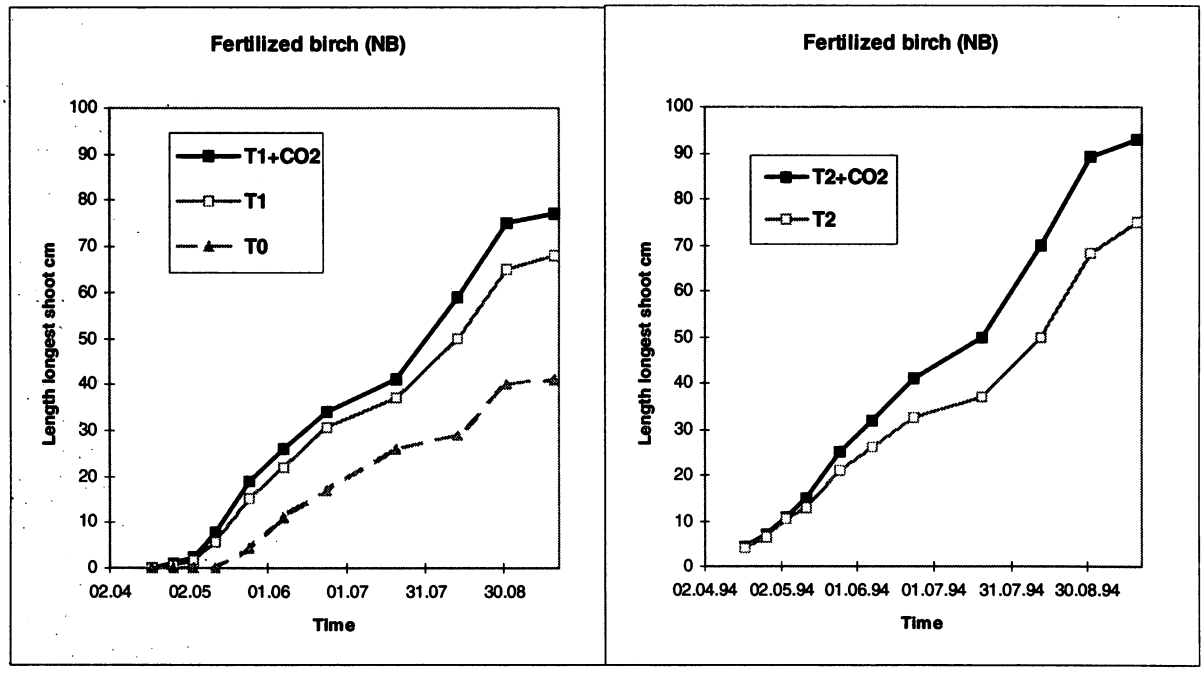


Figure 2.--*Shoot growth (length longest shoot in cm) of 1-year old white birch seedlings* (Betula pubescens) *during the 1994 season. The plants have been grown atambient (T1) and +4°C raised winter temperature (T2) the preceding winter (1993/94), followed by treatment in open top chambers at ambient (350 ppm) and high (600 ppm) CO_2 level. Dashed line indicates plants grown outside open top chambers at 2°C -3 °C lower temperature. The plants originate from a subalpine stand at 750 m altitude in South Norway.*

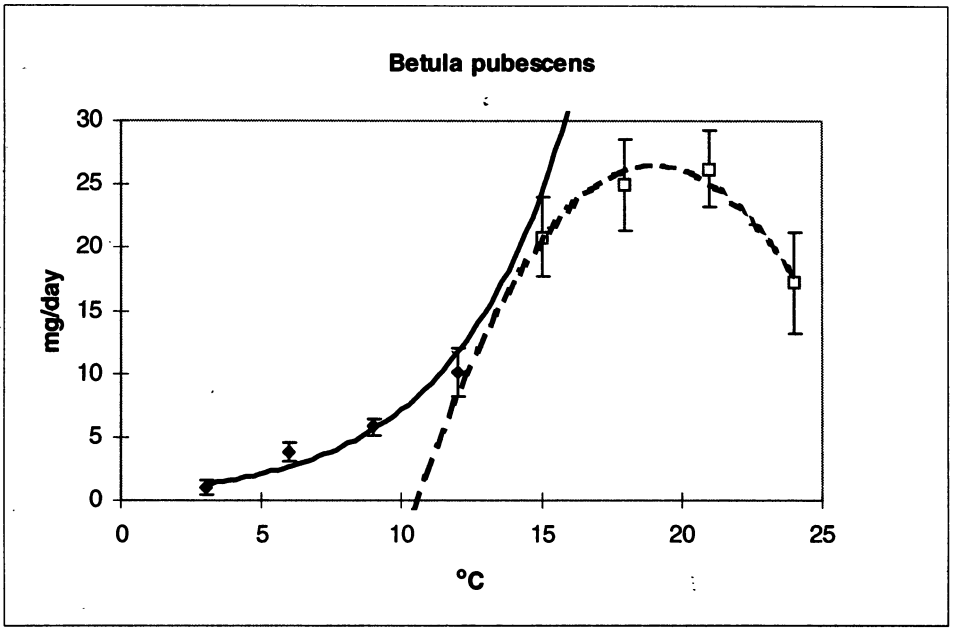


Figure 3.--*Growth in leaf biomass per plant(mg/day) during the linear growth phase in white birch seedlings from Ås, southern Norway (59°N, 95 m altitude) as a function of temperature at 24-hour photoperiod and 300 umolm^{-2}s^{-1} light. The observations are fitted to an exponential regression at temperatures at15°C or lower, and to a polynomial regression at temperatures above 15°C. After Skre (1991a).*

Table 1.--*The temperature coefficients (Q_{10}) of the dark respiration curve in growing shoots of spruce (*Picea abies*), grown at ambient temperatures (Dahl & Mork 1959) and in leaf discs of elm (*Ulmus glabra*) and three provenances of birch (*Betula pubescens*) grown at three different temperatures (9°C, 15°C and 21°C) in 24 hours daylength (Skre 1991, 1993a). In the table is also shown optimum temperatures t_{opt} (°C) and the Y threshold temperatures t_b (°C) for growth, estimated from the relationship between daily growth and dark respiration at different temperatures. The provenances are:*

BM = White birch, Ås (59°43'N, 10°40'E), 95 m altitude
BH = White birch, Blefjell (59°45'N, 9°28'E), 750 m altitude
BT = White birch, Tynset (62°13'N, 11°13'E), 800 m altitude
AA = Elm, Ås (59°43'N, 10°40'E), 95 m altitude
NS = Norway spruce, Hirkjølen (61°14'N, 10°52'E), 800 m altitude

Provenance Temp (°C)	Q_{10}	t_b (°C)	t_{opt} (°C)	Provenance Temp (°C)	Q_{10}	t_b (°C)	t_{opt} (°C)
BM				**BT**			
9	1.74	3.2	22.5	9	2.20	2.0	17.5
15	2.12			15	2.39		
21	2.22			21	1.84		
BH				**AA**			
9	1.87	1.6	21.5	9	1.94	1.2	17.0
15	2.72			15	2.05		
21	1.93			21	1.86		
NS							
ambient	2.65	2.6					

From figure 5 the increase in spruce respiration was found to be 1.47 units, corresponding to 2.6 °C higher July temperature, and in elm 1.55 units, corresponding to 3.0 °C higher temperature.

The spruce diagram shows that naturally-regenerated Norway spruce is absent from many localities along the coast of western Norway where it should be present according to the respiration hypothesis. When planted, however, it generally shows good growth (Robak 1960). Unstable snow cover that may influence survival rates of spruce seedlings, may be one reason (see Hagem 1947). It should be mentioned that in addition to a more unfavorable carbon balance, raised winter temperatures may also cause earlier dormancy breaking in some species, like birch and Norway spruce, that may leave the trees more exposed to spring frost damage (Murray *et al.* 1989, Hänninen *et al.* 1993).

Methods are available for calculating long-term respiration equivalents based on data published in meteorological summaries (Skre 1971, 1972). There is generally a linear decrease in temperatures with increasing altitudes (Werner-Johannessen 1960) varying from 0.4°C per 100 m in winter to 0.7°C in summer. As respiration is an exponential function of temperature, there must be an approximately exponential decrease in the annual accumulated respiration equivalent with increasing altitude. Because of this exponential decrease, the difference in altitude corresponding to one annual respiration unit increases with altitude (see fig. 6). As a result of the expected climatic warming the spruce distribution limit is are therefore expected to be raised by about 400 m over time, the birch limit by 450 m and the elm limit by about 350 m (Skre 1990).

GROWTH RESPONSES IN BIRCH

To investigate the influence of the expected climatic change on the respiration loss during winter and the physiological status of the trees, some results are shown from a field experiment comparing mountain birch seedlings grown in fertilized peat on an oceanic lowland site at Fana (50 m) and on a tree line site at Kvamskogen (450 m altitude).

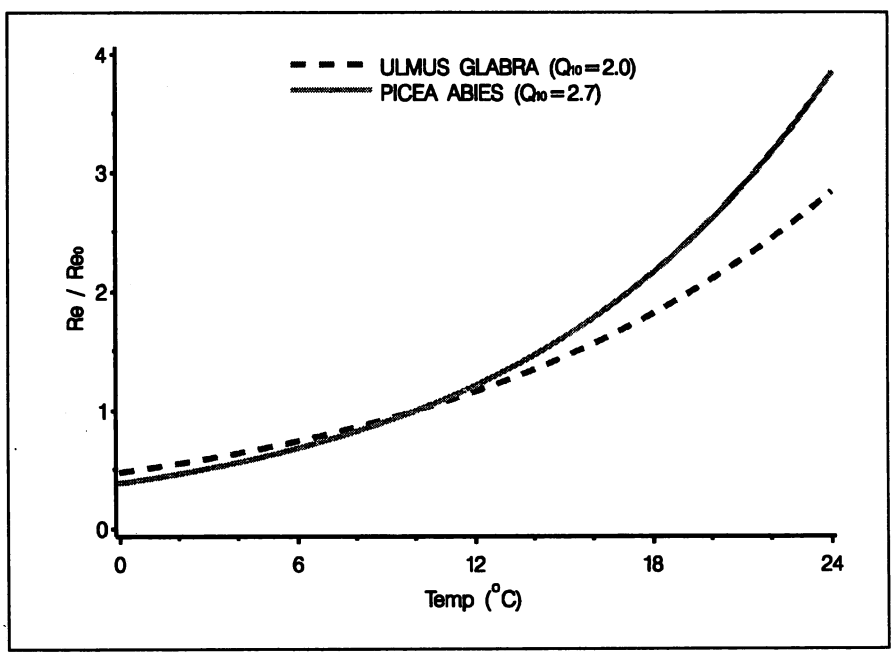


Figure 4.--*Dark respiration rates in growing shoots of spruce (Dahl & Mork 1959) and leaf discs of elm, measured in relative units, where the respiration at 10 °C is taken as unity. The elm leaves were grown at 15°C constant temperature.*

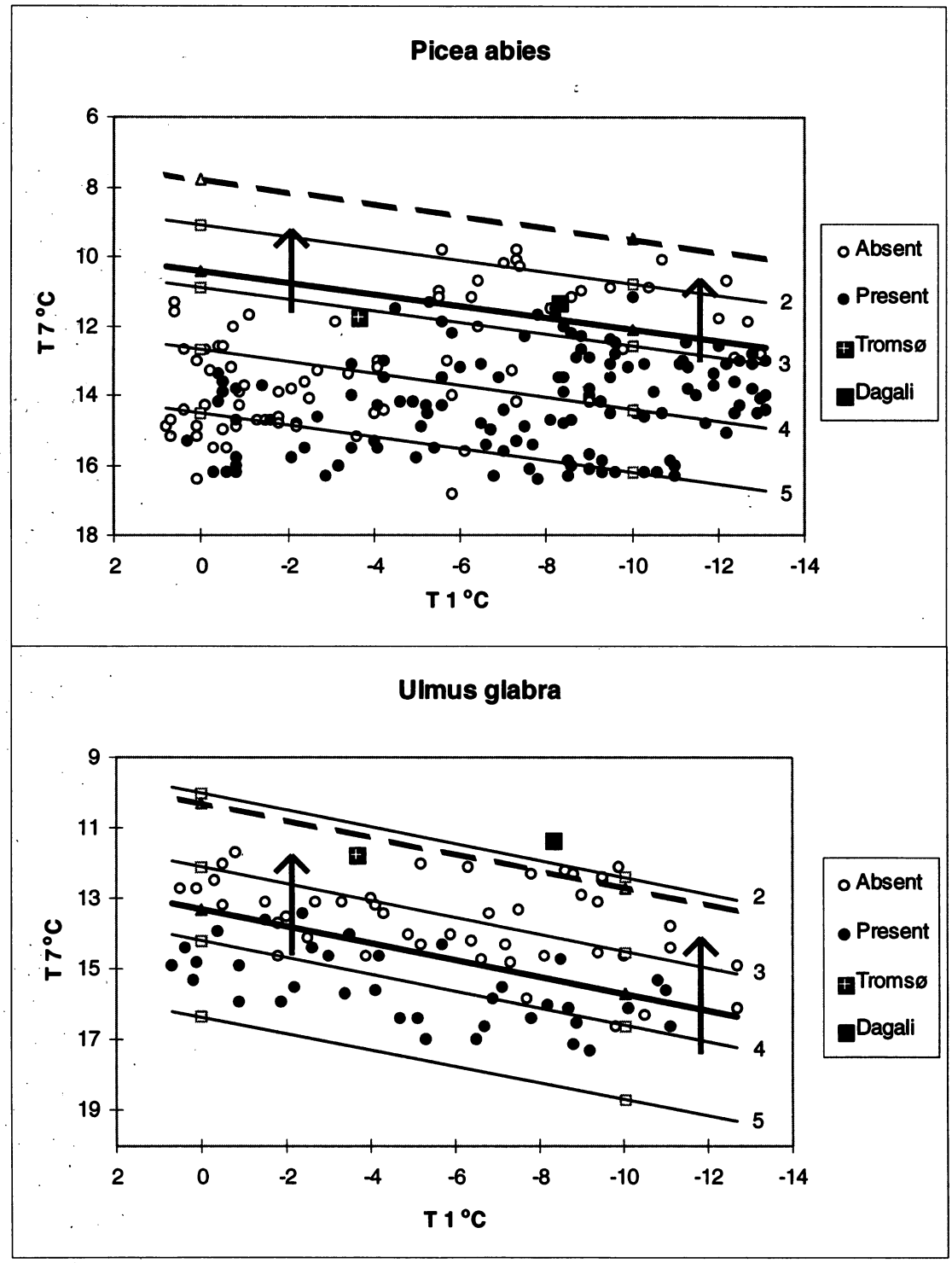


Figure 5.--*The distributions of* Picea abies *(top) and* Ulmus glabra *(bottom) shown in a thermosphere diagram where the mean January temperatures (T1) are plotted against the corresponding July temperatures (T7). The present distribution limits (thick solid lines) coincide with isolines representing a constant accumulated annual growth respiration equivalent to 2.70 and 3.42 units respectively, based on the figures in fig. 3. Main isolines with arrows and the effects of an expected temperature rise of+2 °C in T7 and +4 °C in T1 on the distribution limits are indicated Y are shown with thin solid lines (dashed lines). The two subalpine localities Tromsø and Dagali are plotted on the diagrams.*

A comparison between the sites (fig. 7) shows approximately twice as high growth rates at the former locality. Birch seems to invest a relatively large part of its resources into new growth of structural and photosynthetic tissue. By increasing their photosynthetic capacity, plants are able to replace some of the carbohydrates that were consumed in the growth process (see Thornley 1972), this tendency is usually strongest in fast growing southern ecotypes (BAM) and in lowland birch (BAL).

The second year (1987) growth rates at the lowland site of Fana had increased to about four times the corresponding growth rates at Kvamskogen in lowland birch and southern ecotypes of mountain birch seedlings, while there was almost no effect of altitude on growth in the northern ecotype BJ from Finnish Lapland. This is partly an effect of different daylength reactions, but experiments in a controlled environment also showed that the northern as well as the southern alpine (BH) ecotypes were adapted to a low-growth strategy, where carbohydrates are stored in woody tissue, mainly in roots, rather than being invested in new aboveground growth. This agrees well with earlier findings on alpine plants in e.g., Alaska (Chapin 1979). In a nutrient-poor environment with low competition, plants tend to avoid growth rather than producing nutrient-deficient tissue. However, there is also a strong acclimation effect of temperature (cf. Billings *et al.* 1971). At medium temperatures (10°C -15°C) when photosynthetic rates were high, the shoot/root ratios, as well as the specific leaf area and leaf area ratio, were low, indicating a higher proportion of structural tissue than when temperatures were higher and less favorable for photosynthesis (figs 8-9). By increasing root growth, plants may be able to absorb more nutrients from the soil and increase their photosynthetic capacity (cf. Thornley 1972). The plants grown at high temperatures also increased

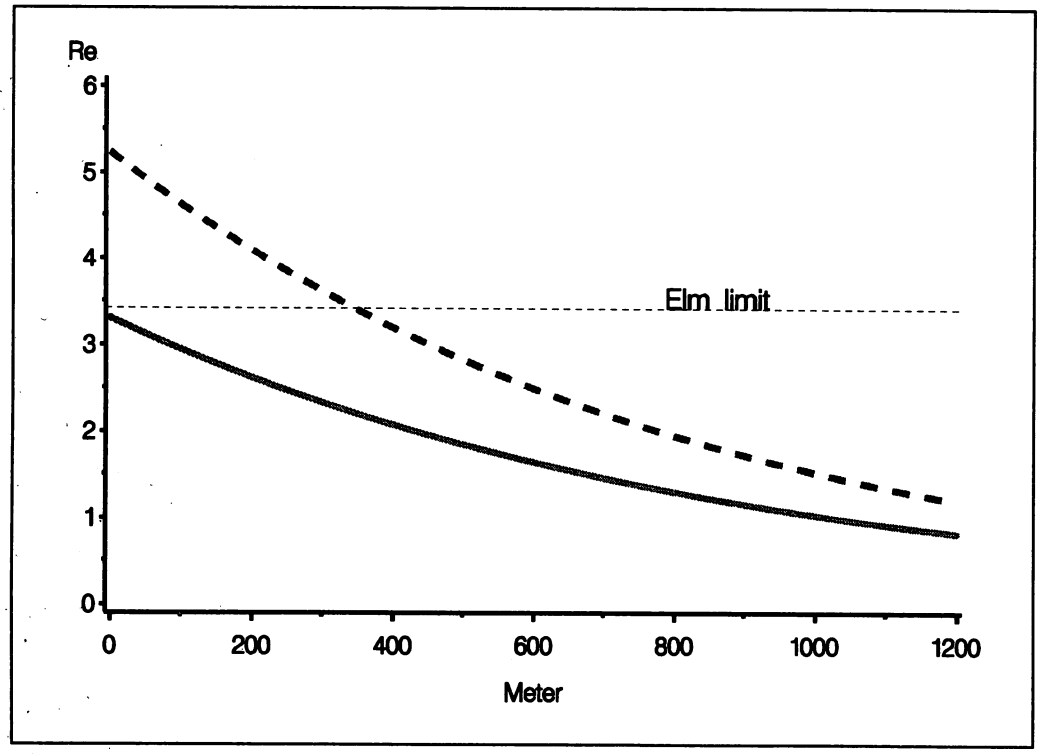


Figure 6.--*Annual growth respiration in elm leaves before (solid line) and after the climate change at sheltered localities, as a function of altitude in inner Troms, North Norway. The tree line of elm is indicated by the horizontal dotted line). After Skre (1983).*

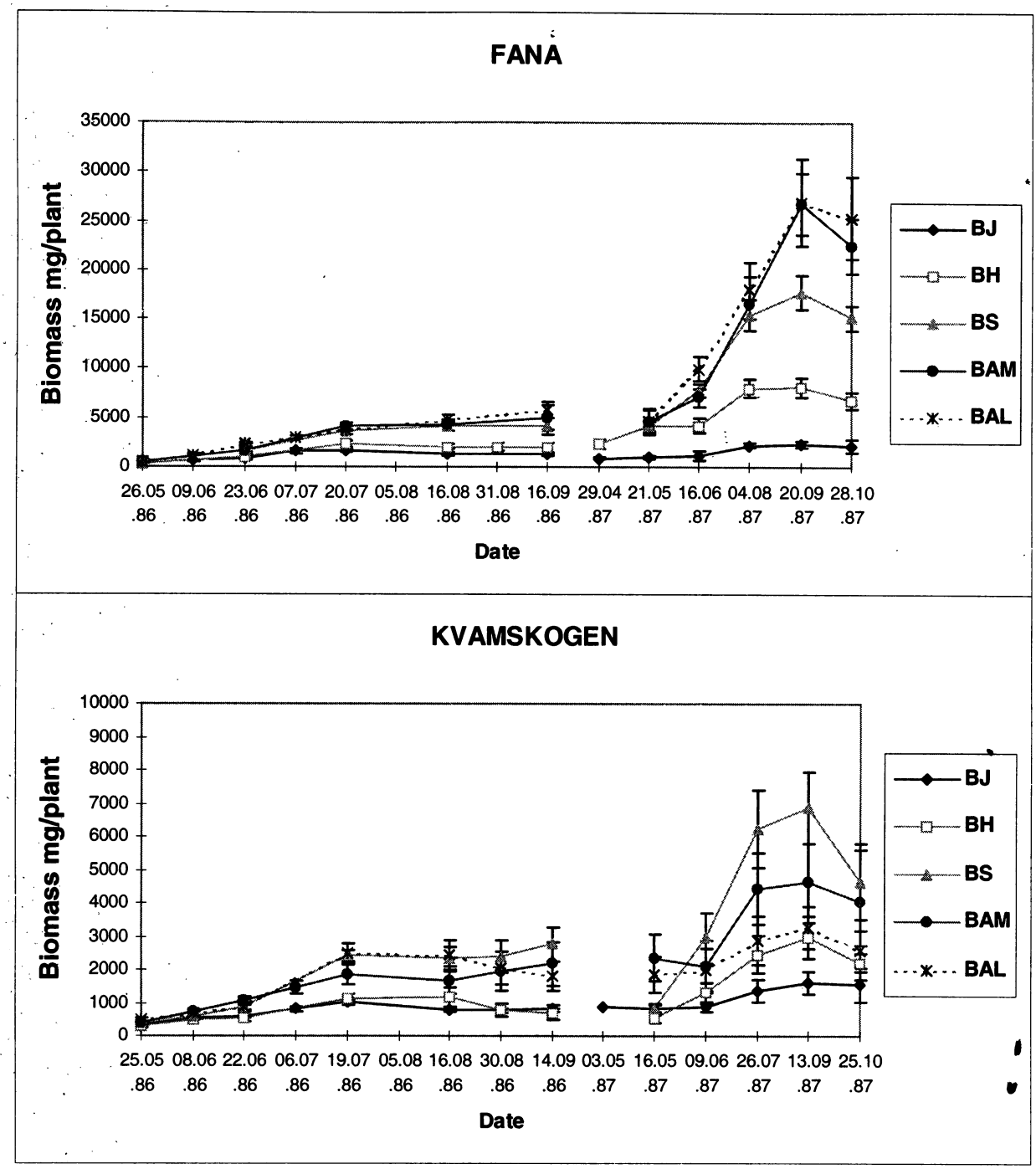


Figure 7.--*Biomass (mg/plant) of white birch (*Betula pubescens*) and silver birch* (B. pendula) *seedlings grown in fertilized peat from the 4-leaf stage during the 1986 and 1987 seasons at Fana (50 m) and Kvamskogen (450 m altitude) in western Norway according to Skre (1993c). The ecotypes are:* Betula pubescens. *BJ:Kevo, northern Finland (69°44' N) 200 m, BH - Blefjell, southern Norway (59°45' N) 750 m. BS: Fana, western Norway (60°16' N) 50 m, BAM - Løten, eastern Norway (60°51' N) 200 m.* B. pendula. *BAL: Løten, eastern Norway (60°51' N) 200 m.*

their photosynthetic rates and their optimum and compensation temperatures for photosynthesis (fig. 10) relative to plants grown at lower temperatures. Respiration rates increased in leaf and stem tissue as a response to low temperature treatment, and decreased in roots (Skre 1993c). The arctic population (BJ) had slightly higher photosynthesis rates that the southern populations. This is called metabolic compensation (Crawford 1989, Mooney & Billings 1961) and is looked at as a compensation for shorter growing season in northern areas.

Chemical analysis of birch tissue confirmed the results of the growth measurements (figs 11-12). White birch seedlings increased their nitrogen and chlorophyll content in leaves as a response to elevated temperatures at the same time as there was a strong decrease in the total nitrogen and carbohydrate content in stem and root tissue. At low temperatures there was also accumulation of soluble proteins in leaves, indicating increased frost resistance, most pronounced in the arctic population (cf. Parker 1963, Volger & Heber 1975).

CONCLUSIONS

In conclusion, experiments have shown that the distribution limits of spruce and elm seem to coincide with their annual accumulated respiration rates, indicating that dark respiration is an important limiting factor. The growth of spruce shoots and birch and elm leaves was closely correlated with their respiration rates at low temperatures.

The results indicate that white birch to a certain extent is able to compensate for decreased photosynthesis and source strength at elevated temperatures by increasing its photosynthetic rates and optimum temperatures for photosynthesis. It also tends to increase its shoot elongation rates, leaf area ratio and the nitrogen and chlorophyll content in

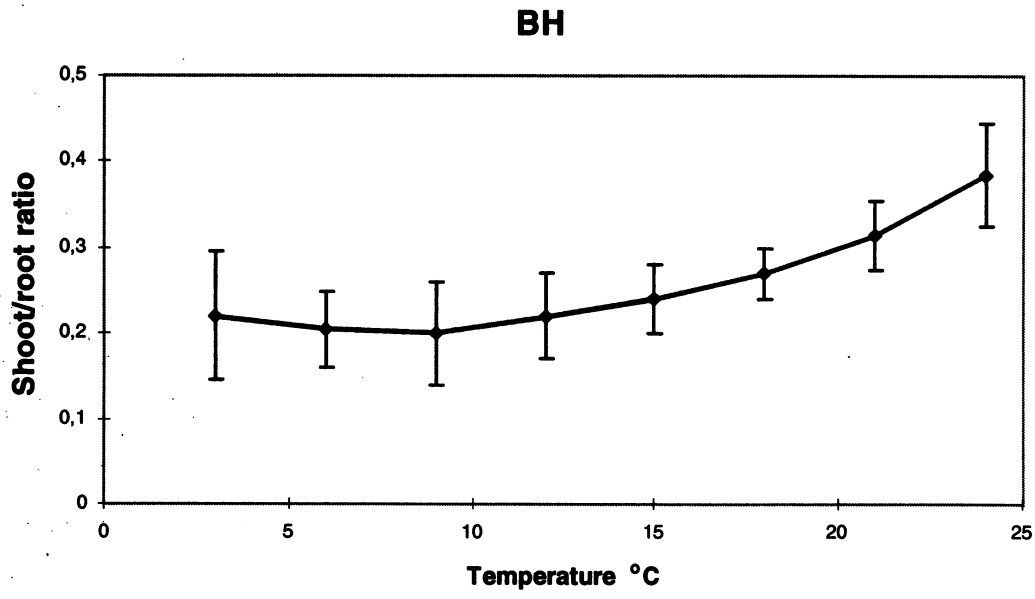


Figure 8.--*Shoot/root ratio with confidence limits ± 2 s.e., shown as a function of temperature in 6-week old sub-alpine white birch seedlings from Blefjell, southern Norway (Skre 1991a).*

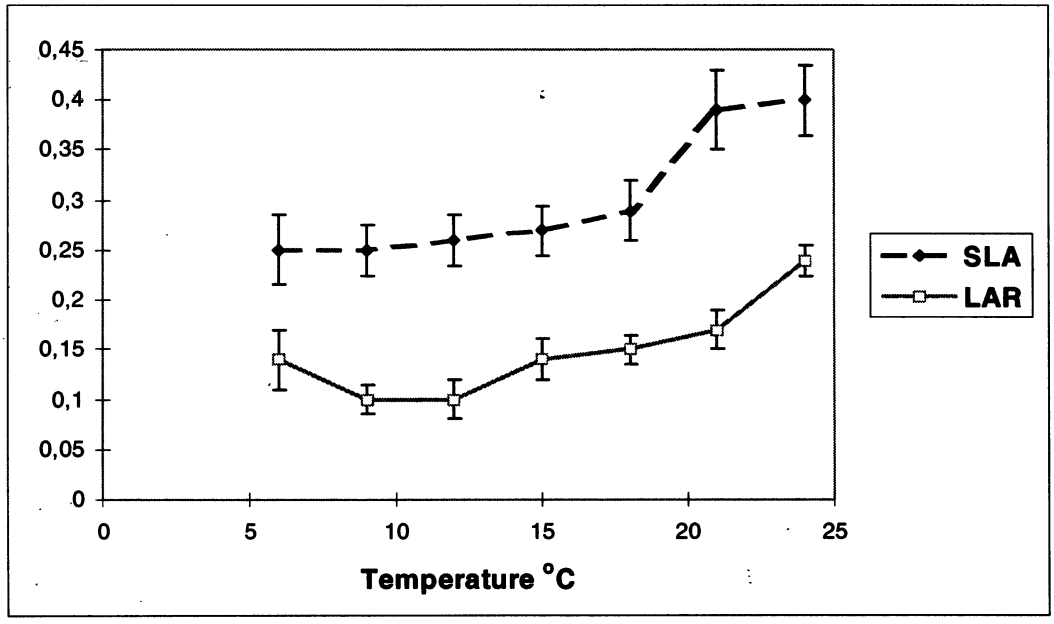


Figure 9.--*Leaf/area ratio (LAR) and specific leaf area (SLA) , Y being a function of temperature, in 6-week old sub-alpine white birch seedlings from southern Norway (Skre 1991a).*

leaves. Compared with evergreens like spruce, white birch seems to have a higher ability to take advantage of increased CO_2 and nitrogen content in soil, and seems to be more tolerant against elevated winter temperatures.

The results indicate that the expected global warming would result not only in much higher growth rates but also in lower sink strength and less available energy resources. Fast-growing species and ecotypes would outcompete slow-growing ones. If the process went too fast, the time would be too short for plants to develop new ecotypes that are adapted to the new climate. Photosynthesis will take over as the growth-limiting process at ambient field temperatures and many plants would therefore be restricted not only by their upper distribution limit but also by a lower limit. Printz (1933) suggested that high winter temperatures might restrict spruce growth in West Norway, but his theory was later rejected (Hagem 1947). However, future climatic change might again lead to a situation like the one predicted by Printz (1933).

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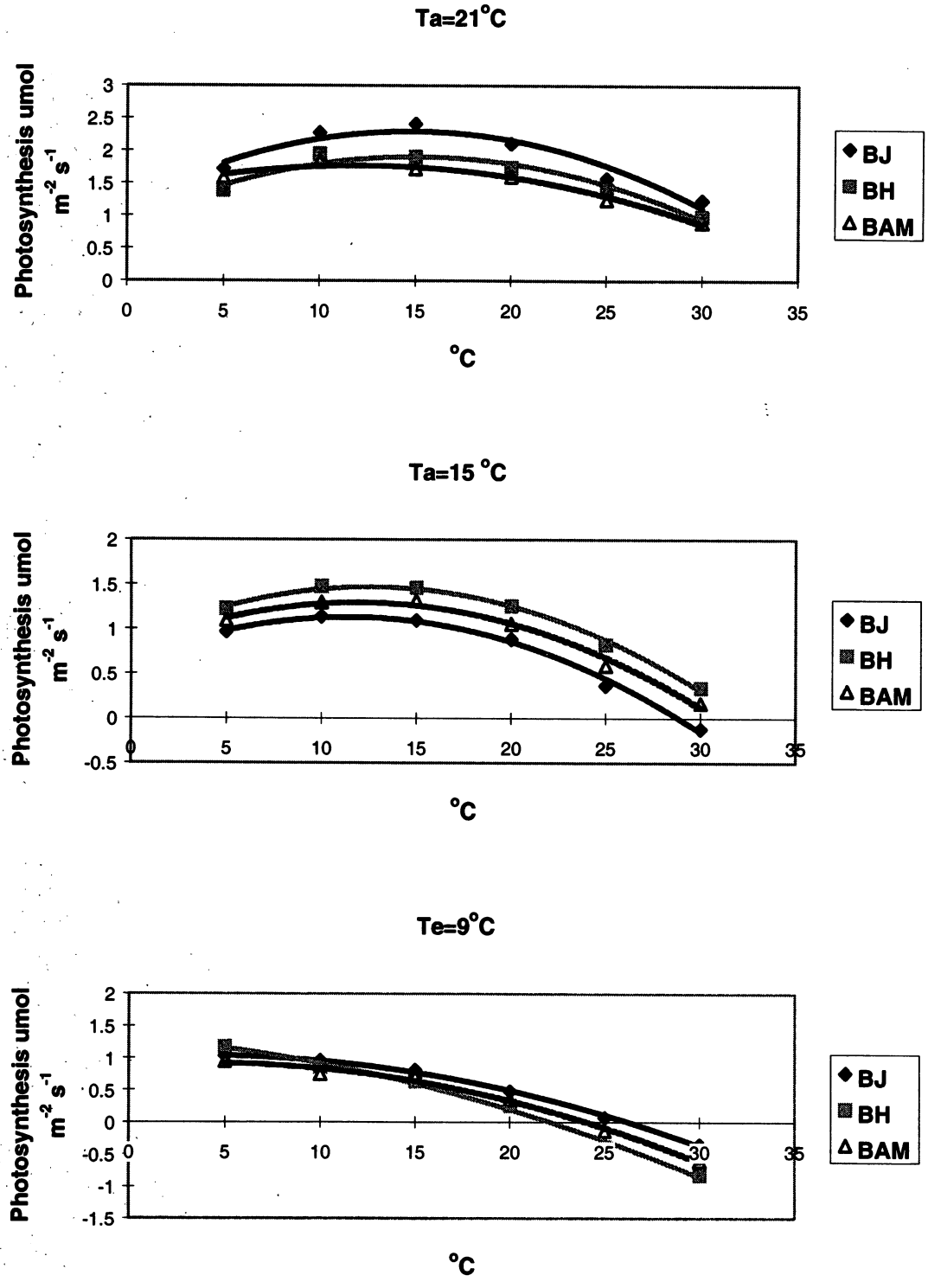

Figure 10.--*Photosynthetic rates (μmol CO_2 $m^{-2}s^{-1}$) at 100 μmol $m^{-2}s^{-1}$ photon flux density in leaf discs of mature leaves from three white birch populations as functions of temperature after growing for 3 weeks in an 18-hour photoperiod at 21°C, 15°C and 9°C. For explanation of symbols, see figure 7.*

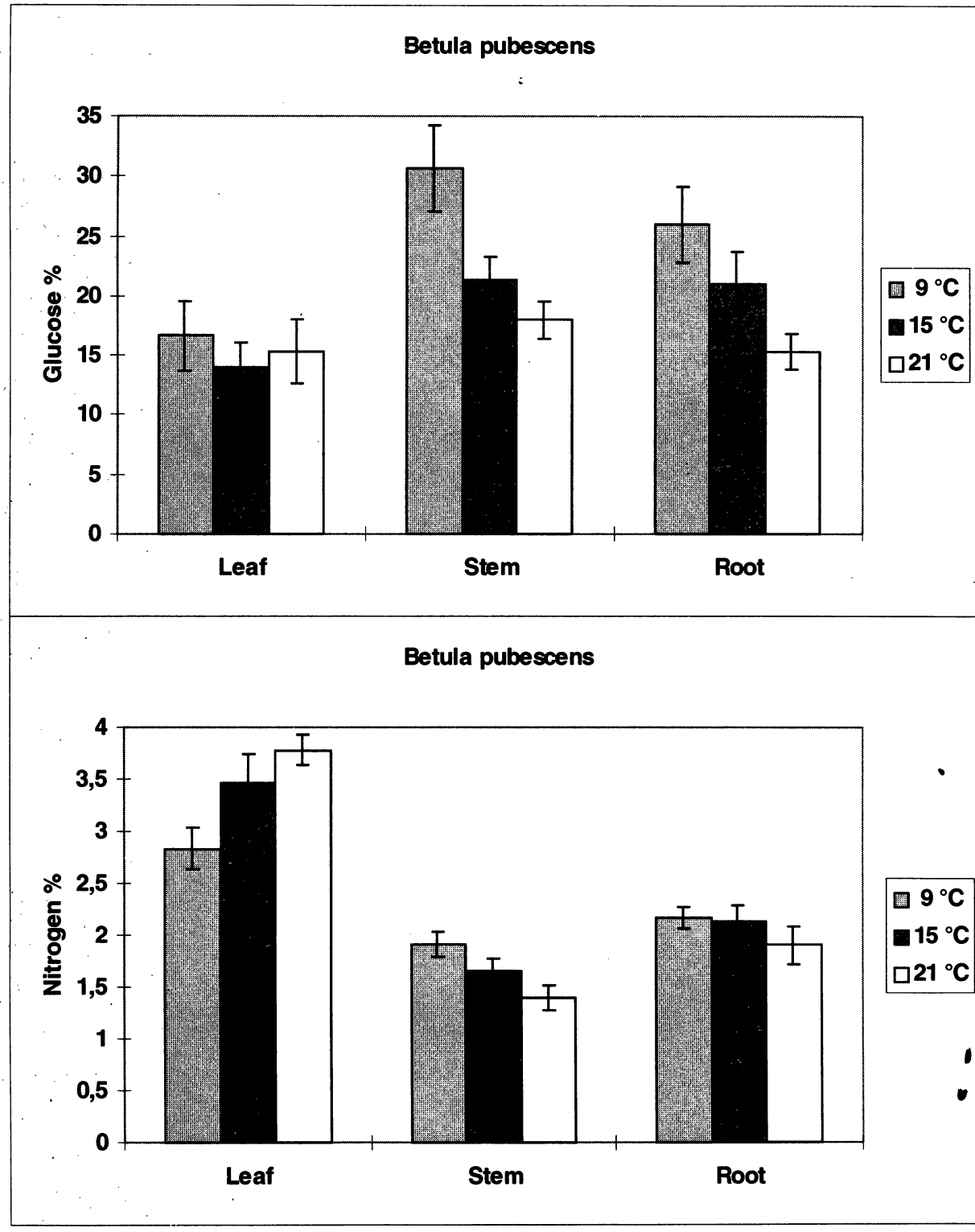

Figure 11.--*The amounts of total non-structural carbohydrates as glucose equivalents (top) and total nitrogen (bottom) with confidence limits ± 2 s.e. in leaf, stem and root tissue of 6-week old sub-alpine white birch seedlings from southern Norway, after growing in 9 °C, 15 °C and 21 °C, when averaged over the entire photoperiod. After Skre (1991c).*

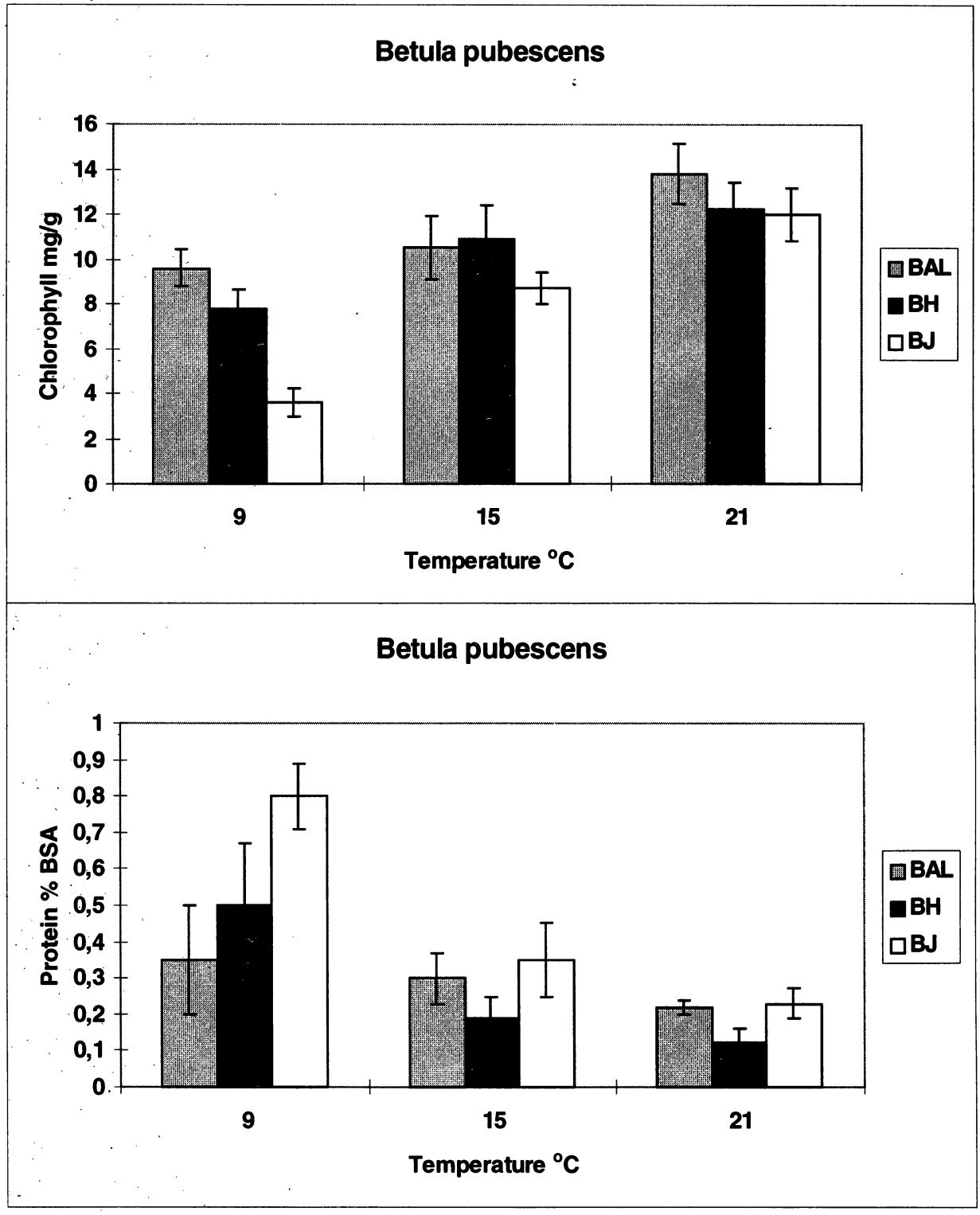

Figure 12.--*Chlorophyll concentrations in leaves (left) and water-soluble protein content as bovine serum albumine (BSA) equivalents (right) with confidence limits of ± 2 s.e. in leaves of three birch populations after growing in 9 °C, 15 °C and 21 °C, when averaged over the entire photoperiod (Skre 1991c). For explanation of symbols, see figure 7a.*

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BIOMASS OF FOREST PINE ECOSYSTEMS OF MIDDLE ZAVOLGIE: TO THE QUESTION OF GLOBAL WARMING

Eldar Kurbanov, Petr Sokolov[1]

ABSTRACT.--Accurate region-wide estimates of forest biomass are the main factor in further understanding of carbon pools and simulations of fluxes. In the procedure for estimating biomass in the Middle Zavolgie region of Russia, we generally obtain data from sample plots established in the pine ecosystems. For each biomass component, analysis of suitability of the most common forms of simple linear regression equations was made. Data obtained on forest pine biomass allows us to analyse main trends in the dynamics of different forest ecosystem fractions. The study indicates a great potential biomass growth in normal pine stands.

INTRODUCTION

The amount of carbon in the forest ecosystems of the Earth in the near future will be of special interest among scientists. Increasing carbon dioxide (CO_2) concentration in the atmosphere is resulting in enhanced greenhouse effect and global warming, which according to pessimistic prognoses could result in more melting of polar ice and flooding over lowlands (Houghton and Woodwell 1989), droughts, and more frequent forest fires.

As for the amount of carbon production and the duration of its sequestration, the forest ecosystems of the planet are considered to be the most reliable system which may help in mitigation of the enhancing greenhouse effect. In fact, several assessments (Tans *et al.* 1990; Kauppi and Tomppo, 1993; Dixon *et al.* 1994) show that forests in the temperate and boreal zones have been carbon sinks for the past few decades.

Carbon has been accumulating in the atmosphere since the large-scale exploitation of forests began, and during the last 100 years carbon accumulation has been enhanced considerably due to increasing use of fossil fuels. Changes in land use and land management, especially deforestation, cause significant changes in terrestrial carbon storage, resulting in exchange of CO_2 between the land and atmosphere. The carbon that was bound in forests and in fossil fuels has been released back into the atmosphere, and growth of the remaining forests and uptake by the oceans have not been able to absorb all the "additional" carbon (Houghton *et al.* 1990). Recent estimates of the global C cycle indicate that combustion of fossil fuels accounts for 5.5 ± 0.5 Pg C/yr (Pg = 10^{15} g) and changes in land use and deforestation for 1.6 ± 1.0 Pg C/yr of the total net anthropogenic sources of 7.1 ± 1.1 Pg C/yr (Shimel 1995). Uptake by the ocean is estimated to be 2.0 ± 0.8 Pg C/yr, and accumulation in the atmosphere to be 3.2 ± 0.2 Pg C/yr.

It is important to study the C sequestration of forest ecosystems since they contain over 75% of the carbon accumulated in terrestrial vegetation (Olson *et al.* 1983), and changes in this reservoir are large enough to affect global climate (Smith and Shugart 1993). On the other hand, the predicted climate change may alter the productivity of forest ecosystems, thus, altering their ability to sequester carbon. Increasing CO_2 concentration in the atmosphere also tends to increase the growth of vegetation, at least in the short term (Houghton *et al.* 1992).

It is well known that by improving methods of forest production, optimization of age and species structure of forest ecosystems, and additional reforestation of the areas there is an opportunity of sequestering a certain amount of CO_2 from the atmosphere into phytomass. In connection with the above, mankind on the border of the third millennium is coming to a new ecologically-oriented paradigm on the utilization of forest ecosystems. That, undoubtedly, would require reconsideration of some points and matters in the policy of forest utilization and forest production due to the development of new tendencies in forest management. In connection with this, at the latest environmental conferences held by the UN much greater attention has been paid to the Earth's forests.

[1] *Eldar Kurbanov, assistant professor, and Petr Sokolov, professor, Mari State Technical University, Department of forest mensuration and inventory, Yoshkar-Ola, Pl. Lenina, 3, Russia. E-mail: eldar@marstu.mari.su.*

Expected warming has been estimated to alter the distribution of vegetation over the globe, especially in the boreal and temperate zones (Smith *et al.* 1992, Monserud *et al.* 1993). Because boreal forests store a large amount of carbon and are located in latitudes predicted to undergo the greatest warming, they may play a key role in future carbon fluxes (Dixon *et al.* 1994).

Russia has approximately 30% of the boreal forests (Murray 1993), and together with Canada and the USA these are known to be one of the main reservoirs of sequestered carbon, which is becoming an object of the world community's attention.

The boreal forests of higher latitudes, unlike the forest ecosystems of the tropical and subtropical zones, are formed by a small number of tree species and are characterized by slow biological cycles. The amount of annual fall of plant residues in such ecosystems exceeds average annual decomposition. In this connection, the boreal forests accumulate carbon not only in the wooden biomass but also in litter, coarse woody debris, soil organic matter and peat.

The aim of this study was to examine biomass dynamics with age in pure normal (fully stocked) and modal (typical) pine stands under current climatic conditions at the stand level in the Middle Zavolgie region of Russia. Ecosystem components that were considered in the assessment include all tree fractions and understory (undergrowth, bushes and forest floor). The investigation of normal and modal pine stands forms the basis for understanding the dynamics of C sequestration for the conditions of Middle Zavolgie and provides a baseline for further simulations of carbon cycles.

FOREST GROWTH CONDITIONS IN MIDDLE ZAVOLGIE

Territory

Middle Zavolgie is located in the eastern part of the Russian plain. This region includes the Volgo-Vyatski and Povolzskij economy areas including the republic of Mari El and Tatarstan, Nizegorodskaya and Kirovskaya oblasts (247 thousand km^2). Middle Zavolgie includes three forest growing zones: taiga (southern part), mixed forest, and forest-steppe, and therefore has appreciable landscape diversity. The Volga river serves as the natural western border of physiographic and natural conditions. The territory north of a line formed by the Volga River (east through Kazan) and the right bank of the Kama River, and extending up to eastern Tatarstan border, belongs to the mixed forest zone. The forest-steppe zone extends to the south of this line. The taiga zone includes the territory of the northern Kirovskaya oblast.

In general, the land surface of the territory is a hilly plain, the height variations being almost unnoticeable (from 45 m to 275 m above the sea level), and transitions from the lower to the higher parts are quite gradual. The highest parts are located in the southeast. From here the surface descends towards the Volga River valley, and along the river there extends central lowland in the form of a wide band with a great number of lakes, marshes and rivers. Above the valley of the Volga-river rises a high right bank in the form of a steep ledge with deep ravines and gullies.

Soil factors rightfully have a main role in forming of certain types of forest ecosystems within congeneric climatic conditions. On the left bank of the Volga, there are mainly sandy and sandy loam soils, and in its eastern part - loam, turf-carbonate, brown-gray and turf-ashen-gray forest soils. The most widespread soils in the taiga zone of Middle Zavolgie are podsol types, which in turn divided into three subtypes: gley-podsol, typical-podsol and humic-podsol.

Climate

The climate of Middle Zavolgie is determined by the continental situation of the region, landscape relief and rotational impacts of the air mass from the Atlantic and Nordic oceans and the dry continental mass from Kazakhstan and Central Asia. The climate of the Mari El Republic is considered to be moderately continental with relatively stable weather in winter and in summer, but changeable and varying conditions in spring and in autumn. The mean annual temperature varies from +2.2^0 C in the northeastern regions, up to +3.1^{0}C in the southwest. In recent years, winter months have been warmer than they used to be. An average amount of precipitation is 450 mm-500 mm, of which 200 mm-250 mm falls during the vegetation period. On the whole, Middle Zavolgie is located in the zone of variable moisture; years and seasons with sufficient and sometimes even with excess precipitation are interspersed with dry ones.

Progress in growth and productivity of forest ecosystems depends also on the duration of the vegetation period. Its beginning and end are roughly determined by mean daily air temperature of +10^{0}C, and soil temperature of +5^{0}C. Accordingly, annual productivity can be related to the sum of active temperatures greater than +5^{0}C or +10^{0}C, or to the length of the period when the plants have green-

leaves. The duration of the period with mean daily air temperature higher than $+10^0$C is typically 150 days.

The Main Forest-Forming Species

On the territory on the right side of the river, dominant tree species are Scots pine (*Pinus sylvestris*), birch (*Betula pendula* and *pubescens*), spruce (*Picea abies* and *obovata*), aspen (*Populus tremula*). Oak (*Quercus robur*) is not as widespread. Tree species proportion of the wood stock in Middle Zavolgie can be seen in fig. 1.

The main landscape background in the natural complexes of the forest group is created by pine stands, which occupy 32 % of the total forest area of $3,197 \times 10^3$ hectares. Pine occupies vast territories in Kirov and Nizegorodskaya oblasts. Birch stands cover 29% of the forest area, mainly young and middle-aged stands appearing on the vast areas of slashes and clearings. Spruce stands occupy expansive territories in the north-western and northern regions of the republic. Aspen is common in mature and young stands. Considerably less widespread are lime (*Tilia*), alder and oak forest ecosystems. With respect to annual carbon sequestration, the principal species are pine, birch, larch, spruce, and aspen. The most promising among them in terms of duration of the annual period of carbon accumulation under the conditions of the Middle Zavolgie is Scots pine.

Among pine species Scots pine occupies the largest area and is believed to accumulate the greatest biomass. Owing to being frost-resistant, drought-resistant, and hardy to a range of soil conditions, it grows on the plains and in the mountains, forming island stands in the forest-steppe and steppe zones. Scots pine's ability to grow in different climatic and soil conditions has contributed much to the formation of both pure stands and mixed stands with spruce and birch--which are very diverse in composition and productivity. Pine possesses greater stability to wind than other species. In the areas near pine forests, the amount of precipitation and the air humidity within the vegetation period is higher than in the open steppe, which allows cultivation of agricultural crops even in dry years. Unlike other species, which are characterized by high increment of wood and short lifespan (predominantly broadleaf softwood species), pine stands are characterized by stable growth and high quality. The pine forest type can be both indigenous and derivative. After harvesting and fires, pine usually regenerates quite well.

THE STATE OF THE PROBLEM--FORESTS AND GLOBAL WARMING

The latest publications and research on the subject of forests and carbon storage could be divided into the following subject areas:

1. Evaluation of the carbon stock and its annual sequestration in the forest biomass. (Karjalainen 1996, Kauppi *et al.* 1992; Alekseyev and Birdsey 1994; Isaev *et al.* 1993; Isaev *et al.* 1995).
2. The prognosis for dynamic changes under different climatic scenarios due to increasing atmospheric CO_2 concentrations (Dixon 1994; Melillo 1993; Cannell 1995; Zavelskaya 1993).
3. Determination of the anthropogenic emission of CO_2 and opportunities for its mitigation through the photosynthesis process and carbon sequestration by vegetation (Dudek 1990; Schroeder 1991; Utkin 1995).
4. Global carbon cycles in natural ecosystems (Dixon 1991; Tarko 1994, Kobak 1988).
5. Biological and chemical carbon emission in the form of CO_2 into the atmosphere caused by forest fires and forest utilization (Fosberg 1996; Furyayev 1996).

Among all the areas listed, the first topic is of the most applied significance in the problem of possible global warming. After determining the patterns and the dynamics of organic carbon sequestration by the phytomass of forest ecosystems, one could mitigate possible global climatic changes through enhancing carbon sequestration in the forest ecosystems by applying science-based methods and approaches.

Recent assessments of the organic carbon stocks in Russia are based on several methodological approaches. Most of the researchers are known to use various maps combined with databases to differentiate all the vegetation layers of the forest ecosystems into phytomass fractions (Bazilevich 1993).

Another group of researchers uses results from forest inventory, taking into account total areas and stocks of wood with respect to age groups of the main species. Then, through conversion coefficients that express relationships between the volume stock of the stemwood and the mass of various tree fractions (wood, bark, branches, roots and leaves) calculation is made of the total phytomass stocks per unit area. Determinations of the carbon have been applied through absolutely dry mass as well as through conversion coefficients proposed by (Kobak 1988).

In Russia there are lot of large research centers engaged in forest studies and investigation. But

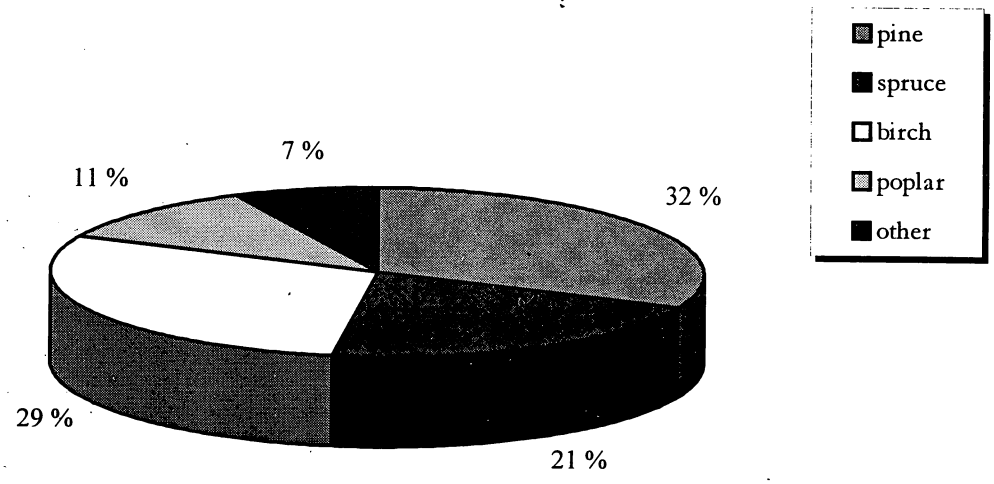

Figure 1. *Main species in the Middle Zavolgie.*

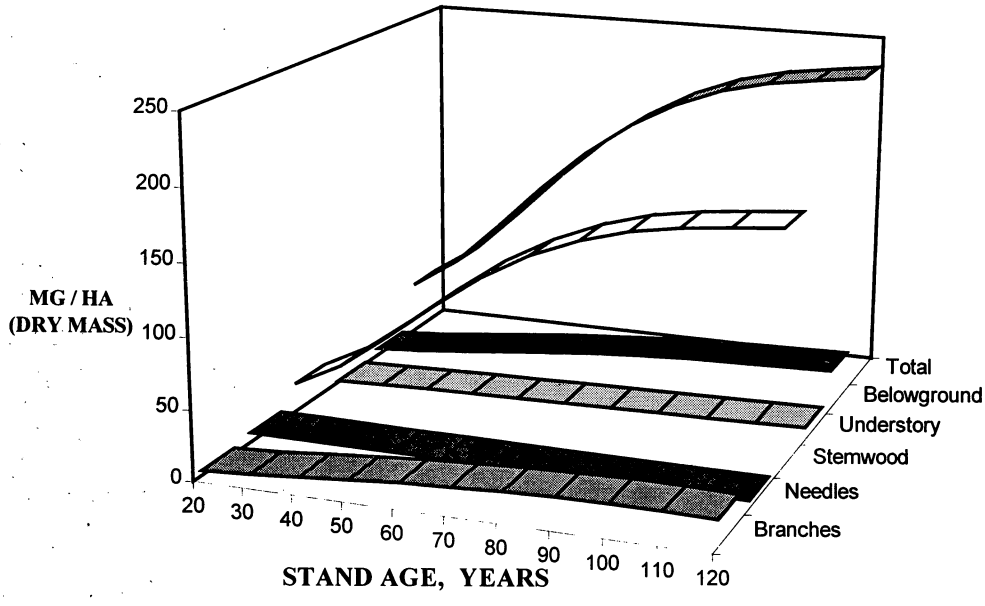

Figure 2. *Dynamics of biomass for the first site class of modal pine stands.*

current situations, such as lack of money, often result in impossibility of implementation and realization of serious investigations. Nonetheless, evaluation and estimation of organic carbon stocks in the forest ecosystems of Russia have been carried out by a number of large research centers. These studies have used various methodologies, however, and results have often been contradictory. These divergences, in our opinion, arise because calculations of the global carbon balance are generally carried out through extrapolation of data from a limited number of samples to estimate phytomass over a rather vast territory. Besides, these estimates typically do not take into account the fractional phytomass composition of the forest ecosystems, litter, current increment, or differences with respect to site, forest types, and forest stands.

Therefore, to obtain accurate estimates of the carbon balance of the forest ecosystems of Russia, it is necessary to carry out research at the regional level, with accurate biomass data for individual studies taking into account all above mentioned factors on the territories of different forestry enterprises. Studies of this kind would contribute much to the development of sound, ecologically–based strategies of nature utilization at the level of economic regions.

MATERIALS AND METHODS

Experimental data were gathered from sample plots established in normal (fully stocked) and modal (typical) pine stands. Field material collection was carried out by using established sample plot methods on pine-dominant forest ecosystems. Materials for the study also included data gathered over different years from sample plots established by the Department of Forest Mensuration and Inventory of the Mari State Technical University.

Based on forest inventory data, we determined the distribution of areas with dominance of *Pinus sylvestris* according to the site classes, basal area and site class of the forest stands. Sample plots were chosen according to Standard Guidelines (1983). Complete reestimation and reevaluation sampling was done by measuring a minimum of 200 trees in middle-aged, maturing and mature pine stands and 400 trees in young forest stands (table 1). Total height and diameter in each of 3 central diameter classes were measured on 5 trees; forest floor cover, soils, litter and canopy were described in detail.

The understory of pine ecosystems includes forest floor vegetation, undergrowth and bushes. In the pine ecosystems of Middle Zavolgie, undergrowth includes regeneration of spruce, birch, *Populus* and oak. Important shrubs include alder buckthorn (*Frangula alnus*), juniper (*Juniperus communis*), filbert (*Corylus avellana*) and *Euonymus verrucosa*. Forest floor vegetation is mainly represented by *Vaccinium vitis-idaea , Vaccinium myrtillus, Calamagrostis arundinacea, Geranium sylvaticum, Vicia sylvatica, Majanthemum bifolium, Hieracium umbellatum, Bryophyta, Fragaria vesca, Lycopodium clavatum and Antennaria dioica.*

Table 1.--*Data from sample plots.*

Stand Age	Number of plots	Number of model trees	Mean diameter	Mean height	Basal area	Volume
Years			*Cm*	*M*	M^2/*hectare*	M^3
20 - 28	25	438	6.4 - 13.1	6.3 - 12.8	12.6- 26.4	64 - 146
30 - 45	26	392	11.6 - 16.3	11.4 -17.6	24.6-33.4	138 - 242
56 - 78	24	322	19.9 - 31.6	17.7 - 26.6	33.6 -42.6	263 - 451
82 - 98	15	196	26.4 - 35.2	18.8 - 27.2	34.6- 44.3	320 - 524
104 - 118	6	78	28.4 - 36.8	20.2 - 28.9	36.4- 46.9	330 - 590

Representative samples were taken from trees beyond the plot boundaries as needed to adequately characterize the diameter classes. 15 to 20 trees were collected per sample area for felling. To obtain this many trees from within the plot areas it would have been necessary to compromise with the requirement of representativeness (Sokolov 1978). Stem defects per meter were recorded for each sample tree.

The crown of each sample tree was cut and divided into three parts: upper, middle and lower. The diameter, weight and length of each branch was measured and divided into sections known as branches (diameter of more than 0.8 cm) and wood greenery (diameter - less than 0.8 cm). Branches without needles were immediately weighed on scales, with a simple comparison made between the weight of branches with needles and those without them. To determine the moisture of the branches and needles of every part of the crown, three middle-sized model branches were taken.

The moisture content of pine wood varies by location in the tree and by length of time since cutting. Moisture occurs in wood as free water in cell cavities and as absorbed water in the cell wall. The condition that exists in a cell when the cell cavity contains no free water and the cell wall is saturated with bound water is known as the fiber saturation point (Husch *et al.* 1982). It is customary to express the moisture content of wood as a percentage of dry weight. The dry weight of pine crown fractions has been obtained by drying a sample of branches and needles at 105 ^{0}C until a stable mass has been reached. The percentage of moisture is calculated from:

$Mc = (W_w - W_d)/W_d$

Where:

Mc - moisture content as a percentage of oven-dry mass

W_w - green mass of wood

W_d - oven-dry mass of wood

The sample areas established in pure *pinetum*/cowberry were characterized by density from 0.5 to 1.0. Undergrowth consisting of pine, aspen, spruce and birch is considered to be within sufficient growth. To investigate pine forest stands, a total of 96 sample plots were established and 1,426 sample trees were harvested (table 1). The sample plots took into account all age classes of forest stands.

Currently, when carrying out forest assessment works in the region of the Middle Zavolgie, it is common to use normal yield and growth tables for pine stands. But as a result of harvesting activity, the forest stands typical of those used to construct these tables are not common. Therefore, the most complete and correct picture of accumulation index growth dynamics can be given by the modal forest stands as the most widespread at present.

Crown Phytomass

Correlation coefficients were calculated for regression equations of interdependence between the dry crown mass and the dendrometry of the stem. Crown mass has a very high positive linear correlation with diameter of crown (D_{cr}) and diameter at breast height (D). For crown length (l_{cr}) this dependence can also be considerable. The dependence of crown mass on the height (H) and the age of trees (A) has been characterised as moderate (Kurbanov 1994).

In recent years, the index, D^2H, has become very popular among researchers as a factor taking into consideration the basal area and the height of the tree. We analyzed the relationship between this index and dry crown mass. While correlation coefficients were high, use of this index can be impractical because of the difficulties connected with its calculation (it is easier to use only D or H). Therefore, we used the dependence of the dry crown mass of the trees on crown diameter, diameter at breast height and crown length in studying crown biomass of pine trees for the Middle Zavolgie Region.

Belowground Mass

Belowground phytomass investigation is a very complicated task. Therefore we did not conduct experimental studies of the pine roots on the sample plots. The phytomass of roots was calculated with the help of the equation of Shvidenko *et al.* (1996) for the European part of Russia:

$$R_{v(bl)} = (a_0 + a_1.A + a_2 . A^2) . A^{a_3} . V^{a_4},$$

where $R_{v(bl)}$ is belowground forest stand phytomass (megagrams of dry matter per cubic meter of green growing stock), A is the average age of a stand in years, V is growing stemwood stock in m^3, and a_0, a_1, a_2, a_3 and a_4 are regression coefficients.

Since neither of the allocation equations considers fine roots, it is assumed that fine roots have about one third of the mass of foliage (Vanninen *et al.* 1996).

Stem Phytomass

Mitscherlich's function was used for pine stand growth calculation in Middle Zavolgie (Chernikh 1992). This function meets all the necessary requirements for the description of general patterns of the growth of living organisms. The function is as follows:

$$T = T_{max}\,[1-e^{(-A*C1)}]^{C2},$$

where T - modeled accumulation index: Indices may represent: diameter (cm), height (m), sum of cross-sectional areas (m^2), or volume stock (m^3); T_{max} - asymptotical value of the accumulation index for a given natural forest stand development; A - age, years; e - base of the natural logarithm; $C1$ - growth parameter; $C2$ - parameter of curve form.

The Mitscherlich's growth function is considered to be non-linear as far as the parameters. To find optimal numerical values of the parameters $C1$ and $C2$ the iteration method is used:

1. For each point of the natural stand with respect to age for the modeled accumulation indices, $C1$ is calculated, for several fixed the of $C2$ (1,2,3,4...10), according to the formula below:

$$C1 = \frac{\mathrm{Ln}(1-(\mathrm{Top}(A)/T_{max})^{1/C2})}{-A}$$

where Top(A) – actual value of the accumulation index in the age A; i – 1,2,3,4.....n; n – the quantity of observations

2. A value of the parameter $C2$ is then chosen to minimize the error of the function.
3. At this constant value of parameter C2, smoothed estimates of parameter C1 are made for all points of the natural stand using a 3rd degree polynomial.
4. Parameter C2 is estimated according to the following equation:

$$C2 = \frac{\mathrm{Ln}\ (\mathrm{Top}(A)/Tmax)}{\mathrm{Ln}(1-e^{-A*C1'})}$$

where C1' - estimated values of parameter C1 according to the 3rd degree polynomial.

5. The estimate of parameter C2 is then smoothed according to the 3rd degree polynomial.

This algorithm was used as the basis for modelling the dynamics of the 4 accumulation indices of the pine forest ecosystem: the mean diameter and height, basal area, and the stem volume.

On the whole, it should be said that approximation of accumulation index dynamics for pine stands showed good results with the help of the Mitscherlich function. This was especially true while fitting stem volume, diameters and heights. The only drawback is the fact that the function has an asymptote parallel to the age axis and, therefore, is considered to be strictly increasing. Therefore, its application is limited by the age of natural maturity, when the growth decreases.

Conversion of stem volume (V_{st}, m^3) to biomass components (M_{st}, Mg) is made by a biomass ratio (R_b, Mg/m^3):

$$M_{st} = V_{st} \cdot R_b$$

The biomass ratio (wood density) varies substantially in relation to the quality of the site, position of the tree, tree age, and the section of stem. Wood density is lower in young trees compared to mature trees, and lower on fertile sites compared to poor sites; i.e., fast growth rate of Scots pine leads to lower wood density than slow growth rate (Kellomaki *et al.* 1992). The overall mean for Scots pine of 381 kg/m^3 was applied in calculations.

RESULTS AND DISCUSSION

Analyses of suitability of the most common forms of simple regression equations were made for the determination of each component of the crown mass. Equation types and variable transformations used to linearize them for determination of correlation coefficients were:

Model	Variable transformations	
	y	x
Linear	y	x
Square root-X	y	square root x
Logarithmic-X	y	log x
Square root-Y	square root y	x
Multiplicative	log y	log x
Exponential	log y	x
S-curve	log y	1/x
Double reciprocal	1/y	1/x
Reciprocal-X	y	1/x

Selection of equations was based on the amount of experimental data available, the statistical criteria of equations, and the distribution of residuals.

After analysing the suitability of the equations for determining needle mass of pine crowns, the best fit to the data is a square root-X equation, followed by the logarithmic and linear equations (table 2), where the independent variable is diameter at breast height.

Of the models fitted, the square root-X model yields the highest R-squared value with 93.24%. This is the model that we selected. There is a statistically significant relationship ($p<0.01$) between needle mass and diameter at breast height. The standard error of the estimate shows the standard deviation of the residuals for the model to be 0.67. The equation of the fitted model is: Table 3 shows the results of fitting several models of dependence of mass of pine branches on diameter at breast height.

Of the models fitted, the exponential model yields the highest R-squared value with 90.16 and therefore this is the model that we selected:

branch mass = exp(0.32 + 0.08*D)

Needle mass = $- 2.85 + 1.92 \cdot (D)^{-2}$

The R-squared statistic indicates that the model as fitted explains 90.16% of the variability in branch mass after transforming to a logarithmic scale to linearize the model. The correlation coefficient of 0.95, indicate a relatively strong relationship between the variables.

Multiple regression models (equations) for phytomass, including a great number of the dendrometry indices, were analysed. It was determined that the combination of such dendrometry indices as age, diameter at breast height and crown diameter are considered to have no essential advantages over the models including only diameter at breast height or crown diameter.

Thus, the equation of form square root-X for pine needle mass and the exponential model for branch mass are considered to be the most acceptable for calculations of the crown phytomass (dry weight basis) of pine stands. Therefore, further calculations of the general crown phytomass were carried out by this type of equation.

In order to examine the relationships of understory and site conditions to stand age, we simulated temporal dynamics of different phytomass components using data from sampling plots.

For the undergrowth the best fit was a square root-Y regression equation:

Undergrowth = $(a + b \cdot \text{Age})^2$,
where a = 3.47, b = 0.42

Table 2.--*Comparison of alternative models relating needle mass to diameter at breast height*

Model	Correlation	R-squared
		%
Square root-X	0.9656	93.24
Linear	0.9604	92.23
Logarithmic-X	0.9428	88.90
Square root-X	0.9156	83.84
Multiplicative	0.8941	79.93
Exponential	0.8458	71.53
S-curve	-0.8077	65.24
Double reciprocal	0,7989	63.83
Reciprocal-X	-0.7704	59.35

Table 3.--*Comparison of alternative models relating branch mass to diameter at breast height.*

Model	Correlation	R-squared
		%
Exponential	0.9495	90.16
Square root-Y	0.9481	89.89
Multiplicative	0.9396	88.28
Linear	0.9117	83.13
Double reciprocal	0.8902	79.24
Square root-X	0.8767	76.85
S-curve	-0.8588	73.75
Logarithmic-X	0.8221	67.58
Reciprocal-X	-0.6842	46.81

Since the P-value is less than 0.01, there is a statistically significant relationship between undergrowth and stand age at the 99% confidence level.

Mass of bushes was best described by a multiplicative model:

$Bushes = a \cdot Age^b$,
where $a = -4.80$, $b = 2.45$

Mass of forest floor for site class 1 of pine modal stands was best described by an S-curve model :

$Floor = \exp(a + b/Age)$,
where $a = 2.44$, $b = -38.76$

The R-squared statistic indicates that the model as fitted explains 95.12% of the variability in forest floor mass after transforming to a logarithmic scale to linearize the model. Forest floor increases with forest age up to 80-90 years. Then due to opening of the forest canopy, which allowed more light to reach the forest floor, the accumulation was stabilized.

The total phytomass of the understory of the normal and modal pine stands increases with stand age. The relationship between forest floor and stand age is modeled as an S-curve, with growth stabilizing after 80-90 years. Undergrowth was better described by square root-Y and X equations with high R-squared values. Accumulation of biomass in bushes has a multiplicative and exponential character.

Data on the crown mass, stem, understory and belowground phytomass (Mg/ha, Mg = megagram = 10^6 g) for the modal pine ecosystems are in table 4. This composite table of forest pine biomass allows us to analyse main trends in the dynamics of different forest ecosystem fractions. As an example, dynamics of the stem and crown fraction mass for site class 1 of

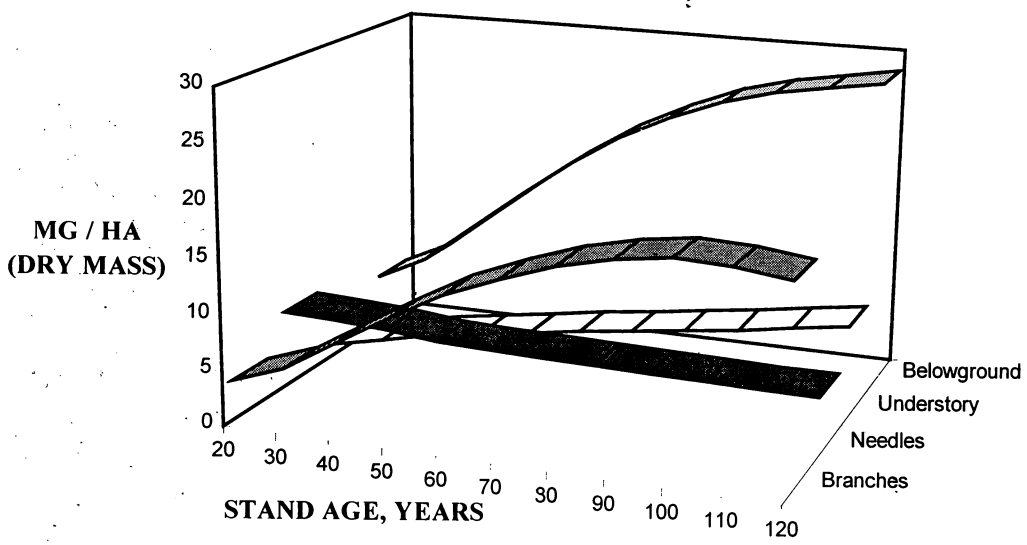

Figure 3. *Dynamics of crown, understory and belowground mass for the first site modal pine stands.*

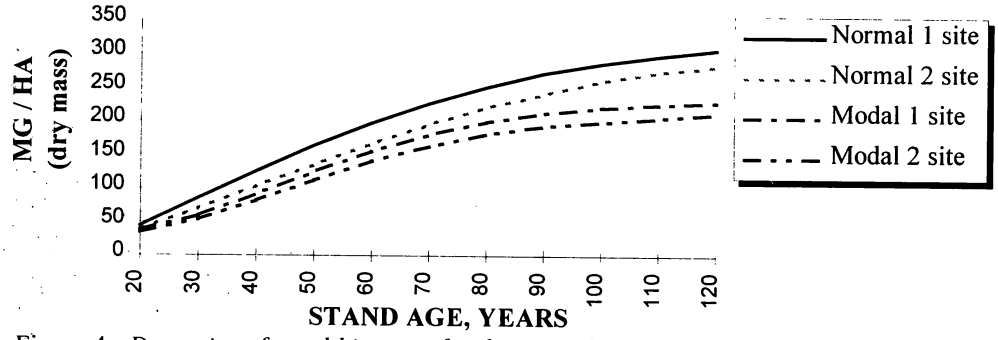

Figure 4. *Dynamics of total biomass for the normal and modal pine stands.*

of the stem and crown fraction mass for site class 1 of modal pine stands are illustrated in fig. 2 and fig. 3. Total stem mass over stand ages of 20-120 years has stable growth. Total crown mass (branches and needles) increases up to 90 years and then drops off slightly. Needle mass is maximum at 20-30 years and then has a stable rate of decrease up to 120 years. Branch mass reaches maximum at 100-120 years. Understory phytomass slowly increases with age.

Normal pine stands have more potential productivity than modal stands. As is seen from fig. 4, normal pine stands of site class 1 have the higher total biomass (tree phytomass and understory). Modal pine stands of site class 2 have the least total biomass productivity. At the end of 120 years normal stands of site class 1 accumulate 308 Mg/ha of total biomass, while site class 2 modal stands accumulate only 213 Mg/ha.

In conclusion, it is necessary to note that the phytomass dynamics of normal and modal pine stands in the Middle Zavolgie can be used for further estimates of carbon stocks and carbon budget on the stand and the regional level.

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Table 4. *Biomass dynamics for the modal pine stands of the Middle Zavolgie, Mg/ha (dry mass).*

Stand age	Crown fractions				Stemwood			Understory				Belowground		
	Branches	Bark	Needles	Total	Stem	Bark	Total	Under-growth	Bushes	Floor	Total	Coarse roots	Fine roots	Total
							Site class 1							
Years														
20	1.8	1.8	7.0	10.6	16.5	4.1	20.6	0.1	0.01	0.9	1.0	2.7	2.3	5.0
30	2.9	2.4	6.8	12.1	32.7	5.2	37.9	0.2	0.03	1.8	2.0	5.1	2.2	7.3
40	5.1	3.1	6.5	14.7	56.8	6.1	62.9	0.4	0.07	2.5	3.0	8.9	2.1	11.0
50	7.4	3.6	6.0	17.0	82.7	6.9	89.6	0.6	0.1	3.0	3.7	12.7	2.0	14.7
60	9.6	4.0	5.8	19.4	105.9	7.7	113.6	0.8	0.2	3.4	4.4	16.3	1.9	18.2
70	11.2	4.3	5.7	21.2	124.6	8.0	132.6	1.1	0.3	3.8	5.2	19.3	1.9	21.2
80	12.6	4.6	5.6	22.8	138.7	8.2	146.9	1.4	0.4	4.0	5.8	21.8	1.8	23.6
90	13.6	4.7	5.5	23.8	148.6	8.2	156.8	1.7	0.5	4.3	6.5	23.7	1.8	25.5
100	14.4	4.6	5.4	24.5	154.7	7.9	162.6	2.0	0.7	4.4	7.1	25.0	1.8	26.8
110	14.4	4.4	5.3	24.1	158.5	7.6	166.1	2.5	0.8	4.6	7.9	25.9	1.7	27.6
120	14.2	4.0	5.3	23.5	161.9	7.4	169.3	2.9	1.0	4.8	8.6	26.7	1.7	28.4
							Site class 2							
20	1.4	1.6	6.2	9.2	14.6	3.8	18.4	0.1	0.02	1.4	1.5	2.6	2.0	4.6
30	2.3	2.0	6.1	10.4	29.3	4.7	34.0	0.2	0.03	2.2	2.4	4.6	2.0	6.6
40	4.3	2.6	5.9	12.8	50.3	5.6	55.9	0.3	0.05	2.7	3.0	8.1	1.9	10.0
50	6.4	3.2	5.8	15.4	74.2	6.3	80.5	0.5	0.07	3.1	3.5	11.7	1.9	13.6
60	8.2	3.6	5.6	17.4	95.6	7.0	102.6	0.7	0.10	3.4	4.2	15.0	1.8	16.8
70	9.6	3.9	5.4	18.9	112.8	7.5	120.3	0.9	0.15	3.6	4.6	17.9	1.8	19.7
80	11.1	4.2	5.3	20.6	125.7	7.6	133.3	1.2	0.23	3.8	5.2	20.2	1.7	21.9
90	12.2	4.3	5.2	21.7	134.5	7.6	142.1	1.5	0.35	3.9	5.8	21.9	1.7	23.6
100	12.5	4.2	5.1	21.8	139.4	7.4	146.8	1.8	0.52	4.0	6.3	23.1	1.7	24.8
110	12.4	4.0	4.9	21.3	144.4	7.2	151.6	2.2	0.77	4.1	7.1	24.1	1.6	25.7
120	12.3	3.8	4.9	21.0	150.5	6.8	157.3	2.6	1.20	4.1	7.9	25.2	1.6	26.8

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FOREST ZONES OF SIBERIA AND POST-FIRE DEMUTATIONS IN CURRENT CLIMATE

D.I. Nazimova, N.P. Polikarpov, M.A. Sofronov, and N.M. Andreeva[1]

ABSTRACT.--One of the most important factors determining the forest cover composition in Siberia is forest wildfires. The characteristics of forest cover are given according to zones and subzones within landscape - climatic sectors. On the base of climatic data the probabilistic model of forest - forming tree species composition is developed. In the whole range of continentality indices the limits of domination these species are determined. The effects of fire disturbances in different sectors and zones are interpreted from the landscape-ecological viewpoint.

INTRODUCTION

Forest wildfires are considered to be one of the most important natural mechanisms of climate-induced successions. Demutation changes are the kind of cyclic successions that end in restoration of initial or close to initial forest type after disturbances if climate and vegetation are in dynamic balance. The dynamic balance is possible both in a stable state and in terms of strong weather fluctuations during comparatively short time intervals.

A new element of investigation is based on the landscape-ecological approach and on the bioclimatic model of Siberian forest zones (Nazimova and Polikarpov 1996). According to the assumption of this approach the composition of potential forest types is mainly determined by the regional climate and biota and - on the lower hierarchical levels - by soils and origin (after fires, insect invasions, wind falls, human impacts). In the Siberian region, because of a continental climate the average duration of the fire danger period is as much as 4 – 6 months in the south taiga and 2 – 4 months in the middle taiga (Valendik 1990). Fire disturbance is a crucial factor determining structure and dynamics of tree stands as well as forest cover of the region (Fires in the Ecosystems 1996).

Every zonal ecosystem has a specific dynamic balance of macroclimate and forest-forming tree species structure. In most of the territory, the balance is being kept stable due to various fire regimes in different zones and altitudinal belts. The goal of this paper is to use a system of territorial units - landscape-climatic sectors and forest zones for study post-fire demutations in current climate. Another goal is to develop a probabilistic model to determine tree species composition of potential forest zones on the basis of the bioclimatic model "BIOME -96".

METHODS

Making use of forest inventory data (Atlas of forests USSR, 1973; Forest fund of USSR,1988) and the subdivision of the Siberian territory into forest regions (Smagin *et al.* 1977, Polikarpov *et al.* 1986) as well as subdivision into the pyrological regions (Sofronov and Volokitina 1990); the authors classified forest regions into bioclimatic sectors, zones and subzones. The most informative climatic parameters for the landscape-climatic sectors is the continentality index Ic, the sums of active temperatures and relative moisture, for zones (Nazimova 1995, Nazimova and Polikarpov, 1996). The forest cover structure was considered as being composed by zonal, extrazonal and interzonal (azonal) combinations of forest ecosystem types. The zonal positions of landscape were defined in topography and soil mosaics and characterized first of all by mature forest stand composition, standard of productivity, structure of lower layers and finally the trends of post-fire demutations.

The zones are characterized by forest-covered area (million hectares), mean percent of area that is forested (%), bogging up ratio, standing volume of mature stands (m^3 /ha), quality class of zonal sites, and post-fire successions.

The landscape topography is of great significance in forest-type pattern. However, it is reasonable to start with ordination of forest cover zonal features determined by the climate. It is the first step to future forest cover prediction in current and in future climate as well. Parallel with this step it is necessary to describe the landscape structure of the territory and to carry out thorough landscape - ecological analysis (Forman and Gordon 1988). According to climatic data, the Nesterov index of forest danger dryness FDI was calculated (see Sofronov and Volokitina 1990 for details).

[1]*D. I. Nazimova, N.P. Polikarpov, M.A. Sofronov, N.M. Andreeva. Institute of Forest SB RAS, 660036, Krasnoyarsk, Academgorodok, Russia. E-mail: ifor@krsk.infotel.ru*

The method is developed by one of the authors and it suggests computation mean monthly estimations using the ratio:

$$FDI = \frac{t^{o}(t^{o} - t_{d}) * 30}{n + 1},$$

where t° – average maximal temperature, t_d – month air temperature of dew-point, and n – number of days with precipitation 3 mm and more during the month. Mean seasonal FDI varies from 300 up to 1,800 in the course of snowless season.

The informational model "BIOME-96" for Siberia has been developed at the Institute of Forest SB RUS using a data base on climatic stations and vegetation, which reveals the climate and vegetation relationship on the zonal and regional levels (Andreeva *et al.* 1997). For zonal forest type complexes such features as forest-forming tree species composition, productivity, and dominant life forms of ground cover are determined. The latter are informative for evaluation forest fuel materials. The climate is characterized by nine parameters describing the annual regime of warmth and humidity supply. The vegetation is presented by main forest forming tree species dominating in the regions characterized by basic climatic stations. For each test area under monitoring 1-4 main tree species were determined taking into account zonal climax or subclimax (Sukhachev 1972). The term "main" suggests dominating mature forest stands of this species in landscape zonal positions (but for sands and supermoistured sites).

On the basis of the informational model, a probabilistic model of the Siberian taiga forest forming species composition has been developed. In this model such integral climatic indices as warmth, humidity and continentality are interpreted as continuous random values. The empirical functions of distributions for these values are obtained. For each given interval of climatic parameters the probability of presence each tree species: Siberian pine, Scotch pine, larch, fir, spruce, birch and aspen is estimated.

RESULTS

Different zonal classes of landscapes are characterized by specific features of natural fire regimes, which results in ratio of burned areas, surface and crown fire distribution, fire return intervals and others. The patterns of past fires determine the patterns of forest cover in many areas of Siberian forest cover. Table 1 represents a system of zonal units in Siberia, some characteristics of zonal forest type structure, and the major trends of post-fire successions in various forest zones (subzones) of Siberia according to three landscape-climatic sectors: West-Siberian, Middle-Siberian, East-Siberian. The West-Siberian forest zones are classified as bog-taiga pedobiome, and West Siberia on the whole is a vast landscape-climatic sector with a specific system of zonality due to high paludification. Nevertheless, even in this area the features related to zonal macroclimate are revealed (table 2).

Latitudinal zonality is vague in the most area of Middle and East Siberia (table 3 and table 4), which is defined by high amplitude of local climatic indices in terms of permafrost and sharp continentality. It is reasonable to mention the hyperzonality of the extremely continental Siberian landscapes, especially within mountainous regions. While in Western Siberia there is the entire set of zones, and latitudinal regularities of demutation shifts are clearly observed, in Eastern Siberia there are no such latitudinal zones as south taiga and mixed forest, but the forest-tundra ecotone occupies a vast area. Larch forest of *L. gmelinii* is the only formation that absolutely dominates. All the other species are lacking in the zonal positions, or occur in interzonal or extrazonal sites. The secondary birch forest fraction is negligible. Larch dominance and lack of dark conifers are caused by high fire danger as well as by severity of climate, mentioned above.

Mountain regions of South Siberia (table 5) are considered to include three sectors of humidity and differ from the neighboring regions due to their less continental climate. They are of interest to compare with the other units of landscape-climatic subdivision of Siberia (see table 2 and table 4). The altitudinal zonal ecosystems are outside the scope of table 1, but one can see some analogues between latitudinal and altitudinal zonal ecosystems in Siberia. On the other hand, the mountain ecosystems are much richer in biodiversity and they include some unique zonal classes: subalpine meadows and woodlands, dwarf-pine formation, dark coniferous herbaceous taiga (called chern taiga), etc. (Shumilova 1962; Smagin *et al.* 1980; Nazimova and Polikarpov 1996). As a whole, the mountains belong mostly to West-Siberian and to Middle-Siberian sectors of continentality but their climate is more humid. Tables 2-4 show that each landscape-climatic sector has its spectrum of forest zones and subzones and is characterized by a specific ratio of life forms and tree species

Table 1.–*The main forest zones in Siberia and their characteristics.*

ZONOBIOMES, ZONO-ECOTONES	Landscape structure (%)	Climatic climax composition		Post- fire successions
		Major tree species	Synusia of lower layers	
TAIGA-TUNDRA, ECOTONE, CONTINENTAL-SUPERCONTINENTAL (C1-C3)	Woodlands and isolated forest tracts in combination with tundra. Peat lands, yernicks - 50-65%. Forest lands - 20-35%.	*L. gmelinii* ssp, *ssp.cajanderi, L gmelinii, L. sibirica, Picea obovata.*	Subarctic dwarf shrubs, lichens, mosses, psychrophytes. *Salix* and *Betula* yernicks.	Longterm deforestation and paludification. Periodically successful regeneration of larch and spruce. (Forest fire can optimize temperature regime of soils and the growth of trees.)
TAIGA, WEST-SIBERIAN CONTINENTAL (C1) HUMID	Dominance of dark coniferous formations in zonal sites; pine forests mostly in azonal sites (sandy and wet peat soils river valleys). Peat lands - 50%, forest lands - 45%.	*Pinus sibirica, Pinus sylvestris, Picea obovata, Abies sibirica.*	Boreal dwarf-shrubs, herbs, green-mosses, *Sphagnum*, Poor biodiversity.	Post-fire pine, birch and mixed pine-birch forests. On the next stages mixed forests with regeneration and lower layer of dark conifers (Siberian pine, spruce, fir). Postfire restoration period - 100-120 years. Wide spreading of birch forest promoted by short interfire period.
TAIGA MIDDLE-SIBERIAN CONTINENTAL (C2) CRYO-HUMID AND SEMIHUMID	Dominance of larch and pine forests in zonal sites, and spruce and pine in the azonal sites. Forest lands - 60%. Peat lands - 9%. Non-forest lands - 31%.	*Pinus sylvestris, Larix sibirica, Larix gmelinii.*	Boreal and hypo- arctic dwarf shrubs, sedges, green-mosses, yernicks.	The North: post-fire larch stands; birch is restricted and pine is absent because of low soil temperature. The South: post-fire birch and pine stands. Sustainable pine subclimax. Dark conifers regenerate in valleys and on shadow mountain slopes.
TAIGA EAST-SIBERIAN SUPER CONTINENTAL (C3) CRYO-SEMIHUMID	Predominance of larch in zonal sites. Local pine and spruce forests, steppe and forest-steppe patches in the azonal sites. Forest lands - 50-60%. Peat lands - 12%. Non-forest lands -30-40%	*Larix gmelinii*, ssp. *cajanderi*. Bushes: *Betula, Duschekia, Salix.*	Boreal and sub-arctic oligotrophic dwarf shrubs, herbs, lichens, mosses. Continental forest and steppe sedges and grasses.	Postfire regeneration of larch is successful. Bushes and yernicks are common at the first stages of demutations. Scotch pine regenerates and forms stands exclusively on warm soils. Spruce regeneration is in valleys. Post-fire woodlands and dwarf-elphinwoods replace larch mountain taiga after fires.
SUBTAIGA CONTINENTAL (C1, C2) HUMID AND SEMIHUMID	Dominance of birch and pine forests in zonal sites, dark coniferous forests (spruce, fir) in the azonal sites. Forest lands - 30-40%. Peat lands - 33% (in West Siberia) and 4 % (in East Siberia).	*Betula pendula, Pinus sylvestris, Populus tremula, Larix sibirica.*	Boreal - meadow grasses and herbs. Lichens and mosses are absent. Rich biodiversity.	Post-fire birch and pine stands and meadows. Primeval forests are absent. Anthropogenic factors determine the forest succession.
FOREST STEPPE ZONOECOTONE CONTINENTAL (C1, C2, C3) SEMIHUMID	Combination of meadow steppe, bunchgrass steppe and patches of subtaiga in low mountains in southern Siberia. Forest lands - 10-30%. Peat lands - 22% (in West Siberia) and 4% (in East Siberia).	*Betula pendula, Pinus sylvestris, Larix sibirica.*	Meadow-forest grasses and herbs. *Caragana, Cotoneaster* and *Spirea* bushes.	Post- fire bushes and steppe. The dynamics of the forests are defined by climatic cycle and human activity.

Table 2.--*Species distribution of forest land in different zones and subzones of West-Siberian sector.*

№	Zone, subzone	Forest land, 10^6. ha	Forest land, area, %	Bogs, area, %	Forest-forming tree species, %								Site quality class	Stock of mature stands, m^3/ha
					Pco	Psb	Asb	Psyl	Lsb	B	P	Oths		
1	Tundra-forest	11.6	20-25	55	24	13	-	7	48	8	-	-	V	50-60
2	Northern taiga	11.0	25	72	12	12	-	51	15	9	-	1	V	110-120
3	Middle taiga	36.3	45-50	46	10	22	-	45	2	25	5	2	IV	140-160
4	Southern taiga	36.0	50-55	42	6	11	8	24	-	40	10	1	III	170-190
5	Subtaiga	8.4	30-40	33	1	-	-	35	-	54	7	3	II-III	170-190
6	Forest-steppe	6.9	10	22	-	-	-	42	-	47	4	7	II-III	160-180
7	Total	110.2	40	46	9	10	2	34	7	30	6	2	IV	150

Conditional signs: Pco – spruce, *Picea obovata*; Psb – Siberian pine, *Pinus sibirica*; Asb– fir, *Abies sibirica*; Psyl – Scots pine, *Pinus sylvestris*; Lsb – larch, *Larix sibirica*; B – birch, *Betula pendula*; P – poplar, *Populus tremula*; Oths – willow, *Salix* spp. and others.

Table 3.--*Species distribution of forest land in different zones and subzones of Middle-Siberian sector.*

№	Zone, Subzone	Forest land, 10^6 ha	Forest land, area, %	Bogs, area, %	Forest-forming tree species, %								Site quality class	Stock of mature stands, m^3/ha
					Pco	Psb	Asb	Psyl	L	B	P	E		
1	Tundra-forest	16.6	30-40	14	4	4	-	-	83	6	-	3	V	60-70
2	Northern taiga	22.0	50-60	10	8	8	-	-	74	8	-	2	V	80-120
3	Middle taiga	38.8	60-65	7	12	10	4	12	50	9	1	2	IV	140-170
4	Southern taiga	38.7	70-80	2	7	12	5	38	25	8	3	1	III(II)	112-240
5	Subtaiga-forest-steppe	8.9	25-30	4	6	7	6	47	13	13	6	2	III	210-230
6	Total	125.0	60	9	8	9	4	19	49	8	1	2	III-IV	160-180

Conditional signs: Pco – *Picea obovata*; Psb – Siberian pine, *Pinus sibirica*; Asb – fir, *Abies sibirica*; Psyl – Scots pine, *Pinus sylvestris*; L – larch, *Larix sibirica*, *L.gmelinii*; B – birch, *Betula pendula*, *B. pubescens*; P – poplar, *Populus tremula*; E – yernicks and others.

Table 4.--*Species distribution of forest land in different zones and subzones of East-Siberian sector.*

№	Zone, Subzone	Forest land, 10^6 Ha	Forest land, area, %	Bogs, area, %	Forest-forming tree species, % L	Psyl	Pco	Psb	B	P	E	Ppum	Site quality class	Stock of mature stands, m3/ha
1	Tundra-forest	41.9	30-35	19	91	-	-	-	-	-	6	3	V	60
2	Northern taiga	32.3	40-50	17	93	1	-	-	1	-	3	2	V	110
3	Middle taiga	94.8	65-70	10	76	11	1	1	3	1	5	2	IV(V)	130
4	Subtaiga-forest-steppe	9.5	30-40	4	61	14	-	-	18	2	2	3	IV(III)	140
5	Total	178.5	50-60	12	81	6.5	0.5	0.5	3	0.5	5	3	V	110

Conditional signs: L – larch, *Larix gmelinii*, ssp. *cajanderi*, *L. gmelinii*, *L. sibirica*; Psyl – Scots pine, *Pinus sylvestris*; Pco – spruce, *Picea obovata*; Psb – Siberian pine, *Pinus sibirica*; B – birch, *Betula pendula*, *B. pubescens*; P - poplar, *Populus tremula* ; Ppum – *Pinus pumila*; E – yerniks (birch and willow bushes and shrubs).

Table 5.--*Species distribution of forest land in different sectors (groups of regions) of South Siberia.*

Group of regions	Forest land, 10^6 ha	Forest land, area, %	Forest-forming tree species, % Asb	Psb	Pco	Lsb	Psyl	B	P	Oths	Site quality class	Stock of mature stands,
Perhumid taiga, chern taiga regions	11,3	70-80	46	23	1	4	3	15	8	-	III	200-220
Humid taiga regions	11,8	55-70	5	39	5	16	18	14	2	1	IV	150-270
Mordern humid taiga and forest-steppe regions	15,3	40-50	-	32	2	46	11	5	-	4	IV-V	130-140
Semihumid taiga and steppe regions	0,8	15-25	-	11	3	81	-	2	-	3	V	60-70
Total forest lands	39,2	55-60	15	30	2	28	10	10	3	2	IV	160-170

Conditional signs: Asb – fir, *Abies sibirica*; Psb – Siberian pine, *Pinus sibirica*; Pco – spruce, *Picea obovata*; Lsb – larch, *Larix sibirica*; Psyl – Scots pine, *Pinus sylvestris*; B – birch, *Betula pendula*; P – poplar, *Populus tremula*; Oths – others.

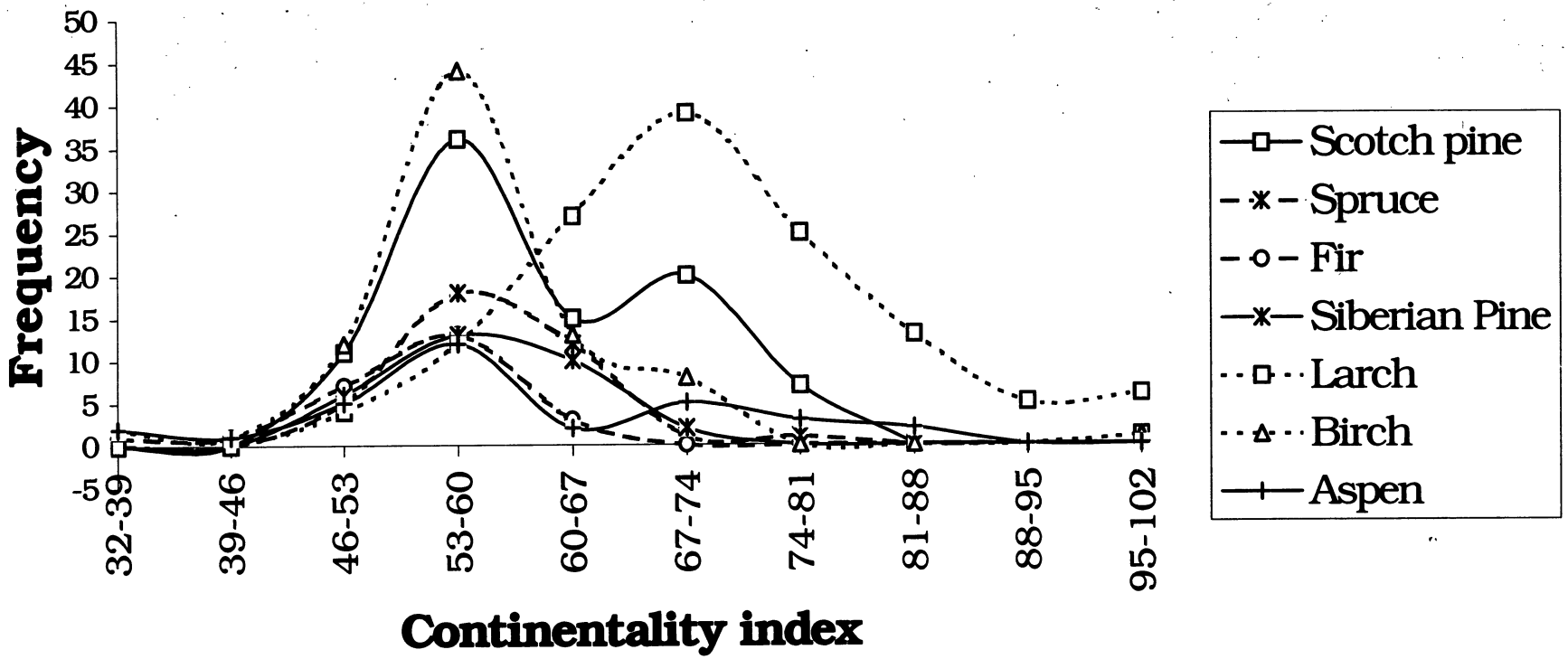

Figure 1.—*Frequency of the zonal forest-forming species across the range of continentality.*

Table 6.--*Nesterov Index of forest drought danger (average for snowless period in zones and landscape-climatic sectors of Siberia).*

Zone	West Siberia	Middle Siberia	East Siberia
Middle taiga	450 - 500	500 - 1000	1000 - 1500
South taiga	500 - 700	1000 - 1300	-
Subtaiga and forest-steppe	750 - 1300	1000 - 1300	1850 - 2050

composition. The more continental is the climate, the less zonal classes can be found within it. The data in tables 1-5 illustrate this, and they also show quantitative ratios of dark conifers, light-conifers and broad-leaved (birch and aspen) trees.

Primary and secondary stands (mainly post-fire ones) form a mosaic pattern in the complicated topography of landscape zones. Nevertheless, their area ratio shows different successions in different landscape - climatic sectors. Within the sectors, the subzones also differ in post-fire successions but these differences are not so expressive. Table 6 shows sharp rise of Nesterov's drought index to the East. The fire regimes determine pine and birch dominance in West Siberia, pine in the south taiga and subtaiga of Middle Siberia, and larch in East Siberia, where pine can not grow on permafrost soils.

In West Siberia fires have always been the major factor of demutations during the Holocene, in spite of high bogging-up. Pine and birch impact is in all zones, and it rises sharply from the North to the South. The Nesterov drought index (FDI) in the north taiga is close to that in the middle taiga, and in the Transurals, where the climate is especially dry, the maximum number of fires and pine dominance are noted. Scotch pine forms sustainable stands, some kind of pyrogenic subclimax, which is analogous to that in the south of Middle Siberia, in the Eastern Priangarye (Shumilova 1962). Birch replaces not only dark conifers but scotch pine as well after fires and cuts in the zonal forest ecosystems of the south and middle taiga of both sectors (Furyaev 1996).

In Western Siberia, (table 2) pine and birch dominance is more than 60% in the taiga and 85% in the subtaiga, and nearly 90% in the forest steppe. Dark conifers are nowhere more than 30% of forest cover in West Siberia. The area ratio of dark conifers proves that none of them, especially fir, was sustainable on zonal sites, at least over the course of the last 100-200 years. Not only wildfires, but also pest invasions caused disturbances on vast areas.

The South-Siberian mountains (table 5) have their peculiarities of composition and demutations. On windward slopes forest types with fir predominance are formed due to perhumid climate.
It is not a latitudinal-zonal phenomenon, but a feature of altitudinal zonality that the chern types of fir and aspen forest are widespread on the northern bottom of the Altaj-Sayan mountains windward slopes. The chern formation of fir and Siberian pine (occurring at 500 m-900 m. elevation) is not exposed to wildfires because of perhumid and even superhumid climate. Demutations run usually through a short stage of aspen, birch, fir, and mixed stands. They are caused not so much by fires, but by windfalls, cuts and pest invasions. The graphic pictures of forest zones and biomes (Nazimova and Polikarpov 1996) illustrate rather narrow climatic niches of sustainable dark coniferous forests dominating in zonal positions. The ecotone between dark- and light-coniferous zonal ecosystems is recognized as a region of birch and mixed forests.

It is advisable to use the forest zone information model (Andreeva *et al.* 1997) as the base for analysis of forest cover in changing climate. The main elements under investigation are the major forest-forming tree species of Siberian forest lands: larch, pine, cedar, spruce, birch, and aspen. The probabilistic model of forest-forming species structure allows estimation of the feasible coniferous tree diversity caused by potential climatic conditions taking into account the existing selective mechanisms in natural zonal landscapes. Fire regimes are an important mechanism that selects species and ecosystems resistant to surface fires. The probabilistic model determines the limits of dominance of each forest-forming species in the range of alteration of each climatic parameter: warmth, humidity supply and continentality. These parameters are interpreted as continuous random values. Empirical diagrams of the distributions for these values are obtained. In fig. 1 the frequency of presence for each forest forming species is shown relative to continentality index (Ic) intervals. Ic varies from 50 to 98 in the plains, and from 35 to 100 in the mountains. The diagram shows that dark conifers decrease with increasing continentality. When Ic values exceed 74, dark conifers disappear.

All the coniferous and small-leaved species occur in the ranges of Ic 46-53, 53-60, and 60-67. When Ic increases, the first to disappear are fir and Siberian pine, next spruce (remaining in valleys). Then birch and pine disappear - their continentality limits are Ic 74-81 and 81-88, respectively. The most resistant to severe climate is larch, which can dominate in the whole range of Ic, from 60-67 up to 95-100 on permafrost.

The method of probabilistic modeling is helpful in any study concerning land cover prediction and forest management planning. It is useful for exploring resistance of natural forest stands to critical climatic situations. Heavy droughts and insect invasions accompanied with forest fire may accelerate replacing zonal tree species (spruce, fir, Siberian pine) by the species more resistant to fire and other disturbances: pine and birch. These species dominate the specific subzone - Siberian subtaiga or mixed light coniferous - small leafed forest, where dark conifers are not resistant (see table 1 and table 2).

However, an alternative direction of post-fire succession is possible, if increasing humidity of climate is more than increasing potential evapotranspiration. The role of dark conifers is possible to increase to 2070 due to demutations in birch and aspen forests, 15%-20% of their area in south taiga of Ob-Yenissey region (Furyaev 1996). The probability of fire occurrence and forest fire extent is assumed close to the current level. Nevertheless, even in this case, the probability of dark coniferous dominance will be dubious in many regions of West and Middle Siberia due to human activity. Fig.1 testifies to the above-mentioned geographical regularities in forest zone composition and provides their ecological interpretation.

Dominating areas of forest-forming tree species may be represented in parametric space of climatic axes. This provides prediction post-fire demutations under current climate. It is useful for predicting different kinds of successions under different climatic trends.

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BOREAL ECOSYSTEM RESPONSE TO ATMOSPHERIC NITROGEN INPUTS AND CLIMATE CHANGE

R. Stottlemyer and R. Herrmann[1]

ABSTRACT.--In 1982, a network of research watersheds was established in U.S. reserves. Four in Michigan, Colorado, and Alaska are northern ecosystems. One research objective is to understand ecosystem response to atmospheric inputs and climate change. N inputs are >6 kg ha^{-1} y^{-1} in Michigan to <0.6 kg ha^{-1} y^{-1} in Alaska. Sites outside Alaska retain N input. Stream water NO_3^- peaks in winter, soil inorganic N pools in spring. Soil N mineralization exceeds precipitation inputs, and most meltwater/precipitation enters the soil. $\delta^{15}N$ and $\delta^{18}O$ studies indicate stream water NO_3^- is the product of soil processes. Soil N mineralization rates increase with temperature.

INTRODUCTION

There is considerable debate about the effects of acidic deposition inputs on terrestrial and aquatic ecosystems (Driscoll *et al.* 1988, Murdoch and Stoddard 1992). Much attention is foocused on surface water acidification as a direct result of precipitation H^+ input. But, most North American surface waters are moderate-to-well buffered and not directly sensitive to precipitation H^+ input (Stoddard 1994). Many terrestrial ecosystems appear more sensitive to chronic inputs of inorganic nitrogen (N) and sulfer (S) (Aber *et al.* 19890. Whle S emissions and deposition are decreasing in many regions of North America (Hirsch *et al.* 1991), those of inorganicN are not (NADP 1982-1992). Increased emissions heighted concerns about the possible effects of inorganic N deposition on ecosystems.

Trends in stream water chemistry reflect not only precipitation inputs, but also the effects of terrestrial ecosystem processes (Likens and Bormann 1995). Stream water ion concentrations reflect the products of processes as soil mineral weathering, disturbance, biotic uptake, change in biomass, precipitation quantity and timing, and snowmelt runoff. Where snowmelt dominates the annual stream hydrograph, two factors account for most seasonal variation in stream water chemistry: the annual hydrologic cycle, and the degree to which snowmelt is partitioned between overland and subsurface flow (Caine and Thurman 1990, Stottlemyer and Toczydlowski 1991, Stottlemyer and Troendle 1992). Meltwater or precipitation chemistry is rapidly altered once it enters the soil or rock substance even in sensitive alpine ecosystems (Caine 1989, Campbell *et al.* 1995, Stottlemyer and Troendle 1992). A third factor in systems with large snowpacks is the effect of unfrozen soils on winter soil processes such as immobilization of essential nutrients, mineralization and nitrification. These processes can produce seasonally large soil reservoirs of inorganic nutrients which further modify the chemistry of passing meltwater.

Few reliable precipitation chemistry data exist before the late 1970s. One can only make educated guesses about chemical conditions of the past. The study of forested ecosystems receiving very different atmospheric inputs can provide some resolution to the question of how ecosystems responded to increased inputs (Driscoll *et al.* 1988).

In 1982, a network of long-term, watershed-ecosystem study sites was established in U.S. protected reserves (Herrmann and Stottlemyer 1991). The network includes four northern watershed ecosystem sites (Isle Royale, MI; Rocky Mountain N.P. and the U.S. Forest Service's Fraser Experimental Forest, CO; and the Noatak National Preserve, AK). Major research objectives are to understand mature ecosystem structure and function, and the effects on processes of change in atmospheric contaminant inputs and climate.

In this paper, we use research results from these sites to assess watershed ecosystem response to anthropogenic N inputs across a gradient of inputs. The magnitude and seasonal distribution of stream water N outputs from these ecosystems suggest they are not yet directly responding to N inputs. N outputs appear to reflect seasonal trends in soil processes such as N mineralization and nitrification.

SITE DESCRIPTIONS

The watersheds are study sites within the ongoing long-term Watershed Research Program, U. S.

[1] *U.S. Geological Survey, 240 W. Prospect Road., Ft. Collins, CO 80526.*

Geological Survey or cooperating long-term watershed sites (Fraser Experimental Forest, CO, U.S. Forest Service; Denali National Park and Preserve, AK, National Park Service).

The Wallace Lake watershed (115 ha) is located in Isle Royale National Park, Michigan, in NW Lake Superior about 120 km N of Calumet. Watershed elevation varies from 190 to 275 m. Watershed vegetation is dominated by trembling aspen (*Populus tremuloides*), white birch (*Betula papyrifera*), and white spruce (*Picea glauca*). The bedrock is metamorphosed volcanics covered by scattered alkaline glacial till originally derived from limestone bedrock immediately south of James Bay. The soils are primarily Alfic Haplorthods, sandy to coarse loamy, mixed, frigid, and are between 3,000 to 5,000-years old. Parent materials are sands and beachline deposits laid down during the post-glacial Lake Nipissing stage. About 50% of annual precipitation occurs as snowfall.

The Fraser Experimental Forest (FEF) is 137 km west of Denver, CO, west of the Continental Divide. Lexen Creek, the study watershed (124 ha) within FEF, has a prevailing easterly aspect and ranges in elevation from 2995 m at the watershed mouth to 3,515 m at the summit of Bottle Peak. The watershed's bedrock includes remnants of sedimentary sandstone at upper elevations, while gneiss and schist parent material underlie the entire watershed. Soils are mostly Inceptisols. About 18% of the watershed is alpine meadow which generally occurs above 3,350 m elevation. The lower and mid-elevation subalpine forested slopes with southerly and easterly aspects are dominated by lodgepole pine (*Pinus contorta*) while upper elevations and northerly aspects are vegetated by Engelmann spruce (*Picea engelmannii*) - subalpine fir (*Abies lasiocarpa*). About 70% of annual precipitation falls as snow. Snowpack water equivalent (SWE) increases sharply with elevation (Stottlemyer *et al.* 1996).

The 660 ha Loch Vale drainage is east of the Continental Divide NE of the FEF in Rocky Mountain National Park. Watershed elevation ranges between 3,100 and 4,000 m. The watershed is primarily alpine, talus, and snowfields with <10% in subalpine forest and meadow. Annual precipitation is about 100 cm with 70% occurring as snowfall. Bedrock is primarily Precambrian gneiss and schist with little apparent ground water storage. Forest soil is a loamy-skeletal, mixed Lithic Cryoboralf. Soil development is very heterogenous ranging in depth from <1 to >25 cm.

The Rock Creek drainage (770 ha) is located in Denali National Park and Preserve, AK. The watershed has a south-southeastern aspect, ranges in elevation from 640 to 1,700 m, and is underlined with metavolcanic quartz shist and some marble. Soils are not mapped, but likely are Alfic Cryochrepts. Mid-age (100+ years) white spruce and 50-year-old trembling aspen dominate vegetation in the lower quarter of the watershed. At lower elevations with eastern or southeastern aspects open black spruce (*Picea mariana*) occurs with white spruce and *Betula glandulosa*. Upstream from the coniferous and mixed forest is a band of closed tall scrub dominated by *Alnus crispa* and *Salix*. Mixed mesic shrub (*Betula, Ericaceae*) occurs at elevations above the tall scrub. Most of the watershed is dominated by patches of mesic to dry *Dryas* tundra. About 50% of annual precipitation occurs as snowfall.

METHODS

There are publications from each of these sites outlining in detail the monitoring and research methods employed (Baron 1992, Stottlemyer 1992, Stottlemyer and Troendle 1992, Stottlemyer *et al.* 1995). Only the measured variables common among all sites are presented here.

Field Procedures

Precipitation amount is monitored with Belfort recording raingauges with strip charts or dataloggers. Precipitation chemistry is generally measured by NADP stations or, in the case of the FEF, using the NADP protocol and analytical procedures.

All watersheds are gauged and continuous flow monitored. Stream water is generally sampled weekly from just before snowmelt to formation of a permanent snowpack. Stream water is sampled more frequently during snowmelt and less frequently during winter depending on access. At all sites, stream water chemistry is measured at a number of sites upstream of the stream gauge. All watersheds are equipped with at least one meteorological station which records wind speed and direction, solar radiation, air and soil temperatures, and relative humidity.

In addition to the routine precipitation and stream water monitoring, all sites monitor snowpack amounts and chemistry, have basic data on biomass and forest floor and nutrient content, canopy and soil modification of incident precipitation, and generally

net soil N mineralization and S mineralization data. All sites currently have some research underway regarding the relationship between climate change and ecosystem function primarily nutrient cycling.

Laboratory Analyses

Samples are brought immediately to a field laboratory. For precipitation and stream water samples, pH, specific conductance, and alkalinity (titration with 0.02 N H_2SO_4 to pH 4.5, stream water samples only) are determined as soon as samples reach room temperature. Separate filtered (pre-rinsed, 0.45 µm) subsamples for ion analyses are refrigerated at 2° C. The filtered subsamples are shipped in coolers to our laboratory at the Rocky Mountain Research Station, Ft. Collins, CO. Calcium, Mg^{2+}, Na^+, K^+, NH_4^+, NO_3^-, SO_4^{2-}, and Cl^- concentrations are determined using a Dionex Model 2020 ion chromatograph. Laboratory and field sampling procedures remained constant during the study results reported here.

RESULTS AND DISCUSSION

Wallace Lake, Isle Royale National Park

The Upper Great Lakes region receives 3 - 4 kg ha^{-1} y^{-1} of NH_4^+, some of the highest wet precipitation inputs in the nation (Lynch *et al.* 1995, NADP 1982-1996). Wet precipitation NO_3^- inputs are about 10 kg ha^{-1} y^{-1}. NH_4^+ - N input equals or exceeds NO_3^- - N input. Wet precipitation N inputs total >6 kg ha^{-1} y^{-1}. The concentration of both NO_3^- and NH_4^+ in precipitation has significantly ($p < 0.05$) increased over the last decade (fig. 1). NO_3^- shows little seasonal change in concentration while NH4+ increases in spring and summer.

Annual stream water discharge is dominated by spring snowmelt and precipitation input. Stream water NH_4^+ concentration is generally below detection. Annual stream water NO_3^- concentration shows no trend (fig. 2). Stream water NO_3^- concentrations are highest ($p<0.05$) in winter and early spring before snowmelt. Watershed retention of N inputs is >90%.

Precipitation NO_3^- - N input is small compared to soil N pools and flux. The organic N pool in the top 10 cm of soil averages 2,080 kg N ha^{-1}. The boreal forest floor (Oi,Oe, and Oa) contains another 550 kg N ha^{-1}, and annual litter fall adds about 20 kg N ha^{-1} (Rutkowski and Stottlemyer 1993). Soil NO_3^- - N pools average >2.6 kg ha^{-1}, and the total inorganic N pool averages >23 kg ha^{-1}. Annual net NO_3^- mineralization averages 4 kg N ha^{-1}, and total net inorganic N mineralization >28 kg N ha^{-1} (Stottlemyer *et al.* 1995). Net N mineralization rates and soil inorganic N pools peak in May and June. NH_4^+ and total inorganic N-mineralization increase ($p<0.01$) with soil temperature indicating the potential for climate change to enhance inorganic N availability.

Gross N mineralization and nitrification rates (fig. 3) also indicate the very high inorganic N cycling rates in this ecosystem. Mean daily N cycling rates range from 0.6 to about 1 kg N ha^{-1}. With such large cycling rates, large soil inorganic N pools, and strong ecosystem N retention, it appears likely any stream water NO_3^- is the product of soil processes, not snowmelt or precipitation inputs.

Lexen Creek, Fraser Experimental Forest, Colorado

Since 1980 to the present, the period of study reported here, there has been no significant trend in annual temperature or precipitation amount (fig. 4). The precipitation amount in May has been increasing ($p<0.05$). Since 1982, the volume-weighted precipitation ion chemistry or loading has not significantly changed except for the decline in SO_4^{2-}. Most high elevation Colorado NADP stations have shown an increase ($p<0.05$) in precipitation amount and NH_4^+ input over the last decade, and some show increases in NO_3^- concentration and loading (Lynch *et al.* 1995). Precipitation ion cencentrations generally increase in spring and early summer (fig. 5).

Snowtel data indicate the length of the snowpack season has decreased since 1982 in the Fraser region. At the FEF, peak SWE has been on the decline since 1965 (fig. 4). Snowpack ion concentration ($p<0.05$) and load increase with elevation and on NE aspects ($p<0.01$, fig. 6).

Canopy modification of incident precipitation amount and its chemical composition is substantial. During a decade of study in the FEF, the canopy intercepted, on average, one third of the snowfall. Snowpack K^+ ($p<0.01$), H^+ ($p<0.001$) and SO_4^{2-} ($p<0.05$) concentrations were greater beneath the canopy, and NH_4^+ ($p<0.01$) and NO_3^- ($p<0.001$) concentrations greater in the opening. Snowpack NH_4^+ and NO_3^- content were also higher ($p<0.001$) in the open.

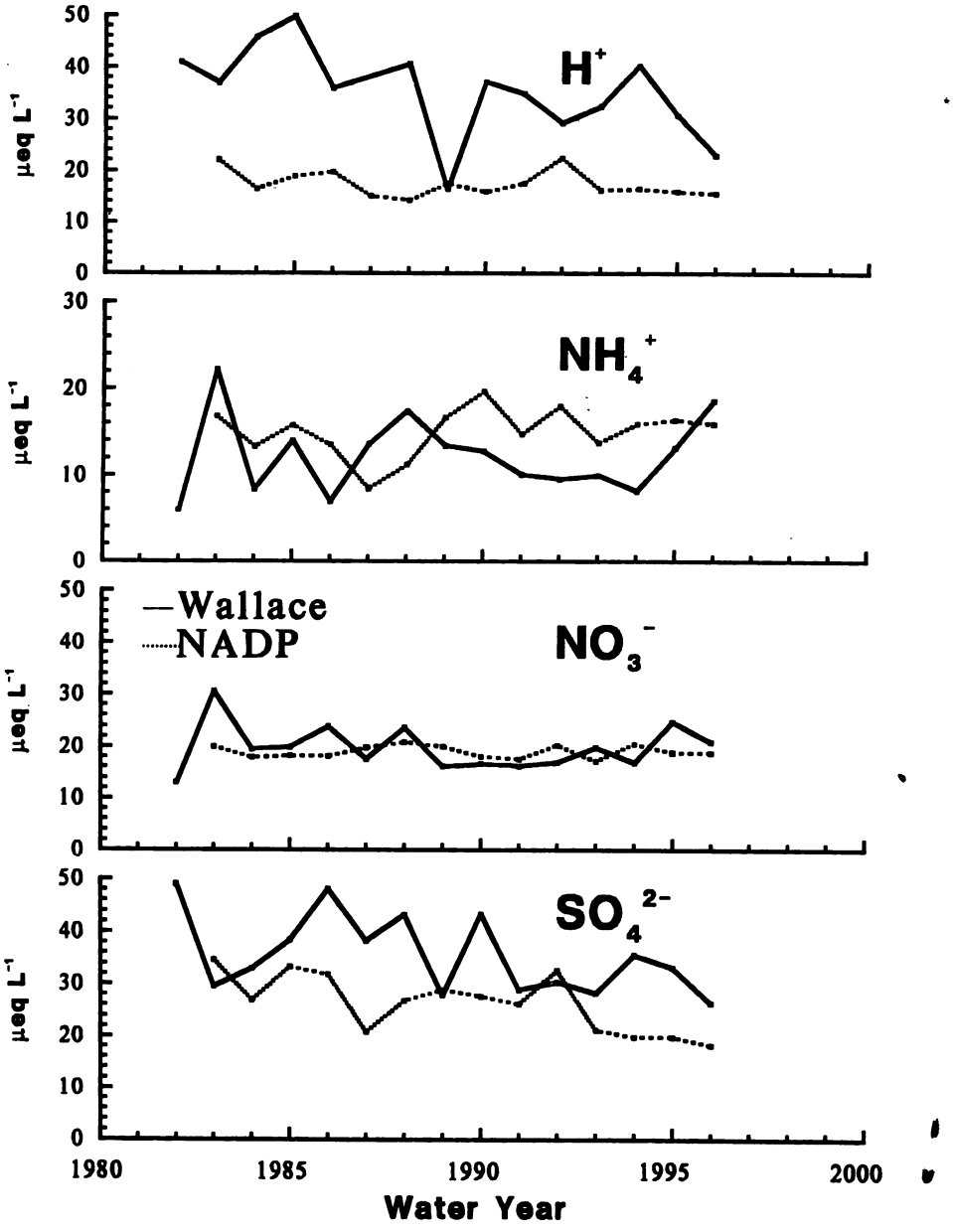

Figure 1. *Trends in annual volume-weighted precipitation chemistry for NADP station MI99 on the south shore of Lake Superior, and the Wallace Lake watershed, Isle Royale National Park, Michigan*

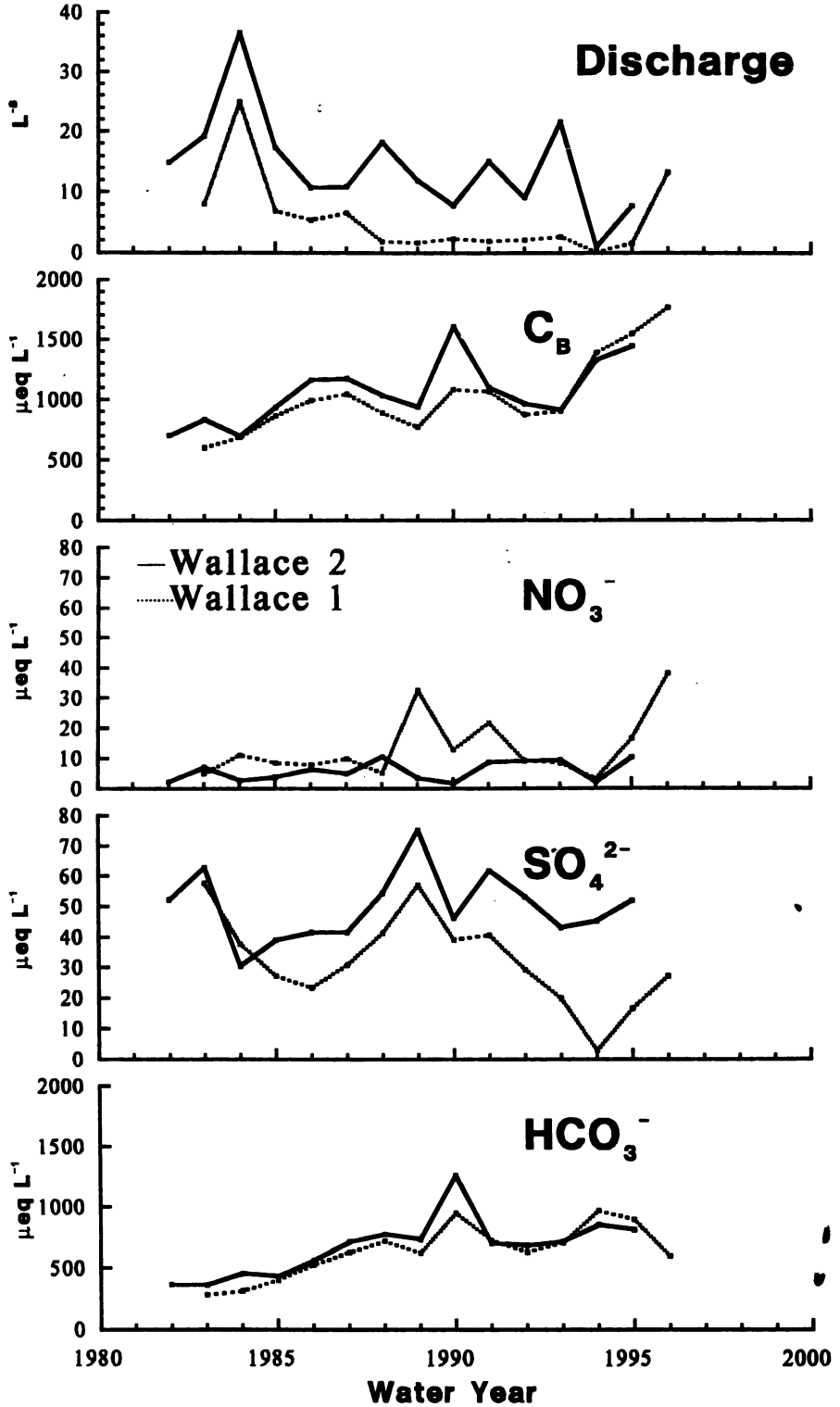

Figure 2. *Trends in annual volume-weighted stream water ion concentration for lower (Wallace 2) and upper gauging station. Wallace Lake watershed. Isle Royale National Park. Michigan*

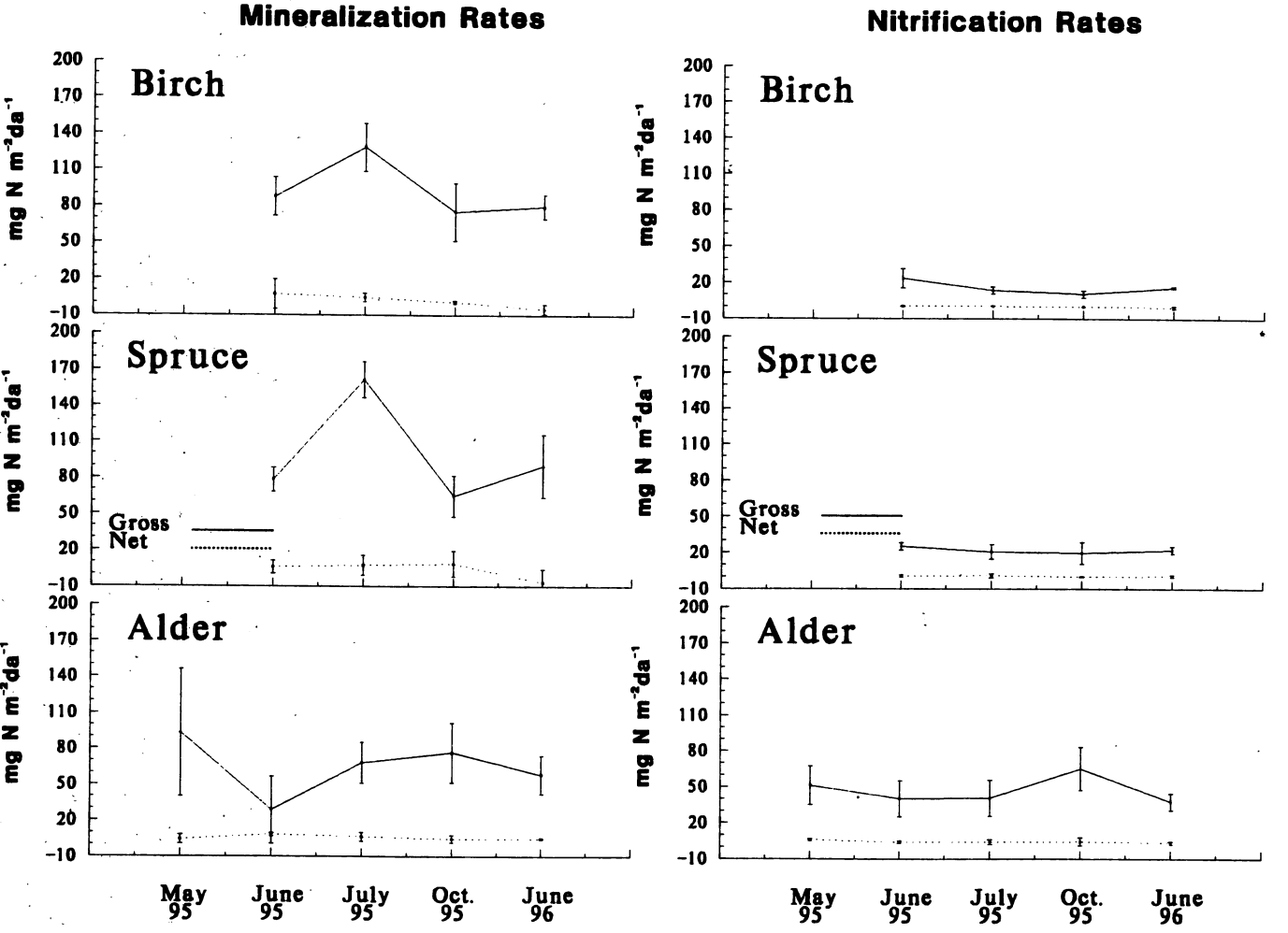

Figure 3. *Gross and net mineralization and nitrification rates in soils beneath major vegetation types, Wallace Lake*

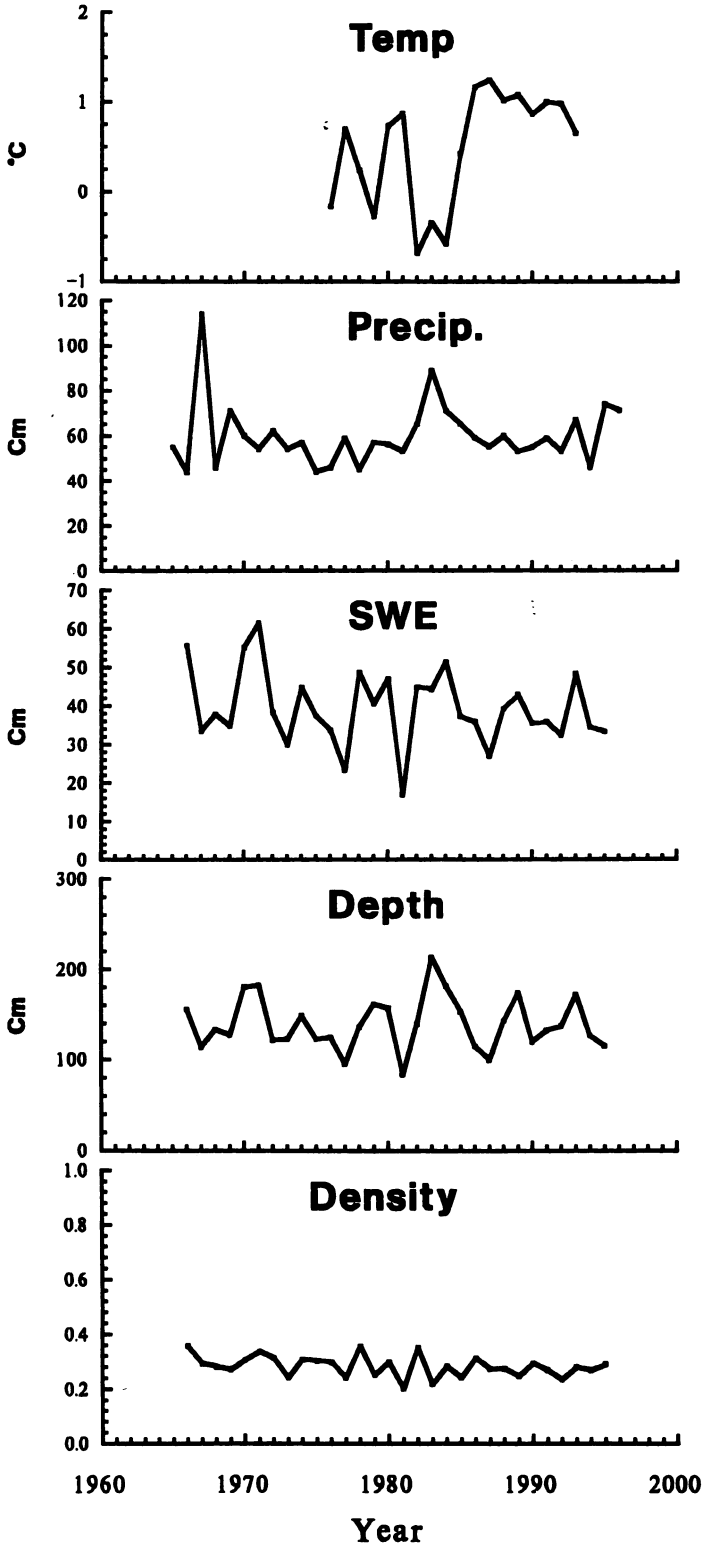

Figure 4. *Long-term trends in temperature, precipitation, snow water equivalent (SWE), snowpack depth and density, Lexen Creek, Fraser Experimental Forest*

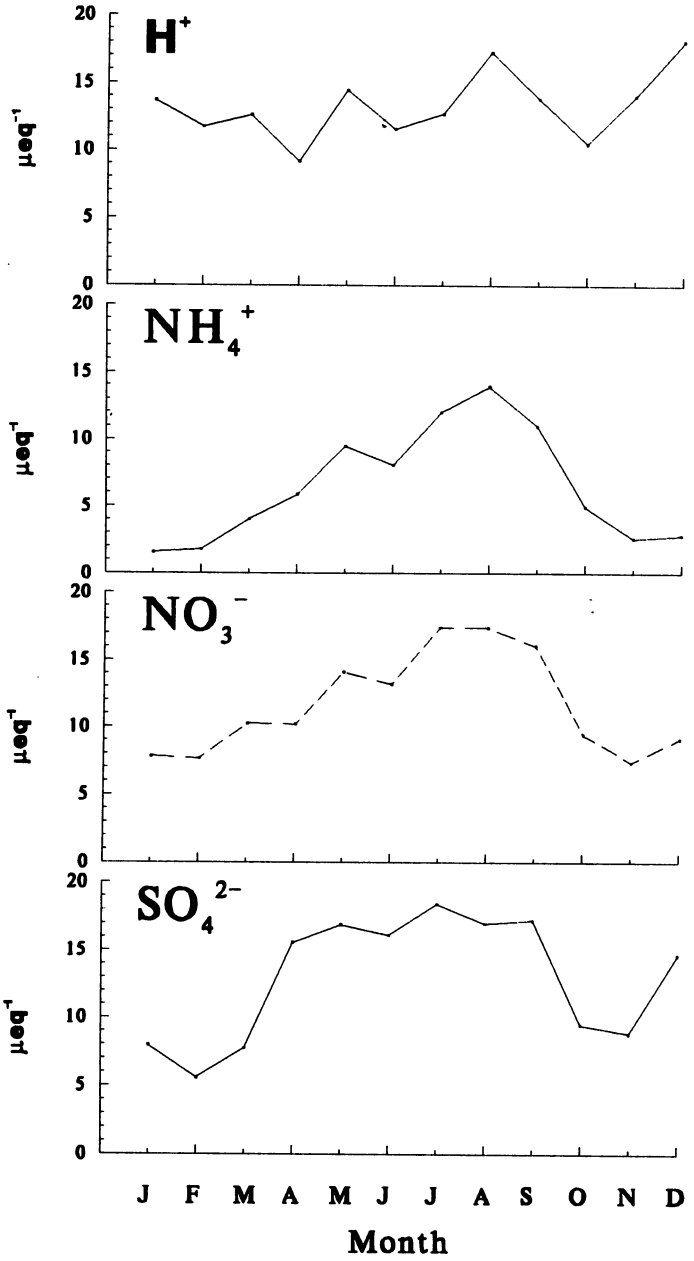

Figure 5. *Seasonal change in volume-weighted precipitation chemistry, Fraser Experimental Forest*

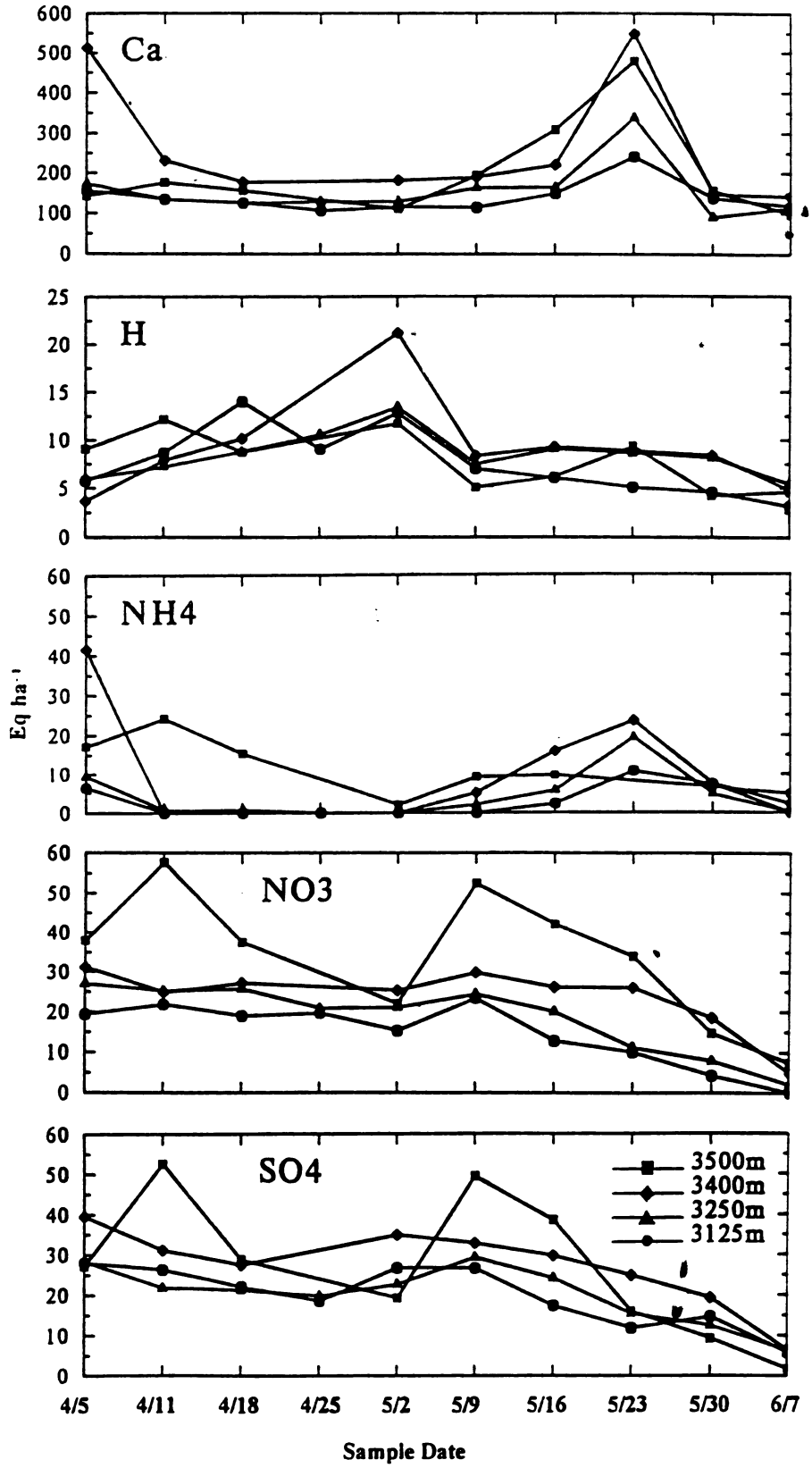

Fig. 6. ***Trends in snowpack ion content with elevation and season, Lexen Creek watershed, Fraser Experimental Forest.***

Significant nitrification within the snowpack has not been observed in high elevation Colorado sites.

In most Colorado subalpine ecosystems, soils remain unfrozen where a snowpack is present. For decades, it has been known most snowmelt in these high elevation ecosystems penetrates the soil or talus. In both the alpine and subalpine zones, most snowmelt passes through the soil before entering the stream. However, many studies of the relationship between precipitation N inputs, snowmelt release, and stream water chemistry have not examined the potential for precipitation chemistry modification by alpine and subalpine soils. Soil modification of precipitation or snowmelt chemistry can occur rapidly even in alpine talus (Campbell *et al.* 1995).

At the FEF, alpine mineral soils have a total N pool >700 kg ha^{-1}. *In situ* net N mineralization rates at Niwot Ridge are >10 kg ha^{-1} y^{-1}, and in subalpine forests of FEF, Loch Vale, and other Colorado sites range from 5 - <30 kg N ha^{-1} y^{-1} (Arthur and Fahey 1993, Stump and Binkley 1993). Alpine soil inorganic N pools are highest in winter and early spring at FEF and Niwot Ridge. In Colorado, the highest recorded wet precipitation N inputs are for Niwot Ridge (3 kg N ha^{-1} y^{-1}). Soil net mineralization rates and inorganic pools are large relative to precipitation inputs.

At the FEF, stream water ion concentrations are highest in winter. During snowmelt, NO_3^- concentrations at the mouth (station 49) of Lexen Creek decline to near detection levels (fig. 7). Alpine (station 49.1) stream water NO_3^- concentrations in Lexen Creek remain high but unchanged through most of the year. The different patterns of C_B and NO_3^- concentrations in alpine and subalpine stream water indicate alpine snowmelt is well mixed before reaching the stream. Watershed N outputs are <5% inputs.

Loch Vale, Rocky Mountain National Park, Colorado

During the last decade at Loch Vale, precipitation amount, NH_4^+ deposition (fig. 8, station CO98) and NO_3^- concentration increased ($p \leq 0.05$). Winter monthly precipitation amounts have increased with a significant gain ($p<0.05$) in January. NH_4^+ deposition increased ($p<0.05$) in March and May and NO_3^- in May. The precipitation NH_4^+/NO_3^- has declined as have summer precipitation amounts and ion inputs.

Where forest occurs in Loch Vale, canopy modification of incident precipitation amount and its chemical composition is similar to the FEF and other subalpine forests (Arthur and Fahey 1993).

Alpine stream water NO_3^- concentrations at Loch Vale and Niwot Ridge are higher than at the FEF (fig. 9). But soil processes appear to dominate stream and lake water quality rather than precipitation quality (Denning *et al.* 1991, Arthur and Fahey 1993, Campbell *et al.* 1995). As at FEF and other sites, studies of seasonal soil water quality show high NO_3^- concentrations in late winter and with initiation of snowmelt. Peaks in stream water DOC, Al, and nutrient species as NO_3^- with snowmelt indicate a pulse of concentrated soil solution reaching the stream. Meltwater infiltration into soils prevents lake and stream water from exhibiting sharp ANC declines. Recent research in Loch Vale found geochemical processes in soil, or even talus, significantly modify stream water chemistry, and these processes occur on timescales of hours or days (Campbell *et al.* 1995). The potential importance of soil processes on stream water chemistry during snowmelt has also been suggested for Glacier Lakes in Wyoming (Reuss *et al.* 1995) and Niwot Ridge in Colorado (Williams *et al.* 1996). In Loch Vale, studies using the distinct $\mu^{18}O$ and $\mu^{15}N$ ratios in precipitation, snowpack, and soil water, found the NO_3^- increase in stream water with early melt mainly comes from "old" NO_3^- in soil water (Kendall *et al.* 1995). At Loch Vale watershed N outputs are about half inputs.

Rock Creek, Denali National Park, Alaska

At Rock Creek, annual precipitation NO_3^- input is <1 and NH_4^+ <0.3 kg ha^{-1}, about the lowest input of any NADP station on the continent. NH_4^+ - N input about equals NO_3^- - N input. Canopy modification of incident precipitation has not been measured, but is likely less significant than at FEF or Wallace Lake since only the lower portion of the watershed is forested.

Net soil mineralization and nitrification during summer alone exceed 3 kg N ha^{-1}, or 30 times annual precipitation input (Stottlemyer 1992). Seasonal soil N mineralization rates are reflected in stream water NO_3^- concentration. Inorganic N outputs from the watershed during summer exceed annual input by 3 times. This is the only watershed in the network where inorganic N outputs exceed inputs. Annual outputs also exceed those from the Wallace or Lexen

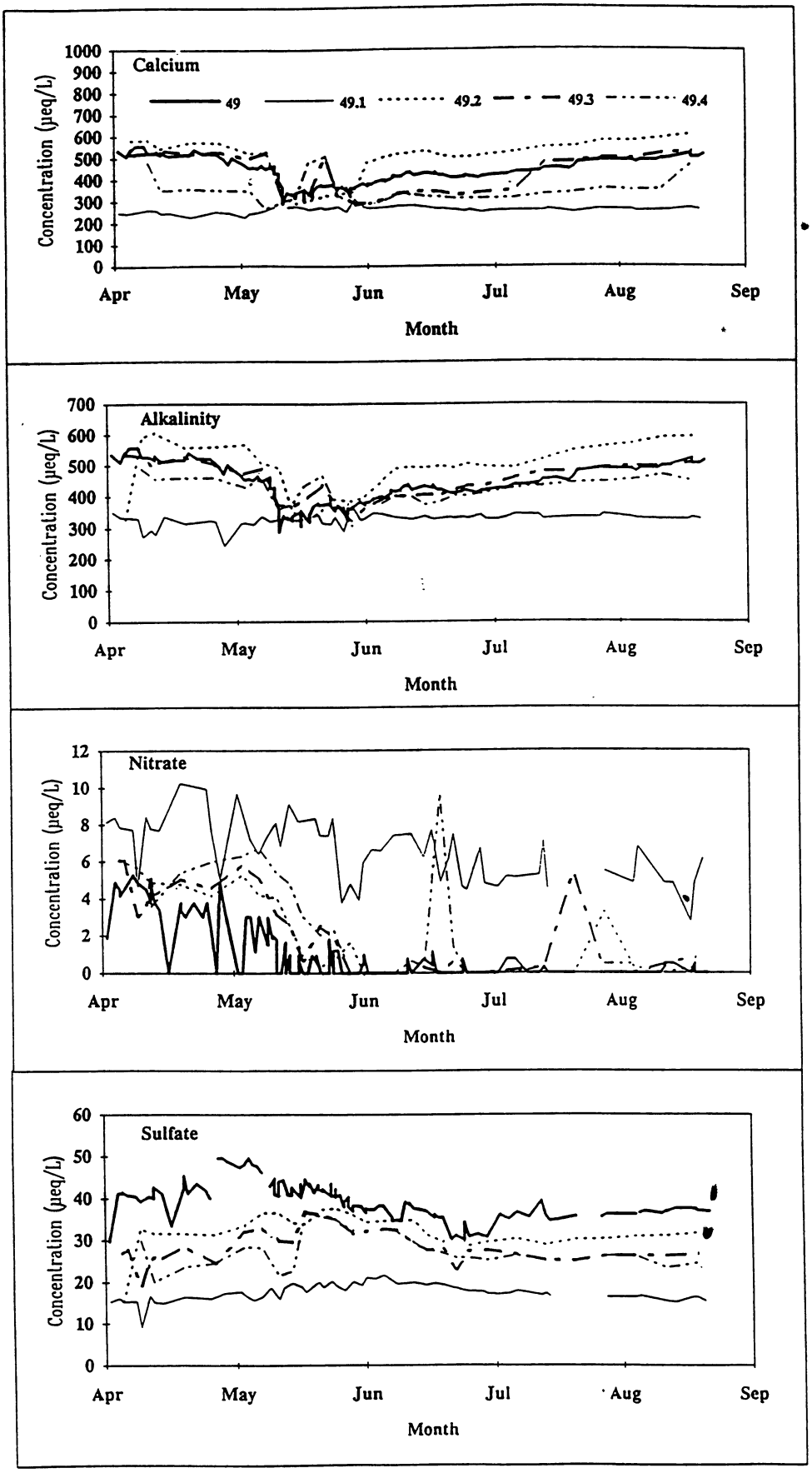

Fig. 7. ***Trends in alpine and subalpine stream water ion concentration, Lexen Creek, Fraser Experimental Forest.***

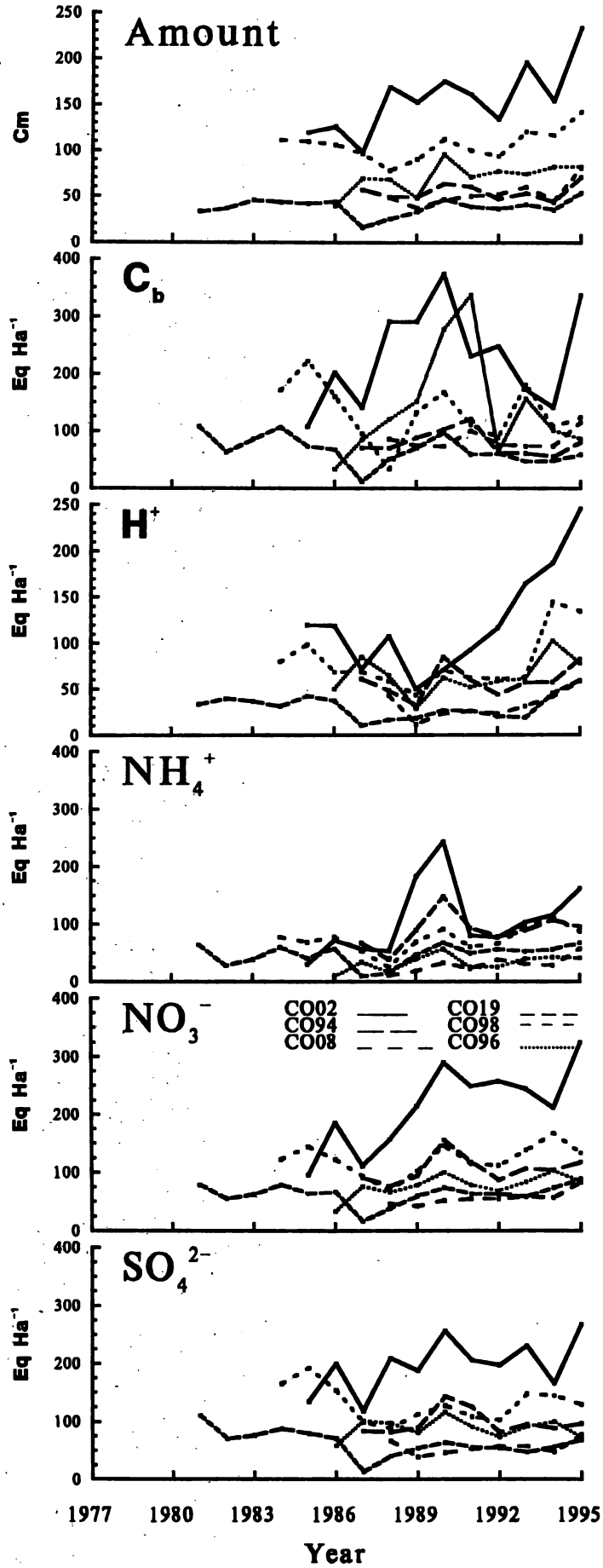

Figure 8. *Trends in precipitation inputs at higher elevation Colorado NADP stations*

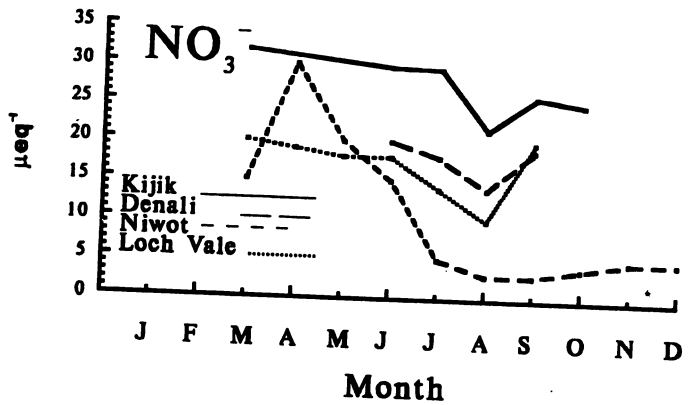

Figure 9. *Seasonal trends in stream and lake water NO_3^- concentration, Niwot Ridge and Loch Vale, Colorado, Rock Creek in Denali National Park and Preserve, and Kijik Lake, Lake Clark National Park and Preserve, Alaska*

watersheds, sites which receive much higher atmospheric inorganic N inputs.

Similar results have been found in other large Alaskan watersheds (Stednick 1981) and lakes, and high stream and lake NO_3^- concentrations (fig. 9) are not unusual in the region despite low atmospheric inputs.

SUMMARY

In most forested and alpine ecosystems, inorganic N is a limiting nutrient. The increase in human-derived atmospheric N inputs is a major concern since some ecosystem components can take more advantage of the input than others. This has potential to directly impact ecosystem biodiversity and processes. Some investigations suggest trends in stream and lake water NO_3^- concentrations are reflecting trends in precipitation inputs. The watershed-level studies reported here, and results from other similar studies, for a series of potentially sensitive sites receiving very different atmospheric N inputs, suggest soil processes likely have much more potential to determine long-term and seasonal change in stream or lake water NO_3^- concentration. Soil processes appear sensitive to modest change in temperature, and climate change could increase available ecosystem N.

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Carbon Cycling and Storage

THE EFFECTS OF FIRE ON ACTIVITIES DESIGNED TO INCREASE CARBON SEQUESTRATION IN THE BOREAL FOREST

Eric S. Kasischke[1]

ABSTRACT.-- This paper discusses the overall effects fire has on the carbon budget of boreal forests. Studies on using the boreal forest as a means to sequester carbon have not adequately accounted for these effects. Among other approaches, it has been suggested that suppression of fire in the boreal forest represent a means to increase terrestrial carbon storage. The impracticality of this approach is illustrated through a discussion of the costs associated with fighting fires in boreal regions.

INTRODUCTION

Study groups within the International Panel on Climate Change (IPCC) have focused on identifying approaches to using the world's forest as sequesters of carbon in order to offset the large amounts of carbon dioxide being released into the atmosphere through the burning of fossil fuels (Brown 1996). Approaches to increase carbon sequestration in forests focus primarily on: (1) planting more acreage in forests or improving forest regeneration after land clearing; (2) increasing the productivity of existing forests; and (3) preventing the clearing or destruction of existing forests, including increasing fire suppression activities (Dixon and Krankina 1993, Krankina and Dixon 1992, Nilsson and Schopfhauser 1995, Winjum *et al.* 1993). In developing carbon budgets for the boreal forest region, carbon sequestration studies to date have severely underestimated the effects of fire, both on current carbon budgets as well as in studies of the effects climate warming will have on future carbon budgets.

This paper presents data and arguments to support the following two premises:

1. Until the effects of fire are properly accounted for, studies on the amounts of carbon which can be sequestered in boreal forests will continue to be seriously flawed; and
2. Suppression of fire in the boreal forest does not represent a feasible means to increase carbon sequestration in the boreal forest.

To support these premises, this paper is divided into four sections, including this introduction. The second section discusses the overall role of the boreal forests relative to other forests in terms of carbon sequestration, and also discusses how fire influence carbon cycling in this biome. The third section discusses how future climate changes are likely to influence carbon storage in the boreal forest. The fourth section presents an example of the cost of fighting fires in boreal forests, which clearly illustrates the unfeasibility of using fire suppression to increase carbon storage. Finally, the fifth section presents the conclusions from this study.

EFFECTS OF FIRE ON THE BOREAL FOREST CARBON BUDGET

Large amounts of carbon are presently stored in the boreal region owing to two factors: (1) the presence of peatlands throughout the boreal forest which store enormous amounts of carbon; and (2) the presence of deep organic soil profiles in many boreal forest types. In terms of understanding carbon storage in the boreal forest, the presence of these large organic soil carbon pools represents unique circumstances. In addition, boreal forests are different from other forested regions because of the large role natural disturbance plays in natural processes. These natural disturbances are more important to the carbon cycle than human disturbances, such as deforestation and acid deposition.

Figure 1 presents a relational diagram of the major carbon pools present in the boreal forest along with the factors that control the exchange of carbon between these pools. The goal of carbon sequestration studies is simple: to identify approaches to decrease the amounts of carbon entering the atmospheric pool by increasing the amounts of carbon remaining in the other

[1]*Eric S. Kasischke, Geography Department, 2181 LeFrak Hall, University of Maryland, College Park, MD 20742. Email: kk169@umail.umd.edu.*

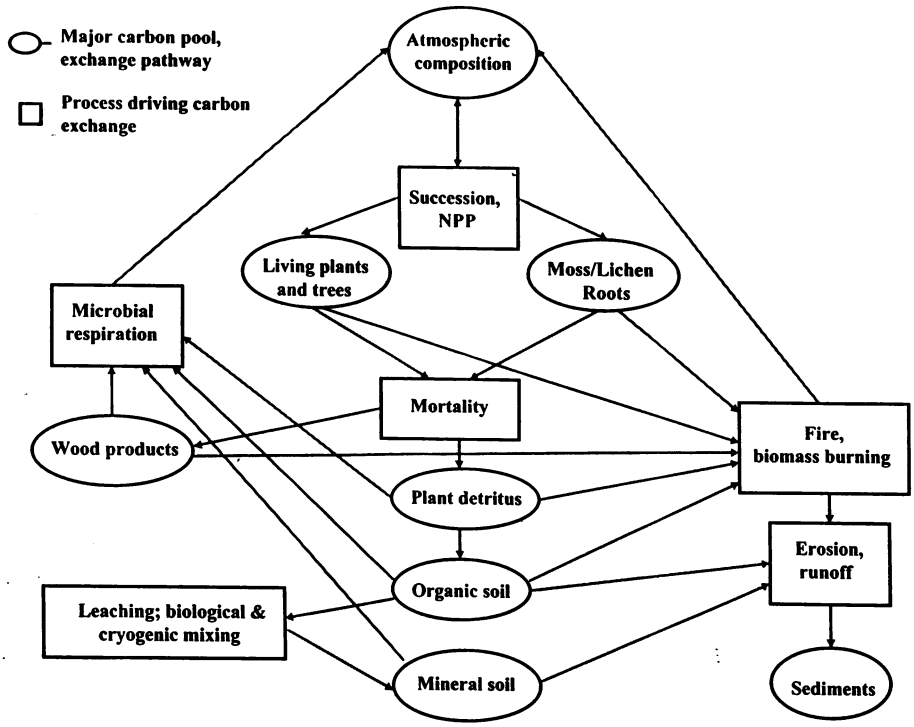

Figure 1.--*A schematic diagram of the major carbon pools in the boreal forest and the processes which control the exchange of carbon between these pools.*

pools (aboveground biomass, moss/lichen/roots, plant detritus, organic soil, mineral soil and sediments). It is important to note that the options for increasing the amounts of carbon residing in non-atmospheric pools require addressing the processes that control the exchanges between carbon pools.

The major shortcoming of most studies on increasing carbon sequestration in boreal forests is they focus on the processes controlling sequestration aboveground carbon pool (e.g., increasing or maintaining production of trees) without fully accounting for the processes which control exchanges of carbon in the plant detritus and organic soil pools. In terms of accounting for exchanges in these latter pools, fire plays an extremely important role whose effects have not been fully accounted for in carbon sequestration studies.

In terms of the effects of fire on carbon cycling in the boreal forest, there are three major areas of concern: (1) total annual area burned in the boreal forest; (2) amounts of biomass consumed during fires; and (3) the effects of biomass burning on post-fire biogenic emissions of carbon.

Annual Area Burned in Boreal Forests

The earliest study on global biomass burning by Seiler and Crutzen (1981) estimated that only 1 million hectares of boreal forest burned annually. One of the difficulties in developing an accurate estimate on the annual area burned in boreal forests is the lack of information from the Russian boreal forest. While maps of fire sizes, locations and boundaries are available for the North American boreal forest (Stocks *et al.* 1996), comparable information is not available

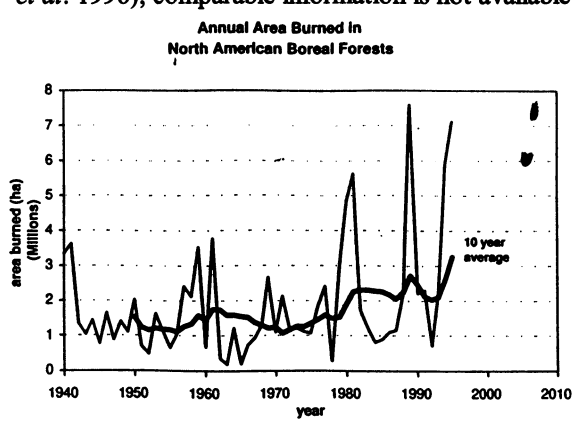

Figure 2.-- *Summary of annual area burned in the North American boreal forest - 1940 to 1996.*

for Russia. An analysis of the North American data record, as well as recent satellite imagery from Russia provides a basis to assess the level of biomass burning that occurs in the boreal forest.

Figure 2 presents a summary of the annual area burned in the North American boreal forest over the past half century. While the accuracy of the information prior to the 1970's is in doubt, over the past 3 decades the record keeping and mapping capabilities of fire monitoring agencies in Canada and the United States has improved considerably. Thus, the estimates of annual area burned in North American boreal forests provided since 1970 have a high degree of accuracy (Stocks *et al.* 1996). An important aspect of the more recent fire record for North America is the significant rise in the average area burned, which has risen from 1.5 million ha yr^{-1} in the mid-1970s to over 3 million ha yr^{-1} in the mid-1990's. This rise corresponds with a > 1.6° (C) rise in annual temperature in detected for the North American boreal region over the past 30 years (Hansen *et al.* 1996), and is consistent with predictions for increased fire activity with increased temperature (Flannigan and Van Wagner 1991; Wotton and Flannigan 1993).

Another important characteristic of fires in boreal forest is their periodic nature. Most of the area burned in boreal regions occurs during years when drought conditions are present. The periodicity of high fire years interspersed with low fire years is an outcome of the periodic nature of fire weather in the boreal region. Analysis of AVHRR satellite imagery of the Russian boreal forest is entirely consistent with this observation. During the extreme fire year in 1987, Cahoon *et al.* (1994) mapped > 11 million ha of burned area in Russia using AVHRR data, while in a year of moderate fire activity (1992), Cahoon *et al.* (1996) mapped 2.4 million ha of fire in Russia.

While most Russian authors estimate that the annual area burned in the boreal forest of their country averages 1 to 2 million ha yr^{-1} (Korovin 1996), the satellite evidence suggests that these estimates are low, by a factor of three or four. If it is assumed that the patterns of fire in the North American and Russian boreal forest are similar (as the satellite data suggest), and given the fact that northern Russia has experienced significant temperature increases over the past 30 years as well (Hansen *et al.* 1996), then the annual area burned in the boreal forest is probably close to 10 million ha yr^{-1} (because Russia contains twice as much boreal forest as North America), or an order of magnitude higher than early estimates.

Consumption of Organic Soil during Boreal Forest Fires

Another unique problem the boreal forest presents in terms of estimating the amounts of carbon released into the atmosphere during fires is the fact that the large volumes organic soil present in this biome are quite susceptible to burning. Most estimates of the amounts of biomass burned in boreal forest fires are based on information obtained during controlled burns. Stocks (1991) and Cahoon *et al.* (1994) use an estimated biomass consumption of 12 t ha^{-1} for both the above and below-ground layers during fires in the boreal forest.

While these experiments have provided a basis for estimating the amounts of carbon released in the aboveground layer for North American boreal forest, they may not provide adequate information to estimate the amounts of carbon released during biomass burning in the ground layer. This observation is supported by observation made on a recent forest fire that occurred in a black spruce complex in interior Alaska. Post-fire ground observations showed that the amounts of carbon stored in the ground layers (which includes litter, moss, lichen and organic soil) black spruce forests found within the fire affected region ranged between 50 t ha^{-1} and 105 t ha^{-1}. The amounts of the ground layer (in terms of total carbon) consumed during fire ranged between 25% and 92%. Data of soil depth and percent carbon consumed for three different areas in this burn are presented in figure 3. The amounts of carbon released during the fire were 25 t ha^{-1} for profile I, 45 t ha^{-1} for profile II, and 72 t ha^{-1} for profile III. Michalek *et al.* (1997) estimated that an average of 43 t ha^{-1} was released during this fire, which is four times the amount used in other studies (Stocks 1991; Cahoon *et al.* 1994).

Post-Fire Biogenic Emissions

The formation of deep organic soils in boreal forests is the result of several factors, including: (1) low average annual temperatures; and (2) the presence of a continuous moss layer which further restricts energy and water exchange between the atmosphere and land surface. These conditions often result in the formation of permafrost in boreal forests. The presence of permafrost under a boreal forest is not continuous over longer time periods. Studies have shown that permafrost often disappears after the occurrence of a fire because: (1) the removal of the tree canopy by fire significantly increases the amount of solar radiation reaching the forest floor; (2) the removal of moss and organic soil decreases the insulating capacity of the ground layer; and (3) the

removal of plant material and replacing it with charred and uncharred organic soil significantly reduces the albedo or reflectivity of the ground layer, meaning more solar energy is absorbed by the ground (Viereck 1983). The result of fires in boreal forests is that there is a significant increase in the ground temperature for about 15 to 25 years after the fire. The severity of burning in the ground layer (organic soil) significantly influences the degree of temperature increase, as illustrated in figure 3. In the area where only 25% of the organic soil was consumed, the ground temperature (to 100 cm depth in the mineral soil) was 2° C warmer than in an adjacent unburned stand. In contrast, in the areas where 92% of the organic soil was consumed, the average ground temperature had risen by 6° C.

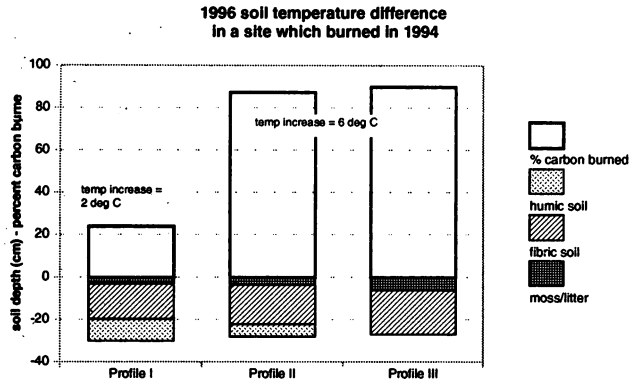

Figure 3.-- *Patterns of ground-layer biomass consumption and effects on ground temperature from three test areas in a 1994 forest fire near Delta Junction, Alaska.*

The rates of soil respiration in organic soils in boreal forests are strongly controlled by temperature (Schlentner and Van Cleve 1985), and the increases in soil temperature following a fire results in net increases in soil respiration. Recent studies by O'Neill *et al.* (1997) showed that the microbial soil respiration in an unburned black spruce stand was significantly lower when compared to respiration in a stand which had burned 5 years earlier.

POTENTIAL EFFECTS OF CLIMATE CHANGE ON BOREAL FOREST CARBON STORAGE

Most studies of the effects of climate change on boreal forests have focused on influences on the distribution and productivity of forests (Smith *et al.* 1992), and have concluded that while increases in temperature would result in increases in average aboveground biomass, they would also result in a net loss of carbon due to increases in soil respiration. However, such studies have not fully accounted for the influence of the effects of climate on the fire cycle and subsequent effects of fire on carbon storage in the boreal forest.

Kasischke *et al.* (1995) presented a simple, but illustrative model on these effects based on looking at several factors: (1) the relationship between fire return interval and stand age distributions in boreal forests; (2) changes in patterns of carbon accumulation in aboveground vegetation and ground-layer biomass associated with changing climate; and (3) the influence of climate on fire frequency and severity.

A high percentage of fires in boreal forest kill all overstory trees, e.g., they are stand replacement fires. In such situations, Van Wagner (1978) and Yarie (1981) have shown that a Weibull distribution can be used to estimate the stand age distribution for such forests. In essence, the shorter the fire return interval, the larger number of young stands that are present (fig. 4). For a given forest type, it is fairly easy to develop a plot of changes in above and below-ground biomass/carbon as a function of stand age (fig. 5). By integrating the two functions of carbon versus stand age with a number stand age distributions for different fire return intervals, it is possible to estimate how average carbon levels vary as a function of fire return interval (fig. 6). Figure 6 illustrates a fundamental relationship: as the fire return interval decreases (e.g., as the average area burned per year increases), the carbon storage potential of a given forest decreases. Therefore, if there is an increase in fire frequency with increased temperatures (as predicted by all climate change models for the boreal region), then the result would be less carbon storage in this biome.

In addition, however, one must account for the changes in the carbon storage depicted in figure 5. Increases in average temperature in the boreal zone will likely result in: (a) increases in the overall pattern of aboveground carbon storage in aboveground vegetation through shifts in forest types and increases in productivity; (b) increased rates of soil respiration, which results in decreases in the patterns of carbon storage in the ground layer (figs. 7 and 8). These changes, in turn, results in changes in the baseline patterns of average carbon storage, as depicted in figures 9 and 10. If it is assumed that the fire return interval decreases from 125 to 100 years, then the increase in average aboveground carbon of 1.3 kg m^{-2} will be outweighed by the loss of 4.8 kg m^{-2} of carbon from the ground layer.

THE COST OF FIGHTING FIRES IN THE BOREAL FOREST - IS FIRE SUPPRESSION AN OPTION FOR INCREASING CARBON SEQUESTRATION IN THE BOREAL FOREST?

Recognizing the role of fires in the carbon cycle, some researchers have suggested that additional fire suppression represent a means for increasing carbon sequestration in the boreal forest (Dixon and Krankina 1993). A simple example illustrates the fallacy of this argument.

The 1990 fire season was the most severe one on record in Alaska, with some 1.3 million hectares being consumed by fires. One of these fires began on 1 July within 1 km of the village of Tok, which is also the site of a regional forest fire-fighting center for the State Department of Natural Resources. In spite of the proximity to the fire suppression unit, because the fire was a distance off the road, by the time that personnel reached the fire it was already big enough that it could not be extinguished or contained using ground crews alone. Fortunately, a large, paved air strip was located within 25 km of Tok and arrangements were made to assign a number of air tankers (both water and retardant) to battle the fire. In addition, Tok is located at the junction of two of Alaska's major highways and it was possible to rapidly deploy additional ground crews from other regions of the state. Because of an extensive drought, the fuel conditions were extremely dry, and the fire quickly spread in spite of intensive suppression efforts. By July 4, the fire had consumed over 20,000 ha of forest and the community of Tok was evacuated. The only thing that saved the village from the fire was a major wind shift.

The fire suppression costs for the Tok fire eventually reached $35 million dollars for a fire whose size was 40,000 ha by the end of the fire season. The suppression costs were significantly reduced (relative to fires in other regions of the state) because of the close proximity to an air field and a logistical support center. This example illustrates a fact that is clear to foresters in both Alaska and Canada: that it is both unfeasible and impractical to suppress all fires in the North American boreal forest. Instead, North American foresters practice modified fire suppression, e.g., fires are actively suppressed only when they directly threaten human life or property.

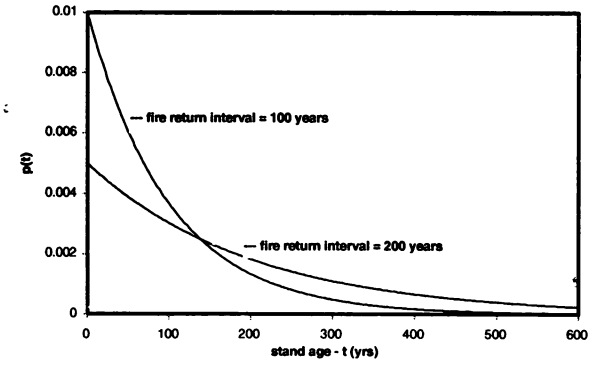

Figure 4.--*Influence of changes of fire return interval on stand age distribution based on a Weibull Distribution.*

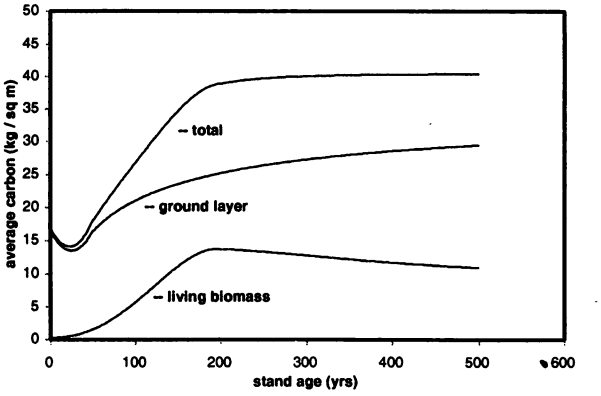

Figure 5.--*Patterns of carbon storage as a function of stand age for a typical boreal forest.*

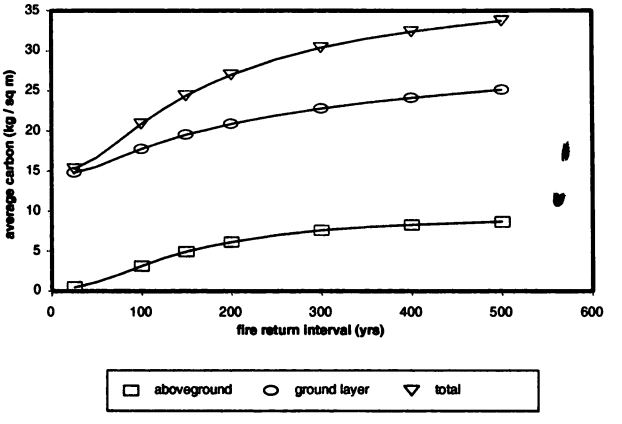

Figure 6.--*Patterns of average carbon storage as a function of fire return interval based on carbon storage patterns presented in Figure 5.*

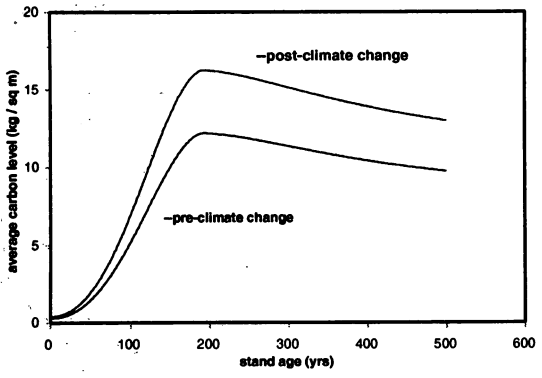

Figure 7.--*Projected effects of climate change on patterns of aboveground carbon accumulation.*

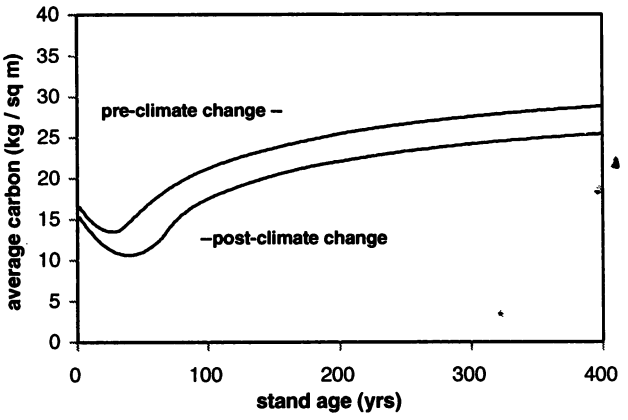

Figure 8.--*Projected effects of climate change on ground-layer patterns of carbon accumulation.*

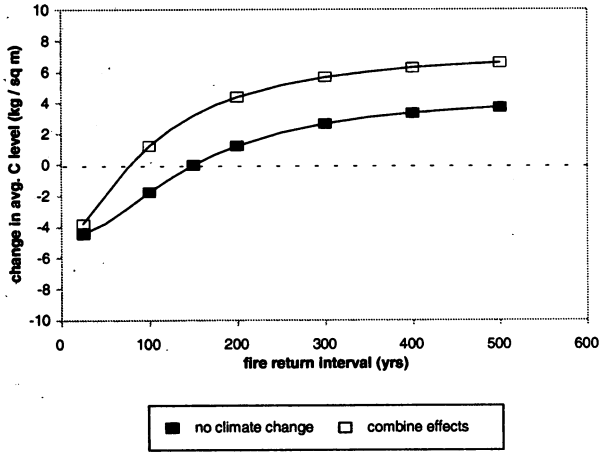

Figure 9.--*Effects of climate change and fire return interval on total carbon storage in aboveground biomass in boreal forests.*

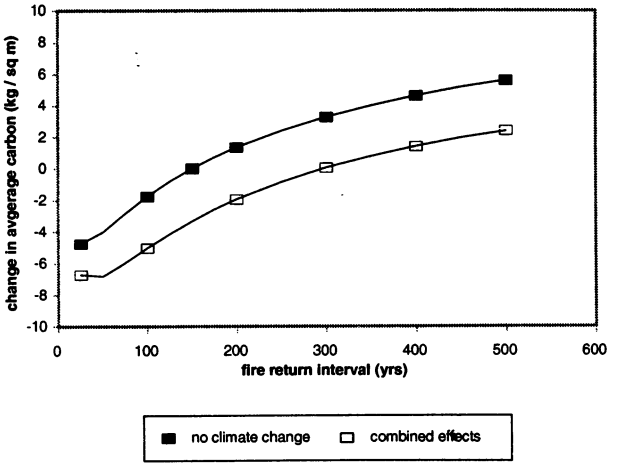

Figure 10.--*Effects of climate change and fire return interval on total carbon storage in below-ground biomass in boreal forests.*

CONCLUSIONS

Understanding the effects of global climate changes and developing viable strategies and options for responding to these changes represents a significant challenge to the scientific and policy communities. The boreal forest region is at the center of studies involved in understanding the role of the world's forests in the global carbon cycle, and developing approaches to mitigating the effects of additions of greenhouse gases to the atmosphere from the burning of fossil fuels through increasing the amounts of carbon stored in terrestrial biomes. The importance of the boreal forest stems from two facts: (1) the large amounts of carbon stored in the boreal region, which is the world's largest reservoir of terrestrial carbon; and (2) the location of the boreal forest in the region where the greatest increases in temperature are expected over the next century.

Fire plays a central role in the exchange of carbon between the terrestrial biomes of the boreal region and the atmosphere. While the importance of this role has only recently been recognized by the broader scientific community, it has not been properly accounted for by scientists developing carbon budgets and approaches needed to sequester carbon in terrestrial ecosystems. While the pathways where fire influences carbon exchange in the boreal forest have been identified, the rates of exchange have yet to be fully quantified. This quantification requires efforts in the following areas: (1) improving data bases of the location and extent of fires in the Russian boreal forest. This information can easily be provided through analysis of satellite imagery; (2) studies of the patterns of biomass burning and consumption of biomass in the different forest ecosystem which comprise the boreal forest. Particular attention needs

to be paid to the extent of smoldering fires in organic soils; and (3) studies on changes in soil moisture and temperature the rates of soil respiration after fires in different boreal forests.

Finally, suggestions have been made that increased carbon sequestration in the boreal forest can be achieved through increased fire suppression activities. This suggestion is totally impractical and undesirable for three reasons. First, most fires in boreal forest occur in remote regions at great distances from fire fighting equipment and manpower required. The cost of deploying these resources is extremely high, and suppressing all boreal forest fires would require financial resources far beyond those that are available. Second, even if unlimited resources are available, once a fire reaches a certain size (e.g., several hundred hectares), it is virtually impossible to suppress without resources beyond those available at the time. And third, because fire is an integral part of the ecology of all boreal forests, fire suppression is not a valued resource management option. In many cases, such as wildlife habitat management, fire is viewed as an extremely valuable process.

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A SPATIALLY EXPLICIT MAP OF FOREST STAND CARBON FOR RUSSIA: A FIRST APPROXIMATION

T. A. Stone[1], P. Schlesinger[1], and V. A. Alexeyev[2]

ABSTRACT.--We have created a map of forest stand carbon for Russia from data on total forest stand carbon by administrative districts from Alexeyev and Birdsey (1994, 1998) and with a 1973 forest cover map of the former USSR (State Committee on Forests 1973). Each carbon value has been applied to the map by tree species or genera. The identifier of each forest polygon was a percentage of the area of forest species per administrative district. Carbon values were applied to the map and then converted to tons of C ha^{-1}. The resulting map shows forest stand C by region and tree genus in a spatially explicit form.

INTRODUCTION

A fifth of the world's forests are in Russia (United Nations 1992). These forests are one of the world's great repositories of terrestrial carbon. It is necessary to understand the spatial distribution of this carbon for forest management and to help ascertain its role in the global carbon cycle. If the spatial distribution of carbon (C) were known, it would improve the accuracy of measuring carbon flux from land use change and help in determining the emissions of gasses and particulates from biomass burning. Also, knowing the distribution of C would aid in defining respiration values for ecosystem models, provide a baseline for changes in C stocks, and allow for validation for ecological process models and dynamic global vegetation models.

In this work, we have combined forest tree stand carbon data with a forest cover map of Russia to produce a map of the spatial distribution of forest stand carbon. We have chosen to use the comprehensive carbon data of Alexeyev and Birdsey (1994, 1998) and to combine it with a map of forest cover from the Forest Atlas of USSR (State Committee on Forests 1973) that we digitized here.

Given the importance of the region, it is not surprising that many authors have developed estimates for the carbon content of the forests of Russia (Shvidenko *et al.* 1996; Isaev *et al.* 1995; Kolchugina and Vinson 1993, 1995; Dixon *et al.* 1994; Lakida *et al.* 1996). Different methodologies have led to widely different estimates of the C contents of forests in spite of the fact that most researchers have use the same statistical data from national inventories as a starting point. Some have determined C for the entire country (Isaev *et al.* 1993, 1995; Dixon *et al.* 1994) while others have used eco-regions or economic regions (Kolchugina and Vinson 1993, 1995; Lakida *et al.* 1996). Isaev *et al.* (1995) estimated that forest carbon was about 35,070 Tg C (35.07 Pg), Kolchugina and Vinson (1993) estimated that forest carbon (above ground woody phytomass) was about 36.7 Pg and Dixon *et al.* (1994) estimated the vegetation carbon pool of Russia to be to be about 74 Pg. The extreme heterogeneity and difficulty of access of forested Russia are also partial explanations for the disparity in these estimates.

The most detailed estimates of the C content of forests have been made by Alexeyev and Birdsey (1994, 1998). They defined the C in forests for the 71 political units of the Russian Federation (oblasts, krays, and republics), for 63 eco-regions, and for each of the forest forming tree species and age class groups.

METHODOLOGY

We constructed an area-weighted map of forest stand carbon using data from Alexeyev and Birdsey (1994, 1998) and a forest cover map of the Soviet Union digitized from the 1973 Forest Atlas of the USSR (State Committee on Forests 1973). The 1973 map (published at 1:15,000,000 scale) is comprised of 22 different cover types: Pine, Spruce, Fir, Spruce/Fir, Larch, Kedra (Siberian and Korean Pine), Juniper, Creeping Cedar, Oak, Beech/Hornbeam, Stone Birch, Sand/Haloxylon, Birch, Aspen, Nonforest, Water, Tilia, other woody and sparse categories. The 1973 map was digitized here using Roots[tm] software and

[1]*The Woods Hole Research Center, PO Box 296, Woods Hole, Mass., 02543, USA.*

[2]*St. Petersburg Research Institute of Forestry, 21 Institutskiy Ave, St. Petersburg, 194021, Russia.*

imported into our geographic information system (GIS). It was the best map available to us at the beginning of this research.

There are three levels of forest maps in Russia: local, regional, and national. Local maps have scales from 1:25,000 to 1:200,000. They are the most detailed maps and are where composition, site quality class, and age-class groups are introduced. However, the composition of stands on such maps is based on dominant species and is at the genus, not species, level. Regional forest maps have scales from 1:500,000 to 1:1.5 million. Such maps also show detailed information, especially maps reproduced as Forest Atlases (State Committee on Forests 1973). But, forest plots less than 500 ha cannot be shown on the maps. Information on the national-level forest maps is the most generalized. We know two Russian forest maps for the whole country, the 1990 USSR Forest Cover Map (Garsia 1990) at 1:2,500,000 and the 1973 USSR Forest Cover Map (State Committee on Forests 1973) with a scale of 1:15,000,000. The smallest size of forest plot that can be displayed on the latter map is 9,000 ha. Thus, such national-level maps can only approximate the situation.

Forested polygons from the digitized 1973 forest map were assigned carbon values according to the percentage of their area in different tree species. The carbon values for forests (in million of metric tons, MT, of C), were taken from table C-1 of Alexeyev and Birdsey (1994, 1998). These values represent the sum of total growing stock of various forest age classes (young, middle-aged, maturing, mature and over-mature). These values were listed by administrative or political district (i.e., republic, *oblast* and *kray*) and by forest type.

Carbon was assigned to each of the 71 administrative districts of Russia using the formula:

Area-weighted Carbon = (Area of forest polygon / Area of forest species) x Tot. MT C per species

Estimating carbon in this manner has certain drawbacks. For instance, we found some disagreement in 68 of 71 administrative districts between the forest tree species described in the 1973 map and those listed by Alexeyev and Birdsey (1994). These disagreements accounted for 4.2% of a published total of 26.1 x 10^9 tons C in tree stands and shrubs. The most likely causes for the disagreement were:

1. the different scales of the forest map we digitized (1:15 million scale) and the administrative district boundary file data (1:8 million);

2. the (unknown) scale of the original data used to create the published values of carbon; and,

3. the differences between the map classifications that were used.

However, these differences accounted for less than 1% of the total carbon mapped.

During the course of this work this work, a digital version of the 1990 Forest Cover Map of the USSR (Garsia 1990) became available. Consequently, this effort could be redone with the higher resolution map. But, before making a high-resolution forest carbon map, we need to consider the scale of the forest carbon data that has gone into the tables of Alexeyev and Birdsey (1994). Using the 1:2,500,000 scale map from 1990 as the basis of the carbon map would provide the appearance of a higher resolution, forest stand carbon map. But, considering the scale of the carbon data, the improvement may be deceptively attractive. Additionally, Shvidenko (personal communication 1997) has indicated that the 1990 Forest Cover map was a hasty and incomplete updating of the maps from the 1973 Atlas.

RESULTS

This newly calculated map (fig. 1) is the only map of forest stand carbon for the Russian territory of which we are aware. The forest stand C mapped by this effort represents approximately 96% of the total of 26,103 Tg (26.1 Pg) forest tree stand C described by Alexeyev and Birdsey (1994) and Alexeyev *et al.* (1995). Of the remaining 4%, nearly half is due to bushes, which were assumed not to be mapped in the 1973 forest cover map.

The primary drawback to this method is that within individual administrative regions we needed to assume that forest density remains constant while we know that, generally, forest density declines as one moves up slope or farther north. For instance, the larch forests of northern Krasnoyarsk Kray in central Siberia contain trees that can be hundreds of years old, of small diameter, and very sparsely distributed (Bondarev 1995). But, our application of C to the map assumes, in this first approximation, that the larch forests in the north are the same as larch forests

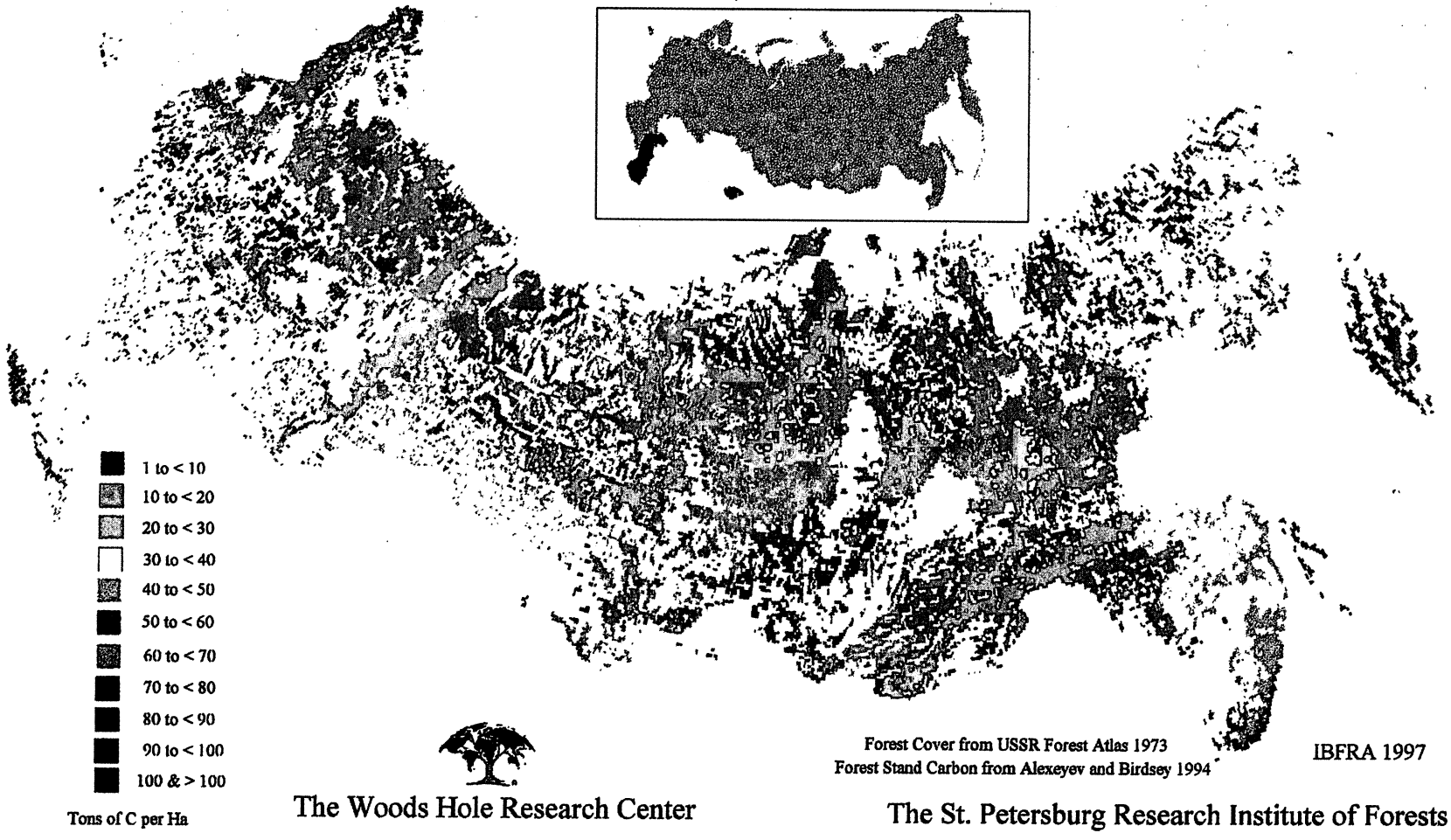

Figure 1.--*Distribution of forest stand carbon in Russia. Forest cover from USSR Forest Atlas (1973). Forest stand carbon for Alexeyev and Burdsey (1994).*

growing 1,500 km to the south. Thus, the map is most generalized in the largest administrative districts and more accurate in the smaller administrative districts.

DISCUSSION

When considering the map developed here, it is important to remember that forest stand carbon is only a fraction of the total carbon in forested ecosystems of Russia. Forest soils alone may store 3 times as much carbon as the forested stands (Alexeyev *et al.* 1995) and total soil carbon in Russia (Rozhkov *et al.* 1996) may be 10 times forest stand carbon.

This map is a static representation of a dynamic region. That is, this map is for a point in time and does not reflect the dynamic nature of the forests of Russia where up to 14 million ha of forests can burn in an exceptional year (Stocks 1991) and 2. 5 million ha burns in an average year (Krankina 1993). Current estimates (Korovin 1995) are that fires burn 2 to 5 million ha yr^{-1} of forests in Russia or about the same magnitude as the rates of forest harvest. Consequently, this map is unsuitable for use in discussions about the sources and sinks of carbon in the forests of Russia.

There are a number of differences between the map and the forest stand C data from Alexeyev and Birdsey (1994). We explain these below.

1. The tree species used in the map from the USSR Forest Atlas (State Committee On Forests 1973) and from the tables of Alexeyev and Birdsey (1994) were not identical. Only forest genera were described in the Forest Atlas map. To adjust for this, we have added the most likely tree species to the C totals of the genera before assigning the data to the map. For example, in Krasnodarski Kray (region #40) and Orenburgskaya Oblast (#48), *Populus tremula* was combined with *Populus spp.* And, in Chelyabinskaya Oblast (#51), where the map had only a spruce-fir category, yet the tables had both tree species listed individually, the data were combined and then mapped. Also, in Krasnoyarsk Kray (#60) and Kamchatskaya Oblast (#68), carbon for *Betula spp.* was not added to *Betula ermanii.* In the regions of Primorskii Kray and Khabarovskii Kray, the genera *Quercus* were mapped, while the tables listed the species as *Quercus mongolica.* In the same regions, the map showed Kedra (*Pinus spp.)*, while the table listed *Pinus korayensis.* In Kamchatskaya, the species *Larix camchatica* was likely mapped simply as Larix.

2. Some species in the tables differed entirely from those on the map. For example, in Irkutskaya (region #61) the tables showed 71.53 Tg C of aspen, while the map did not indicate any aspen, and in Magadanskaya (#69), where *Chozenia arbutifolia* (a poplar species) accounts for 13.62 Tg C, the map did not contain that genus. The map does have an "other hardwood" class, and while it might be appropriate to assign carbon from *Populus* to that class, it is unknown whether the "other" class contained only poplar.

3. The 1973 map is highly generalized, necessitating careful comparisons among administrative regions. Generalization of both the map and C table summaries do not allow for adequate differentiation between the mapped tree species (i.e., *Picea obovata* in Komi and *P. ajanensis* in Sakhalinskaya), or account for the complexity of mixed species at the ground level,

Some other caveats:

(a) None of the small-scale Russian or Soviet maps of forest cover (e.g., Forests of the USSR 1990, Garsia 1990) adequately represent the variation encountered either in the *vydel* (the minimum forest mapping unit in Russia) level database or the actual forest in many locations. Forest type is seen to be constant across a polygon, whereas in reality it is likely only the prevalent genera. Although there are maps at the *leshoz-* level based on *vydels*, they are generally not available.

(b) The 1973 Forest Atlas Map is undoubtedly a generalization of forest knowledge of an earlier time, probably the mid 1960's but perhaps even earlier. It was chosen because it was the only digital forest cover map available at the beginning of this research.

(c) The scale of the 1973 Forest Atlas map was quite small, 1:15 million. At this scale, lines indicating boundaries of polygons are approximately 15 km wide introducing significant errors particularly for

very small polygons. The scale of the source materials for the 1:15 million map are likely the 1:1 - 2 million scale maps of the territories in the Atlas. The quality of those source maps is unknown.

(d) It is not possible to ascertain now whether the 2% of total C which were bushes were actually mapped as Krumholz or class #8 on the 1973 Forest Map; many of the bushes tabulated vary in age from 50 to 100 years.

(e) The map does not indicate carbon in soils, peats, or mortmass (dead trees, branches, and litter).

(f) The carbon map developed here is more appropriate for smaller, rather than larger regions because of the assumptions about forest density and site index or *bonitet*.

FUTURE WORK

This work could be repeated relatively easily with the 1990 Forest Map of the USSR at 1:2.5 million scale. Also, this work could also be repeated with future maps of the forests of Russia at 1-km resolution. To do this, we would use the same administrative datasets and algorithms used in the current effort. A more reliable administrative boundary layer map would help.

Data for forest type stand volume, and increment exists for all Russia at the local forest administrative unit (*leshoz*) level. Given that there are more than 2,500 of these units in the former Soviet Union, the map could be further refined if the boundary files for these management units were available. Although this would be a considerable effort, using these units in a similar manner would result in a 30-fold increase in resolution for forest stand carbon mapping.

Finally, we are planning to merge this map with digital soil carbon maps of Russia available through Russian collaborators. This will allow us to show the spatial distribution of tree stand C and soil C and to examine the spatial relationship between these two major C pools in Russia.

ACKNOWLEDGEMENTS

The authors would like to thank NASA's Office of Mission to Planet Earth (MTPE) NAGW 3856, the W. Alton Jones Foundation and the John D. and Catherine T. MacArthur Foundation for their support.

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SCALING SOIL CARBON STOCKS AND FLUXES AT THE BOREAS NORTHERN STUDY AREA

G. Rapalee, E. A. Davidson, J. W. Harden, S. E. Trumbore, and H. Veldhuis[1]

ABSTRACT. - Boreal forests, currently thought to be significant carbon sinks in global C budgets, could become net C sources as the earth warms. Most of the carbon in boreal forests is stored in the soil and moss. We mapped soil carbon stocks, soil drainage, and fire history at a BOREAS (BOReal Atmosphere-Ecosystem Study under NASA's Earth Science Enterprise) study site in northern Manitoba. To calculate C fluxes, we applied algorithms for inputs and decomposition constants based on fieldwork using radiocarbon analyses. Soil C stocks covary with drainage class, with largest stocks occurring in poorly drained sites. While upper moss layers regrow and accumulate carbon after fires, deep C flux in years between fires can range from a small sink to a significant source large enough to nearly counter the surface moss sink, resulting in an annual carbon flux of nearly zero for the entire soil profile.

INTRODUCTION

High-latitude ecosystems would be expected to experience the largest temperature changes if the earth warms. Boreal forest soils, where most soil carbon (C) accumulates in the moss layer and organic and humic horizons, are both an important source of and a sink for atmospheric carbon. Therefore, responses of boreal forest soils to warming could have important effects on the interactions between climate change and terrestrial carbon storage. Drainage and incidence of fire are the factors thought to be most important for controlling annual accumulation rates of carbon. Understanding the present patterns of soil carbon distribution and the soil properties that covary with soil carbon is essential for projecting future changes.

STUDY AREA

The 733 km^2 study area, located between Thompson and Nelson House, Manitoba, is close to the northern limit of the closed-crown boreal forest. The area includes well-drained upland jack pine stands on sandy soil, black spruce dominated stands on moderately to imperfectly drained soils developed on lacustrine clay, black spruce stands with abundant sphagnum moss ground cover on more poorly drained soils developed on clay, and low-lying wetlands consisting of fens, bogs, and palsas.

PROJECT GOALS

Our goals were: (1) to estimate total carbon stocks by horizon for common soil series based on soil survey data and analyses of data from individual soil profiles; (2) to estimate soil carbon flux based on carbon stocks and a simple model of C turnover derived from radiocarbon studies; (3) to generate area-weighted maps of soil carbon stocks and flux; (4) relate patterns of carbon stocks and flux to patterns of drainage, moss cover, and fire history.

METHODS

To accomplish our goals, we: (1) stratified the study area by drainage class/vegetation type (fig. 1); (2) determined time since last fire (fig. 2); (3) separated the soil profile into two broad layers -- surface and deep -- with distinctly different C accumulation rates; (4) estimated C stocks for surface layers based on a time-dependent model of moss growth after fire; (5) estimated C stocks for deep layers from profile data and soil series map; and (6) estimated annual C fluxes for 1994 from a simple model of input (I), decomposition (k) constants, and C stocks where net annual flux = $I - kC$.

[1]*Gloria Rapalee, The Woods Hole Research Center, and Salem (MA) State College, now at Goddard Space Flight Center, Code 923, Greenbelt MD 20771, grapalee@pop900@gsfc.nasa.gov; Eric A. Davidson, The Woods Hole Research Center, Woods Hole MA 02543, edavidson@whrc.org; Jennifer W. Harden, U.S. Geological Survey, Menlo Park CA 94025, jharden@isdmnl.wr.usgs.gov; Susan E. Trumbore, Dept. of Earth System Science, University of California, Irvine 92717, setrumbo@uci.edu; Hugo Veldhuis, Agriculture and Agri-Food Canada, Brandon Research Centre, Manitoba Land Resource Unit, Winnipeg MB R3T 2N2, VeldhuisH@em.agr.ca.*

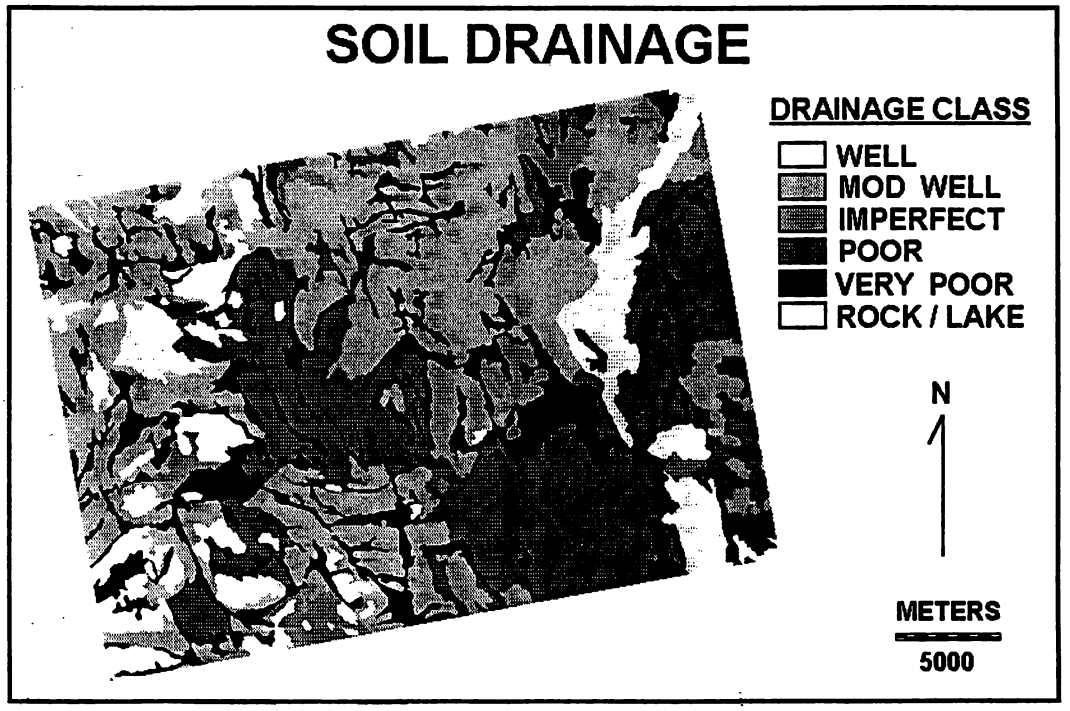

Figure 1. -- *Soil drainage by dominant soil series of soil polygon. Five drainage classes correspond to vegetation type. Figure is from Rapalee et al. (1998), used with permission from the American Geophysical Union.*

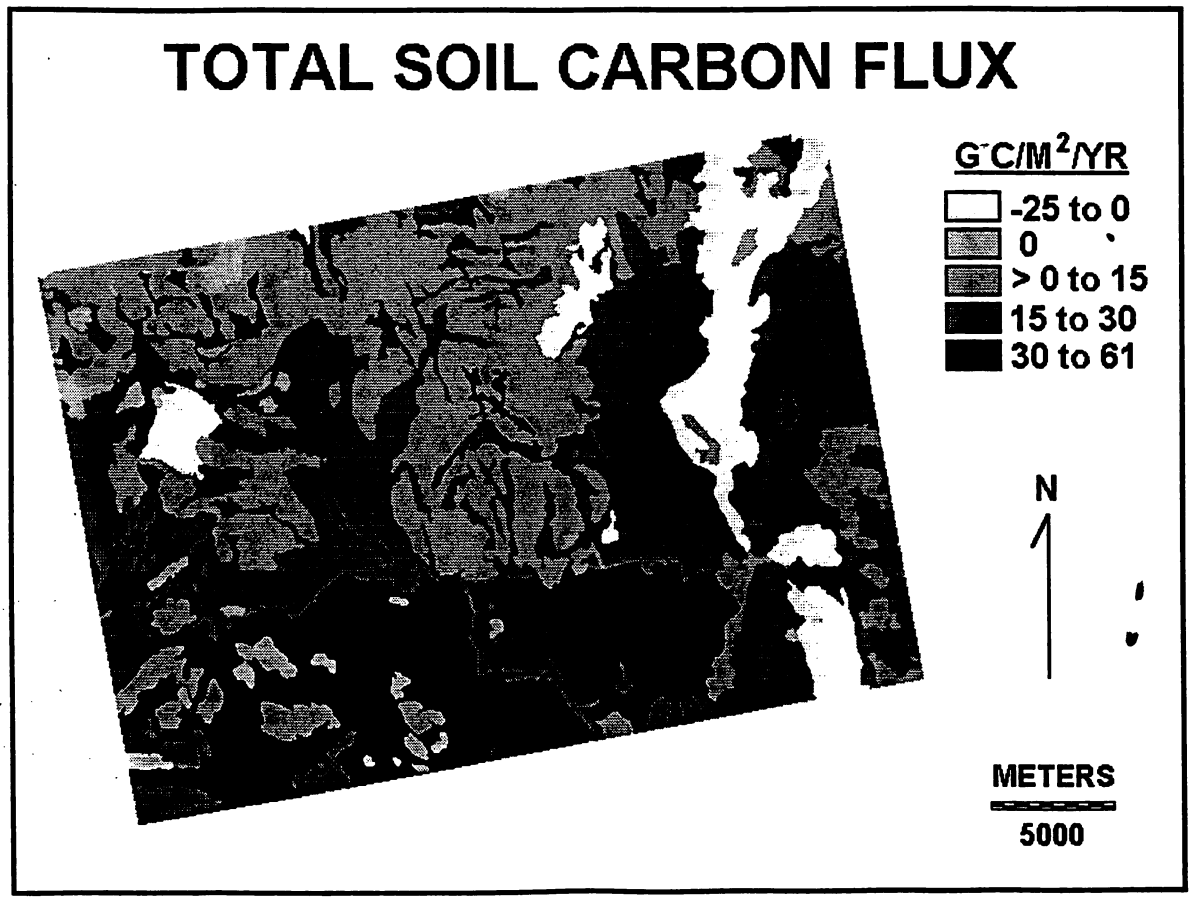

Figure 2. -- *Compiled from satellite images, fire maps, forest inventory, and tree core data, the age ranges represent time since last fire. Figure is from Rapalee et al. (1998), used with permission from the American Geophysical Union.*

The models for surface C flux based on fire frequency and drainage class are derived from the work of Harden *et al.* (1997). The *I* and *k* values are derived from Trumbore and Harden's (1997) radiocarbon field studies.

RESULTS

Our results indicate that soil carbon stocks are linked to drainage, with the largest C stocks occurring in poorly drained sites where humic material and a charred layer derived from fire residues lie at the base of the moss layer (fig 3). Calculations of net C flux are sensitive to the decomposition rate for this large pool of deep soil, which can range from a small sink to a source. Our net flux estimates indicate the entire profile is a small C sink, with the highest annual fluxes in the most recently burned areas and in the oldest, more poorly drained sites (fig 4).

SUMMARY

The study site is a mosaic of drainage classes and stand ages, with no one class comprising more than about one-third of the total area.

Area-weighted spatially referenced estimates of carbon stocks and fluxes account for inclusions and spatial variation within the soil polygons at finer scales than were previously practical. Our results indicate that soil carbon levels are tied to drainage with higher carbon in the more poorly drained sites.

Soil C fluxes depend on both drainage class and time since last fire, with the largest accumulations of soil C in fens and in the surface mosses of recently burned sites. Turnover of deep soil C in these ecosystems and its response to expected warming and altered drainage patterns deserve more attention.

ACKNOWLEDGEMENTS

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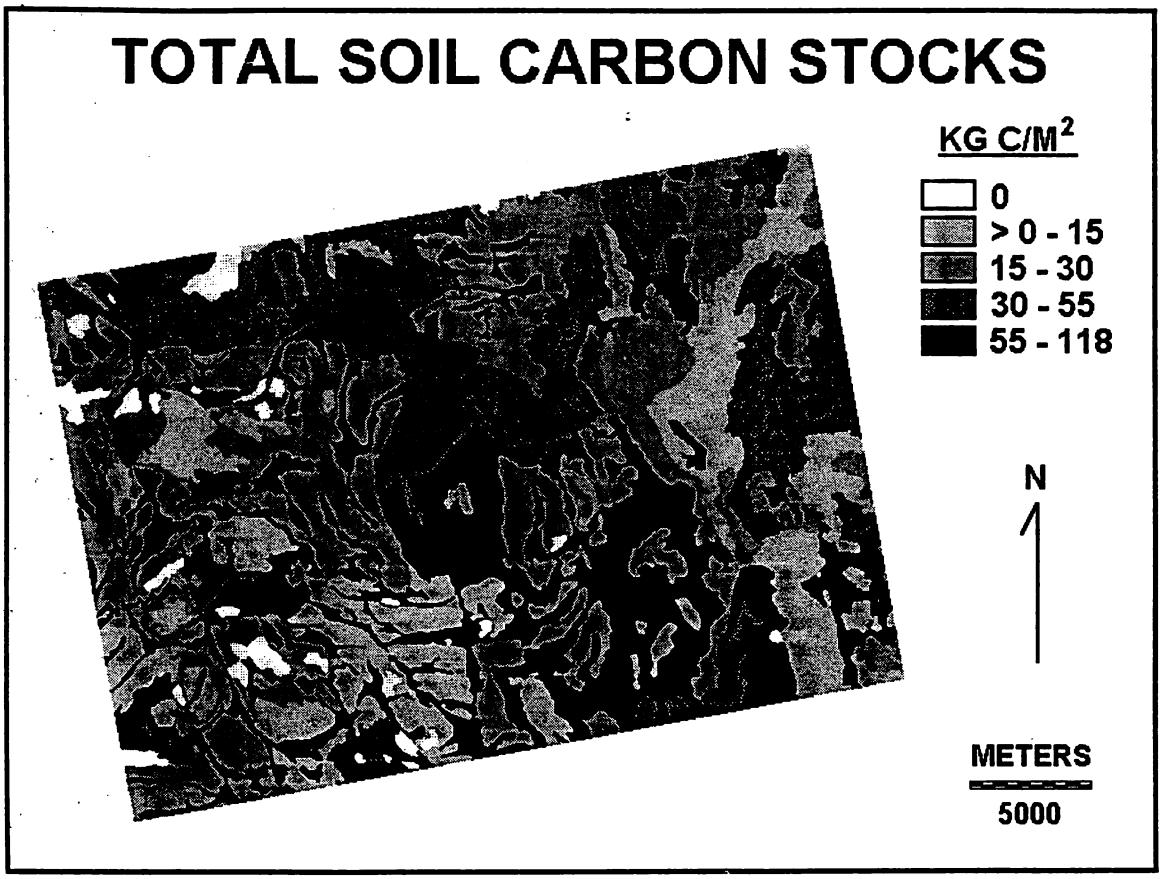

Figure 3. -- *Soil carbon stocks of study area. Values are area-weighted averages (kg C m^{-2}) for entire soil profile. Figure is from Rapalee et al. (1998), used with permission from the American Geophysical Union.*

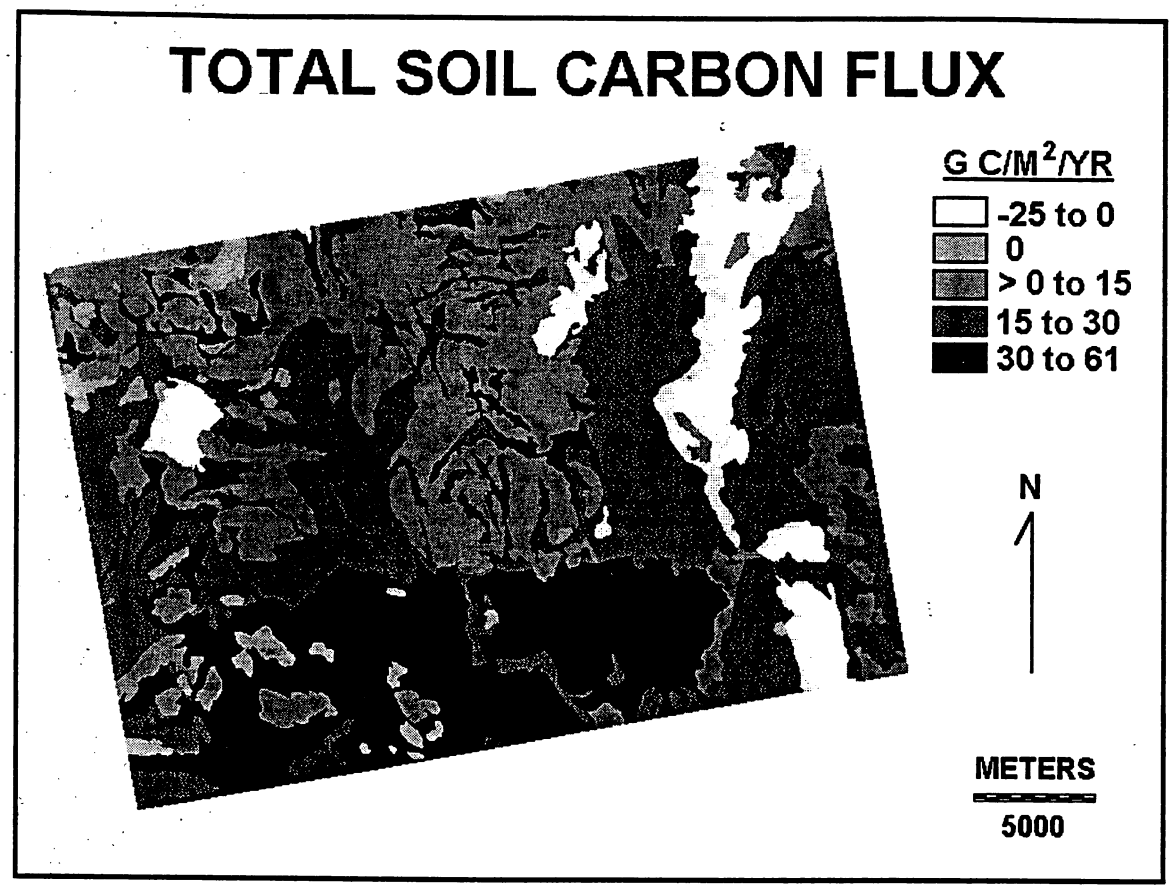

Figure 4. -- *Annual carbon flux of study area. Values are area-weighted averages (g C m^{-2} yr^{-1}) for entire soil profile. Figure is from Rapalee et al. (1998), used with permission from the American Geophysical Union.*

BOREAL FOREST MANAGEMENT

Fire Management and Monitoring

FIRE MANAGEMENT IMPACTS ON BOREAL FOREST PROCESSES AND HEALTH

D.J. McRae and T.J. Lynham[1]

ABSTRACT.--The boreal forest is a fire-dependent ecosystem that has relied on the periodic occurrence of wildfires for maintaining important pioneer plant species. Historically fire has been an important disturbance agent, but modern fire suppression techniques have resulted in a reduction in the number and size of wildfires. Because fire cycles have increased, managers cannot depend on wildfires for initiating and maintaining many forest processes today. Without a renewal of the historical fire cycle, serious changes will result in the forest landscape since life expectancy of individual fire-dependent tree species is much shorter than the current fire cycle. This change will also affect other aspects of the forest, such as wildlife and insect populations. Managers need to understand the consequences of fire exclusion to the overall health of the boreal ecosystem and the possible means of restoring fire as an ecosystem process.

INTRODUCTION

The boreal forest, a fire-dependent ecosystem, has many important pioneer species that rely on the periodic occurrence of wildfires for their survival. Typically, the mechanism for survival is not that an individual tree or even an entire forest stand survives a wildfire, but rather that a process of stand regeneration begins as the result of a lethal wildfire. Tree species such as jack pine (*Pinus banksiana* Lamb.) and black spruce (*Picea mariana* [Mill.] B.S.P.) resist elimination by forest fires through the development of serotinous cones (Cayford and McRae 1983, Viereck 1983) whose plentiful seeds ensure regeneration after fire. These cones protect the seed from being physically consumed by the fire and help maintain seed viability by protecting the seed from the fire's lethal temperatures (Beaufait 1960). However, the heat from the fire is required to melt the bonding material that cements the cone scales closed (Cameron 1953). Only after a second fire, where the interval between fires is shorter than the time needed for young saplings to produce seed-bearing cones, is there a real danger of inadequate natural regeneration. Adequate seed quantities for these two species may not be present until the trees are 25-30 years old. Jack pine stands that have been exposed to short fire return intervals can produce seed cone crops as early as 5 years old (Lynham and Stocks 1991).

Boreal hardwoods, such as trembling aspen (*Populus tremuloides* Michx.) and white birch (*Betula papyrifera* Marsh.), can migrate into a burned area because of light wind-blown seed. In addition, boreal hardwoods can sprout profusely from the base of stems or from root systems that have been killed recently. In other cases, climax tree species such as balsam fir (*Abies balsamea* [L.] Mill.) are delayed from becoming re-established on the site because fire kills the parent trees and consumes the local seed sources. The thin bark of the balsam fir makes it susceptible to be killed even by low-intensity fires. In addition, fire can benefit individual herbaceous and shrub species (Rowe 1983), as well as small mammal and bird species (Fox 1983).

Fire suppression efforts over the last century have curtailed the number and size of wildfires in the boreal forest. This change is a result of increased harvesting pressures and the elimination of wildfires. Both processes affect successional pathways, which in turn can drastically change the vegetation on a site. Artificial regeneration, where planting or direct seeding is applied after some form of site preparation, can often be used to help regenerate some tree species on harvested sites. However, by implementing a total fire exclusion policy, many problems are compounded for the boreal forest ecosystem. Climax species are favored when burning does not occur. Often, this increase in climax species occurs in remote regions that will never be accessed for timber extraction. In other cases, areas have been specifically set aside to protect the fire-dependent pioneer species. Ironically, a policy that does not allow fire to occur may result in a shift to late successional species. A major problem, other than pioneer species regeneration, is that some natural processes require fire to function. Ultimately, the health of the overall ecosystem can be compromised by the elimination of fire. This paper is an exploration of the consequences of eliminating fire from the boreal forest ecosystem based on experiences in Ontario and neighboring parts of Quebec.

[1]*Douglas J. McRae and Timothy J. Lynham, Natural Resources Canada, Canadian Forest Service, Great Lakes Forestry Centre, P.O. Box 490, Sault Ste. Marie, Ontario, Canada. P6A 5M7.*

HISTORICAL AND PRESENT FIRE CYCLES

Presettlement fire cycles in the boreal forest of northern Ontario are poorly understood due to the lack of fire history and palaeoecology data. Jack pine and black spruce are the main conifers that colonize boreal sites following fire. Red pine (*Pinus resinosa* Ait.) and white pine (*Pinus strobus* L.), which are typical of the Great Lakes-St. Lawrence forest, are found at the southern edge of the boreal zone where the Great Lakes-St. Lawrence forest is in transition to the boreal forest. These two tall pines provide evidence of fires as long ago as 400 years due to their long life spans and resistance to decay.

Jack pine (living 100-150 years) and black spruce (living 250-300 years) are far more likely to be killed by fire than either red pine or white pine because jack pine and black spruce have thinner bark and crowns that are closer to the ground. Fire scars, which provide evidence of past fires, can be found on all of these species. Even though jack pine trees are killed easily by fire, some trees usually survive a fire and provide a good source of fire scar material. A fire history study (Lynham and Stocks 1991) of an isolated upland jack pine forest in northern Ontario (53° 50'N, 92° 25'W) showed an average fire return interval of 20 years with individual fires crossing parts of the 4 km by 11 km ridge every 5 to 30 years. Stands that were only 5 years old carried cone crops. This forest has a unique history and geology and is not representative of jack pine stands across the boreal forest.

Despite the lack of published data on fire cycles in the boreal forest, some recent results (Bergeron *et al.* 1997) for 20,000 km^2 of southern boreal forest in the Quebec region of Lake Abitibi, show three fire cycles for three recent time periods. Prior to 1850 the cycle was about 60 years. From 1850-1930 the cycle was about 120 years and from 1930-1997 the cycle was roughly 600 years. Harrington and Donnelly (1978) produced similar estimates of 20th century fire cycles in the boreal forest of Ontario. Their analysis was updated by Alexander (1981) and is updated for the 1980s and 1990s in table 1. These data for northern Ontario show that the mean annual burned area fluctuated by decade; however, when averaged over the period 1920-1996, six of the forest sections (Rowe 1972) have similar fire cycles of about 500-600 years. Two other sections B.4 (Northern Clay) and B.22a (Northern Coniferous) have fire cycles of 2,000 and 233 years, respectively and the fire cycle for the boreal forest region is about 525 years. Fire suppression has also had an impact on the size of forest fires in Ontario. Figure 1 is a graph of the ratio of large (greater than 4 ha) fires to small fires in Ontario from 1920-1996. Although the ratio of large fires was reduced during the era of modern fire suppression (1950-90), the ratio has climbed in recent years. The increase in the ratio of large fires could be partly the effect of higher temperatures due to climate change. Average spring and summer temperatures in the 1980s were 0.5-1.5° C and 0.5° C higher, respectively, in the boreal forest of Ontario than in the previous 30 years (Hengeveld 1995). The increasing proportion of large fires could also reflect changes in fire management policy (Ontario Ministry of Natural Resources 1982) that permit less aggressive attack on wildfires that pose no threat to human life or private property. The 1:1 ratio of large to small fires around 1920 may be the result of a bias in the historical fire database. There is a possibility that not all wildfires were reported as early as 1920. Large fires were more likely to be reported because they would have been more noticeable on the landscape. Conversely, the small fires were more likely to be missed because they may have never been seen or reported. Nonetheless, the data show that the ratio of large fires reached a minimum after 1950 but may be increasing slightly today.

CONSEQUENCES OF FIRE EXCLUSION

The build up of fuel

The elimination of wildfires in a fire-dependent ecosystem affects fuel accumulations. Without fire, fuels tend to build up making high-intensity fires more prevalent (Mutch 1994). In the United States, this has been identified to be a major reason for recent destructive fire seasons. A prescribed fire management program to help in reducing this fuel accumulation has been started to reintroduce low-intensity fires for reducing dangerous fuels (Anon. 1995). Major fuel problems can also exist in the boreal forest when fire is excluded. White spruce and balsam fir are tolerant conifer species that can easily become established in the understory leading to unexpected fire behavior. The conifers act as ladder fuels that can abruptly change a fire from a slow surface fire to a fast spreading crown fire. Boreal hardwoods, considered fire proof except for a short spring window when a surface fire can carry in the dry leaf mat of the forest floor (L-layer of the soil), now are being reported as exhibiting fast-

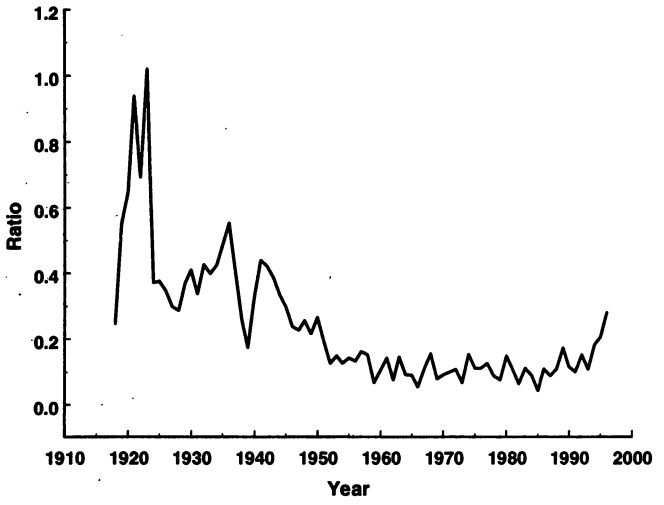

Figure 1.--*Ratio of large (greater than 4 ha.) fires to small fires in Ontario from 1926-1996.*

Table 1.--*Percentage of area burned annually (by decade) and calculated fire cycle for selected geographical regions in northern Ontario based on all fires greater than 200 ha (500 ac) in size that were reported between 1920 and 1996 (adapted from Harrington and Donnelly 1978; Alexander 1981).*

Forest Region and Section (after Rowe 1972)	Area (ha[1])	Percent Mean Annual Burned Area[2]									Fire Cycle[3] (years)
		1920 to 1929	1930 to 1939	1940 to 1949	1950 to 1959	1960 to 1969	1970 to 1979	1980 to 1989	1990 to 1996	1920 to 1996	
Boreal	33,314,588	0.30	0.28	0.14	0.07	0.15	0.20	0.19	0.22	0.19	526
Northern Clay (B.4)	5,393,805	.17	.03	.05	.08	.0	.03	.0	.01	.05	2,000
Missinaibi-Cabonga (B.7)	4,356,171	.44	.16	.49	.19	.02	.02	.05	.08	.18	555
Central Plateau (B.8)	10,069,437	.42	.19	.03	.05	.03	.15	.07	.31	.16	625
Superior (B.9)	2,368,487	.08	.88	.25	.02	.0	.01	.04	.16	.18	555
Nipigon (B.10)	486,031	.17	.02	.27	.69	.0	.0	.0	.34	.19	526
Upper English R. (B.11)	4,123,702	.19	.46	.08	.0	.12	.13	.23	.44	.21	476
Lower English R. (B.14)	1,438,837	.58	.16	.05	.05	.04	.02	.65	.02	.20	500
N. Coniferous (B.22a)[4]	5,078,118	.23	.46	.16	.03	.81	.86	.49	.36	.43	233
Great Lakes-St. Lawrence											
Quetico (L.11)	4,352,403	.51	.60	.03	.01	.01	.14	.71	.12	.27	370

[1]Hectares x 2.471 = acres.

[2]The percentages were computed under the assumption that the portion of the surface covered by water within the boundaries of the burned areas was equal to that in the region at large (Harrington and Donnelly 1978).

[3]The fire cycle is based on the average rate of burning for the period 1920-1996. The reciprocal of the fire cycle is the proportion of the whole region that burned annually (Van Wagner 1978).

[4]The Northern Coniferous refers to the area south of latitude 52°N within the Section.

spreading fires due largely to the increase in the conifer understory. The major reason for this conifer influx can be attributed to harvesting over the last half century that has drastically increased the mixedwood component of the boreal forest in northeastern Ontario to 52% (MacDonald 1996, Spectranalysis 1994). Even when boreal hardwoods are the dominant species on a site, understory species such as balsam fir and white spruce (*Picea glauca* [Moench] Voss) invade with time and become established at high densities. Presently many boreal hardwood stands in Ontario are old enough that we are seeing this shift to a higher fire hazard due to changes in the fuel type. The consequence for fire management is a shift from relatively no concern to realizing that fire can be a major problem. Fire managers must realize that the time when these hardwoods could be considered fireproof is over. Likewise, red pine and white pine ecosystems can accumulate massive amounts of conifer fuel in the understory. The fire regime has changed from one of frequent and benign fires to a modern regime where fire is infrequent. When fires do occur, they have a higher intensity that kills the pines easily. Unfortunately, the latter type of fire has become more prevalent under a fire-exclusion policy. By killing these trees, the fire eliminates the seed trees needed to ensure pine regeneration (M^{c}Rae *et al.* 1994). This example shows that while red pine and white pine are fire dependent, the wrong type of fire can easily eliminate these pines.

Tree Regeneration

Wildfire is an important factor in the renewal of the boreal forest, especially for the shade intolerant pines. While fire is often considered important for seedbed preparation by exposing the mineral soil, this may not be as important as control of the competing vegetation in an unharvested forest ecosystem. The problem is that most of our forest renewal experience stems from a period ranging from 1940 to 1980, where silvicultural research on harvested sites suggested that mineral soil provided the best seedbed (Chrosciewicz 1974). This is no surprise given the harsh conditions experienced on a harvested site. Dry conditions are accentuated by the direct sunlight that creates high surface temperatures. However, fires in natural forests leave standing boles of live and dead trees that remain on the site to moderate the surface temperatures. While complete mineral soil exposure may assist in renewal efforts, it is not required.

Fire reduces the amount and vigor of the vegetation that would compete with the developing pine seedlings. The slower growing pine seedlings require less competition in their first few years, otherwise, they would be quickly overtopped by the older vegetation and they would die. Often red pine and white pine seedlings can be found in an unburned forest. As seedlings, they require low light levels and the seedbed remains moist because it is not exposed to constant, direct sunlight. Seedlings rarely survive to the sapling stage because saplings require higher light levels, which is often unavailable due to the hardwood competition that blocks out the sunlight. Fire exclusion prevents the proper vegetation control that would have occurred historically in the boreal forest. Balsam fir, cited earlier as an added source of fuel, is also a major competitor that is easily eliminated by fire.

Plant and Wildlife Consequences

The adoption of a fire-exclusion policy can lead to the elimination of certain tree species, but many other fire-dependent processes are also affected. Rowe (1983) proposed a very useful concept to deal with fire effects on individual plant species based on their "vital attributes." These attributes help to explain how a plant reacts to fire. According to his scheme, plant species can be classified as invaders, evaders, avoiders, resisters, or endurers. The strategy of invaders is one of copious production of short-lived seeds or spores that are disseminated by wind. Evaders, on the other hand, use the organic or mineral substrate as a seedbank to store their seeds until a disturbance occurs. Their strategy is to evade by rapid post-fire germination when this stored seedbank is exposed usually by a fire that burns the organic material of the forest floor away. Avoiders, such as feathermoss, are species not well adapted to the occurrence of fire and they usually do not appear until the later successional stages. Resisters are plants that are able to withstand fire, while endurers are plants that are able to resprout from stem bases, rhizomes, roots, and other below ground organs, after the passage of fire. Other than the avoiders, most plant species are in fire-dependent categories. Therefore fire exclusion can prevent these species from ever being prevalent in the ecosystem, thereby reducing diversity.

Fox (1983) found an increase in species diversity, species richness, numbers and consuming biomass for bird communities through succession. For small mammal communities, numbers, biomass, and consuming biomass were higher in the herb stage and lower in the shrub and sapling stage than in mature forests.

Sometimes changes are not immediately beneficial but are required for future benefits. The woodland

caribou (*Rangifera tarandus caribou* Gemlin) is an example where fire can reduce the population by half, but long-term benefits allow the population to recover (Cumming 1992). The reindeer lichens (*Cladina* spp.) that caribou feed upon appear to peak in biomass at 60-70 years after a fire and begin to seriously decline after 100 years (Harris 1992). For caribou management, long-term planning of food supplies in the forest ecosystem is required now to ensure that sufficient and suitable age-class structure is present for the future. Under fire-exclusion policies, allowance is not being made for the natural production of younger age classes that will grow to replace the present 60-70 year-old age class in wilderness areas. While allowing fires to burn may not be feasible in some forests, remote areas of the boreal forest should be allowed to burn to ensure the survival of the caribou.

Soil Considerations

Without fire, nutrients from organic materials are cycled slowly by the cool climate of the boreal forest. While fire can reduce the total nutrients on the site, available nutrients can increase (Mroz *et al.* 1980, Viereck 1973). The increase in available nutrients is beneficial because it occurs at a time when new regeneration requires this influx to promote rapid growth. The degree of nutrient change depends upon the amount of organic forest floor material removed by the fire. This material acts as an important nutrient sink, especially on sandy sites that are nutrient poor (Methven 1973). Severe fires that expose the mineral soil will have the greatest impact on nutrients. However, species such as jack pine are adapted for growing under poor soil conditions. On such sites, initial growth may be slow but after several years jack pine can overcome some of the nutrient shortages by establishing extensive root systems that may include a deep tap-root. Jack pine, on sandy sites, will usually out-compete trembling aspen because the low moisture and low nutrient levels cause inferior growth in the aspen (Stoeckeler 1948). Rocky sites also limit root develop in aspen and give jack pine an advantage following fire.

The major problem in the boreal forest, where there has been no recent fire, is that the organic layer accumulates and locks away nutrients needed for the ecosystem to respond properly. In northern parts of the boreal forest, this layer effectively insulates the soil and prevents optimum growing temperatures for the roots (Ballard and Hawkes 1989). The paradox is that the organic forest floor must be burned off completely, exposing the mineral soil, to obtain optimum soil temperatures at the expense of some nutrients that are volatilized. While managers fret over the loss of nutrient, perhaps soil temperature during the growing season is the more important factor affecting site productivity.

Insect and Disease Relationships

Wildfire can be a controlling factor for the elimination of food supplies or sheltering areas necessary for the survival of some insects. In the absence of fire, increases occur in balsam fir, which is the primary food source for the spruce budworm (*Choristoneura fumiferana* [Clem.]). Spruce budworm outbreaks that have normally occurred on an average 40-year cycle, may occur on more frequent cycles because the food source for the insect is never eliminated as it would be by fire. After a budworm outbreak without fire, balsam fir becomes quickly reestablished as the seed is not destroyed. Large fires prior to modern suppression would have eliminated the seed source for balsam fir over vast areas providing an effective block to the budworm outbreaks that now sweep back and forth across Ontario. Blais (1983) blames the increase in budworm to harvesting practices that remove the pine and spruce but leave the balsam fir.

The accumulation of certain organic material on the forest floor can provide ideal shelter during the winter for insects such as the red pine cone beetle (*Conophthorus resinosae* Hopk.) and the white pine cone beetle (*C. coniperda* [Schw.]). These two insects cause developing pine cones to abort and fall to the ground early in the final year of cone development. The larvae pupate in fallen cones and as adults they overwinter in the cones or in the branch shoots. Severe cone beetle attacks can destroy entire cone crops (Hedlin *et al.* 1981) which results in a lack of adequate seed for natural regeneration. Frequent wildfires were probably an important controlling factor in the past for these insects. Today, prescribed fire has been used to control cone beetles by removing or altering the organic matter of the forest floor (Miller 1978, Wade *et al.* 1989).

The loss of red pine stands has affected the feeding habits of the red pine cone beetle. Within the range of red pine, jack pine is also attacked by a cone beetle (de Groot 1991). The cone beetle found feeding on jack pine was thought to be a separate species but taxonomic evidence has shown that it is *C. resinosae* that feeds on jack pine. The author suggested that the attacks on jack pine were an emergency response by the cone beetles, resulting from the loss of red pine due to logging. Although de Groot (1991) pointed out that the red pine cone beetle is not fully adapted to jack pine, possibly the loss of red pine due to

longer fire return intervals could also result in an increase in red pine cone beetle attacks on jack pine.

Benign fires, which do not damage the crowns of young white pine trees, help maintain the medium sunlight levels that are required to prevent serious attacks by the white pine weevil (*Pissodes strobi* Peck). This advantage is created because medium light levels will induce the development of small diameter shoot leaders that are too narrow to accommodate the adult weevil when the weevil is ready to lay eggs (Logan, 1959, Sullivan 1961). Under shaded conditions, the micro-environment is also cooler, which retards weevil development (Katovich 1992). The ability of white pine to grow under a partial canopy also helps to protect the trees from white pine blister rust (*Cronartium ribicola* J.C. Fischer), since dew formation is retarded under the closed canopy, reducing the chances of rust spores from germinating (Stearns 1992).

CONCLUSIONS

Wildfires are an important component of the boreal forest. Without fire, major changes on a landscape level will occur. The pre-settlement fire cycle in the boreal forest was about 120 years whereas the fire frequency in the mid-to-late 20th century is about 500-600 years. There is a sharp contrast between the presettlement fire cycles and the fire cycles of the last century. Most of the difference between presettlement and current fire cycles can be attributed to the effect of modern fire suppression over the past 100 years. The consequences of these changes are not only reflected by the tree composition that we often think of when fire exclusion occurs, but it has implications for every living thing surviving in that landscape. The good news is that in many regions of the boreal forest, the fire cycle can be reestablished. Pioneer species regeneration is still possible on many boreal sites by allowing fire to burn again, since many species have not yet totally disappeared. Older pine sites can be burned to provide younger age classes, while many seeds of lesser vegetation lay buried, waiting for the next fire to expose them. Introducing fire operationally will be a huge challenge to resource managers. However, it will not be as big an obstacle as the ecological challenges that lay ahead if fire is not allowed to burn.

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USING SHORTWAVE AND THERMAL INFRARED SATELLITE IMAGERY TO STUDY THE EFFECTS OF FIRE IN AN ALASKAN BOREAL FOREST

Nancy H.F. French, Eric S. Kasischke, Jeffrey L. Michalek, and James P. Mudd[1]

ABSTRACT.--The importance of measuring and monitoring fire related ecosystem characteristics in boreal forests is recognized and is the focus of several new studies. In this paper we present a discussion on the use of remote sensing for the study of boreal forest fire scar characteristics. Remote measurements of geophysical parameters and model inputs are discussed. It is recognized that remote measurements need to be combined with field measurements and modeling. Specific examples on recent work using infrared remote sensing for fire scar studies are presented and discussed in detail. Results of this work have shown that shortwave infrared systems can be used to map fire severity. Thermal infrared systems can be used to monitor surface temperature changes, to map thermal characteristics of the burn scar surface, and can potentially be used in the modeling of surface energy fluxes. Generally, remote sensing can provide unique information at a variety of scales for the study of fire ecology.

INTRODUCTION

Recent recognition of the boreal forest region as one of the most important and least understood reservoirs of global carbon (Chapin and Matthews 1993) has led to an increased interest in the study of boreal forest ecology, particularly at regional and global scales (Sellers *et al.* 1991, Goldammer and Furyaev 1996b). An extremely important factor controlling ecosystem processes in the boreal region is fire (Van Cleve *et al.* 1986, Payette 1992). On average, over 5 million ha of land burn each year in the boreal region (Stocks 1991). Circumpolar studies of boreal forest fires and fire effects have increased due to attention from the world community through the International Geosphere-Biosphere Program (IGBP) and the International Boreal Forest Research Association (IBFRA) (Goldammer and Furyaev 1996a).

Remote sensing has been found to be an important tool for detection and mapping of fires and fire scars in the boreal region (Cahoon Jr. *et al.* 1994, Kasischke and French 1995, Bourgeau-Chavez *et al.* 1997). Two main approaches have used thermal infrared imagery to locate active fires and multi-spectral imagery to map the location and areal extent of fires after they have occurred. The sensor most often used for these applications is the Advanced Very High Resolution Radiometer (AVHRR) because of its frequent and broad area coverage. Recently, research has focused on using satellite imagery from a variety of sensors to monitor additional surface characteristics at areas affected by fire (Kasischke *et al.* 1994, French *et al.* 1996a, French *et al.* 1996b, Michalek *et al.* 1998). The purpose of this paper is to discuss the use of remote sensing for the study of boreal forest fire ecology, and more specifically, to present some of our recent work related to the use of shortwave and thermal infrared satellite imagery. The paper is divided into three sections. First, is a background discussion on the utility and limitations of using remote sensing data for measuring important ecological factors related to fire and estimating model inputs. Second, we will present results of a study on the use of shortwave infrared (SWIR) imagery for mapping fire severity and estimating carbon release. Third, a discussion, with some preliminary results from our work, on the utility of thermal infrared (IR) systems for measuring surface temperature and estimating surface energy flux will be presented.

BACKGROUND

Effects of Fire on Ecosystem Function

The effect of fire on boreal ecosystems is dramatic (Rowe and Scotter 1973, Viereck 1983). Fire impacts both immediate and long-term functioning of the ecosystem. Due to very slow recovery rates in the cold climate of the boreal region, fire effects can last for many decades. The obvious impacts include, the removal of vegetation, consumption of soil carbon, and changes in surface temperature and albedo. The summation of these effects results in changes in basic ecosystem function, including carbon cycling and energy flux.

[1] *Earth Sciences Group, ERIM International, P.O. Box 134008, Ann Arbor, MI 48113-4008*

Direct changes from fire disturbance lead to changes in carbon and energy exchange. These represent both positive and negative effects. For example, fire will affect carbon cycling by releasing large amounts of CO_2 during and after the fire, and it will also increase carbon fixation rates due to increased net primary production (NPP). The two opposing forces typically result in a net loss of carbon over a 50 to 100 year time frame, and would result in no net change over a longer time frame as the ecosystem recovers to its pre-burn state. If global warming predictions are accurate, however, fire may become more pervasive, leading to a decrease in carbon storage in the boreal forest (Flannigan and VanWagner 1991, Wotton and Flannigan 1993, Kasischke *et al.* 1995a). Because of the uncertainty of the effects of climate change on ecosystem function, it is important that improved estimates of carbon flux and energy exchange be made over as much of the boreal landscape as possible.

The Role of Remote Sensing in Ecosystem Studies

Remote sensing data can be important in ecological studies in several ways. First, remote sensing systems can directly or indirectly measure site condition, such as biomass density or surface reflectance. Second, remote sensing data can be used to help estimate model inputs to answer fundamental questions on ecosystem function (Sellers *et al.* 1995). When using remote measurements it is important to keep in mind the scale of the phenomenon of interest. Coarse-scale systems, like AVHRR, may be useful for estimates of broad-scale phenomenon, such as the overall impact of fire on a region or continent. For determination of fine-scale fire effects, or to look at phenomenon that are typically directly measured in the field, it may be important to use fine scale imagery, from systems such as Landsat Thematic Mapper (TM) or aerial cameras. Remote sensing can be used as a unique tool for scaling locally-derived measurements to landscape, regional, and global scales. This application is of increased interest because of recent attempts to extrapolate from field-derived measurements to global scales. Remote sensing can be used to evaluate the validity of these extrapolations, and to more intelligently apply local measurements to broader scale estimates.

Over the last several years we have been conducting a series of studies on the effects of fire on carbon cycling in the boreal region of Alaska (Kasischke *et al.* 1994, French *et al.* 1995, Harrell *et al.* 1995, Kasischke and French 1995, Kasischke *et al.* 1995a, French *et al.* 1996a, French *et al.* 1996b, Bourgeau-Chavez *et al.* 1997, Michalek *et al.* 1998). One of the most important aspects of our work is to evaluate how information derived from remote sensing systems can contribute to the overall understanding of forest fire ecology and fire-affected ecosystem processes. In our work to date we have concluded that a combination of data from several sensors is needed to adequately address the questions we are asking. In conjunction with the remote sensing data, ground-based measurements are being made for these studies. This is done for two basic reasons; first, to help validate our remote sensing results, and more importantly, to provide inputs to models for estimation of greenhouse gas emission and energy flux that cannot be derived from remote measurements. It is the combination of field measurements, remote sensing-derived measurements, and modeling which will allow an understanding of the role of fire in ecosystem function.

We have used a variety of remote sensing systems in our studies to date. With this work, we have found that synthetic aperture radar (SAR) from the ERS systems has potential for mapping soil moisture condition (French *et al.* 1996). ERS SAR may also be helpful for determination of burn severity. As mentioned above, AVHRR can be used for broad-scale mapping of fire scars, as can ERS SAR (Bourgeau-Chavez *et al.* 1997). Landsat TM can be used for fine-scale mapping of surface characteristics due to its relatively high spatial resolution (30 m) (Michalek *et al.* 1998). The upcoming MODIS sensor will have low resolution, broad scale applications similar to AVHRR, except that it will have a much larger range of spectral channels, so it will provide data for improved temporal mapping of spectrally interesting phenomena.

To help illustrate the use of remote sensing for studying fire disturbed sites we present some recent work on the use of infrared imagery. Much of the work currently underway is taking place at a site which burned in 1994. The site is near the Gerstle River along the Alaska Highway (63°52'N, 145°13'W) in an area adjacent to the Fort Greeley Military Reservation.

SHORTWAVE IR FOR SURFACE CHARACTERIZATION

While shortwave IR data is available from several multi-spectral systems, its utility for vegetation studies has only recently become apparent to the ecological

community. This is likely a result of the fact that many multi-spectral systems lack SWIR bands (1.3 to 3.0μm). The Landsat TM system operates two channels in the SWIR, bands 5 and 7. The Along-Track Scanning Radiometer (ATSR), a relatively new sensor on-board the European Space Agency's ERS satellites, operates a SWIR channel, and the upcoming MODIS instrument will also have several SWIR channels. (For a more complete description of these systems and their specifications the reader is referred to the following world-wide-web sites: For TM: http://edcwww.cr.usgs.gov/Webglis; for ATSR: http://atsrw3.ag.rl.ac.uk; for MODIS: http://modarch.gsfc.nasa.gov/MODIS/)

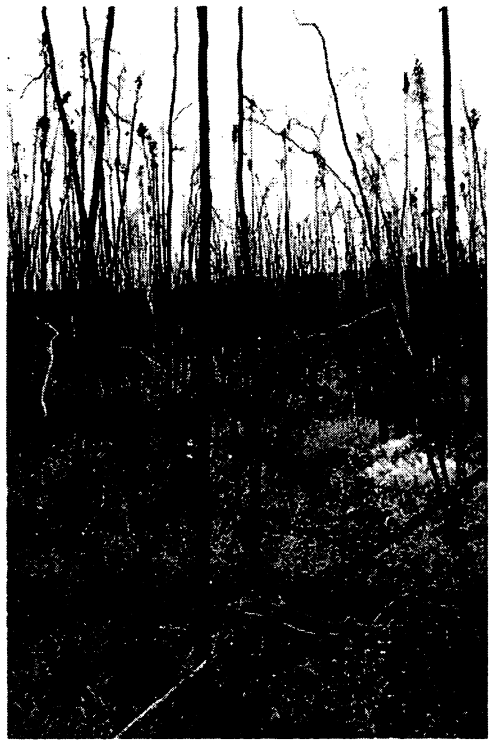
The use of coarse resolution multi-spectral satellite imagery (such as that collected by AVHRR, ATSR, and to be collected by MODIS) for mapping the areal extent of fires in boreal forests is well established (Cahoon *et al.* 1994, Kasischke and French 1995). Since fires result in significant damage to or removal of the tree crown, fire scars are detectable on multi-spectral visible/IR imagery for several years and can easily be used to map the location and areal extent of the large fires occurring in the boreal forest. Epp and Lanoville (1996) have used Landsat TM imagery collected over the boreal forest region of the Northwest Territories (Canada) to map older fire scars (up to 50 years old) using the SWIR and near IR channels.

Besides fire boundary mapping, the fine-scale resolution of the TM sensor allows us to study the variability of burn conditions within the scar. At the Gerstle River fire site we have used TM imagery for within-burn characterization of surface condition (Michalek et al. 1998). Variability in the fire scar signature was found to be related to burn severity. Using pre-burn imagery and field measurements we were able to estimate pre-burn carbon density. This information used in conjunction with the TM-derived burn severity allowed us to estimate carbon release from the burn at a relatively fine scale (table 1). The estimate found using the information derived from TM gave a 52% higher amount of carbon released from this burn than estimates based on a single value for burn severity. The results show that the improved spatial information available using fine-resolution remote measurements of burn severity can improve estimates of carbon release during burning.

Table 1.--*Carbon efflux estimates during the 1994 Gerstle River fire based on pre-burn spruce density and fire severity mapped using Landsat TM imagery (from Michalek* et al. *1998).*

Carbon efflux category number	Pre-burn black spruce density class	burn severity class	Area (ha)	Carbon efflux/ area (T/ha)	Total carbon efflux (T)
1	Sparse	Light	561	27.98	15,697
2	Sparse	Moderate	17	68.88	1,171
3	Sparse	Severe	182	107.82	19,623
(Subtotal)			(760)		(36,491)
4	Medium	Light	1,397	20.96	29,281
5	Medium	Moderate	113	51.38	5,806
6	Medium	Severe	561	79.98	44,869
(Subtotal)			(2,071)		(79,956)
7	Dense	Light	919	15.87	14,585
8	Dense	Moderate	157	38.15	5,990
9	Dense	Severe	731	57.84	42,281
(Subtotal)			(1,807)		(62,856)
Total			4,638		179,303

(a)



(b)

Figure 1.--*Photographs taken at the Gerstle River burn site 2 years following the burn showing (a) an area of severe burn with exposed mineral soil and fallen dead trees, and (b) an area of less severe burn with remaining organic soil and standing dead tree.*

Derivation of burn severity was possible because areas where nearly all organic soil was consumed showed different spectral characteristics than areas where ground layer organic material remained following the burn. The signature differences are due to two factors; the presence or absence of exposed mineral soil, and the presence or absence of standing dead trees, which cause shadows (fig. 1). Areas of severe burn showed a large amount of exposed mineral soil and burned trees had fallen, due to consumption of roots. Areas of less severe burn had little exposed mineral soil, standing dead trees, and shadows from the standing dead.

We found that discrimination of the burn site from the unburned surroundings is most distinct in the near IR (band 4) and particularly the SWIR channels. Also, most of the signal variation within the burned area occurred in these channels. Although burn scar mapping and some within-scar characterization of burn condition is possible with sensors lacking a SWIR channel, more information can be deduced from systems which operate in the SWIR.

THERMAL IR FOR SITE TEMPERATURE MONITORING

Thermal IR satellite sensors in current operation include AVHRR, Landsat TM, and ATSR. Thermal IR, also called long-wave IR (LWIR), operates in the atmospheric window between 8 and 14μm. Energy at these wavelengths is emitted rather than reflected from the Earth.

The energy recorded at the long wavelengths is related to the temperature of the Earth's surface. Since many ecosystem processes are temperature dependent, there is interest in utilizing thermal IR systems for ecosystem studies. Nutrient dynamics, and in particular soil carbon efflux, is temperature dependent. Warmer site conditions will typically cause higher rates of soil respiration, as long as other factors are not limiting (Schlentner and Van Cleve 1985). Also of interest is the effect of fire disturbance on the energy balance at a site (Hall *et al.* 1995). Remote measurements of thermal conditions have potential use in modeling energy flux and in

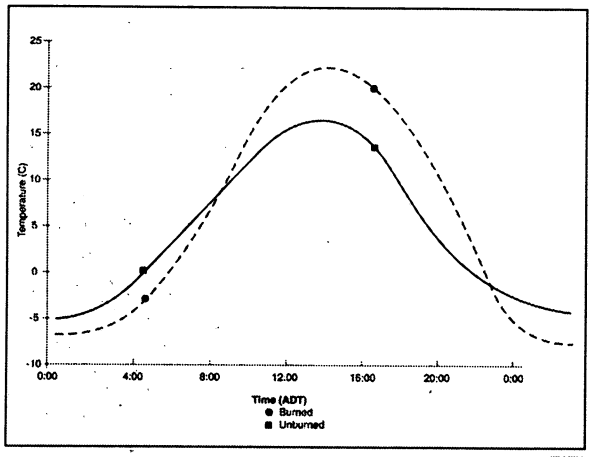


Figure 2. -- *Plots demonstrating the diurnal variation in temperature. Point data is derived from AVHRR thermal imagery at a burned and unburned location. Note the higher temperature variation in the burn area.*

determining the magnitude of fire-effects on the energy budget.

It is well know from field observations that the ground surface temperature increases following fire (Brown 1983). Initial analysis of data from the thermal IR channel of AVHRR has shown increased apparent temperature[2] of as much as 12° C one year following a burn in interior Alaska (French *et al.* 1996b). Further analysis is underway using a better AVHRR data set. The AVHRR sensor collects data over the entire earth twice daily. Therefore, with AVHRR it is possible to gain an understanding of temporal dynamics in site temperature, both daily and seasonally.

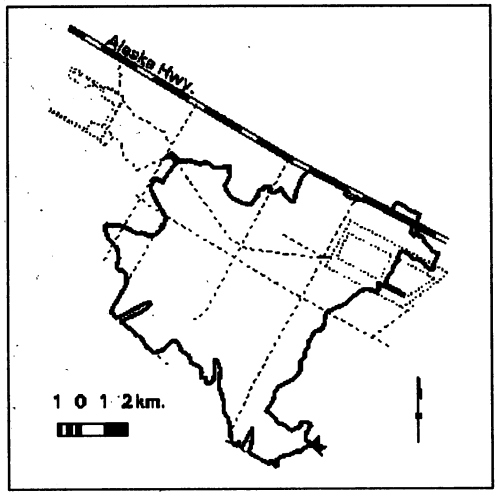
An initial look at a day/night AVHRR image pair from the Gerstle River fire shows that the diurnal variation in thermal signature is greater in the burned area as compared to an unburned adjacent area (fig. 2). Using the day/night image pair it is possible to estimate the diurnal temperature change in a burned and unburned location. For this analysis, the imagery as corrected for atmospheric effects, and radiant temperature estimates were made using the split-window technique (Francois and Ottle 1996). By analyzing the diurnal change in temperature, we may be able to learn more about the thermal characteristics of the surface material. Seasonal temperature changes can also be monitored using the AVHRR's frequent data collection. An increase in seasonal temperature or the diurnal variation in temperature could be significant for soil microbial activity. Warmer soil conditions will allow more microbial activity, thereby increasing biological production of gaseous carbon. The plan for this project includes further analysis of AVHRR imagery to develop methods for modeling temporal changes in surface temperature. A complimentary field experiment is also underway to look at the response of soil microbial respiration to temperature and fire disturbance (O'Neill *et al.* 1997).

The spatial resolution of the AVHRR (1.1 km) limits its utility for looking at fine-scale variability within a burn. Low altitude aerial photos collected one year following the Gerstle River burn show a high amount of spatial heterogeneity due to burning pattern. Spatial heterogeneity can also be a result of pre-burn vegetation pattern. Therefore, it is desirable to look at these sites with higher resolution systems if possible. The TM thermal IR channel (10.5-12.5 μm wavelength) has a resolution of 120 m, which allows for spatial discrimination not possible with the AVHRR sensor.

An initial analysis of TM imagery from the Gerstle River fire one year following the burn showed that the apparent temperature within the fire scar ranged from 12° to 19°C (fig. 3). The burn area has a higher apparent temperature (~2-10°C) than the surrounding unburned area, which includes both vegetated (forest and grass field) and non-vegetated (bare agricultural fields) areas. This indicates that TM data can also be used to delineate burned areas, as demonstrated using AVHRR data but on a much finer scale. Although the

[2]*Apparent temperature is also known as brightness or radiant temperature. It is the actual (kinetic) temperature of a black body which would produce the radiance equivalent to the measured radiant flux. Remote sensing systems measure radiant (apparent) temperature, while thermometers measure kinetic (actual) temperature. In general, radiant temperature is less than kinetic temperature.*

(a)



(b)

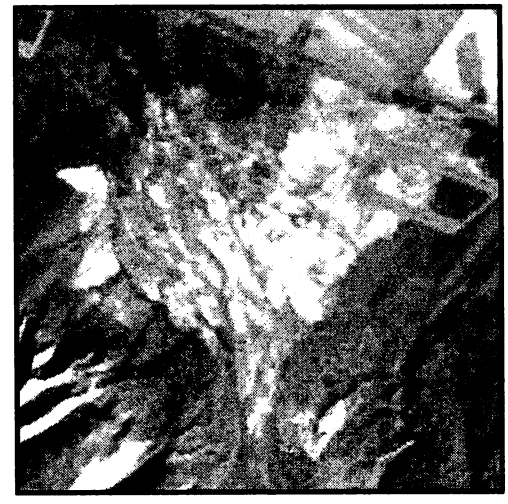


Figure 3. -- *(a) Map showing the outer boundary of the Gerstle River fire scar (solid black line), and (b) the Landsat TM band 6 (thermal) image of the Gerstle River burn collected on 16 September 1995. The area burned is brighter than the surrounding unburned area demonstrating the higher surface temperature (2-10°C) within the fire scar.*

finer resolution of TM is helpful for spatial analysis,image data are readily available due to the satellite's infrequent orbit return interval and cloud cover. For example, for the years between 1991 and 1995 only two usable images were available over the Alaska study sites. The TM imagery, therefore, cannot be used to monitor thermal characteristics over time. The utility of the TM sensor, on the other hand, lies in its ability to more finely define within-site variation at a point in time. A likely approach for modeling temperature variability is to use fine-scale information from TM with coarse-scale AVHRR or MODIS-derived temporal information. Once these systems are fully operational (Landsat 7 and MODIS will launch in summer 1998) this approach will be feasible.

The preliminary analyses we have conducted on AVHRR and TM imagery to date have allowed us to recognize that fire scars exhibit thermal anomalies which are related to temperature conditions at the site. Further work may lead to a better understanding of the thermal properties of the surface. However, many problems exist in using satellite-based thermal sensors for determination of land surface temperature. Some of these problems are correctable, and algorithms exist for deriving temperature, but inherent factors need to be considered when looking at satellite-derived temperature data (Prata 1993). Some of the important considerations are radiometric calibration, correction for atmospheric effects (haze), and determination of surface characteristics, particularly surface emissivity.

Because fire so drastically affects site conditions, it is obvious that the energy flux from disturbed sites is also affected. Although this may seem obvious, it is difficult to adequately quantify energy flux changes. Initial ground-based measurements for energy and water flux modeling are underway. For a complete understanding of the effect of fire on the energy flux changes of an entire fire scar, remote measurements will be needed since ground-based measurements would be difficult to implement at an adequate spatial scale.

The utility of remote sensing, in particular thermal IR sensors, for energy flux modeling is recognized, however, important limitations for using current data is also apparent (Sellers *et al.* 1995). Several

researchers have developed ways of using thermal IR measurements (in conjunction with other measurements, both remote and ground-based) using this data for energy flux modeling (Goward *et al.* 1985, Hall *et al.* 1991, Hall *et al.* 1992, Vining and Blad 1992). Although these results are not considered to be completely accurate due to data limitations, approaches and models are being implemented. This is encouraging for an eventual operational ability to use remote measurements for energy flux estimation.

CONCLUSIONS AND FUTURE RESEARCH

As described above, a variety of tasks are underway to establish the utility of thermal IR data for mapping and monitoring surface temperatures and modeling surface energy flux in fire-disturbed sites. Our work is preliminary, and, due to the state of knowledge and availability of data, may take years to adequately develop. Our short-term goal, however, is to investigate the need for remote sensing data and define the utility of that data for temperature and energy flux estimation.

Work in the shortwave IR realm is better developed and closer to the goal of using remote measurements for broad scale estimates of fire scar characteristics. We have used Landsat TM data, with the SWIR channel in particular, to map fire severity. Further analysis of these data and new data sets will allow us to expand these methods to feed models for estimation of carbon flux from fire-disturbed sites.

Generally, there is a need for improved estimates of model inputs for determination of the role of fire in carbon cycling and effects on energy exchange. This is true especially in the boreal region, since fire is so prevalent and climate change is predicted to be so influential. Additionally, site-specific results made at fire-disturbed sites need to be scaled up, so that results from ecosystem-level studies can become useful at landscape, regional, and even global scales. By including remote sensing as a tool at several stages, better estimates of carbon release, carbon fixation, and energy flux can be made.

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ON IMPROVEMENT OF WILDLAND FIRE MANAGEMENT IN RUSSIA ON THE BASE OF VEGETATION FUEL MAPS

A.V. Volokitina and M.A. Sofronov[1]

ABSTRACT.--The subject matter of this paper is a new area in the field of pyrology (wildland fire science) in Russia, the mapping of vegetation fuels (VF). The possibility of using the VF-maps for improvement of wildland fire management is presented. A new method of surface fire prediction on the base of VF-maps is described. In the classification of vegetation fuels, some components of forest ecosystems (biogeocenoses) are used as groups and types of VF; their role and pyrological properties are taken into account in the spread and development of fire. The necessity of improving wildland fire management, especially in cases of critical fire situations, is indisputable. What is necessary for efficiently dealing with a wildfire? First of all, the behavior of the fire must be predicted. Doing this work needs a large-scale map of vegetation fuel.

VEGETATION FUEL MAPS

Spatial mapping of vegetation fuels is a prohibitively costly job. For this reason we proposed to create such maps on the basis of the largest-scale forest maps produced in the process of forest inventory, by way of identification of the fire properties of the inventory plots or units. Methods for drafting such maps based on forest inventory data have been developed in the Institute of Forest, Siberian Branch of the Russian Academy of Sciences (Volokitina, Sofronov, Klimushin 1996; Volokitina 1996). The basis for this work was a system of vegetation fuel (VF) classification (Volokitina, Sofronov 1996). A key group of VF plays the leading role in fire incidence and spread. We named this group of VF "Prime Conductors of Burning" (PCB - fine fuel) and worked out its classification (table 1).

A typical forest inventory map is not informative enough for fire management uses, because forest cruisers concentrate on the inventory of the timber. So we proposed to improve the usefulness of forest inventory documents for fire management purposes by developing identifiers of the types of the prime conductors of burning (PCB), in order to have PCB's identified and recorded in the inventory documents. These descriptors must be regional. The development of PCB identifiers should include descriptions of vegetation communities as typical fuel complexes, similar to the Canadian fuel types (Forestry Canada, Fire Danger Group 1992). We developed such identifiers for the Krasnoyarsk Region, which comprises 12 typical fuel complexes (TFC) (Volokitina 1990). These complexes are shown in table 2.

The TFC characteristics given here could be very useful for identifying TFC's on aerial photos. This is one way to make development of fuel maps less costly. Another way to reduce the cost of producing large-scale fuel maps is to create a database that can be used to draw operational fuel maps for areas of active fires, rather than fuel mapping of the entire territory (Volokitina, Klimushin, Sofronov 1995).

FIRE SPREAD PREDICTION

To forecast the behavior of a fire, it is necessary to know not only the spatial pattern of fuel complexes but also the fuel conditions (e.g., ignitability and the character of burning), that depend on drought severity. Awareness of potential changes in these conditions with time is also necessary, so a detailed weather forecast must be available. Another important, technically difficult task is correct display of the fire front on the fuel map. This could be done, for example, by thermal scanning from an aircraft.

We have developed a simple method of fire spread prediction, which can be applied "by hand", without the use of a computer. Forecasting of fire spread is based on calculation of positions of the fire front at particular points in time. This is done by calculating the spread of the fire front along several radial lines in different directions. We plan to use the method of the Canadian system (FBI) for prediction of fire development, in particular, for transition between surface fire and crown fire. With this approach, critical courses of fire spread can be revealed, and a calculation of resources necessary to control the fire can be made.

[1] *V.N.Sukachev Institute of Forest, Siberian Branch, Russian Academy of Sciences, 660036, Krasnoyarsk Fax:(3912) 43 36 86; E-mail: ifor@krsk.infotel.ru.*

Table 1.--*The types of prime conductors of burning (PCB).*

PCB type, code its	Typical areas, their attributes	Non-typical areas, their attributes	Critical drought class(CDC)
1	2	3	4
	Mossy PCB Subgroup		
Lichen (Lc)	1. Lichens predominate in the forest floor 2. Lichens are present in the forest floor on dry soil	Very dry, including rock outcrops, with litter of pine needles	I
Dry moss (Dm)	Forest floor is predominated by green mosses, somewhere with lichens, on drained soil in boreal and nothern forests (green-moss and cranberry - green-moss forest types)		II
Moist moss (Mm)	Forest floor is predominated by green mosses, on insufficiently drained soil (forest types - mossy, aulacomnium, moist billberry and similar)	In southern taiga on drained soil, the forest floor may be predominated by green mosses. Thin cover (up to 3 cm) of compact moss	III
Bog-moss (Bm)	Ground cover is predominated by Sphagnum and Hypnum moss species, on boggy and bog soils (without notable presence of sedge or grass)		
Subtype Bm1	Boggy forests and small bogs amidst drained plains, with peat layer up to 0.7 m thick	Ground cover is predominated by Polytrichum	IV
Subtype Bm2	Large bogs and bog systems		Not burnable
	Litter PCB Subgroup		
Cured grass (Cg)	Forest floor is predominated by dry grass or sedge (i.e. naturally cured plants), usually in fall and spring. Not included are sedge forest types with ground cover of evergreen sedge	Bogs and wamps of sedge-sphagnum and sedge-hypnum types, with well-developed cover of sedge in spring and fall (in summer - Bm)	I
Loose litter (Ll)	1. Forest floor is predominated by herbs - in spring and fall. 2. Forest floor is predominated by litter of birch and aspen fallen leaves 3. Forest floor is predominated by litter of pine and cedar fallen needles	1.Forest floor is predominated by evergreen sedge - in spring and fall 2.Matted litter of dry sedge or grass	II
Compact litter (Cl)	1. Forest floor is predominated by compact litter of fallen needles of fir, spruce or larch - in all seasons 2. Forest floor is predominated by compact litter of fallen leaves of birch or aspen, and by matted dry herbs		III
Non-conductor (Nc)	Surface fuel cover too scarce for fire to spread		
Subtype Nc1	Fuels not providing spread of surface fire: duff, humus, turf; usually in summer. Ground fires are possible.	Live grass load exceeds PCB load in summer, so spread of surface fire is impossible	IV
Subtype Nc2	Absence or very scarce presence of any PCB; sands, pebble, rock outcrops, plowed fields, roads, etc.		Not burnable

Comments: Critical drought class (CDC) means such lowest class of drought which provides the burning ability of a particular PCB in typical environment conditions (gentle topography, forest stand of mean stock index (0.5-0.7) with live leafing). Drought class is rated according to the fire drought index (or the Nesterov index, or the LenNIILH PV-1 index), points: I DC - up to 300 p., II DC - 301-1000 p., III DC - 1001-3000 p., IV D' - 3001-10,000 p., V DC - over 10,000 points of the fire drought index.

Table 2.--*Pyrological categories of areas and their characteristics in the conditions of Krasnoyarsk Priangari (fragment).*

Pyrological categories of areas (forest stand and ground cover)	**Topography**	**Overstory species**	**Soil moisture**	**Forest floor, live and dead fuel**	**PCB types, during summer and spring (fall)**
1	2	3	4	5	6
Dry pine stands with ground cover of lichens	elevated	pine, somewhere pine + birch	insufficient or dry	cover of lichens (surface coverage 0.6 -0.8)	Lc-Lc
Pyne stands, usually without admix of fir, spruce or cedar, with ground cover of green mosses	elevated	pine, somewhere with admix of birch, aspen and larch	insufficient or normal	cover of green mosses, somewhere with admix of lichens	Dm-Dm
Pyne stands, with fir, spruce or cedar as the overstory admix and understory, with ground cover of green mosses	gentle and elevated	pine, with admix of fir, spruce, cedar and birch	normal	cover of green mosses	Mm-Mm
Stands of fir, spruce, cedar or birch, or mixed stands with cover of green mosses	gentle	broadleaved, somewhere birch with admix of fir or spruce in overstory, understory or seedlings	normal or somewhere excessive	cover of green mosses	Mm-Mm
Birch stands with cover of herbs and spots of sedge, no litter in summer	gentle	birch, somewhere birch + fir or spruce	normal	cover of herbs with patches of sedge; litter became an upper layer of duff	Ll-Nc1
Aspen stands, no litter in summer	gentle	aspen, somewhere with admix of birch, or spruce	normal	thin cover of herbs and low herbs, litter became an upper layer of duff	Cl-Nc1
Calamagrostis type cuts, selection cuts, young stands	elevated and gentle	cuts, open second-growth stands, sparse stands	normal or insufficient	continious cover of sedge or grass species	Cg-Ll

To draft such predictions of fire behavior, the following data are used:

- a sketch of the fire perimeter; reference points (compartment grid, rivers, lakes, etc.) should be also marked in the sketch;
- a data bank for prompt preparation of the VF map;
- fire weather data for any given day (air temperature, and humidity, wind speed and direction, fire weather draught index and class) and a forecast of those data for the next 24 hours;
- tables of characteristics of burning for PCB types available;
- tables of coefficients of influence of stand canopy closure on wind speed under that forest canopy;
- tables presenting the relative effects of the main factors (wind, slope, humidity) influencing the spread rate of surface fire (Sofronov, Volokitina 1990).

Essential clarifications:

(1) The drought class (DC) is determined by the value of the forest fire drought index (FFDI). Currently, two FFDIs are used in Russia; the Nesterov index and the moisture index of LenNIILKh (HI-1). We have improved those indexes by their integration and inserting corrections accounting for hygroscopicity of fuel, and by simplifying the accounting of precipitation effect. We have named the index "HHI":

$$HHI_n = [HHI_{(n-1)} + (t+10^0)_n] \bullet (t-td-5^0)_n] \bullet (Cr)_n,$$

Where:

- t is the air temperature at 13:00-15:00, ^{0}C;
- td is the dew point at that temperature, ^{0}C;
- n is the number of the day for which this index is being estimated;
- n-1 is the number of the proceeding day;
- Cr is the factor of precipitation, determined by total daily precipitation (R, m) or by daily duration of precipitation (T, hr);

$$Cr = 1.8 / (R + 1) \text{ or } Cr = 1.8 / (1.3\ T + 1)$$

Notes: If R = 0 - 0,5 mm or T = 0 - 0,4 hr, then Cr = 1. In terms of pyrology, the period of 24 hours should be started from a morning hour rather than from midnight. The index for each day is calculated twice: in the morning it is estimated based on the weather forecast, and the next morning it should be recalculated taking into account the factual data.

The three FFDIs mentioned above are expressed in abstract units (u). Forest fire drought classes (DC) are determined by the following FFDI scale: DC 1 - up to 300 u, DC 2 - 301-1,000 u, DC 3 - 1,001-3,000 u, DC 4 - 3,001-100,003 u, DC 5 - 10,001-300,001 u, DC 6 - more than 30,000 u.

(2) Pyrological characteristics of PCB types (table 3) show that the possibility of burning appears at different values of FFDI and heat discharge intensity (E, MJ m^{-2}). It also determines the base rate of fire spread (V_0, m/s) at standard conditions: zero wind speed, horizontal fuel bed, ambient temperature 20^0 C, air humidity 40%.

(3) To estimate wind speed under the forest canopy at a height of 2 m, the value of wind speed at the relevant weather station should be multiplied by the coefficient Kcc, which is a function of canopy closure (C;see tabulation below). The closure is indicated in the pyrological description of the relevant taxon.

(4) The relative contribution of the aforementioned main factors to the rate of spread of surface fire (table 4) is assumed here to be independent and the same for each type of PCB. This contribution, however, is somewhat different for layers of different structures and for different types of surface fire.

The method developed to predict fire spread consists of the following steps:

(1) Data related to the taxon where the fire will burn (or is burning) are selected from the data bank. The boundaries of the territory for which the map is made are sketched more accurately, a copy of a map of that territory is made from the plotting board, and the pyrological descriptions of the taxa in that territory are overlaid to produce a VF map for the area.

(2) Then the VF map is colored to convert it into a map of current fire danger. To perform this, the drought class of the current day defined by weather data is compared with critical drought classes given for each taxon in the pyrological description. Taxa

If	C =	0	0.1	0.2	0.3	0.4	0.5	0.6	0.7	0.8	0.9	1.0
then	Kcc =	0.77	0,70	0.60	0.48	0.38	0.30	0.23	0.16	0.11	0.07	0.05

Table 3.--*Pyrological characteristics of types of prime conductors of burning (PCB).*

Pyrological characteristics	Fire weather drought index FWDI (Nesterov's or HI-1 or HHI)										
	100	200	300	400	500	700	1000	2000	3000	4000	5000
					Lichen type of PCB (Lc)						
E, MJ/sq.m	4.0	5.0	6.5	7.5	8.5	9.0	9.5	10.0	10.0	10.0.	10.0
V_0, m/s	0.003	0.004	0.005	0.007	0.008	0.009	0.010	0.011	0.011	0.011	0.0110.
					Dry moss type (Dm)						
E, MJ/sq.m	-		-	-	3.5	5.3	10.0	16.0	20.0	22.0	24.0
V_0, m/s	-	-	-	-	0.002	0.002	0.003	0.004	0.004	0.005	0.005
					Moist moss type (Mm)						
E, MJ/sq.m	-	-	-	-	-	-	2.3	8.8	12.0	13.2	14.4
V_0, m/s	-	-	-	-	-	-	0.003	0.004	0.004	0.005	0.006
					Loose litter type (Ll)						
E, MJ/sq.m	-	-	-	-	-	4.0	7.5	10.0	11.0	11.5	12.0
V_0, m/s	-	-	-	-	-	0.002	0.003	0.005	0.006	0.007	0.008
					Compact litter type (Cl)						
E, MJ/sq.m	-	-	-	-	-	-	3.0	9.0	12.0	13.5	15.0
V_0, m/s	-	-	-	-	-	-	0.003	0.004	0.004	0.005	0.005
					Cured grass type (Cg)						
V_0, m/s	-	-	0.005	0.005	0.006	0.006	0.007	0.007	0.007	0.007	0.007

Note: Bog-moss (Bm) and Non-conductor (Nc) types never burn. E - heat discharge intensity;

V_0 - base rate of spread.

Table 4.--*Coefficients of influence of wind, slope and relative humidity on the rate of fire spread.*

					1.Effect of wind							
Wind speed, m/s (Height 2m)	0.0	o.4	0.8	1.2	1.6	2.0	2.4	2.8	3.2	3.6	4.o	5.0
K_w, head fire	1.0	1.2	1.4	1.7	2.1	2.6	3.2	4.0	5.0	6.0	7.0	10.0
K_w, back fire, flank	1.0	1.0	1.0	1.1	1.1	1.2	1.2	1.3	1.4	1.5	1.6	1.8
K_w, high-intensity f lank (by Konev, 1984)	1.0	1.1	1.2	1.4	1.6	1.8	2.0	2.2	2.5	3.0	3.5	4.5
					2. Effect of slope							
Slope, degrees	-40	-30	-20	-10	0	10	15	20	25	30	35	40
K_s	0.7	0.7	0.7	0.9	1.0	1.2	1.5	2.0	2.5	4.0	6.o	12.2
					3. Effect of humidity							
Humidity, %	20	25	30	35	40	45	50	55	60	70	80	90
K_h	1.7	1.5	1.3	1.1	1.0	o.9	o.8	0.7	0.7	0.6	0.5	

for which critical drought classes are lower than the drought class of the current day are regarded ready to burn; they are colored red. Taxa for which critical drought classes are higher than the drought class of the current day are considered not ready to burn for a while, and they are colored green. Taxa for which critical drought classes are the same as the current drought class are considered to be in uncertain burning state; they are colored yellow.

(3) The perimeter of the fire is copied from its sketch to the prepared map, recording the time when this perimeter was drawn. An arrow indicates the wind direction, and the wind speed also should be indicated.

(4) The time period for which the fire spread is to be predicted is determined and ordered. In case of large fire, some more directions probably will need to be drawn from its perimeter, each direction normal to the fire line; their angles to the wind direction should be indicated.

(5) Four directions in which the fire is anticipated to propagate are marked in the map by straight lines, usually from the center of fire: 1) heading, downwind; 2) backing, upwind; 3) flanking, right flank, and 4) flanking, left flank, (the latter two are normal to the wind direction).

(6) Those lines may cross various taxa available in the map. To estimate what time (t, hr) it will take fire to propagate along those directions through each taxon, length of the line (L, m) crossing the taxon is divided by the estimated rate of fire spread in this taxon (V_x, m/s):

$$T = L / V_x$$

The estimated rate of spread is determined by the formula:

$$V_x = V_0 \bullet K_s \bullet K_h \bullet K_w,$$

where K_s - factor of contribution of slope to the spread rate; K_h - factor of contribution of humidity to fire spread rate; K_w - factor of contribution of wind speed to fire spread rate. V_0 is taken from table 3, in accordance with the PCB type indicated in the pyrological description of the taxon, and the FFDI value. These factors are shown in table 4. Wind speed should be corrected for canopy closure.

(7) If there is a "yellow" taxon in the direction of fire propagation, i.e., the possibility of burning here is uncertain at present, two versions of fireline spread in that place are to be estimated: 1) assuming the taxon is ready to burn, and 2) assuming it is not ready to burn.

(8) If the taxon in the direction of fire spread is not ready to burn ("green"), the fire will stop spreading in this direction if the barrier is wider than the width of the burning strip of the fire edge. In the other case the fire will propagate along the barrier in either one or two directions. To estimate the width of the burning fire edge in the direction of the barrier, the fire spread rate along neighboring directions should be predicted first. If the spreading fire encounters a barrier (a road, a brook, a river, etc.) its backfire and flanks are usually stopped while the head fireline, particularly a long one (over 100 m) or that of high-intensity, may overcome such barriers except rivers of $\geq$50 meters wide. In summer, river's flooded lands should be added to the width of the river as they are usually not ready to burn.

(9) As a result, a fire perimeter is drafted for the given time, taking into account: the time period for which the perimeter of the fire has been drafted; the period required for fire to propagate across each taxon in the given direction; and the final ordered time for which the fire perimeter is to be predicted.

(10) To estimate the perimeter length (i.e., length of the fireline), the predicted perimeter is drawn in the map through the points that the fireline would reach by the ordered time in each direction. The length of this perimeter is measured (taking into account the scale) and multiplied by the twist coefficient, for which the average value is 1.5.

(11) The rate of perimeter increase (V_p) is determined by the rate of spread of the fire head (V_{fh}) in the last taxon, i.e., closer to the ordered time:

V_{fh}, m / hr	25	40	70	120	200
V_p, m / hr	200	300	400	700	1,000

(12) The power of the surface fire is determined and qualitatively classified by the intensity of heat discharge from a unit of length of its head fire edge (Ifh), where: $I_{fh} = E \bullet V_{fh}$ and E is taken from table 3 in accordance with PCB type in the last taxon in the fire head direction, and the FFDI:

Intensity (Ifh, kW/m)	**Power of Surface Fire**
Up to 35	Low
35-120	Moderate
Over 120	High

(13) Because data on the locality of fire perimeter, its intensity and the rate of perimeter increase for the given time, as well as characteristics of taxa adjoining the fire, are outlined in the map, it is

easier to draft optimum plans to control the forest fire.

We developed special schemes for a simple calculation of the amount of resources necessary for successful control of fires.

DISCUSSION

To predict wildland fire and prescribed burning behavior, "fuel models" and complex "fuel types" are used in USA and Canada (Deeming *et al.* 1977; Forestry Canada, Fire Danger Group 1992). Their pyrological characteristics are based on much factual data on fires. In Russia it is not possible to make such a bank of empirical data. That is why an analytical method is used in our fire behavior prediction system. It is based on the vegetation fuel classification (as components of the ecosystem) and on studying their pyrological characteristics, which depend on natural factors and the mutual influence of seasonal changes in ecosystem components.

In the future we plan to use American and Canadian experience to improve wildfire management in Russia. We have analyzed the CFFDRS already due to training in Canada (Alberta University) and prepared a paper (Sofronov, Volokitina 1996). We would like to do the same with the NFDRS. For this aim we need communications with American scientists who study and improve the NFDRS

CONCLUSION

Unfortunately, because of budget shortage and the intricate economic situation in Russia our work is suspended. For this reason, and because of minimal forestry activity, the newest scientific achievements cannot be introduced into the practice of forestry at present. But we hope for the best in future. In the V.N.Sukachev Institute of Forest, a GIS for the Krasnoyarsk region is being developed through a joint Russian-American project. This GIS incorporates our program for operative large-scale vegetation fuel (VF) map creation (Volokitina, Sofronov, Klimushin 1996). So, we hope using VF-maps for fire management in Russia will start soon.

At present with the financial support of the Krasnoyarsk Regional Science Foundation we are elaborating a theoretical basis for pyrological "thermal land-reclamation" for forests on permafrost soils. The results of this project together with vegetation fuel maps can be used for improving wildfire management in these forests.

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KRASNOYARSK SUBSYSTEM OF WILDFIRE MONITORING FROM SPACE AS AN ELEMENT OF THE WORLD WILDFIRE MANAGEMENT SYSTEM

A.I. Sukhinin[1]

ABSTRACT.--A system being developed for the Krasnoyarsk Region of Russia integrates satellite remote sensing, Geographic Information Systems, and ground-based sampling to provide timely information for wildfire prediction, monitoring, and management on several scales. Hardware, software, and methods of data analysis and interpretation are described. This approach is proposed as a prototype for a potential worldwide wildfire management and monitoring system.

INTRODUCTION

Wildfire as an environmental factor, especially in terms of growing global change impact, necessitates development of a worldwide fire management system. Current technologies concerning main fire problem decisions such as: forest fire danger rating, prediction of forest fire start, spread and behavior, operational and reliable detection of forest fires, choice of strategy and tactics of fire suppression, and ecological and economic estimation of forest fire effects are now at different stages of development in leading forest countries.

We propose to begin developing a world forest fire management system by building a specialized Geographic Information System (GIS) for wildfire monitoring that will require joint efforts of scientists and service staff from a number of countries. The development of an experimental subsystem of wildfire monitoring based on a downlink station installed in Krasnoyarsk through the cooperation of the U.S. National Aeronautic and Space Administration (NASA) is an example of such cooperation. This station receives data from satellites operated by the U.S. National Oceanographic and Atmospheric Science Administration (NOAA), and we can consider it as an element of a future worldwide wildfire management system.

Close correlation between the Canadian Fire Weather Index (FWI) and the Russian (Nesterov factor) index of fire weather, along with information from NOAA satellites such as spatial patterns of accumulated temperatures, the Normalized Difference Vegetation Index (NDVI) serves as a basis for developing a generalized Canadian-Russian fire danger rating system to cover most of the world's boreal forests.

An experimental GIS has been built for the Krasnoyarsk Aerial Forest Protection Service (KAFPS) that uses space information for monitoring wildfires and the postfire state of forests at a regional level. It is intended for:

- Operational assessment of spatial distribution of fire classes through computer-aided synthesis of information in the near and far infrared bands,
- Large forest fire dynamics mapping and a determination of fireline energy parameters,
- Estimation of optical and physical parameters of zones under permanent smoke cover during mass forest fire events,
- Mapping of local precipitation and estimation of storm clouds and clouds for seeding,
- Inventory of burned areas and description of their post fire state,
- Monitoring of snow cover dynamics as an indicator of the beginning and end of the fire season.

REGIONAL SUBSYSTEM OF WILDFIRE MONITORING

The subsystem of wildfire monitoring is a multilevel operational subsystem for spatial assessment of forest area fire danger, wildfire detection, estimation of fire intensity, prediction of fire spread and development, assessments of postfire effects, and recommendations for operational decisions. It is based on the use of multiscale remote sensing and ground data

[1]*V.N. Sukachev Institute of Forest, Siberian Branch, Russian Academy of Sciences, Krasynoyarsk, Russia. E-mail: ifor@krsk.infotel.ru.*

integrated in a single information system applying modern computer technologies.

Organization of the wildfire monitoring subsystem operation is managed by the Regional GIS Center of the Siberian Branch of the Russian Academy of Sciences (Krasnoyarsk) in collaboration with the Forest Committee (Krasnoyarsk), KAFPS, the Committee on Nature Protection and other interested organizations in the Krasnoyarsk Region.

Purposes and Goals of the Wildfire Monitoring Subsystem

The purposes of wildfire monitoring subsystem development in the Krasnoyarsk Region are to:

1. Increase forest protection and wildfire management effectiveness, reduce ecological and economic damage from wildfire, and enhance understanding of processes of fire ignition and development and the influence of fire on forest structure and dynamics.
2. Provide environmental protection and natural resource information to regional and local forest management committees and other users.

The subsystem is aimed at the following primary tasks:

- Development of a multilevel information subsystem for spatial and temporal forest monitoring, including technical instruments for collection, processing, analysis, and storage of remote sensing and ground data, and a system of prediction models combining digital and map databases,
- Spatial assessment of fire danger and prediction of fire dynamics over a large forest region,
- Fire detection and determination of wildfire intensity in unprotected northern taiga forests,
- Prediction of large wildfire behavior,
- Analysis of weather in order to predict and suppress fires,
- Inventory of postfire forest state,
- Distribution of relevant information among users to improve fire management decision-making at different management levels.

Objects of Wildfire Monitoring

The three major areas of concern for remote wildfire monitoring are characterization of wildfires, meteorological conditions that affect fire danger and fire behavior, and landscape characteristics that relate to fire behavior and fire effects. Some of the key objects for database development are:

- Wildfires: mechanisms of ignition and combustion of forest fuels, processes of fireline spread, and turning of fire from one type to another, processes of energy release including dynamics of spectral reflection and radiation characteristics, processes of smoke emission and movement of smoke aerosols,
- Meteorological conditions: weather and climatic factors including air mass transfer, and meteorological parameters with a special stress on identifying storm and resource clouds,
- Landscape characterization: forest vegetation, soils, wildlife, and water in the forest ecosystems.

Levels of Wildfire Monitoring

In order to address the problems discussed here, we are developing three levels of wildfire monitoring: (1) Krasnoyarsk Territory, (2) regional and (3) local.

The Krasnoyarsk Territory level involves the organization of wildfire danger monitoring, weather and active wildfire information in the territory of the Krasnoyarsk Region, including areas that do not receive aerial fire protection (north of about 60^0 latitude). It is intended for general assessment and medium-term prediction of fire situations on a broad scale across the Krasnoyarsk Region, for obtaining integrated assessments of modes and routes of fire detection flights, as well as for proper use of manpower and fire fighting equipment on the basis of assessments of expected fire damage. The main sources of spatial information are map data and data from satellites, scale 1:10 million to 1:2.5 million.

The regional level provides detailed estimates of fire situation severity in regions within the Krasnoyarsk Territory. This is intended to be sent to regional air bases and air crews, and

includes daily recommendations on number and routes of fire detection flights and rearrangement of manpower and facilities within regional air bases. Source spatial information has a scale of 1:1 million to 1:0.5 million.

The local level is implemented at the level of a Forest Exploitation Area (FEA). As a rule this is used only in areas of large fires (burned areas more than 200 ha). Maps and aerial and satellite materials used are of the scale 1:0.1 million and larger. Fire spread and growth are predicted, and, on the basis of expected damage assessments, an optimum plan of fire containment and suppression is worked out.

Stages of Development for the Wildfire Monitoring Subsystem

Stage 1 includes development of technical facilities and an associated database, as well as calculation procedures of factors under monitoring for Krasnoyarsk Region level subsystem by KAFPS (1996-1997). Technical facilities include: (1) a satellite data receiving station, (2) a computer-based work station for preprocessing satellite data, (3) a computer-based work station for a KAFPS dispatcher, (4) netware. The combined database contains mapping materials (maps of forests, forest landscapes, relief, hydrological system, administrative borders, road system, and fixed-wing aircraft and helicopter landing spots, etc.), satellite data, weather information, and information on fire fighting resources.

Stage 2 will be creation of a subsystem for wildfire monitoring at a regional level (1997-1998), and Stage 3 will be development of the local level subsystem (1998-1999).

Organizational Structure of the Remote Wildfire Monitoring Subsystem

The basis of the organizational structure of the wildfire monitoring subsystem is regional jurisdiction organs and a country-wide network of their departments that are in charge of environmental observations and control, such as:

- Krasnoyarsk Aerial Forest Protection Service (KAFPS),
- Regional Center for receiving satellite information (HRPT station from NOAA satellites, Krasnoyarsk Scientific Center, Siberian Branch, Russian Academy of Sciences),
- Krasnoyarsk Forest Committee,
- Krasnoyarsk Committee of Nature Protection,
- Krasnoyarsk Committee of hydrology and meteorology,
- Krasnoyarsk Committee of Extreme Situations and Civil Defense.

Organization and methodological control of the wildfire monitoring subsystem is carried out by a coordinating board created from subsystem leaders or branch territorial monitoring centers and users of the Krasnoyarsk Service of ecological monitoring.

Operational control at the regional level is by a chief of the Forest Area Fire Protection Service (FAFPS), and on the Krasnoyarsk Territory level by the administrator of the Krasnoyarsk Territory.

PRACTICAL REALIZATION OF THE PROJECT: A SPACE WILDFIRE MONITORING SUBSYSTEM FOR THE KRASNOYARSK REGION

The Project is intended to establish fundamental principles of a GIS for spatial assessment of fire danger in boreal forests, detection and description of wildfires, and determination of the postfire state of forests using remote sensing of Earth from satellites.

The first stage of the wildfire GIS development includes the development of a functional GIS and applied mathematical software for the analysis of space images of forest areas, and for solution of above mentioned tasks, as well as operational output of thematic maps.

Spatial dynamics of fire danger in boreal forests is determined by AVHRR and TOVS measurements of forest area and atmosphere infrared and microwave radiation from space platforms. The theoretical foundation of this method is given in previous publications, and a mathematical model of forest fire danger dynamics describing radiative and reflective characteristics of forest fuels is designed.

In order to refine useful decoding procedures for space images, as well as to develop and test

procedures for atmospheric and terrain corrections, a test site is being established in the mountain forests of the Mana river basin. Simultaneous space, aircraft and ground experiments will be conducted to determine fire danger and intensity of combustion during experimental and wildfires and to appraise burned area state.

Structure of the Subsystem Technical Complex

The HRPT station is equipped with a modern computer complex, allowing us to carry out the first stage of the wildfire monitoring GIS in 1997.

The Krasnoyarsk downlink station is based on two computers. A schedule of NOAA satellite passes is done using an IBM PC Pentium 166, as well as orbit tracking and other operational calculations. On-line functioning and processing of AVHRR/HRPT telemetry data are performed on a Micro VAX II computer. In addition, image processing is performed on the Image Visualization and Analysis Station (IVAS) of the International Image Systems (IIS).

IVAS permits visualization and processing of 3-channel color images (1024 by 1024 pixels). MicroVAX II includes digital audiostorage on 4mm magnetic tapes (2.0Gb) for archiving given images. The current station complex is involved in a high-speed (receiving capacity up to 10 megabaud) computer network that connects computing centers of the Krasnoyarsk Committee of Geology and Krasnoyarsk Scientific Center of Siberian Branch of RAS, as well as KAFPS. This has increased computing capability of the HRPT station by an order of magnitude. By the end of 1997, the administration of the Krasnoyarsk Region and Committee of Hydrometeorology, Climate and Environmental Protection will be added to the network.

Available programs and hardware include:

1. Software programs
   - NASA programs for obtaining data and satellite information preprocessing,
   - Software packages GIS ArcINFO 3.4.2, ArcVIEW 3.0, IDRISI for WINDOWS, EPPL-7,
   - Programs of wildfire spread and development,
   - Programs for data collection, preprocessing and transmission from the wildfire patrol aircraft.
2. Hardware
   - A U.S. downlink receiving information from NOAA satellites,
   - A system of satellite data processing (VAX-11, IVAS-Display processor),
   - A system of A0, A4 format image input, output,
   - A system of data collection from aircraft using a radio channel and prediction of wildfire spread,
   - Automatic complex for fire aerial detection in the infrared and microwave bands.

Completion of the proposed subsystem will require expansion of the current complex of technical equipment to include: an airborne large fire detection information and measuring complex working in infrared and microwave bands of the spectrum; a system of information transmission through a radio channel; and a computer-based working station for predicting wildfire spread and modeling of fire containment and suppression.

Algorithms to Carry out Wildfire Monitoring Tasks

Methods are under development for mapping, prediction, and analysis of cloud cover and boundaries, storms, snow cover, surface temperature, fire danger, and fuel moisture.

Cloud Distribution

Creation of a cloud cover composite image: A cloud cover distribution map will be produced by fragmenting full-scale cross-pixel images of the territory in near (channel 2) and far (channel 4) infrared bands and then putting the fragments together. After overlaying of the hydrologic network and coordinates, it will be converted into a stereographic projection.

Determination of storm boundaries: A map of boundaries of storms; assessment of probable number and location of lightning vertical discharges; and predictions of the spatial and temporal dynamics of storm clouds will be produced. Storm clouds will be identified by synthesis of information from visible, near IR and thermal bands of the spectrum. Big vertical cumulus clouds will be recognized based on

spatial and frequency characteristics of image spectral brightness analysis. Boundaries of storm clouds will be identified using cluster analysis; and area, velocity and directions of storm clouds will be calculated. Additional information will be incorporated from lightening locators and from sounders of lightening vertical discharges.

Determination of cloud boundaries: A map of distribution of clouds suitable for cloud seeding (to produce rain in fire areas) will be produced in two stages. The first is a synthesis of cloud images in visible, near IR and thermal bands of the spectrum; computer based analysis of big cumulus clouds will use spatial pattern recognition on spectral and contrast frequency signs. The second is the identification of boundaries of clouds suitable for artificial cloud seeding, i.e., clouds in the radius of 30 km-50 km around large fires and moving toward forest fires.

Snow cover

Determination of snow cover boundaries: A map of snow cover boundaries will be developed by synthesis of underlying surface images in visible and IR bands of spectrum using spectral brightness differences and radiation temperature differences. First, snow cover boundaries will be identifies through automatic image processing in visible and IR bands of spectrum using the standard programs of equalization, contrasting, binarization, and median filtration. Then images will be overlain on the topographic map and geometric distortions corrected using the hydrological network. This approach will ensure an accuracy of positioning of 5 km-10 km in case of clear boundary of snow cover and 10 km-20 km in case of a vague boundary of snow cover.

Movement of snow cover boundaries: Snow cover boundary movement will be determined by overlay of consequent boundary contours. Predictions of snow cover boundary movement will use weather forecast data and analysis of the atmosphere using satellite images. These data will be analyzed using mathematical models describing movement of boundaries of fuzzy sets to predict snow cover boundary movement.

Temperature

Construction of a radiative temperature map: A normalized map of surface layer temperature distribution will be derived through calibration and correction of satellite data. Raw temperature data from the scanner is corrected for the radiometer calibration and geometric distortion connected with the scanning method. Corrections related to limited spectral range of the sensor, the modulated transmission function of the equipment, atmospheric transmission, spectral characteristics of natural background radiation, and reflection are taken into account. Joint calculation of the radiative temperature at the medium and far IR ranges allows estimation of thermodynamic temperature and areas of small-scale thermal anomalies, for detection of recently started wildfires. Construction of the isotherms is produced by the method of radiation temperature averaging in the sliding window and followed by thinning of isolines.

Fire Danger

Determination of the beginning of the fire danger period: The beginning of the fire danger season is judged from the disappearance of the snow cover boundary. The approach of fire danger season is determined for each type of main forest fuels using the sum of radiative temperatures corresponding to the moisture of extinction where combustion can occur. For each fire period, corresponding radiation temperatures will be different. It is necessary to take into account recent precipitation and drying factors, such as wind and solar radiation. The result is maps tracking spatial distribution of the beginning of the fire danger period over time.

Assessment of forest fuel moisture content: Maps of current and predicted forest fuel moisture content distribution for a given region will be based on combining infrared images of the forested area with a contour map of forest fuels, overlaying the hydrologic network, and recognition of precipitation areas. These maps will be modified daily by calculation of ground cover radiation temperatures for each pixel of an image and determination of their correlation with forest fuel moisture content. Analysis will include spectral brightness in the near IR band and landscape thermal inertia factors. Fuel moisture content predictions will be based on weather forecast information.

CURRENT PROGRESS

1. Development of a new information technology of forest fire danger assessment is continuing using remote sensing data on the Earth received from satellites in the thermal band.

2. The objectives of the wildfire GIS, its structure, with the analysis of its characteristics at different levels, and the place of remote sensing facilities in wildfire monitoring have been determined.

3. A functional structure of the wildfire monitoring GIS has been proposed, including key subsystems, such as assessment and predictions of wildlife danger; monitoring of wildfire ignition and spread; and monitoring of wildfire prevention, detection, and suppression.

   Analysis and systematization of the wildfire monitoring GIS information structure are completed, including the organization of digital and map databases obtained at different scales.

4. The general structure of the wildfire monitoring GIS software has been proposed including the system and applied components.

5. A high-speed (up to 10 gbaud) computer network has been created linking the HRPT station receiving data from NOAA satellites and the Krasnoyarsk Aerial Forest Protection Service.

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THE CONCEPT OF DEFINING THE REQUIRED LEVEL OF FIRE PROTECTION INTENSITY IN BOREAL FORESTS

Yuri Z. Shur[1]

ABSTRACT.--An approach to defining desired and actual levels of forest fire protection is considered. The main features of an expert system used for defining protection are described. Variables include fire information, geographic databases, meteorological data, and environmental, economic, and social considerations and may be quantitative or qualitative. The approach is illustrated for a hypothetical site.

The 1992 United Nations Conference on Environment and Development formulated some principles of global development. The main humanity-wide problem was defined as preservation of natural biodiversity. Taking this principle into consideration, we must point out that sustainable management of forest ecosystems is a major consideration for forest landscapes. So, preservation of landscape biodiversity is a main objective of forest fire protection.

Generally, the forest fire protection level in specific areas is designed to attain certain outcomes from the forest fire protection system (FFPS). We report both the planned fire protection level and the level actually achieved. The first is defined before the fire season. The second is defined after the fire season. To define the scale of fire protection level means to classify protected areas according to the type and intensity of fire protection. This scale consists of six fire-protection intensity levels:

- zero (no fire protection);
- extremely low;
- low;
- medium;
- high;
- extremely high.

The main limitation of present methods for evaluating fire protection levels is a lack of calculated indices regarding intensity levels of fire protection. We relate any fire protection intensity level to a calculated index based on values over a certain time interval. This index is the proportion of total fire ignitions that were contained in a timely manner. A timely containment is defined as a fire whose time from the moment of ignition to the beginning of suppression is less or equal to a certain target value. It is also assumed that the number of firefighters is adequate for successful fire containment.

Every fire protection level is related to particular conditions that affect the functioning of the FFPS. The main factors we use to describing the functioning of the FFPS are fuel moisture, wind velocity, and the period of the fire season when the fire occurs. The influence of moisture on the total number of fires on a given day in the fire season affects fire containment efforts. Wind velocity affects the rate of spread, one of the key tactical elements in fire suppression. There are also wind restrictions on the ability to use aerial fire protection methods. The prevailing forest fire type is determined by forest conditions and period in the fire season. Forest fire moisture is characterised by three Fire Danger Levels: low, medium and high. For wind velocity we consider two levels: up to 10 m.p.s. and above 10 m.p.s. The fire season is divided into three Fire Season Periods: spring, summer, and autumn.

The geographic area used for defining the FFPS's intensity level is a "sub-unit". By "sub-unit" we mean the existing and established elementary unit for which we can characterize the functioning of the interconnected complex of methods and capabilities for detection, delivery and suppression. (V.Pokryvailo and Y.Shur 1992). For the purposes of fire protection, Russia is divided into zones based on the primary means of protection: aerial protection, protection by ground forces, or unprotected. The definition of sub-units will depend on the primary means of fire protection in a specific area. For example, the Air Division is the sub-unit for the aerial protection service, and the Forest District is the sub-unit for ground protection.

The following factors are used in defining protection levels:

- Statistics on numbers of forest fires;
- Forest conditions;
- Meteorological conditions;
- Relief;
- Hydrography;
- Ecological significance;
- Presence of enterprises at extreme risk from fire;
- Economic activities.

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[1]*Yuri Z. Shur, Saint-Petersburg Forestry Research Institute, Institutsky Av. 21, St. Petersburg, 194021, Russia*

The decision-making process for defining fire protection levels is based on the use of an expert system that incorporates these factors. This expert system integrates the possibilities of computer data analysis with information derived from expert knowledge and opinion. Such a system can "suggest" reasonable advice or solutions. An additional feature of the system is that it explains the methods used in searching for a reasonable solution. One must receive the answer in clear and simple form.

Any expert system is a synthesis of three elements: a data base, a knowledge base, and a control structure. The data base is the compilation of information about quantitative and qualitative characteristics of an area. In our case the data base is information on the factors listed above for defining forest fire protection levels. The knowledge base is the synthesis of conceptual knowledge, described in terms of rules of formal logic, such as "if – then". For example, if the density of fire ignitions fires is more than 5 for a given type of conditions then the fire protection level would be 'high'. The control structure defines rules concerning the order in which variables are considered in the analysis. For example, all rules connected to radiation danger might be evaluated first, ones connected to ecological significance evaluated second, etc.

CONSTRUCTION OF A DATA BASE

Data base indices are subdivided into two groups: indices that do not require special algorithms and indices requiring special algorithms.

The indices of the first group are introduced into the data base from the computer screen. These include:

- Fire Danger Level (low, medium, high);
- Wind velocity (up to 10 mps above 10 mps);
- Fire season period (spring, summer, autumn);
- Level of radiation contamination (low, medium, high);
- Level of chemical contamination (low, medium, high);
- Presence of atomic power stations or other enterprises comprising possible radiation risk in dangerous proximity to the protected area. (Yes – 1, No – 2);
- Presence of other extremely dangerous or critical production activities;
- Index of other extra risk factors such as economic risk.

The indices of the second group require special algorithms. These algorithms are subdivided into two groups: those that use numerical input data and those that using cartographic input data. The following types of algorithms are considered:

1. Determination if the "elementary unit" is inside a protected area. By "elementary unit" we mean the object for which database information is accumulated. For example, a forest fire is the "elementary unit" for a forest fire database; a forest site is the "elementary unit" for a forestry database; a meteorological station is the "elementary unit" for a meteorological database. Within the framework of our task, we define whether or not a particular fire, site or station lies inside a protected area.
2. Calculation of average index values (e.g., calculation of average burned area for particular conditions of functioning of FFPS);
3. Construction of distribution for index values; (e.g., fire distribution based according to ignition sources or distribution of forest district areas based on categories of lands)
4. Summing values of certain indices;
5. Dividing values of certain indices;
6. Obtaining values of some indices from an appropriate database;
7. Average values of certain indices;
8. Average difference in values between indices;
9. Searching certain index values for a "elementary unit" in the database;
10. Sampling of several indices from the database;
11. Repeating one of the above mentioned algorithms on a region including the calculated area.

AN EXAMPLE

An example of the definition of forest fire protection levels (here defined before the fire season) under different levels of fire danger for a hypothetical sub-unit is presented in table 1. Table 2 describes the proportion of fires that are expected to be contained within time guidelines for different fire protection levels for the same hypothetical area. These tables represent an example for a specific combination of the data base variables described above. Fire protection levels might be changed, for example, by a lower or higher index for economic values at risk or risk of radioactive contamination.

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Table 1.--*Definition of appropriate fire protection level.*

Working conditions		**Fire protection level (for fire season periods)**		
Fire Danger Level	**Wind velocity(m.p.s.)**	**Spring**	**Summer**	**Autumn**
1 (low)	≤ 10	low	low	low
1	> 10	low	low	low
2 (medium)	≤ 10	medium	medium	medium
2	> 10	high	high	high
3 (high)	≤ 10	high	high	high
3	> 10	high	extremely high	high

Table 2.--*Acceptable range in proportions of fires meeting the goal for timely containment under different fire protection levels.*

Fire protection level	**Timely containment (Proportion of fires)**	
	Above	Up to
Zero	N.A.	N.A.
Extremely Low	0.70	0.80
Low	0.80	0.90
Medium	0.90	0.95
High	0.95	0.97
Extremely High	0.97	1.00

USE OF BEHAVE FOR FOREST FIRES AND PRESCRIBED FIRE EXPERIMENTS IN SIBERIA

Eric N.Valendik,Yegor K. Kisilyakhov, Galina A. Ivanova, Valentina D.Perevoznikova,
Serghey V. Verkhovets[1]

ABSTRACT—Prescribed fire was tested as a potential tool for site preparation and for reducing fire hazard after clearcut logging in dark coniferous forests in Siberia. Experimental burns were conducted on 8 sites to evaluate the practicality of fire use and effects of prescribed fires on fuels and on site conditions for regeneration. The BEHAVE system for fuel and fire behavior modeling was used successfully to predict fire behavior and to help determine prescription windows for burning. The application of prescribed fire following clearcut logging in Siberian forests shows considerable promise as a management tool.

INTRODUCTION

Any fire use in forests of Russia was prohibited until 1995, when a permission was given to use early spring burns in grass fuels near roads before the beginning of the fire season (Valendik 1996). Within the framework of the Central Siberian Sustainable Forest Management Project, the V.N.Sukachev Institute of Forest, Siberian Branch, Russian Academy of Sciences, in collaboration with the USDA Forest Service and the Russian Federal Forest Service, began to study experimental prescribed fires in the Krasnoyarsk Region starting in 1996. The main objectives of these experiments in slash fuels were: (1) fire hazard reduction, (2) site preparation for planting and natural regeneration, and (3) developing regulations for using prescribed fire in forestry practice of Russia.

The fire behavior prediction and fuel modeling system BEHAVE developed at the Intermountain Fire Science Lab., USA (Burgan and Rothermel 1984, Andrews 1986, Andrews and Chase 1989, Andrews and Bradshaw 1990) was used for planning experimental prescribed fires in slash fuels of clearcuts. Custom fuel models were developed for these slash fuels and used as inputs to the RXWINDOW program for the determination of an optimal range of weather and fuel moisture conditions. This paper presents some results of these experimental prescribed fires and future plans.

METHODS

Description Of Experimental Sites

In 1996 and 1997 a series of experimental plots was prepared at 8 dark coniferous forest logging slash clearcuts near Predivinsk (Krasnoyarsk Region). Experimental burning sites were selected in Bolshaya Murta leskhoz (Krasnoyarsk Region) and firelines were prepared for prescribed burns by Predivinsk lespromkhoz staff immediately after logging. Slash fuels resulted from clearcut logging of the dark coniferous forest stand (fir, spruce) on the sites 1 to 2 years before burning. Logging was done in winter using LP49 felling machines. Clearcuts were 20 ha to 50 ha in area and surrounded by firelines 4 m to 6 m wide made by a bulldozer. Some clearcuts were further divided by firelines into small 2 ha to 3 ha plots. Advance regeneration left after logging had red needles due to sudden exposures to light in many clearcuts.

Fuel Sampling

Preburn sampling of all fuels was conducted on each experimental site. Fuel loads of surface fuels were determined at 10 m intervals along a transect, using small sampling plots (0.2 m x 0.25 m) for 1 hr forest fuels, and larger plots (0.5 m x 0.5 m) for live forest fuels. Fuel loads of downed woody materials were defined on 15 m long transects from these sampling points. Surface to volume (S/V) ratios were calculated from transect data. Postfire sampling of fuels was also done near the same points to determine fuel consumption.

[1]*V.N.Sukachev Institute of Forest, Siberian Branch Russian Academy of Sciences, Krasnoyarsk 660036 Russia; e-mail: ifor@krsk.infotel.ru.*

Defining Prescription Windows

After sampling fuels and defining fuel loads, custom fuel models were developed using the fuel subsystem of BEHAVE (Burgan and Rothermel 1984; Burgan 1987). After testing the models using DIRECT and TSTMDL we defined prescription windows with the help of RXWINDOW (Andrews and Bradshaw, 1990). Windows of acceptable weather and fuel moisture content conditions then were used in planning and conducting prescribed fires.

Fire Weather Monitoring

Suitable weather patterns for prescribed burn experiments were identified using weather images received from NOAA satellites by the Krasnoyarsk NOAA satellite downlink station (Sukhinin 1996). Using these images we can plan prescribed burn experiments so as to conduct them with minimal equipment and manpower. On each site weather characteristics such as dry bulb temperature, relative humidity, and mid-flame wind speed were determined before, during, and after the fire.

Fire Behavior

After ignition and achieving quasi-steady state at the fire edge, fire characteristics such as forward spread rate, flame length, and width of the burning zone were measured by visual observation and slide and video film recording. In some cases special methods for changing of fireline intensity were used so it was very difficult to define characteristics of the fire. But in many cases it was possible to estimate fire characteristics in a short length of time.

RESULTS AND DISCUSSION

During the summer of 1996 and June-July 1997, 8 clearcuts with a total area of more than 300 ha were burned in dark coniferous logging slash fuels in Central Siberia (Krasnoyarsk Region). Presented below are some results of BEHAVE usage for experimental fire number 1/96 as an example. After defining of fuel loads, a custom fuel model was developed with the help of the FUEL subsystem of BEHAVE. Fuel characteristics are presented in table 1. The S/V ratio of 1-hour fuels was 76 cm^2 cm^{-3}. The custom fuel model was tested using TSTMDL, and the model fire behavior predictions are presented in table 2. A fire behavior characteristics chart (Rothermel 1983) for custom fuel model 14 is presented in fig.1.

The custom fuel model 14 lies between NFFL fuel models 11 and 13 (fig. 1). Using RXWINDOW we can define windows for acceptable weather and fuel moisture conditions from table 3 for a backing fire and similar tables for head and flank fire. After this comparison it was decided to use a backing fire with 1 hr fuel moisture content near 7%-10% and with midflame wind speed up to 5 km/hr.

Suitable weather patterns were determined using images received from NOAA satellites by the Krasnoyarsk downlink station installed by NASA in the Krasnoyarsk Scientific Center, Siberian Branch, Russian Academy of Sciences (Sukhinin 1996). The weather pattern presented in table 2 was observed July 19, 1996 before experimental fire number 2/96. Images were also used for detecting the experimental fire areas. Because the slash fuels burned very intensively, it was possible to detect these fires on small areas.

Immediately after logging there were a lot of slash fuels, and fuels were distributed irregularly so intensity of burning varied. In some cases convection columns developed, and we were able to use SPOT to estimate spotting distances.

Fuel loads were decreased in the burned areas. Down and woody fuel loads decreased from 4.3 kg/m^2 to 0.3 kg/m^2 and initial volume of sound wood from 108.5 $m^3 ha^{-1}$ to 68.6 $m^3 ha^{-1}$ on a 60 ha clearcut (prescribed fire number 1/ 96) near Predivinsk. Based on information in Stocks (1991), we estimated emissions from experimental fire number 1/96. Using emission factors of 445 g/kg^{-1} for carbon dioxide, 45 g/kg for carbon monoxide, 4.55 g/kg^{-1} for methane and total nonmethane hydrocarbons (aerosols) we estimated 1,068 tons for carbon dioxide, 108 tons for carbon monoxide and 10.9 tons of methane and NMHCs emitted only in combustion of fuels up to 7 mm diameter in this experimental fire. So even a prescribed fire in a small area is a powerful source of energy and gas emission.

Table 1.--*Fuel characteristics before burning on the site of experimental fire number 1-96.*

Fuel Loads		Other Fuel Characteristics	
Fuel Class	**T/ha**	**Characteristic (units)**	**Amount**
1 hr	14.3	Depth (cm)	19.81
10 hr	7.1	Heat content (J/g)	17,989
100 hr	8.7	Moisture of extinction (%)	32.0
Live herbaceous	0.0	Packing ratio (PR, v/v)	0.0295
Live woody	0.0	PR/Optimum PR	4.90

Table 2.--*Predicted fire behavior under ambient environmental conditions, using the custom fuel model derived on-site. Slope was 0%.*

Fuel Condition		Predicted Fire Behavior			
			Mid-flame wind speed		
Fuel Type	**Fuel Moisture (%)**	**Fire Variable (units)**	**No wind**	**7 km/hr**	**14 km/hr**
1 hr	5.0	Rate of Spread (m/min)	1	4	11
10 hr	12.0	Flame length (m)	1	2	2
100 hr	17.0	Reaction intensity (kW/m^2)	912	912	912
Live herb	100.0	Heat release (kJ/m^2)	9,382	9,382	9,382
Live Woody	100.0	Fireline Intensity (kW/m)	80	631	1782

Table 3.--*Wind speeds and weighted fuel moistures that result in fire behavior within prescription constraints for a backing fire (flame length ≤ 1 m). In cells with an "X", fire would be in prescription for the indicated windspeed, using data from table 2.*

Weighted Dead Fuel Moisture (%)	Windspeed (6 m height/midflame) (km/hr)						
	0.0/0.0	**9.7/4.8**	**19.3/9.7**	**29.0/14.5**	**38.6/19.3**	**48.3/24.1**	**57.9/29.0**
2	X	X	X	X	X	X	X
3	X		X	X	X	X	X
4	X						
5	X						
6	X						
7	X						

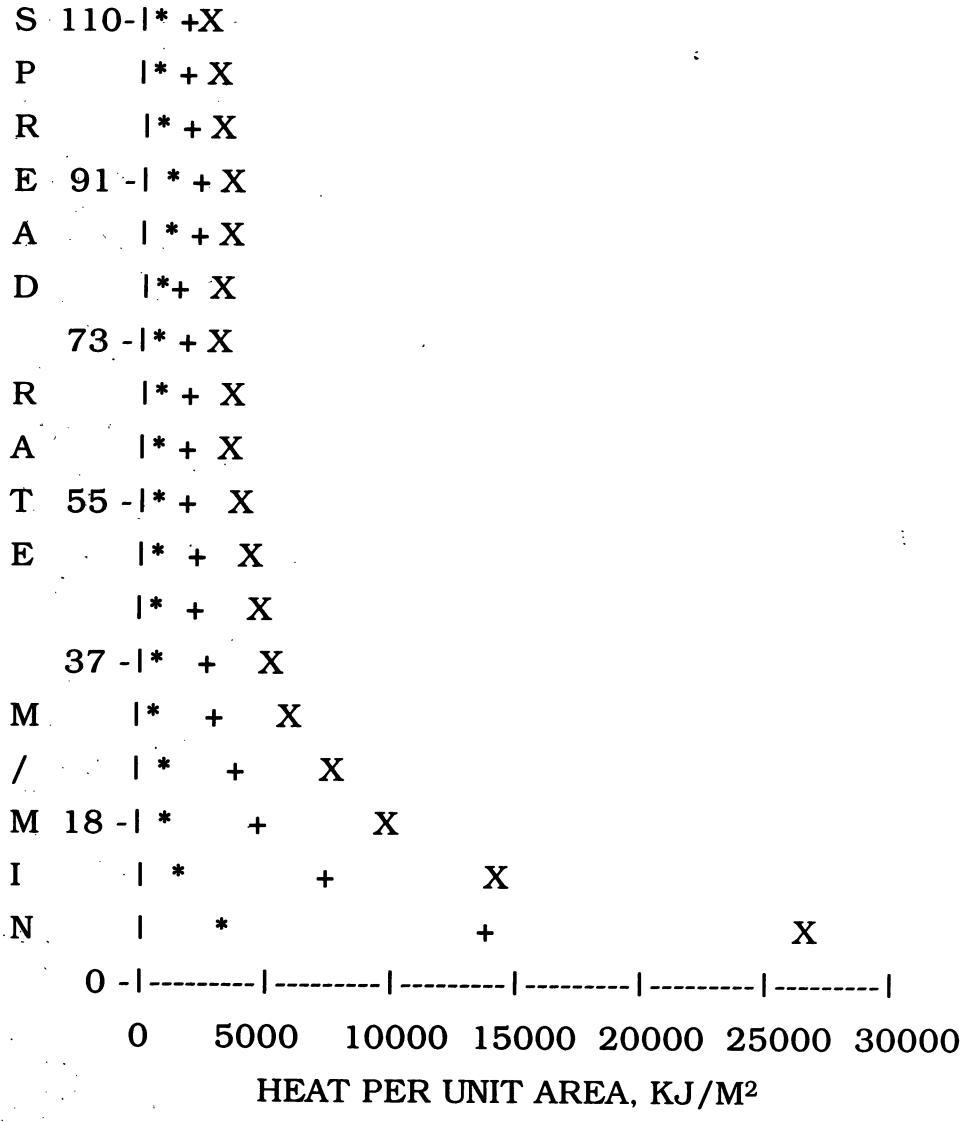

Figure 1.--*Predicted relationships between heat release per unit area and spread rate for three fuel models. Fuel model 13 (slash/heavy fuel loads)--X; fuel model 11 (slash/low fuel loads)--*; custom fuel model 14--+ .*

Natural regeneration after the prescribed fire was successful in the burned area of experimental fire number 1/96. The burned area of experimental fire number 2/96 was planted in fall 1996.

In 1997 understory burning experiments were begun in pine stands (*Pinus sylvestris*) to study the use of prescribed fire for thinning of stands. Low intensity surface fires with flame heights up to 1 m were used for this purpose.

FUTURE PLANS

We will continue prescribed burn experiments in dark coniferous forest clearcuts and begin to study the use of prescribed fires in light coniferous forest clearcuts. Also understory burning in pine stands will be a focus of our research.

CONCLUDING REMARKS

Prescribed fires as well as wildland fires are powerful sources of energy and gas emissions, and if they are included in a forestry practice in boreal forests of Siberia it appears that they will make some contribution to global change. BEHAVE will be a useful tool for planning future IGBP related forest fire experiments on the Siberian IGBP transect and in the introduction of prescribed fires into the forestry practices of Russia.

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Forest Management

DEMONSTRATING SUSTAINABLE FORESTRY IN CENTRAL SIBERIA: A RUSSIAN - AMERICAN PARTNERSHIP

J.C. Brissette[1], V.P. Cherkashin[2], R.M. Babintseva[3], V.V. Ivanov[4], A.J.R. Gillespie[5]

ABSTRACT.--A USDA Forest Service team was established in 1993 to work with Russian counterparts on a Forests for the Future Initiative in the Krasnoyarsk Krai of central Siberia. The goal of the program is to establish pilot projects to demonstrate the principles of forest conservation and sustainable development. One phase of the project in Krasnoyarsk is to provide geographic information system (GIS) capability at the field level. The other phase is to establish demonstrations of sustainable forestry in the southern taiga. This paper reports on our progress so far and our plans for the future of the project, with a focus on the role of research.

INTRODUCTION

In 1993, a team of scientists and managers from the USDA Forest Service's Northeastern Research Station and the Eastern Region was established to work with Russian counterparts on a Forests for the Future Initiative (FFI) in the Krasnoyarsk Krai (Region) of central Siberia. The goal of the FFI program is to establish pilot projects to demonstrate the principles of forest conservation and sustainable development. The first phase of the project in Krasnoyarsk is to provide geographic information system (GIS) capability at the field level in Siberia. Begun in 1995, the second phase will establish demonstrations of sustainable forestry practices in the southern taiga. An initial report of the activities of this Russian - American partnership was prepared by Vaganov *et al.* (1997). This paper will report on our progress and plans for the future, with an emphasis on the role of research in the project.

Currently, members of the American team include scientists from the Northeastern Research Station and managers from Region 1, Region 5, and the Washington Office of the Forest Service. Russian team members include scientists of the Sukachev Institute of the Russian Academy of Sciences, managers from a number of units of the Federal Forest Service of Russia, and the directors of a state-owned logging and sawmilling enterprise.

Members of the American team visited Siberia in 1994 and 1995 to become familiar with the needs for GIS technology, the ecology of the region, and the state of research in the area. We returned to Siberia in 1996 to witness harvest operations in the area. Also in 1995, 1996, and 1997 delegations of Russian researchers, managers, and industrialists visited northern New England and the Lake States to view operational use of GIS, our long-term research and operations on national forest and industrial forest lands, and some visited California and Montana to learn about prescribed burning.

AN OVERVIEW OF KRASNOYARSK KRAI

Krasnoyarsk Krai is in the Asian part of the Russian Federation and occupies 233 million ha or 13.6 percent of the country's territory. The krai lies between 51° N and 81° N, and 78° E and 113° E, and includes the geographic center of Russia. Its southern boundary is the Sayan Mountains and its northern, the Arctic Ocean. Elevation ranges from 100 m to 3,000 m above sea level. Its population is about 3 million people, most living in cities.

The Yenisei River, one of the world's largest with a total length of more than 4,000 km, flows through Krasnoyarsk Krai. It starts in the Sayan Mountains and flows north to the Kara Sea, dividing east and west Siberia. The two largest hydro-electric stations in Eurasia are on the Yenisei. With its many tributaries, the Yenisei forms an important transportation network for goods and passengers. One of its tributaries, the Angara, is the outlet of Lake

[1]*Research Forester and Project Leader, USDA Forest Service, Northeastern Research Station, Durham, NH, USA*

[2]*Research Associate, V.N. Sukachev Institute of Forests, Krasnoyarsk, Russia*

[3]*Professor, V.N. Sukachev Institute of Forests, Krasnoyarsk, Russia*

[4]*Senior Research Associate, V.N. Sukachev Institute of Forests, Krasnoyarsk, Russia*

Baikal. Although the rivers are extremely important for transportation within the region, they are of limited value for exporting goods because the outlet is to the Arctic Ocean. The Trans-Siberian Railway crosses the Krai; however, distances to commercial centers are vast. In a direct line it is 3,200 km west to Moscow and 3,000 km east to Vladivostok.

The area within Krasnoyarsk Krai chosen for demonstrating sustainable forestry is in the Predivinsk lespromkhoz (a timber enterprise) within Bolshoya Murta leskhoz (equivalent to a national forest in the United States). Bolshoya Murta is also the name of a village and the surrounding administrative district, similar to a county. Predivinsk is a village of about 5,000 people on the east bank of the Yenisei River, east-northeast of Bolshoya Murta. The people who live in the village are entirely dependent on the timber enterprise for their livelihoods. They work either in the sawmill, as loggers, or in support of those activities.

PROJECT LOCATION

The Bolshoya Murta leskhoz (approximately 57° N, 93° E) is about 450,000 ha in size and 93% forested. The climate is continental with a mean annual temperature of 1.0° C, and maximum and minimum temperatures of 36.9° C and –48.0° C, respectively (Vaganov *et al.* 1997). Precipitation averages 406 mm per year, the growing season averages 146 days, and soils typically freeze to a depth of 172 cm. The area was not covered by the last continental glacier.

The leskhoz is divided by the Yenisei River. The western part of Bolshoya Murta is southern taiga and subtaiga forest-steppe transition. These sites are quite flat. Soils are Podzolic (Alfisol and Spodosol) and Chernozem (Mollisol) (Vaganov *et al.* 1997). The eastern part is mountain taiga forest. The mountains are not large and only moderate slopes are encountered. Soils there are also Podzolic. The area is south of the permafrost zone.

The southern taiga and mountain taiga forests are comprised of "dark coniferous", "light coniferous", and hardwood stands. Except for scattered trees occasionally harvested for local use, these stands have never been logged. However, because of natural disturbance, average stand age is only about 91 years (E. Vaganov, personal communication). Nevertheless, coring of individual trees when we were there indicated some more than 200 years old. Furthermore, many of the stands we visited had the appearance of being uneven-aged. Trees of the forest-steppe transition are mostly birch, *Betula*, and pine, *Pinus*.

Although they are at higher latitude, dark coniferous stands found in Bolshoya Murta are strikingly similar to spruce-fir (*Picea-Abies)* forests of eastern North America. Siberian spruce (*P. obovata)* is similar to red spruce (*P. rubens* Sarg.) so common in Maine and Atlantic Canada, and to white spruce (*P. glauca)* (Moench Voss) which is common across the northern United States and Canada. Siberian fir (*A. sibirica*) is similar to balsam fir (*A. balsamea)* (L.) Mill. Although it is a stone pine (i.e., 5-needle fascicles, short-winged or wingless seeds), Siberian pine (*P. sibirica*) (called cedar by the local people) seems to have an ecological niche much like eastern white pine (*P. strobus* L.) in mixed conifer forests of eastern North America. These are all relatively tolerant species and the stands tend to have multiple cohorts. Natural disturbances include insect epidemics, wind, and fire. After stand-replacing disturbance, dark coniferous stands are followed by the region's only large hardwoods, birch (*B. pendula*) and aspen (*Populus tremula*).

Light coniferous stands of Scotch pine (*Pinus sylvestris L.*) are usually even-aged and appear similar to natural stands of red pine (*P. resinosa* Ait) in the Great Lakes region of North America. With increasing latitude, Siberian larch (*Larix sibirica*) progressively replaces pine in light coniferous stands. However, in the area where we are working, pine is much more common than larch.

PROJECT ACTIVITIES: GIS

The GIS phase of the project is the furthest along and much background about the development of a GIS database for the Bolshoya Murta leskhoz was presented by Vaganov *et al.* (1997). However, research and implementation of GIS to the field continue and some additional important progress was made during the past year. GIS-generated maps, which combine information from older forest cover type maps and maps of soils drainage and productivity, have been compared with conditions in the field and found to be quite accurate. Testing the accuracy of such combined maps is important to ensure their usefulness to managers.

In the Bolshoya Murta leskhoz, natural regeneration is the primary means of renewing both dark and light coniferous stands. Nevertheless, artificial regeneration is also used on some sites. Results of field studies were used to build a GIS map of regeneration effectiveness. This map is being tested on an experimental site to determine whether it can predict amount and viability of tree regeneration. Combining the regeneration potential map with a previously developed map showing methods of cutting will allow us to identify areas where we should protect advanced regeneration, as well as sites where it will be necessary to plant.

A GIS-based subsystem was developed to help managers decide how to harvest any given forest unit. It includes a forest inventory database, standards and parameters of allowable commercial cutting, and an algorithm to recommend the most appropriate harvest method. Preliminary tests of the system have proven effective for determining optimal harvesting operations. A similar subsystem was developed to help determine sites where salvage cutting should be prescribed. These two subsystems, unparalleled in Siberia, are key elements in developing a much more effective approach to achieving practical forest management objectives.

PROJECT ACTIVITIES: ON THE GROUND DEMONSTRATIONS

Silvicultural treatments in the Soviet Union were mandated based on forest type, stand structure, slope, and whether the site had sufficient advanced regeneration of valuable species. The same or similar rules still apply. However, because of this project, we are able to demonstrate a number of innovative approaches that previously may not have been allowed. The use of prescribed fire is a good example.

Natural forest fire is an important disturbance factor that determines the composition and structure of forests throughout the world. However, in many parts of the United States, prescribed fire has only recently been accepted as a forest management tool. Similarly in Russia, fire has traditionally been considered destructive and prescribed fire is still a radical idea. Nevertheless, in 1996, an experimental prescribed fire was used to reduce slash in a recent clearcut in a dark coniferous stand and to prepare the site for planting. The experiment was successful and led to more prescribed fires in 1997. While still on a small scale, these experimental fires may demonstrate the value of prescribed burning for regeneration and maintenance of forest stands.

In the Predivinsk lespromkhoz, tracts of dark coniferous forest are clearcut with little concern for regeneration. Harvesting is in 50-ha units using tracked feller-skidders. See Folkema and Holowacz (1985) for detailed descriptions of Russian logging equipment and practices. This equipment has several limitations: the operator can only harvest trees to the left; the boom reach is only 5 m; and trees cannot be lifted once cut--so tops are dragged across the site before the butt of the tree is placed in a bunk at the back of the machine. Furthermore, skidding progresses in 5-m wide strips across the harvest unit with each skid trail being the previously felled strip. Thus, the whole unit is covered by skidding and all advanced regeneration is destroyed. Where artificial regeneration is an option, such harvesting might be acceptable. However, there is neither nursery capacity nor infrastructure to ensure that all harvest units get planted, and there is no funding available to release established conifer seedlings from hardwood competition.

After viewing current cutting methods, an experimental harvest that could protect some advanced regeneration was proposed for a dark coniferous stand scheduled to be harvested. The experiment was designed to use existing mechanical harvesting equipment. In this demonstration harvest, 5-m wide skid trails were cut in the conventional way, but they were alternated with 5-m strips where trees were cut and removed while providing some protection for advanced regeneration. The trial was not fully successful but it did demonstrate the willingness of the director of the lespromkhoz to work with our team on problems of mutual concern. Over the longer term, we hope to encourage harvesting equipment manufacturers to demonstrate their machines in the region. Under similar terrain and forest conditions in the industrial spruce-fir forest of eastern North America, state-of-the-art feller-bunchers operating on designated skid trails do a good job of protecting advanced regeneration.

ACCOMPLISHMENTS AND PLANS

Since its beginning in 1993, this project can claim a number of accomplishments. Communications among all the partners are improving, and there is a strong sense that what we are trying to accomplish is extremely important. Undoubtedly, the greatest accomplishment has been bringing together the various agencies responsible for inventorying, protecting, managing, and utilizing the forests to work toward a common goal. Phase One, developing local capacity to build, maintain, and use GIS technology is operational and nearly complete. Additional research in this area will refine existing methods and the scientists involved will continue to work with forest managers to effectively use the technology. Managers of the forests around Bolshoya Murta have been authorized to experiment with treatments other than those normally prescribed by rule; the prescribed burn in 1996 is the first attempt at such innovation. This rule change gives us the opportunity to continue research on prescribed fire on an operational scale. Exchange visits between Russian and American partners have improved our understanding of each other and of our respective forests. These exchanges have shaped our annual work plans, infusing this project with a high degree of flexibility, and we hope that such exchanges can continue.

We have a number of objectives for the future. Within the next year, GIS technology should become available to the lespromkhoz which will enhance their ability to locate and schedule harvest units effectively. Under Phase Two, we have two immediate objectives. One is to plan and conduct a site preparation underburn in a light coniferous stand to demonstrate the potential of fire to enhance natural regeneration of pine. The other is to demonstrate economically-sustainable methods of harvesting in dark coniferous stands while protecting advanced regeneration.

We hope that continued successful demonstrations of GIS technology and on-the-ground techniques will lead to a willingness to consider a longer approach and integrated forest management planning. We would like to identify some areas well before they are scheduled for harvest and prepare a thorough silvicultural prescription, something we have not yet accomplished. The evaluation would consider the state of advanced regeneration and recommend appropriate prescribed fire and cutting methods to control the composition and density of the future stand. In conclusion, the Russian-American partnership has tangible accomplishments and the vision needed to help sustain the forests of Siberia for the future.

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TREE REGENERATION DYNAMICS OF LOWLAND BLACK SPRUCE STANDS FOLLOWING CROWN FIRE AND CLEARCUT LOGGING

G.G. Wang and C.A. Harrison[1]

ABSTRACT.--To determine the differences in tree regeneration after fire and logging, lowland black spruce stands burned (by crown fire) and logged (by clearcut) 6 to 13 years ago in southeastern Manitoba were investigated. Black spruce regeneration was the most abundant on both burned and logged sites, with jack pine as the main associate on burned sites and tamarack and trembling aspen as main associates on logged sites. Other tree species were rarely found on either burned or logged sites. A continous recruitment pattern was observed on both burned and logged sites for black spruce, with a higher number of black spruce seedlings recruited each year on burned sites. As a result, seedling stocking and density was significantly higher on burned sites. When compared to the regeneration standard of Manitoba, all burned sites were considered satisfactory while two of the nine logged sites were unsatisfactory. Advanced regeneration was found insignificant to post-disturbance (either fire or logging) forest recovery. The total height and the 1995 height growth of black spruce seedlings were significantly higher on logged sites, which was caused, at least in part, by the inherent difference in site quality (measured by site index) between burned and logged sites. Relative annual height growth (expressed as % of mean annual height growth) was higher at age ≤ 5 and lower at age ≥ 6 on burned sites when compared to logged sites where it increased linearly with age. The ratio of the 1995 height growth of logged sites to that of burned sites decreased from 165 % to 135 % with the increase of total height, and stabilized at 135 % when total height approached 1 m. These differences in early height growth pattern suggested that black spruce seedlings of burned sites may have encountered more competition from understory plants.

INTRODUCTION

Maintaining the natural diversity of forests while at the same time allowing timber to be harvested is an important challenge in the management of Canadian boreal forests. In searching for a balance between these two interests, it is critical to understand the way in which forests recover after natural disturbances and after timber harvesting. Boreal forests in North America have been profoundly influenced by natural disturbance regimes, predominately fire (Heinselman 1981, Rowe 1983, Bonan and Shugart 1989). Through its control on forest regeneration and succession, fire, interacting with life history traits of tree species and local site conditions, creates and maintains the spatial mosaic of forest patches of different size, age, and type displayed in natural boreal forest landscape. Understanding the effect of fire on forest regeneration and succession thus holds the key to the understanding of the formation of the natural boreal forest landscape. With growing economic demand, increasing areas of southern boreal forests are facing timber harvesting and subsequent silviculture treatments. Increasing logging and silvicultural activities, along with attempt in fire suppression, has raised serious concerns about possible changes in natural diversity and productivity of boreal forests (Thompson and Welsh 1993).

Black spruce [*Picea mariana* (Mill.) B.S.P.] dominates much of the North American boreal forests and is a major source for pulp and paper industry in Canada. Within the Manitoba Model Forest area, black spruce has been subjected to timber utilization for more than 70 years. A significant portion of black spruce stands harvested in the area were growing on lowland organic soils. It is generally believed that lowland black spruce sites usually regenerate naturally to a black spruce-dominant stands after fire (Dix and Swan 1971, Carleton and Maycock 1978, Heinselman 1981) and logging (Stanek 1975, Jeglum 1981). However, previous observations made in the study area suggested that natural regeneration of black spruce on some lowland sites following logging may fail to meet the provincial regeneration standard or to restore into a fully productive forest on some occasions (Synthern Resource Ltd. 1990).

The main objective of the study was to determine if fire and logging significantly differed in their effects

[1] *Department of Biology, The University of Winnipeg, Winnipeg, Manitoba, Canada R3B 2E9. E-mail: wang@uwinnipeg.ca.*

on tree regeneration. Specifically, we examined (1) tree species composition, density, age and height growth, (2) recruitment pattern of black spruce, and (3) height growth pattern of black spruce on burned and logged lowland sites that previously supported black spruce stands. The study is a part of a larger project designed to investigate vegetation, soil and seedbank dynamics of lowland black spruce stands following a crown fire and a clear-cut logging.

MATERIALS AND METHODS

Study Area

The study area is located in the forest management unit 31 of southeastern Manitoba, extending approximately from 50°30' to 51°08' N and from 95°10' to 96°30' W. Winters are cold (with mean temperature of -19.2 °C in January) and relatively dry while summer are warm (with mean temperature of 19.0 °C in July). Average annual rainfall is about 400 mm with 270 mm in growing season (May to August). Average snowfall is about 120 cm. The main frost-free period is about 100 days. The study area has a relatively simple topography, with little change in plateau. The Canadian Shield is near the surface throughout the study area. Rock outcrops typically lack of mineral soils are frequently encountered. Depressions between rock outcrops contain mineral or organic soils depending on drainage. Important soil orders include Dystric Brunisols, Grey Luvisols, Gleysols, Fibrisols, and Mesisols (Canada Soil Inventory 1989). According to the data given in the 1986 Forest Resource Inventory, black spruce, jack pine (*Pinus banksiana* Lamb) and trembling aspen (*Populus tremuloides* Michx) are either the primary or secondary species in 88 % of the stands. Black spruce appeared as the primary or secondary species in 80 % of these stands. Jack pine/black spruce stands were dominant on uplands except on fine-textured soils where aspen mixedwoods are common. Black spruce stands frequently dominate on lowlands with a well developed layer of feathermoss or sphagnum and a poorly developed layer of vascular plants. The term "lowland" in this study refers specifically to those sites with organic soils as defined by CSSC (1978). Lowland black spruce stands are among the most commonly encountered forest types in the study area as well as in Manitoba's boreal forests. According to Manitoba's forest ecosystem classification (Zoladeski *et al.* 1995), lowland black spruce stands consist of vegetation types of V30, V31, V32, and V33 and soil types of S12F and S12S.

Site Selection and Sampling Design

Using available information (cover type map, topography map, fire and logging record) and through field reconnaissance, lowland black spruce stands that were burned or logged in past 20 years were identified on the map within the study area. All burned sites selected in the study were burned by crown fires with 100 % mortality of mature trees of the pre-fire stands. All logged sites selected in the study were disturbed by small patch clearcut (typically < 5 ha). As a result, six burned sites ranged from 7 to 13 years old and nine logged sites ranged from 6 to 12 years old were used in the study. Within each selected site, a quick reconnaissance was conducted to examine the heterogeneity in soil and vegetation conditions. One or more 10x10 m plots were sampled with each site depending on its size and within-site variation. Each plot was oriented in the north-south direction, representing an area with homogenous soil and vegetation characteristics. Within each plot, five 2x2 m quadrats were sampled in the four corners and in the center.

Data Collection

Individual seedling of each tree species were counted on each plot (for seedling ≥ 2 m tall) and on each quadrat (for seedling < 2 m tall). The total height, age, and 1995 leader growth were measured for every seedling. The height up to the end of the 1995 growing season was measured as total height because the 1996 growing season was incomplete at the time of sampling. Age determination for each seedling was accomplished by counting the number of terminal bud scars from the base of the seedling to the end of the 1995 growing season. Using the terminal bud scars allowed for accurate aging in the absence of whorled branches. If any difficulty encountered with this method, destructive sampling was used for age determination. The distance between the terminal bud scars representing the end of the 1994 growing season and the end of the 1995 growing season was measured. This distance represents the amount of growth that occurred during the growing season of 1995 (i.e., the 1995 height growth).

Table 1. -- *Summary of selected site characteristics according to disturbance type.*

Disturbance type		**Fire**	**Logging**
# of sites		**6**	**9**
Site age (year)	Minimum	7	6
	Maximum	13	12
	Mean	9.3	9.3
Groundwater table (cm)	Minimum	10	2
	Maximum	65	85
	Mean	28	29
Major rooting zone depth	Minimum	5	5
(cm):	Maximum	27	34
	Mean	19	20
Depth of organic layer	Minimum	70	45
(cm):	Maximum	>150	>150
Soil moisture regime:		Wet/very wet	Wet/very wet
Soil nutrient regime		Very poor to medium	Very poor to medium
Soil taxon		Fibrisol/Mesisol	Fibrisol/Mesisol

*Field identification of soil nutrient regime did not take nutrient input due to groundwater influence into consideration.

A soil pit was dug near the center of each plot to classify and describe soils. Soils were classified following CSSC (1978). Important soil variables recorded in the field including depth to ground water table and/or seepage, depth of organic layer, depth of the major rooting zone, and soil taxon. Soil moisture and nutrient regimes of each plot were determined using a procedure modified from Wang *et al.* (1994). A brief summary of these site characteristics is given in table 1.

Data Analysis

The presence of a species on each disturbance type was calculated as a percent of total plots in which the species was found. The stocking of a species on each study site was calculated as a percent of total quadrats in which at least one individual of the species was found. Density of a species was expressed as number of individuals per hectare. A simple t-test was used to compare seedling stocking, density, age, total height, and the 1995 leader growth between burned and logged sites. A separate variance t-test was applied whenever group variances were not equal (Wilkinson 1990).

Each black spruce seedling sampled was grouped according to the year since disturbance, which was identified based on its age and the year when fire or logging occurred. Stocking, density, and accumulated density were compared on every year since the disturbance. The success of black spruce regeneration on each site was also assessed using the provincial standard (Delaney 1995).

The relationships between total height and age and between the 1995 height growth and the total height were quantified, separately, for burned and logged sites. To find the best models, scatterplots with LOWESS smoothing were constructed (Wilkinson 1990). Logarithm transformations of dependent variables were made to improve the homogeneity of their variances. As a result, the following two models were selected.

[1] $$\ln(HT) = a(1-e^{bA})^c$$

[2] $$\ln(GTH) = a(1-e^{bHT})^c$$

where *ln* = natural logarithm, *e* = the base of natural logarithm, *HT* = height (cm), *GTH* = the 1995 height growth (cm), and *A* = age (years). These nonlinear models were fitted using the Quasi-Newton procedure (Wilkinson 1990).

To examine growth pattern, relative annual height growth (%) were calculated using Eq. [3] and plotted against age for both burned and logged sites:

[3] $$RAG(\%) = \frac{CAG}{MAG} x100$$

where RAG, CAG, and MAG are relative (%), current (cm), and mean (cm) annual height growth, respectively. CAG and MAG were derived from Eq. [1]. Similarly, the ratio of the 1995 height growth of logged sites to that of burned sites was also calculated using values derived from Eq. [2] and plotted against seedling height.

RESULTS

Current Status of Tree Regeneration

Four species were found among the six burned sites: black spruce on six sites, jack pine on five sites, and tamarack [*Larix laricina* (du Roi) K. Koch] and trembling aspen on one site. Six species were found among the nine logged sites: black spruce on nine sites, tamarack on seven sites, trembling aspen on eight sites, and paper birch, white spruce and balsam fir on one site.

On burned sites, black spruce and jack pine had a high stocking levels (88.3 % and 46.7 %, respectively); tamarack and trembling aspen were found only in one quadrat of one site. On logged sites, black spruce had the highest stocking (83.8 %), followed by tamarack and trembling aspen (21.3 %); paper birch (*Betula papyrifera* Marsh.), white spruce [*Picea glauca* (Moench) Voss], and balsam fir [*Abies balsamea* (L.) Mill] were found only on one quadrat of one site. No significant difference ($p = 0.581$) in the stocking level of black spruce was found between burned and logged sites.

On burned sites, there were 43,150 stem/ha of black spruce and 3,168 stem/ha of jack pine. On logged sites, black spruce averaged 14,450 stem/ha, tamarack 900 stem/ha, and trembling aspen 1,668 stem/ha. Despite large variations among sites within each disturbance type, black spruce density was found to be significantly different ($p = 0.034$) between burned and logged sites.

Similar ages were observed for black spruce, jack pine and trembling aspen seedlings regenerated on both burned and logged sites (table 2). However, tamarack appeared younger than black spruce and trembling aspen on logged sites. There was no significant difference ($p = 0.637$) in the age of black spruce seedlings between burned and logged sites.

Jack pine (on burned sites) and trembling aspen (on logged sites) were taller than black spruce; tamarack (on logged sites), despite of its younger age, had a similar height as black spruce (table 2). Similarly, jack pine, tamarack, and trembling aspen had higher 1995 height growth than black spruce (table 2). Despite of the similar age, total height and the 1995 height growth of black spruce seedlings on burned sites were significantly ($p = 0.012$ and 0.013, respectively) lower than those on the logged sites.

Table 2.--*Means (with standard error in parentheses) of age, height, and the 1995 leader growth of major tree species regenerated 6-13 years after crown fire and clearcut logging.*

Disturbance	Species* Sb	Pj	Ta	At
Age (year)				
Fire	4.8	5.3		
	(1.8)	(1.2)		
Logging	4.7		3.4	4.8
	(1.4)		(1.1)	(1.8)
Height (cm):				
Fire	20.0	74.4		
	(9.1)	(5.9)		
Logging	8.2		11.7	7.7
	(3.2)		(6.0)	(6.0)

*Symbols for species are: Sb-black spruce; Pj-jack pine; Ta-tamarack; At-trembling aspen.

Recruitment Pattern of Black Spruce After Fire and Logging

Most black spruce seedlings were established after fire and logging. Advance regeneration was only found on three logged sites (500 stem/ha) and one of the burned sites (9,000 stem/ha).

Black spruce seedlings regenerated after fire and logging demonstrated a continuous recruitment pattern (table 3). There were higher % of burned sites with black spruce recruitment almost in any year after the disturbance when compared to logging sites. Every studied site had black spruce regeneration at year four since fire and at year six since logging (table 3). The recruitment level of black spruce seedlings on both burned and logged sites increased with time after disturbance, at least until year seven (table 3). More seedlings were recruited each year on burned sites compared to that on logged sites. Significant differences ($p = 0.047$) were found at years five. Because of continuous recruitment, the density of black spruce seedlings on both burned and logged sites accumulated rapidly (table 3). As a result of a higher recruitment rate, density of black spruce seedling on burned sites was consistently higher than that on logged sites when measured at any year since the disturbance. Significant differences ($p < 0.05$) were found at years five, six and seven.

When compared to the provincial stocking criteria (minimum of 55 % stocking for black spruce at year seven), all burned sites were considered successful while two of the nine logged sites were considered unsuccessful (with stocking level at 40 % and 50 % at year seven). Significant difference in black spruce stocking at year seven was found between burned (88 %) and logged (70 %) sites ($p = 0.026$). On average, it took only 4 and 5 years, respectively for burned and logged sites, to reach the suggested minimum density of 2,746 stem/ha (Delaney 1995). When individual sites were examined, it took only 5 years for all the six burned sites to reach the minimum density level while it took 9 years for all the nine logged sites to reach the minimum density level. However, only one logged site did not reach the minimum density level at year seven.

Table 3.--*Black spruce recruitment at different years after fire and logging.*

	Year after disturbance								
	1	2	3	4	5	6	7	8	9
Presence (%) and accumulated presence (%) (in parenthesis)									
Fire	25	33	67	100	100	100	100	90	50
	(25)	(41)	(67)	(100)	(100)	(100)	(100)	(100)	(100)
Logging	0	31	50	75	50	75	79	79	69
	(0)	(31)	(50)	(75)	(81)	(100)	(100)	(100)	(100)
Mean and standard deviation of density (stem/ha):									
Fire	167	208	1433	3083	7391	6291	6416	6670	5050
	(258)	(188)	(1616)	(2166)	(5974)	(5737)	(6129)	(9224)	(9132)
Logging	0	297	833	1083	1037	1056	2250	1681	2157
	(0)	(455)	(1145)	(1159)	(1602)	(1217)	(1866)	(1574)	(2611)
P value*	Na**	0.61	0.453	0.077	0.047	0.076	0.161	0.295	0.525
Mean and standard deviation of accumulated density (stem/ha):									
Fire	167	375	1958	5208	12583	188751	25291	29950	35000
	(258)	(262)	(1600)	(3152)	(8323)	(10701)	(15358)	(24343)	(32595)
Logging	0	297	1129	2212	3250	4306	6781	8458	11261
	(0)	(455)	(1582)	(2408)	(3672)	(4426)	(4302)	(4890)	(5474)
P value*	Na**	0.678	0.345	0.080	0.040	0.019	0.031	0.120	0.179

*Separate variance T test

**Insufficient data for test; na-not available

Height Growth Pattern of Black Spruce Regeneration After Fire and Logging

Age accounted approximately about 50 % (Equation [4]) and 69 % (Equation [5]) of variations in seedling height, respectively for burned and logged sites:

[4] $$\ln(HT) = 4.053(1 - e^{-0.192A})^{0.795}$$

Corrected $R^2 = 0.50$ RME = 0.399
n = 1036p = 0.000

[5] $$\ln(HT) = 6.557(1 - e^{-0.092A})^{0.641}$$

Corrected $R^2 = 0.69$ RME = 0.322
n = 435 p = 0.000

where ln = natural logarithm, e = the base of natural logarithm, HT = height (cm), A = age (year), and RME = root mean square error. The natural logarithm of height was used to improve homogeneity of variance of height at different age.

Although seedling height increased with age on both burned and logged site, seedlings established on logged sites were taller at any age (fig. 1, top). On burned sites, relative annual height growth decreased from year 1 to year 2, increased at relative fast rate from year 2 to year 5, slowed down considerably from year 5 to year 7, and declined somewhat from year 7 to year 8 (fig. 2, top). On logged sites, relative annual height growth maintained at the similar level at initial two years and then increased linearly with age (fig. 2). At age ≤ 5 years, relative height growth on burned sites was higher than that on logged sites. The reverse was true when age ≥ 6.

Total seedling height explained approximately 31 % (Equation [6]) and 30 % (Equation [7]) of the total variations in the 1995 height growth, respectively for burned and logged sites:

[6] $$\ln(GTH) = 2.014(1 - e^{-0.0308HT})^{0.633}$$

Corrected $R^2 = 0.31$ RME = 0.392
n = 1052P = 0.000

[7] $$\ln(GTH) = 2.326(1 - e^{-0.0259HT})^{0.368}$$

Corrected $R^2 = 0.30$ RME = 0.386
n = 458 P = 0.000

where GTH = the 1995 leader growth and all others are previously defined.

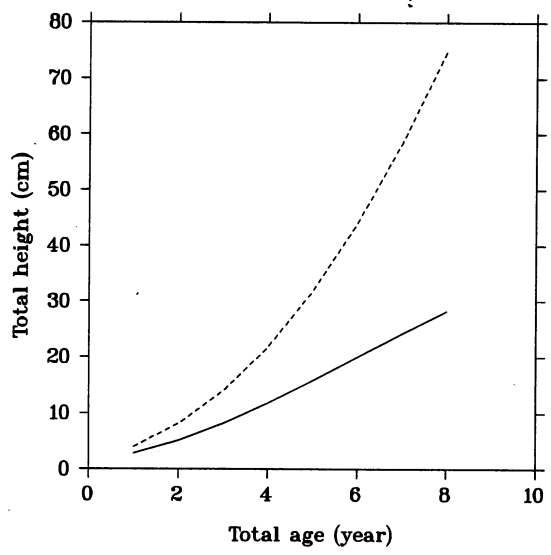
Although increased with the total height on both burned and logged site, the 1995 height growth on logged sites was higher than that on burned sites for any given seedling height (fig. 1, bottom). The 1995 height growth ratio between the two disturbance type were found approximately between 135 % to 165 % (fig. 2, bottom). With the increase of seedling height, the ratio decreased at higher rate initially and then leveled off when seedling height approached 1 m, where the ratio stablized at about 135 % (fig. 2, bottom).

DISCUSSIONS

Species Composition

Our result indicated that a direct succession, in which the same species (i.e., black spruce) re-establish itself as the dominant, was evident on lowland black spruce stands after either fire or logging disturbance. This is in agreement with previous studies on boreal forest succession (Dix and Swan 1971, Carleton and Maycock 1978, Heinselman 1981). Although jack pine (on burned sites), and trembling aspen and tamarack (on logged sites) occurred as common associates, they did not achieve dominance in any burned and logged site. The presence of jack pine on five of the six burned sites and non-existence on any logged site can be explained by the reproduction strategy of jack pine. Fire opens the serotious cone of jack pine, therefore provides seeds to burned sites. The relative higher presence of tamarack on logged sites but not on burned sites may be due to the effect of fire disturbance. Crown fires destroy the seed source of tamarack, therefore reduce its presence on post-fire communities. However, it was doubtful that fire had killed the root of trembling aspen, thus caused the lower presence of trembling aspen suckers on burned sites. Rowe (1983) reported that the death of root system due to fire only occurred on upland sites with extremely shallow soils.

Because black spruce, jack pine, and trembling aspen are well adapted to fire disturbance, they re-established themselves quickly after fire (Burns and Honkala 1990, Sims *et al.* 1990). The lack of significant differences in the age of black spruce seedlings between burned and logged sites suggested that black spruce also regenerated quickly after logging. The relatively late establishment of tamarack on logged sites, which may not be biologically significant due to small difference (1-2 years late) despite of statistical significance, is difficult to explain. One reason could be that some logged sites were lacking *in situ* seed source of tamarack, thus delayed tamarack regeneration because of its dependence on seed inputs from outside sources.



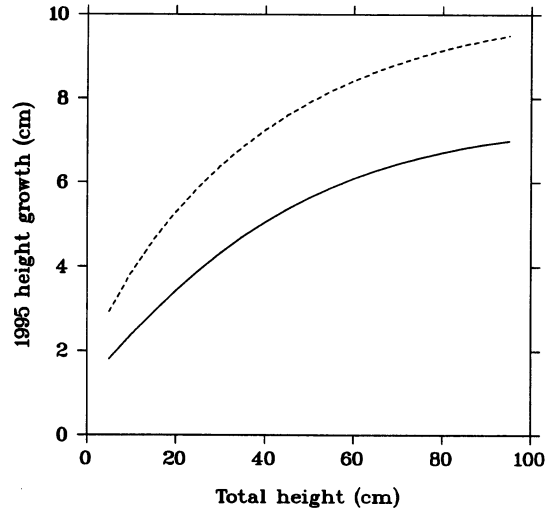


Figure 1.--*TOP: The relationships between total height and total age of black spruce seedlings on burned (solid line) and logged (dashed line) sites, which are defined by Eqs. [4] and [5], respectively. BOTTOM: The relationships between the 1995 height growth and the total of black spruce seedlings on burned(solid line) and logged (dashed line) sites, which are defined by Eqs. [6] and [7], respectively.*

Figure 2.--*TOP: The relationships between relative annual growth (%) and total age of black spruce seedlings on burned (solid line) and logged (dashed line) sites, which are derived from Eqs. [4] and [5] respectively. BOTTOM: The relationship between the ratio of the 1995 leader growth on burned sites to that on logged sites and the total height of black spruce seedlings, which are derived from Eqs. [6] and [7], respectively.*

Recruitment Pattern of Black Spruce

Despite previous studies found that advanced regeneration was important on post-logging sites (Brumelis and Carleton 1988), we found that advanced regeneration was not an important component on either burned or logged sites. Our field observations on advanced regeneration of mature stands suggested that seedlings (< 2 m) of black spruce or any other tree species were low in stocking (36.7 %) and density (1038 stem/ha), and generally poor in vigor.

The continuous recruitment pattern demonstrated on burned and logged sites was further supported by the wide age distribution of black spruce trees in the canopy layer of mature (both logging- and fire-origin) stands (Wang 1996, unpublished data). It is also in agreement with that poor sites take longer time for tree species to fill available niche (Oliver and Larson 1996). Considering the dynamics of seed supply following disturbances, however, a continuous rather than a pulse pattern of black spruce recruitment, especially following fire disturbance, is somewhat surprising. For logged sites, a continuous recruitment pattern may be explained by the fact that all logged sites are located in small-patch clearcuts frequently with mature black spruce trees/stands nearby. Therefore a continuous seed supply can be maintained over time by *in situ* seeds from logging slash and seed dispersal from nearby. However, most burned sites were destroyed by an extensive fire without any living trees/stands nearby. The continous seed supply could not be attributed to seed dispersal from an outside source rather from an *in situ* source. However, black spruce seeds can only maintain its viability on forest floor for < 16 months (Zasada *et al.* 1992), and very few viable black spruce seeds were found in soil seed bank (Staniforth 1996, unpublished data). If soil seedbank is not the *in situ* source, this continous seed supply must come from those black spruces snags on the burned sites. However, a study on seed release from black spruce cones in logging slash indicated that 65 %-90 % of the total viable seeds was released within the first year and ≥ 90 % was released within the first 2 years (Fleming and Mossa 1996).

This result indicates only < 10 % total viable seeds left in the cone after 2 years. Considering the total viable seeds are estimated in a few million per hectare (e.g., Haavisto 1975, Fleming and Mossa 1996), 1 % seed release each year is still at least an order of magnitude greater than the number of recruited seedlings. It is not clear how long black spruce snags can remain as an effective seed supply.

Growth and Growth Pattern of Black Spruce

The consistent poorer growth performance of black spruce seedling on burned sites when compared with logged sites was not expected. To what degree this can be attributed to the effect of disturbance type is not clear. A number of factors that are not related with the type of disturbances may greatly affect seedling growth. To attribute any difference in height growth to the effect of disturbance type, we must be sure that there is no significant difference in site quality between burned and logged sites. This is extremely important since we used naturally burned and logged sites without any control on the similarity of 'experimental units' and completely lack of any mechanism that insures a random assignment of 'treatments' to 'experiment units'. As all burned and logged sites are accessible (i.e., in a walking distance) from logging roads. It is likely that past logging operations may have occurred on those sites with higher productivity. Therefore those sites with lower productivity were left and then burned by wildfire. Ideally, this assumption should be verified by data collected from pre-disturbance stands or some measurements of ecological site quality (e.g., site index, soil nutrient status). Because data from pre-disturbance stands were lacking, an approximate approach was applied to assessing if there was an inherent difference in site quality between burned and logged sites. We assume that the mature stands originated from fire and logging have similar site quality to the burned and logged sites, respectively. Site index calculated from 24 fire-origin and 15 logging-origin lowland black spruce stands sampled for a growth and yield study (Wang 1997) was used as an indicator of site quality.

Table 4.--*Means and standard deviations of site index (m @ 20 years breast height age) and age to breast height for the mature (fire origin) and young (logging-origin) stands.*

	Site index (m)	Age to bh (years)
Mature stands	3.8	18.8
	(1.3)	(7.4)
Young stands	7.1	11.0
	(1.0)	(4.8)
P value (separate variance T test)	0.000	0.000

When measured as the mean height of dominant black spruce trees at breast height age (bha) of 20 years (i.e., SI_{20}), significant difference ($p = 0.000$) in site index was found between the fire-origin and logging-origin stands (table 4). The ratio of SI_{20} of the logging-origin stands to SI_{20} of the fire-origin stands is 187 %. The number of years required to reach breast height (1.3 m) was also found significantly different ($p = 0.000$) between the fire-origin and logging-origin stands (table 4). Before reaching breast height, the average height growth was 12.3 cm/year for the logging-origin stands and 7.2 cm/year for the fire-origin stands, with a ratio of 171 %. Both ratios are higher when compared to the ratio of the 1995 height growth of logged sites to that of burned sites (135 % -165 %), suggesting more competition was encountered by those seedlings that did not become dominant trees in a mature forest stand. Because the number of year to breast height was found significantly correlated with SI_{20} ($r = 0.66$, $p = 0.000$), the differences in the total height and the 1995 height growth between burned and logged sites were resulted, at least in part, from this inherent difference in site quality. Attempting to remove the effect of different site qualities on growth, we examined relative annual height growth and the 1995 height growth ratio. The different patterns of relative annual height growth displayed on burned and logged sites are likely due to the differences in competition from understory vegetation and in temporary changes in soil nutrient regimes. It appeared that black spruce seedlings on burned sites were benefited from initial nutrient pulse (Kimmins 1987) and negatively affected by increasing brush competition. The consistently higher ratio (> 135 %) of the 1995 height growth (logged sites vs burned sites) indicated that the 1995 height growth on logged sites was faster than that on burned sites for any size (height) of seedlings. This could be explained by the inherent difference in site quality as discussed above. However, the ratio changed with the increase of seedling height, rather than remained stable. This suggested some other factors, in addition to the inherent difference in site quality, also affected the 1995 height growth. Apparently, the influence of these factors depended on seedling height. For short seedlings, these factors had more negative influence on seedling established on burned sites when compared to logged sites (as reflected by a higher ratio). As seedling become taller, the differential influence of these factors become smaller, and eventually stabilized at 135 % when seedling height approach 1 m tall. One of these factors may be the level of sunlight, which is a good surrogate of the competition from understory vegetation. Black spruce seedlings require > 60 % of full sunlight to achieve optimal growth (Jobidon 1994). Competition from understory vegetation affect the light level received by each seedling depending on its height, which, in turn, affect seedling height growth. Shrub cover on burned sites (72.4 %) was high compared to logged sites (47.0 %), and the dominant shrub species on both burned and logged sites was *Ledum groenlandicum* (usually < 60 cm tall).

CONCLUSIONS

The composition of tree species regenerated after fire and logging on lowland black spruce stands support a direct succession hypothesis: black spruce re-established itself as the dominant species of future stands, and no other tree species achieved dominance on any burned and logged site.

Burned sites had significantly higher black spruce density compared to logged sites. According to the provincial regeneration standard, all burned sites were considered successful in black spruce regeneration while two of the nine logged sites did not meet stocking criteria, and one of the two did not meet the recommended level of density at year seven.

Advance regeneration was not a significant part of both post-fire and post-logging stand establishment. Black spruce demonstrated a continous rather than pulse regeneration pattern on both burned and logged sites. The recruitment level of burned sites were consistently higher than that on logged sites at the years examined.

Despite of similar age, black spruce seedlings on burned sites were shorter in their total height and lower in their 1995 height growth when compared to those on logged sites. These growth differences, however, may not be entirely attribute to the effect of disturbance type because there was evidence suggesting that logged sites may have higher site quality (measured by site index) compared to burn sites.

Growth pattern analysis suggested that black spruce seedlings on burned sites may have encountered more competition from understory vegetation when compared to logged sites.

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PROBLEMS RELATED TO FOREST REGENERATION IN FINNISH LAPLAND

Eljas Pohtila[1]

ABSTRACT.--The severe natural conditions in Finnish Lapland provide many challenges to regeneration of Scots pine. Good seed crops are rare, natural regeneration is slow, and planted seedlings often experience high mortality. Improved methods are being developed for site preparation and artificial regeneration, including seeding and use of containerized stock. Consequently, regeneration is more successful than it was 40 to 50 years ago.

NATURAL CONDITIONS

Intensive forest management is practised in Finnish Lapland farther north than anywhere else in the world. Scots pine is the main tree species. The absolute timberline is formed by mountain birch, but the outermost conifer on the timberline is Scots pine, not Norway spruce as is the case on the Kola Peninsula, NW Russia, for example. Apart from the most continental parts of Siberia, the belt of contiguous coniferous forests never reaches as far north as it does in Finland. Because of variations in climate, the timberline is in a constant state of flux. The polar timberline may be relatively far away, but when one proceeds straight up the alpine timberline in Lapland is always just a few hundred meters in elevation.

RECURRENCE OF GOOD SEED CROPS

In the case of Scots pine, the development of seed from flowering to maturity takes two growing seasons. For a good cone crop to be produced, the minimum temperature sum of the summer preceding flowering, i.e., during the period when the flower buds are formed, should in Lapland be at least 910 degree-days (d.d.) (threshold value +5°C). To have 50 % of the seeds maturing (which is regarded to be the minimum requirement for seed collection to be worthwhile) the total temperature sum of the summer when the seed mature should be at least 845 d.d. If the recurrence of warm summers follows the normal distribution pattern, the probability of a big yield of mature seed is, according to weather statistics, for example, in Sodankylä (67°22'N, 26°39'E) only 0.18 x 0.3 = 0.06. Practical experience has confirmed this. There may have been two or three periods during the current century during which Scots pine has yielded a good crop of mature seed right up to the timberline.

SLOWNESS OF NATURAL REGENERATION

Research has shown that the regeneration period required for a clearing to be converted to a stand of young pine in Lapland is, on an average, between 30 and 40 years. Natural regeneration can be accelerated by soil preparation. Nevertheless, it seems that the prepared soil's capacity to become restocked deteriorates rapidly. Immediate seeding is required if the landowner wishes to get full benefit of soil preparation. Maturing of the seed crop is uncertain right up to the very last moment and this is why it is difficult to correctly time the soil preparation treatment. The period of time from soil preparation to natural seeding may be so long that the investment in soil preparation is totally wasted.

Natural regeneration is given priority in the instructions for forest management. In practice, artificial regeneration, i.e., seeding or planting, is today more common than natural regeneration. Besides the slowness of the natural regeneration process, other factors promoting this development are the profusion of under-productive stands and the frequency of gales, which during the past few decades have undone numerous natural regeneration plans by bringing down the seed trees.

MORTALITY OF SEEDLINGS AND TRANSPLANTS

It has proved to be difficult to achieve full stocking of the required tree species as set out in the recommendations on planting and seeding density (2,000 seedlings per hectare). Up to half of the initial seedlings may die as a result of unfavourable natural conditions or due to neglect in tending of young stands. The latest forest inventory conducted in Finland has revealed that failures amounted to 14 %of the total area under artificial regeneration in Lapland, and that these

[1]*Prof. Dr. Eljas Pohtila The Finnish Forest Research Institute Unioninkatu 40 A FINLAND. Tel: +358 0 857 051, Fax: +358 0 625 308 Email: eljas.pohtila@metla.fi*

generally occurred on sites about 300 m above sea level.

Planting and seeding densities have now been raised to 2,500 and 3,000 per hectare, respectively. Campaigns are also being conducted to improve the standard of tending of young stands. While it is in principle easier to increase the density in seeding than in planting; large supplies of seed would be required. For the present, we have to rely on natural seed crops, but things will soon be different as newer seed orchards established to meet the demand coming from Lapland attain full production.

Because of its humid climate, Lapland is basically well suited for increased application of seeding as a means of forest renewal. Combined with soil preparation, more interest has been focused on broadcast seeding. Seeding, however, requires efficient advance management of young stands, including control of ground vegetation and undesirable coppice. The general public in Finland disapproves of the use of herbicides to control ground vegetation and coppice shoots, and people are generally opposed to the application of intensive forest management. It is, therefore, difficult for forestry goals to be reached simply by seeding. Chemical control techniques have been investigated to a great extent in Finland and these could be managed.

The adoption of containerized transplants in planting has clearly improved the survival of seedlings. The differences among the various types of seedling stocks appear to be small. The bigger the seedlings are, the faster is the afforestation of the regeneration area.

COMPATIBILITY OF DIFFERENT METHODS

Soil preparation on regeneration sites has proven to be one of the greatest advances in forest regeneration. Site preparation has been common practice in the forests of Lapland since the 1950's, when the concerted, large-scale regeneration of over-mature stands was begun. Wing ploughing has proved to be a very reliable method and one that is not significantly hampered by the stoniness of the glacial till soils.

Experimental research has shown that it is important how the soil preparation and planting and seeding methods are combined. Generally speaking, prescribed burning, scalping and disc ploughing are better suited for seeding than for planting, whereas wing ploughing and other such methods are better suited for planting than for seeding. Depending on the site conditions, different methods are combined in workable pairs. During the past few years, the most commonly used treatment pair has been wing ploughing and planting because regeneration has been targeted on fresh and peat-covered residual forests. Most recently, heavy ploughing has been used less frequently, partly because of the criticism brought forth by conservationists and environmentalists.

A CHANGING FOREST

Despite the difficulties, the development in the forests of Finnish Lapland over the past decades has been in a satisfactory direction. The most essential change has been in regeneration. In the early 1950s, only 15 % of the forests were less than 60 years of age, while today it is 40 %. The proportion of Scots pine as the dominant tree species has also increased by 15 %. The annual increment in stemwood is today greater than ever before.

It seems that pristine forests no longer exist in the timber production areas of Finnish Lapland. About 30 % of all of Finnish Lapland's forest lands have been set aside for conservation or other multiple use purposes.

INTERNATIONAL COOPERATION IN BOREAL FOREST RESEARCH

THE IGBP NORTHERN EURASIA STUDY

Michael A. Fosberg[1], Will Steffen[2], Anatoly Shvidenko[3], Johann G. Goldammer[4], Frank Oldfield[5], and Ernst-Detlef Shulze[6]

ABSTRACT.--The International Geosphere Biosphere Program is planning an integrated hydrological, atmospheric, chemical, biogeochemical and ecological global change study in the tundra - boreal region of Northern Eurasia. The unifying theme of the IGBP Northern Eurasia Study is the terrestrial carbon cycle and its controlling factors, and how these will change under changing environmental conditions. Special emphasis will be placed on the role of disturbance (fire, insect, disease, human activities) in driving the hydrological and biogeochemical cycles and in shaping the ecosystem. This study is intended to develop an integrated understanding of temporal and spatial processes of the carbon cycle in the Northern Eurasian landscape from paleo-records as well as contemporary observations. Integrated modeling studies will be used to both design the experiment as well as interpret findings. Close coordination and collaboration between the IGBP-NES activities and the International Boreal Forest Research Association will promote extended field research, wider involvement of the international research community, and will improve our understanding of global change in Northern Eurasia.

INTRODUCTION

Much interest in Northern Eurasia, and in high latitudes in general, is centered on their role in the carbon cycle. High latitudes contain approximately 25% of the global land area, contain 800 - 900 Gt of carbon, or approximately 35% of the global terrestrial carbon (fig. 1). About two-thirds of the boreal forest and 20% of the global forests are in Northern Eurasia. Because much of the region contains permafrost, ecosystems are very sensitive to changes in climate and land use (Steffen and Shvidenko 1996). In terms of ecosystem physiology, and soil and wetland processes, global change will have direct effects on the carbon cycle, on water and energy exchange and on nutrient dynamics. Atmospheric general circulation model scenarios suggest that warming will be greatest at high latitudes. This could lead to greater soil respiration, with release of carbon dioxide, and melting of methyl hydrates in permafrost regions and subsequent release of methane to the atmosphere. The boreal and tundra region could become a net source of these greenhouse gases under a doubled carbon dioxide atmosphere - climate response and further accelerate warming (Oechel *et al.* 1994, Steffen and Shvidenko 1996, Zimov *et al.* 1996).

Hydrological processes play a key role in the functioning of tundra and boreal ecosystems. Permafrost is widespread, and together with topography, play a large role in water flux rates and level of the water table. These, in turn, are critical factors in determining vegetation type, carbon balance and land - atmosphere exchange of water, energy and carbon dioxide. Global change is also expected to have pronounced effects on the composition and structure of high latitude ecosystems (Steffen and Shvidenko 1996).

Ecosystem disturbances from fire, insects and disease have dominated boreal forest landscapes (Shvidenko *et al.* 1996). Fires in forests and peatland impact the hydrological cycle through changes in rate of evapotranspiration and run-off. Fire has a significant impact on the structure and composition of forests from the stand to landscape levels, on formation and nutrient status of soils, and on the formation and

[1] *Michael A. Fosberg, IGBP BAHC, Potsdam Institute for Climate Impact Research, Potsdam Germany. Present address: 20 Ames Street, Onancock, Virginia, USA 23417.*

[2] *Will Steffen, IGBP GCTE, CSIRO Division of Wildlife Ecology, Canberra, Australia.*

[3] *Anatoly Shvidenko, IIASA, Laxenburg, Austria.*

[4] *Johann G. Goldammer, Max Planck Institute for Chemistry, Biogeochemistry Department, Freiburg, Germany*

[5] *Frank Oldfield, IGBP PAGES, Geographishes Institute, University of Bern, Bern, Switzerland*

[6] *Ernst-Detlef Shultze, University of Bayreuth, Bayreuth, Germany.*

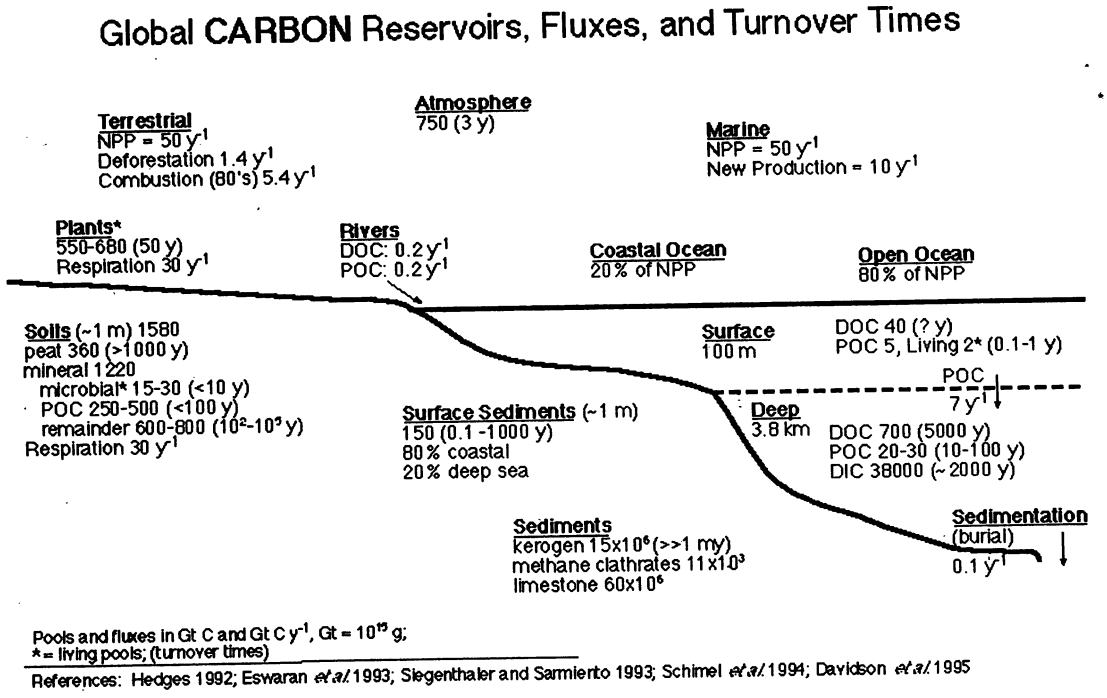

Figure 1.—*The global carbon cycle: reservoirs, fluxes and turnover times.*

dynamics of peatland and permafrost (Goldammer and Furyaev 1996). Fire is a significant factor in the carbon cycle from the release of carbon during the fire, making nutrients readily available following fire (Fosberg *et al.* 1996). Increased human activities at high latitudes are becoming more important, and could well rival fire in disturbing the landscape in the future.

THE STUDY AREA

The region is characterized by a strong temperature gradient from north to south, which is the major determinant of the general biome structure: tundra and four sub - zones of taiga (sparse, northern, middle, and southern). There is strong continentality in an east - west direction exhibiting strong temperature and precipitation gradients. The western region receives more than 400 mm per year and minimum temperatures are generally above -50C. The eastern portion has 200 mm of precipitation and minimum temperatures are below -50C (fig. 2).

These climatic gradients result in a differentiation of vegetation with western tundra extremely wet, changing to colder and drier tundra types in the east. Permafrost is deeper in the east. The tundra - boreal ecotones are also different. In the west the tundra grades into evergreen conifers (*Picea*) while in the east the tundra grades into deciduous conifers (*Larix*). The western taiga is mixed conifer forest and swamp, while the east is monotypic stands of *Larix*.

MAJOR ACTIVITIES OF THE NORTHERN EURASIA STUDY

The Northern Eurasia Study will consist of an integrated set of experiments and observational studies at a variety of scales, modeling, and aggregation activities.

The major goals are:

1. Predict the direct effects of global change on carbon cycling.

2. Predict the effects of global change on land - atmosphere exchange of water and energy, and on groundwater hydrology.

3. Predict the effects of global change on the composition and structure of tundra and boreal ecosystems and the interactions with ecosystem function, and

4. Determine the interactions of human driven land cover change and changes in disturbance regimes (fire, insect outbreaks, thawing of permafrost) with biogeochemical cycles and with the function, structure and composition of tundra and boreal ecosystems.

The study is organized around seven components: transect; network; water, energy and biogeochemical cycles; disturbance (fire); paleo-ecology; and modeling and integration.

Specific activities include:

1. Development and operation of north - south transects in eastern Siberia which include sites in tundra, tundra - forest transition, and in larch, spruce, and pine forests.

2. Development and operation of a network of observing sites throughout Northern Eurasia focusing on trace gas emissions and their controlling factors. The network will be concentrated on the extensive wetland in the west, but also should include a few sites in the east (including areas of permafrost), a site in the evergreen taiga, and a site in the tundra. Special attention should be given to the wetlands in the west.

3. A water, energy and carbon flux study, integrated with the transect and network components. This activity is part of the global FLUXNET to characterize fluxes of water, energy and trace gases between the biosphere and the atmosphere, and should include forest, wetlands, and tundra sites.

4. Studies of disturbance regimes which should include work on all major disturbances. They should include work on both gaseous and aerosols emissions from fires, and on the ecological role of fire in influencing the successional dynamics of the taiga and permafrost formation.

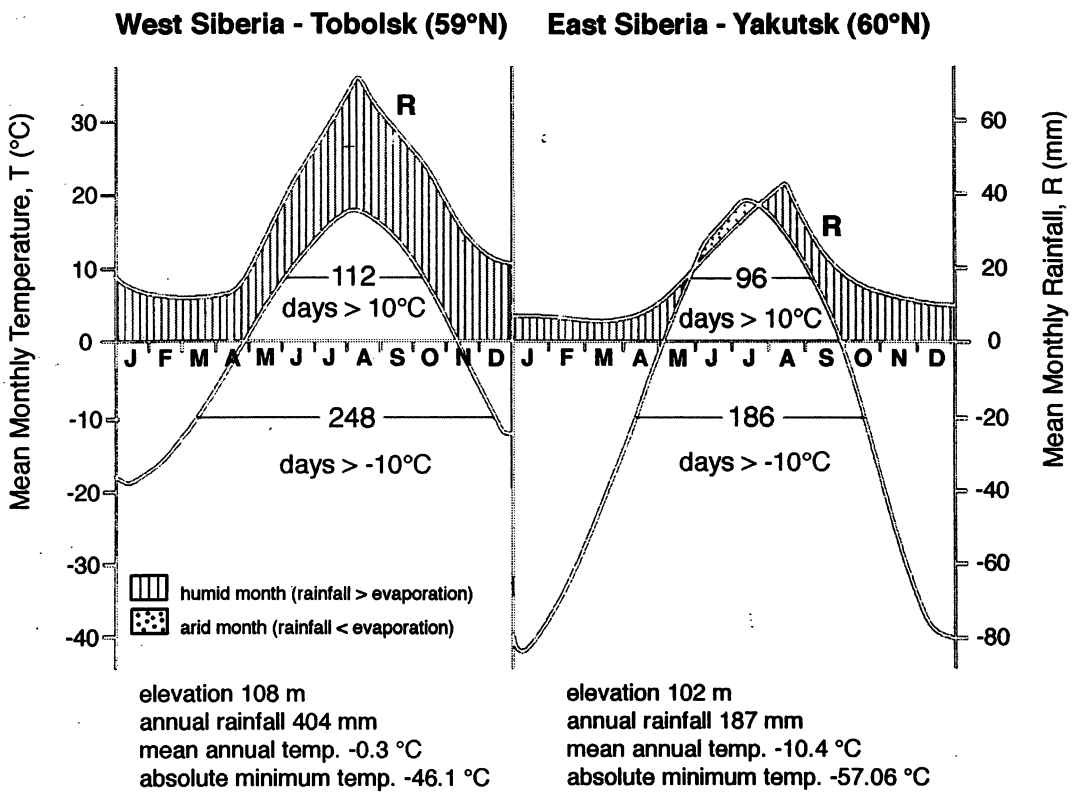

Figure 2.--*Examples of West and East Siberian climate at about 60° N latitude for Tobolsk at the Irtysch River (58° N, 68° E) and for Yakutsk at the Lena River (62° N, 138° E) (Walter 1994).*

5. Aggregation, modeling, and integration will require a large suite of models which will be used to help design the experimental and observational studies, interpret the data and project the impacts of global change on high latitude ecosystems.

6. A paleo-ecological transect to address questions of process, dynamics and change through time, and which will be interactive with contemporary monitoring and experimentation.

The Transect Component

The overall concept of transects are to use spatial gradients as surrogates for temporal change. They are intended to address questions where a long period of prior equilibration is required, where spatial context is essential, identification of thresholds along a continuum, and questions of gradient driven processes. Typically, a transect would be 1,000 km in length and a few hundred km wide. Transects would be laid out along a primary gradient of temperature, precipitation or land use, and could have secondary gradients either parallel or normal to the primary gradient.

The Northern Eurasia Study is one of several IGBP transects which are designed to integrate global change studies consisting of distributed observational studies and manipulative experiments and modeling studies. This transect approach promotes collaborate, interdisciplinary research and leads to efficient use of scientific resources (Koch *et al.* 1995). The Northern Eurasia Study (fig. 3) is based on the primary north - south gradient of temperature with secondary gradients of nutrient status, precipitation and land use change. The Northern Eurasia Study is in fact, composed of two parallel sub-transects, one along the Yenesei River and one along the 135th meridian in the Lena River Basis. The Northern Eurasia Study is one of four high latitude transect studies planned for (Scandinavia and Alaska) or already operational (Canada).

The Network Component

The Network of study sites will form the basis for characterization and quantification of trace gas fluxes from a variety of systems and understanding of controls of these fluxes. These flux measurements will be a part of a global network of flux measurements and will lead to an understanding of the global fluxes and the variations of these trace gas fluxes.

Questions that the network should address are:

1. Is the southwestern Siberian wetland region unique with respect to biosphere – atmosphere exchange of water vapor, carbon dioxide, methane, and other trace gases?

2. What proportion of trace gas emissions are "fossil," that is, from carbon stored under decaying thermokarst lakes in eastern Siberia or released by exploitation of oil and natural gas deposits?

3. What impact do trace gas emissions from forested areas have on atmospheric expansion (ozone and others)?

4. What types of fire experiments and campaigns can be usefully linked to sites established for other purposes, for example, peat fires?

5. Are the vegetation and soil factors that influence carbon, water, and energy exchange in the western, wetter and warmer region, similar to those in the eastern region of Siberia?

Water, Energy and Carbon Flux Component

There are two aspects of water, energy, and carbon fluxes to consider. Water, energy, and carbon fluxes are coupled, water and energy flux studies routinely include fluxes of carbon dioxide. In fact, fluxes of carbon dioxide cannot be interpreted without inclusion of the hydrological cycle. Second, water and energy fluxes between the land surface and the atmosphere influence climate. These bi-directional interactions are a key aspect of this component.

The principal objectives are:

1. Investigate the energy, water, and trace gas exchange processes in the soil – vegetation – atmosphere interface at the patch scale.

2. Investigate and apply methods to aggregate fluxes and land surface parameters from local to continental scales.

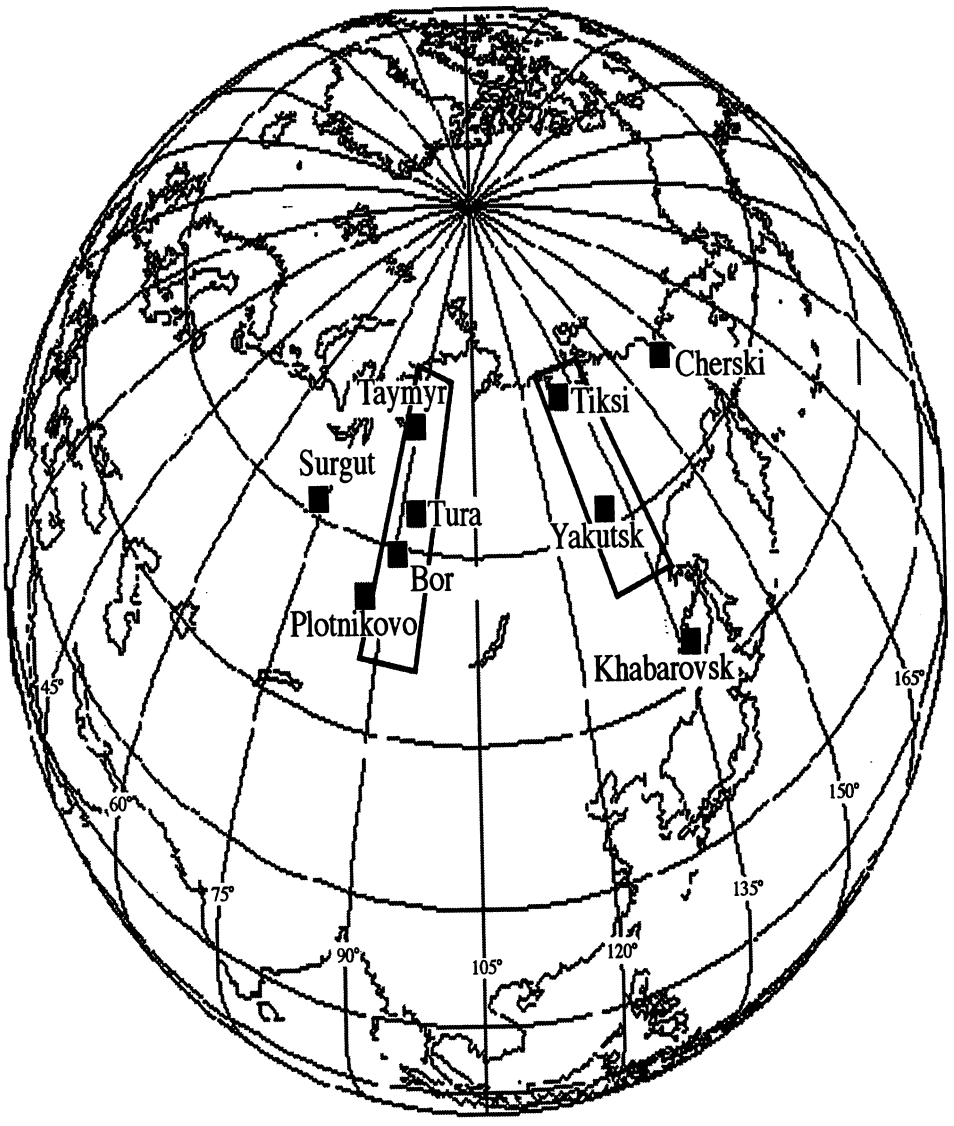

Figure 3.--*Map of Siberia showing the two major North-South transects for the IGBP Northern Eurasia Study.*

3. Investigate below ground processes.

Design of the water, energy and carbon flux component should address:

1. Measurements should include both eddy correlation measurements and nested catchment studies of internal fluxes.

2. Eddy correlation measurements should be made at the intensive study sites along the eastern transect and at the network sites.

3. Eddy correlation and chamber measurements of trace gas fluxes should be made simultaneously at the same locations.

4. Eddy correlation measurements of fluxes should be made at a number of locations around intensive sites, representing the major stages of forest succession. These should be long term measurements.

5. Surface hydrological studies should include catchment experiments at a variety of scales and should be linked to GEWEX studies at larger scales.

6. Measurement of snow fall, snowpack, and melt are essential in understanding the hydrological, energy, and trace gas budgets and fluxes.

7. "Fingerprints" of current fluxes in permafrost are essential.

Fire Component

The fire component of the North Eurasia Study will have four parts:

1. Fire manipulation at individual forest sites.

2. A series of campaigns based on air and space born research platforms.

3. Construction of a fire database relating the extent, frequency, and intensity of fires to vegetation and climate conditions for both contemporary and historic conditions.

4. Development of aggregated models of forest fire frequency and extent, and response to global change.

Modeling and Integration

In general, there are three scales at which modeling will take place; the patch scale with SVAT models, mesoscale, and continental or global scale. For ecosystem dynamics, phenomena such as fire operate at scales larger than a patch, and are difficult to portray at the mesoscale or continental scale. A major aim of the Northern Eurasia Study is to tackle the difficult scale of tens to hundreds of kilometers.

The models that will be considered are:

1. Patch scale; trace gas flux models, ecophysiological models, patch scale models of ecosystem dynamics and semi-empirical forest growth and yield models.

2. Landscape or mesoscale models and aggregation tools consisting of; hydro-climatological aggregation techniques, mesoscale meteorological models, landscape models of ecosystem dynamics, landscape hydrological models, fire models, and land use – land cover changes projections.

3. Continental scale models addressing; ecosystem and regional biogeochemical dynamics and cycles, models of atmospheric chemistry, and models of bi-directional feedback.

Paleo-ecology Component

The paleo-science component will provide more than just a "historic context" perspective for the North Eurasia Study. It will address the full range of science required to answer questions of process, dynamics and change through time. The paleo component will address:

1. Time series of spatial change.

2. Historic changes in terrestrial ecotones.

3. Reconstruction of past atmospheric composition.

4. Hydrological changes over time.

5. Human impacts on terrestrial and aquatic ecosystems.

6. Historic reconstruction of carbon budgets over space.

AN INTEGRATED STUDY OF THE REGIONAL CARBON BUDGET

Each of the components of the Northern Eurasia Study contributes to an understanding of the carbon budget in northern Eurasia. The total pool of carbon, in living biomass, in dead above ground biomass as well as the soil carbon will change over time. Lateral transport, either through riverine systems or by the atmosphere in or out of the region, and release of fossil carbon preclude approaching the regional carbon budget as a closed system in which carbon is just moved from one pool to another. Pools of carbon, change in those pools and the mechanisms of change as well as the vertical and lateral fluxes of carbon must be addressed as a dynamic problem rather than a closure problem. This places emphasis on understanding the mechanisms and interactions of changes in the carbon pools, the ecophysiological controls of the fluxes, the dynamics of how disturbance changes the pools, and the role of nutrients and water availability in controlling the fluxes.

Integrated carbon assessment requires unified approaches for both modeling and experimentation. We will need to use the name or compatible primary and integrated territorial units for all components of the study – hydrology, fire and so on (Rojkov *et al.* 1996).

There is no current model at the regional level that can be applied to an integrated assessment of the carbon-vegetation-hydrological system. Global scale models are too coarse for the heterogeneous landscape, patch scale models are not easily applied because of the aggregation problems in incorporating disturbance, permafrost, among others. We will need a model which incorporates such factors and which will allow us to aggregate information at the larger scales. A basic model of urban change,

$$DC/dt=Input\ (t)-Output(t)$$

must include in-situ transfers between pools, lateral, and vertical fluxes, the controlling mechanisms and rates of the transfers of the fluxes, and the drivers of the processes. The total, or substantive change of carbon over time, must include those spatial ingredients such as proximity of disturbance, competition, and so on.

COOPERATION AND INTERACTIONS WITH THE INTERNATIONAL BOREAL FOREST RESEARCH ASSOCIATION

The International Boreal Forest Research Association was established with the mission of promoting and coordinating research to increase the understanding of the role of the circumpolar boreal forest in the global environment and the effects of environment on boreal forests (Apps *et al.* 1995) This mission is similar to that of the International Geosphere Biosphere Program; to describe and understand the interactive physical, chemical, and biological processes that regulate the total earth system, the unique environment that is provided for life, the changes that are occurring in these systems, and the manner in which they are influenced by human activities.

A partnership between IGBP and IBFRA to design and implement the Northern Eurasia Study, and other high latitude transect studies, partnership in execution of research projects, exchange of information and data, conduct of joint workshops to synthesize knowledge, and to bring that knowledge into decision and policy for a will serve the mission and goals of both organizations.

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Conard, Susan G., ed.
2000. **Disturbance in boreal forest ecosystems: human impacts and natural processes.** Proceedings of the International Boreal Forest Research Association 1997 annual meeting; 1997 August 4-7; Duluth, Minnesota, USA. Gen. Tech. Rep. NC-209 . St. Paul, MN: U. S. Department of Agriculture, Forest Service, North Central Research Station. 435 p.

The papers in these proceedings cover a wide range of topics related to human and natural disturbance processes in forests of the boreal zone in North America and Eurasia. Topics include historic and predicted landscape change; forest management; disturbance by insects, fire, air pollution, severe weather, and global climate change; and carbon cycling.

Key Words: Boreal forest, disturbance, forest management, climate change, forest health.

Our job at the North Central Forest Experiment Station is discovering and creating new knowledge and technology in the field of natural resources and conveying this information to the people who can use it. As a new generation of forests emerges in our region, managers are confronted with two unique challenges: (1) Dealing with the great diversity in composition, quality, and ownership of the forests, and (2) Reconciling the conflicting demands of the people who use them. Helping the forest manager meet these challenges while protecting the environment is what research at North Central is all about.