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The Role of Boreal Forests and Forestry in the Global Carbon Budget

Proceedings of IBFRA 2000 Conference
May 8-12, 2000
Edmonton, Alberta, Canada

C.H. Shaw and M.J. Apps, Editors

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ABSTRACT

The International Boreal Forest Research Association (IBFRA) 2000 conference "The Role of Boreal Forests and Forestry in the Global Carbon Budget" was held on May 8-12, 2000, in Edmonton, Alberta, Canada and brought together nearly 300 experts, scientists, forest managers, and policy advisors from around the world. Manuscripts from the conference have been published in special issues of *The Canadian Journal of Forest Research, Forest Ecology and Management* and *Climatic Change* as well as in this proceedings volume. Collectively these papers represent a snapshot of the state of science at the end of the 20th century. They cover a range of topics including large scale assessment of carbon budgets and cycling, impacts of climate change and natural disturbance regimes, and contemporary issues of importance to policy makers and forest managers. This proceeding volume includes a synthesis paper that summarizes the important findings and future research needs identified in the papers published in this report and the three journal special issues.

RÉSUMÉ

La conférence 2000 de l'Association internationale de recherche sur les forêts boréales (IBFRA) intitulée « Le rôle des forêts boréales et de la foresterie dans le budget global du carbone » s'est tenue du 8 au 12 mai 2000, à Edmonton (Alberta), Canada, et a rassemblé près de 300 experts, scientifiques, gestionnaires forestiers et conseillers en politiques du monde entier. Les communications présentées à la conférence ont été publiées dans des numéros spéciaux de *The Canadian Journal of Forest Research, Forest Ecology and Management* et *Climatic Change* ainsi que dans les présents actes. L'ensemble de ces publications constitue un instantané de l'état actuel de la science à la fin du 20e siècle. Elles couvrent une vaste gamme de sujets allant de l'évaluation à grande échelle du budget et du cycle du carbone aux impacts des changements climatiques en passant par les régimes de perturbations naturelles et les enjeux contemporains importants pour les décideurs et les gestionnaires forestiers. Ces actes contiennent également un article de synthèse qui résume les découvertes importantes et les besoins en recherche identifiés dans les articles publiés dans ce rapport et les numéros spéciaux des trois revues mentionnées.

The Role of Boreal Forests and Forestry in the Global Carbon Budget

Edmonton, Alberta, Canada 8-12 May 2000

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The Role of Boreal Forests and Forestry in the Global Carbon Budget

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FOREWORD

The world's boreal forests, containing large carbon stocks in their biomass and soils, have long been recognized as important contributors to the net balance of carbon exchange between the atmosphere and the biosphere. An international conference on the "Role of Boreal Forests and Forestry in the Global Carbon Budget" was held in Edmonton, Alberta, Canada from 8 to 12 May 2000 to examine the present state of understanding of this subject. Hosted by the Canadian Forest Service, the conference attracted nearly 300 scientists, forest managers and policy advisors from around the world. Notably there was excellent participation from both China and Russia.

The conference was organized under the auspices of the International Boreal Forest Research Association (IBFRA) whose stated goal is to "*promote and coordinate research to increase the understanding of the role of the circumpolar boreal forest in the global environment and the effects of global environmental change on that role.*" Formed in 1991 as a relatively informal association of interested scientists and government agencies, IBFRA has previously organized conferences related to that goal in Canada, Finland, Norway, Russia, Sweden and the USA, and in most cases, published the results in Journal Special Issues (Apps *et al.* 1995, Korpilahti *et al.* 1995) or in edited Proceedings Volumes (Conard 2000, Pisarenko 1997).

Some of the issues identified and discussed in previous IBFRA conferences, especially those dealing with the importance of disturbances and climate change in boreal forest ecosystem dynamics, remain significant scientific challenges today. New scientific insights and findings on these subjects formed a significant component of the conference in 2000. In addition to these ecological issues, however, the development of the Kyoto Protocol in 1997 has created a new dimension, and generated a new urgency for both improving the science of forest carbon cycles as well as its dissemination. Although this IBFRA 2000 conference, by design, did not deal with the intricate policy issues associated with the subject, the management of forest ecosystems for enhanced carbon sequestration and reduced emissions was one of the central considerations of the conference. Related to this management interest was an increased awareness of the pressing need to understand and quantify the influence of a changing environment on the carbon stocks of boreal forest ecosystems.

The Honorable Ralph Goodale, then Minister of Natural Resources Canada, gave an opening address by satellite hookup. Over the next 5 days, 180 presentations (40 verbal and 140 posters) were delivered, covering a wide spectrum of research interests, broken into four themes. Theme 1, dealing with *Boreal Forest Carbon Budgets,* was introduced by a Keynote presentation by Dr. Werner Kurz (Canadian Forest Service). Prof. Paul Jarvis (University of Edinburgh) and Dr. Steve Running (University of Montana) gave a two-part Keynote address to open Theme 2 on *Monitoring and Measuring Boreal Forest Carbon Fluxes and Stocks.* Theme 3 was devoted to *Forestry Practices for Carbon Management* and highlighted by a special address given by Mr. George Weyerhaueser Jr. (Weyerhaueser Company, USA). Keynote speaker Dr. Jagohman Maini (United Nations) spoke to the issues of Theme 4: *Social and Economic Issues, including Implications of the Kyoto Protocol.*

The scientific understanding of the contribution of boreal forests and forestry in the global carbon cycle at the end of the 20th century was presented at this conference. To capture this state of the science, a series of publications were arranged by the Conference Publications Committee. First, an edited book of abstracts for all posters and verbal presentations was published prior to the conference (Apps and Marsden, 2000). Following the conference, authors were invited to submit manuscripts based on their presentations for publication in one of three Special Issue Journals (externally peer-reviewed to the standards of the journal) or in an edited (but not peer-reviewed) Conference Proceedings volume.

The papers published in the *Canadian Journal of Forest Research* Special Issue (Appendix I) focussed on research in Canadian boreal forests with some contributions from large scale studies in the United States, Russia and in Nordic forests. Papers in the *Climatic Change* Special Issue (Appendix I) represent conference presentations dealing with climate change impacts on carbon cycling/budgets and natural disturbance regimes (forest fire and insects) across the circumpolar boreal zone, while those in the *Forest Ecology and Management* Special Issue (Appendix I) were deemed to be of primary interest to forest scientists, managers and policy makers in forestry and dealt with the application of biological, ecological and social knowledge to forest management.

This Proceedings Volume encompasses a diversity of presentations from Canada, Russia, Ukraine, China, Norway and Sweden and provides a forum for new ideas, preliminary research results, testing emerging hypotheses and the description of initiatives for coping with large scale ecosystem research. The first paper in this volume is a synthesis of all the papers published in the three Special Issues and this Proceeding Volume and includes contributions from the keynote speakers. As such, it represents a synthesis of the state-of-science of the circumboreal forest's contribution to the global carbon cycle and the culmination of a highly successful and well attended conference. The synthesis paper is organized into five topic areas; Carbon Stocks and Fluxes; Effects of Natural Disturbances or Management Practices on Carbon Dynamics; Afforestation and Carbon Sequestration and Effects of Climate Change and Elevated CO_2 Concentration on Carbon Dynamics.

The coordinated publication of three Special Issues and a Conference Proceedings Volume, all with different publishers, has been a challenge and while not simultaneous, publication of all four volumes in 2002 has been achieved. The publications committee sincerely appreciates the extraordinary efforts and high degree of cooperation received from reviewers, authors, editors and managers to make this publication effort a success. Special thanks to those directly involved in the publication process including Katarina Kivel (Assistant Editor of Climatic Change) Tuija Lapveatelainen of the European Forest Institute and Lisa Bowker (Assistant to the Editor of the Canadian Journal of Forest Research). We anticipate that readers of this Proceedings Volume will want to refer to the three special issues for a comprehensive snapshot of the state of the science as we entered the new millennium.

On behalf of the Conference Program Committee, we wish to thank the Canadian Forest Service for hosting the conference. We also gratefully acknowledge the generous support the European Forest Institute, Alberta Forest Products Association, Alberta Pacific Forest Industries Inc., the Canadian Pulp and Paper Industry, the Foothills Model Forest, Forintek Canada Corporation, Manning Diversified Forest Products Research Trust Fund, Miller Western Forest Products Ltd., Suncor Energy Inc., the Sustainable Forest Management Network, Weldwood of Canada Ltd., the Weyerhaeuser Company, and the Canadian Government through the Canadian Climate Action Fund, the Energy from the Forest Program, and the Canadian International Development Agency. We wish to acknowledge the volunteer efforts from so many individuals at the Northern Forestry Centre who helped make the conference a success. Finally we wish to acknowledge all contributors to this Proceedings Volume and the three Special Issues and the reviewers of their manuscripts for the excellence of their work.

C.H. Shaw (Senior Editor for the Conference Proceedings)

M.J. Apps (Conference Chair and Senior Guest Editor for the Special Issue in *Canadian Journal of Forest Research*)

T. Karjalainen (Senior Guest Editor for the Special Issue in *Forest Ecology and Management*)

B.J. Stocks (Senior Guest Editor for the Special Issue in *Climatic Change*)

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NOTE

THE ROLE OF BOREAL FORESTS AND FORESTRY IN THE GLOBAL CARBON BUDGET: A SYNTHESIS

I.H. Fyles, C.H. Shaw, M.J. Apps, T. Karjalainen, B.J. Stocks, S. W. Running, W.A. Kurz, G. Weyerhaeuser, Jr. and P.G. Jarvis

INTRODUCTION

The circumboreal forests cover approximately 33% of the Earth's terrestrial surface, contain 25% of the vegetation carbon and 60% of the soil carbon, making up approximately 50% of the carbon stored in biomass and soil globally (Apps et al. 1993; Dixon et al. 1994, Schlesinger 1997). Boreal forests have often been reported as carbon sinks, meaning that carbon uptake in the biomass and soil exceeds carbon removals from those pools. Although the entire boreal forest is thought to be sink of 0.3 to 0.5 Pg carbon per year (Dixon et al. 1994, Apps et al. 1993)Bousquet et al. 2000, Gurney et al. 2002), this sink term varies both geographically and over time (Goodale et al. 2002). Thus for example, Kurz and Apps (1999) reported that Canadian boreal forests became a source in the 1980s as a consequence of large scale disturbances.

It is now widely acknowledged that climate models project the largest global warming trends to be in high latitudes (Running 2000). There is widespread evidence that this warming is already underway. Snow cover reduction, glacial retreat, sea ice cover, growing season length, river break up times, permafrost depth and air temperature records are all showing clear warming trends over the last 50 or more years.

The timing of the thaw in the spring and the freeze-up in the fall is a significant constraint on carbon fluxes in boreal forests (Gower et al. 2000). Because there is usually abundant solar radiation prior to the spring thaw, an early thaw may lead to an increase in the annual net carbon gain in regrowing stands relative to those experiencing a late thaw. By contrast with continental boreal spruce forest, young spruce stands at similar latitudes in the oceanic climate of the Atlantic fringe have net ecosystem fluxes of over 6 t C ha^{-1}, in part because there is no winter freeze up – leading to some carbon gain on almost every day of the year.

The carbon sink in the boreal forests today must be considered very vulnerable (Gower et al. 2000). Over wide areas, the net sink is both small and fluctuating, so that the permanence of the sink as atmospheric CO_2 concentrations and temperature continue to rise has become a crucial question (Jarvis et al. 2001). Short-term experiments demonstrate that the increase in photosynthesis that results from a rise in atmospheric CO_2 concentration tends to diminish at higher CO_2 concentrations, while both autotrophic and heterotrophic respiration increase, perhaps exponentially, with increasing temperature. This has led to the suggestion that the overall capacity to take up additional carbon from the atmosphere may decline progressively as climate changes (Scholes 1999). Thus it has been suggested that by about 2050 respiration will have increased so as to exceed photosynthesis and today's net terrestrial carbon sink will have become a carbon source. There are, however, contrary predictions. For example, analysis of a boreal forest in northern Sweden using the ecosystem model G'day indicates that net primary production will consistently exceed heterotrophic respiration so that the carbon sink rises to an asymptotic maximum and remains there for 100 years without significant decline.

In the boreal forests of Canada and Russia, large-scale natural disturbances play a large role in determining the outcome of carbon source/sink relationships. For example, Kurz and Apps (1999) reported that Canadian boreal forests became a source in the 1980s as a consequence of large scale disturbances. Changes in disturbance regimes bring about long-term changes in the age-class structure of these boreal forests. Forest age-class structures that are heavily skewed towards older ages, as observed in many regions of the circumpolar boreal forest may represent high C stocks that may not be sustainable into the future.

In the latter part of the 20th century the critical links between boreal forest ecosystems, carbon dynamics and global climate change were increasingly recognized by the scientific community. There was a clear need to share information better and to develop international collaborative research initiatives. The conference "The Role of Boreal Forests and Forestry in the Global Carbon Budget" held in Edmonton, Alberta, Canada May 8-12, 2000 was designed to be an end-of-the-century review of the state of a) our understanding of carbon dynamics of boreal systems and b) our ability to influence these through purposeful actions. This paper provides a synthesis of the papers published in the three special journals issues and the Proceedings (this volume) associated with the conference. As such, it represents a state-of-science summary of the circumboreal boreal forests' contribution to the global C cycle at the end of the 20th century.

The published papers and contributions from the keynote speakers have been grouped into five topic areas that form the sections of this synthesis; C Stocks and Fluxes; Effects of Natural Disturbances or Management Practices on Carbon Dynamics; Afforestation and Carbon Sequestration and Effects of Climate Change and Elevated CO_2 Concentration on C Dynamics.

The first section, in keeping with the major theme of the conference, focuses on updated estimates of carbon stocks and fluxes in Russia, Canada, China and western Europe. Estimation of C stocks were made at the national scale, factors controlling variability of estimates were investigated at the regional scale while processes were studied at sites within regions.

Advances in research on the effects of natural disturbances and forest management practices on C stocks and fluxes is the focus of the next two sections. Disturbance type, history and extent are now accepted as major factors influencing large scale carbon dynamics in boreal forests. Under conditions of a changing climate predicted by the IPCC (Watson et al. 2001), natural disturbances such as fire, insect defoliation and windthrow are expected to increase in frequency and extent and will influence C dynamics at all spatial scales. Silvicultural and harvesting practices affect C dynamics directly through changes in mortality and growth trends and may be used as management strategies to mitigate impacts of natural disturbances and purposefully alter source/sink relationships. Afforestation is addressed separately as a mechanism within the Kyoto Protocol for countries to offset net greenhouse gas (GHG) emissions. Finally advances in research on the influence of the predicted climate change and increases in atmospheric CO_2 concentrations on boreal forest C budgets and vegetation response are summarized.

CARBON STOCKS AND FLUXES

Since the last glaciation, large amounts of C have been taken up from the atmosphere by boreal forest ecosystems, accumulating in vegetation and soil organic C pools (Gower et al. 2000). In mature boreal forest stands, there may be up to eight times as much C in the soils as in the trees and other vegetation. With two or three exceptions, the boreal forest stands so far investigated with flux towers in Russia, North America, Europe exhibit C sinks with annual net ecosystem fluxes of 0.5 to 2.5 Mg C ha^{-1}. However, this sink must be considered very vulnerable and over wide areas where landscape scale processes, such as disturbances can result in large short and long-term emissions, the net sink is small and fluctuating. Estimates of C pools are therefore constrained by large temporal and regional variation, but measurements and model estimates of C sequestration, above and below ground C pools, and factors affecting those estimates are rapidly increasing for Russia, Canada, China and Europe.

Russia

Reported assessments of C stocks for the Russian boreal forest suggest that the vegetation was a sink for C during the last 40 years of the 20th century. Much of the C sequestered was accounted for by increases in the total area of rapidly growing young and middle-aged stands, the establishment of deciduous species and increase in the amount of coarse woody debris (CWD).

Using long-term forest inventory data, Shvidenko and Nilsson (2002) show that between 1961-1998 the vegetation of Russian forest lands had an annual average C sequestration of 210 ± 30 Tg C yr^{-1} of which 153 Tg C yr^{-1} was in live biomass and 57 Tg C yr^{-1} was in dead wood. However, the temporal variability of the sink estimate was very large: for the 5 year averages used in the analysis, the C sequestration varied from about 60 to more than 300 Tg C yr^{-1}. Within the forested area, the higher C sequestration was a result of a systematic expansion in the area of young and middle-aged stands and a corresponding decline in mature and overmature stands. Increases in CWD (about 57 Tg C yr^{-1}) that were reported for the last 10 years by Shividenko and Nilsson (2002) were associated primarily with fire and insect defoliation.

The total amount of C stored aboveground in forests managed in the Russian Forest Fund (RFF) is currently about 34 Pg in phytomass and between 5.4 and 3.6 Pg C in CWD (Shvidenko and Nilsson 2002; Utkin et al. 2002). These authors point out that while there was a significant increase in the area of young and middle-aged stands the mature and overmature stands still account for more than half (57%) of the phytomass C in the RFF.

Total stocks of organic C in soils (to 2 m depth) of the Russian Federation was estimated to be 367 Pg of which nearly half (164 Pg) was in the top 30 cm (Stolbovoi 2002). The total amount of soil C was estimated to be 172 Pg in the RFF and 183 Pg for soil C for all forests in Russia (Utkin et al. 2002). The C density in the soils of the Russian boreal forest is about 11.5 kg m^{-2} for the 0-1 m layer, similar to values found for Canada. The trend in the spatial distribution of C density in the soil C pool is nearly the reverse of that observed for C in the phytomass pool and results in a similar total C density (17-20 kg C m^{-2}) in all ecosystem types within subzones, through zones and through provincial profiles across Russia (Utkin et al. 2002).

The large area of peatlands in Russia contributes significantly to soil C storage and and total accumulation of organic C in deep peat (more than 50 cm thick) is estimated to be 156 Pg in the 0-2 m layer (Stolbovoi 2002). Studies such as that by Glebov et al. (2002) that examine peat forming processes, are required to increase baseline knowledge of C stocks in boreal peatlands in Russia.

Regional assessments of the size of forest C pools highlight the importance of geographical differences, forest age-class structure and species dynamics on the evaluation of C dynamics. Krankina et al. (2002) quantified regional stocks of CWD in seven major forest regions of Russia and concluded that variation among regions is so large that extrapolating information from one region to another would be difficult and misleading. They found that stocks of CWD in the western part of Russia were, on average, lower (4-5.8 Mg ha^{-1}) than in eastern regions (11–14.4 Mg ha^{-1}). These differences were associated with differences in the nature of prevailing disturbance (clearcut harvesting in the west and natural disturbance in the east). Lower decomposition rates in the east, due to its harsher climates, may also be a contributing factor.

In Middle Zavolgie (western Russia), forest land use has played an important role. C stocks in stem wood, examined over a 40 year period showed that stem wood C had increased from 277 Tg in 1958 to about 307 Tg in 1995. This increase was ascribed to increased establishment of rapidly growing deciduous species (Kurbanov and Post 2002). The total area of young and middle-aged stands had increased by 27% over this period, resulting in a high rate of annual C sequestration through rapid regrowth. A study carried out over a growing season in central Siberia also showed significant C sequestration occurred in young deciduous forest stands (Meroni et al. 2002). Pine stands in the same region, dominated by young (0-40 yr) and middle-age stands (40-60 yr) also show rapid growth, with maximum C sequestration taking place between 40 and 70 years of age (Kurbanov 2002). Natural, fully stocked and unmanaged pine stands have a maxiumum net ecosystem exchange of 1.5 Mg C ha^{-1} yr^{-1} compared to 1.1 Mg C ha^{-1} yr^{-1} in disturbed and managed stands.

In Lakyda et al. (2002), C storage in the phytomass of Ukrainian forests was estimated at about 500 Tg of which about half (266 Tg) was contained in coniferous forests. A study of litterfall and soil organic matter (SOM) in the Krasnoyarsk region of middle Siberia showed that C accumulation in above and belowground SOM was greatest in larch stands and that coniferous stands contain 3 to 4 times more dead organic matter C in the above- and below-ground woody debris than deciduous stands in which decomposition is faster (Vedrova and Mukhortova 2002).

Two Russian studies used models to estimate future changes in biomass and soil C under current climatic conditions. Korovin et al. (2002) simulated long-term forest vegetation dynamics in Siberia and Far East Russia to show that a management strategy that gives sustainable maximum yield would also lead to stabilization of changes in biomass C. This, they suggest, was the result of an even balance amongst forests of different age structures.

For the Leningrad Administrative Area Chertov et al. (2002) used the model SOMM to assess the C balance in forest soils. After a 100- year simulation, a 30% decline in SOM in the organic layer and a 20% increase in SOM in the mineral soil resulted in a net increase of about 8%, or 20.7 Tg of C. Total C input to the soil and C emission from decomposition were roughly equal and reflect the relative stability of SOM pools in the forest soils of the region under the existing climate, stand structure and forest management.

Canada

The vulnerability of the C sink in boreal forests, referred to above, was illustrated in a simulation model study by Liu et al. (2002). Total forest biomass C in Ontario increased from 1.83 Pg to 2.56 Pg between 1920 and 1970 and then declined to 1.70 Pg by 1990. Carbon in soil and forest floor dead organic matter increased from 8.3 Pg to 11.0 Pg between 1920 and 1985 but then decreased to 10.95 Pg by 1990. Ontario's forest ecosystems sequestered between 41 and 74 Tg C yr^{-1} before 1975 but later became a C source releasing 7 to 32 Tg C yr^{-1}. This study is discussed in more detail in the section on disturbance.

Other Canadian studies examined factors affecting the variability of C estimates with a particular focus on belowground C pools. Global, national and regional scale estimates of C pools and fluxes are commonly based on a notion of average site conditions that are hard to define in highly heterogeneous landscapes. Better knowledge of the magnitude of this variability, factors that cause the variability in the landscape and how they are related to stand growth and C turnover is required to improve estimates of C dynamics at all spatial resolutions.

In central Canada, uncertainties in soil C estimates have been examined at polygon and regional scales (Bhatti et al. 2002). At the polygon scale, one empirical method estimated consistently higher soil C than another method—a difference that may be related to geomorphic and microclimate influences not accounted for in the lower estimate. Regional estimates, made using the two empirical methods and a model simulation, ranged from 6.2 to 27.4 kg C m^{-2}. Higher soil C simulated by the model arises in part because it accounts for components of the forest floor detritus that were not included in the other estimates.

In western Alberta, Banfield et al. (2002) used a model simulation to provide a regional estimate of biomass, forest floor and soil C stocks while forest inventory, plot and soil polygon data provided an estimate of their spatial variation. They found that inventory-based biomass-to-age relationships could be modified using a relationship between clay content and biomass C to assess variation across a region as well as to improve predictions at a higher spatial resolution.

Such a simple correction was not found, however, in a smaller scale study in northern Alberta (Little et al. 2002). Differences in leaf area index (LAI) and soil C between the top and toe of slopes in boreal mixed-wood forest conditions, illustrated small-scale variation that was not easily related to site factors.

Information about the C content, the chemistry of forest soil organic matter, and factors that control later stages of litter decomposition can also be used to assess factors causing variability in soil C dynamics. Tremblay et al. (2002) showed that in productive upland soils of southern Quebec, the C content of the forest floor and mineral soil could be predicted from forest floor thickness, soil horizon colour, texture class and pH — factors easily accessible in soil survey data. Models were developed and used to construct maps that indicated sites where C accumulation in forest soils may be susceptible to forest management practices.

Preston et al. (2002) found that C and N stocks and distribution of C and N amongst the size fractions of organic matter, varied between two Brunisolic soils and a Cryosol in three upland forest sites in central Canada. The clay-textured Cryosol had higher C and N stocks and a higher proportion stored in the very small (< 63 μm) size fraction compared to the sandy-textured Brunisols. An increase in C stocks in the mineral soil horizons appeared to follow a trend of decreasing lessivage (mass transport within the soil profile), increase in pH and increase in exchangeable calcium.

Effects of litter quality and climate on decomposition rates of 10 different plant tissues and 1 wood block, were examined during a 6 year exposure at 18 upland sites across Canada (Trofymow et al. 2002). The mass remaining after six years could be effectively predicted from a limited range of climate (temperature and summer precipitation) and litter quality (acid-unhydrolyzable residue, AUR, and AUR/N) variables. The best set of variables for predicting decomposition, however, varied with the age of the litter, changing over the six year period. In the first year, litter quality properties and winter precipitation were important in predicting decomposition while in subsequent years, temperature, summer precipitation and a variety of litter-quality factors including AUR/N, were most useful.

Measurements of aboveground C have also been related to site characteristics. In Ontario, Chen et al. (2002), found that the relationship between net primary production (NPP) and stand age of black spruce (*Picea mariana* [Mill.] B.S.P.) was modified by site index (SI). Productive sites reached a maximum NPP after 40 years while less productive sites (peatlands) reached a maximum after 160 years. Differences in C allocation to woody components also varied with stand age and SI.

Longer term direct observations of C fluxes using eddy covariance techniques at various sites in the BOReal Ecosystem Atmosphere Study (BOREAS) region are providing useful insight on C dynamics. Measured C fluxes in black spruce, jack pine (*Pinus banksiana* Lamb.) and aspen (*Populus tremuloides* Michx) stands over one year in Saskatchewan have been made as part of the Boreal Ecosystem Research and Monitoring Sites (BERMS) initiative which seeks to extend the BOREAS observations to longer times (McCaughey et al. 2002). The study found that C fluxes were similar in the conifer sites but the C uptake for aspen was double that for black spruce in the summer while during winter and fall aspen lost significantly more C (Arain et al. 2002).

China

The long history of forest use by humans in China has led to large areas of degraded forests in the boreal regions of China (Jiang et al. 2002). However, China has some of the largest afforested/reforested areas in the world and high rates of afforestation and reforestation have resulted in the creation of larger forest areas than were expected under past forest development plans. This has had a large effect on the recent assessment of C budgets of forests in the temperate region of China (Xu and Zhang 2002).

Estimation of the current temperate forest C budget has been made using new data that includes tree C density in different age classes, soil C dynamics, wood utilization and the latest forest development plan (Xu and Zhang 2002). The F-CARBON model was developed to account for variation in the biomass densities and growth rates of the different age classes of forests and calculates the C emissions resulting from harvesting, burning and decomposition. For 1990, the C stock in China's temperate forest was estimated at 11.1 Pg C and the forest was a net sink for C with an uptake of 42.2 Tg yr^{-1}. Net uptake is predicted to increase gradually to 87.7 Tg C yr^{-1} in 2050 largely because of a predicted increase in biomass C.

Larch forests (Larix spp.) are important timber resources in China but are sensitive to global climate change. Forest biomass and net primary production (NPP) in the larch forests of northern China were estimated by Zhou et al. (2002) using models based on forest inventory data (FID). These models take into account the change in the ratio of forest biomass to volume with stand age and the effect of stand age on forest NPP. The relationships between NPP and biomass were not linear and these authors stress that natural and planted forests need to be treated separately when biomass and NPP of the forest are estimated. Wang and Zhou (2002) created a new NPP model based not only on FID but also on climatic data. This biology-climate model successfully simulated observed NPP data from larch forests and the authors suggest it may be applied to existing forest inventory data in other countries.

Europe

The C budget of soils and trees in the forests of western Europe was calculated for 1950 to 2040 (Liski et al. 2002). The C stock of both trees and soils increased over the period but the soil increasingly became the major C sink. In 1990, the estimated soil C sink was 26 Tg yr^{-1} or 32-48% of the contribution by trees but by 2040, the soil sink was projected to be 43 Tg yr^{-1} – about 65% that of the trees. The increase in soil C was attributed to an increase in litter inputs over time as the stands aged.

Joosten and Schulte (2002) caution that changes in management practices and environmental variables in Germany over the past 60 years have led to change in stand and tree characteristics. The authors found that tree density, ratio of height to DBH and average crown ratio were all significantly different from commonly used, but apparently out-of-date yield tables. The study suggested that the most reliable approach for calculating above-ground tree C was a cubic regression function that calculates the growing stock volume, and converts this directly into tree C content.

EFFECTS OF NATURAL DISTURBANCES ON CARBON DYNAMICS

Natural disturbances, in particular fire and insects, play an important role in the C dynamics of the circumpolar boreal forest on both short and long time scales (Kurz 2000). In the short term, disturbances cause direct emissions of C to the atmosphere as well as the transfer of C from living biomass pools to dead organic matter pools resulting in delayed emission through decomposition. In the long-term, changes in disturbance regimes bring about changes in the age-class structure of boreal forests. Major disturbances create large areas of rapidly growing young trees but also large amounts of decomposing residual dead organic matter. During a disturbance, and for a time after it, the net loss of C through decomposition (or combustion in the case of fire) may exceed the uptake of C in regrowth, and the affected stands function as C sources. As these stands age, the trees accumulate C from the atmosphere and replenish the decomposing forest floor stocks of C. Thus, changes in disturbance regimes at the landscape scale can result in periods in which large areas of forests act as either sinks (average age is increasing) or sources (average age is decreasing).

Average disturbance rates in Canada's forests have nearly doubled since 1970, largely as a result of increased incidence of fires, having a profound influence on the C budget (Kurz and Apps 1999). The effect of this increased disturbance was demonstrated by Liu et al. (2002) in their study of the boreal forest in Ontario. They found that between 1970 and 1990, increased disturbance (fire, insects and harvesting) resulted in the C sink changing to a C source. The young average age of the forest in 1990 (36.2 yr) indicates a high potential for C sequestration and under a less severe disturbance regime Ontario's forests would convert back to a sink.

In a study at the national scale, a simulation experiment related the amount of C stored in the Canadian boreal forest landscapes to length of fire cycle (Li and Apps 2002). More ecosystem C was stored in landscapes with long fire cycles than in landscapes with short fire cycles. In addition, more frequent small fires and less frequent large fires appear to be associated with long fire cycles and result in less variation in ecosystem C. Less frequent small fires and more frequent large fires, associated with short fire cycles result in larger interannual variation in ecosystem C.

The effect of changing the fire cycle on tree density and C storage was simulated in aspen stands and white spruce (*Picea glauca* (Moench) Voss) - aspen stands in central Canada (de Groot et al. 2002). In Alberta reducing the fire cycle from 263 yr to 196 yr increased aspen stem density and total C stocks. Because the fire cycle of 263 years is greater than the lifespan of aspen, trees were overmature and declining. Reducing the fire cycle prevented this decline and allowed vigorous regeneration after fire. Reducing the fire cycle from 88 yr to 65 yr in aspen stands in Saskatchewan resulted in a smaller average stem diameter and less C stored. Overall however, the shorter fire cycle of the Saskatchewan aspen stands will promote rapid suckering of aspen and these stands will store twice as much C as the Alberta aspen stands with the longer fire cycle. This suggests that there is a fire cycle between 88 and 196 years that will allow aspen stands to reach a maximum C storage. Overall C storage was not affected by the fire cycle in aspen-spruce stands although aspen density increased relative to spruce in the 65 year cycle.

Fire also influences the energy flow at the soil surface which in turn affects soil respiration, post-fire vegetation regrowth and C sequestration. Growing season albedo, measured before and after fire in a black spruce ecosystem in Alaska, was reduced in 46% of the burned area, increased in 40% and stayed the same in 14% of the burn (French et al. 2002). Whether albedo increased or decreased depended on the pre-burn vegetation and the burn severity. However, after five years of vegetation regrowth, albedo was higher throughout the burned area than in the unburned area.

Changes in vegetation composition can have a significant effect on the fire regime. Rupp and Starfield (2002) simulated the role of black spruce ecosystems in the fire regime of the interior Alaska boreal forest and found that both the number of fires and the total area burned increased as black spruce stands became an increasingly dominant component of the forest landscape. Increasing the areal extent of white spruce stands resulted in fewer fires and smaller area burned. Ecosystem flammability accounted for the majority of the differences in the distribution of the average area burned. These results suggested that not only do differences in vegetation have a significant impact on the fire regime, but that the large scale fire events resulting from increased black spruce forest also have the potential to alter future distribution of the vegetation.

The short term impact of fire on the global C budget is also significant but high interannual variability and difficulties in acquiring accurate data make estimations of the size of the impact problematic. For example, Conard et al. (2002) suggested that the extent and global importance of forest fires in the boreal zone in Russia have often been underestimated. Using remote sensing data, it was estimated that in 1998, a major fire year, 13.3 million ha of forest burned in Russia releasing between 135 and 190 Tg C or 14 to 20% of the average annual global C emissions from forest fires.

These authors believe that this is a conservative estimate and suggest more research is required to improve measurements of the extent and severity of forest fires and their potential impacts on C emissions in the boreal zone.

Assessment of C emissions associated with forest fires has been examined by several Russian authors. In Central Siberia, regional C emissions from fires of differing intensity were estimated from C accumulation in different types of pine stands with various mean fire intervals (Ivanova et al. 2002). These authors reported that emissions can vary by an order of magnitude between fire seasons because of high interannual variability in the burned areas and fire intensity. Estimates of total C emissions resulting from forest fires also require knowledge of long-term, post-fire C dynamics that occur during succession (Sofronov et al. 2002).

Isaev et al. (2002) developed a map of forest fire damage that could be used to assess C emissions in Siberia. The degree of stand damage in a burned area, estimated from national security satellite images and large–scale aerial photography, correlated well with differences in the normalized difference vegetation index (NDVI) for pre- and post-fire images. Superimposition of stand damage and potential ground/ crown fire maps produced a map which shows the spatial distribution of fire types and intensity and which could also be used to produce estimates of C emissions.

Lightening strikes are a major source of boreal forest fire ignition and, in Siberia, start 30% of all forest fires (Ivanov 2002). A study of seasonal and daily storm dynamics (1986-1992), and storm data over a 20 year period, showed that lightening-induced fire ignition was associated with local storms and occurred most often in stands dominated by light needle tree species (Ivanov 2002). Areas with high geomagnetic anomaly also appeared to have a larger number of lightening strikes.

Volokitina et al. (2002) studied the dynamics of biomass consumed during fire, depending on seasonal and weather factors, and proposed a method to construct large-scale fuel maps in Russia, based on the existing classification of vegetation fuels. Vegetation fuel classes, which play a leading role in fire incidence and spread, and a critical class of drought are included on the map in order to estimate how much biomass will be consumed in a fire.

Fire is not the only important disturbance agent in the boreal forests. Over the period 1920 to 1995, Canada's forests experienced some 2.22 million km^2 of stand-replacing disturbances, of which 48% involved fire and 36% insects (Kurz and Apps 1999, W.A. Kurz and M.J. Apps, unpublished data*)*. The importance of insect defoliation on C storage in Canada's forests is currently being modeled to assess and develop forest management options and strategies for mediating the effects of insect disturbance (MacLean et al. 2002).

Insect defoliation is not only a direct disturbance of the forest ecosystem but may also increase the potential for fire outbreaks. In Ontario, 417 thousand km^2 was defoliated by spruce budworm (SBW) (*Choristoneura fumiferana* (Clem.) at least once between 1941 and 1996 (Fleming et al. 2002) and 7.5% of the areas containing trees killed by SBW were burnt in western Ontario compared to 4.8% in the eastern part of the province. These regional differences may result at least in part from slower decomposition of dead fuels in the drier, western climate. The higher decomposition rate in eastern Ontario not only shortens the length of time after SBW-caused tree mortality during which the fire potential remains high, but also reduces the likelihood that such a stand will be burned before it regenerates. In a climate change scenerio of warmer and drier conditions, it is likely that slower rates of decomposition and more frequent drought conditions will result in more fires in post-outbreak stands.

In addition to direct stand mortality and interaction with fire, insect defoliation contributes to the dieback and reduced growth of trembling aspen observed in some Canadian provinces since the early 1990s (Hogg et al. 2002). Trembling aspen is the most important deciduous tree species in the Canadian boreal forest and is a significant component of the C cycle at the national scale with about 1000 Tg of C contained in the aboveground biomass. Dieback in Grande Prairie, Alberta was related to insect defoliation as well as drought and thaw-freeze events, all of which may be increased by climate change and affect C sequestration by the forested ecosystems of the region.

Windthrow is also a major disturbance over much of the boreal region, creating large amounts of CWD and in the process transferring large amounts of C from storage in trees to storage in the soil compartment where it may have a much longer turnover time. Windthrow was the main disturbance in the unmanaged boreal forests of the Saint Petersburg region of Russia where it strongly influenced the dynamics of C in both living and dead wood (Shorohova and Soloviev 2002). C stored in living trees ranged from 30 to 172 Mg C ha $^{-1}$ and was largest in even-aged stands when the growing stock was increasing (as in the immature phase) or relatively stable (as in the mature phase). The amount of C in CWD (4.5 – 61.5 Mg C ha $^{-1}$) was largest in ecosystems undergoing stand breakup (i.e., where the growing stock declines), such as those stands suffering windthrow. This phase lasted from one to several decades depending on the intensity of the windthrow event and other stand conditions.

EFFECTS OF MANAGEMENT PRACTICES ON CARBON DYNAMICS

Uncertainty surrounding the regulatory policy response to increased global CO_2 emissions makes decisions about how to manage forest resources difficult (Weyerhaeuser 2000). However, most of the choices today are straightforward even in the absence of a definitive international forest C accounting protocol. Reducing fossil fuel CO_2 emissions through energy conservation and increasing C sequestration by growing more wood are both likely to be valuable to producers of forest products.

A market price for reduced C emissions will likely provide incentives for maximizing forest regrowth through practices such as early regeneration and careful stocking control, for minimizing forest waste while maintaining soil productivity and for managing fire and pest outbreaks. There may also be incentives to fertilize forests and to shorten rotation age if carbon storage in forest products is taken into account (although this is not the case under present IPCC guidelines for reporting).

Foreward looking forest product companies, however, will not wait for certainty before they act on the carbon issue but instead are beginning now to assess the C budgets of their forest operations and to introduce ways to mitigate their emmissions. To this end, current forest C models must be refined to better predict the effects of management practices at local levels and longterm field observations need to be put in place to confirm model predictions.

The use of three simulation models to examine the effects of silvicultural and harvesting practices on long-term C storage in North American and Chinese forests were reported. The ecosystem simulation model, FORECAST, was used to investigate the effects of harvest frequency and tree species on C storage in a boreal mixed-wood forest type in northeastern British Columbia, Canada (Seely et al. 2002). The initial conditions were created from a mixed-wood stand on a mesic site of medium nutritional quality with a fire interval of 150 years and a total simulation time of 300 years. The model predicted larger total ecosystem C storage with increased rotation length, regardless of tree species. The influences of species on ecosystem C storage and accumulation rates were, however, evident: for equivalent rotation lengths (•90 yr) total C storage was largest for aspen followed by pine and spruce. Species biomass accumulation rates were highest in the shortest rotations for aspen (< 75 yr), in mid-length rotations for pine (75 yr) and longer rotations for spruce (>100 yr). Soil C showed a slow decline in managed stands relative to those undergoing natural disturbances.

The STANDCARB model was used to examine the effect of various management techniques in a coastal forest dominated by Douglas-fir (*Pseudotsuga menziesii* (Mirab.) Franco) and western hemlock (*Tsuga heterophylla* (Raf.) Sarg) in Oregon, U.S.A. (Harmon and Marks 2002). This forest is distinguished from the boreal forest by its wetter climate and longer growing season. Conversion of old-growth forest to any other management or disturbance regime resulted in a net loss of C whereas the conversion of an agricultural system to forest plantations doubled the total C stored in about 100 years. The model found that C storage increased as rotation length increased but decreased as the harvest intensity (fraction of trees harvested and detritus removed) increased.

The CENTURY 4.0 model was used to examine the dynamics of C stocks and the impacts of different harvesting regimes in the Chinese boreal forest (Jiang et al. 2002). Simulations were run for 2000 years to establish a steady state initial condition, and scenarios were run for 5000 years with analyses performed on the last 400 years. Results were similar to the above models with C storage lower in short rotation (30 yr) forest plantations than in long rotation (200 yr) plantations. The effect of different harvest regimes on total C stocks, averaged over 100 yr, resulted in 131 Mg C ha^{-1} under whole tree harvesting (100% removal of stems, 90% removal of branches), 158 Mg C ha^{-1} under conventional harvesting (100% removal of stems, branches left on site) and 462 Mg C ha^{-1} under a no-harvest reference regime. The model simulations suggest that reducing harvest intensity and extending rotation length may be a good strategy for maintaining the sustainability of production and maximizing C storage in Chinese boreal forests.

All three models showed that increasing rotation length increased C stocks in the forest ecosystems. The FORECAST model (Seely et al. 2002) suggested that by selecting appropriate species combinations and rotation lengths, it may be possible to balance the competing demands of fiber production and C storage. A landscape that includes stands with management practices optimized for timber production and stands with management to maximize long-term C storage could meet both these demands. Simulations with STANDCARB (Harmon and Marks 2002) and CENTURY 4.0 (Jiang et al. 2002) also suggested that an adequate supply of wood products may not be incompatible with practices that increase C stocks. For example, partial harvest and minimal use of fire may provide as many forest products as a traditional clearcut/broadcast burn system while at the same time increasing C stocks in the forest ecosystem.

Short term effects of harvesting techniques reported in two Canadian field studies, supported the results of the simulation models. In north central Ontario, Lee et al. (2002) showed that partial cutting (2/3 volume removed) in a second-growth boreal mixed-wood stand resulted in higher C assimilation rates in woody biomass of the saplings and trees than clearcutting over a five year period, although the results of both treatments were lower than in the unharvested control plots. The forest floor, sampled seven years after harvesting showed no differences in organic mass among the treatments likely because of an increase in litter fall from rapidly re-established ground cover and aspen and similar decomposition rates among the treatments. The authors suggested that if harvesting were to occur, ecosystem C assimilation in boreal mixed-wood would be maximized under a partial cutting regime although the lower timber productivity and higher costs associated with this harvesting technique may reduce its attractiveness.

Direct observations of above-ground and below-ground fluxes of CO_2 on a vegetated five-year old sub-Boreal clearcut in central British Columbia showed that the site was a sink for CO_2 during the growing season (Pypker and Fredeen 2002). However, if the entire year was considered, belowground efflux was sufficiently high to make the site a source on an annual basis. Natural deciduous plants were a much more important CO_2 sink than the planted conifers (375 and 48 g C m^{-3} respectively) and the authors suggest that removal of deciduous "non-crop" vegetation should be avoided to minimize the loss of C in the early stages of stand development.

Nitrogen (N) fertilization is a management technique that not only increases tree productivity but also forest C storage. In the FORECAST model described previously, Seely et al. (2002) simulated the application of 250 kg N ha^{-1} in years 10 and 25 of a 30 year harvest cycle of an aspen stand and found that there was a net increase of 13% in average C storage in all ecosystem pools. Most of the increase was attributed to greater C retention in wood products and biomass pools.

Field studies in Canada and Siberia suggest that the effects of N fertilization vary with site quality and change over time. A productive, overstocked, 45-year old jack pine forest in northeastern Ontario did not respond to normally recommended rates of N fertilization (Foster and Morrison 2002). Fertilization with urea at five levels up to 448 kg N ha^{-1} did not affect either the gross and net stand C increment (compared to a control) after about 10 years or the maximum amount of C in the stand over 30 years, although an increase in dbh in co-dominant trees was noted at 10 years. The unfertilized stand was able to capture sufficient inherent site N to expand its crown mass and N content until maturity and, from a management perspective, productive jack pine stands on high quality sites should be given low priority when considering the use of N fertilization to increase short-term C sequestration.

The effect of N fertilizer on even-aged pine stands (15 to 110 years old) in Siberia showed that 200 and 400 kg N ha^{-1} resulted in a transient increase in the overall phytomass of trees during the first 3 years followed by a second peak in growth 7 to 8 years later (Buzikin et al. 2002). This "echo effect" was small in large diameter trees when fertilizer doses were small and the smallest trees showed a decline in increment after fertilization. The "echo effect" may be attributed to increased production of needles resulting in larger amounts added to the litter layer and increased N recycled through the soil.

Annual additions of complete fertiliser containing ca 70 kg N ha^{-1} to young boreal stands of Norway spruce (*Picea abies* (L.) Karst.) over 15 years in northern Sweden have resulted in over four-fold increase in stem volume and C storage (Gower el al. 2000). Large annual net ecosystem C fluxes of 6 – 7 Mg C ha^{-1} have been measured over the past five years in Sitka spruce on a moderate quality site, intermittently fertilised with urea in oceanic Scotland at a latitude similar to that of the BOREAS sites in Saskatchewan (Gower et al. 2000). However, in the overall context of exchanges of GHGs between forests and the atmosphere, some reservations must be made at the present time with respect to increases in C sequestration as a result of fertilisation with N. There is a possibility, as yet not adequately evaluated, that addition of N-containing fertilisers may lead to significant emissions from forest soils of nitrous oxide, N_2O. Molecule for molecule, this gas has a global warming potential 300 times that of CO_2, so that even small emissions may off-set substantial amounts of net CO_2 uptake.

AFFORESTATION AND CARBON SEQUESTRATION

Afforestation and reforestation are recognized under the Kyoto Protocol as potential mechanisms that may be used by countries to offset their greenhouse gas emissions by increasing the strength of their biological C sinks. Afforestation and reforestation can be defined as the conversion of non-forested lands to forests with the only difference being the length of time during which the land was without forest. Reforestation occurs when planting trees on non-forested land that was forested at any time within the 50 year prior to 1990, whereas afforestation occurs on land that has never been forested or at least was not forested in the similarly defined 50 year period. The achievement of real afforestation gains is tied to the anticipated economic returns from both the additionally sequestered carbon and the additional timber produced.

In Saskatchewan, Canada, the land base potentially available for afforestation (estimated as private lands currently under unimproved pasture and within 100 km of existing mills) is about 260,000 ha and could potentially sequester 5.9 x 10^6 Mg of C over the next 80 to 100 years (Johnston et al. 2002). Economic predictions showed that the desirability of afforestation with aspen, white spruce or hybrid poplar plantations declined as the discount rate increased beyond 3%. Distribution of revenues from timber sales to land owners early in the life of the plantation and putting an economic value on sequestered C would increase the viability of afforestation in Saskatchewan.

Another Canadian study of the afforestation potential in Ontario and the Prairie provinces identified the C price needed for the net present value of afforestation to be equivalent to the opportunity cost of agricultural land use (Stephens et al. 2002). For parts of Ontario, afforestation with red pine may be economically viable at C prices between \$15 and \$35 Mg^{-1} C. For the Prairies, afforestation using hybrid poplar was viable at prices above \$25 Mg^{-1}C.

In the Ukraine, potential afforestation and reforestation of about 2 million ha of low productivity land and wastelands and creation of forest stands along rivers and canals would increase the forest area by almost 20% (Nijnik 2002). However, most of this afforestation would not be feasible using a discount rate of 4% unless the C sequestered had economic value. A Swedish study of forest strategies for reducing GHG, that did not include socio-economic considerations, suggested that within a 10-30 year time frame, afforestation with willow and intensive N fertilization of spruce stands would be the most successful strategies (Olsson 2002), but there are potential, but hidden dangers of N_2O emissions with intensive N fertilisation, as mentioned earlier.

Afforestation can also increase C stocks in the soil and such C is generally considered more resistant to changes in forest management than that stored in biomass. A chronosequence study of oak (*Quercus robar* L.) and Norway spruce planted on a former highly productive arable cropland in Denmark showed that, although C storage in the upper 5 cm of the mineral soil increased, at lower depths it declined as stand age increased to 29 years. This led to similar total (but differently distributed) soil C stocks of around 65 Mg C ha^{-1} along the chronosequence (Vesterdal et al. 2002). The decrease in soil C below 5 cm was thought to likely be a result of C inputs from the young stands being insufficient to match the ongoing decomposition of C inherited from agriculture. The authors suggest that afforestation of nutrient rich arable soils does not lead to significant sequestration of C in soils within 30 years although faster sequestration in afforested soils may occur on poorer soils where organic matter decomposition is slower.

Similarly, in a simulation of conversion of an agricultural system to a forest system in the Pacific Northwest, U.S.A., Harmon and Marks (2002) found that total C storage increased significantly in about 100 years but pointed out the importance of initial soil conditions to the end result.

In addition to its role in C sequestration, there are advantages of afforestation that have not been given an economic value, but are important to sustainability of the resource. Factors influencing this sustainability include soil stabilization, restoration of soil organic matter, maintainance of ground water quality, increasing wildlife habitat diversity and expanding the resource base (Johnston et al. 2002; Nijnik 2002; Vesterdal et al. 2002).

To supplement the benefits of afforestation, biofuels that are produced from wood and used to replace a portion of fossil fuel combustion, may be used to reduce net GHG emissions. O'Connor et al. (2002) used Delucchi's full fuel cycle GHG model to calculate GHG emissions from production and combustion of biofuels under different landscape scenarios. They found that over the long term (100–500 yr), use of forest residues (branches, tops, needles) as a biofuel was effective at reducing GHG emissions. These authors suggest that ethanol produced by short-rotation forestry on afforested land was significantly better at reducing GHG emissions than ethanol produced from short-rotation forestry on previously forested lands (O'Connor et al. 2002).

In order to document actual changes in afforestation and deforestation at a national scale, a suitable monitoring system must be put in place. In Canada, the recently developed National Forest Inventory (NFI) with its 20,000 primary sample units, each 2 x 2 km (400 ha) in size, will provide useful land-use data on forest C but may not be sufficiently precise to accurately estimate afforestation or deforestation that often occurs in small and scattered areas (Leckie et al. 2002). Integration of the NFI system with satellite imagery and existing land use records will improve measurements particularly where deforestation activity is high.

Persistent deforestation documented in southern portions of the boreal forest in Saskatchewan between 1960 and 1990 (Fitzsimmons 2002) also highlight the need for a more detailed accounting system of afforestation and deforestation. A comparison of sequential maps from Canada's National Topographic system allowed an estimate of changes in total wooded area but did not provide specific information on vegetation that would allow a calculation of forest C lost. Rates of deforestation were highest in agricultural regions (-1.21 to -1.27%) and negligible in public forests and parks. The authors suggest that in this region, continued clearing of existing forests could outweigh the potential C gains from afforestation and reforestation.

EFFECTS OF CLIMATE CHANGE AND ELEVATED CO_2 CONCENTRATION ON CARBON DYNAMICS

Accurately quantifying the rate and extent of high latitude warming, and the impacts of this warming on ecological systems should be a high priority for boreal scientists (Running 2000).

The studies included in this section modelled or measured effects of increased annual temperature, precipitation and/or CO_2 concentration on vegetation distribution and C stocks, and ranged in spatial scale from national to individual trees and in time from many years to several growing seasons. Results suggested that while climate change may cause shifts in vegetation distribution that could reduce forest area, forest productivity was generally increased by higher temperature and CO_2 concentrations.

In a study of boreal forests in northeast and northwest China, Ni (2002) found that increased mean temperature (3.5-4° C) and precipitation (51-88 mm), modeled by BIOME 3, resulted in a significant northward shift of the boreal forest. Under the present climate, higher and lower CO_2 concentrations (500 ppmv and 200 ppmv respectively), caused a reduction in the area of boreal forest which indicated a nonlinear effect on boreal forest distribution. Both climate change alone and climate change with CO_2 enrichment significantly reduced the C stored in vegetation and soils.

In Russia, Siberian climate change impacts are expected to be important and complex. Models simulating a 2° C increase in summer temperature and a 20% increase in annual precipitation predicted lowland vegetation to shift 250 m upslope and highland vegetation to shift 450 m upslope resulting in considerable reduction of tundra and light needled taiga, and expansion of the forest-steppe (Tchebakova and Parfenova 2002). The mountain forests surrounding Lake Baikal in Russia filter about 90% of the water flowing into the lake and the health and condition of these forests strongly determines the quality and quantity of its water. Thus, under expected warmer summer temperatures, the filtering capacity of the forest-steppe would be weaker than that of the mountain forests with a resulting potential deterioration of the quality of water in Lake Baikal. This could have significant consequences as Lake Baikal represents as much as ¼ of the world's reservoir of fresh water.

In his study of soil C in Russia, Stolbovoi (2002) hypothesized that soil warming caused by climate change may result in deeper peat accumulation in the tundra zone and expansion of the steppe into the temperate forest zone. Such changes he suggests, would result in increased levels of soil C in Russia in the future.

To examine the possible impacts of climate change on economic welfare and land-use shifts between forestry and agriculture in the United States, Alig et al. (2002) used two global circulation meteorological models to feed into two ecological process models, thereby creating four climate change scenarios for 2070-2100. They found that forest productivity, C storage and total economic welfare are all higher under all scenarios.

The effects of an annual temperature increase of 4°C, an 8.7% increase in annual precipitation and a CO_2 concentration of 700 ppm on the dynamics of the forest at the ecotone of the boreal and temperate forest in Northern China, were simulated using a modified adapted gap model BKPF (Chen 2002). After a 50-year simulation, the density and LAI of the forest regenerating from clearcutting were not significantly different, but the productivity and aboveground biomass were higher, under climate change than under current conditions. The effect of climate change on the current undisturbed forest was to reduce stand density by 20% and aboveground biomass by 90%. Althouth the stand productivity did not vary significantly, a significant change in species composition occurred.

Also in China, Jiang and Zhou (2002) used the CENTURY 4.0 model to simulate the effects after 50 years of a 2° C increase in temperature, a 20% increase or decrease in precipitation and a CO_2 concentration of 700 ppm, on cold temperate conifer forests dominated by *Larix gmelini* (Rupr.) Kuzeneva. Regardless of changes in precipitation, increasing the temperature caused soil C to decline, gross plant biomass and NPP to increase and the annual capacity of the forest to sequester C to increase by about 10%. Doubling the CO_2 concentration above current concentrations resulted in a 10% increase in forest NPP and slightly increased the size of the net C sink.

Yarie and Billings (2002) also used the CENTURY model to evaluate the effect of a 5° C increase in mean annual temperature on future C dynamics in the Alaskan boreal forest. All ecosystems (aspen, paper birch (*Betula papyrifera* Marsh.), balsam poplar (*Populous balsamifera* L.), white spruce and black spruce) showed a higher net ecosystem productivity (NEP) and all required less time after a disturbance to shift from being a C source to a C sink at the higher temperature. The model estimated that current vegetation of the boreal forest in Alaska absorbs approximately 9.65 Tg C yr^{-1} but with the temperature increase could absorb 16.95 Tg C yr^{-1}.

Many studies use models to predict the effects of climate change on forest C dynamics but different assumptions made by these models about ecosystem processes, make comparing their results difficult. The CENTURY 4.0 and the FOREST-BGC models, calibrated for a productive black spruce stand in Ontario were used to predict forest productivity under climate change and their outputs were compared (Luckai and Larocque 2002). Both models predicted similar relative increases in C storage with an average temperature increase of 6°C or an increase in CO_2 concentration to 700 ppmv, but they disagreed on the impacts of increased temperature in combination with elevated CO_2 concentration. Comparison of both models indicated that the representation of critical processes in these two forest ecosystem models is incomplete.

Several field studies on the effects of increased temperature and/or CO_2 concentrations on individual tree species were reported. Stands of Scots pine (*Pinus sylvestris),* at their northernmost limits on the border between Norway, Russia and Finland, were well-suited to revealing climatically induced growth trends (Alekseev and Soroka 2002). Recent increases in radial increment growth (over the last few decades) relative to the period of last registered warming (1930-40) was attributed to both climate warming and higher levels of CO_2.

In a long-term field study of the effect of CO_2 enrichment on four tree species, Sigurdsson et al. (2002) found that a CO_2 concentration of 700 ppmv increased the light-saturated rate of photosynthesis (C uptake) by 49-114%. The relative increase of photosynthesis by elevated CO_2 concentration was found to be strongly dependent on temperature and lead to the suggestion that enhancement of C uptake by elevated CO_2 concentration may be less at high latitudes.

Increased winter temperature (+4°C) resulted in earlier bud break in downy birch (*Betula pubescens* Ehrh.) seedlings after one winter while an increase in CO_2 concentration to 650 ppm stimulated shoot elongation and biomass accumulation in subarctic and southern populations of birch from Norway and increased biomass in Norway spruce (Skre and Næss 2002). Strong interaction effects with nutrient levels were also noted.

The most significant changes in forest C sequestration are likely to occur in the spring when temperature strongly regulates events such as the timing of bud burst and the onset of photosynthesis (Leinonen and Kramer 2002). However, the ability to monitor spring freeze/thaw events at high latitudes is at present severely limited because of lack of people and human infrastructure for monitoring stations (Running 2000).

Remote sensing might provide an alternative technology for high latitude monitoring that is consistent, repeatable and complete in coverage (Running 2000). Past research found that persistent cloud cover and winter lack of solar illumination restricts the use of normal optical remote sensing. Radar research in the early 1990s, however, discovered that the backscatter signal from a radar sensor clearly identifies the distinction between frozen and thawed land surfaces, even under clouds and in complete darkness. A prototype radar satellite information system that would cover the entire 50 million km^2 of seasonally frozen land surface daily, and continuously monitor changes in the area of frozen ground has been developed. Daily freeze/thaw information from such a system provides an interesting new hydroecological measure of seasonal phenology by clearly quantifying the onset of spring snowmelt and somewhat later the budburst phenology of vegetation.

State-of-the-art terrestrial process-based models that currently tend to simulate the onset of spring drawdown of CO_2 too early and underestimate the interannual variablility (Dargaville 2002), would benefit from the increased accuracy provided by such information

Both the effect of increased temperature on spring recovery of photosynthetic capacity and interannual variability were studied in modelling and field experiments. Leinonen and Kramer (2002) compared three models, developed and parameterized for predicting the timing of bud burst in birch, and found that all models predicted earlier bud burst as a result of a 2°C increase in temperature. Comparison of the seasonal changes in the photosynthetic capacity of deciduous and coniferous trees resulting from the temperature increase was made using a fourth model, developed for Scots pine. Although the results of all four models are not directly comparable, since different mechanisms determine the photosynthetically active period in conifers and deciduous trees, they do demonstrate the difference in the temperature effects in models developed for different tree species. The pine model predicted a stronger increase in radiation under changing climatic conditions compared to the birch model.

A comparison of two model studies on the effects of air temperature on the recovery of the photosynthetic capacity of boreal conifers during spring, showed differences in the ecophysiological assumptions related to photosynthetic dormancy during winter, reversibility of the recovery and the effects of frost (Hänninen and Hari 2002). Both models predicted an increase in the photosynthetic capacity as a result of rising temperatures although the model developed and parameterized for Scots pine predicted earlier recovery than the model developed and parameterized for Norway spruce. Additional simulations, carried out to compare predictions for the photosynthetic production of Scots pine, showed that the Norway spruce model predicted 23% higher values than the Scots pine model on average. Both these studies (Leinonen and Kramer 2002, and Hänninen and Hari 2002) suggest that predicting future C sequestration in different tree species requires further model refinement.

Significant effects of comparatively high temperatures in the spring were measured in several field studies. In Canada the responses of a temperate mixed-wood stand in Ontario, and a boreal aspen stand in Saskatchewan, to interannual climatic variability from 1996-1998 were examined using eddy-covariance CO_2 flux measurements (Barr et al. 2002). Both sites showed large interannual variability in NEP but the causes of this variability were different. In aspen, a warm spring in 1998 caused early leaf out and increased photosynthesis but had little effect on respiration. In the mixed-wood forest, the same warm spring also caused early leaf out but increased respiration and drought stress. The contrasting impact of the warm spring on annual NEP at the two sites reflects competing influences of climate change on NEP: spring warming, which promotes photosynthesis and increases NEP, and increased soil temperature and drought, which promote ecosystem respiration and reduce photosynthesis, thus reducing NEP.

The responses of NEP to seasonal and interannual climatic variability were also studied by Arain et al. (2002), in an aspen forest and black spruce forest at the BOREAS sites in Saskatchewan. Warm springs enhanced NEP in both forests but high mid-summer temperatures significantly reduced the NEP at the black spruce forest as a result of higher ecosystem respiration. A newly developed C exchange model predicted that the aspen forest was a weak to moderate C sink while the black spruce forest was a weak C sink in cool years and a weak C source in warm years.

SUMMARY

Considerable progress has been made with improving estimation of C stocks in circumboreal forests. These efforts have effectively lead to the identification of key information and knowledge gaps enabling scientists to focus their research efforts. Advances have been made with improving techniques to increase accuracy of tree biomass estimation from inventory data. Challenges remain to unify and standardize estimation procedures and national data sets that must be brought together from a wide variety of sources including private land owners, private industry and highly diverse political jurisdictions.

Peatlands and other organic soils, which are known to contain a large percentage of the

C reserves in boreal forests remain a major challenge. There is a need to improve approaches to estimating C stocks in organic soils and to understanding of contemporary peatland processes in the context of a changing climate.

Progress has been made with integrating the effects of disturbance (natural and anthropogenic) on age-class structure and interpreting the resulting impact on large-scale temporal C dynamics. Future research in this area will have to integrate more complex relationships between predicted changes in weather patterns, their effect on multiple large-scale natural disturbances, changes in vegetation distribution patterns and their interaction with harvesting and silvicultural practices. The emerging and possibly greater challenge may lie in assembling this knowledge in a spatially explicit framework to facilitate the development and application of forest management tools to mitigate GHG emissions.

Researchers are beginning to tackle the problem of identifying regional or site specific factors that determine variablility observed at larger scales. Many of the papers presented at the conference examined factors controlling variability in soil carbon and detritus and the relationship or feedbacks between them and above-ground productivity. Model simulations in western Europe, where forests are intensively managed and impacts from natural disturbances such as fire are markedly less than in North America and Siberia, have shown that sequestration of C in soil increased in importance over time from 1950 to 2040. Understanding the processes that control C sequestration in soil is important, especially if a management strategy is invoked to reduce CO_2 emissions by controlling disturbance regimes in boreal forests where disturbance frequency is predicted to increase.

This trend towards understanding links between tree species, detrital inputs and soil will be pivotal in the development of management strategies for C mitigation and the increased understanding that is required for large scale assessments. The greatest gains will likely be realized by nesting regional studies within a larger-scale spatial framework that adequately reflects large-scale disturbance patterns (anthropogenic or natural) which are now understood to impact C dynamics strongly through their influence on age-class structure.

Although afforestation and reforestation (as defined in the Kyoto Protocol) are recognized as a creditable mechanisms for C sequestration under the Kyoto Protocol, many questions will have to be resolved for such programs to succeed. At a national scale, monitoring and reporting procedures to accurately estimate changes to areas of afforestation and deforestation need to be developed. At a scale where afforestation management practices could be applied, site specific ecological and regional economic considerations must be addressed to simultaneously satisfy the goals of increasing C sequestration and meeting the economic needs of private land owners. Land identified with potential for afforestation frequently occurs in the forest/grassland transition zone — areas that are expected to be highly sensitive to climatic change. Considerable research will be required to determine suitable tree species, rotations, and site types that are most likely to succeed in an afforestation program.

Tax incentives may be required to motivate land-use change from agriculture with short-term economic returns to forestry with long-term economic returns. Some economic value may have to be placed on the additional benefits of afforestation that contribute to sustainability of a broader suite of forest resources including soil and water quality and wildlife and their habitat.

Large-scale model simulations have tended to conclude that the increase in temperature predicted as a result of climate change will increase NEP. Several model simulations indicate that NPP will increase to a greater degree than the increase in heterotrophic respiration, although this is not uniformly agreed. Whilst addition of a carbon cycle to large-scale models and GCMs in real time is a major step forward, confidence in the the predictions is limited by incomplete understanding the C cycle processes, particularly their acclimation to temperature and CO_2, and linkages to nutrient cycles. Furthermore, many of these large-scale models do not yet adequately include the impacts of changing disturbance regimes on both production and respiration fluxes. Such changes are of particular concern with respect to forests in the boreal zone under a changing climate.

Models that predict vegetation distribution patterns in response to climate change indicate that vegetation zones may shift northwards and to higher elevations in response to an increase in mean annual temperature (MAT) and atmospheric CO_2 concentrations. Cross model comparisons for ecosystem response to both climate change and elevated CO_2 concentrations indicate caution should be exercised when broadly applying a model developed in one region or for one species.

Numerous studies show significant, if poorly understood, changes with altered temperature regimes and that temperature may be a more controlling factor than elevated CO_2 concentration in many forest ecosystems. On shorter time scales (decades) variations in the disturbance patterns that accompany changes in MAT and precipitation patterns may play a dominant role.

An increase in MAT is expected to exert a significant influence on NPP in the boreal region through earlier bud burst and earlier onset of photosynthesis. However, several papers showed that this response is species specific and that there are significant differences in response between coniferous and deciduous treesTemperature-induced causes of interannual variability were shown by eddy covariance studies in Canada to be different for a temperate mixed-wood stand and a boreal aspen stand. Another Canadian study indicated that higher temperatures affect NEP throughout the growing season (not just in spring) and that the nature of the effect is a function of stand type (Arain et al. 2002).

Assessments of C fluxes in boreal forest ecosystems suggest that while most stands may currently be sinks for C, natural disturbances such as fire and insects, and tree harvesting, make a forest region suseptible to becoming a C source. In Ontario, Canada, such a situation appeared to have occurred in the last 30 years, while in Russia, the large proportion of mature and over mature stands suggest that this region may also become a source as harvesting and natural disturbances increase.

In contrast to this, other work suggests that climate change will cause shifting vegetation patterns, increased soil C and higher forest productivity that could result in higher C sequestration in the boreal forest. Ongoing research into the effects of a warmer, earlier spring season, changes in C storage in peatlands and mineral soils, and better monitoring of C losses due to natural disturbances will all contribute to a more accurate assessment of future C dynamics. Evaluation of contemporary forest management practices suggest that longer rotations and partial harvesting can sequester additional C, and these gains can be consistent with timber production goals. Management simulations, however, need to be refined to better estimate the role of soil C and understorey vegetation in order to give a more complete picture of C dynamics in managed forest systems.

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INCREASING FOREST PRODUCTIVITY: THE IMPACT OF NITROGEN FERTILIZATION WITH REGARD TO THE "ECHO EFFECT"

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ABSTRACT

The effects of different amounts of nitrogen fertilizer (Carbamid) on even-aged pine stands in the Middle Siberia (Russia) were examined (*Pinus sylvestris*, 15 to 110 years old, lichen and green-moss forest floor). An analysis of the long-term dynamics of radial and height growth revealed an "echo effect" associated with the nitrogen fertilizer application. Large amounts of fertilizer increased the phytomass of trees in the first three years, followed by a second peak of growth 7 to 8 years later. With this "echo effect" a single application of nitrogen fertilizer in Siberian pine stands will result in a steady increase in phytomass growth (radial or height increment) for up to 10 years. A mathematical model taking into account this "echo-effect" was developed. It describes the dependence of increment change on fertilizer dose.

INTRODUCTION

To increase carbon stored in boreal forest ecosystems, there needs to be an increase in productivity. One method of increasing the productivity of forest stands is the use of fertilizers. Fertilizer application must be optimized to provide the greatest possible increase in wood production with the minimum possible expense.

Extensive research of fertilizer application in forests was started in the 50-60s of the XXth century, almost simultaneously, in a number of countries in Europe, Asia, and America. Results from these studies indicated the use of fertilizers is efficient when the productivity of forest coenoses is not limited by unfavorable hydrological, coenotic and climatic factors. The research of the fertilizer's efficacy in forest stands became focused on the definition of a biologically and economically justified dose, additional increment, and of the duration of the fertilizer's impact (Pobedov, 1972). The period of damping and cease of the fertilizer's impact was not studied. Yet, for agricultural crops, it has been shown that fertilizer remnants accumulate in soil due to prolonged application of fertilizers. The result is higher crop yields in comparison with unfertilized soils, for quite a long time (Cook, 1970).

Consequently the present work is devoted to the analysis of long-term effects of fertilization on tree growth.

MATERIALS AND METHODS

Experiments on the estimation of nitrogen fertilizer (carbamide) influence on tree growth were started in 1982 in two study areas in the region adjacent to the Angara river (Siberia, 55°15'N101°30'E, test sites 1-5, and 57°20'N 96°45'E, test sites 6-11, Table 1). Even-aged pine stands (*Pinus sylvestris*) 15 to 110 years old, with a lichen and green-moss forest floor were examined. The sites are located on deep sandy and sandy-loam deposits. Soils are sandy-loam and sandy, loose, light, permeable and aerobic (Buzikin et al., 1983; Buzikin et al., 1996).

Each test site was divided into sections. One section served as a control and the other sections were fertilized by different doses of carbamide. Every test site included at least 800 trees with at least 150 trees on each section (Table 1). Fertilization took place at the beginning of the summer in 1982 or in 1983. Nitrogen was applied at the rates of 0, 50, 100, 200 kg per ha on all sites and the young forests received an additional treatment of 400 kg per ha. The choice of nitrogen for this experiment was based on the fact that most forest soils are deficient in this nutrient, which is necessary for woody plant growth (Prokushkin, 1982; Pshenichnikova, 1991).

Table 1. Test sites characteristics for fertilizing.

Pine forest type	Age (years)	Amount of sections	Amount of trees in section	Section size (ha)	Doses of fertilizer kg per ha	Date of fertilizing
1. Lichen forest	15	5	582,689, 664,552, 550	0.01	0, 50, 100, 200, 400	June,1983
2. Green-moss forest	30	5	1033, 1378, 817, 809, 1035	0.04	0, 50, 100, 200 400	June,1982
3. Lichen forest	35	4	452, 287, 301, 256	0.01	0, 50, 100,200	June,1982
4. Lichen forest	35	4	No date	0.04	0, 50, 100,200	July, 1982
5. Bilberry --green-moss forest	45	4	486, 389, 411, 287	0.04	0, 50, 100,200	June, 1982
6. Bilberry--green-moss forest	45	4	422, 430, 366, 370	0.06	0, 50, 100,200	June, 1983
7. Bilberry –green-moss forest	70	4	197, 270, 287,295	0.06	0, 50, 100,200	July, 1983
8. Lichen –bilberry – green-moss forest	85	4	227, 239, 255, 242	0.06	0, 50, 100,200	June, 1983
9. Lichen forest	85	4	167, 116, 161, 152	0.06	0, 50, 100,200	June, 1983
10. Lichen forest	90	4	No date	0.06	0, 50, 100,200	June, 1983
11. Lichen –bilberry - forest	110	4	145, 143, 147, 121	0.18	0, 50, 100,200	July, 1983

Long-term response of trees to a single application of different doses of nitrogen was studied by measuring radial increment. In addition, height increment was measured on young forests stands. Increment cores or cross-sectional samples were taken from all model trees at a height of 1.3 m. Height increment was measured, for the cut down model trees, as the distance between whorls. Samples were taken 15-17 years after fertilization. Annual increments were measured, including the year preceeding fertilization.

To eliminate a climatic influence and stand coenotic heterogeneity on the assessment of fertilizer efficiency we used a special procedure. The method used to assess the temporal rows (time series) of the tree increments was as follows. The averaged value of a tree increment in diameter (or in height) for the trees of the same diameter (or height) class were divided by the increment registered in the year previous to the fertilizer application. Then, the normalized increment value for any one fertilizer treatment was divided by the value of the normalized control tree increment for the same year. Thus, the variability in tree increments that are due to weather, location of trees in a stand and the age trend were eliminated. Further analyses were performed on the normalized and averaged data.

The $M(t, H)$ value was defined as the impact effect;

$$M(t,H) = \frac{m(t,H) \big/ m(0,H)}{k(t,H) \big/ k(0,H)} \qquad [1]$$

where;

$m(t,H)$ = the average increment of trees with the initial height of H in year t;

$m(0,H)$ = the average increment of trees with the initial height of H in the year preceeding fertilization;

$k(t,H)$ = the average increment of control trees with initial height H in year t;

$k(0,H)$ = the average increment of control trees with initial height H in the year preceeding fertilization.

The formula used for the impact effect on radial increment $E(t,R)$ took a similar approach.

RESULTS AND DISCUSSION

Is the phytomass of the stand increased after fertilization? The impact effect value I (equation [2]) was used to estimate the influence of fertilizers at the stand level.

$$I = \sum_{i}\sum_{j} p(i) \cdot x(i,j) \qquad [2]$$

where;

$p(i)$ = the share of trees having diameter class i in the stand;

$x\ (i,\ j)$ = the average normalized radial tree increment of diameter class i in year j.

If the fertilizer dose is raised, the impact does not increase directly proportional to the dose. The relation "effect (in relative units)/dose (in kg/ha)", falls from a value of 0.022 to a value of 0.0066. This is consistent with the well established effect of diminishing growth response to an increase in fertilizer dose (fertilizer efficiency falls) (Fig. 1). Once fertilizers impact a phytocoenosis directly, the after-effect persists for a long time through their influence on processes such as mineralization of nutrients and long-term nutrient cycling. However, with each subsequent cycle the response of trees to the fertilization effect dampens out.

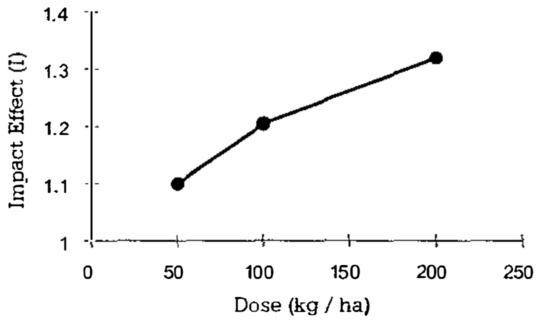

Figure 1. Impact effect of different doses of fertilizer on radial increment of pine tree stand.

Figure 2 shows the effect of fertilizer impacts on tree height increment over 16 years after a single application of **N** applied at 400 kg per ha in a young forest stand. The trees, having initial heights of H = 100 cm, have two peaks for the fertilizer impact effect value. The first peak is observed after 4-5 years following the fertilization and the second peak after 8-9 years. The first peak can be attributed to the direct impact of fertilizers on tree growth, but the second peak is caused by the "echo-effect" alone.

Sixteen years later the fertilization effect value $I(t,\ H)$ of the fertilized trees tends to 1 (i.e. the increment of control and fertilized trees becomes identical).

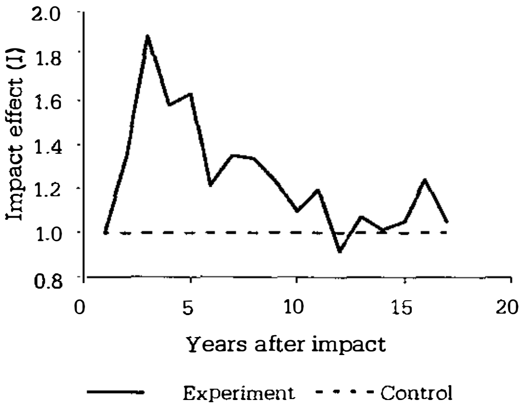

Figure 2. Impact of fertilizer dose 200 kg/ha on normalized radial increment dynamics for trees in a 12-cm diameter class.

The "echo-effect" is also confirmed by the data analysis of the influence of different doses of fertilizers on the radial increment dynamics. The dynamics of the normalized radial tree increments clearly show a double-phase response to nitrogen at a dose of 200 kg per ha (Fig.3). The first response peak is observed 3 years after the fertilizer application and the second peak occurs 7-8 years after fertilization.

The curve characteristics of the trees' response to fertilization are given in Table 2. The maximum of the fertilizer's direct influence (M1) is observed 3-4 years after fertilization, and the maximum "echo-effect" (M2) follows in the seventh year. The "echo-effect" is not observed at small values of M1. With values of M1 up to 1.2, the corresponding values for M2 are similar to those for M1. Thus, the increase in growth due to the direct impact from fertilization does not always result in a similar increase in the "echo-effect".

The "echo-effect" is small or not observed in large diameter trees when fertilizer doses are small (Table 2). When fertilized, the smallest (4 cm) and largest (20 cm) trees show a decrease in their increment when compared with the control. The similar effects indicate that the nature of a tree's response to fertilization depends not only on the fertilizer's dose, but also on the tree's size and it's position in the coenose.

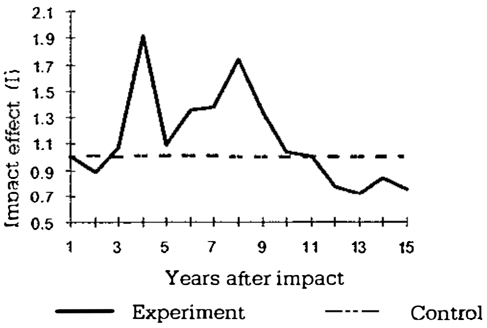

Figure 3. Impact of fertilizers dose 400kg/ha on normalized radial increment dynamics for trees in the 1 m height class in young forest stands.

Thus, these data testify to the existence of an "echo-effect" in trees that respond to fertilization.

How can the "echo-effect" be explained?

A prolonged effect of fertilizers on trees is possible. It is related to the fact that trees absorbed fertilizers not during one vegetation season as do annual plants, but during time T, which is longer than one season. Tree response to fertilization does not take place immediately, but after some time τ. The dependence "dose-effect " can be described by the equation;

$$M(t) = \int h(\tau)*D(t-\tau)d\tau, \qquad [3]$$

where;

$M(t)$ = effect (phytomass changes in relation to control);

$h(\tau)$ = is a response describing assimilation of fertilizers by a plant at any moment τ,

$D(t-\tau)$ = dose of fertilizers, applied in the moment $(t-\tau)$.

The simplest function that describes the core $h(\tau)$ of equation [3] can be as follows;

$$h(\tau) = A\tau \exp(-\alpha\tau), \qquad [4]$$

where; A and α = some of the coefficients

Table 2. Response of trees to different doses of fertilizers.

Dose of fertilizers	Diameter of trees, cm	The curve characteristics [1]					
		M1	T1	M2	T2	R	M1/M2
50 kg/ha	4	1.19	5	1.28	7	1.11	0.93
	8	1.14	4	1.28	7	1.16	0.89
	12	1.12	5	-	-	1.03	-
	16	1.22	3	-	-	1.14	-
	20	1.07	7	1.07	13	0.81	1
100 kg/ha	4	1.28	5	1.3	7	1.12	0.99
	8	1.52	4	1.48	7	1.26	1.03
	12	1.6	3	1.33	7	1.19	1.20
	16	1.71	3	1.24	7	1.15	1.38
	20	1.31	4	1.32	10	1.11	0.99
200 kg/ha	4	1.49	5	1.52	7	1.25	0.98
	8	1.8	4	1.65	7	1.43	1.09
	12	1.89	3	1.35	7	1.25	1.40
	16	2.12	3	1.38	7	1.3	1.54
	20	-	-	-		0.54	-

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[1] M1 – range of the first maximum; M1 – time of the first maximum. M2 – range of the second maximum, M2 - time of the second maximum. R - average reaction of trees to fertilizer application for all years of observations.

The function $h(\tau)=0$ when $\tau=0$, reaches a maximum at $\tau=1/\alpha$ and tends to zero with an increase in growth τ. At high values of α the function $h(\tau)$ quickly reaches its maximum. High values for α describe the case when fertilizer is quickly absorbed by the tree which responds with an increase in growth. When α is small the rate of fertilizer uptake is slow. The coefficient values α may be reduced if the supply of fertilizers to the tree roots is lowered because of leaching out by precipitation, gaseous losses, etc.

Setting equation [4] as a core, and the view of function D (t) describing the dynamics of fertilization, it is possible to solve for A and α in equation [4]. The simplest type of function D(t) can be the instantaneous function $\delta(t)$, which is appropriate to the case of a single fertilizer application. Having solved for these coefficients, it is possible to calculate the impact effect of fertilizers given any of the strategies of their input, i.e. no matter what the view of function D(t) .

For trees, besides the direct influence and delayed effect of the fertilizer itself (described in the linear model [3]), one more impact mechanism is possible. It is connected with the existence of the "echo-effect" which can be described as a second tree growth increment after a single fertilizer application. The echo-effect may be due to an increased fall of tree leaves (needles) from an increase in phytomass production in response to fertilization. After several seasons the larger amount of litter that transforms and decomposes into mineral elements, plays the role of an additional fertilizer influencing tree growth.

The equation for the tree response to fertilizer application at a dose of D at time ($t-\tau$) can be written to account for the "echo-effect":

$$M(t) = \int_0^t A\tau \exp(-\alpha\tau)D(t-\tau)d\tau + \int_0^t BM(t-\theta)\exp(-\beta t)d\theta \qquad [5]$$

where A, α, B, β are coefficients and θ is the characteristic delay time.

The first term on the right hand side of the equation describes the direct influence of the fertilizer, and the second term describes the "echo effect" which depends on the trees' response M at the time ($t-\theta$). The exponential damping coefficient $\exp(-\beta t)$ characterizes the loss of mineral nutrients after several cycles of use by trees.

In general, function [5] will have two maxima at some values $t \neq 0$, and for any values of its factors it will asymptotically tend to zero at $t \rightarrow \infty$ (Fig.4). This describes the obvious physical condition that during an indefinitely long time period the impact response drops to zero.

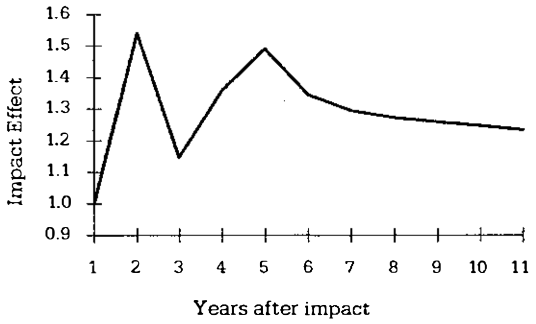

Figure 4. Typical curve described by equation [5]. Modeling parameters in equation [5]: $A = 3$; $B = 0.3$; $\alpha = 1$; $\beta = 0.1$; $\theta = 4$

The function M(t) will vary depending on the relationships between the parameters A, α, B and β in equation [5]. When B is small and α is large there is practically no "echo-effect", and the fertilizer is taken up by the trees right after fertilization. In this case, $h(\tau)$ is approximately equal to $A\exp(-\alpha\tau)$ (i.e. fertilizer efficiency drops exponentially with the time).

The fact that the "echo-effect" does not occur when the impact doses are small is taken into account by [5]. However the absence of the "echo-effect" at the high doses of fertilizers and in the largest trees in a stand is not described by model [5]. It is necessary to enter correction factors into the model [5] to take into consideration the diminishing usefulness of large doses of fertilizers, especially for the dominant trees.

Experimental data for the long-term response of annual tree increment dynamics to a single application of fertilizers testifies to the existence of the "echo-effect".

CONCLUSIONS

Analysis of data on tree increment dynamics following a single nitogen fertilizer application shows the occurrence of an "echo-effect" from fertilizer application.

The "echo-effect" phenomenon can be described as follows. A single application of a fertilizer at doses large enough to produce an observable growth increment within 2-3 years, is followed 7-8 years after the application, by a second growth increment peak. The second peak may be due to the use of mineral nutrients recycled through the ecosystem. Trees and coenoses reaction to fertilization has a fluctuating delay response. The addition of nitric fertilizer represents an external trophic influence on forest ecosystems. The impact of fertilizers does not differ essentially from the impacts of the weather-climatic factors (precipitation, heat). One can assume that after sharp weather anomalies the response of trees and coenoses will fluctuate over the long-term.

The analysis of experimental results on the fertilizer impacts on tree growth has shown that the use of fertilizers can raise the phytomass increment in a stand. The nature of the fertilizer impacts on trees and stands depends both on the dose of the fertilizer input and on the inventory parameters of trees. (For the dominant trees in the stand the impact effect can be insignificant).

The occurrence of an "echo-effect" 8-9 years after a single fertilizer application proves that it makes sense to not carry out repeated fertilization earlier than 12-15 years following the first application. By that time both the direct fertilizer impact effects and the "echo-effect" will equal nil.

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SIMULATION OF ALTERED FIRE REGIMES AND IMPACTS ON BOREAL CARBON DYNAMICS

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ABSTRACT

The physical and ecological effects of future altered fire regimes on boreal tree community and carbon dynamics were examined in a simulation study. Species composition, tree density and biomass for aspen stands in Elk Island National Park (EINP) and Prince Albert National Park (PANP), and white spruce-aspen stands in PANP were modeled under present and future fire regimes. Climatic conditions provided by the Canadian General Circulation Model for the periods 1975-1995 and 2080-2100 were used to estimate fire regimes. Each forest stand simulation was run for 400 years to allow the stand dynamics to reach equilibrium with fire regime. Each simulation scenario was repeated 25 times and averaged. In the aspen stands, average stem density was greater under the 2080-2100 fire regimes for both National Parks, but the total stand carbon decreased in PANP and increased in EINP. Aspen stem density was very low in EINP under the 1975-95 fire regime. In the white spruce-aspen stands of PANP, aspen stem density increased, white spruce density decreased and total stand carbon increased slightly under the 2080-2100 fire regime. All of these changes in stand characteristics were due to changes in fire cycle. The simulations also support previous studies that indicate a future increase in fire severity and fire intensity, but their influence on stand characteristics was minimal. From this study, it appears that fire-induced change in vegetation dynamics will have a greater effect on the boreal carbon balance than any change in carbon loss due to fire.

INTRODUCTION

The northern circumpolar boreal forest stores about 800 Gt carbon, or one-third of terrestrial ecosystem carbon. Because of this high storage level, even small changes in boreal carbon dynamics can have significant effects on atmospheric carbon levels. Fire is a dominant force in the boreal forest as 5-10 M ha are burned annually by wildfire in the circumpolar boreal region. In Canada, stand-replacing crown fires burn 2-3 M ha each year with typical fire cycles of 75-150 years depending on the local fire regime. Over millenia, tree species have adapted to this environment in different ways through fire survival and regeneration strategy. Fuel dynamics also have a feedback effect on fire regime through flammability and fuel load. Because of the prominent role of fire in boreal vegetation dynamics, carbon dynamics are intrinsically tied to fire regime and the fire ecology of its vegetation.

As a result of climate change in the Canadian boreal region, future fire regimes are expected to support a general increase in fire intensity, fire severity (depth of burn) and fire season length (Flannigan and Van Wagner 1991, Flannigan et al. 1998). From a physical standpoint, such a change in the fire regime will have a strong effect on the forest disturbance rate (annual area burned) as well as the carbon release rate that occurs during fires. Ecologically, a change in the fire regime will favour some species over others and cause a shift in species composition. This can affect carbon sequestration rates because of different growth rates in different tree species. Current research on fire and climate change has focused on the physical aspects of altered fire regimes on carbon dynamics but very little has been done to study the ecological impacts on the boreal carbon budget. The purpose of this study was to examine both the physical and ecological effects of future altered fire regimes on boreal tree community and carbon dynamics.

STUDY DESCRIPTION

This study used a simulation approach to compare carbon budgets under current and future fire regimes. Two study locations were chosen in west-central Canada to represent the low and mid-boreal regions at similar elevation and latitude positions. Elk Island National Park (EINP) is located in central Alberta (Figure 1) and represents the aspen Parkland region found at the southern extent of the boreal forest. This forest is dominated by *Populus tremuloides* Michx. (aspen) with some stands comprised of *Picea glauca* (Moench) Voss (white spruce) – *Populus tremuloides* mixtures. In central Saskatchewan, Prince Albert National Park (PANP) represents the mixedwood boreal region with pure and mixed stands of *Populus tremuloides, Picea glauca, Picea mariana* (Mill.) B.S.P. (black spruce), *Pinus banksiana* Lamb. (jack pine), *Betula papyrifera* Marsh. (white birch), *Larix laricina* (Du Roi) K. Koch, *Populus balsamifera* L. and *Abies balsamea* (L.) Mill. (balsam fir). Aspen stands comprise 87% of EINP and 31% of PANP. White spruce-aspen stands represent another 19% of PANP. The climate of Prince Albert National Park is more continental with slightly colder winters and warmer summers, and lower precipitation (Atmospheric Environment Service 1993). For the purposes of this study, carbon budget comparisons were made under current and future fire regimes for aspen-dominated stands in both National Parks and for white spruce-aspen stands of PANP to illustrate climate change effects in the two boreal regions.

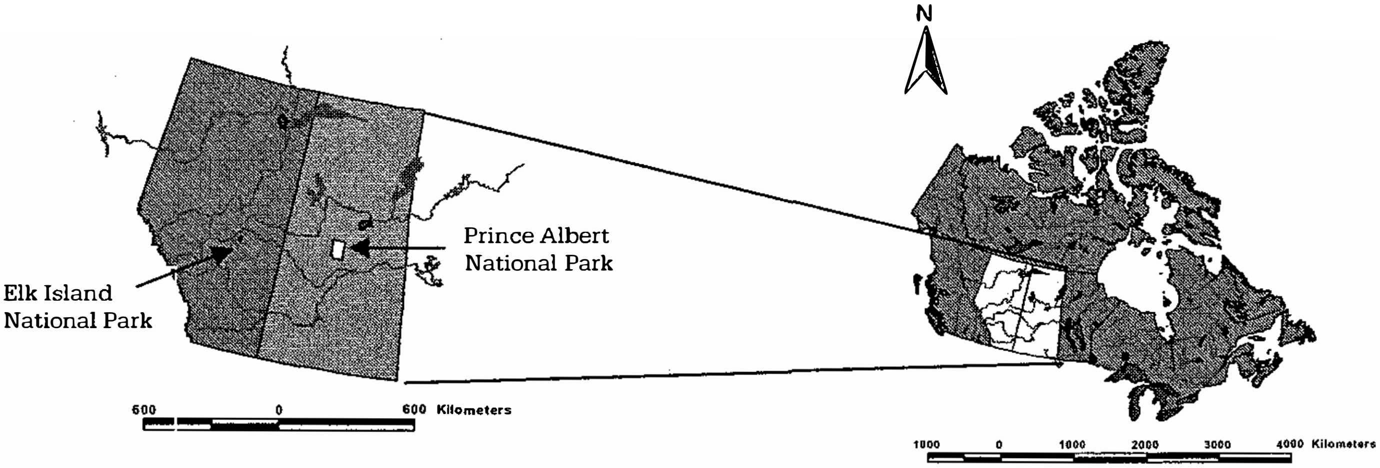

Figure 1. Location of Elk Island Nationl Park and Prince Albert National Park study areas.

CLIMATE AND FIRE REGIME DATA

Fire regime is characterised by parameters of fire frequency, fire intensity, fire severity, season of burn, type of fire (crown, surface or ground fire) and fire size (Malanson 1987, Whelan 1995). Comparison of vegetation under present and future fire regimes for the study was based on climatic conditions estimated by the Canadian General Circulation Model (Flato et al. 2000). The model provided values of daily temperature, precipitation, relative humidity, wind speed and growing degree days (GDD, average of maximum and minimum temperatures above 5°C) for the periods 1975-1995 and 2080-2100 at both study locations. The first four parameters were applied to the Canadian Forest Fire Weather Index (FWI) System (Van Wagner 1987) to provide daily estimates of burning conditions. This included the Duff Moisture Code (DMC) which represents the level of dryness of loosely compacted forest floor organic matter and depth of burn; the Buildup Index (BUI) which is an indicator of the total amount of fuel available for combustion and can be used to estimate dead woody fuel consumption; and the Initial Spread Index (ISI) which indicates the rate of fire spread and is used with fuel load to determine fire intensity. These codes were used to simulate the fire severity, fuel consumption and fire intensity characteristics of fire regime. The daily FWI System data used to describe fire regime was summarized by month for each time period (Table 1).

Table 1. Summary of fire regime data for Elk Island National Park and Prince Albert National Park. Average monthly values (+/- SD) of Duff Moisture Code (DMC), Initial Spread Index (ISI) and Buildup Index (BUI) are presented. Estimates of area burned for 2080-2100 are based on monthly DMC correlation with actual area burned during 1975-95 (see text for procedure).

Elk Island National Park				
Month	Area Burned (ha)	DMC	ISI	BUI
		1975-1995		
April	24,295	1.7 (2.1)	0.6 (0.8)	2.1 (2.4)
May	387,331	5.5 (3.7)	1.3 (1.0)	7.9 (5.2)
June	710,375	13.3 (7.6)	1.8 (1.2)	20.1 (10.5)
July	256,508	18.4 (9.5)	1.7 (1.0)	30.1 (13.5)
August	728,332	17.2 (10.9)	1.4 (1.0)	29.8 (17.4)
September	514,700	15.9 (11.3)	1.7 (1.0)	28.1 (18.5)
		2080-2100		
April	62,881	4.4 (2.7)	1.0 (0.9)	5.3 (3.3)
May	652,724	9.1 (6.4)	1.4 (1.0)	12.4 (8.4)
June	1,052,210	19.7 (13.7)	1.8 (1.2)	28.6 (17.5)
July	347,122	24.9 (14.9)	1.8 (1.0)	39.7 (20.9)
August	885,008	20.9 (13.7)	1.6 (1.0)	35.5 (21.4)
September	550,308	17.0 (12.6)	1.6 (1.0)	30.0 (20.9)

Prince Albert National Park				
Month	Area Burned (ha)	DMC	ISI	BUI
		1975-1995		
April	26,223	3.9 (2.3)	1.7 (1.6)	5.0 (2.7)
May	1,635,324	10.3 (8.4)	4.6 (4.9)	14.2 (10.8)
June	1,825,268	25.3 (13.6)	6.2 (4.7)	34.9 (17.4)
July	1,660,594	34.0 (20.3)	5.8 (3.7)	49.8 (26.4)
August	1,187,969	49.9 (31.0)	5.5 (3.4)	74.2 (40.1)
September	33,445	48.0 (34.1)	5.7 (4.3)	74.4 (46.3)
		2080-2100		
April	69,928	10.4 (7.4)	3.6 (3.7)	11.6 (7.7)
May	3,905,725	24.6 (12.8)	5.0 (3.4)	30.8 (14.3)
June	2,041,703	28.3 (14.8)	5.0 (3.9)	39.8 (18.3)
July	1,768,044	36.2 (19.8)	5.4 (3.6)	53.6 (26.0)
August	1,002,274	42.1 (23.1)	5.3 (3.3)	65.8 (30.6)
September	29,125	41.8 (22.1)	5.3 (4.1)	68.1 (31.8)

The GDD data was used to estimate important season of burn dates based on leaf-flush for aspen and birch, and green-up of herbaceous understory plants. Aspen leaf-flush occurred in the model at GDD=115 when leaves expanded (Parry et al. 1997) and birch leaf-flush occurred at GDD=155 (estimated from Parry et al. 1997, Burton and Cumming 1995). Green-up of understory vegetation was estimated as the time of full leaf expansion of aspen at GDD=225 (Parry et al. 1997) when forest floor shading is highest in the spring and solar drying of understory fuels is lowest. Leaf-fall was estimated as the time when average daily temperature dropped below 3.5°C. These parameters were used in the model to simulate the season of burn characteristics of the fire regime.

Because of the short (20-yr) time periods used in the study and limited size of the two study areas, the fire frequencies were estimated from provincial values. For EINP, the 1975-95 fire frequency represents the annual rate of area burned (as a percent of total area) in Alberta during that time period. This was calculated on a monthly basis using the actual average monthly area burned data for 1975-95. Harrington and others (1983) studied the correlation of monthly area burned to monthly FWI System values and DMC was found to show the strongest relationship in most cases. For this study, average monthly DMC values for 2080-2100 (as provided by the Canadian GCM) were used to estimate future monthly area burned. Any difference in average monthly DMC values between 1975-95 and 2080-2100 was proportionately reflected in the average monthly area burned. Fire frequency for PANP was similarly estimated using Saskatchewan data. The actual fire cycles for 1975-95 were 263 years for EINP and 88 years for PANP; the 2080-2100 estimated fire cycles were 196 years for EINP and 65 years for PANP.

MODEL DESCRIPTION

Simulation of the effects of fire regime on vegetation and carbon dynamics was done using a basic ecological model of boreal fire effects (BORFIRE) which was developed for this study. The model quantitatively simulates tree community dynamics (species composition and stand density, average tree height and diameter) and biomass (above- and below-ground, live and dead organic material) using separate submodels. Changes in those two state variables was based on processes of tree mortality, tree recruitment, tree growth, biomass decomposition, and biomass consumption by fire (Figure 2). Fire was the main forcing variable and was included as the third submodel. Tree community dynamics were driven by fire disturbance events which affected recruitment and mortality, and natural thinning due to competition. Biomass component values were the product of species composition and stand density, and dead organic matter accumulation. The model is process-driven using an annual time-step and simulates conditions at the stand level.

Tree Community Submodel

The model simulates the community dynamics of six major boreal tree species: aspen, white spruce, jack pine, black spruce, white birch and balsam fir. Each simulated stand may include any number of species. Stand density (stems/ha, by species) at the end of each time-step was a result of recruitment and mortality to the initial stand density. Recruitment occurred after a fire event as new seedlings for jack pine and black spruce, and as new sprouts for aspen and white birch. White spruce and balsam fir recruitment was possible at each time-step if a seed source was available and both species could regenerate in the understory of aspen or birch. All other species were shade intolerant and could not regenerate under a canopy. For the purposes of this study, all stands were assumed to be fully stocked. Mortality was separated into fire and natural mortality (or thinning). Natural stand thinning followed the fully stocked stand density algorithms of the Alberta Phase III Inventory (Alberta Forest Service 1985) for aspen, white spruce, black spruce, jack pine and mixedwood stands (any conifer/hardwood mix) which were based on Alberta and Saskatchewan forest inventories. Western boreal databases were not available for balsam fir or white birch, so similar data from Ontario was used (Plonski 1974, MacDonald 1991, Payandeh 1991). In the case of non-pure stands, thinning occurred proportionately to stand density by species.

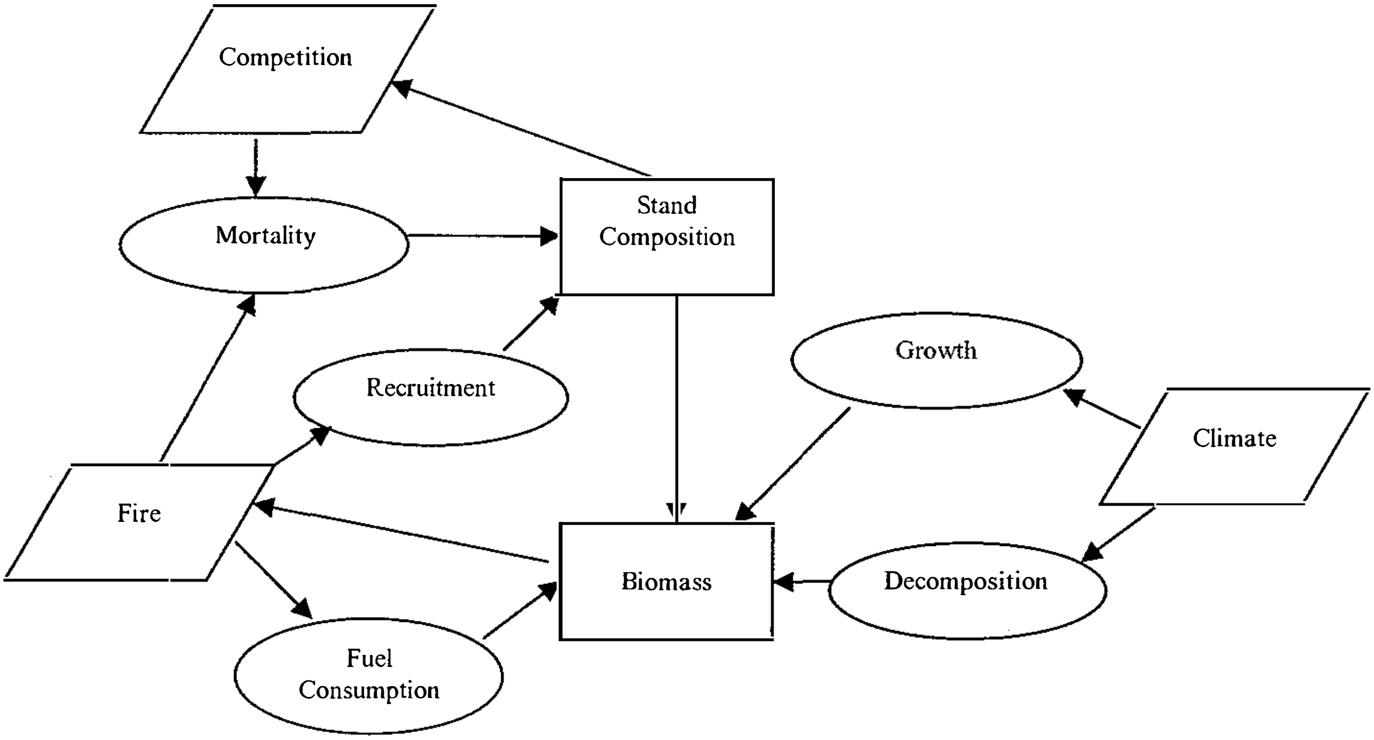

Figure 2. Simplified structure of the BORFIRE model showing state variables (square), processes (ellipse) and driving variables (parallelogram).

Biomass Submodel

Biomass was separated into live and dead, and above- and below-ground components. Live aboveground tree growth rates followed the Alberta Phase III Inventory and Ontario algorithms. The live below-ground (root) component was calculated as a proportion of aboveground biomass using Kurz et al. (1996). As a result of natural thinning, trees were killed and transferred to the appropriate aboveground and below-ground dead biomass pools. Standing dead stems fell at a rate of 10% annually and they were transferred to surface slash pools of branchwood (10% of tree) and coarse woody debris (CWD, 90% of tree). Respiration of slash branchwood used the maximum decay rate of the fast soil carbon pool (representing a half life of 3-20 years) in the boreal west region of the Carbon Budget Model of the Canadian Forest Service (CBM-CFS, Kurz et al. 1992, Kurz and Apps 1999). The CWD respired at the maximum decay rate of the medium soil carbon pool (half life of 20-100 years) and standing dead stems respired at the medium soil carbon pool minimum decay rate.

Dead below-ground biomass was separated into three pools: a surface litter layer of dead foliage and fine (<1cm) woody material, a duff layer of loosely-compacted surface organic matter, and dead roots. Leaf and needle detritus was an annual input to the forest floor litter layer, and litter was similarly transferred to duff using rates of Keane and others (1989). Through decomposition, branchwood and CWD was transferred from aboveground slash to the duff pool at a rate of 10% and 5% per year, respectively. In each timestep, 10% of dead root biomass and 73.5% of live fine roots (Kurz et al. 1996) were input to the duff layer. Respiration of litter and duff used the rates of fine and medium soil carbon pools for the west boreal region of the CBM-CFS (Kurz et al. 1992) with the maximum rate occurring in the first year after fire and declining to the minimum rate by year 100. Dead root respiration similarly used the medium soil carbon pool rate.

Fire Submodel

Fire is the main driving variable affecting post-fire vegetation response in BORFIRE. The tree community submodel uses the physical characteristics of individual fires (intensity, severity, season of burn) to simulate plant response (tree death, recruitment) based on the fire ecology of each species. Fire frequency is incorporated as time since the last fire, or tree age. In the biomass submodel, fire also affects the physical structure of the stand through loss of biomass during combustion and transfer between pools (e.g., live to dead, aboveground to below-ground).

Fire ecology and effects

Because the main purpose of the tree community model was to simulate the ecological effects of altered fire regimes, the interaction of physical fire characteristics and fire ecology is critical. The most important fire ecology traits for each species were summarized (Table 2) and each trait was quantified in the model as it affected tree mortality and recruitment under different fire conditions. Recruitment of aspen, jack pine and black spruce followed the algorithms of Greene and Johnson (1999). To simulate postfire re-sprouting from the root collar in white birch, recruitment was incorporated as a replacement of surviving individual trees. There are no good recruitment models for balsam fir or white spruce, so a basal area approach was used in a very similar fashion to Greene and Johnson (1999). Balsam fir used the same algorithm as black spruce, and white spruce used a simple factor of surviving tree basal area to basal area of a fully stocked pure stand. Long distance seeding of aspen and white birch was considered of minimal importance (see Greene and Johnson 1997, Greene et al. 1999) and was not included in this model. In order for regeneration to occur in the simulated stand, a propagule bank (aspen, jack pine, black spruce, white birch) or a surviving individual (white spruce, balsam fir) was required. Reproductive age and maximum lifespan were used to define the reproductive period for each species (Table 2). Tree mortality was based on the amount of crown scorch and cambium death during fire using algorithms from Ryan and Reinhardt (1988). Crown scorch height was calculated using Van Wanger's (1973) equation and fire intensity as determined by the Canadian Forest Fire Behaviour Prediction (FBP) System (Forestry Canada Fire Danger Group 1992) for the vegetation community under specified burning conditions. Crown scorch was estimated from crown scorch height (Peterson 1985). Bark thickness (Kozak and Yang 1981) and fire intensity were used to calculate cambium mortality. For the re-sprouting species (aspen and white birch), season of burn was important to tree mortality and re-sprouting ability as low intensity fires prior to leaf flush have been shown to girdle aspen stems and prevent suckering (Weber 1990). Therefore, the model includes fire intensity and date of leaf-flush in the mortality and recruitment of aspen and white birch. Also, pure deciduous stands in the model do not burn after spring greenup of the understory, but they may burn after autumn leaf-fall when it is possible for the cured herbaceous understory to dry-out quickly with increased solar radiation reaching the forest floor.

Table 2. Summary of main fire ecology traits for major boreal tree species in the BORFIRE model.

	Jack Pine	Black Spruce	White Spruce	Aspen	White Birch	Balsam Fir
Regeneration Method	canopy-stored seed	canopy-stored seed	unstored seed	root suckers	root collar sprouts	unstored seed
Fire Resistance	moderate	low	low	very low	very low	very low
Seasonal Fire Effect	none	none	self-seeds only after autumn fire	does not re-sprout if burned prior to leaf flush	does not re-sprout if burned prior to leaf flush	none
Reproductive Age (yrs)	20-120	15-200	25-250	5-110	15-110	20-140
Shade Tolerance	intolerant	intolerant	tolerant	intolerant	intolerant	tolerant

Biomass impacts

Fuel (or biomass) consumption during a fire event was estimated in the model using fuel load and burning condition parameters. Slash consumption as correlated to BUI by McRae (1980) was used to calculate biomass losses in branchwood and CWD pools. Total depth of burn was calculated using forest floor depth and duff moisture content (as estimated by DMC) using the same procedure as FIRESUM (Keane et al. 1989). All biomass in the litter layer was assumed to be lost during combustion. The amount of duff biomass lost was determined using duff fuel load, duff bulk density (97 kg/m^3), litter depth (based on 36 kg/m^3) and total depth of burn.

A certain amount of the live biomass was also lost during fire to represent fine aerial fuels. In the case of a crown fire, this amount was estimated as the foliage and stem bark components (Ter-Mikaelian and Korzukhin, 1997) for conifers and white birch, and only the foliage for aspen. In non-crowning fires, the model estimates no loss in aspen and a 2% loss in all other tree species. All live trees that were killed by fire were transferred to the dead standing biomass pool, and all trees that were dead and standing at the time of the fire were transferred to the dead aboveground biomass pools of branchwood and CWD.

Physical fire parameters

Fire was simulated as a stochastic event using a Monte Carlo method and the average fire frequency. If a fire event occurred during any annual time-step, the model determined a Julian 'fire date' by randomly selecting a date from a weighted distribution of the historical monthly area burned. The fire date was used to determine the state of hardwood flushing, understory condition, and the burning conditions as measured by the FWI System. The model randomly selected the BUI, ISI and FWI values for the fire from a normal distribution of monthly values as calculated for the fire regime database (Table 1).

The DMC was used to calculate forest floor consumption, and BUI was used to calculate aboveground dead biomass consumption. The average fire rate of spread (weighted by species) was calculated with the ISI and tree species data using procedures of the FBP System. Fire intensity was calculated using the average fire rate of spread and fuel consumption during the fire (Byram 1959). Fuel consumption was calculated in several steps using FBP System procedures. Foliar moisture content at the time of the fire was calculated using the fire date, latitude, longitude and elevation data. This was combined with live crown base height to determine the critical surface fire intensity for crown fire to occur based on the species composition of the stand. The actual surface fire intensity was determined using the fire rate of spread (from stand composition, age and season of burn) and surface fuel consumption (litter, duff, dead and downed branchwood and CWD). If surface fire intensity was greater than the critical surface fire intensity, then a crown fire occurred and total fire intensity was calculated using rate of fire spread and total fuel consumption (surface fuels plus bark and foliage fuels).

MODEL SIMULATIONS

Each simulation was started with initial conditions of 1,000 seedlings per hectare for each species, and a standard duff biomass of 80 t/ha. This was to simulate the first year of a stand regenerating after fire. The model used fire regime data in Table 1 to drive species composition and biomass dynamics in EINP and PANP aspen stands, and in PANP white spruce-aspen stands during 1975-95 and 2080-2100. Each simulation was run for 400 years to allow the stand dynamics to reach equilibrium under the driving variables. Each simulation scenario was repeated 25 times and averaged. Tree densities during the 400-yr simulation period are presented in Figure 3, and the average tree densities and stand carbon (converted from biomass by 0.5) during the final 50-yr period of the simulation are summarized in Table 3.

MODEL RESULTS

The aspen stand density in all simulations showed the same pattern of high initial values as many new root suckers appeared after the first fire in the simulation period (Figure 3a, 3b). The number of aspen stems declined during the rest of the simulation period as

Table 3. Summary of average stem numbers and carbon storage during the final 50 years of the 400-year simulation scenarios.

	Elk Island National Park		Prince Albert National Park			
	Pure Aspen		Pure Aspen		Aspen-White Spruce	
	1975-1995	2080-2100	1975-1995	2080-2100	1975-1995	2080-2100
			Stem Density (trees/ha)			
Aspen	22	244	768	831	694	923
White Spruce					193	118
			Carbon Storage by Stand (t/ha)			
Dead Woody Material	0.4	3.6	8.5	9.2	11.6	9.2
Forest Floor	12.1	39.0	82.3	80.1	84.3	86.4
Trees	11.5	49.9	111.6	88.3	97.2	102.1
Total by Stand Type	24.0	92.5	202.4	177.6	193.1	197.7

natural thinning occurred between fire events until the stem density reached a dynamic equilibrium range with the fire regime. Average aspen stem density for the final 50 years of the EINP 1975-95 scenario showed an exceptionally low value (Table 3). Average aspen stem density was greater under the 2080-2100 fire regimes for both National Parks, but the total aspen stand carbon decreased in PANP and increased in EINP. In the white spruce-aspen stands of PANP (Figure 3c), aspen stem density increased, white spruce density decreased and total stand carbon increased slightly under the 2080-2100 fire regime (Table 3).

DISCUSSION

The different tree community responses to change in fire regime illustrate the importance of stand type differences and regional differences in fire regime. There is a very large difference in the current fire regimes of EINP and PANP. The very long fire cycle in EINP is unlikely to sustain fully-stocked aspen stands because the maximum life-span of aspen is much shorter than the fire cycle. Although aspen is capable of suckering without fire, sprouting vigour, and therefore average stand density, will be much lower if overmature declining trees become the main source of stand regeneration. A fire return interval shorter than the lifespan of aspen forces it to produce suckers at an earlier and more vigorous age. The 1975-95 fire cycle used for EINP in this study was the result of a province-wide policy of wildfire exclusion but this fire regime is not capable of supporting healthy aspen stands in the longterm. Clearly, the current ongoing prescribed burn program in EINP is required to maintain aspen-dominated ecosystems in the Park.

In PANP, the current fire cycle is shorter than the life-span of aspen. This maintains a high aspen stand density and a much higher level of carbon storage than EINP. Under the future estimated fire regime, aspen stands in PANP are expected to increase slightly in stand density due to greater fire occurrence and more frequent postfire re-sprouting. However, there is also a decrease in total stand carbon. Even though there are more aspen trees, average stem diameter is smaller and there is a lower overall carbon storage level. In contrast, aspen stands in EINP are expected to greatly increase in stand carbon storage and density under a future fire regime of shorter fire cycles. But overall, PANP aspen stands will still store twice as much carbon as EINP aspen stands. This is because the EINP fire cycle is three times longer than the PANP fire cycle, so there is greater promotion of suckering (and associated carbon sequestration) in PANP aspen stands. This indicates that there is a fire cycle threshold of maximum carbon storage. In EINP, the current fire cycle is much greater than this threshold value so any future decrease in fire cycle length will increase carbon storage. In PANP, the current fire cycle is less than the threshold and a future decrease in fire cycle length will move it further away from the threshold value and decrease carbon storage.

Future change in PANP white spruce-aspen stands is as much a result of aspen and white spruce fire ecology as the influence of

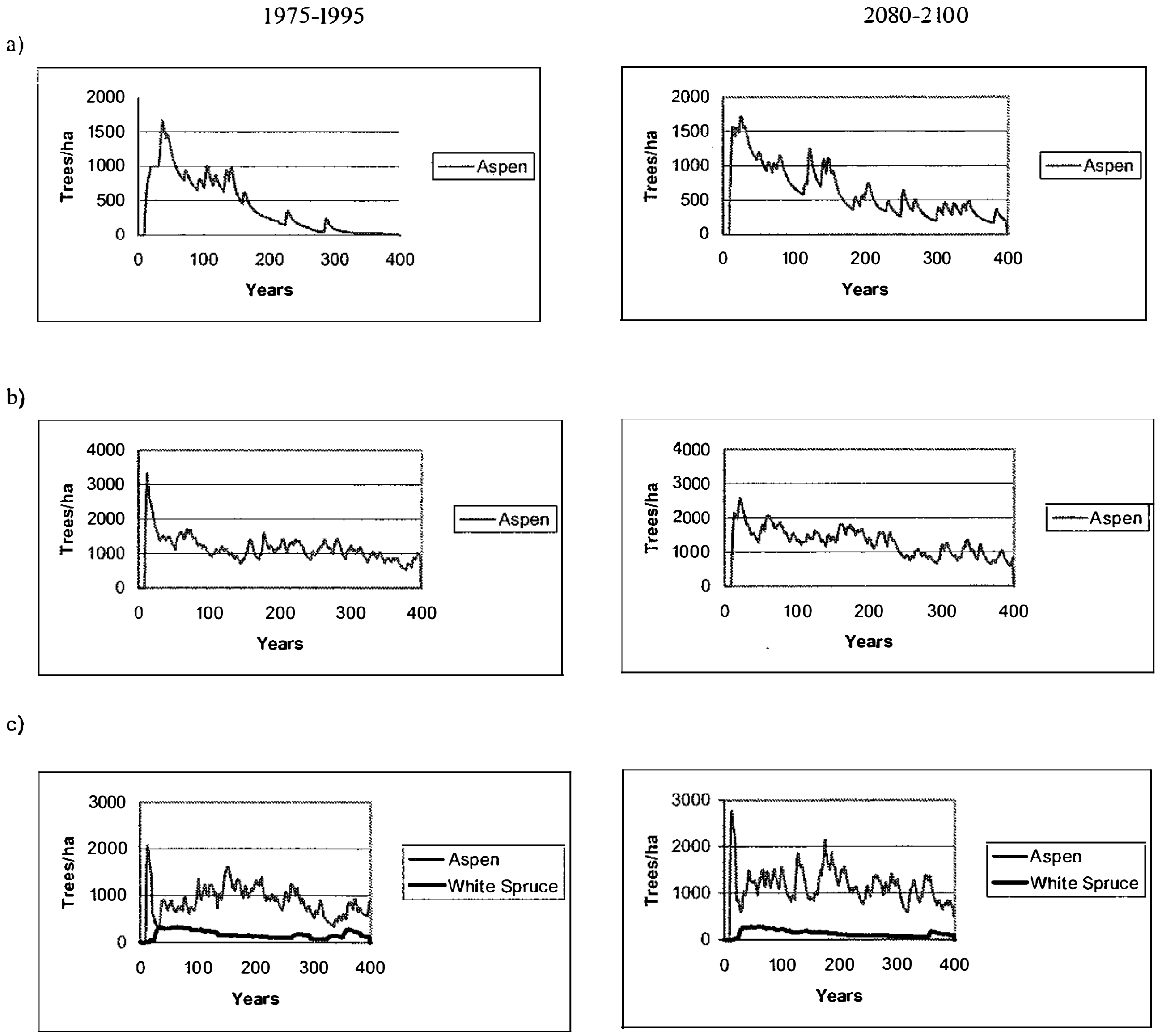

Figure 3. Tree density for the simulation scenarios. Data represents an average of 25 model simulation runs using 1975-95 and 2080-2100 fire regime data for aspen stands in Elk Island National Park (a), for aspen stands in Prince Albert National Park (b), and for white spruce-aspen stands in Prince Albert National Park (c).

altered fire regimes. Aspen is well-adapted to frequent fire by rapid postfire re-sprouting. White spruce is very poorly-adapted to fire because it is easily-killed and does not store propagules on-site. Therefore, any decrease in the fire cycle will benefit aspen at the expense of white spruce. It is also important to note that white spruce is the dominant species in this stand type because it is capable of growing in low light conditions under an aspen canopy, eventually overtopping and dominating the aspen component if the stand remains undisturbed by fire for a long enough time. So, a long fire cycle will benefit white spruce over aspen. As a result of the shorter future fire cycle used in the model, white spruce stem densities decreased by 39% which allowed aspen stems to increase by 34%.

White spruce and aspen also have very contrasting carbon sequestration and storage characteristics. White spruce is long-lived, grows very large and has a high carbon storage level. Aspen is relatively short-lived and quick growing, but it doesn't grow as large as white spruce. Under the current and future fire regimes used in the model simulations, a 34% increase in aspen stems results in slightly higher carbon storage than a 39% decrease in white spruce stems.

The estimated future fire regime data supports previous studies that indicate an increase in fire severity and fire intensity in Canadian boreal forests. Although increased fire severity and intensity will cause an increase in carbon loss from the forest floor and aboveground biomass during fires, those losses can be overshadowed by the longterm growth dynamics of the future stand as evidenced by increases in total carbon storage. In effect, boreal carbon dynamics can be more affected by the ecological influence of forest fires than by their physical effects. From this study, it appears that fire-induced change in vegetation dynamics will have a greater effect on the boreal carbon balance than any change in carbon loss due to fire. Mixedwood stands are expected to have the greatest potential for change in species composition under future fire regimes, and studies are underway to apply the BORFIRE model to other boreal mixedwood stand types to examine the resulting impact on carbon dynamics.

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PRELIMINARY ASSESSMENT OF THE IMPACT OF FIRE ON SURFACE ALBEDO

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ABSTRACT

Wildfire causes extreme changes in surface energy dynamics and has a profound impact on the structure and function of ecosystems. Although it has long been recognized that fire plays an important and immediate role in modifying boreal forest ecosystems, detailed studies on the impact of fire on the radiation balance and energy partitioning have been few. Measurements of the surface energy balance in fire-disturbed boreal sites are underway. Remote sensing is being investigated as a tool for estimating energy budget model parameters and for extrapolation of field-based measurements to broader scales.

Landsat image data as well as tower-based and hand-held instruments were used to measure surface albedo at sites in interior Alaska. These measurements were used to study the impact of fire on surface albedo, a fundamental measurement for estimating net radiation. Results of Landsat data analyses have shown that after a fire, spectral reflectance in the short-wave infrared region (1.5 to 3.0 μm) increase and remain higher than the surrounding unburned forest for many years. In the visible and near IR region of the spectrum (0.4 to 1.5 μm), spectral reflectance typically decreases from burning. Preliminary results of total albedo estimated with both field and image data show that fire typically reduced overall solar albedo directly following the burn. In the case of severe burn conditions, however, albedo could be higher following burning due to a combination of higher albedo surface material and fewer shadows from standing dead trees.

Additional research showed that following the typical initial decrease in albedo due to charring, albedo will increase at a site due to the re-establishment of light-leafed deciduous primary successional plant species. Further analysis is underway to develop methods of modeling the changes in albedo over time (diurnal and multi-year) to understand the impact of fire on net radiation in both space and time.

INTRODUCTION

Wildfire is a prevalent factor in many terrestrial ecosystems and often plays an important role in defining ecosystem structure and function. Changes in surface albedo and energy exchange resulting from fire disturbance can have both short-term and long-term impacts at local, regional, and global scales. This is particularly true in boreal settings where fire scars (areas of the landscape disturbed by fire) are often very large and are very slow in recovering to pre-burn condition.

Fire in the boreal region is now being studied intensively at many scales (Cofer et al., 1996; FIRESCAN Science Team, 1996; Conard and Ivanova, 1997; Chambers and Chapin, 1999; Durant et al., 1999; Kasischke and Stocks, 2000; also see http://www.fsl.orst.edu/fera/

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frostfire.html). The impact of fire on carbon cycling is the focus of many of these studies because of the recent emphasis on understanding the global carbon cycle. Fire's role in carbon cycling is obvious, since it is a major mechanism for the movement of carbon from the terrestrial pool to the atmosphere (Cahoon et al., 1994; Kasischke et al., 1995; French et al., 2000). Fire also impacts carbon dynamics through the alteration of fundamental ecosystem parameters, such as temperature and moisture (Van Cleve et al., 1983; Viereck, 1983), which directly affect soil microbial respiration (O'Neill et al., 1997; Richter et al., 2000) and post-fire regrowth characteristics (Landhaeusser and Wein, 1993; Kasischke, 2000). Because of this, studies of surface energy dynamics, including albedo change, are needed for a more basic understanding of the impact of fire on surface temperature and moisture regimes and, therefore, carbon balance. However, detailed studies on the impact of fire on energy flow in these sites have been few. Recent measurements of energy flux are just now being analyzed and published (Chambers et al., 2000a; Chambers and Chapin, 2000).

Recent results of experiments using global climate models have shown the importance of the boreal forests in the seasonal and annual climate regimes because of their direct influence on albedo (Bonan et al., 1992). Boreal forest canopies mask a snow covered ground surface causing a higher albedo than if they were absent. Removal of the forest canopy can increase albedo substantially and cause much cooler air temperatures. These studies have shown that disturbance related changes in the structure of the boreal forest can be as important as biogeochemical changes (Bonan et al., 1995). Furthermore even small albedo changes (0.10) induced by forest cover change may have important impacts on global air temperatures (Foley et al., 1994).

The purpose of the study presented here was to quantify and map the change in growing season albedo as a result of fire-disturbance in a black spruce dominated ecosystem in interior Alaska. Remote sensing information from Landsat TM imagery was the main source of data for this study. Field measurements were used to complement and confirm the remote sensing results which showed that albedo change can be small and can decrease or increase as a result of fire-disturbance. The results also showed that albedo change is spatially variable based on pre-burn vegetation characteristics and burn severity.

BACKGROUND

The Importance of Albedo

Net radiation (R_n) is the amount of radiative energy (W m^{-2}) available at the earth's surface to participate in energy-exchange processes. Using the convention that radiative fluxes toward the surface are positive, net radiation is calculated by summing the incoming and outgoing components of shortwave (S; 0.3 to 4 μm wavelength) and longwave (L; 4 to 80 μm wavelength) radiation at the surface as follows:

$$R_n = (S_i - S_o) + (L_i - L_o) \qquad [1]$$

where the i and o subscripts refer to incoming and outgoing energies, respectively. Net radiation can be "tracked" through the system, that is, partitioned between a number of energy sinks including energy to warm the air (sensible heat), to evaporate water (latent heat), to grow plants (photosynthesis), or to warm the vegetation canopy (canopy storage) or ground (ground storage). An accurate estimate of net radiation is a valuable independent check of the accuracy of the measurements of each of the individual sink processes when studying the entire energy exchange system of a site. An analysis of the relative partitioning of net radiation between these processes helps in understanding the movement of energy within and between the surface and the atmosphere and to compare the use of energy in different ecosystems or site-types, in our case, energy use in burned versus unburned sites (Monteith and Unsworth, 1990).

Of primary importance in estimating R_n is measuring surface shortwave albedo (α). Albedo is the fraction of incoming shortwave energy (S_i) that is reflected from the surface (S_o). Surfaces with a high albedo value (0.9 or 90%, for example) such as snow reflect most of the incident shortwave solar energy, while surfaces such as forest canopies and soil with a low albedo value (0.1, for example) reflect a small amount of energy; most of the shortwave solar energy is absorbed into these surfaces.

By measuring albedo we know how much shortwave energy is "captured" by the system and how much is reflected away. The net shortwave component of equation 1 can be restated in terms of the albedo in the following manner:

$$(S_i - S_o) = (1 - \alpha)S_i \qquad [2]$$

Determination of albedo can be helpful for understanding processes at many scales. Disturbance-caused changes in albedo due to vegetation change have been estimated in many ecosystems to determine its impact on local, regional and global-scale processes (for example Jackson and Idso, 1975; Couzin, 1999; Chambers et al., 2000a; Michalek et al., 2000a). Fine scale albedo measurements allow a better understanding of fine-scale energy partitioning. For example, albedo determines the amount of net radiation at a site which can influence microclimate. In turn, the microclimate determines fundamental ecosystem characteristics such as vegetation species establishment patterns and succession, as well as disturbance regime. Regional and global-scale albedo mapping allows an understanding of broad-scale energy dynamics, including the impact of terrestrial ecosystems on global temperature. This scale of albedo analysis allows us to determine the total amount of energy available in the Earth System for a better understanding of land/water/atmosphere interaction and global climate patterns.

The determination and interpretation of local ecosystem albedo is the focus of this study, but it is important to note that the techniques presented here for albedo estimation can be applied equally well to investigations at other scales, such as regional impacts of fire on the energy balance. Furthermore, the large areas affected by fire in the boreal region may have a significant impact on the energy balance of the Earth, so broad-scale studies of albedo in fire-disturbed regions are warranted.

It is important to note that the work presented here represents albedo estimates for the growing season months and does not consider winter albedo. Winter albedo is strongly affected by snow cover, since snow has a very high albedo (~ 0.9). Fire-disturbed sites are particularly susceptible to such affects since the snow pack is not masked as much by a perennial spruce canopy as before the disturbance. Consequently, the winter albedo of a fire-disturbed site would be much higher than the forested pre-burn site and the summertime site as long as there is snow cover. The impact of fire-disturbance on winter albedo, therefore, may be of great importance for the wintertime energy flux. At high latitudes, however, low sun angle and short days strongly influence the amount of incoming solar radiation in the winter months. The time of year most sensitive to snow-induced albedo changes is late winter to early spring, when the ground is usually still snow covered and the days are rapidly becoming longer after the vernal equinox (March 21). Although year-round albedo mapping would be useful for studies of the impact of fire-disturbance on energy exchange at regional and global-scales, ecological factors are less affected by changes in wintertime albedo because biological systems are generally dormant during the cold months. For this reason winter albedo is not considered in this study.

Remote Sensing of Fire-disturbed Sites

The research presented here follows a recent set of studies conducted in an area near Delta Junction, Alaska which burned in 1994. Data on pre-burn forest type, burn severity, soil water, soil CO_2 efflux, and a variety of remote sensing data sets have been collected in order to look at the impact of fire on the landscape and carbon flux in particular (French et al., 1997a; French et al., 1997b; O'Neill et al., 1997; French et al., 1999; Kasischke, 2000; Michalek et al., 2000b). In one of these studies, carbon release from burning was determined from information on pre-burn carbon density and burn severity derived from Landsat TM images from before and after the burn (Michalek et al., 2000b). The study used air photos and field measurements to classify the Landsat imagery and to quantify pre-burn carbon and burn severity. Researchers found that an average of 40 t/ha of carbon were released during the burn. Most importantly, the study results showed that a spatially explicit estimation of pre-burn carbon and burn severity is important for a proper estimation of carbon release. As would be expected, carbon release was highest in areas of high pre-burn carbon and burn severity (108 t/ha) and lowest where carbon and burn severity were low (16 t/ha).

The intermediate remote sensing products created for the carbon study were used in the current study to assess the influence of pre-burn and post-burn conditions on surface albedo.

METHODS

The study site is located 190 km southeast of Fairbanks, Alaska along the Alaska highway near the Gerstle River (N 63° 48', W 145° 06'). Several access roads into the burn area have made it possible to sample a variety of locations within the burn. The site is situated on relatively level terrain and is dominated by black spruce forest types with varying canopy densities. The fire occurred in 1994 and burned from mid-June to early October. Various levels of burn severity, as defined by biomass consumption, can be found within the burn scar. Photographs of the severe and moderate burn sites one year after the burn and five years post-burn are shown in Fig. 1. A more detailed description of the study site and burn conditions can be found in Michalek et al. (2000).

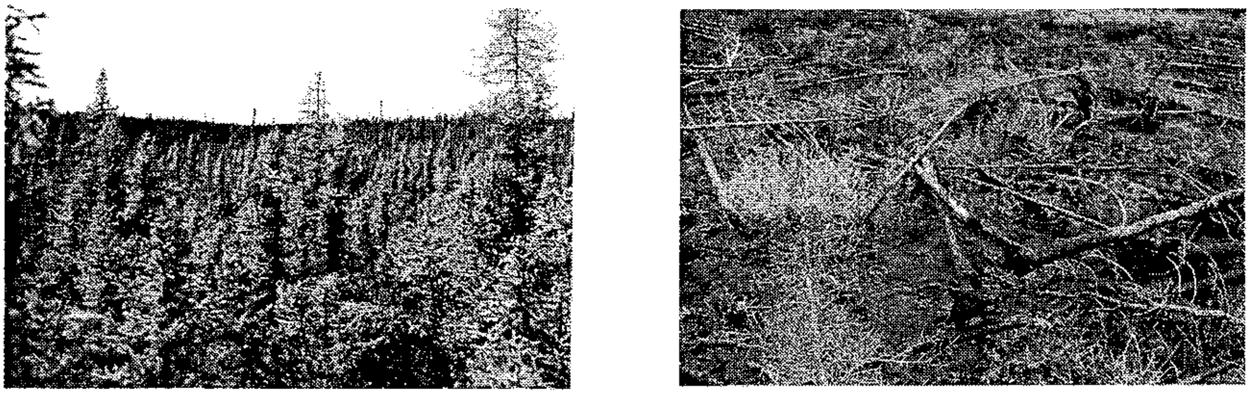
Landsat TM images from 30 August 1992 and 16 September 1995 were the primary data used in this study. Total albedo and albedo change from 1992 to 1995 was mapped using the six reflective bands of Landsat (bands 1-5 and 7). First, reflectance values for each band of the 1992 image were estimated using within-scene targets with low, medium, and high digital values (DN) and whose spectral reflectance was known. A linear regression relationship between TM spectral band reflectances and DN's was determined using these target values, and these relationships were applied to the Landsat data of the study site. Reflectances were initially determined for the 1992 scene; the 1995 scene was then normalized to the reflectance of the 1992 data using unchanged landscape features. This technique allows for reasonable change analysis from pre-fire to post-fire. Albedo was then estimated from the spectral reflectance values by weighting the average Landsat reflectance values in each of the regions of the shortwave solar spectrum by their percent of the solar irradiance as follows:

$$\text{Albedo }(\%) = \left\{\frac{(\text{band1}+\text{band2}+\text{band3})}{3}\times 0.526\right\} + \{\text{band4}\times 0.362\} + \left\{\frac{(\text{band5}+\text{band7})}{2}\times 0.112\right\} \qquad [3]$$

This procedure is comparable to a procedure used by Toll et al. (1997) for estimating albedo from AVHRR imagery. Finally, maps of estimated surface albedo across the burn scar for before and after the 1994 burn were made as well as a difference between the two dates to show the change in total albedo. The patterns of albedo change were then compared with pre-burn vegetation and burn severity maps created for a previous study (Michalek et al. 2000b).

Tower-based field measurements of albedo were collected in 1998 and 1999 at the 1994 burn in two severely burned sites, one moderately burned site and in an unburned "control" site in conjunction with a full array of tower-based meteorological and eddy correlation flux measurements. Upward and downward facing pyranometers were deployed 14m above the surface for 10 days in 1998 and 4m above the ground for 4 days in 1999 during the months of July and August. Data were sampled continuously during the deployment period and logged at 20-second intervals. The data were then averaged to 30-minute intervals for interpretation. A detailed description of the instrumentation and data collection can be found in Chambers and Chapin (2000).

Field albedo measurements of component cover materials in the 1994 burn and a nearby 1999 burn were collected using a modified Kip and Zonen hand-held pyranometer system. The instrument consists of an upward-looking hemispheric pyranometer which measures total incoming solar irradiance and a downward-looking pyranometer with a field of view of approximately 60° which measures directional radiance from the surface material of interest. The pyranometer system was calibrated to albedo using 2% and 50% spectrally "flat" (constant) reflectance panels. Albedo was calculated as π times the ratio of outgoing radiance to incoming irradiance. We measured albedo for common surface cover materials several times at four sites within the 1994 burn and a nearby 1999 burn to look at materials in five-year-old and recently burned sites. Extensive data collection of surface material albedo was not conducted, so these data were used only for general comparison of component surface materials, rather than for direct comparison to the tower-based measurements or Landsat-derived albedo



a) Unburned black spruce forest

b) Severe burn - exposed mineral soil and fallen tree boles

c) Moderate burn site - standing dead trees and charred moss

d) 5-year-old burn scar - vegetation regrowth

Figure 1: Ground photos showing (a) a typical unburned black spruce stand in interior Alaska; (b) a severe burn site with exposed mineral soil and fallen tree boles; (c) a moderate burn site with standing dead tree boles and a charred moss ground surface; and (d) a 5-year-old burn site showing herbaceous and shrubby vegetation regrowth.

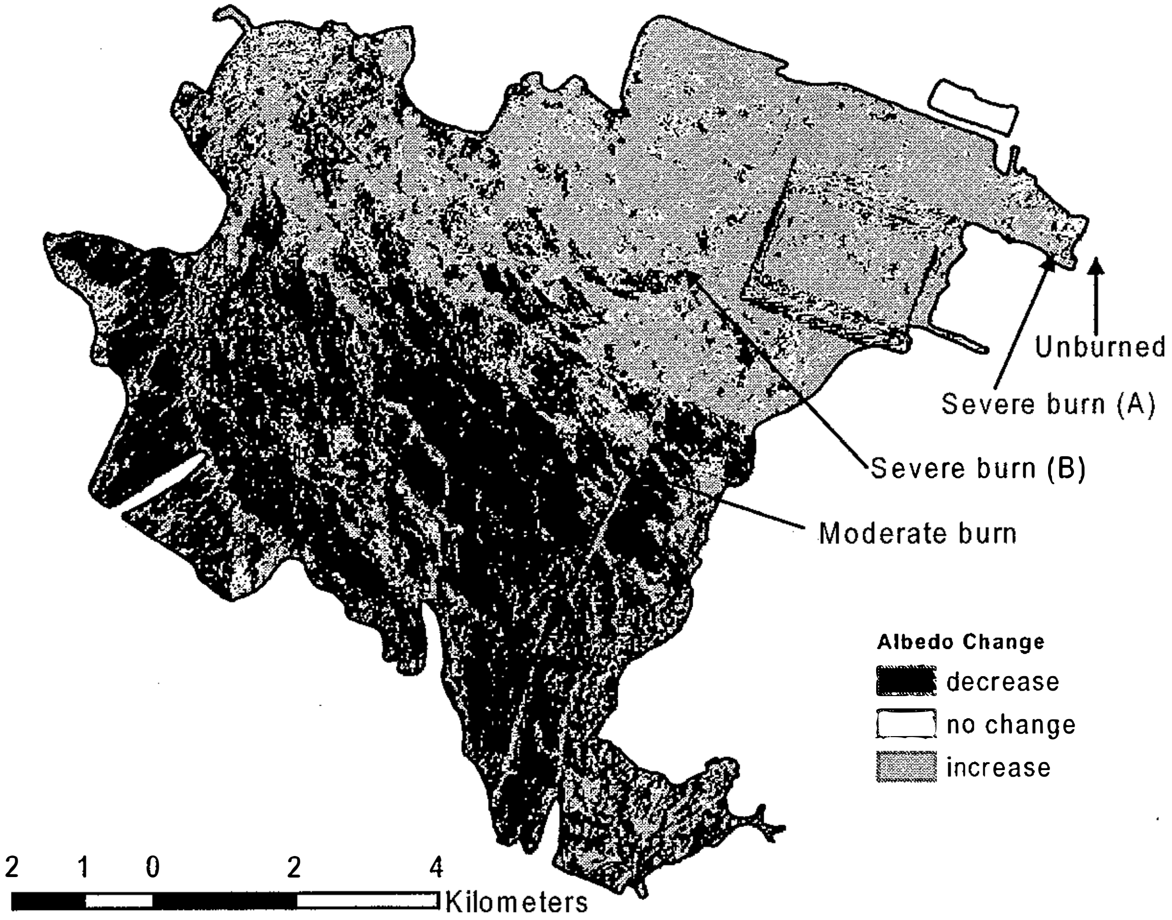


Figure 2: Albedo difference from 1992 to 1995 of the 1994 burn site derived from analysis of 1992 and 1995 Landsat imagery. Grey areas indicate an albedo increase while black areas indicate albedo decrease. Sites of tower-based field measurements are indicated.

estimates. Average albedo for 14 materials were obtained in sunlit conditions and nine in shaded conditions. These included eight types of vegetation, charred wood, charred moss, bare soil and others.

RESULTS

Our Landsat analysis of albedo from before the 1994 burn to approximately one year after the burn shows that the change was small in most areas, with an increase in some areas of the burn and decrease in others. The image-derived map (Fig 2) shows the spatial distribution of these results. The results show that 14% of the burn area showed no change in albedo, while 40% of the area showed an increase and 46% a decrease (Table 1). A comparison of the spatial maps of pre-burn spruce density, burn severity and albedo change shows that areas of high pre-burn spruce density (the category with the lowest pre-burn aledo) most often increased in albedo after the burn, while areas of medium and low density black spruce decreased. Burn severity was also an important factor, with areas of low to moderate burn severity showing an albedo decrease and severe burn conditions (which tend to have the highest post-burn albedo) showing an albedo increase. No significant difference is seen in measurements collected at the unburned site sampled by the field-based sensors indicating hat the Landsat-based measurements and field measurements are consistent (Table 2). The tower data collected the burn areas four to five years following the burnshow that average albedo increased after a few years of vegetation recovery relative to the Landsat measurements made one-year after the fire. All sites measured by the tower-based sensors, whether moderately or severely burned and with various pre-burn conditions, show higher albedo than the Landsat estimatefor the site before the burn, although the change is more dramatic in the severely burned sites which had a high density black spruce canopy before the burn.

Field measurements of ground components using the hand-held pyranometer system show that early successional vegetation has a high albedo (ranging from 0.10 to 0.28) relative to other materials present following a burn, such as charred moss, burned tree boles, bare mineral soil, and bare unburned tree boles (0.05 to 0.18). The exceptions to this are new moss, which has a lower albedo than other vegetation colonizing after a burn (0.12), and dead spruce needles that have a relatively high albedo (0.18) but are a small component of the ground surface. Shadow also has a low albedo (0.02 to 0.08);

Table 1. Albedo change as compared to burn severity and pre-burn spruce density.

Albedo Change	Burn Severity	Pre-burn Black Spruce Density	Number of Pixels	Proportion of total area	Proportion of increase or decrease
Increase	low/mod	high	8,857	11%	28%
		medium	2,838	4%	9%
		low	12	0%	0%
	severe	high	11,721	15%	38%
		medium	7,312	9%	23%
		low	463	1%	1%
		sub-total	31,203	40%	100%
Decrease	low/mod	high	4,820	6%	13%
		medium	17,786	23%	50%
		low	9,233	12%	26%
	severe	high	43	0%	0%
		medium	1,037	1%	3%
		low	2,972	4%	8%
		sub-total	35,891	46%	100%
None			10,576	14%	
Total			77,770	100%	

Table 2. Albedo measurements at 4 sites within the 1994 burn area (see Fig. 2 for site locations).

	Total Albedo (%)		
		1-year	4/5-year
	Pre-burn	Post-burn	Post-burn
Burn Condition	1992 Landsat	1995 Landsat	1998/99 Field(tower)
Unburned	9	9	9
Severe (A)	8	10	12.5
Severe (B)	8	9	13
Low/Moderate	10	8	11

it is a large component of the ground surface in areas with standing dead trees but is virtually absent in the severely burned areas due to fallen tree boles (Fig. 1).

DISCUSSION, CONCLUSIONS, AND FUTURE WORK

Despite the general notion that fire immediately lowers surface albedo, this analysis shows that the result is not always so straightforward. Our results show that fire-disturbance can increase or decrease growing season albedo depending on the pre-burn vegetation condition (canopy density) and burn severity. Results of a study comparing fire-disturbed sites in arctic and boreal sites showed that pre-burn cover type is also critical in determining the impact of fire-disturbance on albedo and therefore radiation flux (Chambers et al., 2000a). And, as mentioned before, wintertime albedo can be impacted by fire-disturbance due to a removal of the forest canopy. Our analysis shows that growing season albedo change from fire-disturbance in black spruce dominated ecosystems is often small compared with fire-induced changes seen in arctic or temperate ecosystems (Chambers et al., 2000a). This minimal impact of fire-disturbance on albedo can lead to either an increase or decrease in net radiation at burned black spruce sites, and it means that the change in net radiation is strongly dependant on the impact of the disturbance on the longwave component of the net radiation equation (Eq. 1).

Although it may be obvious, it is important to acknowledge that change is defined by conditions both before and after the change event. Pre-burn black spruce canopy density defines the initial surface albedo, with dense spruce having a very low albedo (0.08) and sparse spruce a slightly higher albedo (0.10). Post-burn condition, a function of burn severity, influences albedo due to variations in the surface material and the presence or absence of standing dead trees at a site giving rise to shadow. Areas of severe burn at this site had a high amount of exposed mineral soil following the burn and very few standing dead trees leading to a higher albedo following the burn (0.10 versus 0.08 pre-burn). Moderate/low burn conditions are characterized by many standing dead stems and very little exposed mineral soil leading to a lower albedo following the burn (0.08 versus 0.10 pre-burn; Fig. 1). At this study site, albedo increased one year following the burn in sites with moderate to high pre-burn spruce density and severe burn conditions, while albedo decreased in sites with lower pre-burn spruce density and moderate or low burn severity (see Table 1).

Results from the tower-based measurements of albedo at this site several years after the burn, and results from other fire-disturbed black spruce sites in the area have indicated that albedo change over several years of forest stand recovery follows a non-linear pattern (Chambers and Chapin, 2000; Chambers et al., 2000b). After the initial change in albedo from the disturbance, vegetation regrowth can cause a substantial increase in albedo over the pre-burn due to the establishment of light-leafed vegetation. At the 1994 burn site albedo was found to be higher in all of the tower measurement locations after five years of vegetation regrowth regardless of pre-burn condition or burn severity, although burn severity did have some impact on albedo due to differences in vegetation establishment patterns (see Table 2). Using more recent Landsat data of the 1994 burn scar we plan to look at changes due to regrowth and age since burn in order to verify the field-based results and map the impact across the entire burn scar.

The purpose of this study was not to develop or refine remote-sensing albedo estimation techniques, but rather to investigate the impact of fire disturbance on surface albedo. We have used a relatively simple estimation method using the six Landsat reflective bands. Most importantly, our estimates are not meant to be absolutely accurate as much as relatively accurate between the two images in order to look at albedo change. We have compared some of the absolute albedo estimates (in Table 2), but also show that the "control" site is unchanged, indicating some reliability of the Landsat estimates and consistency with the tower-based measurements. Albedo change, however, is the main result of this study.

Although our remote sensing-based analysis shows similar results as the ground-based analysis, the remote sensing study has some advantages. First, with the Landsat images we have both pre-burn information and post-burn information at the same site. The tower-based study used adjacent unburned sites to represent pre-burn conditions and assumed the sites to be similar. Also, remote measurements provide a broad-scale view of a site allowing us to see spatial variations in surface conditions, where ground-based measurements generally look at only a few representative sites.

Remote measurements also present some disadvantages. Remote sensing can only measure radiometric information at one point in time. Field based studies can directly measure a parameter and they can be made over shorter time intervals, thereby capturing hourly, daily and weekly variations. Variations in factors such as surface wetness are not captured in remote sensing data such as Landsat imagery. Surface wetness variation is a potential problem in our albedo change study, since wet surfaces are generally darker (lower albedo) than the same material dry. By using two images normalized to each other we are aiming to minimizing this type of variability.

An important next step for this research study involves analysis of 1999 Landsat ETM+ imagery of the 1994 burn scar and a new 1999 burn scar in the same area. We have identified a mid-summer 1999 image of the region which was collected coincident with the tower measurements and a September 1999 cloud-free image which will compare well with the 1992 and 1995 September images used in this initial study. Using this set of images we hope to show that satellite image data can be used to effectively map changes in albedo as a site ages.

Further work related to surface temperature mapping is also of interest for these fire-disturbed sites. Fire disturbance can dramatically influence the thermal properties of the site thereby changing the longwave radiation flux ($L_i - L_o$) and net radiation (R_n; see Eq. 1). The potential exists to use thermal IR remote sensing data to further explore the impact of fire on net radiation and learn what site factors are of primary importance in driving the direction of net radiation change from fire disturbance.

Additional remote sensing work is possible using ASTER-derived albedo and temperature/emissivity for finer temporal coverage and MODIS albedo and land surface temperature products for regional scale work. These data could be used to look at within-season variability in surface conditions which relate to derivation of net radiation across these fire-disturbed landscapes. This type of broad-scale analysis will be helpful for understanding the impact of fire on regional energy balance. Because fire is so prevalent and affects large areas this impact may be substantial. By looking at the finer-resolution impacts with Landsat-like data, as we have in this study, we will be able to interpret coarse-scale MODIS data products more effectively.

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LIGHTNING FIRE IN FORESTS OF CENTRAL SIBERIA

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ABSTRACT

Lightning fires make up 30% of all fires in the forests of Siberia and contribute considerably to forest carbon budgets. As populations grow and economic and recreational use of forest lands increases, concern about lightning forest fires in Siberia increases. Lightning is a major ignition source in forests of many Siberian regions. Storms are quite often active over big areas and result in sudden simultaneous occurrence of mass fires that cover vast territories. This makes investigation of lightning fire occurrence and spread, as well as storm prediction and early detection, very important issues. Our case study conducted in the Yenisey plain in central Siberia described lightning fire occurrence related to weather patterns and the Earth's magnetic field anomalies.

Keywords: lightning forest fire, dynamics of storms, geomagnetic anomalies, frequency of lightning fires, boreal forests, Central Siberia, Russia

INTRODUCTION

Predicted global warming is expected to increase the storm activity and the frequency of lightning fires, which will influence carbon storage in boreal forests. Lightning is the major natural cause of forest fires. The number of lightning forest fires varies with region and, according to different literature sources, they can account for 1 to 70% of all fires (Gribanov 1954; Uspensky 1959; Kurbatsky 1964; Zakharov and Solarchuk 1977). Data available at the Central Forest Protection Air Base show that lightning fires make up 30% of all forest fires in western Siberia and 7-33% in eastern Siberia.

Plummer (1912) was the first to address lightning impacts on forests in the forestry literature. He studied the influence of lightning on trees and soil and found that most lighting fires start due to ignition of the forest floor immediately around a tree stem. This conclusion has been confirmed by other studies (Gribanov 1955; Uspensky 1959; Listov 1967; Kurbatsky 1976). Storms occurring at the end of droughts were considered the most dangerous, and even precipitation cannot prevent ignitions under these conditions (Nesterov 1949).

The energy flux going along a tree stem down to the ground and duration of exposure to heat are the two factors important for lighting forest fire occurrence. Lightning fires are most frequent in dry sites occupied by *Pinus sylvestris*-lichen and *P.sylvestris*-feather moss-*Vaccinium vitis-idaea* forest types (Zakharov and Stolarchuk 1977). Even heavy rain can fail to extinguish lightning-caused ignitions since lightning, as a rule, ignites a thick duff layer surrounding a tree base, which is protected from rain by the crown of the tree (Listov 1967). The time lag that occurs between the moment lightning strikes a tree and its visible flaming is attributed to the period of time that wood is capable of smoldering (Tchirvinsky 1950). Fire remains active inside downed snags and under the spruce canopy. These hidden flames can start a fire when dry weather comes. Mass lightning forest fire occurrence is dependent on the antecedant weather pattern (Kamyshova and Stolarchuk 1984). Storms activity over large territories can frequently lead to unimaginable mass fire coverage (Stolarchuk, Rakov and Belaya 1989).

STUDY AREA AND METHODS

Instrumental monitoring of seasonal and daily storm dynamics was conducted during 1986-1992 in the Yenisei plain, located in the western part of central Siberia. The study area is located along the Yenisey meridian from the Yeloguy river in the north to the Ket river in the south. It stretches 600 km south-north and 200 km west-east (Figure 1). The western part of the area is covered by bogs and is elevated 100-150 m a.s.l. A watershed exists (150-200 m a.s.l.) in the eastern part that separates the Ob and Yenisey river basins (Central Siberia 1964).A cool and humid climate prevails here. Average annual air temperature ranges from –3.2 to –5.7 °C, total annual precipitation is 450-500 mm, the frost-free period lasts for 86-107 days, and snow cover remains stable for as long as 195-210 days. In summer, the climate is determined by cyclones coming from the Atlantic Ocean and the European part of Russia. Frequent droughts are induced by

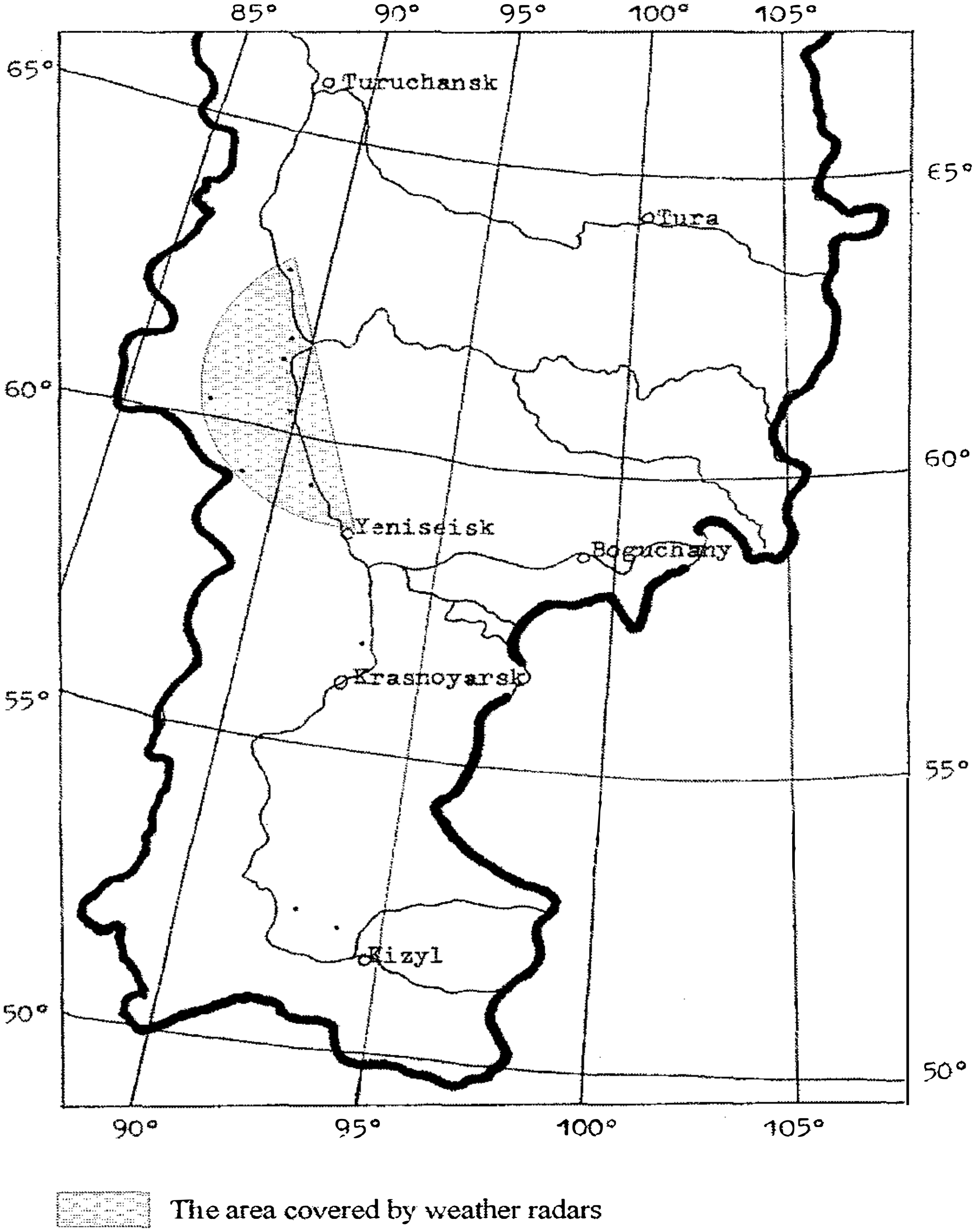

Figure 1. The study area.

dry and warm air masses coming from Central Asia, Mongolia, and the central part of eastern Siberia (Alisov 1956).

Forested land and bogs account for 73% and 27% of the area, respectively. Swamp forest makes up the largest proportion of the forested land. The percentages of the total forest land occupied by different tree species are as follows; 42.5% pine (*P.sylvestris*), 34% spruce, 5.5% siberian pine (*P.sibirica*), 7% larch, 7% birch, and 4% aspen, stands. The *Pinus sylvestris*-lichen forest type is limited mainly to the highest narrow ridges. *Pinus sylvestris*-green moss forest types (with *V.vitis-idaea*, *V.myrtillus*, and *Ledum sp.* in the ground cover) are found on low, narrow, long sandy ridges with podzolic soils. *P.sylvestris*-long moss and *P.sylvestris*-sphagnum forest types are also common. Ground cover in spruce and Siberian pine stands is dominated by long, green mosses and bog grass species. Larch stands, where ground cover is dominated by green mosses, are rare. Secondary birch stands replace dark coniferous forest with grass-bog ground cover after they have burned (Zhukov et al. 1969).

The fire season lasts from May through September, with the peak of fire occurrence in June and July. Krasnoyarsk Forest Protection Air Base estimates forest fires in the western part of central Siberia occur at a rate of about 1000 every year (Table 1). 40% of these are lightning fires.

Fire coverage is generally high in this area. Fire potential is high throughout the entire fire season in the *P.sylvestris*-green moss forest, but it is the *P.sylvestris*-lichen stands that burn most frequently or extensively.

We analyzed data on storms for a 20-year period, provided by six weather stations located in the study area, to identify storm activity characteristics. Seasonal and daily dynamics of cloud-to-ground strikes were investigated using SM-6 lightning strike recorders, with an operation radius of about 100 km. We also used recorders with an operational radius of about 5 km. The time needed to record a strike is one second and the recording frequency is 10 kHz. Data from an MRL-2 stationary weather radar with a radius of 150-200 km were used to determine sources of storm clouds and track their movement. This radar data allowed us to determine the coordinates of stormy clouds and their speed and direction, as well as occurrence and distribution of precipitation during storms. Also, an RPSN-2 weather radar was used, with an operational radius of 100 km. The radars' detection error is ±(0.6 –1.8) km depending on the distance to the object.

In order to relate forest fire frequency with weather patterns, we analyzed data on fire time of occurrence and synoptic maps.

Forest fire extent analysis was based on the past 10-year fire records available at Krasnoyarsk Air Base and the results of field surveys. Forest types and landscape elements most often subject to lightning strikes were identified using data on lightning fire coordinates and simply walking the area.

STORMS AND WEATHER PATTERNS IN CENTRAL SIBERIA

The distribution of storm activity in Siberia is remarkably non-uniform. The greatest number of days with storms occurred in the central and southern parts of Krasoyarsk and Irkutsk regions, as well as in the Tchita region and Buryatia.

In central Siberia, storms occur mostly during the warm season. Storm activity is controlled to a large extent by physical and geographic characteristics of a given area (Filippov 1974). Storm activity increases from north to south in the study area due to increasing air temperature and humidity. The number of days with storms decreases from

Table 1. The number of fires and fire coverage in the western part of central Siberia, Krasnoyarsk region.

	All Forest Fires		Lightning Forest Fires	
Year	Number	Coverage, thousand ha	Number	Coverage, thousand ha
1994	1504	143.0	420	60.9
1995	970	27.8	418	9.6
1996	1412	316.7	299	14.2
1997	1337	126.4	292	12.0
1998	1099	32.1	231	7.7
1999	1772	132.0	533	19.0
2000	850	27.7	342	14.5

west to east because the air is relatively drier over the eastern part of the area. A Storm is a rare event in the northern part of central Siberia; one or two storms per year on average. Some 15-20 storms occur during a year in the Central Siberian Tableland. About 20-23 days per year have storms in the central part of the study area and 30-35 in the eastern and western parts of the Sayan mountains.

Table 2 presents annual duration (hours) of storm activity for 12 weather stations. The total hours of activity are the longest at Sym and Usinsk weather stations. Storms exhibited the highest frequency in June and July – the two months of extreme fire danger. Average continuous duration of a storm is 0.7-1.5 h. The longest total duration of storms occurred between noon and 6.00 p.m.

SEASONAL AND DAILY DYNAMICS OF STORMS AND LIGHTNING FIRES IN THE YENISEY PLAIN, CENTRAL SIBERIA

Front-caused and local storms are characteristic of the Yenisey Plain (Table 3). They start to occur in mid-May and cease in September, with maximum activity in July. Local storms prevail and account for, on average, up to 85% of the total number of recorded storms. Rough surfaces, such as water bodies and forest/field boundaries, are of key importance in developing a horizontal temperature difference that leads to steady convection which, in turn, is a major condition prerequisite for cumulus cloud development. Moreover, distribution of storm activity is related to landscape diversity. Storm occurrence is greater in boggy forest areas and coincides well with boundaries with big forest tracts (Khromov 1940; Yagudin 1967, 1969; Ivanidze 1967; Zhakov 1982). Even forest outskirts contribute to an increase in storm activity by increasing dynamic turbulence (Lobodin 1986). A storm diameter averages about 1 km (Weisberg 1976). Storm clouds of local character can produce several storm cells of a relatively small size (several hundred meters in diameter) (Workman 1965).

Some 50 local storms can occur at any time during a 24-hour period. Up to about 15 storms can occur in just one hour. Front-

Table 2. Annual duration of storm activity (hours) in central Siberia.

Weather Station	Month				Year total
	May	June	July	August	
Turukhansk	0.5	4.1	9.4	6.5	20.5
Upper Imbat	1.2	7.1	13.0	6.7	28.0
Under-stone Tunguska	0.7	10.6	16.2	7.8	35.3
Vorogovo	0.2	8.6	12.5	4.7	26.0
Yartsevo	0.5	11.6	12.3	7.1	31.5
Sym	1.8	17.2	25.6	13.2	58.7
Nazimovo	0.4	10.1	12.5	6.4	29.4
Alexander Lock	2.2	10.8	20.2	12.4	45.6
Yeniseisk	1.8	8.5	16.5	7.2	34.0
Big Murta	4.1	6.4	13.4	5.7	29.6
Turan	1.0	4.7	7.0	6.4	19.1
Usinsk	2.3	17.1	20.4	12.6	52.4

Table 3. Stormy season duration and storm types over the Yenisey Plain, central Siberia.

Years	Number of stormy days	Stormy season duration (days)	Storm types (%)	
			Front-caused	*Local*
1991	35	83	10	90
1992	34	104	19	81
1993	26	71	5	95
1994	29	102	27	73
1995	43	95	9	91

caused storms are most active in the daytime whereas local storm maxima are observed in the evening and at night. This can be due to two factors: 1) underlying surfaces with a large proportion of bogs heat slowly and 2) clouds developing under the influence of solar energy in the daytime appear to be too small to develop further into stormy clouds (Neiburger et al.1985). At night, additional cooling of the upper part of a cloud is sufficient to initiate updrafts in the cloud; the cloud grows and, eventually, a storm occurs. Two maximum storm activity periods can be identified during a 24-hour period; 2.00 p.m. to 8.00 p.m. and midnight to 4.00 a.m.

Storms can occur under any weather pattern. Local storms are characteristic of synoptic patterns such as pressure ridges, troughs, cyclones, low-gradient fields, and warm sectors. Maximum of local storms is observed under highs and lows (59.1% of the total recorded number of storm events) (Table 4).

The average specific frequency of lightning strikes (number of strikes per hour) characteristic of different synoptic patterns is given in Table 4. The highest specific frequency of strikes occurs during storms caused by cold fronts, while the smallest is associated with warm front-caused storms and those occurring in warm cyclone sectors. The increased frequency of cloud-to-ground strikes related to low-gradient pressure fields, highs, troughs, and cyclones is indicative of the considerable intensity (frequency) of local storms.

The number of cloud-to-ground lightning strikes during local storms varies from 2768 to 22147 per year, with 106 to 763 strikes per day per storm. On average, 3 cloud-to-ground strikes per square km are observed for a storm season. 50 percent of these strikes occur between 11.00 p.m and 4.00 a.m. The bulk (81%) of lightning forest fires are associated with local storms. The greatest number of ignitions per synoptic pattern is observed for the low-gradient pressure field (Table 4).

Stormy clouds move mostly in the southwestern and southern direction along with the prevailing air masses in the study area. Average stormy cloud speed is 10 km/hr.

Warm front-induced storms have a remarkably low specific frequency of lightning strikes but they enhance fire starts. Multiple fire occurrence can result if a warm front is preceded by extensive storms and strikes can occur over forest sites with fuels of low moisture content. A warm front is usually accompanied by a small amount of precipitation that fails to extinguish lightning fires when they are still small. While specific frequency of lightning strikes is the highest for storms caused by cold fronts, few ignitions occur because of heavy precipitation and low air temperature characteristic of these storms.

Lightning fires can occur in any forest type but stands dominated by light-needled woody species experience lightning fire most frequently. Pine stands that cover 42% of the forest land in the study area account for 55% of lightning fires, while only 30% occur in spruce and Siberian pine stands that represent 39% of the total forest land. More frequent early summer fires in the *P. sylvestris* stands, as compared to spruce and Siberian pine stands, since snowmelt and fuel drying takes longer in dark-needled forests. During dry summer periods, the number of ignitions in dark- and light-needled stands are about the same.

Table 4. Storm activity and number of lightning forest fires under different synoptic patterns.

Synoptic Pattern	Percentage of pattern occurrence	Specific frequency of cloud-to-ground strikes (# strikes/hr.)	% of the total number of fires	Number of fires per day with storm
Warm front	5.2	96	9.6	1.7
Cold front	4.8	275	0.9	0.2
Occlusion	0.8	-	-	-
Trough	36.6	184	41.1	1.3
High	22.5	205	17.9	1.3
Low-gradient field	15.2	180	19.2	2.3
Warm cyclone	9.3	107	10.9	2.2
Cyclone	4.7	130	0.4	0.1
Anticyclone	0.9	-	-	-

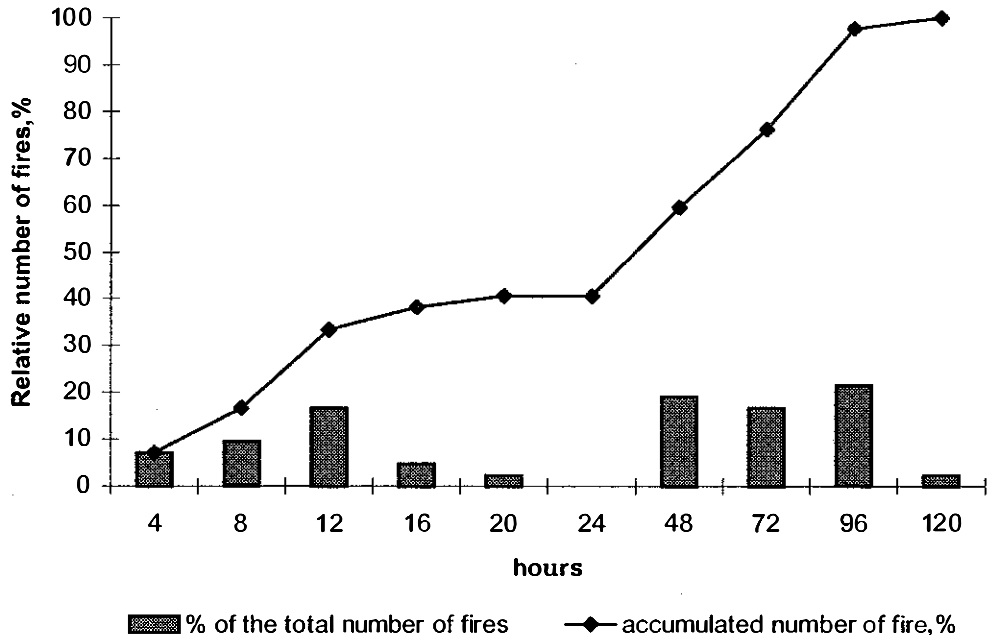

Figure 2. Lag time between a thunder storm and detection of a lightning fire.

Time lags between ignition occurrence and forest fire detection are shown in Figure 2. 40.5% of lightning fires are detected during the first 24 hours after a storm. Initially this time lag is simply due to a lack of detection in forests that are not patrolled after dark. Later on, the time lag is determined by factors such as fuel type and moisture content and weather conditions. Fires ignited by lightning can smolder as long as 5 days after a storm has passed. The majority of them, however, are detected during the first 24-hour period after the storm.

INFLUENCE OF GEOMAGNETIC ANOMALIES ON STORM ACTIVITY AND LIGHTNING FOREST FIRE OCCURRENCE

Some researchers consider that storm activity is related to the Earth's anomalous magnetic fields and geoelectrical heterogeneity of the underlying surface. The highest storm frequencies occur where Quarternary deposits are underlain by rocks of high conductance and low resistance, such as coal layers (Novgorodov et al 1985; Sapozkov and Krechetov 1984; Tchiginsky 1984). Magnetic anomalies are defined as areas where the geomagnetic field is considerably different from that of adjacent regions (Yanovsky 1964). However, in his study of the relationship between storm parameters and soil conductance, Lobodin (1987) found the latter was not a key factor determining variability of storm activity.

We analyzed data on the number of cloud-to-ground lightning strikes recorded with a lightning detector with an operation radius of 5 km diameter. Detectors were established in sites characterized by magnetic anomalies of different intensities. The analysis revealed that a strong anomalous magnetic field contributed to an increase in storm occurrence probability. The higher the magnetic anomaly intensity, the greater the number of storms recorded in the area of that anomaly. Cloud-to-ground lightning strike frequency tended to increase with increasing intensity of a geomagnetic anomaly (Figure 3). The coefficient of correlation between geomagnetic anomaly intensity (x) and the number of cloud-to-ground lightning strikes (y) during a storm season equals 0.88. This close, exponential relationship is described as:

$$y = e^{(2.602+0.410X)}$$

Electromagnetic field turbulence is observed to be the highest, most often, between 5.00 p.m. and 10.00 p.m. (Lipskaya, Babushnikov and Vladimirov 1972). Our data show that daily maxima of both storm activity and cloud-to-ground lightning strike

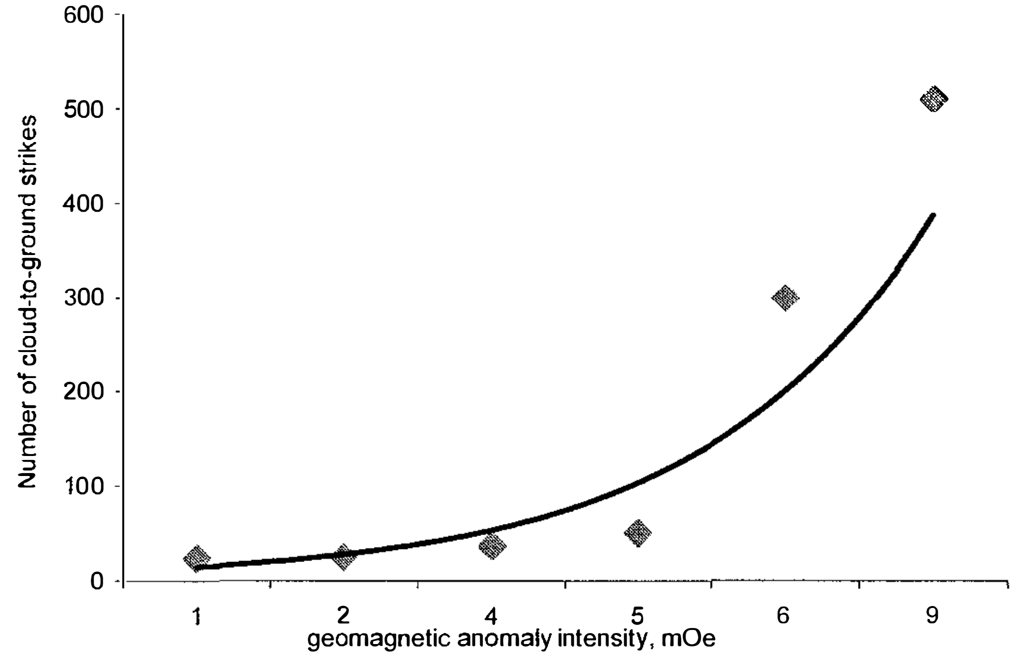

Figure 3. The relationship between the number of cloud-to-ground lightning strikes and the intensity of geomagnetic anomolies.

frequency coincide with the daily peak of electromagnetic activity. This again confirm the relationship identified between storm activity and intensity of geomagnetic anomalies.

Analysis of maps of geomagnetic anomalies of different intensity in relation to lightning fire occurrence revealed an even fire distribution pattern in areas where the geomagnetic field is characterized by uniform intensity. Our results do not confirm the conclusion of Novgorodov et al. (1982) that the number of fires occurring is greater at the boundary between two different intensities of anomalous geomagnetic field as compared with the center of a given anomaly.

Figure 4 shows the number of lightning fires occurring in areas with different intensity of geomagnetic anomalies. A relationship was established between intensity of geomagnetic anomalies and the number of lightning forest fires per unit area (correlation coefficient = 0.69). Strong anomalous magnetic fields increased the probability of lightning strike occurrence over the study area. According to our data, 47% of all lightning-caused ignitions were influenced by the intensity of geomagnetic anomalies. The results of our study and analysis of data from the literature allow us to propose the following theoretical mechanism for the effect of the Earth's magnetic anomalies on lightning frequency.

A storm cloud is known to have three stages of development (Tomilin 1986). The first stage includes development of a cumulus cloud characterized only by air indraft. The second stage is the one of maturity, or the stage of storm. At this stage, both air upflow and downflow occur. The cloud is electrolyzed and an electric charge accumulates. For the electricity to break through the air and produce lightning, the potential difference between the cloud and the ground should be several million volts. After that, the final stage of cloud destruction occurs.

Several electrolysis mechanisms become active during the development of an electric charge in a storm cloud. One of them is breakup of water drops into smaller droplets (Stekolnikov 1955; Kloss and Fasy 1979). Experiments conducted by some researchers (Feinman et al. 1977; Beituganov and Medliev 1977) suggest that electric fields promote breakup of drops into droplets of much smaller sizes. It is possible that when a storm cloud gets into a geomagnetic anomaly area, water drops begin to break up intensively and an electric charge accumulates that is capable of breaking

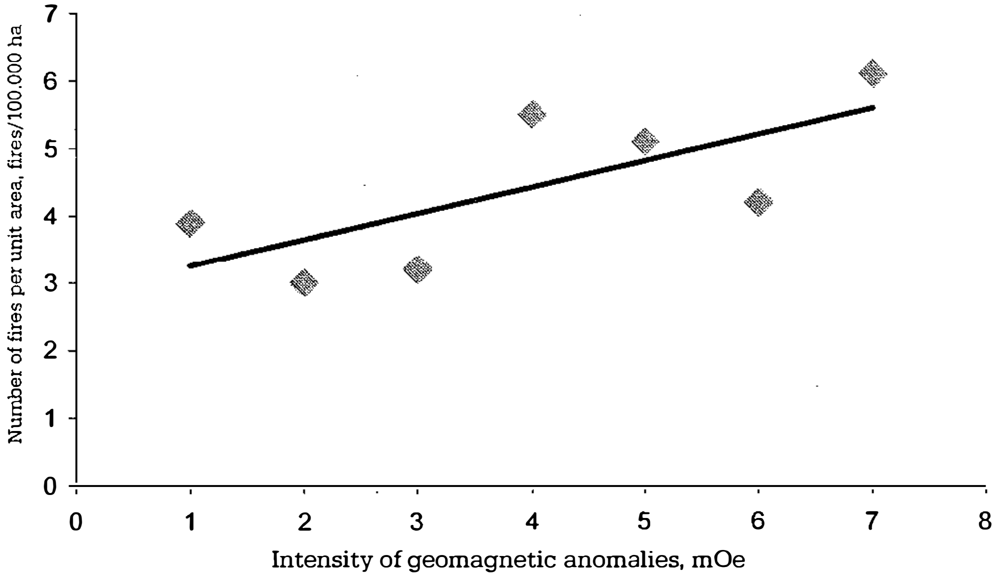

Figure 4. Lightning fire occurrence in areas differing in intensity of geomagnetic anomaly.

through the air. Thus, a highly anomalous magnetic field increases the probability of lightning over a given area.Our finding, that the number of injured trees was closely correlated with intensity of magnetic anomalies, is consistent with this theory.

Observations of tree injuries caused by lightning were conducted along 5-7 km long by 100 m wide transects established in sites differing in their intensity of geomagnetic anomaly (total of 8 transects). In the Yenisey Plain, the most common types of lightning-caused tree injuries include: bark peeling, stem wood stripping off in the direction of wood fibers, cone-shaped bark peeling from the top to the base of a tree, and bark peeling as a discontinuous strip.

Sixty six and eighteen damaged trees per square kilometer were found in sites with geomagnetic anomalies of 6 mOe and 1 mOe, respectively. We found that the number of injured trees was closely and positively, correlated with intensity of magnetic anomalies (correlation coefficient = 0.74).

Larix sibirica and *Pinus sylvestris* appear to be injured most often (51% and 39%, respectively), because individuals of these two species grow overtop of other species in the stands where they occur and they are highly resistant to other disturbances that can kill overstory species. Other tree species were injured by lightning at much lower frequencies: Siberian pine (5%), spruce (3%), and birch and aspen (1% each). Lightning tracks were most often observed on declining individuals and snags.

CONCLUSION

Frontal and local storms are common in the Yenisey Plain of central Siberia. The storm season lasts from mid-May through September. Maximum storm activity is observed in June. Local storms account for the majority of stormy days (85%).

Storms can occur under any synoptic pattern, most of them recorded at pressure highs and lows (troughs). Two storm activity peaks can be identified during a 24-hour period: 2.00 p.m. to 8.00 p.m. and midnight to 4.00 a.m.

Lightning fire ignition during the fire season is caused primarily (81%) by local storms. Lightning fire can occur in any forest type but most often in stands dominated by light-needled species.

Lightning-caused ignitions can smolder for up to 5 days. The majority of these ignitions, however, are detected during the first 24 hours after a storm passage.

Patrol flight route planning should consider characteristics of synoptic patterns and storm dynamics in the region, as well as presence of geomagnetic anomalies. An area where a storm occurred should be flown over 4-5 day periods after the event.

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THE CARBON CYCLE OF *LARIX GMELINI* FOREST ECOSYSTEMS AND IMPACTS OF MANAGEMENT PRACTICES

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ABSTRACT

We used the CENTURY model (version 4.0) to simulate the impacts of climate change and forestry management practices on the carbon (C) cycle of *Larix gmelini* (Dahurian Larch) forests. The model was validated using inventory data collected from *Larix gmelini* forest ecosystems in China.

Results indicated that *Larix gmelini* forests served as a net C sink of 2.65 t ha^{-1} yr^{-1}. Climate change and doubled carbon dioxide (CO_2) increased their net primary productivity (NPP) and capacity as a C sink. The forests served as a net C sink even when ecosystem respiration was taken into account. The gross plant biomass and NPP of *Larix gmelini* forests increased and the soil C content decreased, after the temperature was increased by 2^0C. Different forestry management practices produced different impacts on the forest ecosystem. Forest felling decreased the gross plant biomass and NPP while increased soil C content. When the harvesting rate was increased, the magnitude of the change (increase or decrease) of the forests' gross plant biomass, NPP and soil C content increased and more time was required for the forests' restoration. Annual removal of litter decreased the soil C content and increased the gross plant biomass of *Larix gmelini* forests. NPP increased in the first 10 years and then decreased gradually. After 100 years, it was close to the NPP of the forests without litter removal. There were no differences among the impacts resulting from different litter removal intensities.

Keywords: *Larix gmelini* forest, carbon cycle, climate change, management practices, CENTURY model.

INTRODUCTION

Global change resulted from elevated concentrations of greenhouse gases, especially CO_2, in the atmosphere has become a hot point of ecosystem research. As an important component of terrestrial ecosystems, forest ecosystems play a key role in regulating the balance of atmospheric CO_2 because of their large area, biomass, net primary productivity (NPP) and capacity for carbon (C) storage (Zheng, 1989; Jiang, 1996). Forests are closely linked to the major greenhouse gases in the atmosphere, especially CO_2, which may induce global warming. Forests sequester CO_2 through photosynthesis and release CO_2 through plant and soil respiration and litter decomposition, hence regulating terrestrial uptake and output of atmospheric CO_2. Thus, any change to *Larix gmelini* forests are expected to result in great changes to atmospheric CO_2 concentrations.

Larix gmelini forests located in the Great Khingan Mountains serve as a focal point of forest ecosystem research for the world because it is the dominant vegetation in cold-temperate conifer forests in China, as well as being a major component of boreal forests (Taiga) in the world. They cover about 15.6 x 10^6 ha or 13.2% of the total forested area in China (118 x 10^6 ha) and provide about 4700 x 10^6 US\$ or 4% of the gross annual forest ecosystem services in China (Jiang et al., 1999). It is the most sensitive area to global change in China. According to the study of Zhang et al. (1993), it is likely that *Larix gmelini* forests will move northward out of China under the conditions of 2^0C or 4^0C increase in temperature and 20% increase in precipitation. Consequently, interest has recently developed to study the C cycle and the impacts of climate change and human activities on *Larix gmelini* forests.

Presently, two types of models are being applied to simulate the growth dynamics of forests in China. Gap models, such as NEWCOPE (Yan, 1997; 2000), ZELIG (Yu et al., 1996; 1997), and KOPIDE (Shao, 1989), simulate the dynamics of forest growth and succession. However, they do not simulate the C cycle of forest ecosystems. Biogeochemical cycling models, such as BIOME-BGC (Running and Hunt, 1993), TEM (Melillo et al., 1993) and CENTURY (Parton et al., 1993), can simulate the processes of ecosystems' biogeochemical cycling. They are mainly used to simulate C and nutrient cycles among vegetation, litter and soil organic matter pools. Among the last three models, only the CENTURY model is able to simulate the impact of natural/man-made disturbances (e.g. fire and felling) on forest ecosystems.

In this paper, we used the CENTURY model (version 4.0) to simulate the C cycle of *Larix gmelini* forests in China. The objective of this study is to evaluate the carbon balance and analyze the potential impacts of global change and forestry management practices on the C cycle of *Larix gmelini* forest ecosystems.

STUDY AREA

Larix gmelini forests are mainly located in the Great Khingan Mountains and Xiaoxing'an Mountains (43^0N~53^0N to 118^0E~132^0E). A temperate monsoon climate zone characterizes the climate. Its winter can be as long as nine months and the summer less than one month long. There is almost no summer in most parts of this area. The annual mean temperature ranges from – 4.7^0C to 5.4^0C and the frost-free period lasts from 90-180 days. The annual mean precipitation is about 300-600 mm. The soils are Podsolic with a very thin solum (20-30 cm.). The main vegetation types include *Rhododendron dauricum* (Dahurian Rhododendron) and *Larix gmelini* (Dahurian Larch) forests. The arbor layer generally is pure *Larix gmelini* forest and a small number of forests are mixed with *Betula platyphylla* (Asian White Birch) and *Pinus sylvestris* var. *Mongolica* (Mongolian Scotch Pine). The coverage rate ranges from 0.4 to 0.9. The shrub layer is made up of *Rhododendron dauricum* (Dahurian Rhododendron), with coverage of more than 50% (Xu, 1998).

METHODS

The CENTURY model was validated using observed data for biomass and soil C in *Larix gmelini* forests. Then, the model was used to assess the C source/sink relationships and simulate the impacts of disturbances, including responses to doubled CO_2, climate change, forest fire, forest felling and litter removal (Figure 1).

Model Description

The CENTURY model simulates long-term dynamics of Carbon (C), Nitrogen (N), Phosphorous (P), and Sulfur (S) for different Plant-Soil systems. It can simulate the dynamics of grassland, agricultural crop, forest and savanna ecosystems. The grassland/crop and forest systems have different plant production sub models linked to a common soil organic matter sub model. The model runs using a monthly time step and the major input variables for the model include:

1. monthly average maximum and minimum air temperatures,
2. monthly precipitation,
3. lignin content of plant material,
4. plant N, P and S contents,
5. soil texture,
6. atmospheric and soil N inputs, and
7. initial soil C, N, P and S levels.

The output variables include biomass, NPP, soil organic matter content and soil water (Parton et al., 1988; 1993; 1994).

Input Parameterization

1. Climate parameters; monthly maximum and minimum temperature and monthly precipitation from 1953 to 1995,
2. Location parameters; latitude and longitude,
3. Soil parameters; fraction of sand, silt and clay in soil; bulk density; number of layers and depth of different layers; pH,

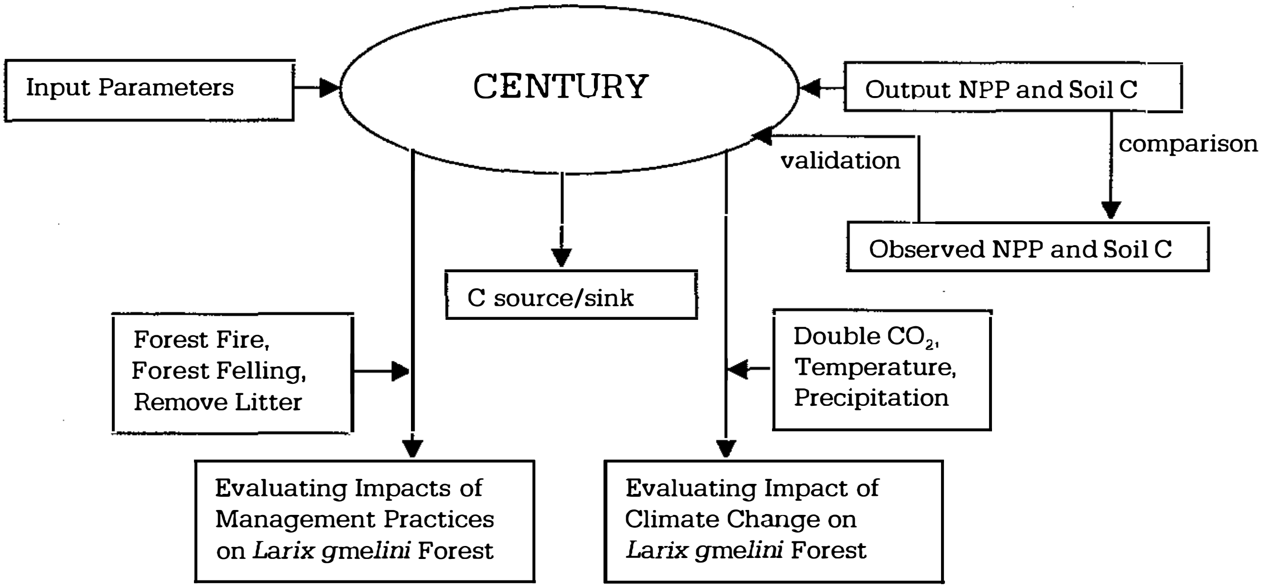

Figure 1. Flow chart for simulating the carbon cycle of *Larix gmelini* forests.

4. Plant parameters; maximum gross and net forest production, lignin fraction in the live forest system components, C allocation fraction of new production for juvenile and mature forests, monthly death rate fractions for forest components, maximum leaf are index (LAI).
5. Organic matter initial parameters; initial value for C, N in forest system components; initial value for soil C, N; initial C, N and C/N ratio for litters; initial lignin content of dead forest components; and so on.

The data for parameterization were collected from the Editorial Committee of "Inner Mongolic Forest" (1989), Fan et al. (1992), Feng et al. (1985; 1994 a,b), Feng (1999), Gu (1987), Institute of Forest Soil (1980), Jiang (1996), Liu et al. (1990), Liu et al. (1997), Wu et al. (1995), Xu (1998) and Zhou (1991).

Simulation Scenarios

This paper examines the simulated effects of doubled CO_2, climate change (temperature, precipitation), and forestry management practices (forest felling, litter removal) on *Larix gmelini* forests, soil C concentration, gross plant biomass, NPP and C sink/source relationships. The scenarios examined include;

1. Doubled CO_2 which refers to 700 ppm compared with the normal concentration 350 ppm,
2. Climate change includes two scenarios (temperature increased by 2°C with precipitation increased by 20%, temperature increased by 2^0C with precipitation decreased by 20%) compared with the unchanged condition,
3. Forest felling at low (fell 20% of the large wood biomass), medium (fell 60% of the large wood biomass), and high (fell 90% of the large wood biomass) rates. The forest was considered restored when the forest's gross plant biomass reached 95% of the level before the forest was cut,
4. Litter removal with low (removing 20% of litter), medium (removing 60% of litter) and high (removing 90% of litter) levels.

Relevant parameters were modified based on these disturbances.

RESULTS AND DISCUSSION

Model Validation

The CENTURY model was primarily developed for grassland ecosystems. It has seldom been used in Chinese forest research although it can simulate the C cycle of forest ecosystems. In China, it has only been used to simulate the C cycle of tropical rain forest ecosystems (Huang, 2000), and has hot been used in boreal forest research. To validate the model for simulating the C cycle of boreal forests, we input the initial parameters and ran the model for 500 years to simulate the change from bare ground to mature forest. A comparison between the simulated and observed values for biomass (age at 120 years) is given in Table 1 and Figure 2.

The results indicate that the simulated values of the plant biomass components are very close to the observed values. The relative errors were less than 0.11 (Figure 2a). The simulated soil C content (5539 g m^{-2}) was also in the range of measured soil C content (5382.4 – 9419.2 g m^{-2} (Figure 2b). So, the CENTURY model could be applied to simulate the C cycle of *Larix gmelini* forests.

C Source/Sink

The simulation results indicate that *Larix gmelini* forests sequester 4.03 t C ha^{-1} yr^{-1} through photosynthesis (after subtracting C released from plant respiration) and release 1.38 t C ha^{-1} yr^{-1} through soil respiration and litter decomposition. Thus, at present, *Larix gmelini* forests in China are a net C sink of about 2.65 t ha^{-1} yr^{-1}.

Impacts of Doubled CO_2 on *Larix gmelini* Forests

Doubling the atmospheric CO_2 concentration increased *Larix gmelini* forest's NPP by 9.8%. The net C sink was about 2.69 t ha^{-1} yr^{-1}. This is an increase of 1.5% compared with the present level even when soil respiration and litter decomposition were taken into account.

Table 1. Validation results of CENTURY model.
(Observed data calculated from Feng et al., 1985)

Compared Items	Biomass Component (g m^{-2})					
	Gross	Coarse Root	Fine Root	Fine Branch	Large Wood	Leaf
Simulated Values	11 000	2 600	49	490	8 000	98
Observed Values	11 000	2 600	48	489	8 000	88
Error	0.00	0.00	0.02	0.00	0.00	0.11

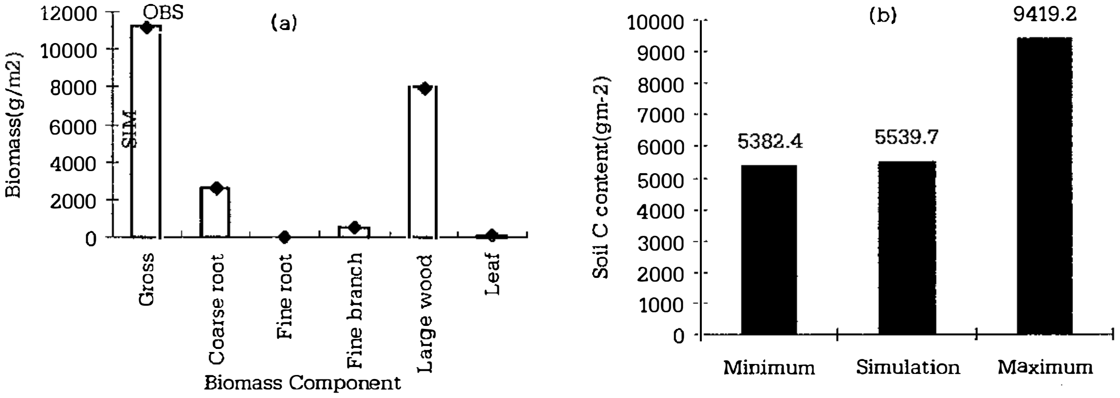

Figure 2. Comparison between the simulated values from the CENTURY model and the observed values.

(a) Plant biomass and,

(b) Soil C content: "min" and "max" refers to the practical minimum and maximum values of soil C content in *Larix gmelini* forests; "simu" refers to the simulated value.

Impacts of Climate Change on the Soil C Content of *Larix gmelini* Forests

1. Soil C content: The soil C content of *Larix gmelini* forests decreased when the temperature increased by 2^{0}C. After 50 years (2050), when temperature increased by 2^{0}C and precipitation increased by 20% or decreased by 20%, the soil C content of *Larix gmelini* forest decreased by 4.4% and 3.2%, respectively compared to that under normal climatic conditions (Figure 3a).
2. Gross plant biomass; The gross plant biomass of *Larix gmelini* forests increased in response to a 2^{0}C increase in temperature. After 50 years (2050), under the conditions of temperature increasing by 2^{0}C with precipitation increasing by 20% or with precipitation decreasing by 20%, the gross plant biomass increased by 0.6% and 1.4% respectively, compared to that under normal climatic conditions (Figure 3b).
3. NPP: The NPP of *Larix gmelini* forests increased in response to a 2^{0}C increase in temperature. After 50 years (2050), under the conditions of temperature increasing by 2^{0}C with precipitation increasing by 20% or with precipitation decreasing by 20%, NPP increased by 8.3% and 10% respectively, compared to that under normal climatic conditions (Figure 3c).
4. C source/sink: *Larix gmelini* forests were still a net C sink and their capacity to sequester C in response to a 2^{0}C increase in temperature. After 50 years (2050), under the conditions of temperature increasing by 2^{0}C with precipitation increasing by 20% or with precipitation decreasing by 20%, the annual capacity to sequester C increased by 9.5% and 11.4%, compared to that under normal conditions (Figure 3d).

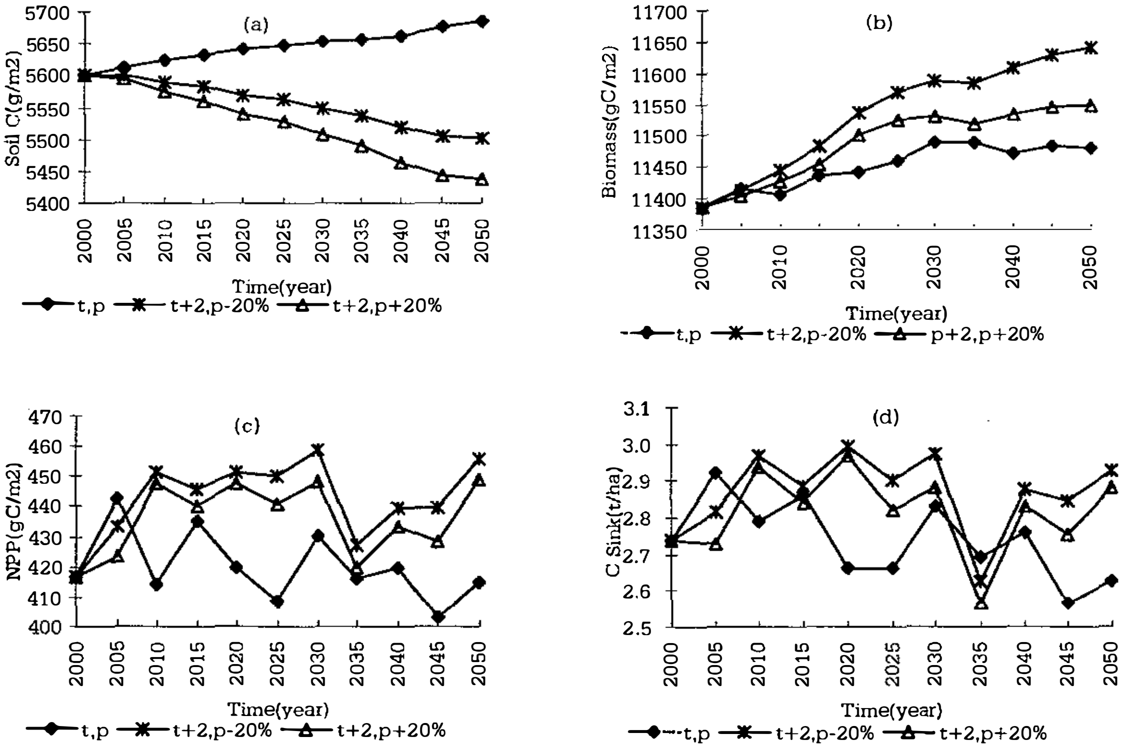

Figure 3. Impacts of climate change on *Larix gmelini* forests.
(a) soil carbon, (b) biomass, (c) NPP(net primary productivity), and (d) carbon sink
t,p refers to normal temperature and precipitation;
t+2, p-20% refers to a temperature increase of 2°C and precipitation decrease of 20%;
t+2, p+20% refers to a temperature increase of 2°C and precipitation increase of 20%.

Impact of Forest Felling on *Larix gmelini* Forests

1. Soil C content: The soil C content of *Larix gmelini* forests initially increased after forest felling and then decreased along with the forests' growth. The higher the rate of forest felling, the more the soil C content decreased. Over 50 years, the soil C content gradually increased as the forests matured. It reached the before forest felling level at the same time as gross plant biomass (Figure 4a).
2. Gross plant biomass: The more the forest was felled, the more its gross plant biomass decreased, and the longer the time to restoration. Light, medium and strong felling decreased the gross plant biomass by 15%, 45.6% and 74.9% respectively and required 96, 173 and 261 years respectively, for restoration (Figure 4b).
3. NPP: In the first year after felling, the NPP of *Larix gmelini* forests decreased sharply. The more the forest was felled, the more its NPP decreased. NPP decreased by 5.9%, 19% and 40.9% in the first year after light, medium and strong felling, respectively. After the first year NPP increased gradually along with the forests' renewal. When the forests' were restored, their NPP was 0.2% lower after light felling, but after medium and strong felling NPP was 1.8% and 6.8% higher respectively, compared to that without felling (Figure 4c).
4. C Source/sink: The capacity of C sequestration of *Larix gmelini* forests decreased in the first year after forest felling. The more the forests were felled, the more their capacity as a C sink decreased. But the forests were still a C sink (Figure 4d).

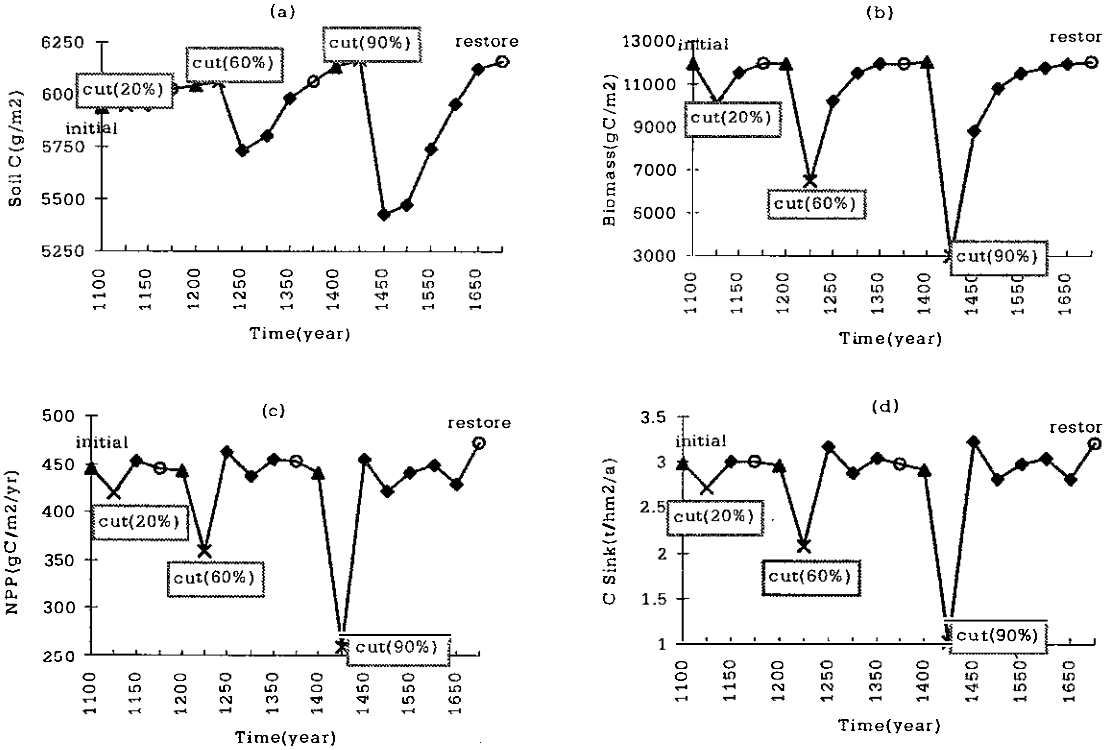

Figure 4. Impact of forest felling on *Larix gmelini* forests.
(a) soil carbon, (b) biomass, (c) NPP (net primary productivity), and (d) carbon sink

▲ "initial" refers to the value before forest felling;

○ "restore" refers to the value when the forest is restored after felling

Impact of Litter-removal on *Larix gmelini* Forests

1. Soil C content: Removing litter resulted in a decrease of soil C content of *Larix gmelini* forests (Figure 5a). After 100 years of continuous removal of litter, the soil C content decreased by about 30% compared with those without removal. There were no differences among the impacts resulting from three different intensities of litter removal.
2. Gross plant biomass: Annually removing litter increased the gross plant biomass of *Larix gmelini* forests, especially in the first 30 years. From then on, it increased by 3% compared with those without removal and remained stable. There were no differences among the impacts resulting from three different intensities of litter removal (Figure 5b).
3. NPP: In the first 10 years with annual litter-removal, the NPP of *Larix gmelini* forests increased sharply and was obviously higher than NPP without litter removal. After 10 years, NPP decreased gradually, but was still higher than NPP without litter removal before 60 years. There were no obvious differences between the NPP of the forest with and without removal after 60 years. There were no differences among the impacts resulting from three different intensities of litter removal (Figure 5c).
4. C Source/sink: In the first 10 years with litter-removal, the capacity of the C sink of *Larix gmelini* forests increased sharply and was obviously higher than for those without litter removal. Later, this capacity decreased gradually, but it was still higher than those without litter removal. After 100 years, the C sink was 3.41 t ha^{-1} and increased by 22% compared with those without removal (2.79 t ha^{-1}). There were no differences among the impacts resulting from three different intensities of litter removal (Figure 5d).

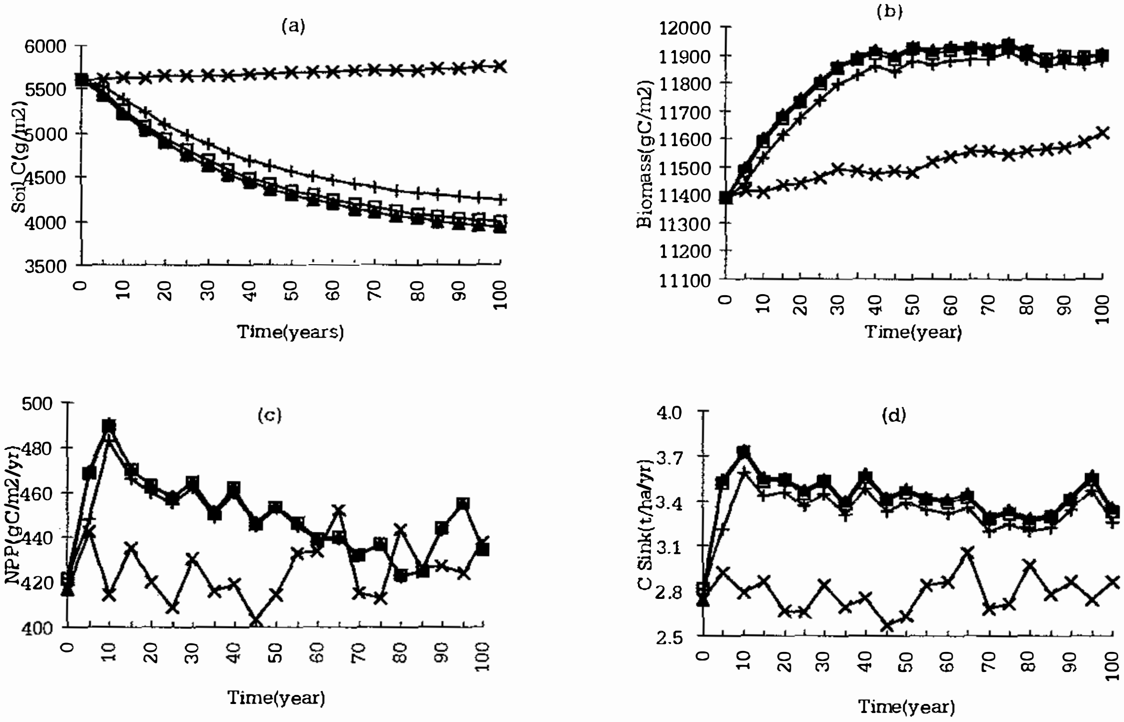

Figure 5. Impact of litter-removal on *Larix gmelini* forests.
(a) soil carbon, (b) biomass, (c) NPP(net primary productivity), and (d) carbon sink
+ - 20% removal; ▲ - 60% removal; □ - 90% removal; × - no removal

DISCUSSION

Larix gmelini forests serve as a net C sink of about 2.65 t ha^{-1} yr^{-1} and will still be a net C sink after climate change.

Climate change and double CO_2 is favorable for the growth of boreal forest. Low temperature is a major limiting factor in this area. The elevation of temperature will stimulate the growth of forests. The impact of precipitation indicates that water in this area is adequate or even excessive.

Forest felling decreased the gross plant biomass and NPP and increased the soil C content of *Larix gmelini* forests. The stronger the felling intensity, the greater the gross plant biomass, NPP and soil C content changed, and also the longer the time required for restoration.

Much litter accumulates on the soil surface because of high precipitation, low temperature and slow decomposition. These conditions favor low soil temperatures and prevent water from running off. But, too thick a litter layer may block seeds from taking root and prevent regeneration. Removing some of the litter might be favorable for seeds to take root and stimulated the forest's renewal. After an extended period of continuous litter removal, the soil fertility probably became unfavorable for forest growth, NPP decreased to the initial level and the biomass did not increase.

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CARBON SEQUESTRATION ON PRIVATELY OWNED MARGINAL AGRICULTURAL LANDS: AN ECOLOGICAL - ECONOMIC ANALYSIS OF AFFORESTATION IN SASKATCHEWAN

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ABSTRACT

The Kyoto Protocol to the United Nations Framework Convention on Climate Change commits Canada to reducing its CO_2 emissions to levels 6% below those in 1990. Afforestation, i.e. planting trees where they previously did not occur, is recognized by the Protocol as an important carbon sink that may be used by countries in meeting their Kyoto targets. In the prairie region of Canada, Saskatchewan has large areas of marginal agricultural land that could be suitable for the establishment of new forest plantations. Our estimates indicate that approximately 260,000 ha may be available for afforestation in Saskatchewan, largely in the form of unimproved pasture. This area would support approximately 5.9×10^6 tonnes of carbon storage over the next 80-100 yr., and timber volume production of approximately 21×10^6 m^3 of merchantable wood. Our economic analysis indicated that conversion of these pasture lands to tree plantations would be profitable to land owners only at a discount rate of less than 4%. Therefore, availability of cheaper credit may play a pivotal role in such decisions. Distribution of revenues from final timber sales to the land owners earlier in the life of the plantation, or a regular payment for carbon sequestration may also lead to more land conversion in Saskatchewan. Our review of existing policy suggested that there are a number of disincentives that may discourage land owners from establishing plantations even if economic benefits exist. These include unfavorable tax regimes, government timber allocation and pricing policies, and the lack of extension and research related to plantation genetics and silviculture appropriate for marginal agricultural land.

Keywords: afforestation, economics of carbon sequestration, climate change, Kyoto Protocol, plantations

INTRODUCTION

The United Nations Framework Convention on Climate Change (FCCC) was ratified by Canada in 1992. The FCCC voluntarily committed the signatories to reducing their greenhouse gas emissions to 1990 levels. The majority of countries failed to meet this target, and in 1997 the Protocol to the FCCC was written in Kyoto, Japan. The Protocol is a legally binding agreement that, when ratified, will require developed countries to reduce their emissions; e.g., Canada's goal for emissions reduction is to be 6% below those in 1990. Carbon sinks are recognized under the Kyoto Protocol as a factor that must be included in calculating a country's net CO_2 emissions. Carbon sinks and sources arise from three types of land use related to forests: afforestation, planting trees where they did not grow previously; reforestation, planting trees following forest harvesting; and deforestation, the permanent removal of forest cover (see Lund 1999 for a discussion of the definitions of these terms). Afforestation is a land use that can sequester large amounts of carbon which can be quantified relatively easily.

In addition to its role in carbon sequestration, afforestation has a number of other benefits. Many areas that were cleared for agriculture in the past have proven to be of low productivity and in some cases have been abandoned. These areas are often prone to soil erosion or have been overgrazed. Afforestation is a land use practice that can help stabilize soil and restore soil organic matter on degraded sites (Johnston et al., 2000). Afforestation has other benefits in providing protection against wind erosion, increasing wildlife habitat diversity and maintaining water quality in riparian areas (Williams et al. 1997).

The objective of this study was to determine the land area available in Saskatchewan for afforestation; to determine how much carbon could be sequestered on this land base; to examine the economic implications of an afforestation program, particularly for individual land owners; and to recommend changes to current policy that could encourage land owners to establish plantations on their land.

METHODS

We began the land base determination by assuming that only land within a reasonable transport distance to existing processing facilities would be economically attractive for tree planting. Current mills in Saskatchewan are in the towns of Meadow Lake, Prince Albert and Hudson Bay (Figure 1). Therefore

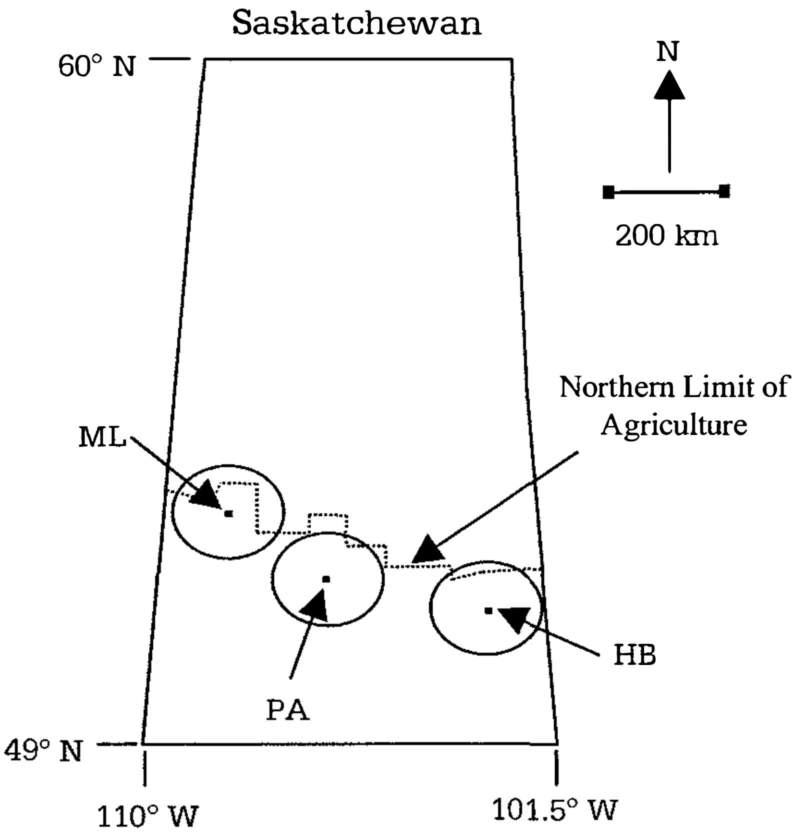

Figure 1. Location of three study sites within Saskatchewan: ML, Meadow Lake; PA, Prince Albert; HB, Hudson Bay. Circles indicate 100 km radius around each city.

we limited our analysis to lands within a 100 km radius of these three towns. This area was further limited to private land; i.e. government (Crown) land was not considered. We then consulted maps from the Saskatchewan Department of Municipal Affairs indicating the location of Rural Municipalities (RMs) within the 100 km radius circles. Data from the Statistics Canada (1997) census showed the land area occupied by various agricultural practices within each RM. We further assumed that only land currently under unimproved pasture would be available for afforestation; these areas have a minimal economic investment and therefore the opportunity costs associated with conversion to plantations are the lowest of any agricultural land. In addition, the existing transportation system for collection of agricultural products and the distance from these areas to ports (e.g., the Great Lakes or west coast ports) suggests that these areas are most vulnerable to increased freight rates. Thus, these regions can be considered marginal using both economic variables and physical characteristics. The area in unimproved pasture in RMs within 100 km of the three centers was considered to be the available land base for afforestation.

The potential for carbon sequestration was determined by estimating tree volume growth per hectare and multiplying by the land base estimates. The forest industry in Saskatchewan processes mostly white spruce (*Picea glauca* (Moench) A. Voss), jack pine (*Pinus banksiana* Lamb.) and trembling aspen (*Populus tremuloides* Michx.), with minor amounts of white birch (*Betula papyrifera* Michx.) and black spruce (*Picea mariana* (Mill.) BSP). We assumed that the most economically attractive species for land owners would be white spruce and aspen. We also included hybrid poplar (*Populus* spp.) due to its rapid growth rate, which provides not only rapid carbon accumulation but also reasonably quick revenues for the land owner. The area planted to each species varied among the three study sites in order to take into account climate and soil limitations. In the Meadow Lake study area, we assumed that spruce would be dominant due to the relatively dry climate; in the Prince Albert area we assumed an equal mixture of the three species; and in the Hudson Bay area we assumed that hybrid poplar would be dominant. This last assumption was based on a recent analysis of hybrid poplar land suitability using soils and climate criteria carried out by the Prairie Farm Rehabilitation Administration (Schroeder 2000). We used Saskatchewan forest inventory data to generate yield curves for aspen and white spruce, using data from the area along the southern boundary of the provincially managed commercial forest nearest to the three processing centres. For hybrid poplar, we used growth data for good sites reported in Peterson et al. (1999). Yield data produced estimates of merchantable volume production, which were then used in the economic calculations.

Carbon sequestration was determined by converting stem wood volume to whole-tree biomass by using conversion factors in Alemdag (1983, 1984), and assuming that biomass is 50% carbon (Kurz et al. 1992). We then multiplied carbon sequestration per hectare by the land area allocations described above. In order to determine the time course of carbon accumulation, we assumed, using common practice in the region, that rotation lengths were 80 years for aspen and spruce and 20 years for hybrid poplar, and that 10% of the land area available would be planted each year. Our analysis was limited to carbon sequestered in above-ground tree biomass.

Economic desirability of converting land from unimproved pasture to trees was determined using the accounting perspective of the land owner. Return to the land owner discounted over the rotation as compared to the return from unimproved pasture was used as the criterion to decide whether land use conversion from pasture to plantation was desirable. If the returns from this conversion were positive, conversion to plantations is desirable; if the criterion shows a value of zero or less, the land owner is better off continuing with the existing land use practice.

Our economic analysis of afforestation was based on a life-cycle analysis of two options: (1), maintain the land under its present use, i.e. unimproved pasture; or (2), break the land and plant with trees, i.e., the afforestation option. Social desirability of the afforestation option was determined by developing a life-cycle marginal net benefits to society. In this approach unimproved pasture was assumed to be producing forage that is utilized by cattle. However, cattle production was not included explicitly in the analysis. Instead, a shadow price of grass for cattle production was derived, and used for determining return to

Table 1. Assumptions for the use of unimproved pastureland in Saskatchewan.

Particulars	Unit of Observation	Low Quality Pastureland	Medium Quality Pastureland
Fencing (life 40 years)	$ per ha	111.20	111.20
Animal unit months	# per ha	0.593	0.815
Forage consumed	Kg/day/animal unit	26	26
No. of days on pasture	#	90	100
Total Forage yield	Tonnes/ha	0.255	0.389
Price of Forage	$/tonne	$35	$40
Gross returns	$/ha	$8.91	$15.57

land and equity under the first type of land use. The assumed development costs for low and medium productivity pasture are shown in Table 1. For the second option, three types of trees – white spruce, hybrid poplar, and aspen, were planted on designated areas. The life-cycle analysis involved costs and revenues from the plantations, and were first converted into return to land and equity for a given time period, and then into present values using various discount rates.

For unimproved pasture, development costs were charged in year one (which is equivalent to t = 1, the initial year of experiment), and the pastures were assumed to be used for cattle grazing from year two through the life of the project. It was assumed no other improvements to the pasture would be required if proper management guidelines were followed. Since pasture productivity is variable, we determined the costs and benefits at low and medium productivity levels. On the cost side it was assumed that since the land is already in its natural state, the only cost to be incurred, prior to using it for grazing, would be fencing.

On the revenue side, no direct estimate of yield of forage from unimproved pasture is available. Therefore, we used data from Abouguendia (1990) giving stocking rates of cattle on low and medium quality pasture.

For forest plantations, we used the costs of establishment and harvesting given in Lindenbach (2000). These costs were assigned in year one of the rotation, which is the start of the simulation for agro-forestry. No other costs were incurred until the harvest of the timber. At this time, costs included harvesting and transportation to the pricing point (mills). The price of timber is difficult to estimate, particularly for the next 80 years, so we relied on the opinion of experts in the Saskatchewan forest industry. The general consensus was that the real price of timber is expected to increase in the future. All calculations were done in 1999 dollars. These assumptions are shown in Table 2.

Table 2. Costs and revenues from forestry rotations, by species.

Particulars	Unit	White Spruce	Hybrid Poplar	Aspen
Rotation length	Years	80	20	80
Site preparation[1]	$/ha	150	150	150
Seedling cost[2]	$/ha	440	1,000	1,000
Total establishment costs	$/ha	836	1,396	1,396
Harvest costs[3]	$/ha	2,421	2,421	2,421
Transport cost[4]	$/ha	711	711	711
Yield	M^3/ha	184.4	251.4	150.0
Price	$/$m^3$	60.00	55.00	58.00

[1] It is estimated that cost on pasturelands, if tilled, could be reduced to $120 per ha. However, unimproved pastures were assumed to be untilled

[2] Assuming planting density of 2,000 trees per hectare. Price of white spruce seedlings was assumed to be $0.22 each, while that for hardwood seedlings was $0.50 each. Data were obtained from Pacific Reforestation Technologies tree nursery, Prince Albert, SK.

[3] Assuming cost per tree of $1.23.

[4] Based on cost for Minnesota, obtained from Lindenbach (2000).

Three alternative rates of discounts were used: a medium level of 5%, plus a sensitivity analysis using low (3%) and high (8%) discount rates. All analyses were carried over an 80-year period from the time conversion took place. Thus, for each year, full lifecycle costs and revenues were included.

RESULTS

Land Base and Biomass Accumulation

We estimated the total land base in Saskatchewan available for afforestation to be approximately 260,000 ha, distributed among the three study areas as shown in Table 3.

Table 3. Area (ha) available for afforestation in the three study areas in Saskatchewan shown in Figure 1.

Study Area	Unimproved Pasture	Per cent
Meadow Lake	111,123	42.9
Prince Albert	89,462	34.5
Hudson Bay	58,726	22.6
Total	259,311	

About 43% of this area is in the Meadow Lake study area, with approximately 33% in the Prince Albert area and the remainder in the Hudson Bay area. Allocation of this area to the three species (spruce, aspen and hybrid poplar) is shown in Table 4, and reflects growing conditions in the three areas as described above. The volume and net carbon accumulation over the next 100 years is shown in Table 5. Production reaches a steady state in 2080, with a merchantable volume of about 21 x 10^6 m^3 and carbon sequestration of about 6 x 10^6 tonnes. The carbon sequestered in this land base under unimproved pasture was determined using data for temperate grasslands from Bolin and Sukumar (2001), and is about 6.3 x 10^5 tonnes, or an order of magnitude less. Merchantable volume was used in calculating the economic value of trees versus pasture for individual land owners.

Table 4. Area (ha) allocated to each species in the three study areas in Saskatchewan shown in Figure 1.
(Totals may not agree with those in Table 1 due to rounding).

Study Area	Spruce	Aspen	Hybrid Poplar	Total
Meadow Lake	50,000	30,000	31,000	111,000
Prince Albert	30,000	30,000	30,000	90,000
Hudson Bay	15,000	15,000	30,000	60,000
Total	95,000	75,000	91,000	261,000

Economic Analysis

Results of economic desirability criterion for three discount rates are shown in Table 6. As we expected, forest rotations lose their economic desirability at higher discount rates. This is because any benefits (or costs) that are received (or incurred) in the long run (30 years or longer) become insignificant. At a discount rate of 3%, the three areas showed a positive benefit from afforestation. However, as the discount rate increases, net marginal benefits (afforestation minus pasture) become negative, suggesting that afforestation is not in the best interest of the land owners.

These results can be explained by two factors. First, we assumed that society has not placed any economic value on the amount of carbon being sequestered. Therefore, land owners are forced to look at the commercial products produced under the two options. Second, there is a lack of revenues from afforestation during the earlier part a given tree rotation. Land owners receive no benefits under this option until the trees are harvested and sold. In discounted terms, this reduced the desirability of the afforestation option significantly, particularly for the longer rotations.

To show regional differences in the relative economic gains from afforestation, net benefits in Table 6 were converted to a per hectare basis, shown in Table 7. Generally speaking, net benefits were lower when afforestation was on poor quality pasture, as compared to medium quality lands. However, even when the pasture was of poor quality, if the discount rate was equal to or higher than 5%, afforestation was not an economically desirable option. The Hudson Bay area had a consistently higher net benefit per ha, whereas the Meadow Lake area had the lowest. In part, this reflects the mix of species planted in various areas. The Hudson Bay area has the

Table 5. Volume and carbon accumulation for the three study areas shown in Figure 1.

Year	Volume (m^3)				Carbon (t)			
	Meadow Lake	Prince Albert	Hudson Bay	Total	Meadow Lake	Prince Albert	Hudson Bay	Total
2000	0	0	0		0	0	0	0
2010	409,400	396,200	396,200	1,201,700	127,900	123,800	123,800	375,500
2020	1,234,800	1,186,700	1,168,400	3,589,900	383,300	369,000	364,200	1,116,500
2030	1,438,700	1,345,500	1,247,900	4,032,100	435,500	410,400	384,900	1,230,800
2040	2,656,700	2,437,900	2,171,100	7,265,700	790,900	734,000	664,500	2,189,400
2050	3,362,600	2,975,900	2,440,100	8,778,600	970,300	873,300	734,200	2,577,900
2060	5,110,500	4,459,700	3,559,000	13,129,200	1,459,200	1,297,400	1,064,000	3,820,600
2070	6,313,900	5,355,600	4,007,000	15,676,500	1,763,100	1,527,800	1,179,300	4,470,200
2080	8,489,500	7,137,200	5,274,800	20,901,600	2,358,000	2,026,900	1,546,600	5,931,600
2090	8,489,500	7,137,200	5,274,800	20,901,600	2,358,000	2,026,900	1,546,600	5,931,600
2100	8,489,500	7,137,200	5,274,800	20,901,600	2,358,000	2,026,900	1,546,600	5,931,600

Table 6. Net benefits to land owners (\$ x 10^6) from afforestation on private unimproved pasture lands in the three study areas. Values represent low and medium pasture quality, respectively.

Study Area	Discount Rate		
	3%	5%	8%
Meadow Lake	95.5; 117.0	- 32.8; - 19.0	- 84.0; - 75.5
Prince Albert	98.3; 115.8	- 18.3; - 7.1	- 67.3; - 60.4
Hudson Bay	121.2; 132.8	14.6; 22.0	- 34.6; - 30.0
Total	314.9; 365.7	- 36.5; - 4.2	- 186.0; - 165.9

Table 7. Net benefits (\$/ha) from afforestation of private unimproved pasture lands in the three study areas. Values represent low and medium pasture quality, respectively.

Study Area	Discount Rate		
	3%	5%	8%
Meadow Lake	868; 1,211	- 298; - 140	- 763; - 715
Prince Albert	1,092; 1,286	- 203; - 79	- 748; - 671
Hudson Bay	2,019; 2,213	243; 367	- 577; - 500
Average	1,211; 1,406	- 140; - 16	- 715; - 638

largest proportion of hybrid poplar, relative to the Meadow Lake and Prince Albert areas (50% vs. 28 and 33%, respectively; see Table 4). This again supports the earlier premise that unless benefits are generated over a shorter period of time, afforestation is not a preferred option of land owners, particularly as the discount rate increases. This is also supported by the nature of the net benefit function, as shown in Figure 2. As the discount rate increases beyond 3%,

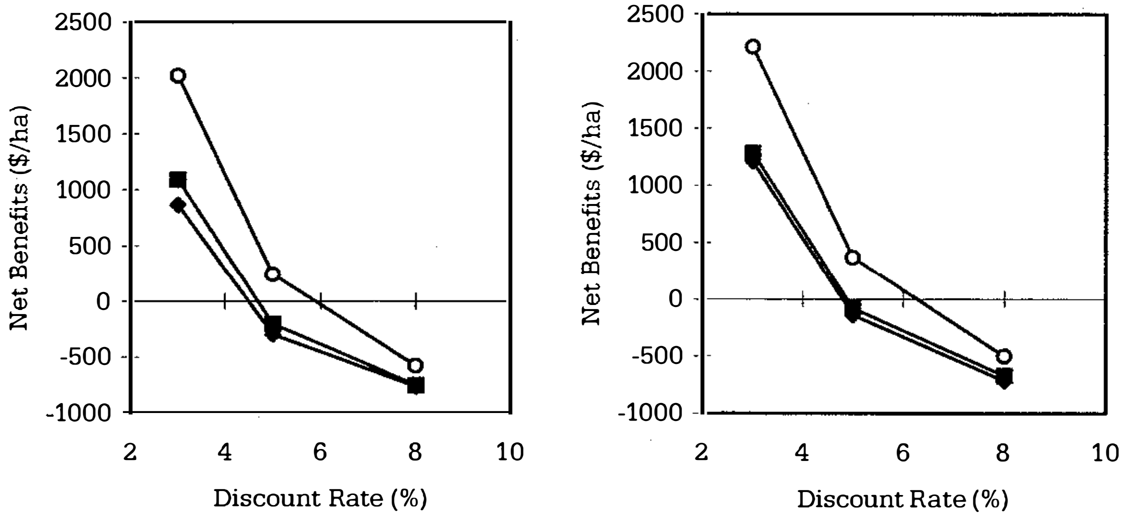

Figure 2. Net benefits from afforestation under alternative discount rates: A, low quality pasture; B, medium quality pasture.

net benefits become lower and eventually become negative. The rate of decline in the three areas is virtually the same for the medium quality pasture. However, for low quality pasture, the Hudson Bay area has a distinct advantage over the other two areas. Also the rate of discount at which net benefits go to zero is slightly higher (5.5 to 6%) in this area, compared to the other two (4.75% for Prince Albert and 4.25 to 4.75% for Meadow Lake).

DISCUSSION

The current annual allowable cut (AAC) in Saskatchewan is approximately 6 x 10^6 m^3. This study indicates that the potential volume production from afforestation would add considerably to the current AAC. With agricultural economies suffering throughout North America, afforestation may provide a realistic alternative to current land use practices. In addition, the Kyoto Protocol recognizes that emissions trading is a legitimate way for countries to meet their emission reduction target. Emission trading based on carbon sequestered in afforested agricultural land appears to have potential. For example, Canada's projected CO_2 emissions for 2000 are approximately 694 x 10^6 tonnes (AMG 1999), so that an afforestation program utilizing the land base described here could potentially sequester on the order of 1% of the nation's emissions. Combined with other carbon sinks and direct emission reductions, afforestation could play an important role in assisting Canada to meet its international obligations.

The economic analysis indicated that afforestation could be a feasible alternative for land owners in the forest fringe area of northern Saskatchewan on poorer quality pasture. However, the rate of discount would have to be rather low (less than four percent) to encourage conversion to plantations. This could be achieved through subsidized credit by lending agencies, with incentives from federal or provincial governments. The other way in which to improve the net benefits from afforestation is to devise some way of providing a cash flow to the land owner during the time of planting trees and their harvest. Such a scheme may involve paying a portion of the gross revenues to the land owner during this period. Sharing a portion of the establishment costs, either through subsidized seedlings (similar to the current practice under the farm shelterbelt program of the Prairie Farm Rehabilitation Administration), or through direct incentives by various levels of governments, may act as an incentive to land owners. Another possible solution would be for forest companies to enter into a contract with a group of land owners which would see a portion of the harvest revenues paid out early to defray costs of establishment.

In this analysis, we assumed that additional carbon sequestration has no value to society. However, in light of the Kyoto Protocol, and international efforts to curb emissions of greenhouse gases (where Canada has been an active participant), afforestation produces another service, i.e. reduced carbon dioxide emissions into the atmosphere. If land owners are paid for this service (either fully or even a fraction of it), economics of afforestation would probably swing in favor of converting poor (and possibly medium) quality pasture to tree planting.

Our analysis of current policies indicated that in Western Canada, governments have typically avoided regulation or incentives to encourage management of private land forests, and that some existing federal, provincial and municipal policies act as disincentives and deterrents to private land afforestation. For example, favorable tax treatment of agricultural lands often provides a disincentive to convert lands to forest where the higher assessed values of plantations could result in increased property tax.

In some provinces, governments have allocated Crown timber resources to the forest industry in volumes that are significantly greater than that required to supply existing mills. In addition, many forest companies pay relatively low royalties on wood harvested from Crown land. Low royalty rates and the over-allocation of timber rights has meant that forest industry has had access to a large supply of low-cost Crown timber, and consequently, forest industry has had little need for wood from private lands. Agreements with companies to limit their licensed harvest to, say only 80% of required wood volume would provide strong incentives for private land owners to supply the remaining 20%.

In Canada, the federal Income Tax Act often disadvantages woodlot owners relative to farmers engaged in traditional agriculture. Some woodlot owners are classed as farmers, while others are classed as logging operators, and different rules apply for each class of tax payer. Many woodlot owners are unable to claim legitimate deductions from income and face unfavorable treatment with respect to capital gains. In addition, federal rulings have effectively eliminated the deductibility of silvicultural expenses for some tax payers. Woodlot owners who are classed as logging operators are not eligible for the capital gains exemptions available to farmers. This creates a disincentive to establish a woodlot, and in some cases, may encourage liquidation of some woodlots to avoid capital gains taxation. In the U.S., policies affecting profitability in agriculture may play an important role in the land owners' decision to proceed with afforestation. For example, Plantinga (1996), using a case study of Wisconsin dairy farmers, found that lowering the milk support would reduce the incentives for farmers to keep marginal land in agricultural production.

The level of technical expertise related to plantation establishment and maintenance is low in areas where traditional agriculture has dominated land use practices. Governments may need to consider extension programs to help farmers identify and realize the opportunities available through the establishment of woodlots, and more research is required to develop varieties of hybrid poplars suitable for the northern prairies. Governments may also need to consider establishing programs to identify and inventory lands suitable for afforestation.

Other workers have shown that the opportunity cost of the land to be afforested depends upon the desired amount of carbon sequestration. Higher levels of sequestration require land to be taken from more productive uses, thereby increasing the cost per unit of carbon sequestered. In the U.S., McCarl and Callaway (1993) estimated the cost of sequestering a tonne of carbon at $US 17.38 if 39 million tonnes of carbon are sequestered, but increases to $US 25.95 per tonne if the target is 308 million tonnes. Freedman and Keith (1996) have also indicated the limited scope of planting on agricultural lands, which is also supported by findings of van Kooten (1995) for the boreal forest region. Furthermore, in the context of climate change, one time planting may not contribute much to help Canada meet its Kyoto Protocol commitment (van Kooten 2000)

In the U. S., land set-aside programs have provided areas for planting trees. Dudek and LeBlanc (1990) investigated the possibility of planting tree on such lands. The social cost of this type of activity was estimated to be between $US 6.64 and $US 10.67 per ton (equivalent to 2,000 lbs) of carbon dioxide removed from the atmosphere. This translates into a cost of $US 28 to $US 43 per tonne (equivalent to 1,000 kg) of carbon sequestered. Van Kooten et al. (1992) suggested that in Canada, although there are no land set-asides, there are lands that have not been improved (i.e., still in a native state). Planting trees on these lands would yield a sequestration of 3.4 Mt per year, at an initial cost of $1.3 to $2.2 billion.

Planting of hybrid poplar on marginal agricultural lands was investigated by van Kooten et al. (2000). The cost of sequestering a tonne of carbon was $18.82 if no discounting is used to $32.97 per tonne for a 4% rate of discount. Assuming a cost of $20 per tonne, it was estimated that 5.24 million ha of marginal agricultural land would be converted to forests. Using an optimal control theory model, van Kooten (2000) indicated that carbon sequestration benefits of planting hybrid poplar on marginal agricultural lands are somewhat optimistic in terms of meeting Canada's commitments under the Kyoto Protocol.

A study of 94 nations by Dixon et al. (1993) showed a large variability in carbon sequestration among nations as well as among various types of agro-forestry. Long-term storage of carbon in agroforest systems (such as wooded pastures) may be greatest in boreal ecosystems because of longer rotations (Krankina and Dixon, 1992). For Canada, Dixon et al. (1993) estimated a cost of $11.5 per tonne of carbon sequestered through forest systems.

These studies of afforestation on marginal agricultural lands suggest that several factors need to be considered besides the dynamics of carbon sequestration process. Agricultural policies, discount rates, and efficacy of afforestation relative to other means of reducing greenhouse gas emissions, are important. In addition, monocultures (such as hybrid poplar) may not be in the best interest of either farmers or society. If commercial benefits are the prime goal of the conversion, demand for timber and the capability of the land base are two important considerations.

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CARBON SEQUESTRATION IN PINE STANDS OF MIDDLE ZAVOLGIE OF RUSSIA

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ABSTRACT

The aim of this study was to estimate the carbon sequestration for pine ecosystems at stand and regional levels in Middle Zavolgie of Russia. Our approach estimated the remaining amount of carbon in vegetation, litter, and soil organic matter (SOM) after the carbon returned back into the atmosphere (respiration and decay processes) had been subtracted from the gross production. Sample plots were established in normal (full stocked) and modal (managed) pine stands of medium to high productivity within a relatively homogeneous area. The total sequestered carbon during a 120 years period for the I site modal pine stands was 160 Mg C/ha, which consists of 107 Mg C/ha in vegetation storage, 28 Mg C/ha in litter storage and 28 Mg C/ha in SOM storage. The total regional carbon store in pine forests in Middle Zavolgie was 128.4 Tg C. The largest proportion of the total carbon was in the Kirovskaya oblast (70 TgC), while Tatarstan had only 5.2 Tg C. Pine stands of the Nizegorodskaya oblast and Republic Mari El had 37.4 Tg C and 15.8 Tg C of carbon stores, respectively.

Keywords: Scots pine; carbon sequestration; global warming; forest inventory

INTRODUCTION

It is important to study carbon (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).

Expected warming is anticipated 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 are a large carbon reservoir and are located in latitudes predicted to undergo the greatest warming, they may play an essential role in future carbon fluxes (Dixon et *al*., 1994; Post et *al*., 1997).

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 characterised by a slow biological cycle. The amount of annual fall of plant residues in such ecosystems typically exceeds the annual norm of their decomposition. Therefore, the boreal forests accumulate carbon in the wood biomass, litter, coarse woody debris (woody detritus), soil organic matter and peat.

Russia has approximately 30% of the world's boreal forests (FAO, 1999), and together with Canada and USA is known to be one of the largest carbon reservoirs on Earth. Therefore, estimation of the carbon budget of the forest sector of Russia is very urgent for international strategies aimed at mitigation of the consequences of predicted global climate change.

The latest publications and research show that in spite of the fact that the Russian forest sector has had tremendous changes recently, it still has great potential for mitigating the consequences of the global warming. During the period from 1700 to 1910 the European part of the Former Soviet Union lost 63 ± 3 million ha of forests, mainly by transfer to agricultural land use (Shvidenko and Nilsson, 1998). From 1961 to 1993 forested areas increased by 68 million ha (9.8%), mainly in forests under state forest management. During the same period the total growing stock of all forests increased by 3.2 billion m^3, although growing stock of forests under state forest management decreased by 1.1 billion m^3 (Shvidenko and Nilsson, 1998). The total carbon stock in the forest biomass of the Russian Federation was estimated as 38.6 Pg C (Isaev et *al*., 1995) with sequestration of 262 Tg C/year. According to the estimation of the Sukachev Institute of Forest (Alexeyev and Birdsey, 1998) the weighted average carbon density in Russian forests is 153.2 Mg/ha. Most of this carbon is in forest soils, which accounts for 46% of the carbon in the entire area and 62.3% in the stocked area.

Recent forest policy of the Russian

Federation is oriented towards giving more authority to the regional forest committees and ministers of different republics and oblasts (Federal forest service of Russia, 1998). That, in turn, allows development of regional approaches and strategies for the implementation of international agreements on global warming mitigation. Therefore, regional approaches for estimation of carbon budgets of forest ecosystems are an important part in the investigation of Russian forests.

The aim of this study was to examine C sequestration in pure pine stands in the Middle Zavolgie region of Russia, under current climatic conditions, at stand and regional levels. Ecosystem components that were considered in the assessment include total tree biomass, understory (undergrowth, shrubs, and forest floor), litter, and soil organic matter (SOM). The assessment was based on total carbon sequestration, i.e. the amount of carbon left in vegetation, litter, and soil organic matter when the flows back into the atmosphere (respiration and decay processes) have been subtracted from the gross production.

STUDY AREA

The Middle Zavolgie (Figure 1) is located in the eastern part of the Russian plain where it occupies 287,000 km^2 between 55-60^{0}N and 44-52^{0}E. The administrative units in this territory are the republics of Mari El and Tatarstan and the Nizegorodskaya and Kirovskaya oblasts (Figure 2, Table 1). Middle Zavolgie can be subdivided into three ecological zones: southern taiga, mixed forest and forest-steppe. The Volga River serves as the natural physiographic border to the west. The territory north to the line Volga - Kazan - Right Bank of Kama up to eastern Tatarstan border belongs to the mixed forest zone. Forest-steppe zone is located to the south of this line. The northern part of the Kirovskaya oblast belongs to the taiga zone.

Forest cover of the territory varies widely (Table 1). On average forests cover 44% of the region's land area. The maximum forest cover is in Kirovskaya oblast, followed by the republic of Mari El with 53% of forest cover (FFSR, 1995). The Nizegorodskaya oblast has 46% forest cover and the Tatarstan has the lowest proportion (16%) of forest area in the region.

On the territory of the Middle Zavolgie the dominant species are pine (*Pinus sylvestris* L.), birch (*Betula pendula* Roth. and *Betula pubescens* Ehrh.), spruce (*Picea abies* Karst. and *Picea obovata* Ledeb.), and aspen (*Populus tremula* L.). Oak (*Quercus robur* L.) is not common. Pine forest stands occupy 32% of the total forest area in the region (3.2 million ha) and form the main landscape feature of the forest ecosystem. Pine is especially common in Mari El and Nizegorodskaya oblast (Table 1).

Most of the territory investigated is occupied by young pine stands which comprise 47% (1.5 million ha) of all pine forests of Middle Zavolgie (Table 2). Middle-aged and immature stands occupy 23% and 10%, respectively. The largest territory of young pine stands (58%) is in Nizegorodskaya oblast and republic Tatarstan. Middle-aged and immature stands are spread uniformly across Middle Zavolgie. Kirovskaya oblast has the largest area of mature and overmature pine stands - 454,000 ha (33% of all pine stands area).

This distribution of pine stand age groups is the result of unregulated harvesting of mature forests during the second half of the 19th century. Especially large clearcutting took place in Middle Zavolgie during World War II and after (Kurbanov and Sokolov, 1996).

Currently, when carrying out forest assessment work in the region of the Middle Zavolgie, it is common to use normal pine stands defined as natural, fully stocked and unmanaged forest ecosystems (Forest encyclopaedia, 1986, p. 19, 2 volume). This kind of forest stand occurs infrequently in nature and are used to derive yield and growth tables (Table 3). We are using normal stands in our research in order to facilitate comparison with other pine stands of Middle Zavolgie.

At present the main forest ecosystems in the region are *modal* stands, which are mostly managed, disturbed and have lower stand density and stocking than unmanaged ecosystems. Taking into account all kinds of forests on the investigated territory, both normal and modal pine forest ecosystems were used in our calculations.

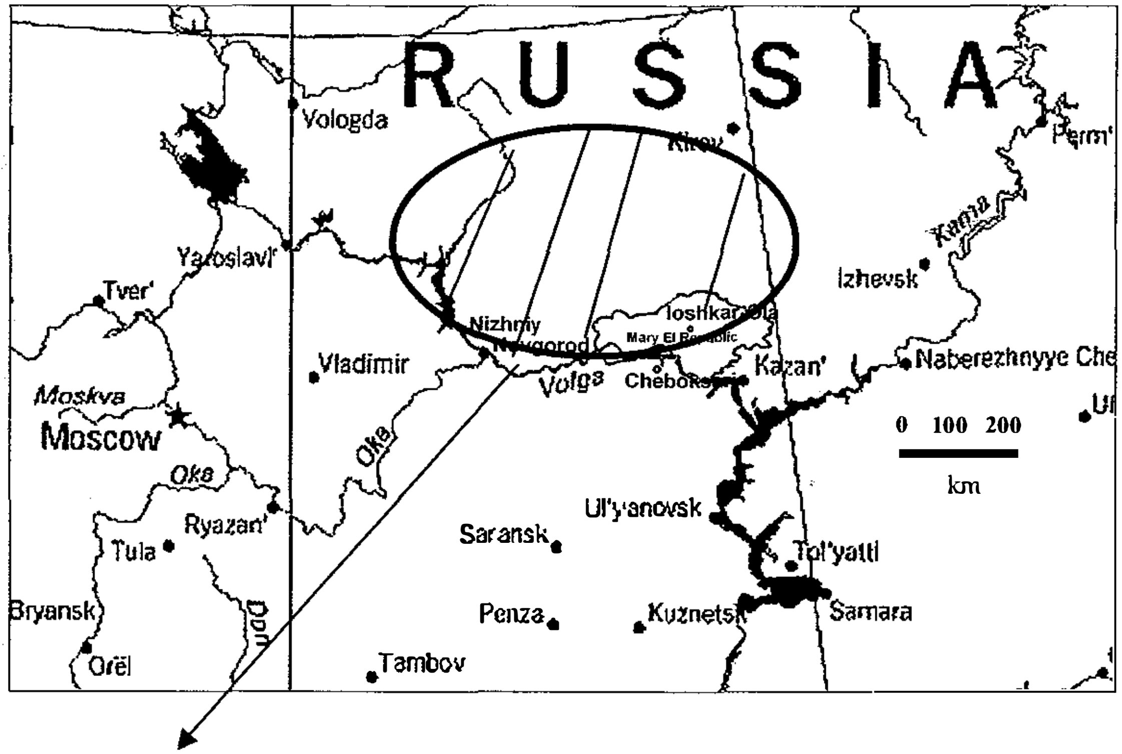

Figure 1. Middle Zavolgie on the map of Russia.

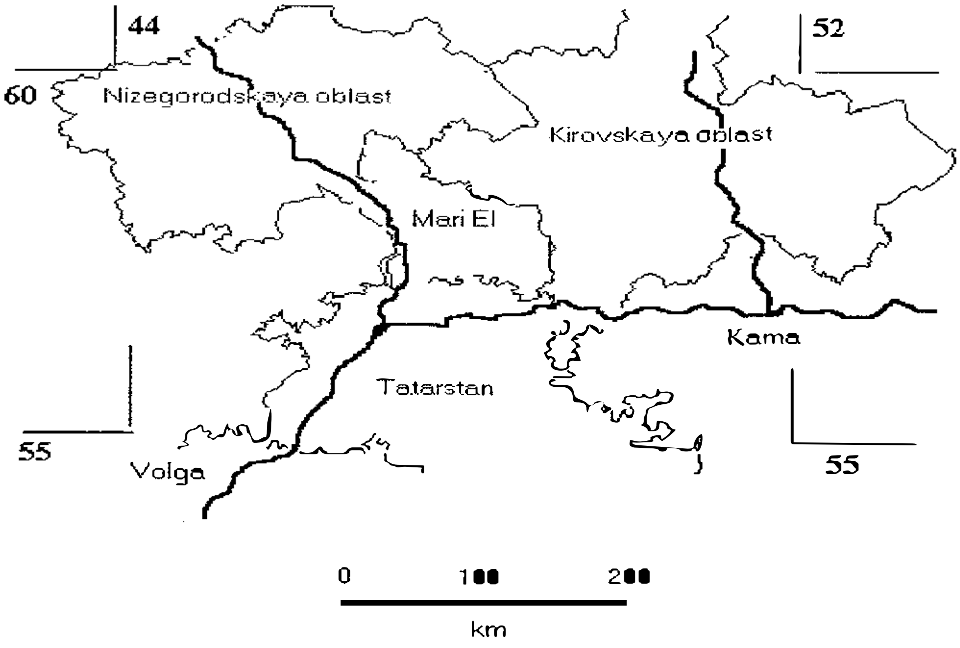

Figure 2. Administrative areas in the Middle Zavolgie.

Table 1. Characteristics of the forest cover in the republics and oblasts of Middle Zavolgie region (FFSR, 1995)

Parameter, units	Mari-El	Kirovskaya	Nizegorodskaya	Tatarstan
Area, thousand km^2	23.2	120.8	74.8	68
Closed canopy forests, % of total area	53	62	46	16
Area dominated by major species, % of total area				
Pinus sylvestris	20	11	16	3
Picea abies	5	13	3	1
Betula pendula	15	16	13	2
Populus tremula	3	5	4	3
Other species	1	1	1	5

Table 2. Distribution of pine stands by the age groups* in Middle Zavolgie, thousand ha (FFSR, 1995)

Republic, oblast	Young stands, 0-40 years	Middle-aged stands, 40-60 years	Immature stands, 60-80 years	Mature and overmature stands, 80 and >years	Total
Republic Mari El	226	127	45	54	452
Kirovskaya oblast	495	303	124	454	1376
Nizegorodskaya oblast	699	277	133	96	1205
Republic Tatarstan	94	38	18	13	163
Total	1514	745	320	617	3196

*Common age intervals for the age groups of pine forests in Russia.

Table 3. Distribution of pine stands in Middle Zavolgie region by the site classes (bonitet), thousand ha (FFSR, 1995)

Site classes	Normal stands	Modal stands
II and higher	610	1682
III	-	479
IV	-	291
V	-	102
VA-VB	-	32
Total	610	2586

METHODS

Sampling

Fieldwork was done in July and August 1990-1993 and 1995-1999. Sample plots with a minimum of 200 trees were located in the middle-aged, maturing and mature pine stands and a minimum of 400 trees in the young pine stands, were set up at a distance not less than 30 m from forest block boundary, trails, roads, and forest edges. Sample plots (SP) were established in the normal and modal pine stands of medium to high productivity within a relatively homogeneous area. Most pine stands were selected to represent the full range of stand age groups, tree species composition, and structure typically occurring in these ecosystems. The fieldwork was carried out conforming fully to Russian forest inventory guidelines (Otraslevoi standart, 1983). Data gathered over different years on sample plots established by the department of forest inventory and measurements of the Mari state technical university were also used in this study (Sokolov and Kurbanov, 1997; Sokolov and Kurbanov, 1999; Kurbanov and Sokolov, 2000; Kurbanov and Krankina, 2000).

Based on the forest inventory data, we determined the distribution of areas dominated by *Pinus sylvestris* according to the bonitet (site classes) and basal area of the forest stands. Fifteen to 20 trees were selected per sample area for felling. The selection criteria for the sample trees were that the trees should appear healthy, with one main stem and their size distribution should reflect the size distribution of the measured plots. Destructive samples were taken from trees beyond the plot boundaries in proportion to their stem diameter. 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 classified as large branches (diameter ≥ 0.8 cm) or small branches (diameter < 0.8 cm). Branches without needles were immediately weighed. A simple relationship between the weight of branches with needles and those without needles was used to predict needle mass for branches without needles. Three middle-sized model branches were taken to determine the moisture content of the branches and needles which was applied to other parts of the crown.

The sample plots established in pure pinetum cowberry were characterized by density from 0.5 to 1.0. Pine, aspen, spruce and birch undergrowth were measured following the same procedures used for the dominant tree species.

To investigate pine forest stands, a total of 96 sample plots were established and 1426 destructively sampled trees were cut on the territory of republic Mari El and Tatarstan, Nizegorodskaya and Kirovskaya oblast. The sample plots took into account all age classes of the forest stand (Table 4).

For our calculations we pooled all data from sample plots into two bonitet classes of normal and modal pine stands. In the Russian classification system potential productivity of a forest ecosystem is characterised by site class (bonitet) corresponding to Orlov's scale (Tretiakov, 1965). There are seven site index classes (Bonitet classes - Ia, I, II, III, IV, V, Va) in this system which are defined by various combinations of average stand age and stand height. The highest class (Ia, I) characterizes the best-known conditions for growth of the forest stand andthe lowest class (V, Va) the poorest conditions. Bonitet class of the stand is not a characteristic of wood quality. For instance, a rotted aspen stand might have I bonitet class.

Table 4. Main characteristics of the sample plots.

Age, years	Number of plots	Number of sample trees	Mean diameter, cm	Mean height, m	Basal area, m^2	Stem volume, 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

Regression Analyses

Gross carbon production statistics were calculated from the regression equations of interdependence between the absolutely dry mass of different tree components and the dendrometry parameters. We concluded that the dependence of the absolutely dry crown mass of the trees with the crown diameter, diameter at breast height and crown length should be used in studying the crown of pine trees for the Middle Zavolgie Region. The amount of foliage and branches of trees have often been predicted through allometric equations with breast height diameter as the independent variable (Shugart, 1984; Landsberg, 1986; Makela and Vanninen P, 1998). In this study we also used allometric relationships between biomass components and diameter at breast height for the modal and normal pine stands.

Analysis of the suitability of different regression equations was done for each component of the crown mass (Kurbanov, 2000). Selection of equations was based on the amount of experimental data available, the statistical criteria of equations, and the distribution of residuals. All further analysis and plot fittings were done for the time frame between 20 and 120 years since the average ages of sample plots established in Middle Zavolgie were between 20 and 120 years.

Calculation of Carbon Sequestration

Sequestration of C in forest ecosystems has been assessed using a carbon bookkeeping system that accounts for C flows in and out of forest ecosystem components (Karjalainen, 1996). The carbon stocks indicate the momentary amount of C in any one component of the system (Table 5). Annual flows (MgC/ha/a) are the rate variables. Cumulative flows (Mg C/ha) describe the accumulation of carbon in the flows over longer periods. For instance, cumulative litterfall, cumulative gross productions, cumulative net production and so on.

We calculated the carbon budget for the young (0 - 40 years), middle-aged (40-60 years), immature (60-80 years), mature and overmature (80 and > years) pine stands of the Middle Zavolgie. Then, based on these budgets and the pine forest area in the respective age groups (Table 2), the total regional carbon store in the pine stands of the Middle Zavolgie was calculated.

RESULTS

Figure 3 shows an example of annual dynamics of gross carbon production in I site modal pine stands in Middle Zavolgie as a result of regression analyses (Kurbanov, 2000).

The stand level allocation of carbon at the end of a 120-year simulation in different tree components was similar in both modal and normal pine stands. Stemwood carbon accounted for 55-65% of the carbon stock in pine trees while coarse and fine roots contained 12-22%, branches 8-12% and needles 4-8%. The proportion of understory fraction of the C stock was only 2-6%. These data for the normal and modal pine stands were used in further calculations of the cumulative carbon store and carbon fluxes in the pine stands of Middle Zavolgie.

The study shows that total carbon sequestration varies during pine stand growth, with the maximum rate occurring from 40-70 years (Table 6). Average gross annual carbon flux is 1.9 Mg C/ha/a for modal pine stands and 2.5 Mg C/ha/a for normal. Average annual net ecosystem flux over the 40-70 years is 1.1 Mg C/ha/a in the modal pine stands and 1.5 Mg C/ha/a in the normal stands.

The harvest age for the pine stands in Middle Zavolgie is 80-100 years. By this time pine ecosystems slow down their rate of annual C sequestration (Table 6) and it may be assumed that it would be beneficial to replace mature pine forests with young fast-growing plantations. However, evidence exists that much more carbon is stored in old, undisturbed forests than in any of the plantations considered so far (Cannell and Milne, 1995; Harmon et. *al*, 1990). Taking this into account, carbon sequestration capacity should be assessed over long term intervals of forest development, using criteria such as forest ecosystem net sequestration and carbon store at the end of a certain time period.

Over the entire 120-year period, differences in emission profiles were noted for different biomass components. As a rule, vegetation respiration varied from 35-40%, litter decay 23-30% and SOM decay 32-45%, of the total 120 years of C flow to the atmosphere.

Accumulation of carbon in different ecosystem components of the modal pine

Table 5. Scheme for calculation of annual and cumulative C flows (Mg C/ha/a and Mg C/ha) in the pine ecosystems of Middle Zavolgie (Karjalainen, 1996).

1. CARBON FLOWS OF VEGETATION

GROSS PRODUCTION: GROWTH (TREES, GROUND VEGETATION) + RESPIRATION
- RESPIRATION: GROWTH AND MAINTENANCE RESPIRATION
= NET PRODUCTION

NET PRODUCTION
- LITTERFALL
= NET PRODUCTION OF VEGETATION

2.CARBON FLOWS OF LITTER

FORMATION OF LITTER
- EMISSIONS OF DECAYING LITTER INTO THE ATMOSPHERE
- CHANGE TO SOM (SOIL ORGANIC MATTER)
= NET PRODUCTION OF LITTER

3.CARBON FLOWS OF SOIL ORGANIC MATTER

FORMATION OF SOM (CHANGE OF LITTER TO SOM)
- EMISSIONS OF DECAYING SOM INTO THE ATMOSPHERE
= NET PRODUCTION OF SOM

*4.CARBON BALANCE OF FOREST ECOSYSTEM**

C FLOW INTO FOREST ECOSYSTEM (GROSS PRODUCTION)
- C FLOW OUT OF FOREST ECOSYSTEM (RESPIRATION, EMISSIONS OF DECAYING LITTER AND SOM INTO THE ATMOSPHERE)
= NET FOREST ECOSYSTEM PRODUCTION (NEP)

*Not used in the further calculations

stands varied throughout the 120-year simulation. At 40 years modal pine stands' allocation of C was 73-75% in vegetation, 15-17% in litter and 12-8% in SOM. At the end of the simulation the C allocation was 65-67% in vegetation, 17-19% in litter and 14-18% in SOM. In normal pine stands distribution of carbon during the 120-year simulation was even with 75% in vegetation, 13% in litter and 12% in SOM.

The total store of carbon during a 120 year period for I site modal pine stands was 160 Mg C/ha, which consists of 107 Mg C/ha in the vegetation store, 28 Mg C/ha in the litter store and 25 Mg C/ha in the SOM store (Table 7). The largest total for C store of 254 Mg C/ha was for the I site normal pine stand.

Amongst the regions studied the total C store in pine ecosystems were higher in the Kirovskaya oblast than in other republics and oblasts. The proportion of different stocks of carbon varied with the age of the pine stands of the Middle Zavolgie. Young, immature and mature stands contained 19, 20 and 48% of the total carbon stock in pine stands in Middle Zavolgie, respectively (Figure 4). The percentage of total carbon store in pine stands of Middle Zavolgie found in immature stands was low, but still significant.

DISCUSSION AND CONCLUSIONS

This study has shown that pine ecosystems of Middle Zavolgie make a positive and significant contribution to carbon sequestration in Russia. The pine forests of Middle Zavolgie of Russia store large amounts

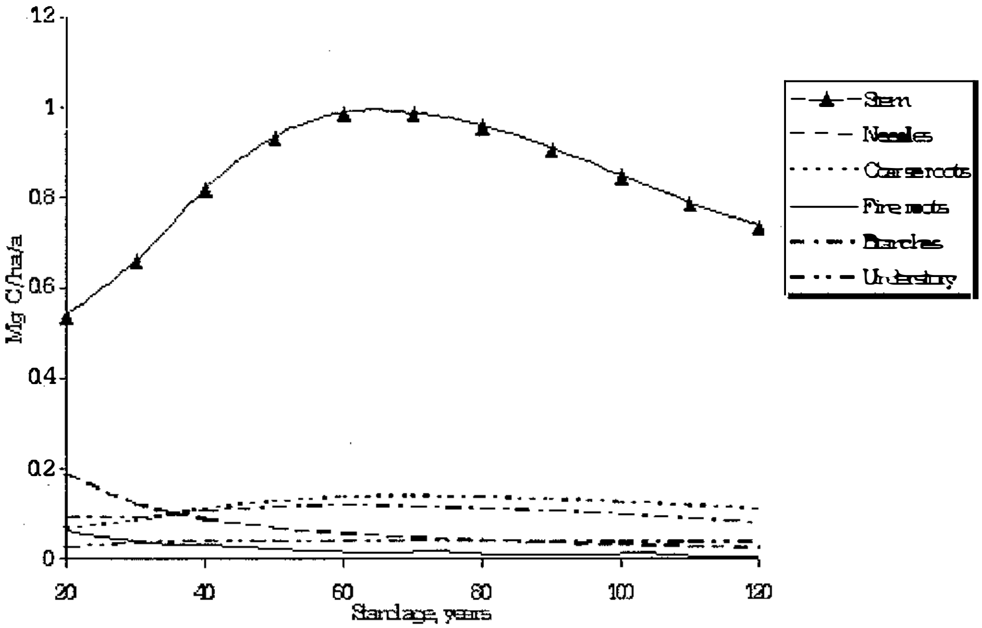

Figure 3. Dynamics of gross carbon production in I site modal pine stands in Middle Zavolgie.

Table 6. Gross and net ecosystem production over 120 years for the pine stands, Mg C/ha/a.

Stand age, Years	Modal I site	Modal II site	Normal I site	Normal II site
		Gross ecosystem production		
20	1.8	1.5	2.2	2.0
30	1.9	1.7	3.0	2.5
40	2.4	2.2	3.1	2.6
50	2.6	2.4	3.0	2.7
60	2.4	2.3	2.8	2.6
70	2.2	2.1	2.7	2.5
80	1.9	1.8	2.4	2.5
90	1.7	1.6	2.4	2.2
100	1.5	1.5	2.1	2.1
110	1.4	1.4	2.0	2.0
120	1.4	1.4	1.9	1.8
		Net ecosystem production		
20	1.0	0.9	1.5	1.5
30	1.1	1.0	1.8	1.5
40	1.4	1.3	1.9	1.6
50	1.5	1.4	1.8	1.6
60	1.4	1.3	1.7	1.6
70	1.3	1.2	1.6	1.5
80	1.2	1.1	1.5	1.5
90	1.0	1.0	1.4	1.3
100	0.9	0.8	1.3	1.3
110	0.8	0.8	1.2	1.2
120	0.8	0.8	1.1	1.1

Table 7. Carbon store for the normal and modal pine stands in Middle Zavolgie. Cumulative C flows (Mg C/ha) are shown for 120 years and stores after 120 years simulation.

	Cumulative Stock	Modal I site	Modal II site	Normal I site	Normal II site
	Gross Production	868	789	1130	995
-	Respiration	295	268	384	338
=	Net production	573	521	746	656
-	Litterfall	466	424	555	489
=	Vegetation Stock	107	97	191	167
	Formation of litter	466	424	555	489
-	into the atmosphere	186	169	222	196
-	Change to SOM	252	230	300	264
=	Litter Stock	28	25	33	29
	Formation of SOM	252	230	300	264
-	into the atmosphere	227	207	270	238
=	SOM Stock	25	23	30	26
Total Stock		160	145	254	222

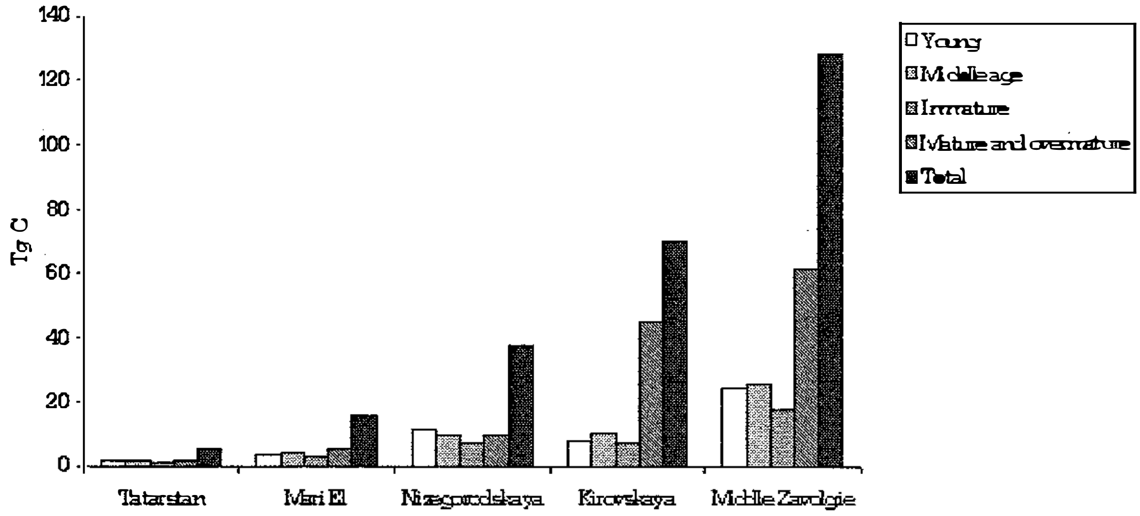

Figure 4. Carbon stores in pine stands by age groups in republics and oblasts of Middle Zavolgie.

of carbon in the vegetation (82.4 Tg C), litter (24.1 Tg C) and SOM (21.8 Tg C) with a considerable storage rate over 120 years. The total carbon store of pine forests in Middle Zavolgie was estimated at 128.4 Tg C. Predominance of young and middle age stands in pine ecosystems in the region (Table 2) suggests that current sequestration rates are high.

Sequestration of carbon in pine stands of Middle Zavolgie has been assessed using a C bookkeeping system (Karjalainen, 1996). The results of this study were compared with various other studies. They are generally within the range of those reported for other forest regions and countries. In our calculations the average NEP for 120-year modal and normal pine stands was 1.3 and 1.6 Mg C/ha/a. Net ecosystem production for conifer stands in the Russian Federation reaches 1.2 Mg C/ha/a (Isaev *et al.*, 1995). Nabuurs and Mohren (1995) reported the net annual carbon stock flux for slow-growing long rotation forests as 1 Mg C/ha/a. In Britain net annual carbon flux for the Scots pine over the first rotation including trees, products, litter and soil was estimated at a rate of 2.7 Mg C/ha/a (Cannell and Milne, 1995a). Diversity in results can be explained by differences between forest types, climate, site productivity and level of forest management.

Carbon sequestration is higher in normal pine stands than in modal ones if evaluated to the end of the investigated period (up to 120 years), that confirms high potential productivity of the fully stocked normal stands. Nevertheless, more attention should be paid to the modal pine stands since they are more prevalent in the Middle Zavolgie region.

This study of carbon sequestration of pine stands forms the basis for a method that can be applied to further regional assessments. However, several considerations should be taken into account. Our results are only for pure evenaged stands of I and II site classes. Representation of the proportion of mixed pine stands in Middle Zavolgie might be essential. Assessment of mixed pine stands with other species for all site classes would require additional research in these types of forest ecosystems or require the use of a correction factor for the mixed species. Forest fires should be considered in more detail by taking into account climatic conditions and frequency of fires on the investigated territory.

Estimates at stand and regional levels might be further improved by taking into consideration biomass removed from the forest as wood-based products (Karjalainen, 1996; Cannell and Dewar, 1995) and disturbance from insect outbreaks and diseases (Isaev *et al.*, 1995).

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THE ROLE OF THE CONIFEROUS FORESTS IN THE CARBON BUDGET OF UKRAINE

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ABSTRACT

Coniferous forests in the Ukrainian Polissian and Carpathian regions are situated at the southern edge of the boreal forest zone of Eastern Europe. Forests are dominated by pine (*Pinus silvestris* L.) and spruce (*Picea abies* (L.) Karst.). Since forest cover in Ukraine is low (average of 15.6%), the coniferous forests in these regions represent a major resource. According to the State Forest Accounts (01.01.1996), they occupy over 42% of the total forest area, and 52% of the overall timber stock. Apart from the highly valued resources of these coniferous forests, their ecological significance is extremely high and is characterized by high general forest productivity, volumes and the magnitude of accumulation of biomass components and stored carbon. Our experiments were aimed at investigating these parameters.

Overall state phytomass resources and accumulated carbon were calculated using an original method, modern mathematics and computerized methods of data processing. A large data set was collected from more than 300 temporary sample plots and stored in an electronic database (State Forest Inventory).

The results obtained showed that the overall conifer forest phytomass of Ukraine equals 538 Mega tonnes (Mt) and contains over 266 Mt of carbon (43% of its total volume). Average carbon density of the forested area of the boreal forests is 6.7 kg m^{-2}. Estimates for this parameter showed significant fluctuations since it depends on the tree species (physical and mechanical properties of the timber), forest productivity, forest age and other factors. For instance, the estimated values for carbon density of coniferous stands in the Zhytomyr region was 6.3 kg m^{-2}, and for spruce forests in the Transcarpathians it was 10.9 kg m^{-2}.

Trends in phytomass dynamics in pine and spruce stands, and the magnitude of their annual carbon stocks were also investigated.

GENERAL INFORMATION ABOUT UKRAINE FORESTS

Ukraine is a new state located in Eastern European. The area of the country is 608 thousand km^2 and it's population is about 50 million. Ukraine has 10.8 million ha of forests with 1736 million m^3 of growing stock. The average forest cover rate is low at about 15.6%.

The most forested regions in Ukraine are Polissia and Carpathia. Polissia is a northern region where the average percentage of forest land cover is 34.7%. Since Polissia is located in the southern extent of the boreal forests, it is mostly occupied by coniferous forests. Forests are dominated by pine (*Pinus sylvestris* L.) and their plantations occupy an area of 3.1 million ha with 377 million m^3 of total growing stock. The second forest region in Ukraine is Carpathia. This is a mountainous region with well defined mountain forest zones. The Middle and the Top zonal belts of these mountains are occupied by coniferous forests. Spruce (*Picea abies* (L.) Karst.) forest types dominate the forests in Carpathia. The spruce stands occupy an area 700 thousand ha, and their total growing stock is 205 million m^3. Thus, coniferous forests in Carpathia and Polissia form 37.0% of the area and 52.9% of the growing stock of all Ukrainian forests.

METHODS

The estimation of phytomass and carbon storage in coniferous forests of Ukraine was calculated using special methods, which facilitated analysis of a complex collection of experimental data from temporary sample plots (Lakida, 1996). Field and laboratory

Table 1. Sampling design for the forest phytomass database and carbon estimation.

		Phytomass Fraction (number of plots sampled) with measurements of				
Forest species	Total number	Stem	Branches	Needle (leaves)	Roots	Under canopy vegetation
Pine stands	205	195	174	195	66	20
Spruce stands	54	44	43	44	15	14
Sum	259	239	217	239	81	34

research provided biometric data and data for biological parameters. Data from the literature were used to develop some indices for pine stands in Polissia (Mjakushko, 1978) and for spruce stands in Carpathia (Pitikin, 1987). The data were processed on personal computers using the special programs (PERTA, ZRIZ, PLOT) and stored in a database. General information about this database is presented in the Table 1.

Phytomass storage in Polissian and Carpathian forests was evaluated using the conversion coefficients ($R_{v(fr)}$) (Lakida at al., 1997a; Lakida, 1996) which is equal to:

$$R_{V(fr)} = \frac{M_{fr}}{V_{st}} \quad [1]$$

where:

$R_{V(fr)}$ is the conversion coefficient of a tree fraction;

M_{fr} is the dry mass of a phytomass fraction (tonnes);

V_{st} is the stem growing stock under bark (m^3).

This method gives us certain benefits. The only information needed to calculate the phytomass are the growing stock under bark and the conversion coefficients. As long as growing stock data is available phytomass can be calculated at different scales (stand, plantations or region).

Using data collected in coniferous forests of Ukraine, dependencies between biometrics parameters of stands and phytomass fractions conversion coefficients were established. For approximation of dependencies three types of equations was chosen:

$$R_{V(fr)} = a_0 \cdot A^{a_1} \cdot B^{a_2} \cdot \exp(a_3 \cdot A) \quad [2]$$

$$R_{V(fr)} = a_0 \cdot A^{a_1} \cdot B^{a_2} \quad [3]$$

$$R_{V(fr)} = a_0 \cdot A^{a_1} \quad [4]$$

where:

$R_{V(fr)}$ is the conversion coefficient of a tree fraction,

A is the age of stand,

B is the site index of a stand and,

a_0, a_1, a_2, a_3 are coefficients of the regression equations.

From these equations we selected the one which best described the dependence of $R_{V(fr)}$ on biometric indices of stands. Multiple regression was used to calculate the equation coefficients (a_0, a_1, a_2, a_3). The best model was selected based on the coefficient of determination (R^2) and *F-criterion* of Fisher.

Total phytomass and phytomass carbon of fractions in the Polissia and Carpathia coniferous forests were estimated using data from the State forest account as of January 1,1988 (Minleshoz UkrSSR, 1989) and January 1,1996 (Ukrderglisproect, 1998). Using the conversion coefficients and statistical analyses the stand phytomass for each species within the climatic regions was estimated and then extrapolated to the entire country.

The carbon storage in coniferous forests of Ukraine was determined as a function of phytomass type, its growing stock and the conversion coefficients (Matthews, 1993). The proportion of carbon per unit of dry organic mass of living plants was assumed to be 0.5 for stems, branches, bark, trunks and roots and 0.45 for needles and undercanopy vegetation (living ground vegetation, young seedlings, undergrowth).

RESULTS

Results of phytomass and carbon estimates are presented in the Table 2.

The results indicate that in 1996 for all Ukrainian forests the total stored phytomass was 1.2 Gt and it contained 0.5 Gt of carbon. Coniferous forests contained 0.5 Gt of phytomass and 0.3 Gt. of carbon.

The average deposited carbon density per m^2 in coniferous forests is slightly higher than the average for all forests. This is explained by higher productivity of coniferous forests compared with deciduous forest under the climatic conditions of Ukraine. A comparison of average deposited carbon density divided by forest region, in Ukrainian coniferous forests, is presented in the Figure 1.

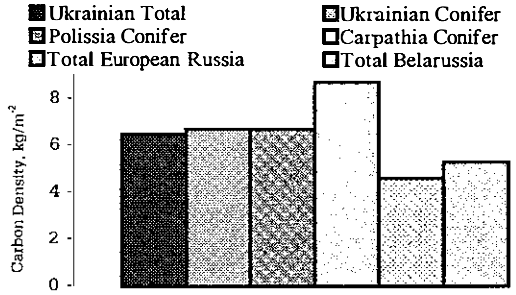

Figure 1. Deposited carbon density.

The average deposited carbon density in coniferous forests of Ukraine (Ukraine Total) is greater than the average for Belorus forests, the European part of Russia (Figure 1), and neighboring regions (Lakida at al., 1997b). The difference between Polissia and Carpathia forests can be explained by differences in forest management in these regions.

Total phytomass and carbon storage in coniferous stands is greater in 1996 than 1988. For eight years the balance (growth minus mortality) of carbon increased by 0.009 Gt in Carpathian coniferous forests, by 0.025 Gt in Polissian coniferous forests and by 0.010 Gt in all other Ukrainian coniferous forests. The observed increase in carbon storage reflects not only the natural growth of pine and spruce stands, but also differences between the two data sets. In 1996 field-protecting forests, forests along roads and railways etc (in total more than 400 000 ha) were included in the State forest account but were not included in the 1988 inventory. In addition, the 1996 forest account data reflect changes in stand growth and/or mortality that occurred between inventory moment and 1996 (forest inventories are provided in different periods, e.g. forest inventory for Zitomir region was in 1994, for Sumi – in 1995). Then one can be sure, that estimation of carbon storage in Polissian and Carpathian coniferous forests based on the 1996 data is more accurate, due to a more complete forest data inventory.

Carbon distribution amongst tree factions is of interest because these data are closely connected to estimation of litter in coniferous forests of Ukraine. Litter, when decomposing, is a source of carbon. To study this flux of carbon it is necessary to estimate the general carbon balance in Ukrainian forests. Distribution of general amounts of carbon in coniferous forests by fractions is shown in the Figure 2.

Wood and bark of stems and roots are the most stable fractions and represent 67.7% and

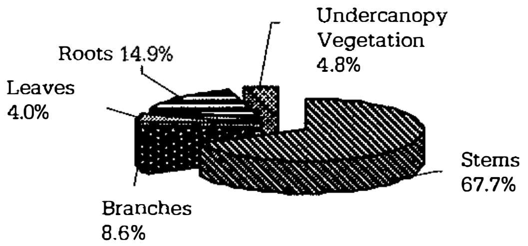

Figure 2. Percent of the carbon content in the phytomass fractions of conifer forests (data from January 1,1996).

Table 2. Phytomass and carbon storage in Ukrainian forests.

	All Forests			Coniferous Forests		
Year	Total phytomass (Gt)	Total carbon (Gt)	Total deposited carbon density (kg m^{-2})	Conifer phytomass (Gt)	Conifer carbon (Gt)	Conifer deposited carbon density (kg m^{-2})
1988	0.9	0.5	5.4	0.4	0.2	5.6
1996	1.2	0.6	6.5	0.5	0.3	6.7

14.9% respectively, of the total carbon storage in coniferous stands. The percent of total carbon in branches is 8.6%, needles 4.0% and under canopy vegetation 4.8%.

CONCLUSION

Coniferous forests are important to Ukraine. They constitute 52.9% of the total forest growing stock in Ukraine. Consequently, they play a crucial ecological and economic role. Use of the conversion coefficient method and multiple regression technique can give anaccurate evaluation of forest phytomass. From 1988 until 1996 the conifer forest phytomass storage increased by 0.1 Gt and carbon storage increased by 0.044 Gt. The observed increase from 1988 to 1996 is due to both forest growth and a change in the forest inventory technique. Most deposited carbon is located in the stems and roots of trees (82.6 %).

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POSSIBLE SYSTEMS FOR MEASURING AND REPORTING ON DEFORESTATION IN CANADA UNDER THE KYOTO PROTOCOL

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ABSTRACT

A national system for determining and reporting on areas of deforestation is needed to fulfill Canada's Kyoto Protocol reporting commitments. An enhanced National Forest Inventory (NFI) forms a reasonable national framework on which to build a deforestation reporting system. The NFI consists, at its core, of a grid system of 2x2 km plots on a 20 km spacing. The base design calls for forest parameters to be determined from aerial photo interpretation. A subset of plots are sampled on the ground. This core can be enhanced with data from other sources. One possible enhancement is the integration of the NFI plot system, medium resolution satellite remotely sensed data (e.g., Landsat TM), and existing land use records to improve measurements of deforestation in the context of the Kyoto Protocol. Important in such a system are what data are available and how to integrate the data.

Key issues related to the appropriateness of public land use records are: what records are available; from who; their content, coverage and reliability; are they spatially explicit; are they yearly; are they legislated, regulated or voluntary; and are there access restrictions. Questions related to the potential use of satellite remote sensing include: what types of deforestation can be detected, how accurately and at what minimum mapping unit; for what types of deforestation can you infer deforestation using only one image; how long a time interval do you need to prove deforestation or alternately disprove deforestation; what information can one infer regarding remaining carbon stocks. Issues related to integration into a system within the National Forest Inventory structure are: sampling system design (random, systematic, focused); scaling; sampling interval; what to do when different sources give different answers; and how area and location of deforestation can be related to forest type, biomass, and remaining carbon on site so the impact on the carbon budget can be determined. In this paper these factors were considered and viable integrated systems outlined.

INTRODUCTION

Recently, a need for data in response to Canada's commitment to reduce greenhouse gases has emerged. Under the Kyoto Protocol, Canada has agreed to reduce greenhouse gas emissions to six percent below the 1990 level by 2008. A significant effort will be required to meet this goal; carbon (C) emissions must be reduced and C sequestration must be increased. Canada's forests play a major role in this effort. Through increased afforestation, additional C can be sequestered in the new forests, and through decreased deforestation, C emissions can be reduced.

To document the 1990 baseline data and subsequent C changes, a national reporting and monitoring system is required. The Canadian Forest Service (CFS) has been cooperating with the provinces and territories in the development of a new plot-based National Forest Inventory (NFI). A new approach is needed because the current design cannot meet the new needs for data and information (e.g., Criteria and Indicators of Sustainable Forest Management). The NFI provides a suitable framework on which to build a forest carbon monitoring and reporting system but the capability of identifying changes in the area of land cover classes and land use classes must be improved, in particular deforestation and afforestation. This study focuses on deforestation but most of the procedures also apply to afforestation.

The basic NFI design includes complete interpretation (stratification and classification) of approximately 20,000 primary sample units, or one percent of Canada's land mass using mid-scale aerial photography. Primary sample units (photo plots) are 2 x 2 km in size and are located on a 20 x 20 km network. Estimates of land cover and other forest stand attributes are acquired. A ground-based sub-sample of approximately one in ten photo plots will be established for the estimation of species diversity, biomass, and other detailed data not available from aerial photography. Twenty-five core attributes will be derived from the photo interpreted and ground-based estimates. The initial measurement of photo and ground plots will provide an estimate of the current state of the resource. An estimate of change will be derived from repeated measurements of both photo and ground plots (all plots are permanent). Estimates will be reported for Canada as a whole and by ecozone. They will also be available for each province/territory.

This design provides a good framework for achieving the Kyoto deforestation objectives: it includes a network (grid) of photo plots from which area estimates are obtained, a sub-sample of ground plots for estimating other attributes, and re-measurements to estimate changes over time. However, the accuracy or reliability of the change estimates depends very much on the attribute in question.

Area of deforestation is, at the national level, a small quantity. For example, for the period 1986-91 Lemprière and Booth (1998) estimated that the annual area of deforestation in Canada was about 110,000 ha, which is only about 0.01% of the total forest area. The Lemprière and Booth estimate was based on data from the Canadian Council of Forest Ministers (1997) which indicated that 88,000-103,000 ha of forests was permanently converted each year to non-forest uses in Canada. An extensive sample such as the NFI, which does include deforestation as an attribute, covers only a small proportion of the population area (1%). Although the NFI will provide useful deforestation data, it cannot be expected to pick up and sample reliably such small and likely scattered areas. A useful way to get an estimate of deforestation is to use NFI as a framework, but it must be enhanced.

Enhancements to the NFI are required to increase the reliability of the estimates of small and scattered areas of deforestation.

Enhancements include:

1. improving the capability of determining changes in land cover,
2. improving the capability of identifying land use classes and their changes, in particular deforestation and,
3. including carbon stock and carbon change in the estimation of attributes.

This paper describes a design for a measurement and reporting system based on an enhanced NFI structure using remote sensing information to report on deforestation for the Kyoto Protocol. Details of the design options and the remote sensing studies are contained in Leckie et al. (2000). The following presents various design options based around a core design. Issues related to the designs are discussed. The paper concludes with a description of the next steps in the development of a national system for determining and reporting on areas of deforestation under the Kyoto Protocol.

THE CORE DESIGN

The core design should be simple and cost-effective. It should make use of the information available from the NFI, specifically the deforestation data available from the 2 x 2-km plots. This would avoid having two different estimates and should make the design more cost-effective.

A core design must consider the sources of information, the attributes of interest and any design constraints. The core design illustrated in Figure 1 is an integrated design with the following characteristics:

1. Initial stratification of the population into areas of high and low deforestation activity
2. The use of NFI and remotely sensed data to obtain an estimate of deforestation in high activity areas. The two estimates of deforestation are integrated using a ratio estimator.
3. The use of NFI photo plots to obtain an estimate of deforestation in low activity areas.

The population is that area of Canada that can be expected to grow trees. Deforestation is expected to occur mostly on the forest fringe (i.e., on the borders with agriculture, industrial

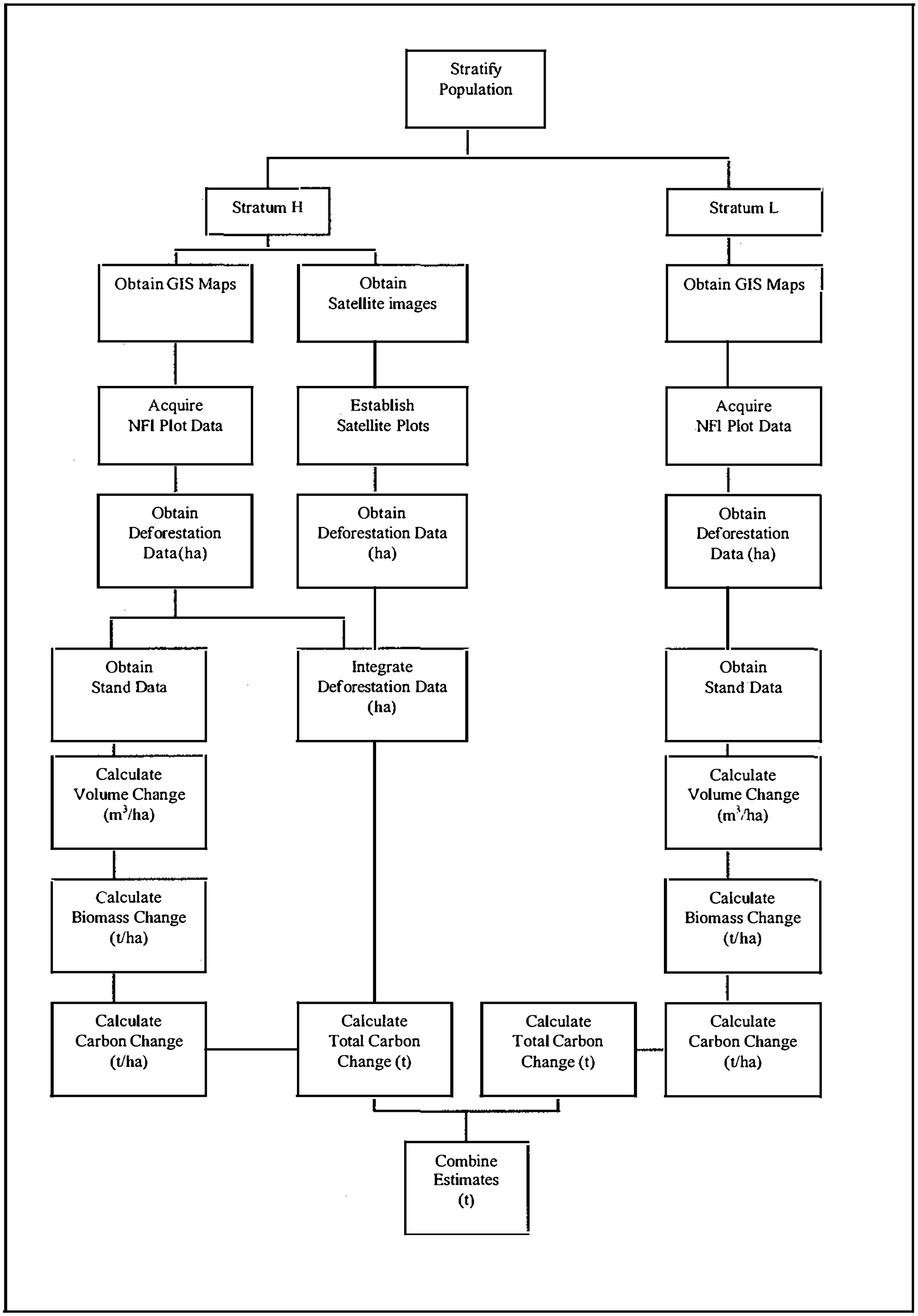

Figure 1. A schematic of the integrated stratified core design.

and urban areas). The population will be stratified into areas of high deforestation (Stratum H) and areas of low deforestation (Stratum L). Different sampling approaches and intensities will be used in the two strata – Stratum H will be sampled intensively, while Stratum L will be sampled extensively.

The specifics of the shape and size of Stratum H remains to be determined, but it is anticipated that there will be a number of areas in Stratum H, and that the area of Stratum H will be much smaller than Stratum L. NFI stratification (by Ecozone and by province/territory) will be retained.

Stratum H - Expected High Deforestation

Estimate deforestation area as follows:

1. Obtain suitable multi-date satellite imagery covering the area of Stratum H.
2. Establish NFI size plots on a 10 x 10-km grid and extract from the satellite imagery areas of any change that could possibly be deforestation. Using a sequential approach[1], label the change areas as deforestation. One quarter of these plots will overlap the NFI plots.
3. Obtain the NFI derived deforestation data for the NFI plots.
4. Use a double sampling estimator to obtain an integrated estimate of the deforestation area.

The 10 x 10-km spacing of the satellite based plots means that there are four times as many of these as of the NFI plots. This is therefore the larger sample. One-quarter of the satellite plots reside at the same location as the NFI plots. This comprises the smaller double sample, having deforestation estimates from both the satellite and NFI sources.

The derivation of an integrated estimate is accomplished through the double sampling estimation formulae. Several versions are available. An appropriate one is double sampling for ratio estimation, in which the average deforestation area is estimated by:

$$y_R = (y/x)\,x'$$

where:

y_R = average deforestation area;

y = average deforestation area from the NFI plots (small sample)

x = average deforestation area from the satellite plots (small sample)

x' = average deforestation area from the satellite plots (large sample).

Having identified the areas of deforestation, the amount of carbon sequestered on these areas before and after the deforestation (the carbon change) must be determined. Carbon is usually measured indirectly, from biomass (in general, one tonne of oven-dried biomass is equal to one-half tonne of carbon). Biomass is estimated from biomass equations, which tend to use the same independent variables as wood volume equations. Biomass may be derived from the volume equations; therefore carbon can be derived from the volume data available from forest inventories.

The average deforestation area (y_R) must be multiplied by the average carbon stock change for that area. Average carbon change is derived from the stand characteristics of the deforested areas on the NFI plots. These in turn are based on forest inventory data. For each plot, on each measurement occasion, stand characteristics of the deforested areas are determined and applied to yield models to derive volume estimates. Biomass estimates are derived from the volume estimates and, from the biomass estimates, the carbon estimates are derived.

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[1] The sequential approach to labeling of change areas involves:

1. Use the imagery/maps and other readily available sources of information to determine
   - which change areas are definitely deforestation
   - which change areas are definitely not deforestation
   - which change areas are doubtful
2. Use other sources of information (land use data, air photos, local agencies) to determine which of the doubtful ones are deforestation.
3. If some change areas remain doubtful or uncertain, set up a procedure to handle them.

Next, calculate the carbon change for each plot and the average for all plots. This average is expanded to represent the total carbon change due to deforestation for the strata represented by the plots.

Stratum L - Expected Low Deforestation

Only the NFI plots are used in this stratum so the procedure for estimating deforestation area is simple; obtain the deforestation data from the NFI plots in the stratum and calculate the average.

Unlike Stratum H, carbon change can here be derived for each NFI plot and averaged. The procedures is as follows:

1. Obtain the stand characteristics for deforested areas of each plot, on each measurement occasion
2. Apply stand characteristics to yield models, determine biomass from the resulting volume estimates and, from the biomass estimates, the carbon estimates
3. Calculate the difference for each plot and the average for all plots. This is the average multiplied by the area of the strata.

This approach is standard simple random sampling so formulae for the derivation of precision estimates are commonly available.

Population Statistics

Population statistics are arrived at by combining the estimates from the two strata. Population means and totals are weighted (by area) averages of the two stratum values. Precision estimators, if not available, can be derived.

DISCUSSION

This approach utilizes the satellite imagery to sample intensively where deforestation activity is high (Stratum H), and integrates these estimates with the NFI based values using a double sampling estimator. In Stratum L, where deforestation activity is low, only the NFI plots are used. Due to the large area of Stratum L, the sample size will be large and the anticipated precision good. This is a simple base design on which to build and discuss options.

Many modifications may be made to the core design including eliminating the stratification and changing sample size and plot size. The exact nature of the stratification strongly influences cost and the number of satellite images needed. This needs to be explored in more depth. The total area of satellite imagery interpreted impacts the time and cost of interpretation. How this area is best distributed in terms of number and size of samples is a design consideration. Indeed, it may be viable to delineate deforested areas for the entire coverage (Stratum H). It must be noted that the vetting of possible deforestation to confirm them as deforestation by closer scrutiny or use of other sources of information, as in the sequential approach, is one of the most time consuming elements. As well, determination of the carbon change for sites requires that the forest cover be known so volume to biomass to carbon relationships can be applied. This implies that the location of the deforestation be related to the forest inventory or at a minimum a broad forest type interpretation from the imagery. The work load involved in relating sites to forest inventory data depends on the information access and extraction tools and protocols available. The trade-off between careful vetting and leaving some areas unconfirmed is an important issue. Options range from: a) applying the double sampling to all sites identified as possible deforestation without any vetting, to b) assuming all unconfirmed sites are deforestation (if the area is small it may be more cost effective to accept the negative carbon consequence than expend effort in determining the carbon change more precisely), to c) a case where time and money is spent to confirm all cases and the double sampling is not needed. The most appropriate options will depend on, among other factors, the number of possible sites that remain doubtful after simple image interpretation of the satellite imagery along with the use of easily available ancillary information (local records). The most likely operational scenario will lie somewhere between these extremes; the following gives an example of such an integrated system.

Applying the double sample to all plots and deforestation sites seems unnecessary, as the satellite and NFI interpretation will be correct in most cases. Using the double sampling for only the uncertain areas/areas of doubtful deforestation may be a good option. Figure 2 outlines this approach. The derivation of an integrated estimate is accomplished through the double sampling estimator in which the average uncertain deforestation is estimated by:

$$y_R = (y/x)\, x'$$

where:

y_R = average uncertain deforestation area;

y = average uncertain deforestation area from the NFI plots (small sample)

x = average uncertain deforestation area from the satellite plots (small sample)

x' = average uncertain deforestation area from the satellite plots (large sample).

Having identified the areas of uncertain deforestation, the amount of carbon sequestered on these areas before and after would be determined from the forest inventories as above. The average deforestation area (y_R) must be multiplied by the average carbon change for that area. Average carbon change per hectare of deforested land is derived from the stand characteristics of all the deforested stands identified on the NFI plots. These are based on forest inventory data. Stand characteristics of the deforested sites of each NFI plot are determined on each measurement occasion. Volume estimates for each site are then determined by applying the stand characteristics to yield models. An average over all the plots is determined and applied to the average uncertain deforestation area of the remote sensing plots (y_R). Within this example there are design variants related to, for example, whether the carbon change should be derived using an average of all deforested areas of the NFI plots or just an average for the uncertain, or whether it is determined by calculating the carbon stock an all uncertain areas identified on the satellite plot and then using the double sampling parameter (y/x) to estimate what proportion of this is related to actual deforestation. This procedure or variants of it gives an estimate of the carbon change on the uncertain areas of the satellite plots.

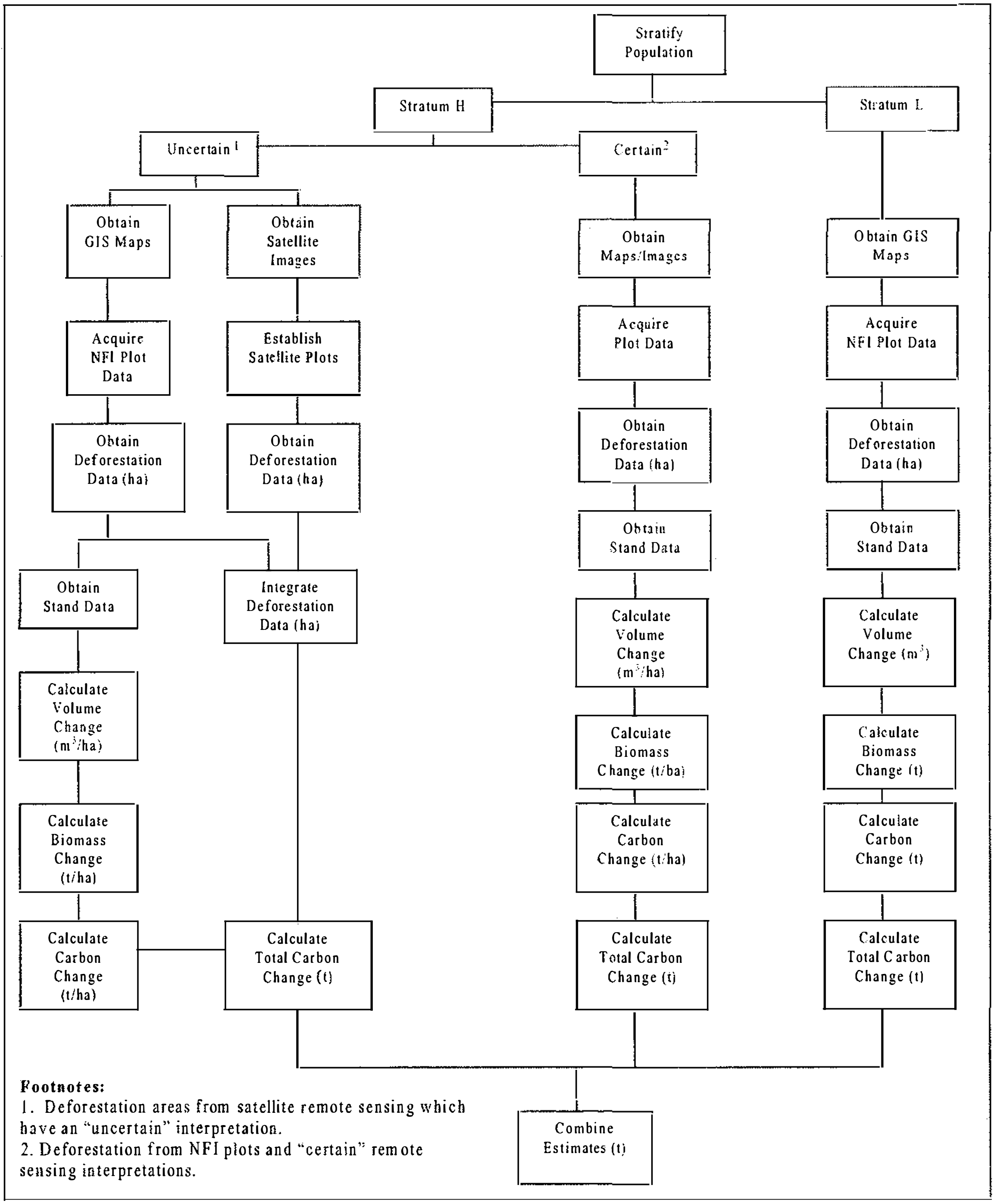
The estimate of total carbon change due to deforestation for all the plots would be the sum of the carbon change for the "certain" deforestation areas calculated for the NFI plots and for confirmed areas of the satellite plots plus the estimated carbon change (through the double sampling method) for the uncertain areas of the satellite plots. This would then be prorated/expanded to estimate the total carbon change for the strata represented by the plots.

NEXT STEPS

The next steps in developing a deforestation measuring and reporting system are to test the design and to arrive at a refined, tested and ready-to-implement system design using NFI, remote sensing, and other data sources to address deforestation.

Testing with three components is proposed.

1. Testing the core design methodology on two study areas. This involves establishing 100 - 200 NFI photo plots (linked with NFI pilot projects) and two Landsat TM scenes per study area. This process exercises the mechanisms and procedures, providing an opportunity to refine the system and reporting the test results. The steps include:
    - Establishing NFI plots (drill existing provincial inventory databases or interpret from aerial photography)
    - Expanding the coverage using satellite TM imagery. Define areas of change; label the type of change
    - For the deforestation areas, extracting information about forest cover and using this to derive wood volumes, from which biomass and carbon estimates can be derived
    - Combining satellite data sources with NFI data sources
    - Generating statistics
    - Assessing accuracy and efficiency of system design
    - Identifying areas of refinement
    - Adjusting and reporting



Footnotes:

1. Deforestation areas from satellite remote sensing which have an "uncertain" interpretation.
2. Deforestation from NFI plots and "certain" remote sensing interpretations.

Figure 2. A schematic of the integrated, stratified core design with double sampling for uncertain satellite interpretations

2. Conducting a pilot project (operational trial) on a region of high deforestation activity. This would involve a complete remote sensing/NFI photo plot incorporated trial.
3. The testing and the operational trial are undertaken over a limited range of conditions. To address this and to demonstrate some national capabilities, a national survey using approximately 40 satellite images would be conducted. This would provide additional experience on the remote sensing analysis under a wide variety of conditions. This would also help identify the types, locations, patterns and rates of deforestation. A national survey could also help define where to focus the sampling and identify the sample design. This could also provide a first approximation of the amount and rate of change associated with deforestation in Canada. A minimum number of NFI plots would also be established as part of the survey.

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FIRE REGIMES AND THE CARBON DYNAMICS OF BOREAL FOREST ECOSYSTEMS

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ABSTRACT

Whether a forest ecosystem functions as a sink or source for atmospheric carbon is determined by the relative amounts of carbon taken up from and released to the atmosphere. Stand-replacement disturbances play an important role in the function of Canadian boreal forest ecosystems. Here we consider landscapes in which forest fires play the dominant role and test a hypothesis that in such landscapes, carbon dynamics are correlated with fire disturbance patterns. Specifically, we hypothesize that at the landscape scale, higher mean values and small variations of ecosystem carbon are associated with long fire cycles, while lower mean values and larger variations of ecosystem carbon are associated with short fire cycles. The hypothesis was supported by a simulation experiment using a spatially explicit model of landscape dynamics (SEM-LAND) based on a study area in west central Alberta, Canada.

INTRODUCTION

The role of terrestrial ecosystems in the global carbon budget has been extensively discussed in the literature (e.g., Watson et al, 2000, Ciais et al. 1995, Houghton 1996). A number of estimates at national or regional scales have concluded that temperate and boreal forests are carbon sinks (e.g., Kauppi et al. 1992; Sedjo 1992; Dixon et al. 1994) while other studies have reported the forests function as a carbon source (e.g., Harmon et al. 1990). Kurz and Apps (1999) suggested that the role of forests in the carbon budget could change over time depending upon changes in the prevailing disturbance regimes. For instance, the forest ecosystems in Canada have been a sink of atmospheric carbon for the period 1920-1980, and a source for the period of the 1980s due to a sharp increase in forest fire and insect disturbances starting about 1970.

Whether a forest ecosystem functions as an atmospheric carbon sink or source is determined by the relative amounts of carbon taken up from and released to the atmosphere through changes in the biological state. For the biomass carbon pool in the boreal forests of Canada, the uptake of carbon is determined by the net biomass increase resulting from forest vegetation growth. The amount of carbon released to the atmosphere is largely determined by the decay of dead organic matter and emissions associated with stand-replacement disturbances such as fire. Some types of timber harvesting also cause stand-replacement, and transfer carbon stored in the forest biomass pool to other pools including forest products and slash, thereby increasing the decomposition emissions from these pools. Insect attacks and disease can also result in stand-replacement, and like harvest, transfer forest biomass carbon to the dead organic matter carbon pool. The carbon dynamics of a forest landscape is the result of the balance of total carbon sequestered through forest growth and the carbon release through all mechanisms (including direct and post-disturbance release) overtime.

This research focuses solely on the influence of forest fires and ignores other stand replacing disturbances that may be present. Forest fires interrupt forest growth and release biomass carbon into the atmosphere immediately through combustion and with a delay, through post-fire decomposition. The objective of this study is to test the hypothesis that landscape carbon dynamics are correlated with fire disturbance patterns: at the landscape scale, higher mean values and smaller variations of ecosystem landscape total carbon are associated with long fire cycles, while lower mean values and larger variations of total landscape carbon are associated with short fire cycles. The hypothesis was examined using a spatially explicit model for landscape dynamics (SEM-LAND) and tested for a forest landscape in west-central Alberta,Canada. Two descriptors, fire frequency and fire size distribution, are used to characterize the fire disturbance patterns. The fire frequency is defined as the mean annual percentage of area burned. The fire cycle is defined as the average time required to burn an area equivalent to the total area under investigation and is numerically equal to the reciprocal of the fire frequency. A theoretical analysis of the relationships among fire cycle, fire size distribution, forest biomass production, and carbon dynamics at the landscape scale is given, followed by a brief summary of the SEM-LAND model, the design of the model experiment, results and discussion. We conclude with suggestions for further research.

THEORY AND HYPOTHESIS

Long-term Perspective of Carbon Sink and Source Function

The concept of carbon sink and source is illustrated in Figure 1a which shows changes in ecosystem carbon stocks over long periods of time. The total ecosystem carbon at the landscape scale at the end of year *t* can be calculated by the equation [1]:

$$C_{Total,t} = C_{Total,t-1} + \Delta C_{Increment,t} \qquad [1]$$

where $C_{Total,t}$ and $C_{Total,t-1}$ are the total ecosystem carbon at the end of years *t* and *t-1* respectively, and $\Delta C_{Increment,t}$ represents the net increment in ecosystem carbon at the end of year *t*. The increment in ecosystem carbon represents net changes in both living biomass and dead organic matter pools. It may be calculated as the difference between net carbon sequestration (or net primary productivity NPP, $\Delta C_{NPP,t}$) and the net carbon loss in year *t*. The net carbon loss in year *t* is the sum of direct carbon emission from combustion ($\Delta C_{DE,t}$), and carbon released through decomposition ($\Delta C_{Decomp,t}$) of accumulated dead organic matter (containing organic carbon transferred by disturbances as well as non-disturbance related litter input), thus:

$$\Delta C_{Increment,t} = \Delta C_{NPP,t} - (\Delta C_{DE,t} + \Delta C_{Decomp,t}) \qquad [2]$$

As indicated in Figure 1, the net increment can be either positive ("sink") or negative ("source") over any given time interval.

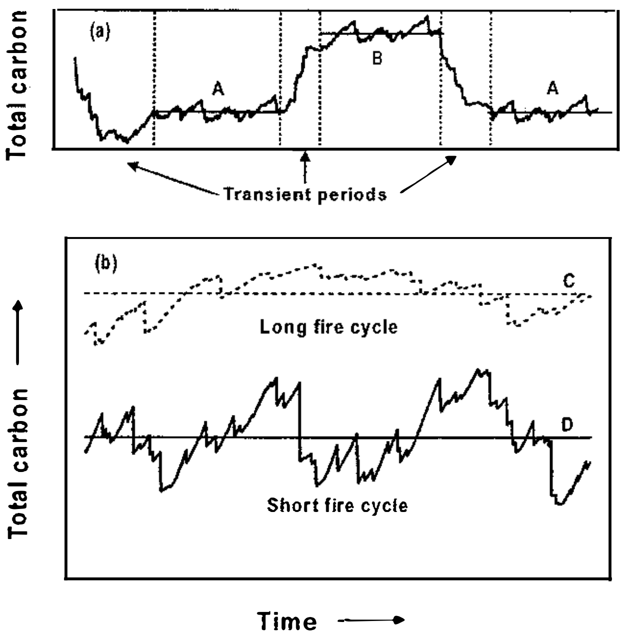

Figure 1. Diagram of carbon sink and source concepts from (a) a long-term perspective, and (b) a short-term perspective. In (a), the total carbon fluctuates around a mean value A under current conditions. Changed disturbance regime and climate conditions result in a higher total carbon associated with a longer fire cycle, hence the mean total carbon value changed from A to B. When circumstances are restored to the previous conditions, the mean total carbon value may change back from B to A. In (b), a long fire cycle (low annual burned area) usually results in higher mean total carbon value C and smaller variation around the C. A short fire cycle often results in a lower mean total carbon value D and larger variation.

The decomposition term ($\Delta C_{Decomp,t}$) typically involves a coupled set of decomposing pools, each distinguished by their origin or source of detritus input and characteristic turnover rates (e.g., the slow, medium, fast and very fast dead organic matter pools associated with humified soil organic matter, dead boles, dead roots and branches and fine roots and foliage, as used by Kurz and Apps 1999). Because these decomposing pools are coupled, changes in $\Delta C_{Decomp,t}$ and

$\Delta C_{Increment,t}$ in a given year t depends on the changes that have taken place in previous years. In particular large pulses of carbon enter the linked decomposing pools are associated with disturbance events, and give rise to large changes in $\Delta C_{Decomp,t}$ over time, depending on the actual disturbance history. Thus $\Delta C_{Decomp,t}$ for a given location is a complex function that involves an integration over the history of that site (Bhatti et al, 2001).

Under an unchanging climate and disturbance regime, the ecosystem carbon at the landscape scale C_{Total} would tend over time towards a steady state at which uptake of carbon from the atmosphere just balances, on average, the releases of carbon through decomposition and direct release. The C_{Total} then fluctuates around this mean value (e.g., level A in Figure 1a). The fluctuations arise from changes in carbon stocks associated with individual ecosystems making up the landscape as they undergo variations in rates of growth, death, regeneration, and decomposition.

If an abrupt change occurs in the disturbance regime for the landscape, the net balance (equation 2) would be altered, and over time, the system would again tend towards a new steady state with fluctuations around a new mean value. If fire led to a decrease in disturbances by fire, for example, the ecosystem carbon stocks may rise to a new mean level B as in Figure 1a. During the transient period between the two disturbance regimes, the slope of the changes in ecosystem carbon determines the magnitude of the landscape sink (as in the transition from A to B).

Note however, that if the disturbance regime returns to its previous level (e.g., discontinuance of fire protection, or an increased fire hazard), the landscape carbon stocks would return to the previous mean level (A) and during this transient period, the landscape is a source (B to A). Thus it is the changes in disturbance regimes and hence between mean values (A) and (B), that determine whether a forest functions as a carbon sink or source, on average, over longer time scales. Kurz and Apps (1999) published an application of this concept to Canadian forests and showed that an increase in fire and insect disturbances during the 1970s and 1980s relative to earlier periods caused Canadian forests to act as a carbon source. During this period of increased disturbances (analogous to the transitory phase above), total ecosystem carbon decreased, resulting in a change in Canadian forests from a carbon sink to a carbon source. Exploratory analyses of future scenarios indicated a strong dependence on the assumptions about future disturbance regimes (Kurz and Apps 1994).

Short-term Perspective of Carbon Sink and Source Function

Using changes in mean values as an indicator of source or sink is less meaningful when the short-term fluctuations of ecosystem carbon (C_{Total}) around the mean value (A or B in Figure 1a) are of a similar magnitude to the difference between the two mean values (A) and (B). However, the concepts described above can also be applied to examine annual changes as shown in Figure 1b. The hypotheses examined in this paper are presented visually in this figure: 1) long fire cycles usually result in higher mean values (C) than those values (D) associated with short fire cycles and 2) the fluctuations in ecosystem carbon (short-term sources and sinks) are smaller with long fire cycles than with short fire cycles.

Assessment of carbon sink or source

The amplitude of inter-annual forest C_{Total} fluctuation over a landscape in which fire is the dominant disturbance agent can be determined by three factors: (1) rates of forest growth; (2) size, intensity (hence tree mortality), and severity (hence the direct carbon emission ΔC_{DE}) of fires (both natural and human caused); and (3) the carbon release through the decomposition processes ΔC_{Decomp}. If the net carbon sequestration from forest growth ΔC_{NPP} is higher than loss from ΔC_{DE} and ΔC_{Decomp}, then the C_{Total} will increase, resulting in a net carbon sink (equation [2]). However, if the ΔC_{NPP} cannot compensate for the loss from ΔC_{DE} and ΔC_{Decomp} in a given year, then the C_{Total} must decrease making the forest appear as a carbon source. When ΔC_{NPP} is precisely equal to $(\Delta C_{DE} + \Delta C_{Decomp})$, the landscape is in steady state, with a net sink (or source) strength of zero.

Fire cycles and size distributions

The biomass dynamics of forest ecosystems are influenced by fire regimes that are characterized by cycle, size distribution, intensity, severity, type, number, and season (Weber and Flannigan 1997). Over a long time and in the absence of human intervention, the number of fires plotted against their size (fire size distribution) has been shown to decrease exponentially (Li and Corns 1998). (Li and Corns also point out that this does not imply, however, that the fire size distribution can be characterized by a negative exponential probability distribution.) Assuming fire size distributions have a negative exponential shape, Li et al. (1999) demonstrated that the fire size distributions are correlated with fire cycles under natural conditions. Short fire cycles generally have larger mean fire sizes and hence have relatively more large fires than long fire cycles.

Under the influence of human activities, two interacting factors must be examined. One is the increased risk of fire initiation by industrial, recreation, and residential causes. The other is the decrease of fire sizes as a result of fire suppression efforts. Consequently, human activities may be expected to alter natural fire regimes from shorter fire cycles to longer fire cycles. Changes in the associated fire size distributions can also be expected. The following are examples of the possible relationships between fire size distribution and fire cycle as affected by human activities:

1. Baker (1992) reconstructed 3 fire-size distributions from Heinselman's (1973) fire-year maps for the Boundary Waters Canoe Area (BWCA) during 3 time periods. The Pre-Settlement period during AD 1727-1868 was associated with a short fire cycle. The Settlement period during AD 1869-1910 was associated with a longer fire cycle. The Fire Suppression period during AD 1911-1972 was associated with an even longer fire cycle. The differences among the 3 fire-size distributions indicated that the short fire cycle in the Pre-Settlement period had less small fires and more large fires than the longer fire cycles in the Settlement and Fire Suppression periods.
2. Ward and Tithecott (1993) compared fire-size distributions under different fire suppression efforts, and found various levels of fire suppression have significantly increased the number of fires smaller than 200 ha, and considerably reduced the number of fires larger than 200 ha. Consequently, fire cycles have been increased from 65 to 580 years. This conclusion has been qualitatively supported by the comparison of fire regimes under various suppression efforts in the Red Lake District of Ontario (Li 2000a) and in Ecoregion 90 of northwestern Ontario (Wotton 2000, personal communication). Li (2000a) showed that fire cycles have changed from 51 to 135 years under fire suppression, and mean fire size has been reduced from 1993 to 182 ha in the Red Lake District. Wotton's results showed that the mean fire size for Ecoregion 90 has been reduced from 3,462 to 246 ha under fire suppression. He thus suggested that the effect of fire suppression on the fire size distribution is to shift fires from the large to the very small size classes.
3. In Ontario, Canada boreal forests without the influence of human activity, have shorter fire cycles (50-100 years) than the Great Lakes-St. Lawrence forests (100-350 years) with the influence of human activity. In the boreal forests fire sizes are usually larger than in the Great Lake-St. Lawrence forests (Thompson 2000).

These examples suggest that the relationships between fire size distribution and fire cycle might be similar under both natural and anthropogenic conditions, i.e., more frequent small fires and less frequent large fires under long fire cycles, and less frequent small fires and more frequent large fires under short fire cycles. The consequence associated with the different fire size distribution may be that frequent small fires and less frequent large fires in long fire cycle situations would cause more frequent small variations around the mean value (C) as illustrated in the Figure 1b. In contrast, less frequent small fires and more frequent large fires associated with short fire cycle situations may cause more large variation about the mean value (D) in Figure 1b. Because by definition, short fire cycles imply larger areas are burned annually, they are associated with lower mean values (D) of ecosystem C than are long fire cycles (C in Figure 1b). Therefore, smaller variations in amplitude may be expected with higher mean values (C) than are expected with short fire cycles and lower mean values (D).

Fire size distributions and annual carbon increment

The slope of total carbon increase $\Delta C_{Increment}$ may be smaller under long fire cycle conditions as a consequence of the pattern of biomass increase at different forest ages (Figure 2). The net balance of changes in all the affected carbon pools determines the carbon dynamics at the landscape scale. Figure 2a, from a typical forest growth equation, shows that standing biomass increases with forest age, and eventually reaches a relative stable level at maturity (in some forest types, the mature phase is followed by an over mature state in which biomass declines with the death of old but large trees and the regeneration of young but small ones). Figure 2b shows the annual biomass increment (which can be converted into total carbon inputs into the ecosystem ΔC_{NPP}) at different forest ages. This annual ΔC_{NPP} pattern indicates that the largest ΔC_{NPP} occurs in young to intermediate age forest stands, with a sharp drop off in very young stages and a gradual decline approaching maturity. Thus we might expect that a higher slope, or a more rapid increase of C_{Total} might be observed with a conversion to these younger stands (ignoring changes in the second and third terms in equation 2). Since on average the forest is younger under short fire cycles (more area burned annually, hence higher mortality) than under long fire cycles, more of the forest is a state of rapid accumulation of ecosystem carbon than under long fire cycles.

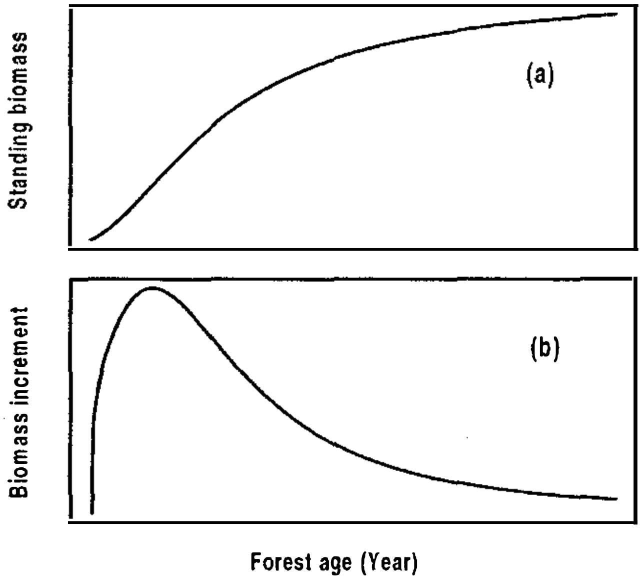

Figure 2. Patterns of (a) standing biomass accumulation and (b) annual biomass increment at different forest ages.

Two caveats apply to the above arguments that can, in some ecosystems, complicate the analyses. First, there is an implicit assumption that the vegetation growth characteristics (Figure 2) are the same under the two disturbance regimes (no successional replacement by faster or slower growing vegetation). Second, changes in the disturbance regime dramatically alters the second term in equation 2 in the early stages of stand replacement by the changes in the size of the decomposing dead organic matter pools. Hence the shape of net balance (equation 2) is significantly different than the biomass increment curve (Figure 2b) and typically goes strongly negative for the first decade or so after disturbance as the additional pulses of dead organic matter decompose.

Scale effect

At a stand level, most living biomass becomes dead organic matter after disturbance by fire (only on the order of 20% or so of the biomass is fully combusted). For some time after the event, stand growth is small relative to the increased carbon release from heterotrophic respiration (ΔC_{Decomp}) from soil due to increased microbial activities induced by increased soil temperature and the large inputs of fresh organic matter from the disturbance event. The consequence of the increased post-fire carbon release through ΔC_{Decomp} is a reduction in the $\Delta C_{Increment}$ of the stand during the years following fire disturbances. The $\Delta C_{Increment}$ can be negative for up to 5 years before returning to positive values (Barnes et al. 1998), and for boreal forests this can be much longer, i.e., 20-30 years (Amiro et al. 2000, Kurz and Apps 1999).

At the landscape scale, however, the changes in $\Delta C_{Increment}$ depend on the relative size of burned areas over time and the biomass density in these areas. If the burned area is a small enough portion of the landscape, the reduction in total ecosystem carbon in the burned area can be compensated by the net carbon taken up through forest growth ΔC_{NPP} in the unburned area, thus the annual carbon increment may still be positive. In this case, the forest landscape functions as a carbon sink. When the burned area is a larger portion of the landscape, however, the regrowth (ΔC_{NPP}) in the unburned area cannot fully compensate for the loss in total ecosystem carbon in the burned area. Therefore, the annual total carbon balance C_{Total} will decrease, i.e., the forest landscape will function as a carbon source in that year. This emission phase may last for several years, depending upon the geographical location and conditions of the affected landscape, until ΔC_{NPP} of the landscape exceeds ($\Delta C_{Decomp,t} + \Delta C_{DE,t}$). For a forest landscape in the boreal region, the changes in the dead organic matter pools characteristically play out over long time scales (decades and longer) and are similar to the long (and slow) periods vegetation regrowth. The fire size distribution and its temporal pattern will influence the forest carbon dynamics, if the fire ignition source level for a given landscape is assumed not to change significantly. Under short fire cycles, more frequent large fires could result in large total carbon reductions in large fire years, followed by large rates of total carbon increase. The large rates of increasing C_{Total} are due both to faster growth rates of the younger forests, and the reductions in area burned in the off-fire years. In contrast under long fire cycles, less frequent large fires would be expected to result in smaller amplitudes of total carbon reduction in fire years, followed by lower slopes of total carbon increases between fires.

Hypothesis and Method of Test

The hypothesis based on the concept and theoretical analysis presented above is that the carbon dynamics of the fire-dominated landscape are correlated with fire disturbance patterns. Specifically, it is hypothesized that long fire cycles tend to result in a higher mean carbon storage with smaller amplitude fluctuations, while short fire cycles result in a lower mean carbon storage with larger amplitude fluctuations.

The hypothesis was tested by conducting a series of simulation experiments with the SEM-LAND model. The model was suitable for testing the hypothesis because it can simulate both forest and fire dynamics within a forest landscape under various conditions (Li, 2000b; Li et al., 1999). A brief overview of the model and the model experiment for testing the hypothesis shall be described in the next section.

SEM-LAND MODEL AND MODEL EXPERIMENT

The SEM-LAND model was designed to simulate the dynamics of both forest growth and fire spread at the landscape scale over a long period of time. Since the model has been described in Li (2000b) in detail, we will provide only a brief overview. The premise of the model development is that forest landscape dynamics are the results of the interactions among ecosystem components, and the modeling should focus on representing these interactions.

The SEM-LAND model simulates forest growth and fire disturbance processes. The model employs the published Alberta provincial forest growth equations for major tree species (Alberta Forest Service, 1984) to simulate average forest growth under various climatic conditions. Fire disturbance processes are simulated according to the relationships summarized in the Canadian Fire Weather Index System (FWI) (Van Wagner, 1987) and the Canadian Forest Fire Behavior Prediction System (FBP) (Forestry Canada Fire Danger Group, 1992; Hirsch, 1996). The formulae can also be found in Li (2000b) and essentially describe the influence of weather conditions on fuel moisture content, fuel types on the rate of fire spread, and landscape topography and wind speed on the directional rate of fire spread. The vegetation types are assumed to be unchanged after fires hence a self-replacement process is hypothesized.

In the raster-based SEM-LAND model, a

fire process is simulated according to the current understanding of natural fire events. A fire process is simulated in two stages: initiation and spread (Li and Apps, 1995, 1996). The fire initiation stage starts with the presence of a fire ignition source that is randomly located within the landscape. The program calculates the fire-initiation probability as a function of vegetation cover type (hence the fuel type), stand age, and weather conditions. Whether this fire ignition source results in a fire initiation (i.e., kills most trees in the forest stand represented by the pixel) is determined by this fire-initiation probability.

Once a fire is initiated, it has the potential to spread to its eight neighboring stands, and the spread stage begins. For each of the neighboring stands, the program calculates a fire-spread probability as a function of fuel type, stand age, topography, and weather conditions. This fire-spread probability determines whether the adjacent stand will be burned. If it is burned, then the fire spread process is repeated, and the new eight neighboring stands are checked. This process continues until the fire stops in all directions or reaches the boundary of the landscape.

The model experiment was carried out based on a study area: the Athabasca Working Circle Compartment 24 of Weldwood of Canada Limited (Hinton Division) in the Rocky Mountain Foothills of west-central Alberta, Canada. Our hypothesis was tested using the spatial data sets from this area since the SEM-LAND model was also validated in this study area (Li 2000b). This study area is located between latitude 53.7161 to 53.809 and between longitude 117.335 to 117.534, with a total area of 7,432 ha. The major tree species that determine the forest cover types are: lodgepole pine (*Pinus contorta* Dougl. Ex Loud. var. *latifolia* Engelm.) occupies about 58% of the total area, black spruce (*Picea mariana* (Mill.) B. S. P.) about 16%, white spruce (*Picea glauca* (Moench) Voss) about 4%, balsam fir (*Abies balsamea* (L.) Mill.) about 2%, and trembling aspen (*Populus tremuloides* Michx.) only 1%. The rest of the area (19%) is covered by other vegetation types such as timbered muskegs, open muskeg or swamp, and brushland.

Four map layers including vegetation cover type, Digital Elevation Model (DEM), stand age, and site index of the study area were used as current forest conditions. Other model input includes data of weather and fire ignition sources. The daily weather data of the fire season (from May 1 to September 30) from the Hinton weather station during the period of 1980-1989 was used to calculate the daily Fine Fuel Moisture Code (FFMC) of the FWI system. The daily noon wind speed was also compiled as a frequency distribution and used in the simulation. The annual fire ignition sources were represented by a simulated time series that follows a normal probability distribution.

In testing our hypothesis, the amount of biomass increase from forest growth and decrease by fire burns need to be simulated. Dynamics of the biomass carbon pool simulated by the SEM-LAND model provide input data for a dynamic simulation model that links the dynamics of biomass carbon pool to different belowground carbon pools. This dynamic simulation model was constructed as a simplified representation according to the CBM-CFS model (Kurz et al. 1992). The simulated forest stem wood dynamics in volume (cubic metre per ha) were converted into total biomass (including stem wood, stump wood, stump bark, stem bark, top, crown, branches, and foliage in kilograms per ha) using conversion factors for different tree species (Penner et al. 1997). The amount of biomass was then converted to carbon using a factor of 0.5. Under the influence of fire disturbances, living biomass is transferred to dead organic pools differentiated by their turnover time and origin: fast (i.e., half-lives of 3-20 years for detrital material that is less then 10 cm in diameter, such as foliage and submerchantable biomass), medium (i.e., half-lives of 20-100 years for detrital material that is greater than 10 cm in diameter, such as merchantable stemwood), and slow (i.e., half-lives greater than 100 years and the humified soil organic matter in forest soils) pools (Kurz et al. 1992). The fractions of the live biomass pools transferred by a given fire event were 9.9%, 34.6%, and 4.9% respectively (Kurz et al. 1992). The decomposition rates for fast and medium carbon pools used in the model experiment were 0.119 yr^{-1} and 0.024 yr^{-1}, and the percentage of unburned biomass was defined as 19.7% (Kurz et al. 1992).

The SEM-LAND model simulates a fire regime as the result of interactions among landscape structures, fire events, and weather conditions. Under an unchanged climate scenario, different fire regimes can most likely be caused by different overall landscape fire susceptibilities. Therefore, we manipulated the fire susceptibility of the overall landscape to achieve different fire regimes. In the simulation experiment, a range of fire cycles from 50 to 5,000 years is within the range of observations that cover fire regimes under natural and management scenarios. Seventy simulation runs were conducted to test our hypothesis. Twelve hundred years of forest and fire dynamics were simulated in each simulation run, but only results from the last 1,000 years were used in the analysis to avoid possible artifacts associated with initial conditions.

SIMULATION RESULTS

Dynamics of $\Delta C_{Increment}$ Under Different Fire Cycles

Figure 3 shows four samples of the SEMLAND simulation results of forest carbon increment ($\Delta C_{Increment}$) for the entire forest landscape under different fire cycles. Each sample represents one realization of a random fire process under a set of given environmental conditions. The dashed lines in the four graphs indicate the steady state condition, $\Delta C_{Increment} = 0$. The curves above the dashed lines indicate a positive $\Delta C_{Increment}$, thus indicating a carbon sink. The curves below the dashed lines indicate a negative $\Delta C_{Increment}$, and thus a carbon source.

Under a long fire cycle, such as 3,159 years shown in Figure 3a, biomass increase from net forest growth is small in most of years, and negative $\Delta C_{Increment,t}$ values can easily result from the biomass loss from small fires. Consequently, both small negative and positive $\Delta C_{Increment,t}$ values exist. Under a short fire cycle (106 years in Figure 3d), the total number

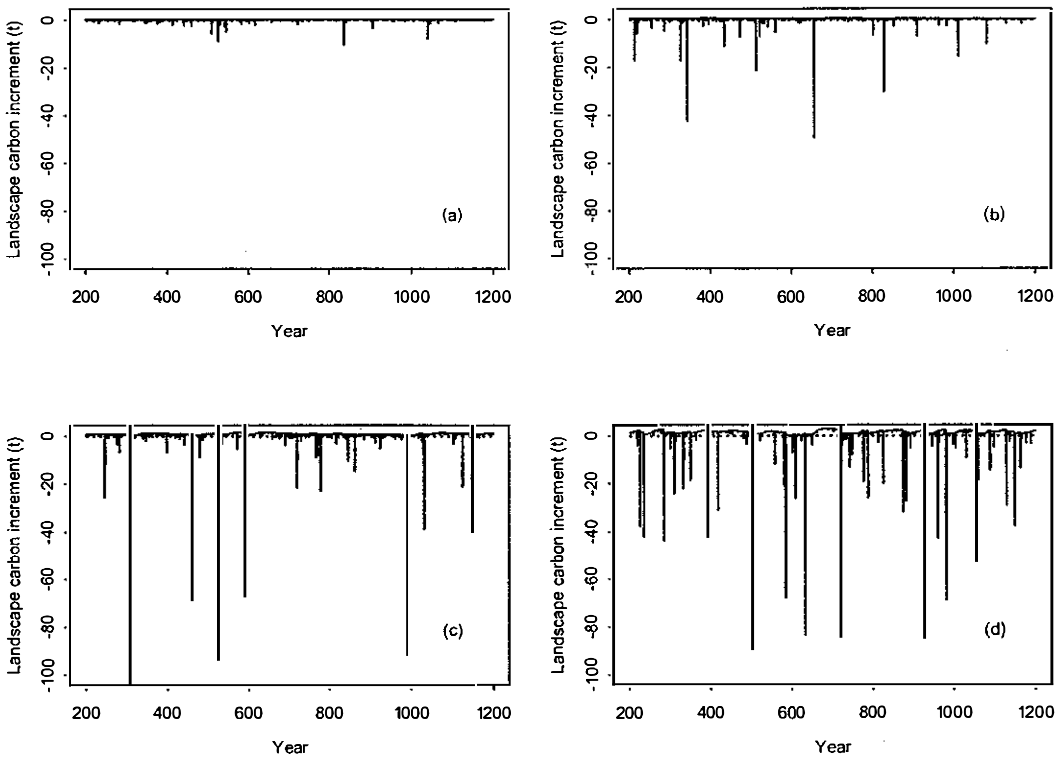

Figure 3. Dynamics of landscape carbon net increment under different fire cycles: (a) 3,159 years; (b) 1,414 years; (c) 426 years; and (d) 106 years. Dashed lines indicate zero increment.

of years with negative $\Delta C_{Increment,t}$ values was significantly reduced, and the annual biomass increase from forest growth was significantly increased relative to a long fire cycle (Figure 3a). Because the total numbers of fires (but not the area burned annually) during the 1,000-year simulations were similar in the four conditions, the lower number of years with negative $\Delta C_{Increment}$ values under shorter fire cycles is a result of the higher biomass increase from forest growth compensating for biomass loss (due to fires). This larger annual biomass increase from forest growth occurring in most years is a result of the increased percentage of younger stands (with higher rates of biomass accumulation) produced by the shorter fire cycle (larger burned area annually).

Figures 3b shows the dynamics of net biomass increase under a fire cycle of 1,414 years, where the total number of negative $\Delta C_{Increment,t}$ values was reduced compared with Figure 3a, but the amplitude of the negative and positive values became larger. Figure 3c shows the dynamics of $\Delta C_{Increment,t}$ under a fire cycle of 426 years. The amplitude of the negative $\Delta C_{Increment,t}$ values became even larger than in Figure 3b and similar to that in Figure 3d.

Dynamics of Total Landscape Biomass

Figure 4 summarizes the mean values and associated variations of landscape total carbon as simulated by SEM-LAND under different fire cycles expressed as the percentage of mean annual area burned. With smaller mean annual area burned, i.e. with long fire cycles, higher landscape carbon and smaller variations occur. Increasing the mean annual area burned was associated with lower total landscape carbon and larger variations around the means. The dynamics of total landscape carbon under various fire cycles differed significantly as indicated in the Figure 4. Under the long fire cycle of 3,758 years, the total landscape carbon was as high as 599 t, and the associated standard deviation was small at 13 t. With a shorter fire cycle (762 years), the total carbon decreased to 404 t with a standard deviation of 28 t. When the fire cycle was shortened to 140 years, the total carbon was about 212 t with a standard deviation of 32 t. The simulation results also suggested that the standard deviation might not change significantly with fire cycles shorter than 726 years. However, the number of large fires could increase with the decreasing fire cycles (see Figure 3) because of increased mean annual area burned.

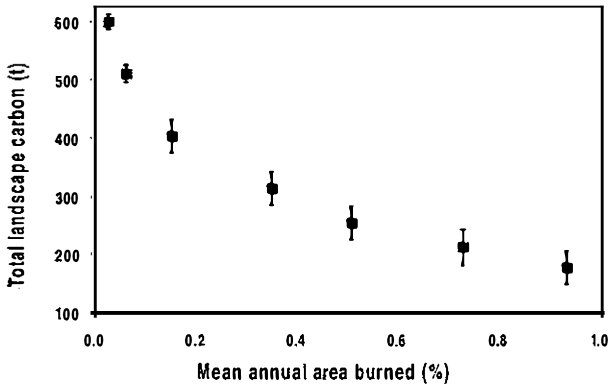

Figure 4. Total landscape carbon (T) under different mean annual area burned and their associated standard deviation (indicated by the error bars) over 1,000 years.

DISCUSSION

Our simulation results support both the theoretical analysis presented in section 2 and the hypothesis that the carbon dynamics of a forest landscape are correlated with fire disturbance patterns. Simulated stem wood biomass and total carbon storage, under long fire cycles was high with smaller variations, and the total carbon under short fire cycles was low with larger variations (see Figure 4). Our results suggested that the variations around the mean total carbon of the landscape are similar when the fire cycle is within the range of 100 to 762 years, and this is usually the range of fire cycles empirically observed under natural conditions and when fire suppression is practiced. However, the frequency of large fires could be higher under shorter fire cycles (e.g., 20 fires generated larger than 25 t carbon loss under the 106-year fire cycle, Figure 3d) that result in lower total landscape carbon, than under longer fire cycles (e.g., 8 fires generated larger than 25 t biomass loss under the 426-year fire cycle, Figure 3c) that result in higher total landscape carbon.

The expected theoretical net carbon increment averaged over a long period is 0 if the landscape dynamics achieve steady state conditions. Our simulation results indicated that the mean values of carbon increment over 1,000-year simulations were 0.02, 0.01, 0.02, and 0.1 t yr^{-1} for fire cycles of 3,159, 1,414, 426, and 106 years, respectively. The results were very close to the theoretical expectations.

To the extent that the Canadian boreal forest is dominated by stand-replacing fires, our simulation results suggest that the fire disturbance history in these forests has influenced the level of total carbon considerably. If, as is likely, the fire cycles before settlement were likely shorter than we observe today, the level of total carbon in forested landscapes would have been lower at that time. For example, fire-dependent tree species such as jack pine and red pine generally require frequent fires to maintain their presence in the landscapes and prevent invasion of other pioneer vegetation. Persistence of these species across boreal landscapes suggested that frequent fire must have existed, and indeed they were reported in the literature (Weber and Stocks 1998). Therefore, the levels of total landscape carbon would have been at relatively low values before the era of settlement.

With the introduction of fire suppression policies, fire cycles usually lengthen, and the total carbon levels across the forest landscapes would be expected to increase as a result. (We note, however, that some or all of this ecosystem gain may be offset by logging if, as is likely, it replaces fire as the dominant disturbance agent. Analyses of the combined effects of multiple disturbances on the landscape is beyond the scope of this paper but has been considered by Kurz and Apps (1999) and Kurz et al (1995). Our simulation results also indicated that because forest growth is a relatively slow process, increasing the length of fire cycles could preserve the carbon accumulated during this slow process. Ward and Tithecott (1993) concluded that a 65-year fire cycle in the umnanaged forestlands was changed to a 580-year fire cycle due to fire suppression in Ontario, Canada. If other factors are unchanged, such a change in the disturbance regime would result in less large fires and thus increased total carbon in the province.

The understanding of the long-term ecological effect of fire suppression, however, is still being developed. Accumulated fuel load resulting from fire suppression might increase the vulnerability of forest landscapes to subsequent fire disturbances and thus larger fires could be observed at later times. This could be one of the reasons that the Canadian boreal forests became a carbon source after 1980s as indicated in Kurz and Apps (1999). Potential climate change impacts on these altered fire regimes and landscape conditions adds another uncertainty. For forest landscapes with warmer and dryer climate conditions, chances are higher that fires escaping from fire suppression could result in more frequent large fires. For forest landscapes with cooler and wetter climate conditions, however, lower probabilities of fires escaping from fire suppression could result in less frequent large fires. Consequently, more complex scenarios of carbon dynamics in the Canadian boreal forests could be expected under changing climate conditions.

The hypothesis test presented in this paper was mainly focused on a short-term perspective of carbon sink and source function, i.e. each simulation was carried out under a given fire and climate condition until steady state status was achieved. The test represents a step towards the goal of developing a better understanding of the carbon dynamics of the Canadian boreal forest under different disturbance regimes and climate conditions. A long-term perspective of carbon dynamics at the landscape scale, however, is needed to support future policy development of managing forest and disturbance regimes, as well as to determine an appropriate adaptation strategy to climate change impacts. One issue, for example, is whether the previous levels of mean total landscape carbon could be re-established after the restoration of disturbance regimes through appropriate management.

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CLIMATE CHANGE EFFECTS ON INSECT OUTBREAKS AND MANAGEMENT OPPORTUNITIES FOR CARBON SEQUESTRATION[1]

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ABSTRACT

We describe a project currently underway to quantify the effect of insect disturbances on the carbon (C) sink potential of Canada's forests. Existing methodologies for forecasting effects of pest outbreaks on stand and forest timber volume development are being scaled up to the regional/national level and interpreted in terms of C dynamics, using the Spruce Budworm Decision Support System. We are developing a modeling system to analyze effects of climate change on pest outbreak dynamics and C sequestration patterns on a large regional scale. The system is being used to predict effects of climate change on the extent and impact of future insect disturbance regimes, and thereby on forest C sink potential. Scenario planning analyses are being used to determine effects of enhanced management/ protection against pests on C sequestration. Results will contribute to the development of a national action program on forest sinks and contribute to Canada's program to meet its Kyoto Protocol greenhouse gas emission reductions target.

INTRODUCTION

Canada's National Forest Strategy for 1998-2001 stated that "We will work toward meeting Canada's commitments arising from the Kyoto Protocol to the Framework Convention on Climate Change... by identifying and implementing cost-effective options for forest sector carbon sequestering and emission reductions to help meet Canada's targets" (Canadian Council of Forest Ministers 1998). Understanding effects of natural and human-caused disturbance on carbon (C) sequestering and emissions is a key element towards achieving this goal. Forest fire and pest outbreaks are two major natural disturbances that strongly influence C balances. Outbreaks of forest insects and diseases in Canada cause losses of 80-110 million m^3 of timber per year (Sterner and Davidson 1982, Power 1991, National Forestry Database 1995), or roughly one billion m^3 over a 10-year period. In comparison, about 25 million m^3/year were lost to fire (Bickerstaff et

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al. 1981), 161-180 million m^3/year were harvested, and sustainable harvest levels were an estimated 240-255 million m^3/year (Rotherham 1991, National Forestry Database 1995). Any serious attempts to influence C sinks in Canada must consider forest and pest management.

Climate change will influence insect outbreak and fire cycles, extent, and severity, and may well worsen the current situation (Fleming and Volney 1995, Fleming 1996). However, few data exist and improving knowledge and understanding in this major gap is a high priority (Volney and Fleming 2000). There may well be opportunities to increase both timber supply and C sequestration through enhanced management and protection against pests and fire. A key question is how climate change will affect the frequency, extent, and intensity of insect outbreaks. Because younger forests contain a smaller amount of biomass and C compared with older stands, and because stand-replacing disturbances result in younger stands, it follows that an increased frequency and severity of insect disturbances will decrease the amount of C stored in forest biomass. This will result in C from the forest biomass being released into the atmosphere where it may accelerate climate warming. Carbon budget studies have shown that changes in these disturbance regimes, especially those that may be significantly influenced by human activities, such as modification of climate, have a significant effect on the sink versus source status of Canada's forests (Kurz and Apps 1999).

The C loss following insect outbreaks may be a large source of CO_2, but no direct measurements are available. However, considerable information exists to quantify the effects of pest outbreaks and management on timber volume. Spruce budworm (*Choristoneura fumiferana* Clem.) and boreal spruce-fir (*Picea* sp. – *Abies balsamea* (L.) Mill.) forest constitute one of the best-understood systems, with data (e.g., MacLean 1980, 1985, 1990), models (MacLean 1996, Erdle and MacLean 1999, MacLean 1999), and a decision support system (MacLean and Porter 1995, MacLean et al. 1997, 2000a, 2000b) available to help predict effects of insect outbreaks on timber volume. Translation of timber volume model output and pest volume loss estimates into C estimates is feasible.

This paper will describe a project, supported by the Canada Climate Change Action Fund (CCAF) "*Greenhouse gases sources and sinks call*", that is currently underway to quantify the effect of insect disturbances on the C sink potential of Canada's forests. Existing methodologies for forecasting effects of pest outbreaks on stand and forest timber volume development, using the Spruce Budworm Decision Support System (SBW DSS) (MacLean et al. 2000a, b), are being scaled up to the regional/national level and interpreted in terms of C dynamics. Objectives of the project are:

1. to develop a modeling system to analyze effects of climate change on pest outbreak dynamics and C sequestration patterns on a large regional scale;
2. to predict the extent and impact of future insect disturbance regimes on forest C sink potential;
3. to use scenario planning ("*what if?*") analyses (Schoemaker 1995, MacLean 1998) to quantify effects of enhanced management and protection against pests on C sequestration; and
4. to estimate probable effects of climate change on pest outbreaks and C sequestration opportunities from enhanced management programs.

The project is still underway, so this paper will describe background information, plans, and current status.

USE OF PEST DSS TO ESTIMATE EFFECTS OF INSECT OUTBREAKS ON C SEQUESTRATION

Two of Canada's most damaging insects will be modeled: spruce budworm (SBW) and forest tent caterpillar (FTC, *Malacosoma disstria* Hubner). Spruce budworm is the most damaging pest nationally; at the peak of its last outbreak, it caused losses of 44.5 million m^3/yr, or 41% of total losses (Sterner and Davidson 1982, Hall and Moody 1994). Consecutive-year analysis showed that, of the 96 and 81 million ha defoliated by spruce budworm and forest tent caterpillar over the 1980-1996 period (Figure 1), 9 and 21% were defoliated for more than 6 and 3-4 consecutive years, respectively (Simpson and Coy 1999). Substantial background work has already been carried out in terms of developing spatially explicit, climate-dependent models of SBW defoliation regimes in Ontario (Candau et al.

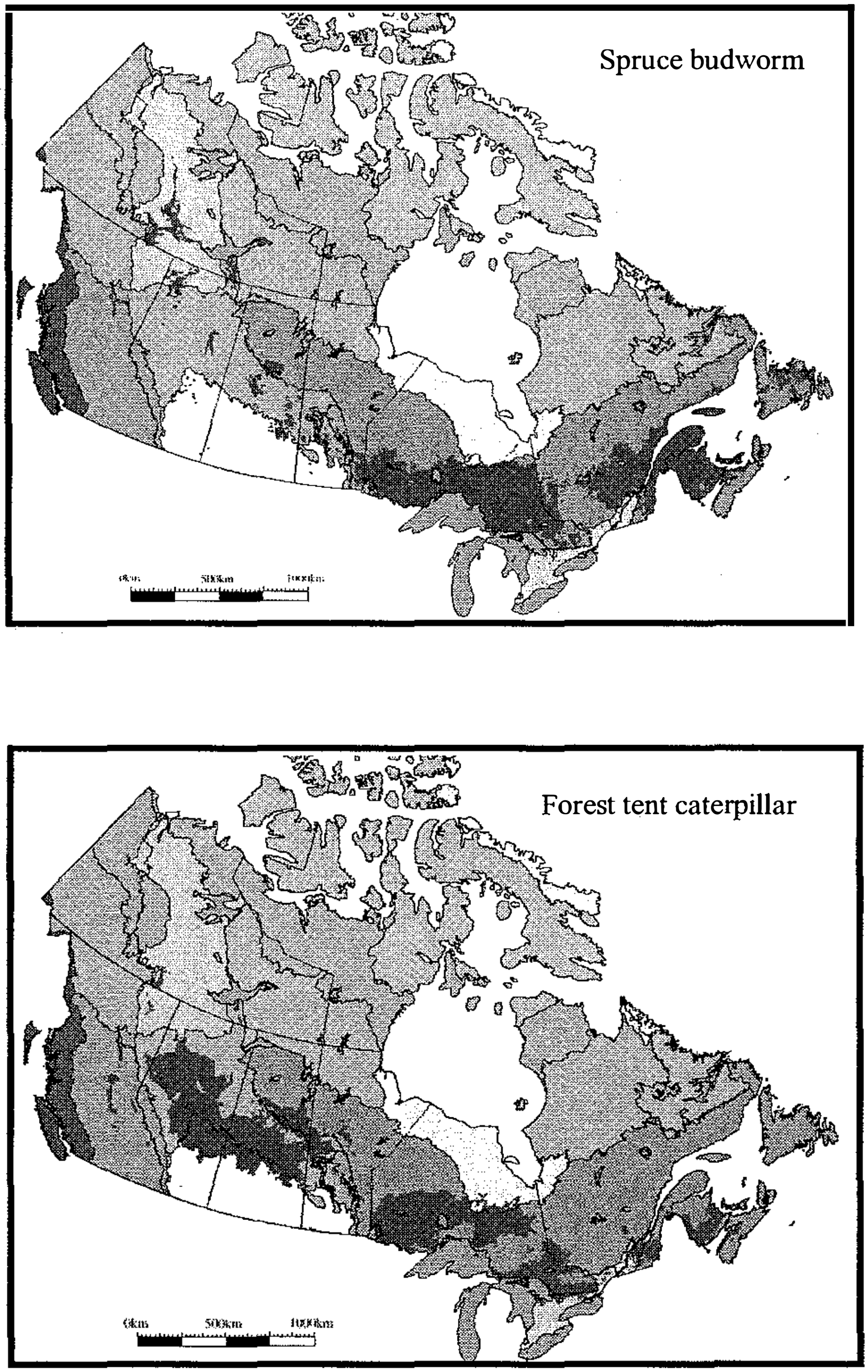

Figure 1. Extent of spruce budworm and forest tent caterpillar outbreaks in Canada, from 1980-1996 (shown in darkest gray, excluding BC coast; from Simpson and Coy 1999). Outbreaks of these insects encompass parts of nearly all vegetations zones in Canada (other shades).

1998, Fleming and Candau 1998) and Quebec (Gray et al. 1997, 1998, Gray and Boulet 2000) and developing a DSS for SBW (MacLean et al. 1997, 2000a,b) that can forecast total wood volume losses, given a series of annual defoliation intensities as input. Forest tent caterpillar has major effects on the boreal forest/aspen parkland ecotone (an area that will probably see the most rapid climate change effects) (Volney and Fleming 2000).

The focus here is on SBW as a case study/prototype for four reasons:

1. there have been SBW outbreaks in forests of all provinces and territories,
2. it causes over half of the average annual total of forest losses to insects in Canada (amounting to 1.5 times the losses to fire),
3. it has been studied more intensively and surveyed more extensively than any other insect in Canada, and
4. climate-dependent models of SBW defoliation and the DSS are already under development.

In progress to date, historical records of SBW and FTC defoliation in Ontario, Quebec, and the Prairie provinces, originally stored in different databases, have been merged. Characteristics of outbreaks such as duration and severity have been described in a spatial and temporal context. Plausible future SBW outbreak scenarios as influenced by climate change are being developed. Gray and Boulet (2000) determined a correlation between climate (temperature), forest habitat type, and budworm outbreak scenarios for Quebec, and similar relationships for New Brunswick are under development. In this development, temperature and precipitation are considered important factors in determining outbreak patterns of SBW and FTC within specific stand types. For example, an increase in temperature is predicted to cause a small, but statistically significant, increase in outbreak duration of the spruce budworm in balsam fir/paper birch stands in eastern Quebec, and a small, but statistically significant, decrease in outbreak duration in spruce stands (Gray and Boulet 2000; Fig. 2). An increase in forest concentration has also been shown to increase mean outbreak duration in fir stands (Gray and Boulet 2000; Fig. 3). The question to be addressed is: "What effect will an *x* degree increase in temperature have on frequency, duration, location, and severity of budworm

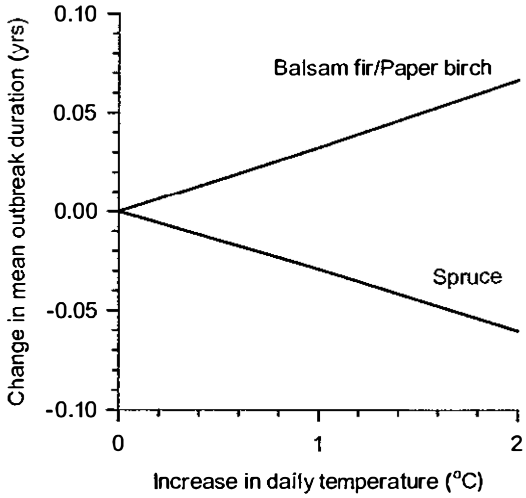

Figure 2. Effect of an increase in mean daily temperatur on mean duration of spruce budworm outbrea in balsam fir/ paper birch and spruce habitat types in Quebec (from Gray and Boulet 2000)

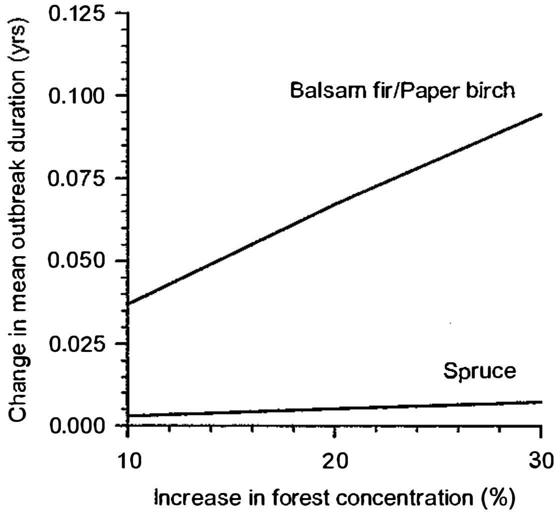

Figure 3. Effect of an increase in forest concentratio on mean duration of spruce budworm outbreaks in balsam fir/ paper birch and spruce habitat types in Quebec (from Gray and Boulet 2000)

outbreaks?" Data exist to help determine effects on duration and severity, while frequency and location will be evaluated using scenario planning analyses.

Methods for forecasting effects of insect outbreaks on timber volume development are being scaled up to the regional/national level and interpreted in terms of C dynamics. The Pest Management Decision Support System program of the Canadian Forest Service (CFS) provided the basis for this (Shore and MacLean 1996). The SBW DSS was developed at the CFS's Atlantic Forestry Centre (AFC) from 1992-96, with over $450K of Green Plan funding. It uses stand- and landscape-level prediction and interpretation models, the ARC/INFO GIS, and an ArcView graphical user interface. In partnership with the provinces and forest industry, and with funding from both, the SBW DSS has been operationally implemented for all forest land in New Brunswick, and on test landbases in Quebec, Ontario, and Alberta. The existing SBW DSS determines volume change resulting from growth reduction and tree mortality caused by defoliation. A user can specify alternative defoliation scenarios (as a series of annual defoliation values for a sequence of years). The current project will develop the means to forecast future SBW and FTC defoliation patterns in a changed climate, and determine methodology to translate timber volume into biomass and thereby to C. The SBW DSS spatially and temporally combines:

1. climate-dependent outbreak regimes of SBW and FTC;
2. spatial forest inventory data;
3. models of forest growth and succession;
4. foliage protection program (pest management) options; and
5. silvicultural options to predict the changes to live fiber biomass in the forest.

The SBW DSS will require "scaling up" (Fleming et al. 1999) to National Forest Inventory-type grid cells (Gray and Power 1997) and adaptation for FTC. Defoliation sequences will be input into the SBW DSS to estimate wood volume losses. Scenario planning ("what if?") analyses will be used to examine effects of climate change on pest outbreak patterns, and thereby on growth and mortality at a large regional scale, as well as potential for alteration by management activities and effects on C sequestration. Scenarios will include salvage harvesting to alter the conversion rate from the forest biomass pool to forest products, insecticide protection to minimize mortality and growth reduction, and forest restructuring to modify outbreak scenarios.

Biomass predictions from the above scenarios will be input to the C Budget Model of the Canadian Forest Sector (CBM-CFS2) (Kurz and Apps 1999). Data on area, volume, and forest type (hardwood, softwood and mixedwood) disturbed (consumed especially), and rules (age or site preferences) by which to spatially allocate mortality and disturbance will be provided for input into CBM-CFS2. The present CBM-CFS2 model operates with distributions (forest stand characteristics, disturbances) within very large spatial units (intersection of administrative and ecoclimatic boundaries). Our study will improve impact conversion factors (mortality rates, presently modeled as stand replacing disturbance) in CBM-CFS2 (Kurz and Apps 1999, Table 5). These conversion factors (mortality tables) in CBM hide complex relationships, and the conversion of area defoliated to area killed can be improved. Further, growth reduction is not accounted for. The model STAMAN (Erdle and MacLean 1999) is being used to determine volume loss estimation factors.

In predicting defoliation patterns under climate change, three time frames are relevant. Over the short term, insects will probably respond to changes in climate before the forest can adapt. Over the longest time duration, equilibrium between insects and the new forest (under a new climate regime) will develop. In the intermediate, medium-time duration, however, insect outbreaks may devastate the forest, which will have difficulty regenerating. Effects along ecotones may be particularly significant in this regard, and may well precipitate changes in forest types. FTC outbreaks in the Prairies/forest ecotone area are thus of particular interest.

"Secondary" effects of pest outbreaks, such as stand opening warming soils and increasing decomposition rates, may have a huge (albeit temporary) effect on C, but no data are available and estimation is beyond the scope of our study. Sensitivity analyses would be relevant here, whereby effects of a plausible range of decomposition rate changes could be evaluated. This would create a framework for data use, and provide justification for future study.

INTERACTIONS BETWEEN INSECT OUTBREAKS AND FIRE

The interaction of insect outbreaks with fire under climate change is also important. If insect-killed stands are prone to fire (Furyaev et al. 1983, Stocks 1987) and climate warming increases insect outbreak frequency, then the increased occurrence of insect-killed stands as the climate warms may lead to increases in fire frequency (Fleming 2000). The effects of these interactions may substantially exceed those due to any direct effects of climate warming on fire-disturbance regimes. While fires are more intense disturbances, their frequency and especially their spatial extent are quite limited compared to SBW outbreaks (Fleming et al. 2000b, Fig. 8.7). Budworm outbreaks encompassed moderate-severe defoliation of 58 million hectares of forest in Canada in 1975. This is a dramatic increase from the 10 and 25 million hectares defoliated in outbreaks of the 1910s and 1940s, respectively (Kettela 1983).

There is ample reason to believe that fire and insect outbreak disturbance regimes do interact. Increased host tree stress (e.g., from chronic defoliation or greater insect outbreak frequencies) may accelerate host tree mortality, and greater tree mortality adds to the fuel load and ultimately leads to more frequent and more intense forest fires. An Ontario study of fire in SBW-killed balsam fir showed that such stands have significantly higher spring fire potential for many years after an outbreak (Stocks 1987). In contrast, during summer, SBW defoliation opens up the canopy and lets understory vegetation proliferate, reducing fire potential during this period. The moist green layer of understory vegetation inhibits fire spread by isolating the dry surface fuels from the crown fuels. After about 5 years of crown breakage and windthrow, however, so much dead and downed material has accumulated on the ground that the surface fuel can overcome the dampening effect of the understory vegetation. The potential for summer fires peaks between 5 and 8 years after stand mortality. After that, the potential for forest fires begins to gradually decline as the understory vegetation continues to proliferate and the dead and down surface fuel begins to decompose and maintain moisture. In summary, the best available information suggests that the interaction between forest fire activity and SBW outbreaks is complex, is critically important to understanding and estimating past, present, and future C storage rates in Canada=s boreal forest, and is in dire need of quantification (Fleming et al. 2000a).

CARBON DYNAMICS AND NATURAL DISTURBANCE

Carbon dynamics within Canada's forest ecosystem are strongly influenced by species composition and age class structure. Natural disturbances, such as fire and insect outbreaks, alter the species composition, age class structure, and patch size of the forest in which they occur. Thus, predictions of natural disturbances, and their impacts, are important factors in estimating C dynamics. Insect outbreaks are major disturbance agents in Canada's boreal forest, and frequency, duration, and severity of outbreaks are all likely to be affected by climate change.

In broad terms, the amount of C stored by a forest ecosystem is the difference between the amount produced through photosynthesis and the amount lost through total ecosystem respiration (including plant, animal, and microbial respiration). Insects such as SBW decrease C storage over vast areas by reducing photosynthesis through defoliation and by increasing total ecosystem respiration through their own respiration and through acceleration of microbial respiration. The latter occurs during decomposition in the soil as SBW larvae convert eaten foliage into frass, clip off and kill other needles that would normally stay on the tree for up to 8 years, and during heavy infestations, cause widespread top-kill and tree mortality. Tree mortality during uncontrolled SBW outbreaks typically has averaged 85% in mature balsam fir and 15-40% in immature fir or spruce and mature spruce stands (MacLean 1980). In addition, defoliation opens up the canopy and alters ground vegetation and probably soil temperature relations (Fleming 2000). Intense fire has a much more spectacular effect by almost immediately eliminating photosynthesis and releasing large amounts of C directly to the atmosphere. After a forest fire, total ecosystem respiration may exceed the accumulation of C through plant re-growth for some time. To some extent, the same can also be expected after a SBW outbreak.

A key question is how climate warming will affect the frequency, extent, and intensity of both insect outbreaks and forest fires. Because forests accumulate biomass (C) as they age, and because increasing the frequency of (stand-replacing) disturbance results in younger stands, it follows that the greater the disturbance frequency, the lower the amount of C stored in forest biomass, and the greater the accumulated amount released into the atmosphere where it may accelerate climate warming (Fleming 2000). Changes in these disturbance regimes, which may be significantly influenced by human activities including modification of climate, are critically important in determining the net atmospheric C exchange (Kurz et al. 1995).

Carbon balances and dynamics are significantly different in healthy forests than in dead and severely damaged forests. However, insect-caused mortality is a major stand-replacing event. Thus, overmature forests that represent a large, but diminishing C pool may be replaced by young, rapidly growing forests that represent a small, but growing C pool.

In addition to direct effects on C during insect outbreaks, C loss following the infestation is perhaps a large source of CO_2, but no direct measurements are available. The large spatial scale of budworm outbreaks results in even minor post-outbreak changes having large implications to C budgets and sequestration. Data from the BOREAS and Boreal Forest Transect Case (BFTCS) studies provide a baseline for C fluxes in different boreal forest types without disturbance. What is needed urgently is a measure of post-disturbance fluxes that can be used to scale the BOREAS and BFTCS measurements up to the boreal forest as a whole and include fire and insects.

MANAGEMENT / POLICY IMPLICATIONS

Forest management is one way of ameliorating the impact of insect outbreaks. The concentration of forest stands within an area was shown to be a significant factor in determining the duration of spruce budworm outbreaks (Gray and Boulet 2000; Fig. 3); altering species composition will affect duration (Fig. 2 and Fig. 3); and pest management will limit tree mortality and growth loss (Fig. 4).

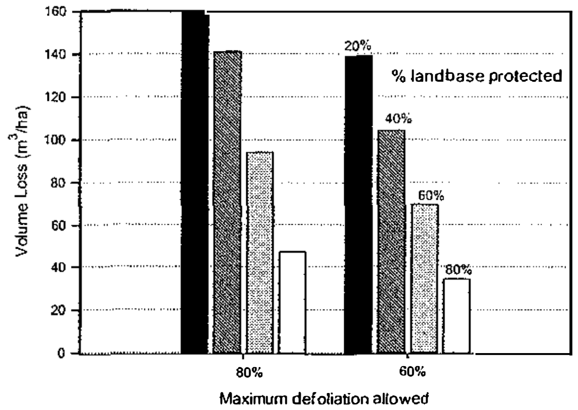

Figure 4. Volume loss from simulated spruce budworm outbreaks on Crown License 6 in New Brunswick, under varying protection (using the insecticide B.t.) regimes. Scenarios depict two protection levels (maximum defoliation allowed as 60% or 80% of current-year foliage) and four proportions of the landbase protected (20% to 80%).

There is the possibility that climate change may result in longer or more severe insect outbreaks, which would have implications for the annual allowable cut and the economy. A larger number of trees under stress may have a positive influence on insect outbreaks. The interrelationship with fire also comes into play: climate change increases both fire and insect outbreaks, while insect outbreaks also increase fire severity (more positive feedback). Management via afforestation and reforestation offers potential for increasing sequestration of C. Insecticide protection programs could benefit both C sequestration and forest productivity, whereas the lack of protection from insect attack could exacerbate CO_2 transfer to the atmosphere. Given the large amount of C tied up in mature forests, investment in better forest fire and insect protection for existing forests might well make much more economic sense than planting new forests to sequester C. Scenario planning analyses (Schoemaker 1995) can help to convey the consequences of alternative approaches to decision makers.

PROGRESS TO DATE

The project is still underway. We have obtained access to the National Forest Inventory (CanFI) database for all of Canada. Preliminary analysis of CanFI data is complete. Assembly of historical records of SBW defoliation in Canada (1960 to present) in GIS (ARC/INFO) format is also complete, as is assembly of historical records of FTC defoliation in the Prairie provinces, Ontario and the Atlantic region. Required modifications to the SBW DSS have been determined and are underway. Impact and defoliation information on SBW feeding on northern white spruce (*Picea glauca* (Moench) Voss) stands (Volney, unpublished data) has been acquired and set up of regression and classification trees to explore the climate dependencies of defoliation in our database has begun (Candau and Fleming, personal communication). There are indications of both temperature and precipitation dependencies - season may also be important.

CONCLUSIONS

This project will: improve the scientific understanding of the role of insect disturbance regimes in determining the C sequestration potential of Canada's forests; assess and develop forest management options and strategies for reducing insect disturbance losses on C sequestration; and provide information and recommendations for the development of Canada's international negotiating position on forest sinks as specified by the Kyoto Protocol.

Expected deliverables from the project include:

1. improved understanding and reduced uncertainties regarding the effects of insect disturbances on the sequestration of atmospheric C by Canada's forest ecosystems;
2. recommendations for forest managers for optimizing C sequestration by reducing insect losses;
3. experimental data- and knowledge-bases on the trends in area and extent of insect disturbances and the release of C to the atmosphere by these disturbances for Canada's major forest ecosystems;
4. improved ability to project the impact of climate change on insects and forest C sequestration potential; and
5. advice and recommendations for Canada's international negotiating position on forest sinks and development of a forest sinks component of Canada's national action program on climate change.

ACKNOWLEDGEMENTS

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THE BOREAL ECOSYSTEM RESEARCH AND MONITORING SITES (BERMS) INITIATIVE: SCIENTIFIC ACCOMPLISHMENTS

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INTRODUCTION

The BERMS program is a Canadian research initiative dedicated to the study of the cycling of carbon, water, and energy in Canadian boreal forest ecosystems.

The core of BERMS is to provide long-term, continuous climatological and flux data (heat, water, and carbon) from target ecosystems in Canada. Such data will be used to study carbon and water cycling in forests and to parameterize and test models of ecosystem carbon and energy exchange in a multi-year perspective. BERMS will also examine the role of interannual climate variation and site disturbance on forest-atmosphere interactions (water and energy) and carbon sequestration (seasonal and annual basis). As well, the BERMS database serves as a test bed for the development and testing of Land Surface Parameterization (LSP) schemes, specifically the Canadian Land Surface Scheme (CLASS).

Three tower flux sites are being used in the program. They are located in south-central Saskatchewan and were part of the Boreal Ecosystem-Atmosphere Study (BOREAS) program. Climate and flux data are being monitored at these three sites situated within pure, mature forest stands: the old aspen (OA) site of *Populus tremuloides*, the old black spruce (OBS) site of *Picea mariana*, and the old jack pine (OJP) site of *Pinus banksiana*.

OBJECTIVES

The objectives of this study can be summarized as follows:

1. measure climate variables and fluxes at the stand level over complete annual cycles
2. use the data to develop and validate process models of the carbon and water cycles
3. scale fluxes to the regional level, using remote sensing where possible
4. use models from #2 to estimate the impact of climate change on the carbon and water cycles.

PARTICIPANTS AND THEIR ROLES

Meteorological Service of Canada

Alan Barr, Barry Goodison, Joe Eley, Charmaine Hrynkiw and Craig Smith
- interannual climatic variability
- carbon, water, and energy fluxes
- data management, Web page development
- program management
- carbon, water and energy fluxes, southern old black spruce (A. Barr)

Diana Verseghy
- Development of CLASS
- BERMS sites as a test bed for CLASS algorithm development

Canadian Forest Service

Ted Hogg
- Forest ecosystem carbon budget modelling
- Climate variability and aspen decline

Brian Amiro
- Carbon, water and energy fluxes following disturbances

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- Ecosystem scale carbon modelling

Bob Stewart
- Program management

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Tim Griffis
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Harry McCaughey
- Carbon, water and energy fluxes, southern old jack pine
- Forest ecophysiology

Paul Bartlett
- Development of CLASS model
- Regional estimates of energy fluxes

Willy Cheng
- Soil respiration

Parks Canada

Norm Stolle
- Program management
- Maintenance of infrastructure and regular site visits

University of Toronto

Jing Chen
- Ecosystem scale carbon modelling

University of Wisconsin

Tom Gower
- Below-ground carbon dynamics

Erica Howard
- Carbon budgets in a logged chronosequence

SCIENCE ISSUES AND PROGRAM GOALS

The scientific goals of the program can be summarized as follows:

1. to understand the sequestration of carbon by the boreal forest in relation to vegetation type and climate
2. to characterize interannual variability of carbon and water exchange of the chosen sites
3. to establish experimental programs to examine the impact of disturbance (harvesting and fire) on the carbon and water balances of the sites, and to track the recovery of the sites following the initial disturbance
4. to resolve the net ecosystem exchange into its components; in particular to measure the flux of carbon from the understory and soil layers
5. to resolve uncertainties in eddy covariance measures of flux under low wind speed conditions, especially at night
6. to resolve the complexities of in-canopy CO_2 storage in the soil and canopy air space
7. to provide high-quality climate and flux data for the further development of the evaporation routine in CLASS
8. to provide data for the future inclusion of a carbon model into CLASS

TOWER SITE LOCATIONS

The sites are relatively close to each other (Figure 1), especially in the case of the OJP and OBS, facilitating the routine servicing of the sites. Regular site visits are conducted by Parks Canada staff from Prince Albert Park, local contractors, and maintenance visits are conducted by staff from the Meteorological Service of Canada (Saskatoon), UBC, and Queen's University on an "as needed basis".

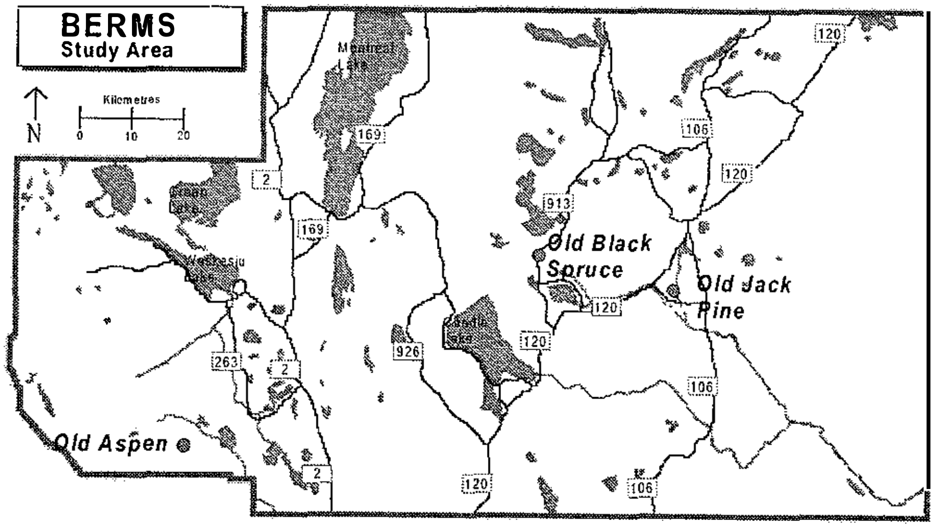

Figure 1. Locations of BERMS sites in Saskatchewan.

MEASUREMENT PROTOCOL

At the start of the program, it was decided to standardize the measurement protocol at all sites to facilitate comparison of conditions between the sites. We believe this is an essential aspect of the work. All fluxes are being measured continuously by eddy covariance with identical treatment for calculation of final flux values. Similarly, the supporting climate data (radiation fluxes, temperature and humidity profiles, wind speed and direction, precipitation, soil heat flux and soil moisture) are done in the same way. Leaf area index is sampled on a regular basis.

Flux data above the canopy have been collected at OA since the end of BOREAS in 1997, and measurement was started at OBS in April 1999 and at OJP in August 1999. Profile CO_2 storage was measured on a continuous basis (summer only) at OA and OBS in 1999 and will be added to OJP in summer 2000. A major effort is now planned to add semi-continuous soil respiration at all sites in summer 2000 using automatic chambers with a minimum of six chambers per site.

RESULTS

Figures 2, 3, and 4 illustrate the seasonal behaviour of the fluxes from the three BERMS sites in 1999 in terms of five-day average values. The extent of the comparison is limited because of the different lengths of time that the flux data are available at each site. However, there are several interesting aspects to note.

In terms of sensible heat exchange (Figure 2), in the late winter and spring the flux from the aspen is substantial. The black spruce and aspen sites are similar from April to July after which the flux from the aspen is much lower, often going negative, while those from both conifer sites remain substantial and remarkably similar.

Once leafout occurs in late April at the aspen site, the evaporative flux increases rapidly through May and is approximately twice the size of the comparable flux from the black spruce through the summer period (Figure 3). The conifer sites have very similar values through August and September, and in the late fall and early winter there are no discernible differences in evaporation between any of the sites.

The behaviour of the net carbon flux data demonstrates the dominance of the aspen as a carbon sink during the summer (Figure 4). However, it also loses a substantial amount of carbon in the shoulder seasons. In the spring, following thaw but before leafout, there is a bulge of carbon efflux, and there is a similar pattern in the fall following leaf senescence and before freeze-up. In contrast, the black spruce site has much more modest carbon uptake in the summer, often only one-third of that from the deciduous canopy. However, the

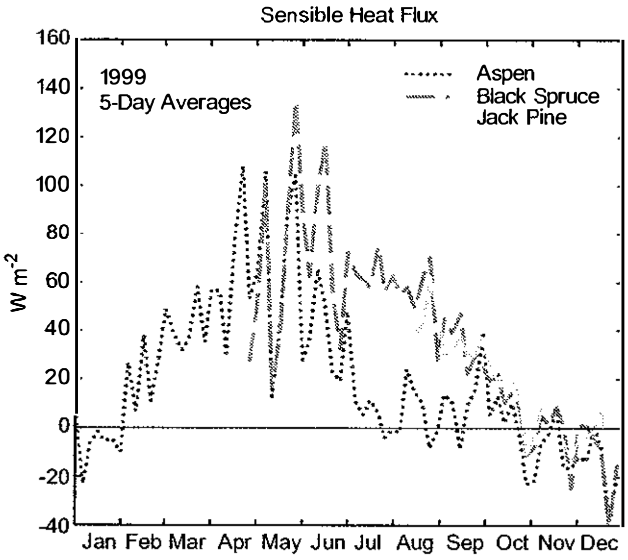

Figure 2. Sensible heat flux from the BERMS sites in 1999. Data are 5-day averages.

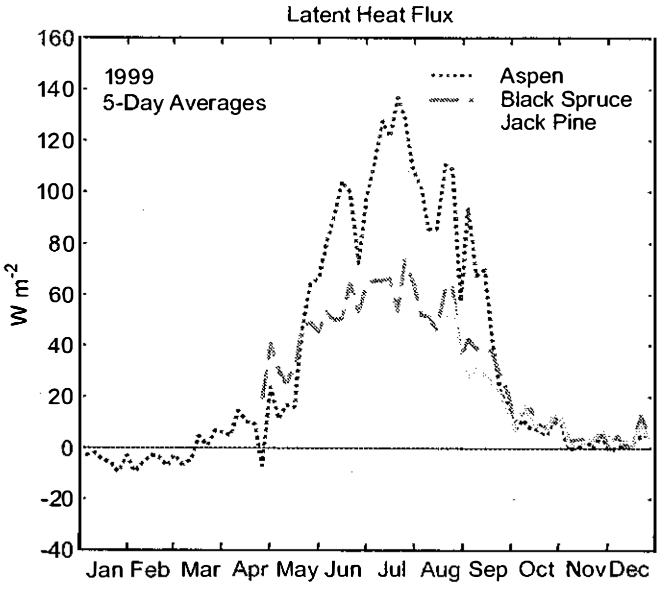

Figure 3. Latent heat flux from the BERMS sites in 1999. Data are 5-day averages.

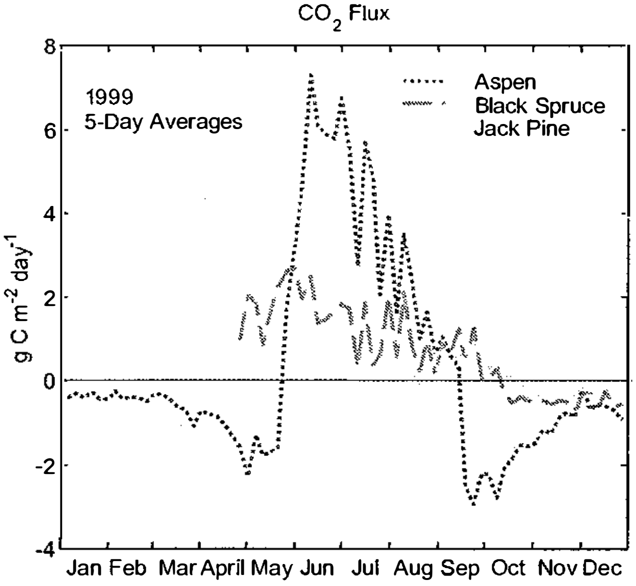

Figure 4. Net flux of carbon dioxide from the BERMS sites in 1999. Data are 5-day avera

conifer site maintains carbon uptake at least one month more than the aspen, even if at low levels. There is a notable similarity between the pattern of the net carbon fluxes from both conifer sites.

The comparison (Figure 5) of the daily net carbon fluxes from the aspen site and the old black spruce site in both experimental years of BOREAS (1994 and 1996) and the BERMS data from 1999 clearly demonstrates the relative behaviour of carbon exchange. The summertime uptake for the aspen is at least double that for black spruce, but the aspen loses more carbon in the fall and winter.

CARBON FLUX FOLLOWING DISTURBANCE

An example of the effect of site disturbance on the short-term carbon balance is provided by the results of an experiment to measure the net carbon dioxide flux from paired towers over a 10-year-old, burned, mixed forest and a nearby, undisturbed forest near Prince Albert National Park (Amiro, personal communication). The eddy covariance technique was used to measure the fluxes over a one-week period in 1998 and 1999. In Figure 6 the combined results of 30-minute averages with their standard errors are shown (here a negative value indicates a sink and a positive value an efflux from the surface). The 10-year-old burned site had values of sensible and latent heat, and soil heat flux that were similar to the undisturbed surface (not shown). The carbon dioxide flux on both surfaces are quite similar and there was no significant difference in the daily totals of carbon exchange (-1.3 g carbon m^{-2} day^{-1} at mature site and –2.9 g carbon m^{-2} day^{-1} at 10-year-old burn site). There are small differences in the diurnal amplitude of the flux, being slightly lower for the burned surface. These data indicate that most of the change in the carbon balance occurs in the first ten years following disturbance, but the evolution of the flux regime will require continuous data, and the most appropriate approach would be to follow the evolution from pre- to post-disturbance, one of the long-term aims of BERMS.

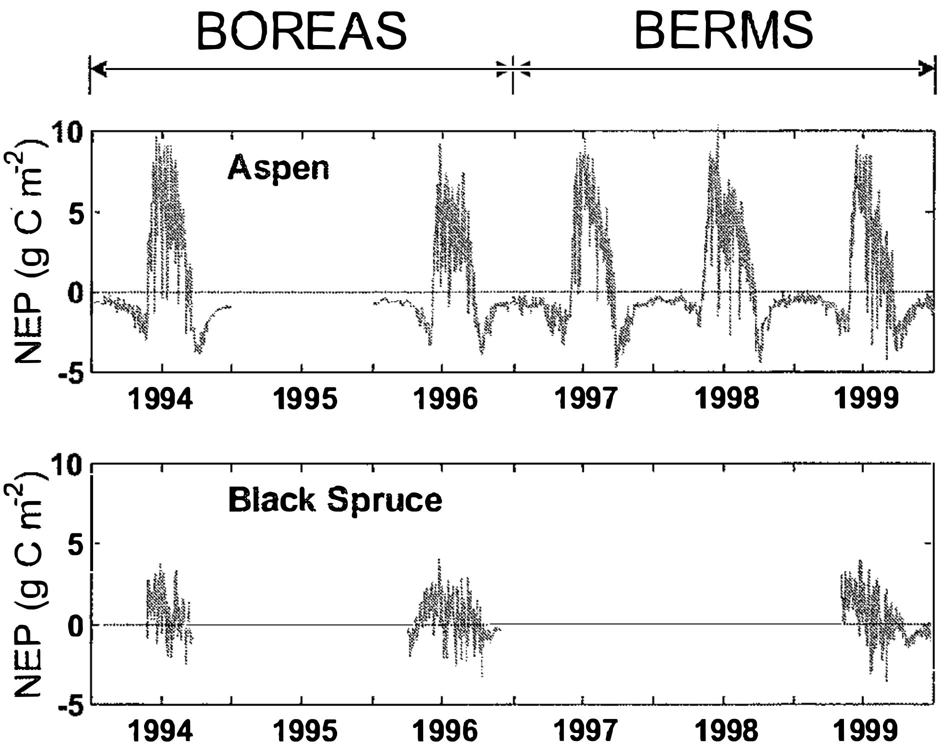

Figure 5. Comparison of net carbon fluxes from the aspen and old black spruce sites.

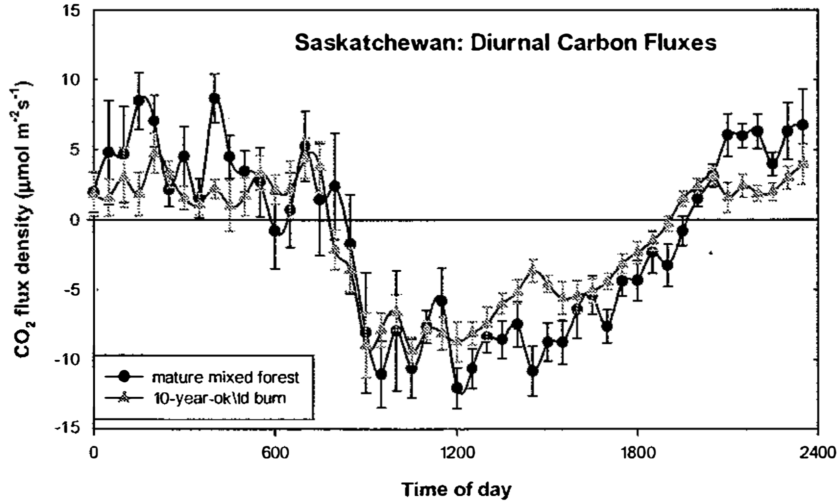

Figure 6. Diurnal pattern of carbon dioxide fluxes from mature forest and a 10-year-old burn near Prince Albert National Park. Data collected in one one-week period in the summer of 1988 and 1989.

FUTURE DIRECTIONS OF RESEARCH

The future work in the BERMS Project is summarized in the following points:

1. continue continuous measurements at all sites for several more years in order to develop robust data sets that include the full range of climate drivers (e.g., precipitation, temperature, and soil moisture)
2. include semi-continuous CO_2 profiles (summer only) at all sites, starting in 2000
3. add semi-continuous soil respiration measurements (summer only) at all sites using dynamic chambers, starting in 2000
4. continue to examine the issue of the lack of full energy balance closure at all sites
   – is it an issue of underestimation of fluxes by eddy covariance?
   – are the minor terms of the energy adequately measured?
5. Continue to examine the problems of flux underestimation by eddy covariance at night under low wind speeds
   – is it a measurement problem or is the carbon dioxide storage in the canopy and soil the issue?
6. continue to develop the CLASS model using BERMS data sets; the main focus will be on the evaporation routine and the development of a carbon model
7. use the BERMS data sets to examine the issue of upscaling fluxes to the regional level; the CLASS model will be used in this work
8. examine the possibility of expanding the program to include the experimental examination of the impact of disturbance on water and carbon fluxes
   -this will involve twinning the existing towers with new towers over disturbed sites (harvested, burned, and insect-damaged)
   -we are considering the possibility that the new towers will be portable and will be moved around in order to cover the widest possible range of disturbance conditions
   -the establishment of new permanent towers over disturbed sites is not a feasible option.

CONTRIBUTION TO CLIMATE STABILITY VIA EXPANSION OF AZONAL BOREAL FORESTS IN THE UKRAINIAN CARPATHIANS

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ABSTRACT

The communist regime established in the western territories of the Ukraine after the Second World War subjected the Carpathian forests to disastrous overexploitation. Excessive felling decreased the vitality of forests considerably by changing their age structure, density, and forest stand productivity. The upper border of the Carpathian forests has crept to a lower elevation with the contraction of the azonal boreal forests of the Ukraine. Forests of the Carpathian Mountains have become less able to perform their shelter functions, including those of climate stabilization. Forest distribution in the mountains could be expanded via afforestation and reforestation of wasteland and highly eroded unwooded lands within the State Forest Fund, and marginal agricultural lands locally used as pastures.

The objective of this study was to assess the potential effects of the expansion of azonal boreal forests in the Carpathians on the carbon cycle and climate stability, and to estimate the policy implementation costs. The first step of the research was to consider a "storage option", where trees are planted for a period of 50 years. The study did not consider future wood-use and land-use after this period.

The results of the research have shown that afforestation and reforestation in the Carpathian Mountains for the single objective of carbon uptake would hardly be beneficial. However, we highly recommend expanding the wooded area of azonal boreal forests of the Ukraine for other purposes, such as soil and water protection. Thus, besides a certain contribution to climate stability, the afforestation program would significantly enhance the environmental role of the Carpathian forests. In future studies, we intend to employ a dynamic optimization model (an optimal control tool) to indicate optimal expansion of azonal boreal forests in the Carpathians.

INTRODUCTION

In the Ukraine, afforestation and forest regeneration are traditional environmental and forest policy measures. Efforts to establish and grow the Steppe forest, that began as far back as in the 18th century, have been highly successful. However, up to the 1920s the efforts of foresters concerning afforestation were isolated and episodic. Protection of lands by means of afforestation was considerably hampered by socio-economic conditions. Forestry was competing constantly with agriculture in the struggle for land, and agriculture got priority. Despite the fact that the revolution in 1917 resulted in the impoverishment and famine of the peasantry, conditions were created that were favourable for afforestation and reforestation. In 1922-1941, 686,000 hectares of forests were planted, excluding western regions of the Ukraine[1]. However, the areas cut and destroyed by the war were greater. After World War II afforestation and reforestation were considerably intensified, as was timber harvesting. Silvicultural operations also intensified over this period when, on average, more than 70,000 hectares per year were subject to silvicultural treatment. During the period between 1948-1980, 2.3 million hectares of forest cultures (seedlings) were converted into wooded areas. Gradually all the cutover areas were reforested and low-productivity agricultural lands partially afforested. The highest level of forest regeneration was reached in 1951-1955. After 1955 afforestation activities decreased, partially because of the suspension of the field protection afforestation program and also because the area of non-

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[1] The heavily forested lands mainly of Galich Rus were returned to the Ukraine after WWII. These comprise some 15% of the total area of the country.

forested lands decreased.

Later, methods of logging operations moved away from clear cutting and were partially replaced by selective and gradual cutting, reducing the area which required reforestation. Recent dynamics of afforestation and regeneration in the Ukraine are presented in Table 1. It was planned that the afforestation practices would continue. Afforestation and forest regeneration are also important environmental policy measures with respect to global concerns for climate change, considering that the additional carbon stored in the newly planted forests is a net addition to the terrestrial forest sink. Studies have also shown that for the excessively ploughed and sparsely wooded territory of the Ukraine, afforestation is one of the most reliable strategies for protection of land from erosion and environmental degradation. The focus should remain on adherence to the sustainability of the forest resources use and on optimizing the balance between arable, fodder and forest lands in the Ukraine's rural planning.

The proposed program of afforestation and reforestation in the Ukraine (Gensiruk and Nijnik, 1995) is aimed at afforestation of about 2 million hectares of low-productivity and waste lands and the creation of forest stands along rivers, canals, water bodies etc. The so-called afforestation potential of the Ukraine includes the following areas:

1. highlands not covered by forests in the Carpathian mountains;
2. previously productive lands in the Forest-Steppe zone that originally were covered by forests but, later on, were converted into lands of other categories including waste lands;
3. marginal agricultural lands mainly located in the Polissja, where agricultural activity is not economically efficient;
4. eroded and contaminated areas in various regions of the Ukraine;
5. lands on sands, slopes, along roads, around water basins etc., where it is wise to create nature protected belts and keep the areas under conservation;
6. zones around industrial agglomerations and lands that have been used in the mining industry for storage of wastes and which today are under recultivation.

It is expected that by planting trees on waste and marginal lands of the Ukraine, the forest area will increase by almost 20% and the environmental situation in the country will improve considerably. An afforestation program will diminish the threat of further degradation of the upper layer of fertile soil, stabilize microclimate conditions and the water balance of the rivers and increase the forest resource base of the country. Implementation of the afforestation program will also address the Kyoto Protocol and The Climate Change Convention (Nijnik, 2000).

The Kyoto Protocol to the United Nations Framework on Climate Change (1997) strengthened international response to the fulfilment of the emissions reduction targets. Together with other measures it encouraged countries to protect forests as carbon "sinks". Establishing forests on degraded and unwooded lands and improving "storage management" adds to the amount of carbon already stored in trees and soils of managed forests (IUC, UN Environmental Programme, 1997). Since it is difficult to meet the Kyoto targets through emissions reduction, afforestation and reforestation is becoming an important environmental policy measure with respect to carbon sequestration. Forests can be seen as "sinks" that absorb carbon from the air. Forests whose carbon flows are in balance act as "reservoirs".

Table 1. Reforestation and regeneration in 1970-1997, '000s ha

Type of reforestation	1970	1980	1985	1987	1990	1997
-planting and sowing	54.7	41.6	40.0	35.8	34.9	28.8
-natural regeneration	5.3	4.4	4.0	2.3	2.5	3.5
Total Reforestation	60.0	46.0	44.0	38.1	37.4	32.3

Source: The State Committee of Forestry (1998)

Carbon can be sequestered over the very long term in buildings, furniture, or other wood products until they either burn or decompose.

Since the Kyoto Protocol, afforestation of marginal agricultural lands, waste lands and unwooded lands has taken on an important role in the carbon dioxide reduction policy of such countries as the USA[2], Canada and some European countries. The potential for afforestation and reforestation in the Ukraine has been studied per forestry zones of the country, which were defined by means of a comprehensive forest zone classification system (Gensiruk and Nijnik, 1995)[3]. Particular attention, with respect to afforestation, has been paid to the Carpathian forests that occupy 41% of all the forested area of the country. These forests are productive, but suffered from overcutting during the period between the WWII and the early 1970's. Harvesting in some regions exceeded annual allowable norms by 2 to 2.5 times. Intensive exploitation of the forests has led to their considerable depletion with the contraction of the azonal boreal forests. This particular study addresses azonal boreal forests of the Carpathians. These forests exist as highland wooded belts between 800-900m and 1350-1500m above sea level and include the Chornogora, Gorgany, Chyvchutsky and Marmarsky mountains.

GROWTH FUNCTIONS OF SPRUCE IN THE CARPATHIANS

In this study, as a first step, the so-called storage option is considered. A one-time conversion of wastelands and grasslands into forest plantations was investigated (Sedjo et. al. 1995; Guy and Benowicz, 1998; Binkley and Van Kooten, 1994; Van Kooten et al., 1993; Van Kooten and Bulte, 2000). The rate of carbon sequestration positively correlates with the rate of tree growth. Therefore, spruce, as the fastest growing species in the boreal forests zone of the Carpathians, has been investigated.

Spruce will grow as monocultures or mixed forest types. These forests grow fast and are productive between 650-1200m above sea level (asl) on loamy and sandstone soils. Spruce is a shade tolerant tree, but it needs special soil, rich with minerals and high in humidity. A tree of the first class of quality of locality[4] is usually 30-40m high. Productivity of spruce forests is 650-900m^3 of timber per hectare around the age of 100-120 years. According to continuous observations (Gensiruk, Nijnik and Kopij, 1998), the best conditions for growing spruce are those at 750-1100m. Stock productivity of spruce timber is around 550m^3 per hectare at 50 years age. Annual average increment is about 5.33m^3 per hectare. The most acceptable index of forest productivity (stock of timber from 1 hectare) in the mature spruce forests of the Carpathian highlands is around 945m^3. However, spruce timber is not resistant to various injuries, insects and the consequent wind falls, which lower stand productivity. The highest elevation where spruce grows in the boreal Carpathian region is 1675m asl.

The estimation of productivity of the Carpathian spruce forests and the definition of optimal age of harvesting were conducted on the basis of the investigation of spruce plantations in the forestry enterprises. 32 experimental areas were chosen with differences in soil, growing conditions, forestry management etc. Results from this investigation showed that with increasing elevation above sea level, productivity of spruce forests decreases (Gensiruk, Nijnik and Kopij, 1998). These results also show that high productivity spruce forests usually grow fast until 50-60 years of age. However, after 60 years these forests become heavily damaged by various injuries, and growth rapidly decreases. This pattern of growth response is common for stands at 650-750m asl. At altitudes of 850-1100m asl the optimal conditions for growth of boreal forests could also be found. Spruce forests, on some sites at this elevation are fast growing, stable and resistant to various injuries and insects and have high enough timber stock. As elevation increases from 1150 m asl to 1450m asl

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[2] However, starting from the COP-6 Conference in the Hague and on, the current US administration has decelerated its efforts in climate change mitigation and in environmental protection.

[3] Spatial classification of forests has been developed as a separate project with the aim to improve spatial planning of forests in order to enhance sustainability of their use. A hierarchical method of multi-criteria classification, cluster analysis, was applied in the study.

[4] Class of quality of locality is officially recognised in the Ukraine as a measure of forest productivity. It is comparable with site index.

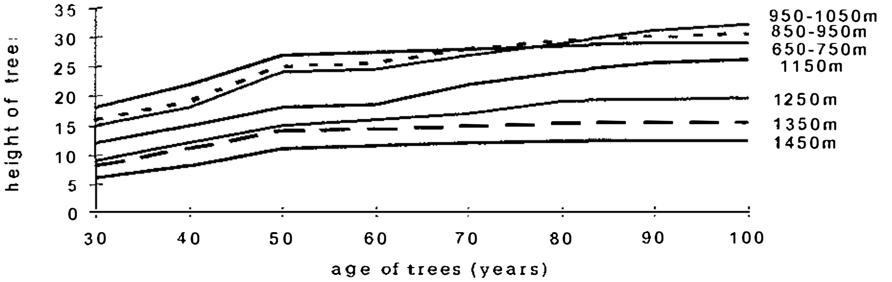

Figure 1. Dependence of forest productivity on elevation (asl).

highland spruce forest productivity declines (Figure 1).

Results from this investigation indicate that at 700-1150m above sea level boreal forest belts are comprised of highly productive spruce trees of 1st classes (1^a and 1^b) of quality of locality. Higher than 1200m above sea level, the productivity of highland forest decreases for each 100m. Thus, at 1250m above sea level, spruce forests usually have the 2nd class of quality of locality, at 1350m the 3rd, and at 1450m asl. the 4th class of quality of locality. The analysis also shows that during the first years tree growth is low, and after 5 years, trees reach a height of only 0.3-1.0m. The maximum rate of tree growth in the Carpathian boreal forest occurs around the age of 50 years and after that it slowly decreases (Table 2).

Table 2. Mean and current increments by classes of quality of locality (m^3/ha).

	Classes of quality of locality									
Age	1b		1a		1		2		3	
(years)	mean	current	mean	current	mean	current	mean	current	mean	current
30	13.0	19.4	9.2	14.8	6.5	11.1	4.3	7.7	2.7	5.1
40	14.5	19.0	10.8	15.5	7.9	12.3	5.6	9.5	3.8	7.3
50	14.7	15.5	11.4	13.6	8.6	11.5	6.4	9.6	4.6	7.6
60	14.3	12.3	11.4	11.4	8.9	10.1	6.8	8.8	5.1	7.3
70	13.7	9.9	11.1	9.5	8.9	8.7	6.9	7.8	5.3	6.7
80	13.0	8.0	10.7	7.7	8.7	7.3	6.9	6.8	5.4	6.0
90	12.2	6.3	10.2	6.1	8.4	6.1	6.8	5.8	5.4	5.3
100	11.5	4.9	9.6	4.8	8.0	4.9	6.6	4.7	5.3	4.3
110	10.8	3.6	9.1	3.6	7.6	3.7	6.3	3.7	5.1	3.3
120	10.1	2.5	8.6	2.6	7.2	2.7	6.0	2.6	4.8	2.2
max. growth	14.7		11.4		8.9		6.9		5.4	
Maturity age	52		61		69		78		89	

Source: Gensiruk, Nijnik and Kopij, 1998

The highest rate of tree growth is observed at 850-1050m asl. Trees usually approach 25m in height at 50 years of age. These are optimal conditions for the growth of spruce forests, and the lands at these altitudes are considered suitable for the expansion of forests in the Carpathians for the purpose of carbon uptake. The age of maturity of highland spruce forests correlates with the class of quality of locality. Usually the higher class of quality of locality corresponds to a lower age of maturity of the trees. Between 650-1250m asl forest stands of the 1^b class of quality of locality approach the age of maturity in 52 years. However at 1350m asl, the same trees with the 3rd class of quality of locality, approach the age of maturity in 89 years. Growth functions for different types of spruce stands in boreal forests, depending on the quality of locality, are shown in Figures 2 and 3.

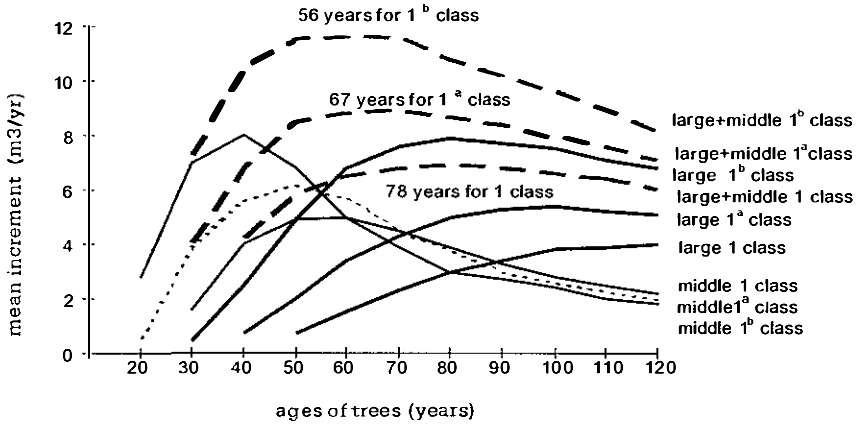

Figure 2. Growth functions of trees 1a, 1b and 1 classes of quality of locality of boreal spruce forest stands.

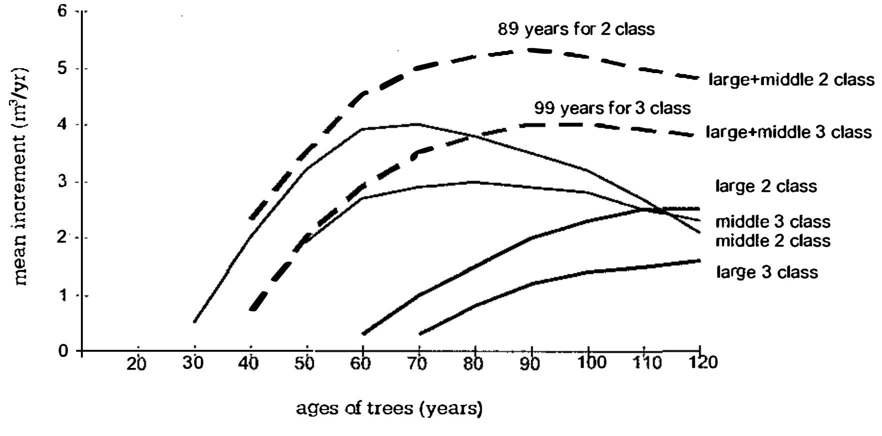

Figure 3. Growth functions of spruce trees in classes of quality of locality 2 and 3.

So, the age of maturity depends on the growing conditions of the forest stands, and these depend on the elevation above sea level and on soil. Soil and climate conditions are unfavourable for growth of boreal spruce forest stands at higher elevations, consequently the trees need more years to reach maturity (Figure 3).

Usually, the maturity of forest stands is used to estimate the harvesting age of a forest. Spruce stands in the Carpathian mountains reach maturity about 28 years sooner than spruce stands on average (Gensiruk, Nijnik and Kopij, 1998). In our study, we consider that highly productive spruce stands, reach a maximum rate of growth in 50 years. Benefits of carbon uptake, that were estimated in physical terms of carbon sequestration, depend on the tree growth functions just presented.

ASSESSING CARBON SEQUESTRATION POSSIBILITIES

Carbon that is taken from the atmosphere is stored in the stems, branches, leaves and roots of trees and in the understory, soil and litter. The bole continuously sequesters new carbon until the time of tree harvesting. This process is assumed to continue indefinitely with new generations of trees growing to replace old ones. The proportion of the above ground biomass of the tree that includes the branches and leaves only, is assumed to be equivalent to 0.57 of the bole volume (Guy and Benowicz, 1998). When trees are cut, the above ground biomass minus the commercial part of the bole that constitutes a log, enters the litter account. Later on, when a new generation of trees grow, there is regrowth of the non-bole biomass and the stem wood volume. Thus total above ground biomass that includes bole, branches and leaves can be treated simultaneously. Let η (=1.57) be a coefficient that translates bole biomass into total above ground biomass and (ϕ =0.207) a coefficient that converts tree growth into carbon. The total discounted carbon per hectare for the above ground biomass (Cg) (Van Kooten et al., 1993 and 2000) is then:

$$Cg = \frac{\eta\phi\left(\int_0^t V(s)e^{-rs}ds - V(t)\,e^{-rt}\right)}{1-e^{-rt}} \qquad [1]$$

where:

t = is rotation age;

s = growing age;

V = volume of stem wood, m^3;

r = rate of growth.

The first term represents carbon that accumulates during the growing stage and the second term accounts for carbon released to another account at harvest time. Dividing by $1-e^{-rt}$ we obtain the sum of the infinite series of returns that take place every t years.

Total discounted carbon per ha for the root account (C_r) (m^3) is:

$$C_r = \phi\int_0^t R\,(G)\,e^{-rs}\,ds \qquad [2]$$

(Van Kooten and Bulte, 2000),
where:

R = root biomass, m^3; (R=.2317Cg)

G = above ground biomass, m^3;

s = growing age;

r = rate of growth.

The constant 0.2317 relates root biomass to above ground biomass for softwoods (in this case spruce).

When land is converted into forest, certain changes occur in soil carbon. However, according to the Kyoto Protocol and its follow-up, a consensus on the methodology of quantifying carbon sequestration benefits has not been reached. The question, of which pools to count, or if one should consider only the commercial component of the tree, has not been answered (Greenhouse Challenge, 1997). Besides, data on soil carbon is hardly available. It is difficult to determine soil carbon associated with different uses of land. Therefore, as a first approximation, changes that occur in soil carbon after tree planting were not considered. The litter pool that consists of dead and dying biomass is a comparatively small pool (AACM International Pty Limited, 1998). Therefore, it also was not considered in the current research.

An important question that arises when using cost-benefit analysis is that of carbon discounting. If the value of the benefits from net emissions reduction were known, it would be easy to discount the expected benefits flow, the same way we discount the costs. However, the benefits of carbon uptake are not certain, and because of time preference, the majority of economists argue that future carbon reductions decrease in value rather rapidly. At high discount rates (10-20%), the value of any amount of carbon sequestered in some 40-50 years rapidly approaches zero. Therefore, when timing of carbon uptake is important, the forests that capture carbon rapidly have an advantage over those which do not (i.e. spruce stands in the Carpathians). This is because spruce stands can take up large quantities of carbon and consequently accumulate substantial biomass, but only over hundreds of years, rather than over a short period of time.

This study considers the case without discounting physical carbon. By not discounting carbon, we assume that the value of carbon damages in the future will increase at the real rate of discount. This suggests that all amounts of carbon sequestered are valued equally, no matter when it is captured. Many policy institutions, such as the Global Environmental Facility of the World Bank and the United Nations, also do not recommend discounting of physical carbon for environmental reasons. Therefore, this study considers the case without discounting physical carbon, while a rate of 4% is applied for discounting of the afforestation costs. Initial estimations of cumulative carbon uptake over a 50-years period by spruce on the highlands of the Carpathian Mountains are presented in Figure 4.

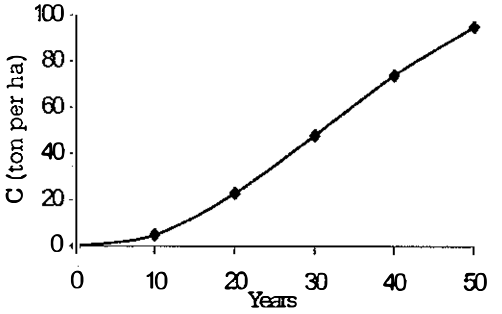

Figure 4. Cumulative carbon uptake by spruce on the highlands of the Carpathian Mountains

The figure indicates that in 50 years about 100 tons of carbon, equivalent to 367 tons of CO_2 emissions, are sequestered per ha of spruce forest in the Carpathians. The following section considers the costs of carbon uptake.

COST-BENEFIT ANALYSIS OF CARBON SEQUESTRATION

Lands of the Forest Fund of the Carpathian mountains suitable for reforestation could hardly be used for any other purpose than growing trees. Therefore, costs of afforestation of these lands are the direct costs of tree planting. Marginal agricultural lands could also be used for pasture. Thus, for marginal agricultural lands, the opportunity cost of afforestation (net returns associated with the current agricultural activity) and direct costs of tree planting were considered. It is impossible to evaluate market value of pasture, because there is no market for this type of land use and there are no prices involved. Thus, we based our estimates of net returns from pastures, on the productivity of lands and on prices that Ukrainian agricultural enterprises pay for similar cattle feeding. On this basis, the opportunity costs of lost pasture was estimated at about 7 Euro/ha. To estimate direct silvicultural expenses, detailed observations on the process of natural reproduction of spruce in the Carpathians were made at 75 experimental plots. Results of the investigation have shown that natural regeneration and growth of spruce is successful in the azonal boreal forests area because of fertile soils, proper climatic conditions, a sufficient level of precipitation and high humidity (Gensiruk, Nijnik and Kopij, 1998). This allows us to consider afforestation costs as the only direct costs of carbon sequestering, without taking into account any further expenses that might occur while trees are growing.

Carbon sequestration costs were computed over a single 50-year period, using a 4% discount rate for the opportunity costs of marginal agricultural lands. According to the estimates, the present value of carbon sequestration costs in the Carpathian highlands vary from some 2.5 Euro to 6.2 Euro per ton, while a willingness-to-pay estimate, per ton of carbon not released, is almost 5 times higher. The maximum carbon sequestration costs are for the lands that are currently used as pastures, because of the opportunity costs of land. In general, costs of afforestation, including both direct and indirect costs, are comparatively low in the Ukrainian Carpathians, because of good forest growing conditions and a cheap labor force. However, the present value of carbon uptake costs, is highly dependent on the discount rate that is employed.

In general, afforestation of waste and marginal agricultural lands is competitive with other carbon reduction options. However, on average, afforestation in the boreal forest zone of the Carpathians is only feasible in the area 100m above the present upper border of forests in the mountains. Thus, the magnitude of possible expansion of azonal boreal forests is not comparable with the territory of the country, and benefits from afforestation in the boreal forest zone of the Ukrainian Carpathians are small compared with the Ukrainian emissions of CO_2 into the atmosphere (about 106.2 million metric tons of energy-related carbon, US Energy Information Administration, 2000) .

However, afforestation and reforestation in the Ukraine, including the expansion of azonal boreal forests in the Carpathians, provide a long list of other economic and environmental benefits besides those associated with carbon sequestration. These benefits include the protection of lands from erosion; enhancing the sanitary-hygienic functions of forests; the development of recreation; additional wood production; possibility to substitute wood for fossil fuels that is very important in economic conditions of the country and should be investigated further; and receiving credits from the world's community for the afforestation program with the possibility to use this foreign currency to promote sustainability in forestry and in the Ukraine's economy in general.

On this list of benefits from afforestation, protecting the water and the soil is a particularly important function of forests under mountainous conditions. Forests decrease surface water flows, slow down the process of snow melt, allow water to penetrate deeper into the soil, stabilize the stocks of groundwater, decrease flooding and support river regimes. The water regulatory impact of forests is especially important for the Carpathian region, where the average annual amount of rainfall is 800-1100 mm and up to 1400-1600 mm in the mountains. During winter time, about 1000 m^3 of water per 1 ha is accumulated in the Carpathians. Forests retain snow, allowing it to melt slowly and control the rate of water flowing down into the valleys. The Carpathians annually evaporate almost 20 km^3 of water, providing moisture to the unwooded areas of Europe (Nijnik, 2000).

Afforestation also prevents the spread of erosion, which is a gradual and silent catastrophe. If erosion spreads further in the Ukraine, at its present rate it will cause considerable losses in productivity of 1/3 of arable lands (Gensiruk, 1992). Forests use a tremendous volume of ground water for transpiration. They decrease soil moisture and prevent erosion. Again, it is especially important for the Carpathian mountainous region, where erosion causes windfall and floods, which in recent years resulted in annual damages of over 200 million hryvna[5]. Considering the aforesaid, it is highly recommended to expand the wooded area in the Carpathians for soil and water protection. The multifunctional role of forests and their shelter functions raise considerably the benefits from afforestation measures. Afforestation in the boreal forests zone of the Carpathians, with species that are relatively fast growing and suitable for the particular conditions (such as spruce), will protect the environment and provide, at the same time, benefits with respect to carbon sequestration.

CONCLUSIONS AND DISCUSSION

The impact of greenhouse gases, such as carbon dioxide, methane, nitrous oxide and chlorofluorocarbons, on the environment result in climate change and in global warming. Forests remove carbon dioxide from the atmosphere and store it as carbon in terrestrial ecosystems. In this respect forests play an important role in climate stabilization. Growing trees act as sinks, absorbing carbon from the air. Mature forests, whose carbon flows are in balance, play the role of reservoirs. Thus, the forest sector offers opportunities for climate mitigation and, apart from economic and environmental benefits, the expanded wooded area puts forward an attractive economic alternative to reducing carbon emissions. Furthermore, in addition to enlarging carbon sequestration benefits of growing and mature forest stands, the expansion of wooded areas contributes to timber supply. Use of wood in products such as furniture and buildings, serves as a carbon sink for many years after timber harvesting.

Though it remains unclear if the carbon sink option will be on par with CO_2 emissions

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[5] 1 hryvna roughly corresponds to 0.2 Euro

reduction, we expect the intergovernmental decision will be positive. Therefore, the implementation of the proposed afforestation program in the Ukraine (Gensiruk and Nijnik, 1995) is promising. Expansion of the wooded area in the Ukraine is an important policy measure for the following reasons. These lands are available, and expanding the wooded area in a country where the land is available should be of interest to the world community. There exists the possibility of a trade between highly developed countries where carbon dioxide emissions are high and, for instance, the Ukraine if it implements the program of afforestation. The world community might be willing to pay for maintainence and expansion of Ukrainian forests for carbon sequestration. In this respect, it is very important to analyze the Ukraine's potential to mitigate climate change via creating the plantations, to assess stock and flows of carbon in Ukrainian forests and to estimate the required investments. An investigation of the economics of creating forest plantations for CO_2 sequestration in the Ukraine, including the Carpathians, is required as a starting point for substantiating and submitting a program of afforestation for consideration of the world community.

The afforestation program could become a key sustainable forest policy measure for the country, because the Ukraine could receive credits for planting trees. The country could acquire hard foreign currency and use the currency obtained from such a trade to develop its forest sector. The expansion of the Ukraine's wooded area is a rational forest policy proposal. The costs of afforestation, including both direct and indirect costs, are comparatively low in the Ukraine, because of the good forest growing conditions, especially in the Carpathians, and also because of a low-cost labor force.

Spruce should be selected for planting in the azonal boreal forests zone of the Carpathians, if the purpose is to enhance carbon sequestration, since it is the most productive and fast-growing species. However, the expansion of azonal boreal forests in the Ukraine is highly recommended mainly for purposes other than carbon uptake. The afforestation in the Carpathians will give a long list of economic and environmental benefits besides those from carbon sequestration. The objectives of the afforestation program in the mountains are environmental and include the prevention of erosion and the enhancement of the hydrological functions of the forests. To meet these environmental objectives, we recommend planting spruce as part of mixed stands. This is because planting spruce monocultures has proven to be unwise in the Carpathians. This practice was very popular during the period of Soviet industrialization, but caused disastrous windfalls in the mountains.

This research does not consider the question of what to do with the trees once they reach maturity (indeed, the commercialization of forests worldwide seems to pre-ordain the end result of a mature spruce forest). In this study, we evaluated a 50-year period and one time planting of trees, the so-called storage option. However, the results of this study have shown that it is important to determine what to do with the trees and land afterwards. This question will be the focus of our future research. In the future, we intend to employ a dynamic optimization model. Its objective will be to maximize the flow of the present and all future net benefits of the afforestation program in the Ukraine, including those of climate stabilization. The study will provide an indication of the optimal areas to plant trees in the country. The project will be carried out by forestry zones of the Ukraine, including the Carpathians and their azonal boreal area.

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STRATEGIES IN SWEDISH FORESTRY FOR REDUCING NET CO_2 EMISSIONS

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ABSTRACT

The aim of this report is to outline possible strategies in Swedish forestry to reduce greenhouse gas emissions. The three main recognised ways are substitution, sequestration, and conservation. Evaluation was carried out only with respect to biological potential with no socioeconomic considerations. Current accumulation of carbon (C) in forest biomass and soil was discussed. Specific strategies evaluated were use of harvest residues for fossil fuel replacement, Salix or silver birch plantations on set-aside farmland, intensified nitrogen (**N**) fertilisation, mitigated site preparation and drainage of wet land. The results showed that forestry might substantially contribute to reduce greenhouse gases in the atmosphere but that the effectiveness of a strategy strongly depends on the time frame. The accumulation of C in biomass and soil might, for several strategies, be initially large but in the long run marginal. Use of biomass for fossil fuel showed the opposite pattern, i.e. initially rather small but in the long-term substantial. It was emphasised that short-term solutions should not be preferred at the expense of more sustainable ways to reduce emissions.

INTRODUCTION

Forestry might play an important role in reduction of greenhouse-gas (GHG) emissions through its potential to produce biomass for fossil fuel substitution and by sequestering carbon (C) in standing biomass and soil organic matter including peat. Sweden has a land area of approximately 411 000 km^2 of which 233 000 km^2 (57%) is forest land, defined as land suitable for forest production and with a potential mean stem production > $1m^3$ ha^{-1} yr^{-1}. With a population of around 8 million inhabitants, Sweden has the second largest forest land per capita in Europe. Since Sweden's forest land area is so large, it should have great strategic potential to mitigate CO_2 emissions through forest land use and management.

The three main recognized ways for GHG reductions are i) sustainable use of biofuels as replacement for fossil fuels – *substitution*, ii) storage of C in biomass, humus and peat- *sequestration*, and iii) avoidance of strategies that increase emissions – *conservation*. Strategies are preferred that favour biomass harvest fossil fuel substitution over carbon sequestration. In the long term storing carbon in growing biomass may reduce the potential to use it for fossil fuel. Additional arguments for biomass harvesting are that i) cutting of wood is not necessarily a loss of a C stock – wood C might be stored elsewhere e.g. as building materials, ii) stock keeping of forest C pools is unsafe with respect to wild fires, iii) storage capacities are limited, and iv) it is economically more progressive to use wood as raw material - forming prerequisites for technical development.

The aim of this report is to outline possible strategies in forestry with respect to their biological potential without considering any technical or socioeconomic constraints. The main target is to assess the possibility to substitute fossil fuel with biomass fuel, however spin off effects on sequestration and conservation are considered.

METHODS

The impact of selected scenarios on GHG emissions was analysed for the time periods of 10, 30, 100 and 500 years. It was assumed that the dry weight of biomass is 0.4 ton m^{-3} (Eriksson 1991; Marklund 1988) and the C content of the biomass is 50% of it's dry weight (Eriksson 1991; Liski & Westman, 1995). It was further assumed that the energy production from 1 ton biomass dry matter corresponds to the energy from 0.42 M ton fuel oil (0.44 m^3) with a C content of ca. 0.4 ton. Consequently 1 ton biomass C can replace 0.8 ton oil C.

Decomposition of soil organic matter was determined using the Q-model (Ågren & Bosatta, 1996).

The net effect of each strategy was calculated as the sum of substituted fossil fuel C plus the effect on C sequestration in soil and biomass and emissions of N_2O and CH_4. The results were expressed as CO_2-C global warming potential where the effect of N_2O was set to 320 times, and CH_4 24 times, that of CO_2 (Lashof & Ahuja, 1990)

RESULTS AND DISCUSSION

Carbon sink in forest stands – business as usual

According to a review by Ståhl (1999) the present stock of whole-tree C amounts to 1000 M ton in Swedish forest land (23 M ha). This represents an accumulation of 400 M ton C since the 1920's, equal to about 6 M ton C year^{-1}. The accumulation was caused by the growth rate exceeding harvest rate. Factors potentially contributing to increased rates of growth are successful silviculture and atmospheric nitrogen (N) deposition. Annually sequestered C corresponds to 38% of the CO_2-C annually emitted amount through fossil fuel burning (16 M ton C). The stock-growth rate is considered to successively decline to zero over a 100-year period (Table 1).

Table 1. Carbon accumulation in biomass due to increasing stocks of forest biomass (M ton C year^{-1}).

Years	10	30	100	500
Forest biomass	5.7	5.1	3.0	0.6

Carbon sink in soils – business as usual

Results from the National Survey of Forest Soils and Vegetation indicate that the humus layer (O horizon) thickness is increasing in naturally drained forest land in Sweden, at a rate of 2–5 M ton C yr^{-1} over the last few decades (Eriksson et al., 1999). The biggest increase has occurred in southern Sweden, and the smallest in the north. Recent research indicates this rate may have been overestimated (Erik Karltun, 2001). Possible reasons for the accumulation are increased biomass production, N-deposition, afforestation of set-aside farmland, and lower frequencies of forest fires. However, conclusions are somewhat restricted due to the fact that only the O horizons were evaluated. Most of the soil C occurs in A and B horizons (Liski & Westman, 1995; de Wit & Kvindesland, 1995) and changes in these stocks might overshadow changes in the uppermost horizon. The current C sink in the growing O horizons in Sweden corresponds to 10--30% of the emissions that are caused by burning fossil fuels. The size of this sink will most likely gradually decline to zero over 500 years (Table 2). In Sweden around 2.5 M ha of wet and peaty forest land are drained. It is expected that these soils might emit CO_2 and N_2O due to enhanced decomposition of peat. The impact of the drainage on GHG emissions is described in section "drainage of wet land" below.

Table 2. Carbon accumulation in growing O horizons (M ton C year^{-1}) at naturally well drained sites.

Years	10	30	100	500
O horizons	3.5	3.4	3.2	1.8

Potential C sink due to use of harvest residues for substitution of fossil fuels

At the present harvest level (ca. 70 M m^3 year^{-1}) it is estimated that the amount of slash (e.g. branches, tops and needles) amounts to 9.9 M ton dry weight (d.w.) year^{-1} (Egnell et al., 1998). It is not technically or economically feasible to harvest all slash. At some vulnerable sites the slash should be left for ecological reasons. The estimate is therefore based on a situation where 70% of all branches and tops and 30% of the needles are harvest. The C content in the slash corresponds to 4.9 M ton C year^{-1}. If used as biofuel it can substitute for the emission of 3.9 M ton C year^{-1} (Table 3) from oil. This represents the maximal potential under present forest management and harvest regimes. The effects on emissions will not decline over time. The removal of forest residues may induce negative consequences such as biological acidification and nutrient losses. Thus, the removal of forest residues has been shown to decrease forest production by 2-5%, mainly as a result of N losses (Egnell et al., 1998). This in turn means reduced C accumulation in the biomass and in the soil. Biological acidification is mitigated through liming or recycling of the ash from biofuels. The N losses are compensated for either via excess N through atmospheric deposition or by fertilisation..

The removal of slash will decrease the amount of litter supplied to the soil. This will temporarily reduce the soil humus content. The main part of C sequestered to the soil will eventually return to the atmosphere through microbial respiration. However, removal of slash will also change the quality of the litter supply which eventually will affect decomposition rates and humus content. The disadvantage of lost sequestration might be large in a very short perspective but negligible in a longer run (Table 3). It has also been argued that slash removal might suppress decomposition of organic matter in the humus layer because it might result in less favourable moisture and temperature conditions. If so, it would partly compensate for the lower supply of new litter.

Table 3. Potential reduction of GHG emissions through use of harvest residues for substituting oil. Positive values indicate decreased, and negative values indicate increased emissions (M ton C year $^{-1}$).

Years	10	30	100	500
Fossil fuel substitution	3.9	3.9	3.9	3.9
C sink in soil	-2.8	-1.0	-0.3	-0.1
Sum	1.1	2.9	3.6	3.8

Potential C sink due to growth and use of short-rotation forestry (Salix) at set-aside farmland

Set-aside farmland in Sweden, potentially available for Salix plantations, was estimated at 385 000 ha. The biomass production is in practice around 8 M ton d.w. ha^{-1} year^{-1}, but with an expected potential in the range 10–12 M ton d.w. ha^{-1} year^{-1}. The C content in the above-ground biomass of 10 ton d.w. ha^{-1} year^{-1} corresponds to 1.92 M ton C year^{-1}. If used as biofuel it can substitute for the emission of 1.54 M ton C year^{-1} (Table 4) from oil.

The establishment of Salix stands would result in a C accumulation of the total Salix biomass amounting to 4.38 ton C ha^{-1} year^{-1}. With respect to the available land area this is equal to 1.6 M ton C year^{-1}. After a 4-year establishment phase no further C accumulation is expected. The production of Salix will also generate a soil C sink in relation to the use of land in agricultural production. The annual accumulation of the soil C stock was, for a period of 50 years, estimated at 0.23 M ton C year^{-1}. After this period only marginal changes in the soil C stock were expected.

Table 4. Potential to change GHG by the establishment and use of Salix on set-aside farmland (M ton C year $^{-1}$).

Years	10	30	100	500
Fossil fuel substitution	1.54	1.54	1.54	1.54
C sink in biomass	0.64	0.21	0.06	0.01
C sink in soil	0.23	0.23	0.11	0.02
Sum	2.41	1.98	1.71	1.57

Potential C sink due to birch plantations on set-aside farmland

Set-aside farmland in Sweden, potentially available for silver birch plantations, was estimated at 385 000 ha. It was further assumed that production will take place without any fertilisation and that 15% of the stem wood production (e.g. precommercial thinnings) plus 75% of harvest residues as branches and tops will be used as biofuel (technically and economically accessible). Provided an average production of 6.2 ton d.w. ha^{-1} year^{-1}, the amount of available biofuel was estimated to 1.2 M ton d.w. year^{-1}. If used as biofuel it will substitute for the emission of 0.5 M ton C year^{-1} (Table 5) from fossil fuels.

Table 5. Potential to change GHG by the establishment of silver- birch stands on set-aside farmland (M ton C year $^{-1}$).

Years	10	30	100	500
Fossil fuel substitution	0	0	0.2	0.5
C sink in biomass	1.0	1.0	0.3	0.1
C sink in soil	0.12	0.12	0.06	0.01
Sum	1.1	1.1	0.6	0.6

The establishment of birch results in a C-sink in total birch biomass that amounts to 2.5 ton C ha^{-1} year^{-1}. The calculations were based on a situation where birch instantly is established on all available land and that harvest takes place after 30 years. With respect to the available land area, the sink would amount to 1.0 M ton C year^{-1}. After a 30-year establishment phase no further accumulation in biomass is expected. The production of birch will also accumulate C in the soil in relation to farm-land use. The annual accumulation was, for a period of 50 years, estimated at 0.12 M ton C year^{-1}. After this period only marginal changes in the soil C stock were expected.

Potential C sink due to intensive N-fertilisation of spruce stands

This strategy is based on a frequent (e.g. every second year) and demand adapted fertilisation with N and other nutrients. The increase in stem wood production was estimated at 4 ton d.w. ha^{-1} year^{-1} (Bergh et al., 1999). The effect is stronger at sites in northern Sweden where water supplies are ample. The fertilisation will reduce rotation periods to 70-90 years in northern Sweden and to 50-60 years in southern Sweden. Specific site requirements are rather young stands of Norway spruce, groundwater table at 0.5–2 m and texture not coarser than fine sand or loamy sandy till. Potentially available forest land in Sweden meeting these criteria, was estimated at 1.0 M ha (ca. 5% of total forest land area).

The harvest will produce residues that might be used for fossil fuel substitution. Provided that 75% of the residues as branches and tops can be removed (technically and economically accessible) the extra amount due to fertilisation would be 1.2 M ton d.w. year^{-1}. If used as biofuel it can substitute for 0.48 M ton C year^{-1} (Table 6) from fossil fuels. No harvest of residues is likely to be possible until 30 years after the fertilisation program is initiated.

Table 6. Potential to change GHG emissions by intensive N fertilisation of Norway spruce stands (M ton C year $^{-1}$).

Years	10	30	100	500
Fossil fuel substitution	0	0	0.34	0.45
C sink in biomass	2.0	2.0	0.0	0.0
C sink in soil	1.40	0.72	0.29	0.08
Sum	2.40	2.72	0.63	0.53

Carbon will accumulate in biomass due to increased stand volumes following fertilisation. This is likely to happen mainly within the first 30 years. Over the long-term the accumulation will be marginal due to shortened rotation periods, i.e. the stands will generally not increase C stocks. In this assessment, the initial sink in biomass was estimated at 2 ton C ha^{-1} year^{-1}, equal to 2 M ton C year^{-1} in Sweden.

The elevated growth rate will increase the supply of organic matter to the soil and affect decomposition rates in a way that more soil organic matter is accumulated. The accumulation was estimated at 20 ton C ha^{-1} after 30 years. The annual average increase would be 1.4 ton C ha^{-1} during the 10-year period but will successively decrease over time. However, there are many uncertainties concerning GHG emissions from the soil. Denser stands might decrease soil temperatures leading to slower decomposition and lower emission losses. There is also a risk that elevated N levels might give rise to emissions of nitrous oxides from occasionally wet soils.

Mitigation of site preparation

Soil scarification seems to decrease the amount of C stored in the soil. Based on results from disk trenching experiments in south and central Sweden it was concluded that the amount would decline by 4.5 ton C ha^{-1} during a 15-year period. Consequently the average increase in C emissions due to increased decomposition was set to 0.3 ton C ha^{-1} year^{-1}. In Sweden around 70 000 ha yr^{-1} are prepared using disk trenching. This would lead to emission of 0.02 M ton C year^{-1}. However, because the prepared area is accumulating from year to year, the average annual losses will be higher. Conversely, site preparation might, through improved regeneration and forest growth, result in a stronger C sink in biomass. However, this is not included in the assessments in this report (Table 7). The effect of mitigated or ceased disk trenching is quite small compared to other strategies in forestry, mainly due to the small area treated.

Table 7. Potential to change GHG emissions by mitigated site preparation (M ton C year $^{-1}$).

Years	10	30	100	500
C sink in soil	0.12	0.24	0.29	0.31

Drainage of wet peaty forest land

The total area of drained wet forest land in Sweden is, according to the National Forest Inventory, around 2.5 M ha (11% of total forest land). Lowering the groundwater table has a complex influence on GHG fluxes. Rates of decomposition of peat or peaty soil material resulting from drainage are initially high but then decline over time. Consequently, prevented or counteracted drainage is a strategic measure that might decrease emissions of CO_2 and N_2O, but increase emissions of CH_4.

This evaluation is based on counteracted or not maintained drainage of 0.25 M ha (10 % of total drained area) resulting in the groundwater table being raised to a depth around 0-0.5m. Drained wet forest land in Sweden loses an average of 1-4 ton C ha^{-1} $year^{-1}$ (Eriksson, 1991). Investigations in Estonia show that losses through decomposition might amount to 7-10 ton C ha^{-1} $year^{-1}$ during the first decade following ditching, but that they might later drop to 2-7 ton C ha^{-1} $year^{-1}$ (Tomberg, 1992). According to Laine (1996) losses are, on average, around 0.5 ton C ha^{-1} $year^{-1}$ during a century after drainage. Based on these findings, it is concluded that present emissions from drained wet forest land might be around 1 ton C ha^{-1} $year^{-1}$. In addition, the amount of emitted N_2O-N was estimated at, on average, 2 kg ha^{-1} $year^{-1}$. The sum of GHG amounts to 1.27 M ton C (as CO_2 equivalents) ha^{-1} $year^{-1}$, equal to 0.32 M ton C $year^{-1}$ for 0.25 M ha. Due to the continuous decomposition losses of peat, it was assumed that GHG emissions would decline over 100 years.

The recently drained wet forest land was assumed to have a mean forest production around 2.8 ton d.w. ha^{-1} $year^{-1}$ (7 m^3). This production might sustain removal of harvest residues to replace fossil fuels. Provided that 75 % of the residues as branches and tops can be removed (technically and economically accessible), the available amount would be 0.83 ton d.w. ha^{-1} $year^{-1}$. Used as biofuels from 0.25 M ha forest land, it will substitute for 0.17 M ton C $year^{-1}$ (Table 8) of emissions from fossil fuels.

The transition to wet conditions would result in forest production losses (i.e. the forest biomass will no longer accumulate C at the same rate). In addition, no harvest residues for fossil fuel substitution will be available. It is assumed that the present forest will not be harvested but left for biodiversity purposes. However, the present forest biomass will likely decompose slowly during 500 years with a rate of 0.04 M ton C $year^{-1}$ (0.25 M ha). Increased CH_4 emissions were estimated at 0.22 M ton C (CO_2-equivalents) $year^{-1}$ (0.25 M ha). Finally, the wet land might also accumulate litter remains (peat) amounting to 0.05 M ton C $year^{-1}$ (0.25 M ha).

Table 8. Potential to change GHG emissions by prevented drainage on previously drained wet forest land. Positive values indicate decreased, and negative values indicate increased emissions. (M ton C year $^{-1}$).

Years	10	30	100	500
Presently drained:				
Emissions of CO_2 and NH_4	-0.31	-0.27	-0.16	-0.03
Fossil fuel substitution	+0.17	+0.17	+0.17	+0.17
Sum	-0.14	-0.10	0.10	0.14
Prevented drainage				
Emissions of CH_4	-0.22	-0.22	-0.22	-0.22
Peat growth	+0.05	+0.05	+0.05	+0.05
Decomposition of forest biomass	-0.04	-0.04	-0.04	-0.04
Sum	-0.21	-0.21	-0.21	-0.21
Difference: not drained - drained	-0.07	-0.11	-0.31	-0.36

Other strategic measures

Burning of forest land seems to result in a loss of around 25% of the humus cover. Wild fires occur only marginally in Sweden but controlled burning is frequently carried out with the purpose to improve biodiversity. The C emissions caused by burning were estimated at 0.07 M ton C $year^{-1}$, provided that 5% of the annually harvested area is burned before regeneration, and that the humus layer, on average, contains 25 ton C ha^{-1}.

Tree species affect GHG emissions through their biomass production and impact on decomposition processes in the soil. The scientific basis for quantification of this effect is still weak. However, provided similar production rates, the soil sink seems to be most strong under birch, followed by Norway spruce and is the least strong under Scots pine.

Liming might increase or decrease the soil C sink due to its impact on decomposition. With good site conditions (i.e. low CN ratio), liming decreases the sink strength, whereas the opposite is true at poor sites. The overall effect of liming on GHG emissions in Sweden is considered to be small.

CONCLUSIONS

The present C accumulation in forest biomass and soils is large. However, it will decline over time because the storage capacity limit will be approached. Increases in storage capacities over 100 years or more, are therefore rather marginal. In contrast, the effect of using biomass for fossil fuel substitution will grow over time. Over a 100-year time frame the use of biomass for fossil fuel substitution is more effective than storing C in soils and vegetation (Table 9). Storage of C in biomass and soil is therefore considered a poor long-term solution in relation to intensive harvesting and use of residues as biofuels. In Sweden, only about 70% of the present forest growth is cut. Hence, there is great potential to increase cutting and reduce emissions by the use of residues as biofuels. Intensified harvest might be preferable also from a socioeconomic standpoint. Wood might, through the replacement of other materials such as concrete, reduce GHG emissions. From a strategic point of view it is important to form international rules for reduction of GHG emissions in a way that short-term solutions are not are preferred at the expense of more sustainable strategies.

In the short term (10–30 year) solutions with Salix plantations and intensive N fertilisation seem to be successful strategies to reduce GHG emission. These methods are interesting because they involve the use of harvest residues for fossil fuel substitution. They also contain a substantial storage component with large amounts of C accumulation in the biomass and soil. Still, over the longer-term, the use of harvest residues in "ordinary" forestry is superior. This is because Salix on set-aside farmland and intensive N fertilisation is most likely only to occur on small areas in relation to ordinary forestry.

Prevented or not maintained drainage on wet forest land turned out to be a poor solution (Table 8). The main reason for this is that one would lose the potential to use harvest residues for fossil fuel substitution. In addition and consistent with the views above, forestry is profitable and will generate economic resources. Also, wood might not only substitute fossil fuel but also other products in the society such as concrete and plastic materials.

Over a 10 year time frame the potential to reduce GHG emissions through forest strategies was around 15 M ton C year^{-1}. This is almost 95% of the present emissions through burning of fossil fuels. Even over the longer-term forestry might compensate for more than 50% of the emissions from fossil fuels. Thus, it can be concluded that forestry in Sweden might substantially contribute to combating GHG emissions.

Table 9. Potential to reduce GHG emissions in Swedish forestry through different strategies (M ton C year $^{-1}$) (for explanation: see Tables 1-8).

Positive values indicate decreased, and negative values indicate increased emissions.
The sum does no include the strategies "Prevented drainage (neg. values) and birch plantation
(The altemative "Salix" excludes a simultaneous use of birch)

Years	10	30	100	500
Forest biomass sink (23 M ha^{-1} year^{-1})	5.7	5.1	3.0	0.6
Soil C sink (23 M ha^{-1} year^{-1})	3.5	3.4	3.2	1.8
Fossil fuel substitution (23 M ha^{-1} year^{-1})	1.1	2.9	3.6	3.8
Salix plantation (0,385 M ha^{-1} year^{-1})	2.41	1.98	1.71	1.57
Birch plantation (0,385 M ha^{-1} year^{-1})	1.1	1.1	0.6	0.6
Intensive N fertilisation (1 M ha^{-1} year^{-1})	2.40	2.72	0.63	0.53
Mitigated scarification (0,07 M ha^{-1} year^{-1})	0.12	0.24	0.29	0.31
Prevented drainage (0.25 M ha^{-1} year^{-1})	-0.07	-0.12	-0.31	-0.36
SUM	15.23	16.34	12.43	8.61

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SOIL C AND N POOLS IN FORESTED UPLAND AND NON-FORESTED LOWLAND SITES ALONG THE BOREAL FOREST TRANSECT CASE STUDY IN CENTRAL CANADA

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ABSTRACT

Despite its importance in global carbon budgets, little is known about the carbon (C) chemistry of forest soil organic matter (SOM) in relation to the conceptual pools used in modelling. We characterized soil C and nitrogen (N) stocks in six sites along the Boreal Forest Transect Case Study (BFTCS) (three forested upland, three lowland) by solid-state ^{13}C nuclear magnetic resonance spectroscopy with cross-polarization and magic angle spinning (CPMAS NMR) and δ^{13}C and δ^{15}N values. NMR indicated that C in the soils' organic horizons is much more influenced by moss and lichen than by root, foliar, or woody litter. For upland soils more carbon was present in the mineral A and B horizons than in the LF(H). Mineral soils were fractionated by water-floatation and size fractionation. For the two sandy-textured upland Brunisols under jack pine, 8-17% of total C was in floatables, the fraction most susceptible to further decomposition. The <63 μm fraction (41 to 49% of C) was hydrophobic (i.e., very high in alkyl C). There was little preservation of lignin or carbohydrates, but some indication of pyrogenic C (char). No floatables were obtained from the clay-textured upland Cryosol under black spruce, but poorly-decomposed plant material (2.8% of C) was found in the 2000-250 μm fraction. The quality of carbon distributed amongst the other size fractions of the Cryosol indicated preservation of a broader range of SOM from plant, microbial and possibly char origins, but the majority of carbon in the mineral soil of the Cryosol was in the <63 μm fraction . Except for a high value for the lowland calcareous Mesisol, δ^{13}C values were within the narrow range expected for C3 forests, with some enrichment at depth and in smaller size fractions. The δ^{15}N values ranged from near-zero in LF(H) horizon of the Brunisols to +11‰ in some mineral soil size fractions; the higher values at depth may be indicative of long-term SOM decomposition and an open N cycle. An increase in carbon storage in the mineral soil horizons of the upland soils appeared to follow a trend of decreasing lessivage, increase in pH and increase in exchangeable calcium. Both climate and soil texture affected C and N stocks, which were lower for the two sandy-textured Brunisols than for the clay-textured Cryosol. Further studies should examine more soil/vegetation types to understand the effects of soil parent material, climate, and tree species on SOM and C-N linkages.

INTRODUCTION

The importance of boreal forests in the global carbon cycle is widely recognized, and there is great interest in the potential of soil organic matter (SOM) to act as a carbon (C) sink (Bhatti and Apps, 2000; Schulze et al., 1999; 2000; Vucetich et al., 2000; White et al., 2000). While boreal soil carbon stocks are large, whether a site functions as a net carbon source or sink in any given time-frame depends on what may be a relatively small difference between two much larger fluxes. One of these fluxes (Rh-heterotrophic respiration) is commonly estimated by modelling decomposition of detrital C inputs from wood and roots in several size classes and foliar litter, with kinetically (not biochemically) defined pools grading from fast or active, to slow and passive (Peng et al., 1998; Schimel et al., 1994). The former are generally associated with light fractions (specific gravity 1.0 to 1.8) which are still mainly of plant origin, whereas slower SOM pools are associated with mineral particles and characterized by increasing microbial and chemical transformation of the original plant biopolymers (Falloon and Smith, 2000).

Several recent publication have challenged the paradigm underlying this approach to modelling SOM or offer alternate approaches, especially for temperate and boreal forest ecosytems (Sanger et al. 1996; Ponomarenko and Anderson 2001; Chertov et al. 2001). Frequently SOM models use carbon-to-nitrogen ratios (C:N) to predict decomposition rates, which has proven to work well for agricultural systems where C and nitrogen (N) in crop residues occur in compounds succeptable to microbial attack, but may not work as well in forest ecosystems where C and N occur in highly heterogeneous compounds (Almendros et al. 2000; Lorenz et al. 2000; Preston et al. 2000, 2002; Taylor et al., 1989). Sometimes lignin:N is used to reflect litter quality and is invoked to slow down decomposition when modeling SOM dynamics (Parton et al., 1987; Sanger et al. 1996) but recently the use of the term "lignin" has been subject to criticism and it's recalcitrance in soil questioned. The term "lignin" is often (mis)used to represent the acid-insoluble residue (AIR) resulting from the final stage of proximate SOM analysis (Preston et al., 1997). However, if present, cutin and tannins will also contribute to the AIR. They may constitute a higher proportion of foliage than true lignin and likely decompose at different rates, and through different mechanisms than for lignin. Conversely, many studies have shown that foliar lignin can be readily decomposed in the forest floor, while lignin accumulation is associated with high input of coarse woody debris. Nierop et al. (2001) showed that within the aromatic fraction of Ah horizons, along a primary successional sequence (algae to pine), lignin is degraded and non-methoxyphenolics (i.e. tannins) accumulate. Studies of mineral SOM consistently show low preservation of lignin (loss of phenolic and methoxyl intensity), often accompanied by accumulation of alkyl structures (Baldock and Preston 1995; Guggenberger et al. 1995; Mahieu et al. 1999; Preston, 1996; ; Preston et al., 2002).

Since many stages are required in the biosynthesis of lignin (and lipids), they tend to be depleted in ^{13}C compared to carbohydrates. Therefore, selective preservation of lignin should favour depletion of ^{13}C in SOM (Balesdent et al., 1993; Ehleringer et al., 2000; Garten et al., 2000). However, the opposite tends to occur which is consistent with the lack of preservation of lignin, while the accumulation of alkyl C (Almendros et al. 2000; Baldock and Preston 1995; Preston 1996) and the ^{13}C enrichment indicate an increasing microbial contribution. Indeed, some research has provided evidence that the relative proportions of the constituent moieties in lignin itself are significantly different depending on the ecosystem under consideration, and that lignin quality should be considered when modeling SOM dynamics in forest ecosystems (Sanger et al. 1996).

Recent studies have also suggested that charcoal and deep carbon, which are currently not considered in most SOM models, may play significant roles in the SOM dynamics of boreal forests. Charcoal may represent a significant source of passive C (Harden et al., 1997; Nalder and Wein, 1999; Schulze et al., 1999), influence the magnitude of C sequestered in SOM (Schmidt and Noack, 2000; Schulze et al., 2000) and play other ecological roles important to forest SOM dynamics (Pietikäinen et al., 2000; Zackrisson et al., 1996). In a comparative study of uncultivated (under aspen and shrubland) and cultivated Black Chernozemic soils, Ponomarenko and Anderson (2001), suggested that the current conceptual model for humus, and the models rooted in that conceptual model, may have to change significantly in order to accomodate the role of char in the organic matter dynamics of these soils. Radiocarbon studies undertaken as part of the BOREAS project (Hall, 1999) have shown the importance of decomposition from deep soil carbon (Winston et al., 1997), and its high sensitivity to timing and duration of soil thaw (Goulden et al., 1998).

These studies suggest that it is important to assess the quality of SOM in boreal forests and wetlands to better define the conceptual pools used in modelling and to contribute to our understanding of how C is sequestered in borest forest soils. This preliminary study was carried out on soil samples from three upland forested and three lowland non-forested sites along the Boreal Forest Transect Case Study (BFTCS) in Canada (Halliwell and Apps 1997a; 1997b) to estimate C and N stocks, separate SOM fractions by size and density, and determine natural-abundance levels of ^{13}C and ^{15}N. Solid-state ^{13}C nuclear magnetic resonance spectroscopy with cross-polarization and magic angle spinning (CPMAS NMR) was used to characterize C in SOM (Kögel-Knabner, 2000; Preston, 1996; 2001; Quideau et al., 2000).

METHODS

Sites and Sampling

Three upland and three lowland sites were chosen from the Canadian Intersite Decomposition Experiment (CIDET) that were also located along the BFTCS (Halliwell and Apps, 1997a) (Figure 1). General site information is shown in Table 1 and further details can be found in Trofymow and CIDET Working Group (1998).

Site description and soil sampling (from four subplots per site) were carried out in 1992, during establishment of CIDET. Briefly, the lowland sites were sampled at 0-10 cm depth. Upland site samples were collected for the LFH on all sites, the A and B horizons of the Brunisols, and a combined Ah+Bm horizon sample for the Cryosol (Ah thin and discontinuous).

The samples which had been stored in an air-dried state until 1999, were used to prepare one composite sample per site for each horizon. Although the sampling protocol was not designed for the present study, and the samples were stored for some time, we considered them adequate for a preliminary study of organic matter, especially for general characterization by NMR. Additional soil chemical data are available in Trofymow and CIDET Working Group (1998).

Sample Fractionation and Chemical Analysis

Sonication and wet-sieving were used to separate the <2 mm portion of the mineral soil into the following fractions: water-floatables, 2000-250 μm, 250-63 μm and <63 μm. Air-dry soil (90 to 130 g) was weighed into a 400 ml beaker, 250 mL deionized water was added, and the sample was stirred to break up larger lumps. It was then left to slake for 24 h at room temperature, after which floatables were skimmed off the top. The remaining material was sonified for 5 minutes using a SONICS & MATERIALS VIBRA CELL model #375 with a disrupter tip of 12 cm diameter and 206 W power output. No rise in temperature of samples was observed during sonication. Further small amounts of floatables generated during sonication were combined with those previously collected. Samples were then wet sieved through 250 μm and 63 μm sieves, and the water was combined with the <63 μm fraction to recapture any dissolved organic C. The fractions and portions of the original samples were dried at 70° C; organic samples were ground in a Wiley mill to 30 mesh (600 μm) and mineral samples were finely ground in a Siebtechnik mill. For the organic samples, subsamples were further ground under liquid nitrogen with a mortar and pestle.

Sample C was determined by dry combustion using a LECO CR-12 analyzer. Samples were analyzed for total N by the semimicro-Kjeldahl method, followed by Rittenberg conversion of ammonium N to N_2 gas and ^{15}N analysis using a Vacuum Generators Sira 9 mass spectrometer (Chang and Preston, 2000). The instrument was standardized using a nitrogen isotope reference gas ($\delta^{15}N$ = -0.67‰ with respect to air, 0.36605‰ ^{15}N) from OzTech Trading Corp., Dallas TX. Carbon isotopic composition was measured using a Fisons NA1500 elemental analyzer linked to a Finnigan MAT 252 isotope ratio mass spectrometer. Values of $\delta^{13}C$ are reported relative to the Pee Dee Belemnite (PDB) standard, and are accurate to ± 0.2 per mil. Very small samples (such as some of the floatables) were analysed for C and N using a Carlo Erba 1106 CHN analyser. Values of total C and total N are accurate to ± 0.1 and 0.5 mg g^{-1}, respectively. Carbonate in the Typic Mesisol sample was analysed by manometric measurement of the CO_2 evolved upon treatment of the sample with 4M HCl (Kalra and Maynard, 1991).

Carbon-13 CPMAS NMR spectra were obtained at 75.47 MHz on a Bruker MSL 300 spectrometer. Dry, powdered samples were packed into a zirconium oxide rotor of 7 mm OD. Acquisition conditions were: 4.7 kHz spinning rate, 1 ms contact time, 2 s recycle time, and 5,000-36,000 scans. For samples with the lowest C content, over 100,000 scans were obtained and the recycle time reduced to 1s. Standard pulse sequences were used to obtain spectra with dipolar dephasing (DD), or DD combined with total suppression of spinning sidebands (TOSS). The Typic Mesisol sample was also run under quantitative conditions, using Bloch decay (BD), a simple acquisition without CP enhancement. Spectra of samples low in C, and those obtained using BD were corrected by subtraction of the spectrometer background signal produced by running an empty rotor (Preston, 2001). Spectra were processed with 30-40 Hz line broadening and baseline correction. Chemical shifts are reported relative to tetramethylsilane (TMS) at

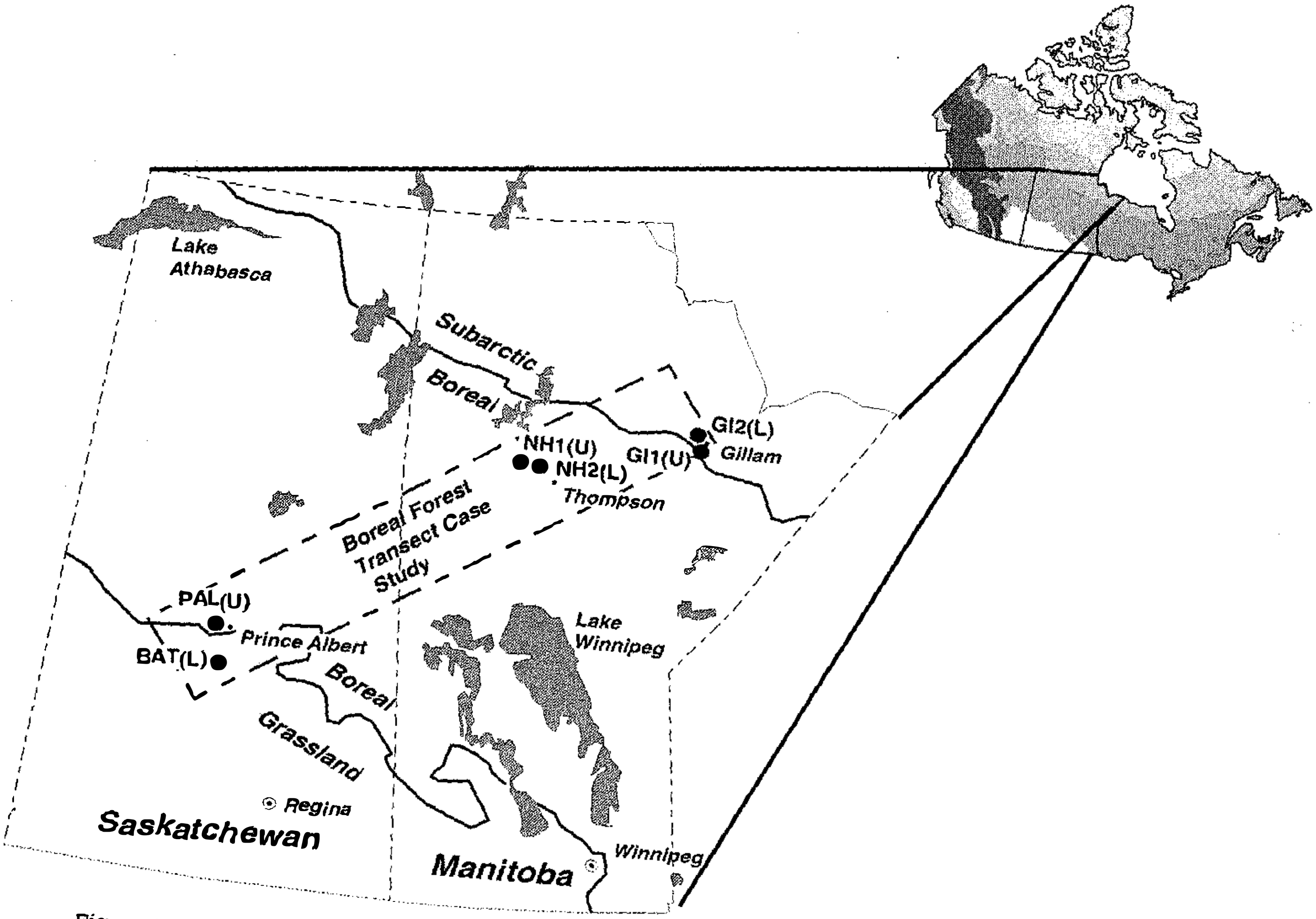

Figure 1. Location of sites sampled along the Boreal Forest Transect Case Study (BFTCS).

Table 1. General site descriptions for upland and lowland sites[a].

Landscape Position	Ecoclimatic province (CIDET Site)	CSSC Soil Classification	Vegetation Type	Latitude (N)	Longitude (W)	Elevation (m)	Total Annual Precip.[b] (mm)	Mean Annual Temp.[b] (°C)
Lowland	Transitional grassland (BAT)	Typic Mesisol (calcareous)	Shrubby Lowland *Betula glandulosa, Potentilla fruticosa, Carex spp., Tomenthypnum nitens, Drepanocladus spp.*	52.72°	106.13°	472	398	0.1
	Subhumid high boreal (NH2)	Typic Fibrisol	Treeless fen *Betula glandulosa, Carex spp., Sphagnum warnstorfii, Tomenthypnum nitens*	55.91°	98.42°	260	542	-3.9
	Low Subarctic (GI2)	Typic Fibrisol	Treeless fen Chamaedaphne calyculata, Rubus chamaemorus, Sphagnum anjustifolium, Tomethypnum nitens	56.33°	94.51°	125	484	-5.2
Upland	Subhumid low boreal (PAL)	Orthic Dystric Brunisol	Jack pine (*Pinus banksiana*) *Vaccinium myrtilloides, Arctostaphylos uva-ursi, Pleurozium schreberi, Cladina*	53.23°	105.98°	476	398	0.1
	Subhumid high boreal (NH1)	Eluviated Dystric Brunisol	(Same as above)	55.93°	98.61°	288	542	-3.9
	Low Subarctic (GI1)	Brunisolic Eutric Static Cryosol	Black spruce (*Picea mariana*) Larch (*Larix laricina*) *Vaccinium uliginosum, Ledum groenlandicum, Hylocomium splendens, Cladina mitis*	56.32°	94.85°	140	484	-5.2

[a] From Trofymow and CIDET Working Group (1998).

[b] 1951-80 annual normals

0 ppm, with the reference frequency set using adamantane.

Spectra of organic horizons and floatables were divided into the following chemical shift regions: 0-47 ppm, alkyl; 47-93 ppm, methoxyl, amide CHN and O-alkyl; 93-112 ppm, di-O-alkyl; 110-140 ppm, aromatic (C or H substitution); 140-165 ppm, phenolic and N-substituted aromatic; 165-185 ppm, carboxyl (including acids, amides, esters), and 185-215 ppm, aldehydes and ketones. The 185-215 ppm region accounted for only 1.0-2.3% of the total intensity, and was combined with the carboxyl region. Spectra of mineral fractions were divided into fewer regions: 0-47, 47-112, 112-165 and 165-215 ppm. The relative area of each region was obtained by integration.

Where spinning sidebands were large enough to be measured (mainly for the carboxyl signal, and for samples higher in C), the relative areas were corrected by assuming equal areas for the upfield and downfield sidebands. Several factors can affect the quantitative reliability of cross-polarization spectra, due to signal loss caused by paramagnetic species including iron, and variable CP efficiency which distorts the relative areas (Hu et al., 2000; Mao et al., 2000; Kögel-Knabner, 2000; Preston, 1996; 2001). Therefore, the NMR data are only used qualitatively, to compare relative areas among similar samples.

RESULTS AND DISCUSSION

Mass, C, and N Distribution

Chemical and isotope data are shown in Tables 2-4. Mass recoveries for the mineral soil fractionations were close to 100% (Table 2). A slightly lower recovery (96.8%) for the Cryosol was attributed to the sample's high clay content which made quantitative transfer difficult.

For all mineral soil horizons of the Brunisols, floatables had low values for SOM mass% (mean=.7%) but the highest average concentrations for C (mean = 250 mg g^{-1}) and N (mean = 7.2 mg g^{-1})(Table 3). Conversely, the 2000-250 μm fraction from the same horizons accounted for the highest proportion (mean = 82%) of the SOM mass% but had the lowest concentrations for C (mean = 1.75 mg g^{-1}) and N (mean = 0.1 mg g^{-1}). Data for % distribution of C and N in the Brunisols show that, regardless of horizon, the largest proportion of C (49.1-56.8%) and N (45.6-59.7%) is found in the <63μm fraction.

As one moves from the the A to B horizons in the Brunisolic soils there is an increase in the 2000-250μm fraction with a concomitant decrease in the floatables and the 250-63μm fractions. In addition, the C:N for the 2000-250μm size fraction decreased from the A to B horizons. This phenomenon is more strongly expressed for the Orthic, than for the Eluviated, Dystric Brunisol.

SOM from the clay-textured Cryosol exhibited distinctly different patterns. No floatables were recovered and 97% of the SOM mass, 91% of the C distribution and 96% of the N distribution occurred in the <63 μm size fraction. The C:N ratio of this size fraction (18) was considerably lower than the C:N of the two larger size fractions (50.1:35.5) recovered

Table 2. Percent recovery of mass, carbon and nitrogen after fractionation of mineral soil.

Ecoclimatic Province (CIDET Site)	CSSC Soil Classification	Horizon	Mass	Carbon	Nitrogen
Subhumid low boreal (PAL)	Orthic Eutric Brunisol	Ah	99.0	82.0	90.4
		Bm	99.6	83.5	120.8
Subhumid high boreal (NH 1)	Eluviated Dystric Brunisol	Ae	99.6	84.0	91.7
		Bm	99.4	92.6	94.8
Low Subarctic (GI 1)	Brunisolic Eutric Static Cryosol	Ah+Bm	96.8	94.7	98.1

Table 3. C,N and isotope data for upland mineral soil horizons by size fraction.

CSSC Soil Classification (CIDET Site)	Horizon	Size Fraction	SOM Mass (%)	Carbon		Nitrogen		C/N	Isotopes	
				Concentration (mg g^{-1})	Distribution (%)	Concentration (mg g^{-1})	Distribution (%)		$\delta^{15}N$ ‰	$\delta^{13}C$ ‰
Orthic Dystric Brunisol (PAL)	Ah	floatables	1.8	279.0	20.7	9.8	19.4	28.4	3.5	-26.0
		2000-250 μm	80.3	3.1	10.2	0.1	8.7	30.9	[a]NR	-25.9
		250-63 μm	8.6	43.7	15.3	1.3	12.2	33.6	2.2	-26.5
		<63 μm	9.3	142.0	53.8	5.9	59.7	24.1	4.1	-25.3
	Bm	floatables	0.1	320.0	9.3	9.1	3.8	35.4	[b]ND	-25.8
		2000-250 μm	86.0	0.9	22.7	0.1	35.5	9.1	NR	-25.3
		250-63 μm	6.8	5.7	11.3	0.2	5.6	28.5	3.7	-26.2
		<63 μm	7.0	27.9	56.8	1.9	55.5	14.7	7.3	-24.3
Eluviated Dystric Brunisol (NH 1)	Ae	floatables	0.8	173.0	17.3	5.0	14.6	34.5	3.8	-26.8
		2000-250 μm	82.7	1.8	18.5	0.1	30.1	18.2	NR	-26.7
		250-63 μm	13.3	9.2	15.1	0.2	9.6	46.2	7.1	-25.9
		<63 μm	3.2	123.0	49.1	3.9	45.6	31.6	5.1	-25.9
	Bm	floatables	0.2	228.0	12.3	4.8	5.1	48.0	ND	-26.6
		2000-250 μm	81.0	1.2	26.1	0.1	42.5	11.5	NR	-25.4
		250-63 μm	12.6	2.7	9.1	0.1	6.6	27.1	NR	-25.5
		<63 μm	6.2	31.4	52.5	1.4	45.8	22.4	8.8	-25.9
Brunisolic Eutric Static Cryosol (GI 1)	Ah+Bm	2000-250 μm	0.8	140.0	2.9	2.8	1.1	50.1	7.2	-25.1
		250-63 μm	2.0	113.0	5.5	3.2	3.0	35.3	5.8	-25.2
		<63 μm	97.2	37.8	91.6	2.1	96.0	18.0	6.6	-24.8

Mass(%) and Distribution(%) of carbon and nitrogen standardized to the sum of recovered = 100%

[a] NR, not repeated

[b] ND, not determined due to insufficient sample size

Table 4. C, N and isotope data for lowland and upland sites by horizon.

Landscape Position	Ecoclimatic Province (CIDET Site)	Soil Classification	Horizon	C (%)	N (%)	C/N	$\delta^{15}N$ (‰)	$\delta^{13}C$ (‰)
Lowland	Transitional grassland (BAT)	Typic Mesisol (calcareous)	LFH	19.5	.73	27	2.8	-20.3
	Subhumid high boreal (NH 2)	Typic Fibrisol	Of	45.0	1.20	38	1.2	-26.9
	Low subarctic (GI 2)	Typic Fibrisol	Of	45.0	.92	49	0.3	-25.2
Upland	Subhumid low boreal (PAL)	Orthic Dystric Brunisol	LFH	22.6	.70	33	0.3	-26.7
			Ah	2.9	.10	29	3.8	-25.7
			Bm	0.4	.02	21	5.2	-25.0
	Subhumid high boreal (NH 1)	Eluviated Dystric Brunisol	LF	31.8	.69	46	0.6	-26.6
			Ae	1.0	.03	32	1.7	-26.5
			Bm	0.4	.02	20	11.7	-25.6
	Low Subarctic (GI 1)	Brunisolic Eutric Static Cryosol	LFH	36.9	1.10	35	2.5	-26.0
			Ah+Bm	4.1	.21	20	5.6	-24.8

from this soil (Table 3).

For the lowland sites, Of horizons of the two Typic Fibrisols had higher total C and N with higher C:N ratios than the Typic Mesisol (Table 4). LFH horizons at the upland sites had lower C:N than the LF of the Eluviated Dystric Brunisol. In the case of the Orthic Dystric Brunisol, the C:N was lower due to lower %C, but in the case of the Cryosol, the C:N was lower primarily due to higher %N (Table 4).

Carbon and Nitrogen Isotope Data

Except for the Typic Mesisol (calcareous) (-20.3‰), values of $\delta^{13}C$ were between –24.3 to –26.9‰ (Table 3) which is consistent with those widely reported for litter and SOM in C_3 forests (Balesdent et al., 1993; Bird and Pousai, 1997; Ehleringer et al., 2000; Flanagan et al., 1996; Garten et al., 2000) and *Sphagnum* species (Kracht and Gleixner, 2000; Rice, 2000). The exceptional value for the Typic Mesisol may reflect carbonates originating from CO_2 or bicarbonate produced from fermentation and possibly sulfate-reduction (Rask and Schoenau, 1993), whereas secondary carbonates resulting from oxidation of methane are much more highly depleted. It may also reflect a higher input or preservation of C from C4 plants (Bird and Pousai, 1997) for this site in the Transitional Grassland ecoclimatic province (Wolfe et al., 2001).

For the sandy-textured upland Brunisolic soils, the general trend (with a few exceptions) was for $\delta^{13}C$ to increase with depth, and for the Eluviated Dystric Brunisol to be more depleted than the Orthic Dystric Brunisol (Table 4). $\delta^{13}C$ in the Orthic Dystric Brunisol increased from –26.7‰ in the LFH to –25.7‰ in the Ah horizon, whereas there was no real difference for the Eluviated Dystric Brunisol (LF, –26.6‰; Ae horizon, –26.5‰). As one moves from the A to the B horizons in these soils, $\delta^{13}C$ increased an average of 0.8‰ for whole soil (Table 4), 0.2‰ for floatables and 0.4‰ for the 250-63 μm fraction (Table 3). $\delta^{13}C$ also increased with depth for the 2000-250 μm fraction(mean =

+0.9) but the values may be less reliable because of the very low C content of these fractions (0.9-3 mg g^{-1}). The $\delta^{13}C$ values for the <63 µm fraction increased with depth (+1.0‰) in the Orthic, but not the Eluviated, Dystric Brunisol.

The Cryosol was less depleted compared with the Brunisols, with $\delta^{13}C$ of −26.0‰ in the LFH and within a narrow range of −25.2 to −24.8‰ for the Ah+Bm horizons (Table 4). The higher value for $\delta^{13}C$ of the whole mineral soil compared with the LFH reflects the dominating influence of the <63 µm fraction (-24.8‰; 97.2 SOM mass%).

Increases of $\delta^{13}C$ with depth have been widely reported (Balesdent et al., 1993; Ehleringer et al., 2000; Garten et al., 2000), and are considered to result mainly from a shift from plant to microbial components during decomposition. A similar ^{13}C enrichment effect can occur in fine soil size fractions (Bird and Pousai, 1997).

Similar trends were found for the upland soils in this study. Compared to the whole soil and the LF(H), the $\delta^{13}C$ values showed depletion in the floatables and coarser SOM fractions from the mineral soil. These fraction would be dominated by plant components. The <63 µm fraction was enriched, compared with the LF(H), to a greater degree for the Orthic Dystric Brunisol and the fine-textured Cryosol than for the Eluviated Dystric Brunisol. The <63 µm samples from both horizons in the Orthic Dystric Brunisol, and the Ae of the Eluviated Dystric Brunisol are enriched with $\delta^{13}C$ and are are high in alkyl C (discussed later) which may indicate a microbial rather than plant origin for this fraction. If the organic matter were of plant lipid origin ^{13}C would be depleted, rather than enriched, relative to the whole plant material (Cayet and Lichtfouse, 2001).

These results suggest microbial C was retained to the greatest degree in the Cryosol, followed by the Orthic Dystric Brunisol and then the Eluviated Dystric Brunisol. For the latter, it may be possible that dissolved organic carbon of a microbial origin was removed in the more acidic and net leaching environment at this site as indicated by the eluvial nature of the soil and the high annual precipitation in the region (Table 1). The Orthic Dystric Brunisol may have stabilized more microbial C due to less lessivage, and more calcium on the exchange complex to facilitate clay-organo complexing (Table 5). The greatest degree of C stabilization occurred in the Cryosol and this C is most likely of a microbial origin. Our results suggest that significant microbial activity must have occurred in this soil in spite of cold temperatures, since no floatables were recovered from this soil and the majority of the C and N in the <63 µm was high in alkyl C. The high capacity of this mineral soil to stabilize carbon likely reflects the combined effect of a high concentration of calcium on the exchange complex and a high percentage of clay in the presence of cyro- and pedo-turbation facilitating the recombination of organic and inorganic components of the soil. Smith et al. (1991) and Brewer and Pawluk (1975) concluded that cryoturbation and faunal activity were two major processes affecting SOM dynamics and stabilization in Cryosolic soils.

Typic Fibrisols at the lowland sites have $\delta^{13}C$ values (Table 4) comparable to the larger sized fraction extracted from uplands soils at the same sites (Table 3). The Fibrisol in the low subarctic site was less depleted ($\delta^{13}C$, -25.2‰) than the one in the subhumid high boreal site ($\delta^{13}C$, -26.9‰).

The values for $\delta^{15}N$ were all positive, ranging from 0.3‰ to 3.8‰ for organic and mineral A horizons. Values for the Bm horizon and the Ah+Bm were higher, ranging from 5.2‰ to 11.7‰ (Table 4). Vogel and Gower (1998) found comparable $\delta^{15}N$ values of around +4‰ for mineral soil (unspecified depth) in jack pine sites close to our locations. In the upland soils the lowest values were found for the LF(H) horizons of the Brunisols (0.3 and 0.6‰). Values ranged from 4.1‰ to 8.8‰ for the <63 µm fractions.

Like $\delta^{13}C$, soil $\delta^{15}N$ values generally increase with depth (Högberg et al., 1996). Near-zero or negative $\delta^{15}N$ values are often found for organic horizons in temperate and northern soils (Högberg et al., 1996; Martinelli et al., 1999; Nadelhoffer and Fry, 1988). These are usually associated with high rates of biological N-fixation, very young or recently disturbed soils and ecosystems with a closed N cycle. Values of $\delta^{15}N$ tend to increase with increasing decomposition and age of SOM, in systems with an "open" N cycle: i.e., higher N losses. Generally, higher foliar and soil $\delta^{15}N$ values are reported for tropical and have been associated with high N availability and N losses.

Table 5. C and N stocks, pH ($CaCl_2$) and exchangeable calcium, by horizon derived from CIDET[a] site and soil data.

Landscape Position	CSSC Soil Classification (CIDET Site)	Horizon	Thickness[b] (cm)	C ($Mg\ ha^{-1}$)	N ($Mg\ ha^{-1}$)	C:N	pH	Exch. Ca (me/100 g)
Lowland	Typic Mesisol (calcareous) (BAT)	LF(H)[b]	10.0	4.02	0.13			
	Typic Fibrisol (NH 2)	Of[b]	10.0	13.01	0.26			
	Typic Fibrisol (GI 2)	Of[b]	10.0	6.94	0.17			
Upland	Orthic Dystric Brunisol (PAL)	LFH	2.5	8.98	0.20	45		
		Ah	4.0	9.22	0.44	21	5.5	6.9
		Bm	56.0	7.20	0.32	22	5.2	1.8
		Total		25.40	0.96			
	Eluviated Dystric Brunisol (NH 1)	LF	1.0	2.79	0.05	56		
		Ae	2.0	1.60	0.08	20	4.0	.4
		Bm	27.0	14.24	0.97	15	4.7	.2
		Total		36.97	1.98			
	Brunisolic Eutric Static Cryosol (GI 1)	LFH	15.0	6.10	0.17	36		
		Ah&Bm	20	124.53	5.63	22	6.6	43.8
		Sum		130.63	5.80			

[a] Trofymow and CIDET Working Group (1998).

[b] Standardized to 10 cm for organic soils and bottom of the B horizon for mineral soils

Our results show that some of the organic horizons have near-zero values (Table 4). Low δ^{15}N in the LF(H) of the Brunisolic soils may reflect the presence of N-fixation activity possibly associated with lichens and mosses (Billington and Alexander, 1978; Dawson, 1983; Huss-Danell, 1977;Karagatzides, et al., 1985) and a relatively closed N cycle between the humus form and the vegetation, but further research would be needed to substantiate this claim. δ^{15}N values were highest for N associated with the <63 μm fraction suggesting older nitrogen, possibly of microbial origin, is stabilized by association with clay particles.

Results for samples with 0.1 mg g^{-1} N are not reported individually in Table 3, but were between 9‰ and 19‰. These higher than expected values may be due to artefacts in the analysis of samples very low in total N. However, δ^{15}N data on such soil fractions are non-existent and worthy of reporting. The high values may reflect disproportionately high contribution from inorganic material in small samples (ammonium in metamorphic rocks is highly enriched (δ^{15}N around 11‰)) (Boyd, 2001).

The wide range of δ^{15}N values found in this preliminary study are consistent with other investigations, and suggests that a more detailed investigation may provide insights into controls on N cycling in these northern ecosystems. Studies now in progress include analysis of individual organic horizons within the forest floor, and the origin of the high δ^{15}N values found in the very-low-N mineral fractions.

Carbon-13 CPMAS NMR Spectroscopy

The NMR spectra are presented in Figures 2-8, and the relative areas in Table 6. The features of the spectra have been interpreted based on many studies of soil organic matter (Baldock and Preston, 1995; Kögel-Knabner, 2000; Preston, 1996; 2001), litter (Almendros et al., 2000; Huang et al., 1998; Lorenz et al., 2000; Preston, 1999; Preston et al., 1997; 2000; 2002), peat (Bergman et al., 2000; Preston et al., 1989) and wood (Hatcher, 1987; Preston et al., 1998). For this reason, only features pertinent to this study will be discussed.

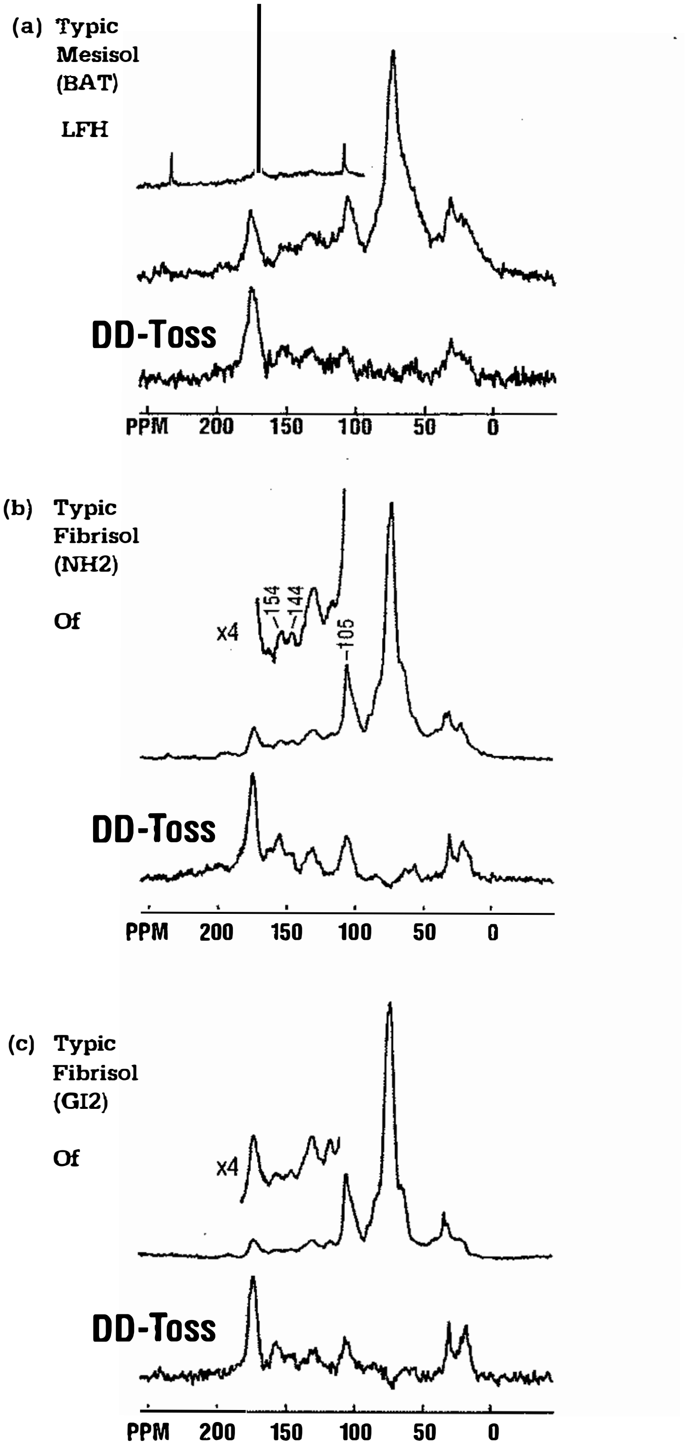

Figure 2. Normal and DD-TOSS ^{13}C CPMAS NMR spectra of lowland 0-10 cm horizons: (a) Typic Mesisol (BAT) with inset showing carbonate peak obtained using BD, (b) Typic Fibrisol (NH2), (c) Typic Fibrisol (GI2).

NMR spectroscopy of lowland sites

The lowland Fibrisol samples (Figures 2b, 2c) are typical of poorly decomposed sphagnum peat, with high intensity due to carbohydrates (O-alkyl peak at 73 ppm and di-O-alkyl peak at 105 ppm), and low alkyl, aromatic, phenolic and carboxyl intensities. These two samples have more than 50% of intensity in the O-alkyl region, and low ratios of alkyl/O-alkyl C. The DD spectra show almost no intensity at 55-57 ppm for methoxyl C, characteristic of lignin structures. This indicates that the weak, largely unresolved intensity in the 50-60 ppm region of the normal CP spectra derives from CHN of amino acids, rather than methoxyl structures. There is some intensity in the aromatic and phenolic regions from carbon substituted with C, O or N. However the splitting in the phenolic region in both the normal and DD spectra (maxima at 144 and 154 ppm) indicates a dominance of condensed tannins, rather than lignin. The broad peak at 106 ppm in the DD spectrum is also a marker for condensed tannins (Lorenz et al., 2000; Preston, 1999; Preston et al., 1997) and other hydroxybenzene structures from mosses (Wilson et al., 1989). The tannins must have originated from other species of higher plants at the sites, since they are not produced by *Sphagnum* (Williams et al., 1998).

The Typic Mesisol LF(H) spectrum (Figure 2a) has broader features, higher aromatic and phenolic content, and a higher alkyl/O-alkyl ratio relative to the Fibrisols, indicative of more advanced decomposition, consistent with its lower C:N ratio. However, signals for lignin and tannin are not prominent, there is no resolved intensity for methoxyl at 57 ppm, and the DD spectrum also shows little intensity in the methoxyl, aromatic and phenolic region. Compared to the Fibrisols, it has the weakest intensity at 105-106 ppm in the DD spectrum, which indicates no memory of lignin and tannin structures. This means that either higher plants had little input to SOM at this site or that they are very strongly decomposed.

For Batoche, the signal-to-noise ratio (S/N) was lower than that expected from its total C content of 195 mg g^{-1}. Due to its remoteness from hydrogen atoms, carbonate carbon is barely detected by a CP experiment (Preston, 2001), but a quantitative BD spectrum (Figure 1a, inset) showed a large signal for carbonate at 168.5 ppm. The presence of carbonate was consistent with the higher pH of the Batoche site, and chemical analysis of 313 mg g^{-1} of $CaCO_3$ (37 mg g^{-1} of carbonate C).

NMR spectroscopy of upland LF(H)

For the upland sites, the LF(H) spectra from the Brunisols (Figures 3a,b) have the strong alkyl and O-alkyl peaks typical for the forest floor; however their aromatic and phenolic intensities are unusually low and similar to those for the Typic Fibrisols. Their DD spectra are also similar to the Fibrisols, with low aromatic, phenolic and methoxyl intensity, and the strongest signals for carboxyl C and a mobile fraction of alkyl C (methyl, acetate, and long-chain CH_2). By contrast, the LFH of the Cryosol (Figure 3c) has higher aromatic and phenolic intensities in both the normal and DD spectrum. The DD spectrum also shows the tannin marker peak at 106 ppm, and a weak but well-resolved methoxyl peak at 56 pp.

A high alkyl/O-alkyl ratio is often used as an indicator of decomposition restricted by factors such as unfavourable temperature, moisture or oxygen status, or nutrient limitations (Baldock et al., 1995; Preston et al., 1989). This leads to an accumulation of relatively recalcitrant alkyl C from residual microbial biomass and plant polymers such as cutin, suberin, and the even less soluble suberans and cutans.

In this study, the low alkyl/O-alkyl ratios for the Fibrisols probably reflects the low alkyl content of *Sphagnum* (Bergman et al., 2000) ,while the higher values for the upland LF(H) (0.42-0.55) are more typical of the forest floor (Lorenz et al., 2000) and may result from accumulation due to decomposition restricted by edaphic factors. The low intensities for tannin and lignin peaks in the forest floor spectra for the Brunisols may reflect the dominance of moss inputs to SOM of the upland soils (Harden et al., 1997; Williams et al., 1998). There is relatively little indication of input or persistence of woody debris, tree and shrub foliar litter, or charcoal in the LF(H) at these sites (Preston, 1999; Lorenz et al., 2000; Preston et al., 2002).These results are consistent with those of Nierop et al. (2001) who found that moss had a significant influence on SOM accumulation in the Ah under heather and pine, whereas above ground litter was of negligable importance.

NMR spectroscopy of mineral horizons

Water-floatable fractions were obtained from the mineral horizons of the two sandy-textured Brunisols (Figure 4). Three of the four horizons (except Bm of Orthic Dystric Brunisol, Figure 4b) are high in alkyl C, probably largely

Table 6. NMR spectral regions (percent of total area) for organic and suitable mineral fractions.

Landscape Position	CSSC Soil Classification (CIDET Site)	Horizon	% of total	area in chemical shift range (ppm)						ratio
			0-47	47-93	93-112	112-140	140-165	165-185	185-215	alkyl/O-alkyl
		Organic Horizons								
Lowland	Typic Mesisol (BAT)	LFH	18.4	43.5	10.6	11.0	6.3	8.1	2.1	0.42
	Typic Fibrisol (NH 2)	Of	16.5	51.7	13.5	7.5	4.5	5.3	1.0	0.32
	Typic Fibrisol (GI 2)	Of	13.1	56.3	14.8	7.6	3.8	3.4	1.0	0.23
Upland	Orthic Dystric Brunisol (PAL)	LFH	24.1	44.0	10.9	7.4	5.9	7.1	0.6	0.55
	Eluviated Dystric Brunisol (NH 1)	LF	20.9	50.1	12.2	7.3	3.5	4.7	1.3	0.42
	Brunisolic Eutric Static Cryosol (GI 1)	LFH	17.4	37.4	10.3	14.3	9.3	9.5	1.8	0.46
		Floatables								
Upland	Orthic Dystric Brunisol (PAL)	Ah	24.7	32.5	9.1	15.4	9.6	6.4	2.3	0.76
		Bm	15.3	41.5	13.2	16.5	8.0	4.3	1.2	0.37
	Eluviated Dystric Brunisol (NH 1)	Ae	36.8	23.4	7.1	13.3	11.1	6.6	1.7	1.57
		Bm	23.5	37.8	12.9	11.6	7.5	5.2	1.5	0.62

			% of total area in chemical shift range (ppm)				ratio alkyl/O-alkyl
			0-47	47-112	112-165	165-215	
Uplands	Orthic Dystric Brunisol (PAL)	Ah whole soil	24.9	44.1	19.6	11.4	0.56
		Ah 250-63μm	37.5	38.3	15.9	8.3	0.98
		Ah <63 μm	42.3	39.5	12.5	5.7	1.07
		Bm <63 μm	25.6	48.9	18.0	7.5	0.52
	Eluviated Dystric Brunisol (NH1)	Ae <63 μm	41.4	35.6	15.4	7.6	1.16
		Bm <63 μm	40.8	41.7	10.6	6.9	0.98
	Brunisolic Eutric Static Cryosol (GI 1)	Ah+Bm (whole soil)	24.3	48.3	19.6	7.8	0.50
		Ah+Bm (2000-250 μm)	17.0	47.8	28.4	6.8	0.36
		Ah+Bm (250-63μm)	18.6	39.5	35.2	6.7	0.47
		Ah+Bm (<63 μm)	28.3	47.1	17.2	7.4	0.60

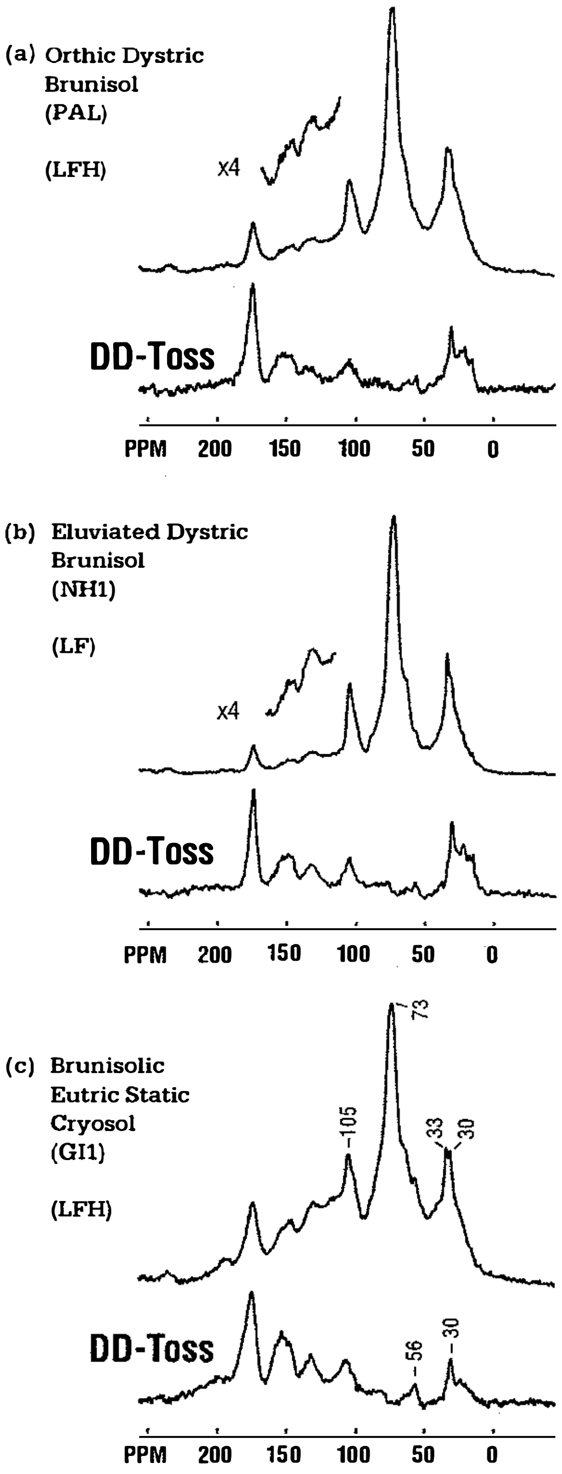

Figure 3. Normal and DD-TOSS ^{13}C CPMAS NMR spectra of upland LF(H) horizons: (a) Orthic Dystric Brunisol (PAL), (b) Eluviated Dystric Brunisol (NH1), (c) Cryosol (GI1).

derived from suberins of roots (Augris et al., 1998; Nierop, 1998). Highly decomposed lignin or low input of lignin is indicated by low intensity at 130 ppm, and at 57 ppm (methoxyl) in the DD spectra. Nierop et al. (2001) provided evidence that lignin in the mineral soil (possibly from roots) decomposes at a much faster rate than lignin introduced above the mineral soil. The broad phenolic peak (145-155 ppm), and the 105 ppm peak in the DD spectrum are tannin indicators, as seen in root spectra from other forest sites (Preston, 1999; Preston et al., 2002).

For mineral fractions, spectra could be obtained down to 30 mg g^{-1} total C, using a shorter delay time of 1 s and acquisition times up to 36 h for samples with the lowest C contents. The normal CPMAS spectra of all mineral samples were corrected by subtraction of the spectrometer background signal, the effect of which is illustrated for two samples with C contents of 29 and 123 mg g^{-1} (Figures 5a,b, respectively). It was possible to obtain DD-TOSS spectra for some samples higher in C; for the <63 μm fraction from the Bm horizon of the Eluviated Dystric Brunisol (31.4 mg $^{-1}$ C, Figure 7a), the spectrum was obtained with DD and subtraction of the background signal.

Compared to organic horizons and floatables, the spectra of mineral fractions (Figures 6-8), have lower S/N and broader features, but peak positions typical for mineral soils and their fractions (Mahieu et al., 1999; Preston, 1996), with maxima at approximately 30, 72 and 171-174 ppm. The exception is the <63 μm fraction from the Bm of the Orthic Dystric Brunisol (279 mg g^{-1} C, Figure 6d); its spectrum has a hump at 103 ppm which does not correspond to any reasonable chemical structure. It is probably an artefact resulting from the subtraction procedure, and therefore its relative intensities (Table 6) are not reliable. For many samples, the aromatic region is both broad and weak, with some samples having a more well-defined peak at 126-130 ppm.

For the Brunisols, the most striking feature is the high alkyl intensity, especially for the <63 μm fractions. For the Ah horizon of the Orthic Dystric Brunisol, alkyl intensity increased from 24.9% for whole soil, to 37.5% for 250-63 μm and 42.3% for <63 μm (Table 6), while the corresponding values for the alkyl/O-alkyl intensity ratio were 0.56, 0.98 and 1.07. As noted previously, the relative areas of the

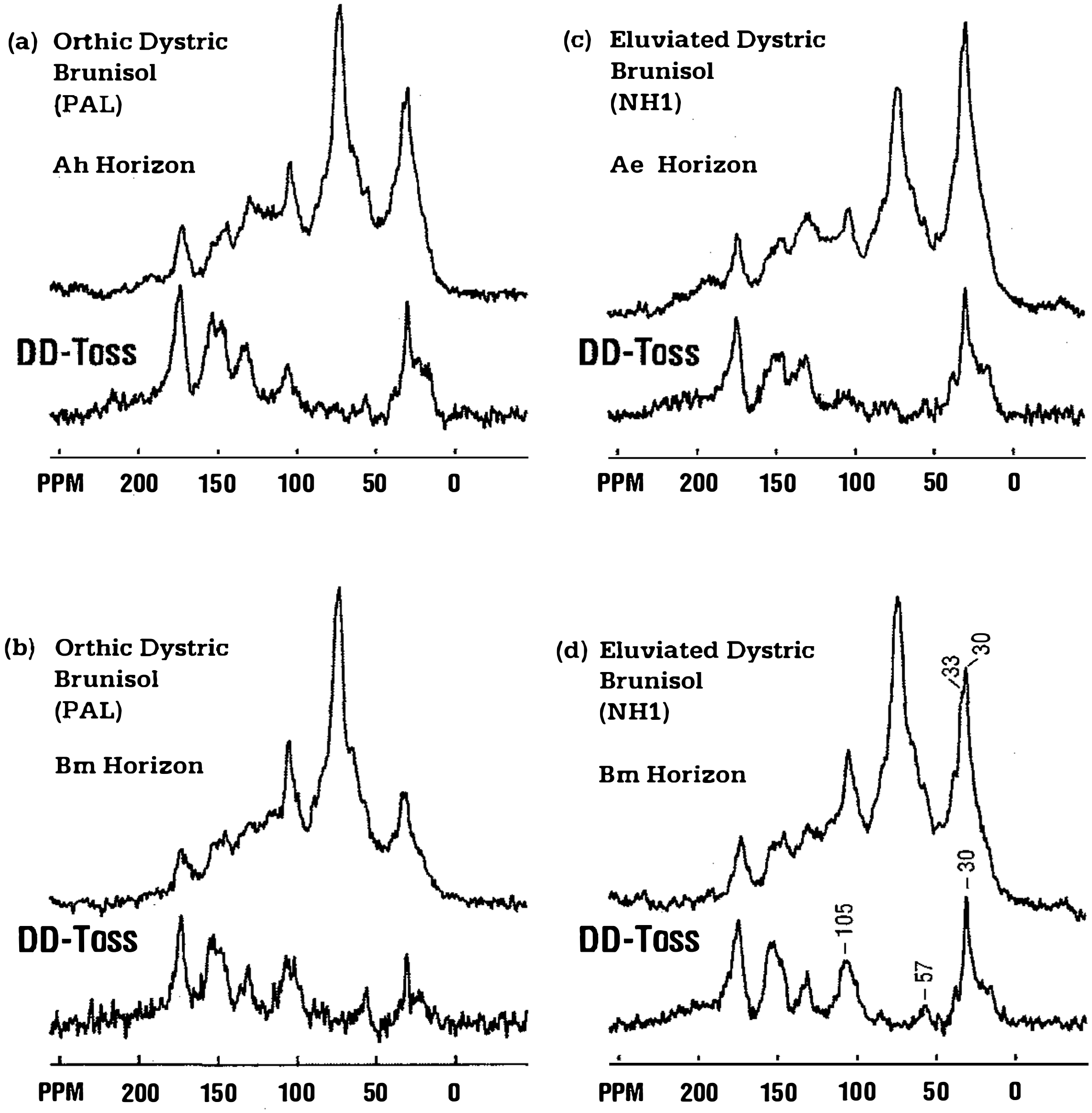

Figure 4. Normal and DD-TOSS ^{13}C CPMAS NMR spectra of floatables from mineral soil of the Orthic Dystric Brunisol (a) Ah and (b) Bm horizons and the Eluviated Dystric Brunisol (c) Ae and (d) Bm horizons .

<63 μm fraction from the Orthic Dystric Brunisol Bm are not reliable but the relative peak heights of the alkyl and O-alkyl peaks are, at least, comparable. The <63 μm fractions from the Ae and Bm horizons of the Eluviated Dystric Brunisol also had high proportions of alkyl C (alkyl/O-alkyl ratios of 1.16 and 0.98, respectively), and low proportions of aromatic + phenolic C (15.4% and 10.6% respectively). The DD spectra that could be obtained had low intensity for non-protonated aromatic C, no residual methoxyl signal (consistent with the low phenolic intensities), and greater intensity for carboxyl C and for more mobile C (maximum at 29-30 ppm), indicating the presence of long-chain CH_2. Aromatic intensities were generally weak, consistent with the normal CPMAS spectra.

In contrast to the Brunisols, the highest enrichment of $\delta^{13}C$ and the best spectra were obtained from the 2000-250 μm and 250-63 μm fractions for the Cryosolic Ah+Bm horizon. Spectra were obtained for all Cryosolic samples (Figure 8), and had lower ratios of

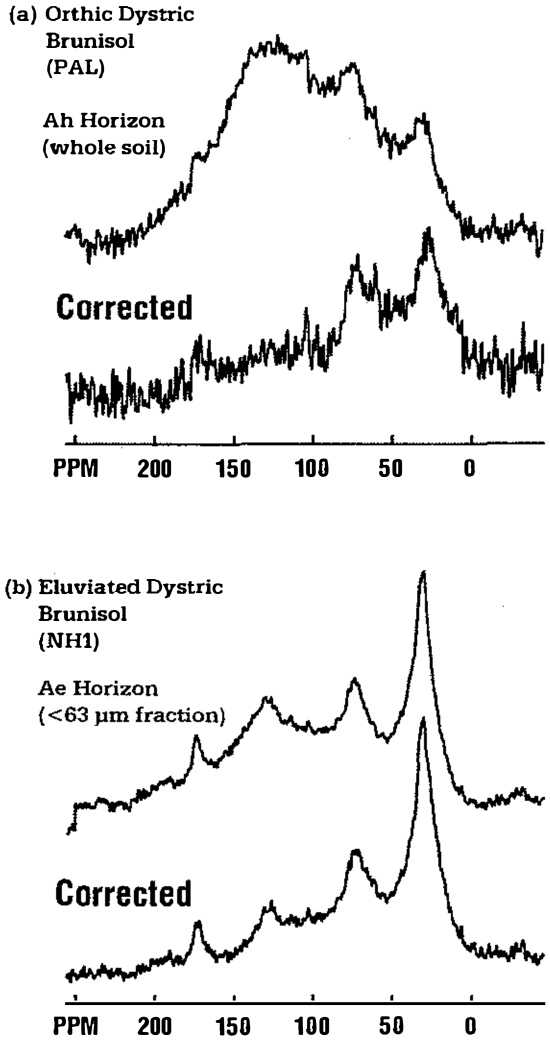

Figure 5. Effect of spectrometer background subtraction for samples low in total C from the A horizons of the (a) Orthic Dystric Brunisol (29.4 mg g^{-1} C) and the (b) Eluviated Dystric Brunisol (123 mg g^{-1} C).

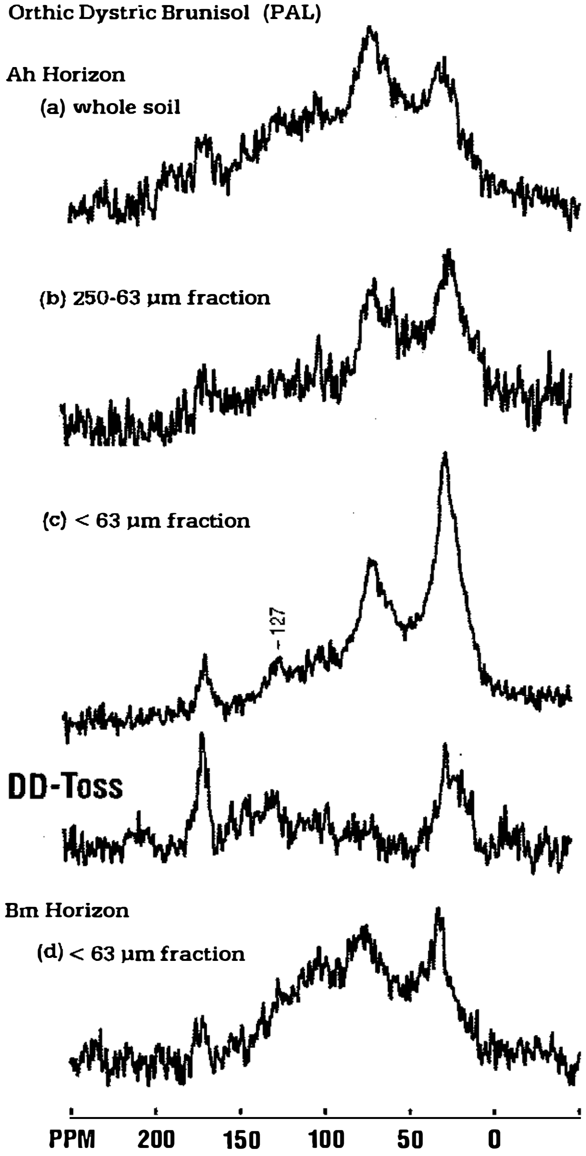

Figure 6. ^{13}C CPMAS and DD or DD-TOSS NMR spectra of mineral SOM from the Orthic Dystric Brunisol Ah horizon (a) whole soil, (b) 250-63 μm, (c) <63 μm; Bm horizon, (d) <63 μm.

alkyl/O-alkyl C and higher aromatic and phenolic C than samples from Brunisols. The normal and DD-TOSS spectra of the 2000-250 μm fraction (Figure 8b, 2.8% of total C) are similar to those of the floatables from the Brunisols (Figure 4) and indicate considerable preservation of plant biopolymers, including long-chain alkyl C (31 ppm), O-alkyl and di-O-alkyl C (74 and 105 ppm, respectively) from carbohydrates, and both lignin and tannin structures, indicated by the broad phenolic signal (144-154 ppm). This is supported by the DD-TOSS spectrum which has a small but sharp methoxyl signal at 58 ppm, and the broad signal characteristic of tannins at 105 ppm. Plant biopolymers may be preserved in these soils through occlusion in structural units formed as a result of cryoturbation and/or soil faunal activity (Brewer and Pawluk, 1975; Smith et al. 1991). Although there was considerable preservation of plant biopolymers in these size fractions, they represent a relatively small proportion of the total C and N in this mineral soil horizon (Table 3).

As noted previously, accumulation of alkyl C is

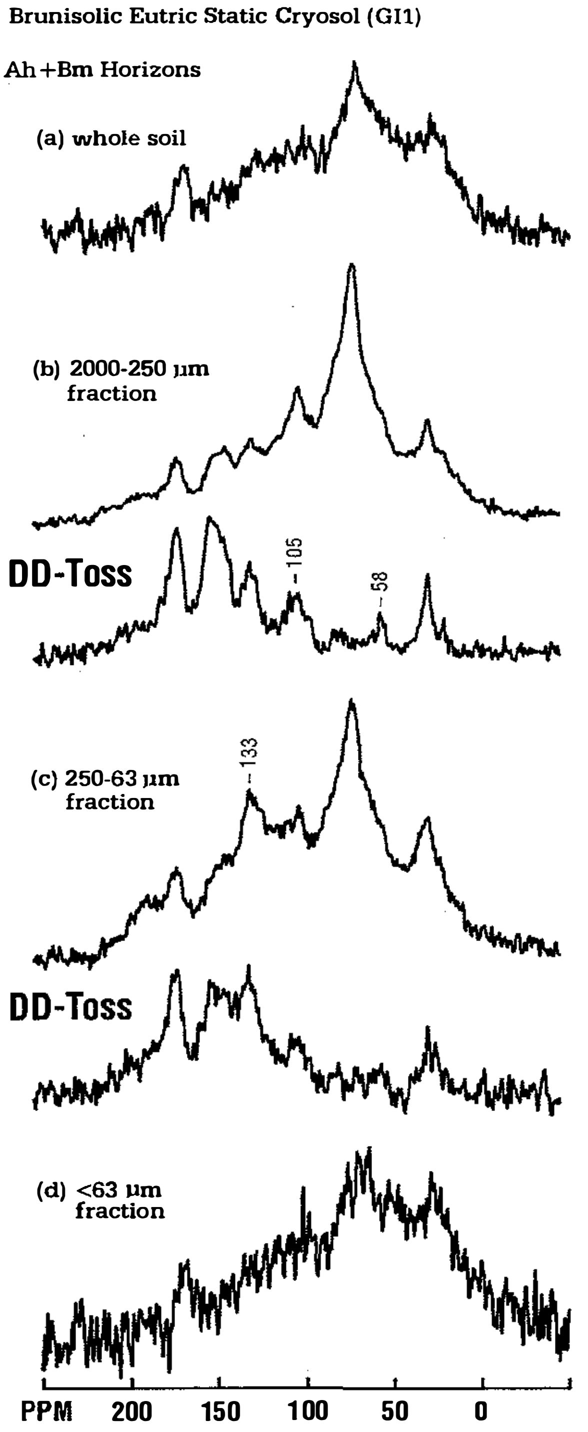

Figure 7. ^{13}C CPMAS and DD or DD-TOSS NMR spectra of mineral SOM from the Eluviated Dystric Brunisol <63 µm fraction from the (a) Ae and (b) Bm horizons.

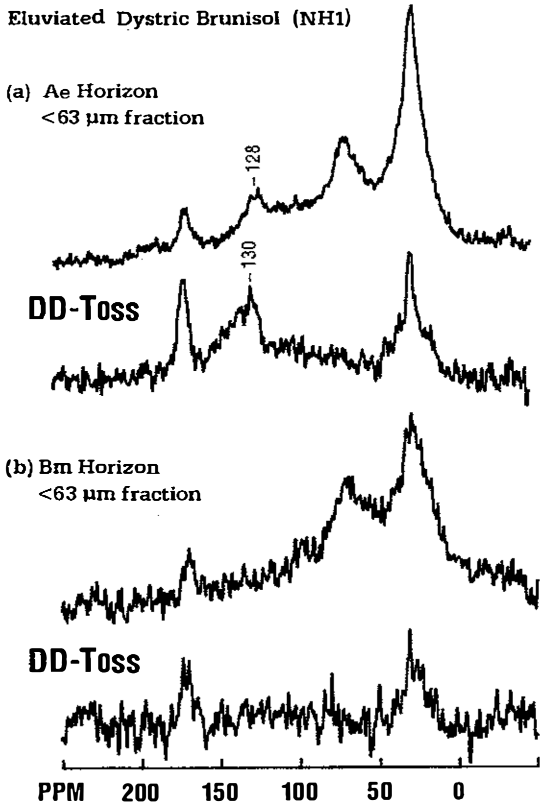

Figure 8. ^{13}C CPMAS and DD or DD-TOSS NMR spectra of mineral SOM from A+B horizon of Gillam 1: (a) whole soil, (b) 2000-250 µm, (c) 250-63 µm, (d) <63 µm.

often associated with restricted decomposition, and Hu et al. (2000) have recently identified a crystalline polymethylene fraction of soil organic matter as a possible contributor to the very slow or passive C pool. Huang et al. (1999) used radiocarbon dating to demonstrate that aliphatic hydrocarbons extractable by dichloromethane constituted a "passive" fraction much older than total organic C. The association of alkyl C with clay has long been known, (Guggenberger et al., 1995; Randall et al., 1995; Schnitzer et al., 1988; Quideau et al., 2000) and in particular, with the clay component of sandy soils (Kögel-Knabner, 2000).

The SOM in the mineral horizons of the Brunisols may have been influenced by fire. A small but well-defined aromatic peak occurs at 127-131 ppm in the normal and DD spectra of the <63 µm fraction of the Ah and Ae horizons from the Brunisols which may be an indicator of pyrogenic carbon (char). The spectrum of the

250-63 µm fraction (Figure 8c, 5.4% of total C) is notable for a sharp aromatic signal at 133 ppm, which may originate from char. The highly condensed aromatic structures of char are poorly detected by CPMAS NMR (Kögel-Knabner, 2000; Preston, 2001; Preston et al., 2002; Schmidt and Noack, 2000; Skjemstad et al., 1999), but it was not possible to obtain a usable BD spectrum from this sample, or others with <200 mg g^{-1} total C.

Pyrogenic C may constitute a very slow or passive soil C fraction, although there is little information on its movement or degradation (Bird et al., 1999; Schmidt and Noack, 2000). Its identity could be confirmed by specific wet-chemical analysis (Schulze et al., 1999) or BD NMR following HF treatment to reduce sample ash (Kögel-Knabner, 2000; Skjemstad et al., 1999). Fire can affect SOM dynamics by promoting particle aggregation which could lead to occlusion of SOM, and in particular "passive" or char C. Hydrophobicity can be promoted by fire through distillation of organics that condense in the deeper, cooler soil layers. (Everett et al., 1995) and hinder further decomposition by limiting access of water. The hydrophobic character of the <63 µm fraction from the sandy-textured Brunisols was demonstrated by an ethanol-drop test that placed them in the range of oil-contaminated soils (C. Langford, U. of Calgary, *pers. comm.*).

Carbon and Nitrogen Stocks

The high degree of climatic variation that accounts for much of the vegetation and productivity differences along the BFTCS could also account for variation in soil organic matter decomposition rates and C and N stocks (Yu et al 2002 and Bhatti et al 2002) The upland Brunisolic Eutric Static Cryosol at the northeastern end of the BFTCS contained the highest amount of C (Table 5) followed by the Orthic Dystric Brunisol and Eluviated Dystric Brunisol at the southwest end. The majority of the C in these upland sites, is in the mineral soil horizons (Table 5). An increase in C sequestration in the mineral soil horizons follows a trend of decreasing lessivage, increase in pH and increase in exchangeable calcium (Table 5). The C and N content for both upland and wetland sites in this study are within the range reported by Bhatti and Apps (2000) for the BFTCS sites and Bhatti et al. (2002) for the region. For the sites considered in this preliminary study, the average annual temperature gradient decreases from the northeast to southwest while total annual precipitation is highest at the subhumid high boreal site followed by the low subarctic and then subhumid low boreal/transitional grassland sites (Table 1).

Soil C stocks are strongly related to soil texture. In general, soils with sandy texture have lower soil C relative to soils with higher clay content (Bhatti and Apps, 2000). In part this is attributable to low detrital inputs to sandy soils due to poor productivity associated with low nutrient and moisture reserves. Oades (1988), however, observed that the positive correlation between clay and soil C content may not represent a true cause-effect relationship because clay content is usually highly correlated with other factors, such as the presence of polyvalent cations, that influence soil C accumulation. Fine-textured soils not only accumulate higher amounts of SOM through binding to clay and silt surfaces, but they preserve a greater proportion of proteins and carbohydrates, and thus, SOM with a more hydrophilic characteristic than sandy soils (Capriel, 1997; Franco et al., 2000).

Soil texture has a major influence on drainage conditions; unstructured clay soils yield poor drainage conditions and reduce decomposition rates (Van Veen and Kuikman, 1990). In this, and other studies along the BFTCS, the largest C stocks occurred in soils in poorly drained conditions (Bhatti and Apps, 2000; Harden et al., 1997; Rapalee et al., 1998; Tarnocai, 1998). Decomposition rates are constrained more by physical conditions (cold and wet) than by inherent recalcitrance of the organic matter structures (Moore and Knowles, 1990). As discussed previously, the alkyl/O-alkyl ratio was found to be low for the organic soils at the Nelson House and Gillam sites which is consistent with high C accumulation on these sites due to restricted decomposition because of unfavorable conditions such as poor drainage.

Drainage conditions also modify the soil temperature which has a major influence on decomposition rates. In another study along the BFTCS the highest rates of decomposition were observed for soils under well-drained conditions (Trumbore and Harden, 1997; Orthic Dystric Brunisol in this study) that experienced average July soil temperature of 13 to 16° C at 10 cm depth(Bubier et al., 1995). The coldest soils on the BFTCS (6° to 10° C at 10 cm depth), found in conjunction with permafrost or poorly drained conditions under mature black spruce stands (Brunisolic Eutric Static Cryosol in this study), had the lowest decomposition rates (Trumbore and Harden, 1997).

CONCLUSIONS

As expected the organic soils have higher carbon stocks than the upland soils (Table 5). SOM of the two Fibrisols is characteristic of poorly decomposed sphagnum with preservation of plant carbohydrate and dominance of condensed tannins rather than lignin. SOM of the Typic Mesisol (calcareous) reflected more advanced decomposition with lower C:N, higher aromatic and lower phenolic content than the Fibrisols. Signals for lignin and tannin were not prominent suggesting that either higher plants had little input to SOM or they were strongly decomposed. $\delta^{13}C$ and BD spectrum results for CO_3-C indicate that inorganic carbon may play a role in carbon sequestration in the calcareous Mesisol.

NMR spectra of the LF(H) from the Brunisols have unusually low aromatic and phenolic intensities for a forest floor and are similar to those for the Fibrisols. This may reflect the dominance of moss inputs to SOM in these soils and little input or persistence of woody debris or tree or shrub litter. Low $\delta^{15}N$ values for the LF(H) of the Brunisolic soils may be indicative of a significant presence of free-living nitrogen fixation activity associated with mosses or lichens and a tight nutrient cycle between the humus form and vegetation.

Within the upland soils C stocks increased from the Eluviated Dystric Brunisol to the Orthic Dystric Brunisol to the Brunisolic Eutric Static Cryosol. The majority of this carbon is in the mineral soil horizons; not in the LF(H) (Table 5). An increase in carbon sequestration in the mineral soil horizons follows a trend of decreasing lessivage, increase in pH and increase in exchangeable calcium (Table 5).

Mineral soil horizons for Brunisols showed no evidence for persistence of lignin, an increase in $\delta^{13}C$ with depth, high alkyl intensity in the $<63\mu m$ fraction and floatables suggest that there may be significant contributions of root and microbial carbon to the mineral soil in Brunisols. $\delta^{13}C$ of the $<63\mu m$ fraction increased with depth for the Orthic Dystric Brunisol but not for the Eluviated Dystric Brunisol. This indicates that the leached, acidic and Ca poor Eluviated Dystric Brunisol has a relatively lower inherent ability to stabilize or sequester carbon compared with the Orthic Dystric Brunisol.

In contrast, there was considerable preservation of plant biopolymers in the Brunisolic Eutric Static Cryosol, although they represent a relatively small proportion of the total carbon in this soil. Most of the carbon in this soil resides in the $<63\mu m$ fraction. The high carbon stocks in the mineral soil of the Cryosol likely reflects the combination of low lessivage, high exchangeable calcium and pH (already mentioned) plus high clay content and the pedoturbation from soil fauna and cryic action. Pedoturbation provides the mechanism to recombine mineral soil and organic matter leading to stabilization of organic matter through clay-organo complexing and occlusion of OM in soil aggregates.

Preliminary results provide further evidence to support the notion that pyrogenic carbon most likely makes a significant, but poorly understood, contribution to carbon sequestration in the Brunisolic soils studied. For mineral horizons in all sites, lignin does not accumulate, and increases in aromaticity may be more influenced by char.

Our results are consistent with previous studies showing considerable heterogeneity in different soil compartments, and a shift to greater microbially-derived C in the finer, denser size fractions (Bird and Pousai, 1997; Cayet and Lichtfouse, 2001; Quideau et al., 2000; Trumbore, 2000). Further work should prepare finer size fractions, and concentrate their organic matter by HF treatment. Litter and root inputs also need to be characterized, and molecular-level analysis is necessary to confirm the broad trends shown by NMR. This approach to the study of SOM sequestration in forest soils should be extended to a broader suite of forest soil types.

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LIVING AND DEAD WOOD CARBON DYNAMICS IN PRISTINE BOREAL NORWAY SPRUCE FORESTS SUBJECTED TO WINDTHROW DISTURBANCES

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ABSTRACT

Today, in view of global environmental changes, investigations into the structure, function and dynamics of pristine forests are especially topical. Results of these investigations may be used to improve estimates of the role of forests in the global carbon budget, to optimize management strategies to maintain the productivity and stability of existing forests, to restore degraded forests and to protect biodiversity in forests. Middle boreal pristine Norway spruce (*Picea abies* (L.) Karst.) forests were studied. Experimental data were collected in the "Vepssky Les" reserve (Saint Petersburg region, Russia). The main types of stands in terms of site conditions, age structure and dynamic phase following windthrow disturbance were explored. The carbon (C) dynamics of living trees and coarse woody debris (CWD) including stems, roots, branches and bark were described using data from permanent sample plots. A 5-20 year period was taken into account. Carbon stored in living trees ranged from 29.9 to 171.5 Mg C ha^{-1}, and in CWD ranged from 4.5 to 61.5 Mg C ha^{-1}. The average annual stand increment was 0.1–2.5 Mg C ha^{-1} yr^{-1}. The average annual mortality was 0.0-4.6 Mg C ha^{-1} yr^{-1}. The average annual carbon flux of CWD decomposition varied from 0.1 to 0.9 Mg C ha^{-1} yr^{-1}. It may be concluded that windthrow disturbance regimes strongly influence living and dead wood carbon dynamics in these pristine forests.

INTRODUCTION

Data on the structure, function and dynamics of pristine forests are essential for many scientific and applied purposes including; i) optimizing the productivity and stability of forests, ii) conservation and restoration of biodiversity and, iii) choosing a standard ecological monitoring system, etc. (Schmidt-Vogt 1985; Backmeroff & Peterken 1988; Lähde et al. 1998; Angelstam 1997; Hansson & Larsson 1997; Linder et al. 1997; Seymour & Hunter 1999; Bengtsson et al. 2000).

Today, in view of global environmental changes, the problem of estimating the role of forests in the carbon budget is especially topical. Because boreal pristine forests are in the late successional state, they are unable to quickly respond to high atmospheric CO_2 levels. Carbon stock changes are likely to consist of long-term carbon accumulation in phytomass, coarse woody debris (CWD) and soil (Harmon et al. 1986; Kobak 1988; Utkin 1995). Some carbon cycling parameters in second-growth managed forests are known, and models of carbon dynamics at the regional level have been developed (Apps et al. 1991; Kolchugina & Vinson 1991; Alexeyev et al. 2000; Harmon 2000; Korovin & Isaev 2000; Nilsson & Shvidenko 2000; Utkin et al. 2000). These models are based on forest inventory records, the volume of harvest of forest products, data obtained from sample plots established chiefly in second-growth forests and historical disturbance data.

However, this is insufficient for a full description of carbon dynamics during successional changes of pristine forests because information on the different carbon stores, the rates of carbon release and uptake, the carbon balance of forests in different states, and disturbance regimes for each successional stage in each ecosystem type, ideally should also be taken into account. A pristine forest may be revealed only on a quite large area, where human activity in the neighboring territories does not affect its functional integrity (Leibundgut 1993). One has to consider a pristine landscape, or its parts -

woodland, as a spatial complex of different stands. Before describing the carbon dynamics of a pristine landscape, an understanding of the unique functional characteristics of the stands is needed. The questions are; i) What factors influence the carbon dynamics at the stand level? ii) What parameters of carbon cycling are to be considered in the first place? and consequently, iii) How to classify the stands for a carbon cycling study?

One of the major boreal forest ecosystem compartments is wood. Presumably, the carbon fluxes connected with living and dead wood which include growth, mortality and CWD decomposition, largely determine the long-term carbon dynamics of boreal forests. This study focuses on long-term ecosystem development and represents a preliminary attempt to estimate carbon dynamics of living trees and CWD in pristine European boreal Norway spruce (*Picea abies* (L.) Karst.) forests at the stand level. The main objectives were; i) estimation of the carbon (C) stored in living trees and CWD including stems, roots, branches and bark, and ii) estimation of the carbon fluxes of growth, mortality and CWD decomposition, in stands of different eco-types, structures and dynamic states.

MATERIAL AND METHODS

Study Sites

The experimental study was conducted in the "Vepssky Les" reserve situated in the Saint-Petersburg region of Russia, within the middle boreal zone (35°09' - 35°13' E, 60°14' - 60°11' N; Figure 1). A description of the reserve is given by Fedorchuk et al. (1998). The reserve represents a part of Vepsovsky Hill (the northern part of Valdaisko-Onezhskaya ridge). Elevation is mainly between 220 – 260 m, rising to a maximum of 300 m. The soils are mostly coarse humus podzolic sandy-loam and loams on morainal tills. The climate is less warm (average annual temperature is + 2.6°C) and more humid (annual precipitation is over 750 mm) than in other Northwestern European landscapes. Low average monthly temperatures, frequent occurrence of frosts, short growing season and high precipitation (including seasonal snow cover), typify the local climate of Vepsovsky Hill.

The forests of the "Vepssky Les" woodland have never been commercially harvested providing a unique opportunity to study pristine forest ecosystems.

Dominant eco-types of these pristine forests are classified as *Vaccinium myrtillus*, *Sphagnum-Myrtillus* and *Polytrichum-Myrtillus*. The shrub layer of the *Vaccinium myrtillus* forest is characterized by *Juniperus communis* (L.), *Sorbus aucuparia* (L.), *Frangula alnus* Mill, and *Alnus incana* (L.) Moench. *Vaccinium myrtillus* L., *V. vitis-idae* L., *Pteridium aquilinum* (L.) Kuhn., *Melampirum sylvaticum* L., *Deshampsia flexuosa* L., *Majanthemum bifolium* (L.) F.W. Schmidt, *Dryopteris dilatata* (Hoffm.) A. Gray, dominate the herb-dwarf-shrub layer. The dominant

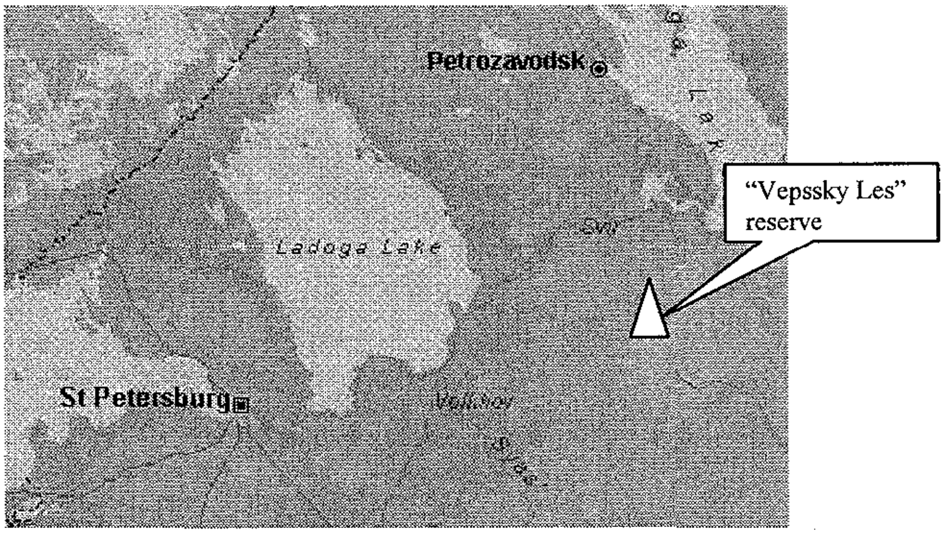

Figure 1. Location of the Vepssky reserve.

mosses are *Pleurozium schreberi* (Brid.) Mitt., *Hylocomium splendens* (Hedw.) B.S.G., and *Dicranum* Hedw. spp. Shrub layers of *Sphagnum-Myrtillus* and *Polytrichum-Myrtillus* forest sites are represented by *Sorbus aucuparia, Salix carpea* L., and *S. cinerea* L. The *Polytrichum-Myrtillus* forest is characterized by the presence of *Vaccinium myrtillus, V. vitis-idae, Majanthemum bifolium, Equisetum sylvaticum* L., while *Dryopteris dilatata* dominates the herb-dwarf-shrub layer and *Pleurozium schreberi* (Brid.) Mitt., *Hylocomium splendens* (Hedw.) B.S.G. and *Sphagnum* (L.) spp. dominate the moss layer. The *Sphagnum-Myrtillus* forest site is characterized by the presence of *Vaccinium myrtillus, V. vitis-idae, Carex globularis* L., and *Rubus chamaemorus* L. which dominate the herb-dwarf-shrub layer and *Sphagnum* spp. dominates the moss layer.

Dominant tree species in all these forest sites include Norway spruce (*Picea abies* (L.) Karst., *P× fennica* (Regel.) Kom.), birch (*Betula pubescens* Ehrh.), aspen (*Populus tremula* L.), with the rare occurrence of Scots pine (*Pinus sylvestris* L.). In the very few stands where deciduous species are dominant, Norway spruce is found in the second vegetation layer. Presumably, these stands originated in large (more than 0.2 ha) gaps after severe windthrows events. Pine stands occur on sphagnum sites chiefly as ecotones between bogs and spruce forests. Single old Scots pine trees established after fire form part of some spruce stands on well-drained sites. However, windthrow, not fire, is the main type of disturbances in the "Vepssky Les". Most of the forested area experienced windthrow events about 140, 160 and 180 years ago, since most of the even-aged stands are of these ages, and the uneven-aged stands observed have these age groups present. Recently, the territory was destroyed by wind in November of 1983 and in 1985 and 1986. The total percentages of gaps, more than 0.015 ha, were 0.5% of the woodland area in 1970, 0.3% in 1981 and 9.8% in 1991.

Prior to sampling, all the stands were classified by site conditions, age structure and dynamic phase. The main types of the stand age structures were distinguished by Dyrenkov (1984) as follows:

1. Relative even-aged stands (from here on referred to as even-aged). The age variation of the overstory trees does not exceed 40 yrs. The diameter and age distributions are bell-shaped, or normal.
   At the present time all even-aged stands in the "Vepssky Les" are old-growth, with an average age over 140 years. The maximum life-span of Norway spruce is about 400-450 yrs. Even-aged old-growth forests amount to 26.3% of the total woodland area (Fedorchuk et al. 1998).
2. Relative uneven-aged stands (from here on referred to as uneven-aged). The population of the predominant tree species is continuous up to over 200 yrs, but 50-90% of the total biomass is concentrated in one of the 40 year old cohorts. The diameter and age distributions have "several bells". Young or old cohorts may dominate in these stands and few peaks, or waves, are pronounced in their dynamics.
   In the "Vepssky Les" uneven-aged Norway spruce stands are associated mainly with *Vaccinium myrtillus* sites and, sometimes, with *Polytrichum-Myrtillus* and *Sphagnum-Myrtillus* sites. Windthrow is a key factor in their dynamics. 48.7% of the total "Vepssky Les" woodland area is occupied by uneven-aged stands which are dominated by young cohorts and, consequently, their total biomass is increasing. 10.8% of the total "Vepssky Les" woodland area is occupied by uneven-aged stands which are dominated by old cohorts and, consequently, their total biomass is decreasing (Fedorchuk et al. 1998).
3. Absolutely uneven-aged stands (from here on referred to as all-aged). All 40 year groups of the predominant tree species population up to the life-span limit are presented, but no one of them exceeds 40% of the total biomass. The diameter and age distributions are exponential or reverse J-shaped. The tree age spatial distribution may be even or patchy. Successional development of these stands is relative even, without clear peaks. In the "Vepssky Les" the all-aged stands occur only on *Sphagnum-Myrtillus* sites and amount to 14.2% of the total woodland area (Fedorchuk et al. 1998).

Three main phases following disturbances, such as windthrow events, may be distinguished in the dynamics of Norway spruce stands. These include growing stock (GS) increasing, GS stabilization and GS decreasing (Fedorchuk et al. 1998). These dynamic phases are pronounced in even-aged and in uneven-aged stands, and are less clear in patchy all-aged stands.

Sampling

The experimental data were collected from 21 sample plots ranging from 0.1 to 0.2 ha. The sample plots were established on permanent sample plots of the Saint-Petersburg Forestry Research Institute in pristine Norway spruce forests (Table 1). These forests represent different types of age structures (all-aged, uneven-aged and even-aged); each at the different dynamic phases following disturbances (GS increasing, GS stabilization and GS decreasing). The plots also represent different ecotype (*Vaccinium myrtillus, Polytrichum-Myrtillus,* and *Sphagnum Myrtillus*).

Estimation of Carbon Stores

Carbon stored in the living trees' stems in each sample plot was estimated by multiplying GS separately for each tree species (data from Saint-Petersburg Forestry Research Institute) by wood bulk density. Stand GS was estimated as a function of the average height and basal area, independent of age and site conditions. It was determined using special forest mensuration tables developed by Moshkalev (1984). The species height curves were constructed for each sample plot using 3-4 replicates of height measurements for each diameter class (4-cm interval). The average height corresponded to the average basal diameter, which had been calculated from the experimental measurements of diameters of all trees.

The bulk density of stem wood was calculated on an oven dry basis (105°C) and standard volume. For cross-section sampling we used trees recently uprooted and not colonized by wood decay fungi. 80 cross-sectional samples were taken for Norway spruce, 16 for birch, and 10 for aspen.

The carbon of living trees' coarse roots and branches was calculated by multiplying GS by the special coefficients depending on tree species and stand (or cohort) age. Tarasov (1999) generalized the data of many authors to obtain the following equations for the Saint-Petersburg region:

Norway spruce roots carbon store =
$0.5 * \text{stem volume} * 0.396 * \text{age}^{(-0.345)}$
$r^2 = 0.6$; N = 49;

Norway spruce branches carbon store =
$0.5 * \text{stem volume} * 0.662 * \text{age}^{(-0.559)}$
$r^2 = 0.6$; N = 70;

Birch roots carbon store = 0.5 * stem volume
$* (0.121 - (0.005 * 10^{-3})* \text{age} + 2/\text{age})$;

Birch branches carbon store =
0.5 * stem volume *
$(0.051 - (0.21 * 10^{-3}) * \text{age} + 2/\text{age})$;

Aspen roots carbon store = 0.5 * stem volume
$* (0.3 - (0.2 * 10^{-3}) * \text{age} + 1/\text{age})$;

Aspen branches carbon store =
0.5 * stem volume *
$(0.09 - (0.02 * 10^{-3}) * \text{age} + 1/\text{age})$;

Scots pine roots carbon store =
0.5 * stem volume *
$(0.079 - 0.00016 * \text{age} + 0.953/\text{age})$
$r^2 = 0.4$; N = 62;

Scots pine branches carbon store = 0.5 *
$\text{stem volume} * 0.161 * \text{age}^{(-0.349)}$
$r^2 = 0.5$; N = 90;

There is a lack of information on birch and aspen phytomass in Russian boreal forests. Tarasov (1999) developed these equations using corrected data from Zamolodchicov et al. (1998). Values for r^2 and N were not always available.

Bark carbon was calculated separately for different tree species by multiplying bark volume (as definite percent by volume from the stem, and, consequently, from the stand: Moshkalev 1984) by bark bulk density.

It is necessary to note that this approach does not take into account the stem and root rots. The percent of rotten living trees in pristine forests may be very high. Thus, some overestimation may have occurred in our estimates of stand carbon stores because the bulk density of rotten wood is less than the bulk density of sound wood. We assume all wood to be sound in our calculations.

CWD carbon was estimated from the inventory of all above-ground CWD (not buried) for each sample plot using the guidelines of Harmon (1996). A total of more than 100 CWD units for each sample plot was inventoried. The CWD species composition varied from 100% Norway spruce to 30-40% birch and aspen, reflecting the present and historical tree species composition (see Table 1 for living tree species composition). In

Table 1. Characteristics of the sample plots.

sample plot number *, year established	Tree species composition (biomass), %	Type of site conditions	Tree basal area, m^2/ha	Age structure type, age of main cohort	Dynamic phase (GS - growing stock)
46, 1991	N. spruce 70, birch 30	*Vaccinium myrtillus*	35.7	even-aged, 170	GS stabilization
100, 1973	1st layer: Aspen 30, Sc. pine 30, birch 10; 2nd layer: N. spruce 100	*Vaccinium myrtillus*	40.4	even-aged, 140	GS stabilization
3, 1991	N. spruce 76, birch 24	*Vaccinium myrtillus*	30.6	even-aged, 180	GS decreasing
16, 1991	N. spruce 57, birch 10, aspen 33	*Vaccinium myrtillus*	31.5	even-aged, 200	GS decreasing
15, 1991	N. spruce 55, Sc. pine 28, birch 12, aspen 5	*Vaccinium myrtillus*	36.8	even-aged, 200	GS increasing
17, 1991	N. spruce 60, Sc. pine 38, birch 2	*Vaccinium myrtillus*	41.5	even-aged, 200	GS increasing
49, 1991	N. spruce 91, birch 9	*Vaccinium myrtillus*	26.4	uneven-aged, 180+250	GS stabilization
198, 1981	N. spruce 86, birch 14	*Sphagnum-Myrtillus*	37.5	uneven-aged, 150	GS stabilization
11, 1981	N. spruce 95, birch 5	*Polytr.-Myrtillus*	35.3	uneven-aged, 140	GS stabilization
10, 1977	N. spruce 94, birch 6	*Vaccinium myrtillus*	17.4	uneven-aged, 200	GS decreasing
73, 1991	N. spruce 80, birch 20	*Vaccinium myrtillus*	40.8	Uneven-aged, 160+200, 250	GS decreasing
72, 1991	N. spruce 62, birch 34, aspen 4	*Vaccinium myrtillus*	27.8	uneven-aged, 250+160, 100	GS decreasing
8, 1977	N. spruce 96, birch 2, aspen 2	*Vaccinium myrtillus*	36.1	uneven-aged, 100	GS increasing
4, 1991	N. spruce 91, birch 9	*Sphagnum-Myrtillus*	40.8	uneven-aged, 180+270	GS increasing
44, 1991	N. spruce 88, birch 12	*Vaccinium myrtillus*	8.3	uneven-aged, 50+220	GS increasing
98, 1973	N. spruce 90, Sc. Pine 2, birch 8	*Sphagnum-Myrtillus*	27.9	all-aged (160)	GS stabilization
50, 1991	N. spruce 68, birch 32	*Sphagnum-Myrtillus*	28.2	all-aged (180)	GS stabilization
53, 1991	N. spruce 86, Sc. pine 9, birch 5	*Sphagnum-Myrtillus*	30.8	all-aged (150)	GS stabilization
2, 1991	N. spruce 94, birch 6	*Sphagnum-Myrtillus*	26.2	all-aged (180)	GS decreasing
91, 1973	N. spruce 92, birch 8	*Sphagnum-Myrtillus*	28.0	all-aged (140)	GS decreasing
9, 1977	N. spruce 94, birch 6	*Sphagnum-Myrtillus*	30.8	all-aged (160)	GS increasing

* The sample plots were numbered according to the numbers assigned by researchers at Saint Petersburg For. Res. Inst.

addition, the volume, decomposition stage and type of wood pieces colonized by wood decay fungi were determined. CWD volumes (each CWD unit was considered as cylinder, cone or frustum of cone) were converted into carbon stores values by multiplying bulk density which varied according to the 5-decay class system of Shorohova & Shorohov (1999, 2001). The carbon stored in coarse dead roots and branches was estimated using special coefficients and decomposition models developed by Tarasov (1999, 2000); a logistic (S-shaped) decomposition model for Norway spruce, a lag-exponential one for birch and an exponential one for aspen. The CWD distributions amongst decay classes and tree species and experimental 25-yr decomposition curves for *Picea abies* and *Betula pubescens* were also used. The decomposition curves were built from chronosequences of *Picea abies* (452) and *Betula pubescens* (37) samples aged from permanent sample plot records (Shorohova & Shorohov 2001). For example, we know the volume distribution of Norway spruce CWD in the 5 decay classes (experimental data from sample plot CWD inventory) and the average decomposition time for each decay class (Shorohova & Shorohov 1999, 2001). We can estimate the average percent of mass loss for each year on the basis of the decomposition curve, if the decomposition time is less than 25 years, or on the basis of Tarasov's model, if the decomposition time is over 25 years. Thus, the roots carbon may be calculated as:

Norway spruce roots carbon store =

Sum $_{i=1-5}$ (0.5 * CWD stem volume of i-th decay class * 0.396 * age $^{(-0.345)}$ * mass loss percent in i-th decay class).

The carbon of CWD coarse roots of other species and the carbon of CWD branches were calculated by analogy. However, our estimations of decomposition of coarse roots and branches were quite rough, because the same rates of decomposition were assumed for the stem wood, roots and branches (for separate diameter groups with intervals of 10 cm).

Dead bark carbon was estimated from "non-decomposed" bark (evaluated by analogy to the living bark carbon, but from CWD volume), bark decomposition–fragmentation models (Tarasov 1999, 2000) and CWD distributions among decay classes and tree species.

Estimation of Carbon Fluxes

The annual carbon increment of the living trees stems for each sample plot was calculated separately for different tree species by multiplying stand GS increment (the data from Saint-Petersburg Forestry Research Institute over 5 – 25 years) by bulk density. The annual carbon loss from living trees stem via mortality was calculated by multiplying stand mortality by bulk density. The increment and mortality of bark, coarse roots and branches were calculated by analogy.

The carbon fluxes of CWD decomposition for each sample plot were estimated separately for all CWD fractions (stems, bark, coarse roots and branches) using data from current CWD, and the experimental decomposition curves and decomposition models already described.

RESULTS

Carbon Stores

The carbon stores in living trees and CWD of different pristine Norway spruce stand types are given in Tables 2a, 2b and 3. The total carbon store in living trees ranged from 29.9 to 171.5 MgC ha^{-1} (Table 2a) depending on the interrelating factors of site condition, age structure (and, consequently, the successional stage of the ecosystem) and dynamic phase of the stand affected by disturbance.

The maximal carbon stores were achieved in even-aged stands in the GS-increasing and GS-stabilization phases. All old-growth even-aged stands of our sample plots were on well-drained *Vaccinium myrtillus* sites associated with steep slopes. In the "Vepssky Les" woodland these stands are strongly affected by windthrow. Intervals between windthrow events are shorter than the life-span of Norway spruce. Therefore, these stands are generally unable to reach the all-aged state. Comparatively low carbon stores in living trees were observed in all-aged stands of all dynamic phases. Stands on boggy *Sphagnum-Myrtillus* sites, associated primarily with low topographic positions, are the only all-aged on the "Vepssky Les" territory. Thus, we could not distinguish the effects of site conditions and age structure on the living trees carbon store, since it was impossible to compare stands of the same age structure, that were not associated with the same forest sites. The uneven-aged stands represent a variety of ecosystems distinguished by composition, structure and dynamic phase. The carbon stores of these stands varied greatly.

The total carbon stores in CWD ranged from 4.5 to 61.5 MgC ha^{-1} (Table 2b).

Table 2a. Carbon stores (Mg C ha^{-1}) in different fractions of the living trees.

Sample plot number	Distribution by species				Distribution by fractions				Total
	Norway spruce	Scots pine	Birch	Aspen	Stems	Roots	Branches	Bark	
46	82.5	0.0	46.0	0.0	91.2	18.5	8.3	10.5	128.5
100	41.2	24.3	17.6	88.4	104.0	41.4	12.9	13.2	171.5
3	65.7	0.0	29.5	0.0	68.3	13.4	6.0	7.5	95.2
16	69.0	0.0	15.8	62.5	93.6	32.3	10.6	10.8	147.3
15	64.2	31.0	19.0	9.4	89.2	17.4	7.4	9.6	123.6
17	77.2	48.6	3.3	0.0	97.7	14.7	7.4	9.3	129.1
49	76.1	0.0	10.0	0.0	63.9	10.9	5.4	5.9	86.1
198	85.4	0.0	23.1	0.0	79.4	14.4	7.1	7.6	108.5
11	103.7	0.0	8.7	0.6	83.8	14.4	7.9	6.9	113.0
10	46.8	0.0	9.0	0.9	41.6	7.5	3.5	4.1	56.7
73	102.9	0.0	33.7	0.0	99.5	18.4	8.4	10.3	136.6
72	64.8	0.0	45.7	5.8	78.2	20.1	8.3	9.7	116.3
8	104.9	0.0	2.3	4.8	79.8	15.3	9.9	7.0	112.0
4	113.5	0.0	13.3	0.0	96.1	15.1	7.5	8.1	126.8
44	25.6	0.0	4.3	0.0	19.5	5.1	3.4	1.9	29.9
98	55.3	0.9	7.5	0.0	47.7	7.9	4.1	4.0	63.7
50	47.5	0.0	27.6	0.0	54.1	10.4	4.6	6.0	75.1
53	71.9	7.2	4.6	0.0	62.6	10.2	5.6	5.3	83.7
2	77.3	0.0	5.8	0.0	63.0	9.9	5.1	5.1	83.1
91	71.8	0.0	7.2	0.0	58.5	10.1	5.5	4.9	79.0
9	74.5	0.0	7.5	0.0	61.3	10.2	5.4	5.1	82.0

Table 2b. Carbon stores (Mg C ha^{-1}) in different fractions of the CWD.

Sample plot number	Distribution by species				Distribution by fractions				Total
	Norway spruce	Scots pine	Birch	Aspen	Stems	Roots	Branches	Bark	
46	2.6	0.0	10.0	0.0	9.7	1.2	0.4	1.2	12.5
100	8.8	6.7	0.9	20.5	27.0	5.4	2.2	2.3	36.9
3	27.2	0.0	5.0	0.0	27.1	2.4	0.9	1.8	32.2
16	10.2	1.5	9.4	9.4	23.5	3.6	0.7	2.7	30.5
15	2.3	0.0	2.0	0.2	3.4	0.5	0.2	0.4	4.5
17	16.2	0.0	3.4	0.9	17.1	1.6	0.7	1.1	20.5
49	12.5	0.0	0.9	0.0	11.4	1.0	0.4	0.6	13.4
198	23.2	0.0	1.0	0.0	20.5	1.7	1.0	1.0	24.2
11	18.0	0.0	0.9	0.0	16.2	1.5	0.2	1.0	18.9
10	26.9	0.0	1.4	0.0	24.3	2.0	0.8	1.2	28.3
73	20.9	0.0	0.7	0.0	18.4	1.6	0.6	1.0	21.6
72	55.2	0.4	6.3	0.0	51.0	5.2	2.5	2.8	61.5
8	10.1	0.0	0.4	0.2	9.1	1.0	0.4	0.6	11.1
4	7.1	5.9	0.5	0.0	6.4	0.6	0.2	0.4	7.6
44	35.1	0.0	1.5	9.5	41.5	5.5	2.8	2.2	52.0
98	10.1	0.0	0.2	0.0	8.8	0.7	0.3	0.5	10.3
50	31.5	0.0	3.8	0.0	30.0	2.4	1.2	1.7	35.3
53	17.3	0.0	4.7	0.0	18.3	1.7	0.7	1.3	22.0
2	14.8	0.0	4.6	0.0	16.2	1.5	0.6	1.1	19.4
91	28.2	0.0	0.4	0.0	24.2	2.1	1.0	1.3	28.6
9	8.8	0.0	0.3	0.0	7.7	0.6	0.3	0.5	9.1

Table 3. Living and dead wood carbon stores (Mg C ha^{-1})* in the stands of different types.

Dynamic phase	Age structure and site conditions: Old-growth even-aged *Vaccinium myrtillus* stands	Uneven-aged *Vaccinium myrtillus* stands	Uneven-aged *Sphagnum-Myrtillus* and *Polytrichum-Myrtillus* stands	All-aged *Sphagnum-Myrtillus* stands
		Living trees		
GS increasing	126.4 ±2.75 (2)	71.0 ±41.05 (2)	126.8 (1)	82.0 (1)
GS stabilization	150.0 ±21.50 (2)	Na**	102.5 ±8.11 (3)	74.2 ±5.79 (3)
GS decreasing	121.3 ±26.05 (2)	103.2 ±23.98 (3)	Na	81.1 ±2.05 (2)
		CWD		
GS increasing	12.5 ±8.00 (2)	31.6 ±20.45 (2)	7.6 (1)	9.1 (1)
GS stabilization	24.7 ±12.20 (2)	Na	18.8 ±3.17 (3)	22.5 ±7.22 (3)
GS decreasing	31.4 ±0.85 (2)	37.1 ±12.34 (3)	Na	24.0 ±4.60 (2)
		Living trees + CWD		
GS increasing	138.8 ±10.75 (2)	102.5 ±20.60 (2)	134.4 (1)	91.1 (1)
GS stabilization	174.7 ±33.7 (2)	Na	121.4 ±10.94 (3)	96.7 ±11.43 (3)
GS decreasing	152.6 ±25.20 (2)	140.3 ±28.24 (3)	Na	105.1 ±2.55 (2)

*Values are means ±SE with sample size in parentheses

** Na = not available (no sample plots)

The total CWD stores, as well as distributions amongst CWD types (namely snags, natural stumps, fallen and leaning stems) and decay classes depended mainly on the stand dynamic phase. Our results suggest that CWD stores are highest in ecosystems with the phase of GS decreasing (i.e. following windthrow disturbances) (Table 3). This phase lasts from one to few decades depending on the intensity of the windthrow event, and on the state of the stand at that time. In the case of severe windthrow in even-aged stands without spruce seedlings and suppressed trees under the canopy, the duration of this phase will be maximum. This is because of the long time required to establish new seedlings and for their biomass to accumulate and increase the total GS. In the case of a mid-intensity windthrow event in uneven-aged or all-aged stands, with many young and suppressed Norway spruce trees under the canopy, the duration of this phase will be minimal. This is because the subsequent GS increase will result from a growth release of the young and suppressed trees. The lowest CWD stores were observed in ecosystems in the phase of GS increasing (Table 3), where decomposition of post-disturbance CWD is finished, and current mortality rates are low. The duration of this phase is restricted either by the ecological features of the predominant tree species – Norway spruce, or by subsequent windthrow disturbance. The second factor acts more frequently. Therefore, the GS stabilization phase may not be pronounced in the dynamics of some stands.

Age structure also affected CWD stores, since dynamic phases and successional processes are expressed differently in the stands with different age structures. The proportions of different fractions (stems, roots, branches and bark in the living trees) and CWD carbon were influenced by the species composition of living trees. CWD was also influenced by the age of predominant tree cohort (Tables 1 and 2).

Carbon Fluxes

The carbon fluxes in Norway spruce stands of different types are given in Tables 4 a-c and Table 5. Estimates for average annual stand carbon increment changed with forest site and ranged from 0.1 to 2.5 MgC ha^{-1} yr^{-1} (Table 4a). This flux averaged 1.1 (0.1-2.3) MgC ha^{-1} yr^{-1} in the Norway spruce stands on *Sphagnum-Myrtillus* and *Polytrichum-Myrtillus* sites and 1.2 (0.3-2.5) MgC ha^{-1} yr^{-1} in the Norway spruce stands on *Vaccinium myrtillus* sites. The annual mortality ranged from 0.0 to 4.6 MgC ha^{-1} yr^{-1} (Table 4b). Mortality values differed depending on the dynamic phase and site characteristics (i.e. well-drained vs. boggy sites). Mortality values were similar in all forest types for GS increasing (0.6,0.7) and GS stabilization (1.5, 1.3) (Table 5). In the GS decreasing phase the values were similar to those for GS stabilization except for the *Picea abies Vaccinium myrtillus* type, where the

Table 4a. Stand carbon fluxes (Mg C ha^{-1}) associated with growth.
(see Note bottom of Table 4c)

Sample Plot Number	Norway spruce	Scots pine	Birch	Aspen	Total	Average annual
2	5.8	0.0	2.3	0.0	8.1	1.6
3	4.5	0.0	0.9	0.0	5.4	1.1
4	11.3	0.0	0.2	0.0	11.5	2.3
15	4.1	0.0	2.0	0.0	6.1	1.2
16	6.7	0.0	0.0	2.5	8.8	1.8
17	6.2	1.9	0.0	0.0	8.1	1.6
44	4.7	0.0	0.0	0.0	4.7	1.0
46	3.5	0.0	1.1	0.0	4.6	0.9
49	1.3	0.0	0.0	0.0	1.3	0.3
50	2.6	0.0	1.0	0.0	3.6	0.7
53	0.6	0.0	0.0	0.0	0.6	0.1
72	0.6	0.0	2.1	0.0	2.7	0.5
73	1.5	0.0	1.3	0.0	2.8	0.6
198	7.3	0.0	4.7	0.0	12.0	0.8
11	16.5	0.0	2.7	0.0	19.2	1.2
10	13.1	0.0	3.4	0.2	16.7	1.0
8	28.8	0.0	0.3	3.9	33.0	2.1
9	17.1	0.0	2.6	0.0	19.7	1.2
100	10.8	3.9	4.3	32.8	51.8	2.5
98	17.5	0.2	2.1	0.0	19.7	1.0
91	20.1	0.0	1.9	0.0	22.0	1.1

Table 4b. Stand carbon fluxes (Mg C ha^{-1}) associated with mortality.
(see Note bottom of Table 4c)

Sample Plot Number	Norway spruce	Scots pine	Birch	Aspen	Total	Average annual
2	1.3	0.0	4.4	0.0	5.7	1.1
3	18.7	0.0	3.8	0.0	22.5	4.5
4	1.0	0.0	0.3	0.0	1.3	0.3
15	0.8	0.0	0.0	0.0	0.8	0.2
16	0.0	0.0	0.0	0.0	0.0	0.0
17	2.3	0.0	6.1	0.0	8.4	1.7
44	0.0	0.0	0.0	0.0	0.0	0.0
46	1.4	0.0	6.0	0.0	7.4	1.5
49	1.4	0.0	1.2	0.0	2.6	0.4
50	13.6	0.0	2.8	0.0	16.4	3.3
53	3.5	0.0	0.0	0.0	3.5	0.7
72	4.4	0.0	0.0	0.0	4.4	0.9
73	10.3	0.0	0.0	0.0	10.3	2.1
198	18.7	0.0	0.3	0.0	19.0	1.2
11	20.5	0.0	1.3	0.0	21.8	1.4
10	70.3	0.0	2.8	0.0	73.1	4.6
8	11.6	0.0	0.7	0.0	12.3	0.8
9	12.1	0.0	0.9	0.0	13.0	0.8
100	12.4	8.3	6.8	51.1	78.6	3.9
98	13.3	0.0	0.6	0.0	13.9	0.7
91	25.4	0.0	2.7	0.0	28.1	1.4

Table 4c. Stand carbon fluxes (Mg C ha^{-1}) associated with CWD decomposition.

Sample Plot Number	Norway spruce	Scots pine	Birch	Aspen	Total	Average annual
2	0.9	0.0	0.9	0.0	1.8	0.4
3	1.6	0.0	1.0	0.0	2.6	0.5
4	0.5	0.0	0.0	0.0	0.5	0.1
15	0.1	0.0	0.5	0.0	0.5	0.1
16	0.7	0.0	1.9	2.0	4.6	0.9
17	0.9	0.0	0.7	0.1	1.7	0.4
44	2.1	0.0	0.4	1.9	4.4	0.9
46	0.1	0.0	2.1	0.0	2.2	0.4
49	0.8	0.0	0.1	0.0	0.9	0.2
50	1.8	0.0	0.8	0.0	2.6	0.5
53	1.0	0.0	1.0	0.0	2.0	0.4
72	3.3	0.0	1.4	0.0	4.6	0.9
73	0.8	0.0	0.1	0.0	0.9	0.2
198	6.4	0.0	0.5	0.0	6.9	0.4
11	5.2	0.0	0.4	0.0	5.6	0.4
10	7.3	0.0	0.7	0.0	8.0	0.5
8	2.8	0.1	0.1	0.0	3.0	0.2
9	2.4	0.0	0.1	0.0	2.5	0.2
100	3.0	2.2	0.3	7.0	12.5	0.6
98	3.4	0.0	0.1	0.0	3.5	0.2
91	9.5	0.0	0.1	0.0	9.6	0.5

Note: the data from the sample plots 2, 3, 4, 15, 16, 17, 44, 46, 49, 50, 53, 72, and 73 are for the 5-yr period;
the data from the sample plots 198, 11, 10, 8, and 9 are for 16-yr period;
the data from the sample plots 100, 98, and 91 are for the 20-yr period.

value was highest (3.4) (Table 5). The amount and the dominant kind of mortality (snags, uprootings or stem breakage) were determined by the intensity of the windthrow disturbance regime and stand dynamic phase. The annual carbon flux associated with CWD decomposition varied from 0.1 to 0.9 MgC ha^{-1} yr^{-1} (Table 4c). Tree species and site conditions affected this flux, which averaged 0.3 (0.1-0.5) MgC ha^{-1} yr^{-1} in the Norway spruce *Sphagnum-Myrtillus* and *Polytrichum-Myrtillus* stands, where spruce dominated the CWD; 0.4 (0.1-0.9) MgC ha^{-1} yr^{-1} in the Norway spruce *Vaccinium myrtillus* stands with the dominance of spruce in CWD; and 0.8 (0.6-0.9) MgC ha^{-1} yr^{-1} in the Norway spruce *Vaccinium myrtillus* stands with the dominance of deciduous species in CWD. Annual carbon accumulation (stand increment) exceeds carbon loss (CWD decomposition) by between 0.1 and 0.6 MgC ha^{-1} yr^{-1}, in most cases, without taking into account the carbon fluxes from soil to atmosphere.

Although we do not have data on the carbon flux from CWD to soil, its order of magnitude may be estimated. Experiments with pure cultures of wood decay fungi on small wood samples showed that white rot fungi decompose wood completely to CO_2 and H_2O. Brown rot fungi leave a remnant (mostly lignin) of about 30% of the original dry mass (Soloviev 1992). Our results (unpublished) showed that the average volume ratio of brown and white rots in the rotten CWD in pristine boreal Norway spruce forests is about 2:3. Let the CWD store be X MgC ha^{-1}. 40% of this CWD is assumed to be decompose by brown rot, and 60% by white rot. This means that 60% of this CWD will decompose completely. 40% corresponds to 0.4X MgC ha^{-1}. 30% of this value (0.12X MgC ha^{-1}) is not decomposed by brown rot fungi. The total Norway spruce decomposition period in the middle boreal zone is about 50 years. Consequently, about 0.12X MgC ha^{-1} during the 50-yr period will transfer to the soil. We do not know the rate of

Table 5. Carbon fluxes (Mg C $ha^{-1}yr^{-1}$) in the stands of different types.
(values are means ±SE, with sample size in parentheses)

	Forest type					
	Picea abies Sphagnum-Myrtillus and *Polytrichum-Myrtillus* forests			*Picea abies Myrtillus* forests		
Growth	1.1 ±0.20 (9)			1.2 ±0.19 (12)		
	Stand dynamic phase					
	GS increasing	GS stabilization	GS decreasing	GS increasing	GS stabilization	GS decreasing
Mortality	0.6 ±0.25 (2)	1.5 ±0.48 (5)	1.3 ±0.15 (2)	0.7 ±0.38 (4)	1.3 ±1.03 (3)	3.4 ±0.94 (5)
	Substrata characteristics					
CWD decomposition	Spruce dominated in CWD	Deciduous species dominated in CWD		Spruce dominated in CWD	Deciduous species dominated in CWD	
	0.3 ±0.05 (9)	Na*		0.4 ±0.08 (10)	0.8 ±(2)	

* Not available (no plots)

this flux. Thus, we cannot estimate the annual flux, but the order of magnitude is in the order of 10^{-2} MgC ha^{-1} yr^{-1}, since the order of magnitude for the CWD stores is about 10^{-1} MgC ha^{-1}.

DISCUSSION

Comparisons with Other Studies

The results obtained in this study may be considered a preliminary assessment of the living and dead wood carbon dynamics in some types of European pristine boreal Norway spruce forests. Previous investigations in the territory of the pristine woodland of "Vepssky Les" allowed us to choose the main types of presented stands and to attach our sample plots to the previously established ones.

Unfortunately, there is a lack of information on carbon dynamics in pristine Norway spruce forests of different age structures, dynamic phases and site conditions. Therefore, we cannot adequately evaluate our findings in comparison with the results of other authors. However, some comparisons can be made.

The CWD carbon stores (range 4.5-61.5 MgC ha^{-1}) are comparable to those estimated in other studies in Northwestern Russian boreal forests. Krankina and Harmon (1995a,b) examined CWD in southern boreal old-growth Norway spruce forests and found CWD ranges from 17 to 36 MgC ha^{-1}. Unfortunately, these authors did not provide information on the successional state of the old-growth stands they studied and they did not specify factors influencing this range. Krankina and Harmon (1995b) also reported that mortality in Northwestern Russian old-growth boreal spruce forests was 23-60 MgC ha^{-1} over a 60 year period of time. This corresponds to an average annual mortality of 0.4-1.0 MgC ha^{-1} yr^{-1}. In our study mortality rates were higher and more variable (0.0-4.6 MgC ha^{-1} yr^{-1}). These differences in mortality rates may stem from differences in stand age structure and dynamic phase of the ecosystems studied, as well as from the differences in site conditions and topographic position of the stands. The integral carbon flux from CWD to atmosphere at the ecosystem level was not estimated.

NPP and net ecosystem productivity (NEP) is calculated by measuring or modeling all carbon fluxes that characterize the carbon balance of forest ecosystems. Using such an approach, Jarvis (2000) has shown that with few exceptions, boreal forests represent carbon sinks in the range of 0.5 to 2.5 tC ha^{-1}. In our study, carbon fluxes were not directly measured. The annual carbon accumulation and loss important for long-term dynamics for living trees and CWD were estimated from the experimental mass difference. Our results show that in pristine Norway spruce forests, annual carbon accumulation (stand increment) exceeds carbon loss (CWD decomposition) by between 0.1 and 1.6 Mg C ha^{-1} in most cases. However, this estimate does not take into account the carbon fluxes from soil to atmosphere. These fluxes may affect the balance in a way that many stands of our sample plots may be considered as carbon sources. This effect is thought to be observed in forests driven by disturbances (Lugo & Brown 1986; Harmon et al. 1991).

Methodological Considerations

The great range in values for both the carbon stores and carbon dynamics in the studied stands requires an explanation. We have organized a sampling in such a way to include all stand types observed in the "Vepssky Les" woodland. But, even in one stratum (one site type, one type of age structure and one dynamic phase), the parameters of carbon cycling varied greatly. Evidently the stand dynamic phase associated with a certain type of age structure and site conditions can significantly influence the carbon dynamics of pristine boreal Norway spruce forests. The dynamic phase reflects the time since the last windthrow disturbance and the relationship between the intensity of the windthrow event, and the ecosystem response. These phases are expressed differently in stands of different successional status and site conditions. The age structure reflects successional status of the stand and some consequences of the previous disturbances. In other words, the above characteristics represent some parameters of past disturbance regimes at the stand level. Consequently, we need better dynamic stand classification that more clearly takes into account disturbance regime in terms of kind of disturbance, disturbance intensity and magnitude, time since the last disturbance, disturbance specificity (connection with site conditions, topographic position and successional status of the stand) and ecosystem legacy (survived trees, CWD), etc. (Pickett & White 1985; White & Harrod 1997). Once the proportion of the stands with different disturbance regimes in the woodland or landscape is known, we can better describe the disturbance regime and carbon dynamics at the woodland or landscape levels. Thus, more appropriate stand classification and information on the percentage of different stands in the woodland, may reduce uncertainty in carbon cycling assessment at the stand and woodland levels.

Another source of variation in the studied parameters may be the sampling procedure. Permanent sample plot records are especially valuable, since they provide data on stand increment and mortality and solve the problem of CWD dating. The transect method allows one to avoid many errors connected with the choice of sample plots, determination of their size, classification problems, and allow one to inventory not only stands but canopy gaps as well. In addition the survey method is easier and less time consuming. The combination of plot and transect sampling may give the best results for carbon dynamics studies in pristine boreal Norway spruce forests.

Estimating the carbon fluxes from mass difference may be accepted at this time and spatial scale. But, the results from sample plots should be transferred to other stands with caution. Many factors that determine the disturbance regime need to be taken into account. The greater the differences between sample plots, in regards to both time (successional) and space, the better the results. When exploring CWD decomposition one should keep in mind that this process at the stand level is influenced by climate and weather peculiarities, site conditions, stand features (species composition, basal areas, age, age structure) and CWD features (total amount, distributions among species, decay classes and position types) (Krankina et al. 1999; Næsset 1999; Tarasov 2000; Shorohova & Shorohov 2001). Taking these factors into consideration may significantly improve results.

The next step in our carbon dynamics investigations in pristine boreal Norway spruce forests will be estimation of the litter and soil carbon pools.

Potential Management Implications

Our findings are of some interest for management practices in European boreal forests. Supporting the role of forests in global carbon budget is an important goal for sustainable management. The "Vepssky Les" case study shows that stands with different disturbance regimes and, consequently, with different carbon cycling parameters are present in these natural woodlands. The current functional woodland integrity was created over a long period of time reflecting the stands' history. Practicing forestry in woodlands like the "Vepssky Les" by emulating natural disturbance regimes means that one may have to conserve the all-aged-*Sphagnum*-*Myrtillus*-Norway spruce stands, which are assumed to have huge carbon pools and be in steady state with respect to carbon. The even-aged and uneven-aged Norway spruce forests subjected to windthrows may be harvested using the single cohort and multi-cohort systems.

Boreal forest management that mimic natural disturbance regimes has been widely discussed from the aspect of biodiversity (Hansson & Larsson 1997; Angelstam 1998a; Seymour & Hunter 1999; Bengtsson et al. 2000). It was shown that "natural structures", such as large trees, deciduous trees, snags and downed wood play an important role in biodiversity. These structures change with land use history and are maximized in pristine forests (Angelstam 1998b). Management to maintain such structures in stands with different disturbance regimes within landscapes, ensures biodiversity at the stand and landscape levels. Our "reference" study provides information on CWD – one of these natural structures, and on the processes crucial for biodiversity – stand mortality and CWD decomposition in the pristine forest. These data may be applied in boreal forestry management.

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AN APPROACH TO THE ASSESSMENT OF THE CARBON BALANCE AND FOREST DYNAMICS OVER LARGE TERRITORIES AS INFLUENCED BY FOREST FIRES AND OTHER DISTURBANCES

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ABSTRACT

Forest fires (and other types of disturbances) can significantly affect carbon dynamics in and between atmospheric and vegetative systems and therefore, affect the carbon balance of boreal forest ecosystems. Carbon fixed in different stages of succession over the long-term, may compensate for carbon emissions from forest fires. An approach to estimating forest ecosystem carbon balances over large territories is proposed that takes into account biomass (organic material) dynamics during succession. For this purpose, we propose to use forest inventory data to develop our concept of modal successions. Then, vegetation plots can be distributed among modal successions and their stages to determine typical biomass dynamics at different stages of succession within a given region.

INTRODUCTION

At present, due to concerns about global warming, the world's scientific community is paying great attention to the fluxes of carbon in and between vegetation and atmosphere systems. Research activity is focused on the estimation of carbon stocks in vegetation and soil, sources of carbon emissions to the atmosphere, and carbon sinks. Only a sufficiently prolonged one-way disparity between carbon sinks and sources can influence the carbon content in atmosphere.

Hydromorphous ecosystems generally have a positive balance of carbon and act as a sink due to the preservation of part of the biomass and its accumulation as peat. In automorphous ecosystems, where accumulations of peat do not occur, the difference between the sources and sinks of carbon are approximately equal to zero.

DISTURBANCES

The parity between sources and sinks of carbon in automorphous ecosystems is destabilized by disturbance factors that cause partial or total destruction of phytocoenoses. As plants die and their biomass decays, the rate of carbon emission greatly increases and carbon sinks slow down. Conversely, during active restoration of phytocoenoses, sinks exceed sources. Such oscillations in the carbon balance occur over a short period in grassland ecosystems, and over a long period in forest ecosystems. The long-term carbon balance oscillations of individual plots overlap and as a result, long-term oscillations of the summed balance that occur over large territories, may affect the carbon content of the atmosphere. Destabilizing regimes in forest ecosystems are caused mainly by fires, insect pest attacks and logging (Table 1).

TYPICAL DYNAMICS OF CARBON BALANCE AND BIOMASS IN PYROGENEOUS SUCCESSIONS

The most destructive kind of disturbance regime is caused by fire. Usually forest fires consume only 10-20% of the vegetative biomass. Decomposition of the remnants of the biomass of dead and dying plants takes many years. New vegetation develops at the same time, so that new biomass forms and accumulates. Post-fire restoration of forests is slow, taking from 5 to 40 years. In this period carbon sinks from forest regrowth could be less than the carbon released due to decomposition of the dead plants. The complete rehabilitation of biomass is observed 50-100 years after the fire.

Fires in boreal forests induce long-term postfire (pyrogeneous) successions. Four

Table 1. Effect of disturbance regimes on destruction of biomass in ecosystems

Biomass categories in ecosystems (vegetation fuel groups)	Disturbance regimes		
	fires	logging	insect pest
1. Layer of litter (detritus), mosses and lichens on the soil surface	F_m	D_{mp}	-
2. Duff (layers F & H), peat, humus	S_p	D_{mp}	-
3. Small shrubs and herbs (green)	F_m	D_m	-
4. Snags and dead-standing trees	$F_m/S_m/D_m$	D_m	D_m
5. Understory and shrubs	F_p/D_m	D_{mp}	-
6. Needles, leaves and twigs in tree crowns	F_p/D_p	D_{mp}/T_p	D_{mp}
7. Trunks and limbs of growing (live) trees	D_p	T_m/D_{mp}	D_m

Key to modes of biomass destruction in ecosystems:

1) flaming combustion - F
2) flameless combustion (smouldering) – S
3) decay (rotting) - D
4) removal (transportation) - T
m - must be (typical)
p - possible

Biomass - organic matter contained in phytomass, detritus, litter, humus, peat.

carbon balance-related stages may be distinguished in the pyrogeneous succession cycle (Figure 1);

1. Intensive carbon emission stage (combustion period). The stock of biomass decreases rapidly and large amounts of carbon are rapidly released. This is a very short stage that lasts for minutes, hours or days.
2. Slow carbon emission stage (postfire destruction of phytocoenoses). Decay of the dead plants prevails, biomass stock continues to decrease, carbon sources are greater than sinks. This stage lasts for years but the process subsides over time.
3. Slow carbon sink stage (postfire recovery). The stock of biomass builds up due to the active growth of vegetation and accumulation of the duff. Carbon sinks prevail. This stage lasts for tens of years.
4. Carbon balance equilibrium stage (climax). The processes of formation and decay of the biomass are at equilibrium, biomass stock is stable, and the carbon balance is practically "nil". Theoretically this stage can be indefinitely long, but is shortened due to the inevitable occurrence of fire, logging, insect pest attack or other disturbances.

The loss and uptake of carbon for the period of the full cycle is generally at equilibrium. But this cycle may be interrupted by another fire in the second or third stage, and start as a new cycle.

Disturbance by fire in boreal forests is common and periodic. A forest in any region consists of areas which are at different stages of different types of successions. Consequently the areas will have differences in the sign and value for the carbon balance. In boreal forests the third stage forest phytocoenoses (the stage of slow carbon sink) prevails in terms of area, because this is the longest stage. This fact seems to be the reason for the common belief that boreal forests ecosystems are the main consumers of carbon and producers of oxygen on a global scale. In reality, mass emissions of carbon during a period of increased forest fire incidence may last a long time and not be compensated for by carbon sequestration. This may result in climate warming, more severe droughts and more forest fires (Mitchel, 1989; King & Neilson, 1992). A minor warming of 0.5 - 0.7° C in the period of 1980-1991 in the Northern hemisphere increased the number of forest fires in Russia by 60 % (Kalinkina, 1992) and the areas burned in Canada by five times (Auclair & Andrsko, 1993). In the case of a continuous build-up of forest fire occurrence, a dangerous self-escalation of the "greenhouse effect" may result.

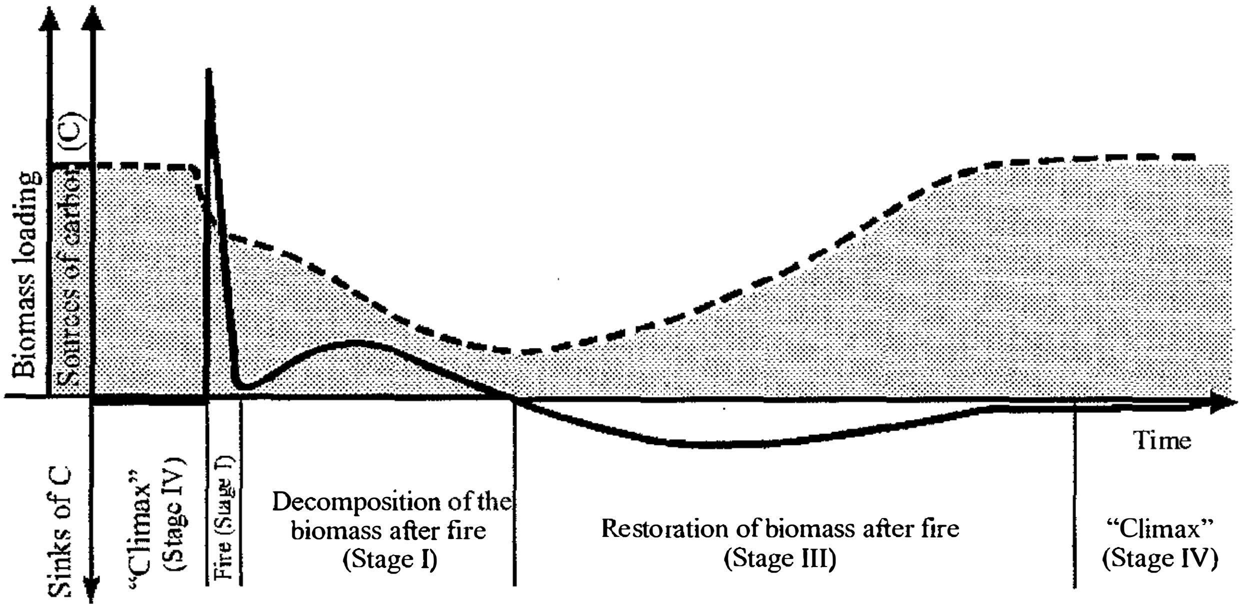


Figure 1. Effect of fires on biomass and carbon balance dynamics during the succession (theoretical diagram).

DYNAMICS OF THE CARBON BALANCE IN LARGE TERRITORIES: ASSESSMENT METHOD

The dynamics of the carbon balance in large forest territories should be investigated, and an information base should be created for forecasting the dynamics of the carbon balance under different disturbance regimes.

The usual methods of carbon budget evaluation are based either on differences in carbon stocks or on estimates of carbon fluxes. The stock-based method operates with a rather small difference of two large approximate values, and the uncertainty of the difference can be significant. For large territories, estimates of fluxes (sources and sinks) are very approximate. Therefore, over large territories, it is sometimes difficult to judge not only the magnitude of the carbon balance, but even its sign.

A different approach for examining the carbon budget, based on the estimation of biomass dynamics during succession is suggested. Postfire successions are highly variable even within one natural region. Such variations are caused by;

1. pre-fire types and ages of biocoenoses,
2. kinds of fires, their intensity and time of impact (season),
3. the role of the vegetation adjacent to a burned area as the source of seeds and,
4. the impact of weather fluctuations upon the conditions of forest regeneration in the postfire period.

However, it is difficult to describe and quantify all of the numerous succession lines and their interconnections within a region.

A landscape-ecological approach, based on bioclimatic parameters of regions (ecoregions), is most desirable and appropriate for our purposes (Nazimova, 1995, 1996; Isachenko, 1996). The advantage of this approach is that it deals with concrete climatic parameters, which are meaningful for zonal ecosystem succession.

Borders of the zonal ecosystems are defined by the landscape-climatic sectors of continentality and humidity, which present a matrix for prediction of the direction and rate of succession at sites within the zonal landscape, using information from a climate and vegetation database. Forty-four zonal classes were established in boreal Eurasia (Figure 2), which are interpreted through the BIOCLIMATIC portrait models. Using two-dimensional and three-dimensional climatic

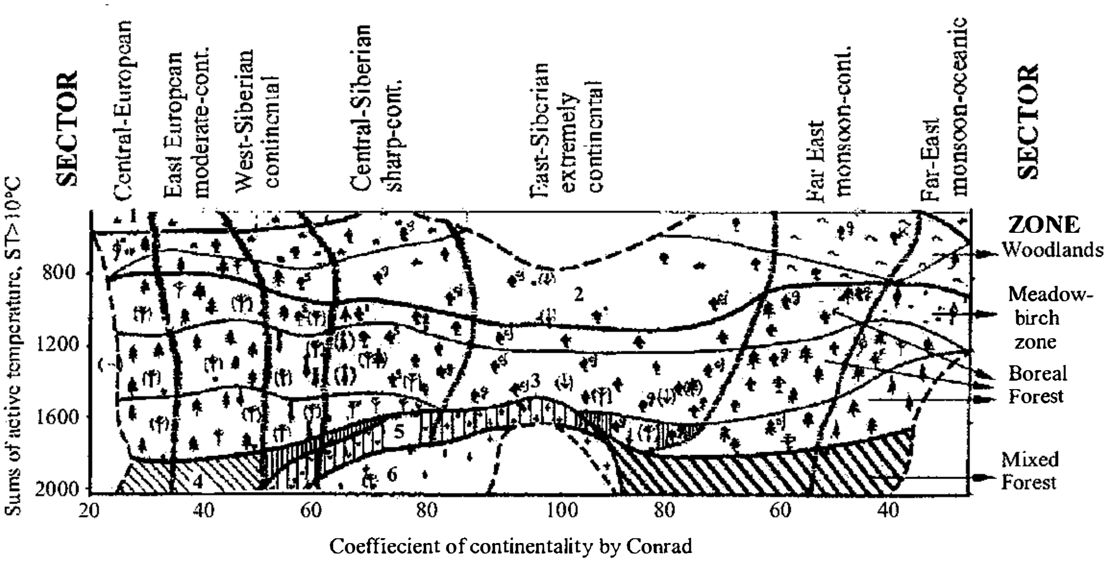


Figure 2. Landscape-climatic sectors and zones of Boreal Eurasia. 1. tundra; 2. woodlands; 3. taiga with subzones of north, middle and south taiga; 4. mixed zones; 5, forest-steppe; 6. steppe. Sectors of continentality – on the upper margin of the diagram.

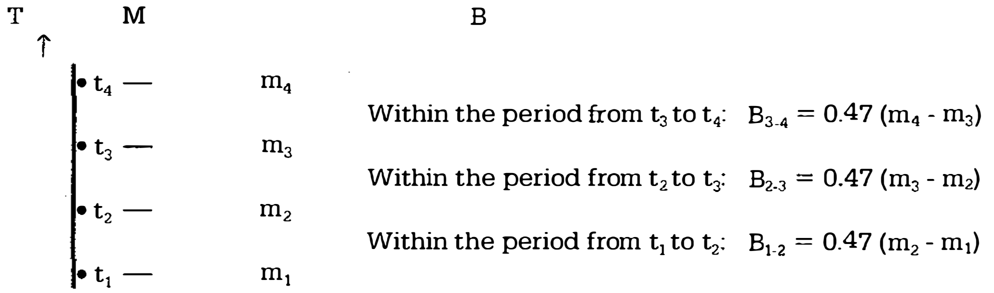
ordination, ecoregions are classified depending on the continentality, warmth supply and humidity. Meteorological data (approximately 500 stations for Siberian ecoregions and the same amount for the other ecoregions of Russia) are drawn from normal records of monthly means for temperature, precipitation, moisture deficit etc.

Bioclimatic models reveal that continentality and humidity of the zonal climate determine the forest composition at climax and quasiclimax stages of succession, while warmth supply mainly determines the level of productivity and rate of growth.

MODAL SUCCESSIONS

We suggest a modal approach, in which all vegetation plots within the region are distributed according to a small number of modal (i.e., prescribed *a priori*) successions, and the dynamics of the biomass by the phases and stages of the successions are investigated. Ideally the modal successions could be harmonized with forest inventory data. Within the modal approach, the quantitative description of biomass dynamics should be made on the basis of the major phytocoenoses' components with separation of generalized typical variants according to biomass dynamics during forest floor restoration, decay and build-up of duff, decay of dead-standing trees and snags, tree stand regeneration, and so on. The biomass dynamics of different successions could be "synthesized" in general from the typical variants of biomass dynamics on the basis of the components of biocoenoses. Methods for linear description and measurements in field investigations have been developed (Sofronov & Volokitina, 1998; Sofronov, Volokitina & Kajimoto, 1999). To reduce the amount of work we do not plan to investigate those biomass components which we assume would undergo minor changes under the influence of fire (for example, the humus of the mineral layer of soil or a water-logged layer of a peat deposit).

The typical dynamics of the carbon balance in each succession could be calculated on the basis of the dynamics of the biomass load (Figure 3).



Within the period from t_3 to t_4: $B_{3\text{-}4} = 0.47\,(m_4 - m_3)$

Within the period from t_2 to t_3: $B_{2\text{-}3} = 0.47\,(m_3 - m_2)$

Within the period from t_1 to t_2: $B_{1\text{-}2} = 0.47\,(m_2 - m_1)$

Figure 3. Dynamics of carbon balance during pyrogeneous succession (scheme of balance assessment based on biomass dynamics) T - time; M - biomass (stock); B - carbon balance.

CARBON BALANCE PREDICTION: INFORMATION BASES

Regional information bases for carbon balance prediction must contain;

1. a description of modal successions and their stages,
2. the set of attributes and characteristics (of those that are used in the forest inventory description) which could be utilized to distribute the inventory plots over modal successions and their stages, with the help of a computer and,
3. characterization of the stages of every modal succession by the sign and value of the carbon balance.

Information about vegetation plots at each stage of the regeneration process must be supplemented with information about the plots related to effects on, and response to fire, including;

1. the combustible biomass load and
2. emission of carbon, depending on burning conditions (phenological period, drought severity, wind speed);
3. a probable succession and its probable stage after a fire (in the case of partial damage to the vegetation).

The amount of burned organic matter can be evaluated in two ways;

1. evaluating the organic matter stock on fresh burns and comparing it to pre-fire stock or,
2. by fire experiments.

A safe technique for fire experiments has been developed (Sofronov, Volokitina, 1990).

Accomodations must be made for the possibility of additional carbon emissions to the atmosphere from mass forest fires due to seasonal dryness plausible under different scenarios of climate warming (consistent with scientific forecasts). The additional carbon emissions would be due to additional "above average" areas burned. It is this additional fire area that should be used in the calculation.

We are able to forecast total area of forest fire within the boundaries of pyrological districts in response to a forecasted dryness level. The calculations should make use of averaged, typical characteristics of the forest area predicted to burn. Most important of these are,

1. the amount of carbon emitted from a unit area in the course of fire and,

dynamics of carbon emissions per unit burned area through all years over the course of all stage of the postfire destruction of biogeocenoses. We should emphasise that the aforementioned parameters are strictly regional

CONCLUSION

A certain role in the dynamics of greenhouse gases in the atmosphere may be played by the carbon cycle in and between, the atmosphere and vegetative systems, more precisely — by the carbon balance. Therefore, its estimation and prediction are very important. Fire in the boreal forests usually exerts its greatest effect on the magnitude and duration of the variations of carbon balance, because rehabilitation stages that make up for fire emission of carbon, occur over the long-term. Correct estimation of the carbon balance can be achieved on the basis of biomass dynamics over succession. In order to estimate and predict the carbon balance of large territories, it is desirable to create information bases that characterize the regions' vegetation in terms of modal successions and their stages, according to the biomass dynamics as a function of the processes of combustion, decay and regeneration of biomass. Distribution of vegetation plots by modal successions and their stages may be achieved using computers and forest inventory data. The landscape-ecological approach facilitates a detailed analysis of forest succession as the basis to forecast forest dynamics.

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AFFORESTATION POTENTIAL IN CANADA: A SPATIAL ANALYSIS OF ECONOMIC LAND SUITABILITY WITH CARBON SEQUESTRATION BENEFITS

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ABSTRACT

Afforestation for carbon sequestration has been identified as a policy option in the context of meeting greenhouse gas emission reduction targets under the Kyoto Protocol. A spatial analysis of economic land suitability for afforestation is presented for several regions in Canada. The regions have been identified as having a base of cleared agricultural land potentially available for afforestation, and include southern Ontario and the Prairie regions of Alberta, Saskatchewan, and Manitoba. Agricultural land values by local area were compared with simulated net present values from afforestation taking into account plantation establishment costs, stumpage values and potential returns from carbon storage in both standing and harvested biomass. Break-even carbon prices for afforestation are mapped to the Canadian Census Consolidated Sub-division unit for four scenarios: red pine (*Pinus resinosa*) in Ontario; green ash (*Fraxinus pennsylvanica*) in the Prairies and hybrid poplar (*Populus* sp.) across both regions. Afforestation using hybrid poplar tended to result in a higher proportion of land identified as economically suitable although red pine with good site potential showed superior results in Ontario due to the benefits of long term growth and delayed emissions from forest product decay. Relatively higher rates of tree growth in Ontario compared to the Prairies was partially offset by the higher opportunity cost of agricultural land, particularly in southern and central Ontario. This information can be used to assist national afforestation policies and programs aimed at enhancing forest sinks. Additional research required to improve the spatial models is identified.

Keywords: afforestation, economic land suitability, spatial variation, carbon benefits.

INTRODUCTION

In response to concern over global warming, the Framework Convention on Climate Change (FCCC) was established at the 1992 United Nations Conference on Environment and Development (UNCED). Under the FCCC, the Kyoto Protocol was signed in 1997 which committed most industrialized countries to reduce their collective net greenhouse gas (ghg) emissions by at least 5 per cent below 1990 levels during the commitment period 2008-2012. The commitment by industrialized countries to reduce ghg emissions under the Kyoto Protocol reflects the growing international recognition of the likelihood of human interference with the global climate and the need to address potential long-term impacts.

The Kyoto Protocol focuses mainly on reducing fossil fuel emissions from energy production and industrial activities, but also recognizes sinks and sources from land use change and forestry. In an attempt to provide incentives to enhance and conserve forest sinks and reservoirs, negotiators agreed on three land use change and forestry activities to be included when accounting for emission reduction commitments - these were the carbon stock changes in 2008-2012 resulting from *reforestation*, *afforestation* and *deforestation* since 1990. Further detailed discussion of issues and options for carbon accounting for land use change and forestry activities is contained in the Special Report on Land Use, Land Use Change and Forestry, prepared by the Intergovernmental Panel on Climate Change (Watson *et al*., 2000).

This paper focuses on one particular forestry activity – that being *afforestation* for sequestering carbon from the atmosphere[1]. Many countries, including Canada, are investigating the potential for afforestation as part of a suite of possible policy options for achieving ghg emission reductions.

Previous afforestation studies in Canada have identified a potentially large base of marginal agricultural or other degraded land that may be suitable for tree planting for carbon sequestration and related purposes (e.g. provision of wildlife habitat and recreation opportunities). Nagle (1990), for example, identified around 7 million hectares (Mha) of marginal agricultural land considered suitable for tree planting, comprising mostly pastures, water margins and other areas difficult for tillage or with low fertility. Similar studies identify almost 11 Mha of marginal agricultural land available for afforestation although a realistic national planting program over the next fifteen years is crudely estimated at no more than 843,000 hectares (National Climate Change Process Forest Sector Table, 1999). Their estimation takes account of the historical tree planting of trees by landowners and other ad hoc factors such as landowner attitudes and alternative uses on higher fertility land. In general, these studies have tended to focus on the biophysical capability of marginal agricultural land to support tree growth with limited consideration of socio-economic factors affecting the overall availability of the land for afforestation. A recent economic evaluation has identified between 2.3 to 5.8 Mha of land in western Canada that may be suitable for afforestation assuming carbon sequestration costs of $20 and $50 per m^3 are tolerated respectively, from a biologically capable land base of 7 Mha (van Kooten *et al*., 1999). However, most assessments have been undertaken at a national or regional (e.g. provincial) level with limited spatial analysis at a finer scale. There is clearly a need for better information on the amount of land potentially available for afforestation taking into consideration the costs and benefits of alternative lad uses; and the spatial dimensions of land availability at a sub-provincial level.

This paper addresses these issues by providing a spatial analysis inclusive of an assessment of costs and benefits of plantation forestry for wood production and carbon sequestration in comparison to agricultural production values. The specific objectives were:

1. to demonstrate, through a simple model, the integrated use of spatial data on economic and capability factors in assessing land suitability for afforestation;
2. to explicitly consider the opportunity costs of land for agriculture as well as potential benefits from carbon sequestration and forest products; and
3. to present the results spatially as a means of identifying prospective areas for further investigation.

This type of information is of interest to policy makers, land use planners and landowners interested in afforestation initiatives and forestry carbon sequestration potential.

METHODS AND DATA

Given the potential importance of growing trees to sequester carbon as a low cost option in addressing ghg emission reductions, numerous approaches have emerged for assessing land use trade-offs between agriculture and forestry uses. Recent empirical studies on land availability and costs of carbon sequestration from afforestation include the use of land rental rates or forgone returns from agriculture for representative land types (e.g. Moulton and Richards, 1990; Parks and Hardie, 1995; van Kooten *et al*., 1999) and econometrically estimable models of landowner behaviour in private land markets (Newell and Stavins, 1999).

Advances in mapping technologies and the integration of bio-physical and economic information within geographic information systems has also enabled more detailed analysis of the spatial dimensions of economic

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[1] For the purposes of the study, afforestation was taken to mean the establishment of trees on previously non-forested land such as cleared agricultural land.

issues such as efficiency of land use. Relevant applications include an assessment of land availability for plantations across Australia using spatially referenced data on agricultural land values with rainfall and representative tree growth data (Burns *et al.*, 1999); and the spatial integration of tree growth and soil type distributions with relevant carbon flux models for afforestation across Wales (Bateman and Lovett, 2000).

This study represents a contribution to more detailed spatial analysis of afforestation potential in Canada by taking into account local variation in the opportunity costs of land for agriculture, as well as potential returns from carbon sequestration and sale of forest products. The sub-provincial areas considered suitable for afforestation were identified by comparing the net present value of afforestation benefits (NPVAB) with the net present value of agricultural land uses (NPVAL). The cost-benefit framework for identifying economically suitable land can be represented simply as:

$$S = \sum_{j} A_j \ \{j : \text{NPVAB}_j > \text{NPVAL}_j\} \qquad [1]$$

where S is the total amount (hectares) of the suitable land area (A) of sub-provincial land units j ($j = 1, 2, \ldots J$). The analysis proceeds in a number of linked steps and represents a simulation approach, rather than optimization modeling, to identifying economically suitable land. The analysis is intended to provide only a strategic overview of suitable land due to uncertainty in the underlying data and coarse scale for some variables, such as the growth curves for afforestation species. The steps underlying the cost-benefit framework and output maps on land suitability are outlined in the following sections.

Identification of Agricultural Land Values

Agricultural land values were identified by local area for each of the case study regions. The 1996 Census of Agriculture provides detailed farm production and socio-economic information by Census Consolidated Subdivision (CCS) for each of Canada's provinces and territories (Statistics Canada, 1999). The CCS units generally correspond with rural county or local municipality boundaries within each province. For the Prairie provinces of Alberta, Saskatchewan and Manitoba, there were a total of 66, 297 and 117 CCS units respectively with an average size of 214,000 hectares. In Ontario, there were 439 CCS units with an average size of 170,000 hectares. Estimation of the value or opportunity cost of agriculture is difficult given the dynamic nature of commodity markets and technological inputs to production. To account for uncertainty in the opportunity cost of agriculture, three reported measures from the 1996 Census were used to provide estimates of the range of likely agricultural land values for each CCS. The three measures used were:

1. land capital values (estimated by landowner);
2. net annual cash receipts (total business receipts less expenses); and
3. annual land leasing expenses.

In the case of net cash receipts and land leasing expenses, these values were converted to a net present value using the standard discount factor for an infinite series of annual payments. That is:

$$\text{NPVAL} = a / i \qquad [2]$$

where,

NPVAL = the net present value of agricultural land,

a = net annual receipts or annual lease expenses and

i = the discount rate (rates of 4% and 6% were used in the analysis).

Using the three estimates of agricultural land value for each CCS, a triangular probability distribution was assumed to generate a likely mean expected land value (this type of Monte-Carlo analysis is discussed later). Expected agricultural land values by CCS varied from less than $10 per hectare to over $20,000 per hectare in those areas close to metropolitan markets or predominantly semi-urban landscapes. The high land capital values of CCS boundaries close to metropolitan centres reflects a combination of agricultural and other potential income sources and market benefits, such as opportunities for residential and other urban development.

Estimation of Carbon Sequestration from Afforestation Scenarios

The carbon sequestration potential was estimated for each of the afforestation scenarios. The afforestation scenarios included red pine (*Pinus resinosa*) in Ontario; green ash (*Fraxinus pennsylvanica*) in the Prairies and hybrid poplar (*Populus* sp.) in both regions. Red pine has considerable potential as a plantation species and a commercial lumber resource in selected areas of Ontario as a result of previous planting programs. Green ash is a popular shelter belt and farm tree species in the Prairies and represented over 30 per cent of seedlings distributed by the Prairies Farm Rehabilitation Administration in 1997 (Peterson *et al.*, 1999). Hybrid poplar has been identified as a fast growing species with considerable potential for carbon sequestration and other uses such as a source of industrial fibre and bio-energy (Resource Efficient Agricultural Production, 1999).

To investigate the sequestration of atmospheric carbon (C) in afforestation ecosystems and forest product pools, a number of assumptions were made to ensure tractability of the analysis and account for scientific uncertainty in transfers (or fluxes) between C pools. The afforestation of previously cleared land initially results in atmospheric carbon being stored in various ecosystem pools, including above ground biomass (i.e. tree stems, branches and leaves), below ground biomass (i.e. roots) and forest litter and soils. Over the first and subsequent rotations, some of the initial C uptake will be transferred from one pool to another, while other C pools will release some C back into the atmosphere. Overall, the analysis tracks the carbon uptake and release in five separate pools and adopts a similar model to that used by van Kooten *et al.* (1998) for western Canada. In addition to providing a consistent basis for comparison, there is no a *priori* basis for assuming otherwise for central-eastern Canada. The five pools included stem wood; above ground biomass (not including the stem wood); below ground biomass; litter and soils.

Starting with the establishment of trees on cleared land, representative growth and yield curves reported in Berry (1984) and Freedman and Keith (1995) for red pine and Peterson *et al.* (1999) for green ash were used to estimate stem wood volume for chosen site classes. Provincial wide growth data for a good quality site was used for both red pine and green ash to reflect an upper bound on afforestation potential.

In the case of hybrid poplar, a generalized growth function developed by Guy and Benowicz (1998) was used to estimate merchantable stem volume. They employed the Chapman-Richards function:

$$V(t) = A\,(1 - \exp^{-kt})^{m} \qquad [3]$$

where,

t = time (in years),
$V(t)$ = volume at time t (m^3/ha),
A = maximum stem wood volume (m^3/ha) and,
k and m are model parameters.

Parameter values used in this study were A=329 and k=0.156 for Ontario; A=270 and k=0.143 for the Prairies; and m=3 for both regions (see Guy and Benowicz, 1998). Carbon builds up in the stem wood until rotation age when harvesting results in a transfer of C from merchantable biomass to forest products, which subsequently release a proportion of C back into the atmosphere as products decay over time. This process of tree growth and periodic harvest is assumed to continue indefinitely.

Conversion factors drawn from the available literature were used for converting stem wood volumes to total above and below ground biomass (Table 1). For all scenarios, the treatment of C allocated to below ground tree biomass (roots) was to include this growth for the first rotation only, reflecting an assumed balance between the rate of decay of roots from harvested trees and new growth of roots from second and subsequent rotations. The carbon allocated to root biomass was calculated as a proportion of annual stem wood volume equivalent to a ratio of 0.396 for green ash; 0.35 for red pine and 0.43 for hybrid poplar (see Peterson *et al.*, 1999; Freedman and Keith, 1995; and Guy and Benowicz, 1998).

When trees are harvested, the non-merchantable biomass is left on the site as slash and enters the litter account. The litter account consists of dead or dying biomass on the forest floor that releases some C to the atmosphere as well as to the soil pool. It was assumed the litter account would increase at a constant annual rate for the first 50 years after which time it would be in equilibrium. At that point it was assumed the litter pool was equivalent to one half the non-merchantable biomass at harvest age, or age 50 in the case for red pine. The annual rate of accumulation in litter for the first 50 years was 0.20 tonnes/ha for hybrid poplar and green ash in the Prairies and 0.26 t/ha for hybrid poplar and 0.44 t/ha for red pine in Ontario. In addition to this initial build up in the litter pool, the dynamics of periodic additions to the litter pool from post harvest slash and subsequent decay needs to be taken into account (van Kooten et al., 1998).

A spike in the litter pool from post-harvest slash (i.e. the non-merchantable component of harvested biomass) after each rotation was assumed to decay at a constant annual rate so that it was depleted by the time of the next harvest (refer Table 1).

A further carbon pool considered was the amount of soil C on afforested land. Small decreases in soil C may be observed for a few years following planting and initial site disturbance (Hansen, 1993), although over the longer term a build up in soil C may be expected. Agricultural soils are usually depleted in C content compared to forested landscapes as a result of previous land use history. A gradual accumulation in soil C was assumed for the first 50 years for each afforestation scenario. A linear rate of accumulation was assumed due to the difficulty in determining soil C associated with various agricultural and forestry land uses (see van Kooten *et al.*, 1999). The annual rate of accumulation in soil C was 0.96 tonnes/ha for the Prairies region and 1.36 t/ha for Ontario. These rates reflect estimated differences in soil C (t/ha) on marginal agricultural land and second growth forest of 60 and 108 respectively for the Prairies; and 66 and 134 respectively for Ontario (Guy and Benowicz, 1999). After the first 50 years the net carbon flux from the soil pool was assumed to reach a long-term balance between annual C sequestration and emissions. A summary of stem wood volumes and ecosystem C pools at rotation age for each of the afforestation scenarios is provided in Table 1.

The planning horizon and afforestation land use (e.g. biodiversity, land rehabilitation, recreation, harvesting activities and so on) also needs to be considered. In this case, it was assumed plantings would be systematically harvested for forest products on a rotation of 60

Table 1. Carbon fluxes in afforestation ecosystems.

Scenario	Red pine (good site)	Green ash (good site)	Hybrid poplar (medium site)
Forest ecosystems			
Merchantable volume at rotation age (m^3/ha)	(60 years)	(50 years)	(15 years)
- Ontario	739	na	243
- Prairies	na	213	185
Above and below ground biomass : first rotation (t C/ha)			
- Ontario	296	na	95
- Prairies	na	85	72
Above ground biomass : second and subsequent rotations (t C/ha)			
- Ontario	230	na	66
- Prairies	na	66	50
Slash at end of each rotation (t C/ha)			
- Ontario	53	na	26
- Prairies	na	21	20
Soils: accumulation over 50 years (t C/ha)			
- Ontario	48	48	48
- Prairies	68	68	68
Litter: accumulation over 50 years (t C/ha)			
- Ontario	22	na	13
- Prairies	na	10	10

Notes: In summary, a conversion factor of 0.39-0.40 was used to convert merchantable stem volume to carbon, taking into account expansion factors for above and below ground carbon pools and conversion of dry matter biomass to carbon. For above ground carbon only, conversion factors of 0.31 (red pine, green ash) and 0.27 (hybrid poplar) were used. More detailed data from which these factors were derived is contained in Freedman and Keith (1995); Peterson *et al.* (1999) and Guy and Benowicz (1999). The cost of emissions from wood burning were offset by the benefits of fossil fuel substitution and bioenergy returns. na - not assessed.

years for red pine; 50 years for green ash and 15 years for hybrid poplar[2]. Red pine and green ash would be expected to provide a combination of lumber and pulp products on good site classes. Hybrid poplar is a short rotation species mostly used in North America as a source of fibre for pulp and wood panel products (e.g. oriented strand board). Table 2 summarizes the assumed proportion of harvested forest products for each of the afforestation scenarios. For conventional species, total merchantable harvest (m^3/ha) consisted of 50 per cent lumber and wood products; 30 per cent pulp and paper products and 20 per cent as bioenergy. For hybrid poplar, total merchantable harvest comprised 20 per cent wood panel products; 60 per cent pulp and paper and 20 per cent as bioenergy.

Finally, the fate of the merchantable biomass converted to forest products needs to be considered. The inclusion of forest products requires data on the types of product mixes as well as C decay cycles for each product category. Skog and Nicholson (1998) identify an average half-life of one to six years for paper products and 80 to 100 years for sawnwood used in housing construction. The half-life is the time after which half the carbon stored in the product is no longer in use and has been released to the atmosphere. A similar survey of forest product uses in the boreal zone has identified annual oxidation rates of 0.005 for paper products that end up in landfills, 0.005 for sawnwood and 0.01 for wood based panels (Winjum *et al*., 1998). These latter rates were adopted in the study for solid wood products. Paper products were further partitioned into two categories: fast and slow decay pools. The fast decay pool assumed an annual oxidation rate of 0.50 for paper products in short term use consistent with similar studies for western Canada (Kooten *et al*., 1999). The slow decay pool assumed a large proportion (75 per cent) of total paper products would be deposited in landfills within one year releasing carbon at a very slow rate. These rates are outlined in Table 2 and reflect short and long term C decay cycles for typical product uses in North America. Carbon emissions from the decay of forest products were treated as a carbon cost or tax to the landowner in the same way as C sequestration from tree growth was treated as a carbon benefit or subsidy using assumed carbon prices (see, for example, van Kooten *et al*., 1995).

In addition to lumber and pulp related products, a proportion of log residues was assumed to be used as a source of bioenergy, thereby replacing an equivalent amount of fossil fuels in energy generation. It was assumed 3.78 m^3 of wood was needed to replace the energy equivalent of 1 t of coal, thereby offsetting the release of 0.707 t of carbon to the atmosphere (van Kooten *et al*., 1999). The substitution of fossil fuels with renewable log residues is an additional carbon benefit of afforestation, together with the sale of bioenergy at an average price of $7.50 per m^3 of wood used in energy production.

There is considerable uncertainty as to how or if forest products might be included in a carbon accounting framework under the Kyoto Protocol. This analysis assumes C storage in forest products is recognized in any future potential carbon trading system. Commercial forest harvesting and multiple rotations were included in the scenarios to reflect a structure of financial incentives to landowners and promote a long-term commitment to afforestation land use. It is recognized there are a range of equally significant non-wood and other environmental benefits from afforestation in addition to forest products (see, for example, National Climate Change Process Forest Sector Table, 1999), although these types of afforestation benefits were not addressed here.

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[2] At a stand level, van Kooten, Binkley and Delcourt (1995) have contributed to a better understanding of the economics of carbon sequestration and optimal rotation decisions. Using generalized growth functions for intensively managed stands of (a) coastal forests in British Columbia; and (b) black spruce forests in the northern boreal region of Alberta; they found that the optimal rotation age increased the higher the value of stored carbon to a point where it may be socially desirable never to harvest the trees for scenarios with low timber prices and high carbon values. Furthermore, the time between harvests was reduced the higher the assumed amount of carbon stored in forest products, mainly because the tax penalty (or cost) of releasing carbon at time of harvest was lower.

Table 2. Proportion of harvested forest products and carbon decay rates (%).

Harvested products (p) at rotation age	Red pine, green ash (%)	Hybrid poplar (%)	Annual decay rate (β)
Sawnwood	50	-	0.005
Wood panel products	-	20	0.01
Paper products	30	60	
- fast decay pools	7	15	0.50
- slow decay pools (e.g. landfills)	23	45	0.005

Simulated Net Present Values from Afforestation with Carbon Benefits

Simulated net present values from afforestation required estimation of two categories of benefits:

1. benefits from the sale of standing timber; and
2. carbon sequestration benefits in forest ecosystem and forest product pools.

The benefits from forest products (NPVFP) were estimated using the formulae:

$$NPVFP = \sum_{t=0}^{r} \{(1+i)^{r-t} * (R_t - C_t)\} / [\,(1+i)^r - 1] \qquad [4]$$

where,

i = discount rate;
r = rotation age;
R_t = stumpage revenues in year t (\$/ha) and,
C_t = establishment costs in year t (\$/ha).

Forest product revenues were given by:

$$R_t = P^S_t * V_t \qquad [5]$$

where,

P^S_t = stumpage price (\$/m^3) and
V_t = merchantable volume (m^3/ha).

The NPVFP provides a useful measure of net returns from a perpetual series of rotations and is equivalent to the soil expectation value of forest land (Samuelson, 1976).

The net present value of the flow of carbon benefits (NPVCB) was calculated as follows:

$$NPVCB = \qquad [6]$$

$$\sum_{t=1}^{r} \{(1+i)^{r-t} * ((C^A + C^F) * P^C)\} / [(1+i)^r - 1] + \sum_{t=1}^{r} \{(C^B * P^C) / (1+i)^t\} + \sum_{t=1}^{50} \{((C^S + C^L) * P^C) / (1+i)^t\} - \ldots\ldots$$

$$- \sum_{p} \{(C^P p * P^C * \beta p) / (\beta p + i)\} * (1/((1+i)^r - 1))\} - ((C^N / \partial) * P^C) * [((1+i)^{\partial} - 1) / (1+i)^{\partial} * (1/i) * (1 / ((1+i)^r - 1))]$$

where,

P^C = carbon price (\$/t);
C^A = total above ground carbon (t/ha);
C^B = below ground carbon (t/ha);
C^S = soil carbon (t/ha);
C^L = litter carbon (t/ha);
C^F = fossil fuel substitution from bioenergy (t/ha);
C^P = carbon in forest product p at harvest (t/ha);
β = decay rate for forest product p (t/year);
C^N = non-merchantable above ground carbon at harvest (slash) (t/ha); suppressing t for convenience.

The last term in equation (6) captures the present value of the assumed period ∂ (years) emissions occur from the slash component of litter following each *r* over a perpetual series of rotations. In this case, the assumed period was equivalent to the rotation age implying that all slash is emitted by the time of the next harvest.

The net present value of afforestation benefits (NPVAB) was calculated by summing NPVFP and NPVCB. The NPVAB was then compared with agricultural land values in (1) to determine economic land suitability. The estimation of economically suitable land took account of the amount of existing forest and broadly non-suitable land within each CCS unit. Using the digital land cover map of Canada produced from the Advanced Very High Resolution Radiometer (AVHRR) (Cihlar *et al.*, 1999), areas of non-suitable land classes were excluded. Land considered broadly available for afforestation included AVHRR land cover classes 23-28 (described as Developed Land), comprising mostly cropland and mosaics of cropland-woodland. Remaining AVHRR classes, such as forest, urban and built-up areas, wetlands and barren land, were excluded from the area estimates.

Accounting for Risk and Uncertainty

Due to uncertainty surrounding afforestation costs and returns in Canada, a stochastic approach was used to account for the likely variation in uncertain variables as a whole. These variables included establishment costs, stumpage values and future potential carbon prices. A triangular probability distribution was assigned to each of these variables defined by low, medium and high values (see Table 3). These ranges reflect reported estimates from the available literature (see, for example, National Climate Change Process Forest Sector Table 1999; McKenney *et al.*, 1997). Using the software package @RISK (Palisade Corporation, New York), a Monte-Carlo analysis was used to identify likely outcomes and the expected mean NPV of afforestation. For each scenario, 500 simulations were undertaken using randomly selected values from the assigned distributions. The expected mean NPV for each afforestation scenario was used in the base case analysis.

The minimum, maximum and expected mean NPVs for afforestation for each scenario are presented in Table 4. Red pine on good sites was shown to generate the highest mean NPV with similar results for hybrid poplar on medium quality sites, particularly in Ontario. Green ash on good sites in the Prairies generated the lowest range of NPVs reflecting lower overall growth and site biomass carbon.

In addition, a sensitivity analysis on carbon prices was included. Future potential carbon prices are highly uncertain with estimates ranging from $20 to over $100 per t based on a survey of economy wide trading models and climate change policy scenarios (Climate Change Economic Analysis Forum, 1999). Predicted mean carbon prices of around $30/t are reflected in the findings of bioeconomic modelling studies such as Fankhauser (1994, 1995) who has yielded carbon sequestration values for a variety of scenarios over the next few decades. Fankhauer's best estimate of these values rises from around $20/t in 1991 to $27.80/t in 2021, so reflecting the predicted increase in climate change damages during that period. Consequently, average carbon prices of around $30 have been adopted in recent afforestation analyses (Bateman and Lovett, 2000).

Suitable land areas identified from the analysis are therefore presented for low ($20/t), medium ($30/t) and high ($100/t) carbon prices, as well as the expected ($50/t) carbon price from the assigned probability distribution. A break-even analysis was also conducted for each scenario and mapped by CCS unit (maps 1-4). The break-even analysis identifies the carbon price needed for the net present value of afforestation to be equivalent to the opportunity cost of agricultural land use (see maps 1-4). This type of sensitivity analysis allows the user to assess the economic suitability of land based on his or her expectations of future carbon prices. Break-even prices are presented assuming a 6% discount rate while results for carbon price scenarios are presented for discount rates of 4% and 6% (see Table 5). All costs and prices are reported in constant 1998 dollars.

The issue of discounting future carbon values has received considerable attention in the literature, focusing on the relationship between marginal damage functions and emission rates over time. In particular, most bottom up studies of afforestation strategies have measured the cost-effectiveness of

Table 3. Range of cost and price assumptions for afforestation land uses.

Variable	Low	Medium	High
Plantation establishment costs ($/ha)			
- conventional species (red pine, green ash)	900	1500	2100
- fast growing species (hybrid poplar)	1200	2000	2800
Stumpage values ($/m^3)			
- red pine	15	20	25
- green ash, hybrid poplar	10	15	20
Carbon prices ($/t C)	20	30	100

Table 4. Variation in afforestation NPVs from stochastic (Monte-Carlo) simulations[1].

Species and site quality	Red pine (good site)	Hybrid poplar (medium site)	Green ash (good site)	Hybrid poplar (medium site)
NPV of afforestation ($/ha)	*(Ontario)*		*(Prairies)*	
Expected mean	4608	3542	689	2054
Minimum	1097	(215)	(695)	(1241)
Maximum	10239	8238	3001	6004
Standard deviation	2067	1806	807	1411
10% percentile	2148	1391	(232)	340
90% percentile	7583	6269	1890	4144

[1]The afforestation NPVs presented here exclude the opportunity cost of land for agriculture. Agricultural opportunity costs by sub-provincial area are included in the subsequent assessment of break-even prices (maps 1-4) and suitable land areas for afforestation (tables 5-6).

carbon sequestration using three main methods: the flow summation method, the average storage method and the levelization/discounting method (Richards, 1997). The flow summation method treats all carbon capture as equivalent, regardless of when it occurs. The average storage method divides the present value of implementation costs by the mean carbon storage over the period. The levelization/discounting method differentiates when carbon is stored and released and gives more weight to carbon captured earlier than carbon captured later in the period. The latter approach is used in this study and included within a stochastic framework to account for variation in the range of uncertain parameters.

RESULTS

The results of the analysis are intended to provide an overview of economic land suitability in the case study regions and are summarized in Tables 5-6 and the following figures of break-even carbon prices for afforestation scenarios:

Figure 1: red pine (good site) in Ontario;
Figure 2: hybrid poplar (medium site) in Ontario;
Figure 3: hybrid poplar (medium site) in the Prairies; and
Figure 4: green ash (good site) in the Prairies.

In Ontario, there was little difference in economic land suitability patterns assuming either red pine on good sites or hybrid poplar with medium site quality. The benefits of long term growth from plantations of red pine and delayed emissions from forest product decay tended to offset the benefits of rapid early growth from hybrid poplar. From a roughly 6.2 Mha land base, between 1.8 to 2.4 Mha of agricultural land may be economically suitable for afforestation assuming a carbon price of $30/t and 4% discount rate, although these areas drop by almost half using a 6% discount rate (Table 5). At a sub-provincial level, the opportunity costs of agricultural land use generally restricted afforestation opportunities in southern and central Ontario where break-even prices were in the order of $45 to over $100/tonne (Figures 1-2). The relatively high

Table 5. Suitable areas for afforestation under assumed carbon price and site class scenarios (Mha).

Carbon price	Low ($20/t)	Medium ($30/t)	High ($100/t)	Expected value ($50/t)[1]
Suitable land areas		6% discount rate		
Ontario				
Red pine (good site)	0.41	1.32	6.09	3.68
Hybrid poplar (medium site)	0.12	0.79	5.86	2.40
Prairies				
Green ash (good site)	0.00	0.00	53.29	18.46
Hybrid poplar (medium site)	0.00	13.20	53.63	52.74
		4% discount rate		
Ontario				
Red pine (good site)	1.38	2.46	6.12	4.85
Hybrid poplar (medium site)	0.84	1.81	6.00	3.49
Prairies				
Green ash (good site)	0.00	8.71	53.63	47.72
Hybrid poplar (medium site)	7.25	46.49	53.63	53.32
Study area land base	Total land area[2] (Mha)	Forested area (Mha)	Developed land area (Mha)	
Ontario	89.1	58.0	6.21	
Prairies	176.3	93.3	53.63	

[1] This carbon price represents the expected value of the assigned probability distribution of carbon prices.

[2] Total land area does not equal the sum of forest and developed land area due to exclusion of urban and built up areas; water bodies; grassland and barren land from the analysis. The developed land area (AVHRR classes 23-28) was used to identify land broadly available for afforestation and approximates the assumed site classes for afforestation growth. Further research is needed on spatial estimates of forest site productivity.

Table 6. Total suitable land areas for break-even carbon price classes (Mha)[1].

Break-even price class ($/t C)	Red pine (Figure 1)	Hybrid poplar (Figure 2)	Hybrid poplar (Figure 3)	Green ash (Figure 4)
	(Ontario)		*(Prairies)*	
0-15	0.06	0.02	0.00	0.00
15-25	0.80	0.39	0.28	0.00
25-35	1.10	0.82	36.68	0.00
35-45	1.08	0.87	14.19	6.45
45-70	2.23	2.38	2.46	42.23
70-100	0.82	1.38	0.01	4.61
100+	0.12	0.35	0.00	0.34

[1] Based on a 6 per cent discount rate.

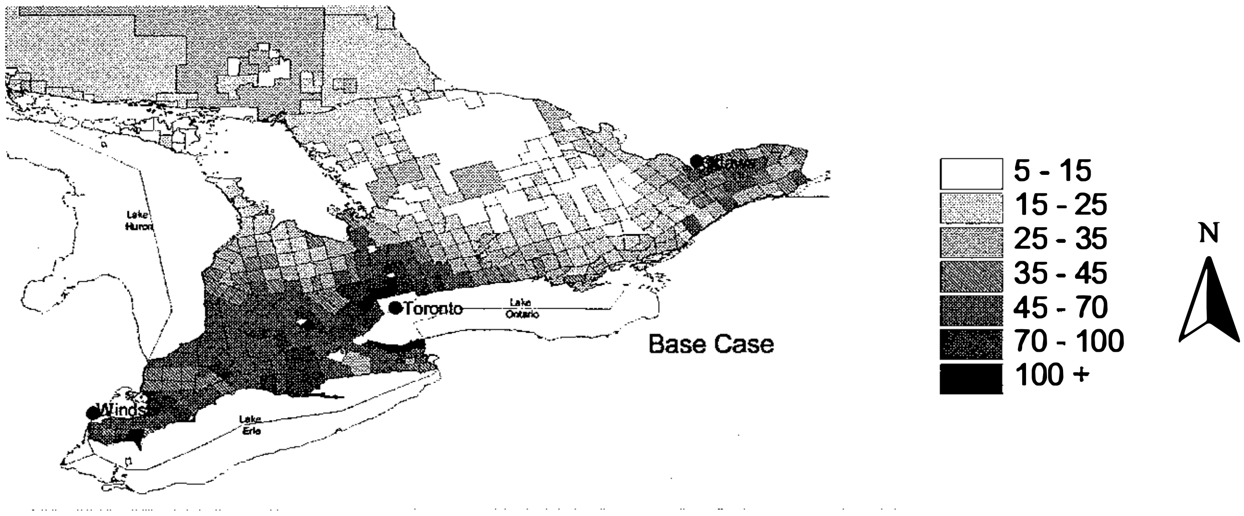

Figure 1. Break even values by Census Consolidated Subdivision (CCS) for Red Pine (good site), Southern Ontario, ($/tonne C).

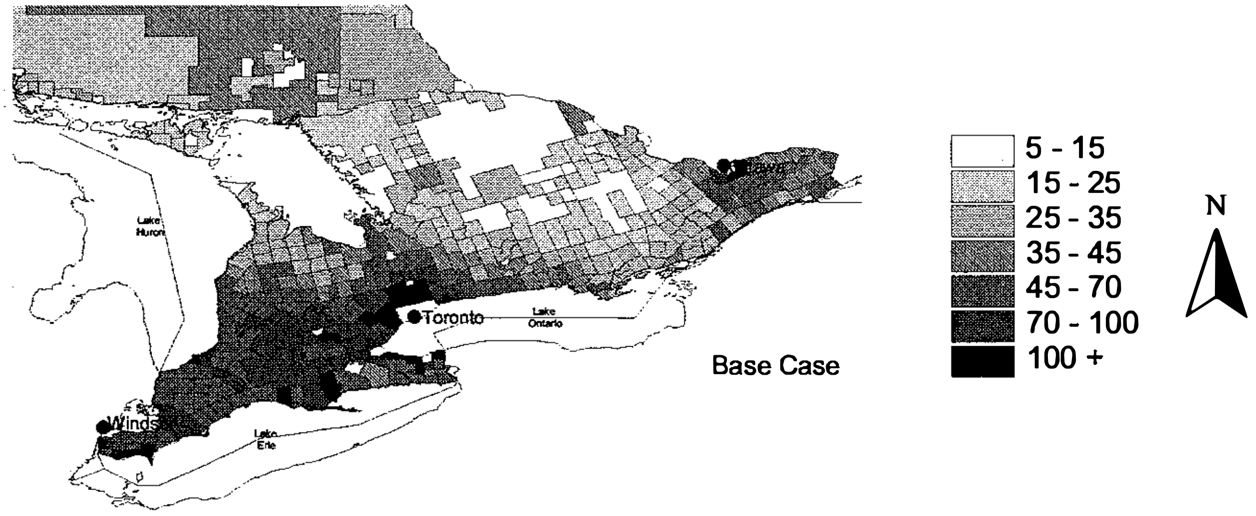

Figure 2. Break even values by Census Consolidated Subdivision (CCS) for Hybrid Poplar (medium site), Southern Ontario, ($/tonne C).

land values in these sub-regions partly reflect the proximity of agricultural land uses to metropolitan markets and a diversified rural landscape, including crops, horticulture and viticulture industries.

By comparison, afforestation may be economically suitable across many areas of north eastern Ontario at carbon prices between $15 to $35/tonne. Agricultural land values in these sub-regions reflect a predominance of commodity based industries (e.g. dairy, forage crops etc) with relatively lower land opportunity costs compared to southern Ontario.

The break-even analyses also indicate that at very high carbon prices (e.g. >$100), over 90 per cent of broadly available land in Ontario may be suitable for afforestation. However, it is important to note that the analysis presented here does not include other carbon sequestration options on available lands, such as enhanced soil conservation and livestock management. It should be possible to independently compare the carbon sequestration benefits from afforestation with other possible energy use and land management strategies.

For the Prairies region, afforestation using hybrid poplar was shown to be economically viable for carbon sequestration in many parts of the region at prices above $25/tonne assuming land with medium site quality and the availability of local markets for short rotation fibre products (Figure 3). From an available land base of 53 Mha, for example, 13 Mha of agricultural land was found to be economically suitable assuming a carbon price of $30/t and a 6% discount rate; and up to 46 Mha assuming a 4% discount rate (Table 5). However, these estimates are likely to be an over estimate of suitable land due to variation in land capability and growth within each CCS unit. Local level variation in forest site classes was not captured in the analysis due to the limited availability of information on spatial estimates of site productivity. Consequently, the results are intended to provide a broad overview of potential areas with good prospects for further investigation.

Relatively lower agricultural land values in the Prairies compared to Ontario were also partially offset by lower overall growth and site potential for afforestation species, particularly for conventional species such as green ash. Afforestation opportunities using green ash were shown to be unsuitable as an economic land use at carbon prices below $35/tonne assuming a 6% discount rate, although 8 Mha was found to be suitable at a 4% discount rate. Spatial patterns of economic land suitability for afforestation in the Prairies also reflected the influence of higher land values close to metropolitan centres and markets, such as around Winnipeg and Edmonton (Figure 4). Afforestation land uses in these sub-regions were constrained by the opportunity costs of agriculture to carbon price scenarios above $70/tonne. A summary of suitable land areas for each of the break-even carbon price classes contained in Figures 1-4 is presented in Table 6.

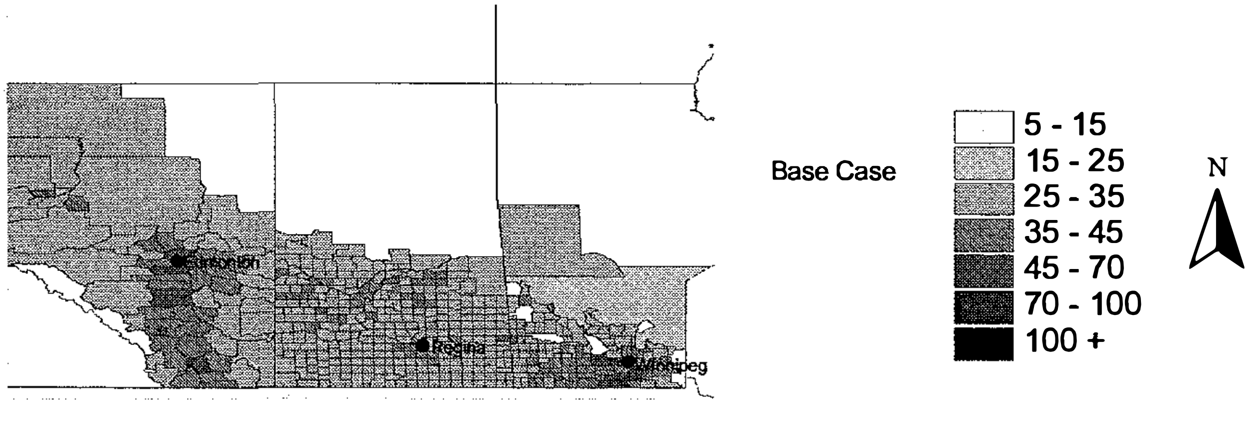

Figure 3. Break even values by Census Consolidated Subdivision (CCS) for Hybrid Poplar (medium site), Prairie provinces, ($/tonne C).

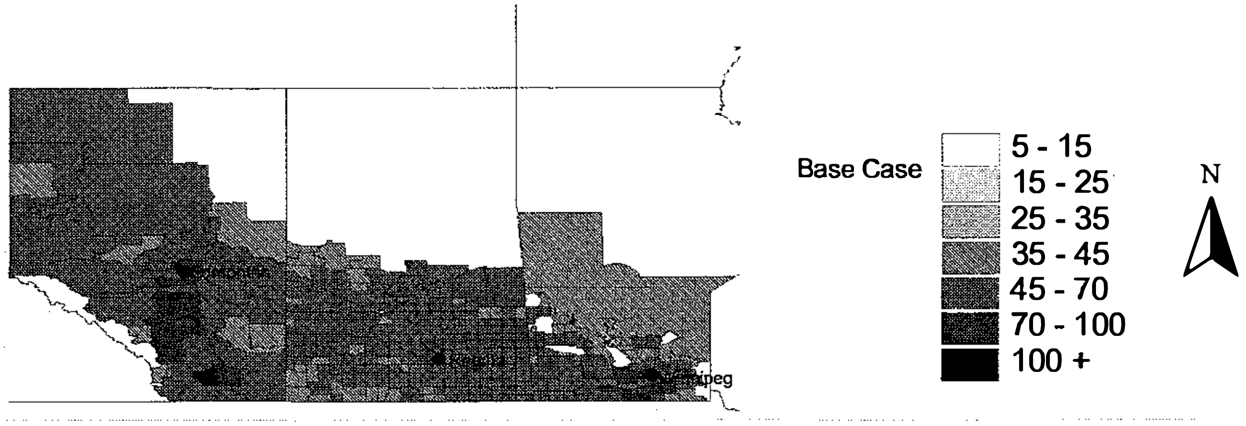

Figure 4. Break even values by Census Consolidated Subdivision (CCS) for Green Ash (good site), Prairie provinces, ($/tonne C).

Finally, it is worth noting the objectives of the study were to contribute primarily to the long term development of spatial models when evaluating land suitability for carbon sequestration. While the analysis provides a broad upper bound on the identification of suitable agricultural land under given site class and carbon price scenarios, there exist other important social and institutional factors that will determine the rate of afforestation uptake by prospective landowners. This includes such factors as land ownership patterns, availability of suitable growing stock, technical expertise and the structure of financial incentives for promoting large scale planting programs over the medium term (National Climate Change Process Forest Sector Table, 1999).

CONCLUSIONS

This analysis demonstrates the benefits from integrated use of local area data on land opportunity costs and capability factors in assessing land suitability for afforestation. The framework draws upon provincial growth and yield estimates to simulate C fluxes and afforestation benefits from both standing and harvested biomass. Land opportunity costs were explicitly included by comparing the relative economic values from agricultural production to afforestation inclusive of a flow of benefits generated from carbon sequestration. Given considerable uncertainty over future carbon prices, the framework allows the analyst to assess land suitability by referring to break-even carbon prices and can therefore make their own judgement on the problematic issue of future carbon credit values. The discount rate also had a large impact on aggregate land suitability indicative of the economic trade-offs involved in land-use conversions. To improve spatial models like these further research is needed to develop higher resolution data on forest growth and carbon pools by species; and identify spatial variation in private landowner attitudes and non-financial benefits (e.g. wildlife habitat). Research is being undertaken to develop better spatial estimates of forest site productivity for selected species across forested and non-forested landscapes in Canada (Bernier, P., personal communication, Canadian Forest Service, May 2000) drawing on generalised physiological growth models (Landsberg and Waring, 1997). A better understanding of potential markets for biomass products and carbon trading benefits will also help to limit the scope of these types of analyses.

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RELATIONSHIPS BETWEEN VEGETATION AND CLIMATE CHANGE IN TRANSBAIKALIA, SIBERIA

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ABSTRACT

Two bioclimatic models of different ranks were developed that predict orobiome (a mountain biome) and forest stand characteristics (tree species composition and wood volume) based on two climatic indices - Temperature Sums (base 5°C) and Dryness Index. Mountain vegetation and live phytomass distribution over the Lake Baikal basin and stand characteristics distribution over a mountain range in Central Transbaikalia were modeled under current and future climate, coupling the bioclimatic models and climatic layers of different resolution. Lowland vegetation is predicted to shift 250 m upslope and highland vegetation is predicted to shift 450 m upslope resulting in considerable reduction of tundra and light-needled taiga and expansion of forest-steppe. Total phytomass within the entire basin is predicted not to change much - 2.3 Pg of current and 2.5 Pg of future phytomass. Stand phytomass across the range is predicted to increase from a current value of 80-160 t ha^{-1} to 200-280 t ha^{-1} in a new climate.

INTRODUCTION

The Lake Baikal basin, an area of outstanding universal value, has been designated as a World Heritage Site. Lake Baikal has 20% of the world's fresh water and more endemic species of flora and fauna than any other lake in the world (Galazii, 1984). About 90% of the water that flows into the lake is filtered by the surrounding forests. These mountain forests play a unique protective role. To a great extent their health and condition determine the quality and quantity of water in Baikal. The forest distribution and productivity, as well as changes of forest functions under environmental impacts, are of a great importance for evaluating the future of Baikal.

Potential global climate warming could alter distribution and productivity of vegetation as well as associated biodiversity of the basin. Global mean temperature is expected to rise by as much as 3°C with a current rate of warming of 0.26°C per decade (Woodwell and Mackenzie, 1995). As predicted in numerous climate change studies, consequences of such sharp warming can be very serious and sometimes even catastrophic, especially in the boreal zone where the most intense warming is predicted to occur.

In this study, we will address how vegetation of the Lake Baikal basin might respond to climate change at two levels: an altitudinal belt, orobiome, over the entire basin and a stand in a locality. For this purpose, we developed an orobiome vegetation model and a higher resolution stand model based on climatic parameters. Both vegetation models were coupled with a climate change scenario (+2°C summer temperature and +20% annual precipitation) to predict potential vegetation types, tree species distribution and phytomass change.

METHODS

We modeled regional climates based on physiography (elevation, latitude, and longitude) and site climates based on topography (elevation, aspect, and slope) and then developed bioclimatic models to predict regional vegetation and local forest stand characteristics under current and future climate scenarios.

An Orobiome Vegetation Model

Eight orobiomes were identified by biogeographers over the Lake Baikal basin from their physiognomy, forest and ground layer structures (Smagin et al. 1980): mountain tundra, dark-needled subgolets (open) taiga, light-needled subgolets (open) taiga, dark-needled mountain taiga, dark-needled chern ("black" in Russian) taiga, light-needled mountain taiga, forest-steppe with light-needled subtaiga, and mountain steppe. Earlier studies of the mountains of South Siberia (Tchebakova and Parfenova, 1991; Monserud and Tchebakova, 1996) showed that two climatic indices, Dryness Index (DI, the proportion of available energy, radiation balance, to energy required to evaporate annual precipitation) and Temperature Sums (accumulated for the growing season with daily temperature above 5°C (TS_5)), were good predictors of vegetation distribution.

Temperature Sums were taken from Reference books on climate (1967-1972) and dryness index (DI) was calculated from the Budyko's theory (1974). Data of 40 weather stations within the basin having records of temperature, humidity, and cloudiness were used to calculate radiation balance (B) necessary to calculate DI. Approximated radiation balance error does not exceed 10% and primarily varied between 2 and 4% (Tchebakova and Parfenova, 1991) when compared with values observed at 14 actinometric stations over a large region around Lake Baikal. Based on estimated radiation balance, a regional regression was developed that predicted annual radiation balance (B) from Temperature Sums, base 5°C (TS_5), with $R^2 = 0.68$:

$$B(\mathrm{kcal}{*}\mathrm{cm}^{-2}{*}\mathrm{yr}^{-1}) = 19.0 + 0.0118 * TS_5 \quad [1]$$

Dryness Index was calculated using equation (1), for 90 stations having records on temperature and precipitation. An orobiome name was assigned to each of these stations using a Vegetation Map (Belov et al. 1972) and geobotanic literature (Smagin 1980). 90 orobiomes were displayed as a climatic ordination in the Temperature Sums (base 5°C), and Dryness Index axes. Border values of climatic indices between adjacent orobiomes were identified (Table 1). This table serves as a regional vegetation model determining an orobiome from climatic parameters.

To delineate orobiome distribution over the basin we used a Digital Elevation Model (DEM) with pixel size of 30 seconds. The Lake Baikal basin is located east of the lake between 102°nd 114°E longitude and 49°N (the southern border of Russia) and 56°N latitude (Fig. 1). We used only the Russian portion of the basin in our analysis.

Table 1. Climatic limits for the orobiomes in the Lake Baikal basin.

	Temperature Sums, base 5°C		Dryness Index	
OROBIOMES	Lower bound	Upper bound	Lower bound	Upper bound
Tundra	none	less than 800	none	none
Open Dark-needled taiga	800	1150	none	less than 1.0
Open Light-needled taiga	800	1150	1.0	2.0
Mountain Dark-needled taiga	1150	1500	none	less than 1.0
Mountain Light-needled taiga	1150	1750	1.0	2.3
Forest-steppe/subtaiga	1750	1950	1.0	2.3
	1950	none	1.0	2.0
Chern (black) taiga	1500	None	none	1.0
Steppe	1950	None	2.0	3.0

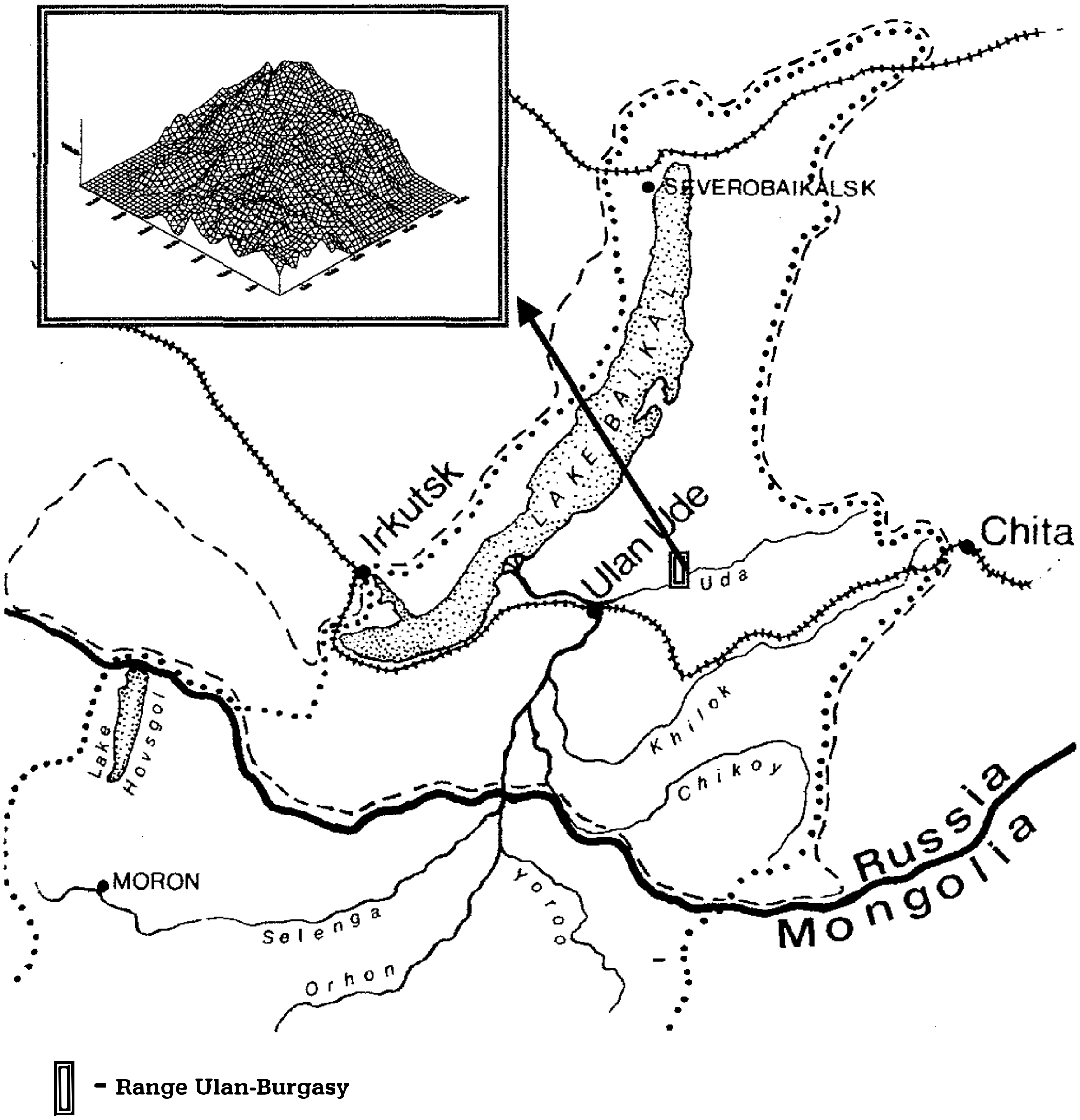

Figure 1. The Lake Baikal basin (The Lake Baikal region.., 1993) and location of Range Ulan-Burgasy.

Based on data of 90 weather stations we developed multiple linear regressions that predicted Temperature Sums (eq. 2) and Dryness Index (eq. 3) from physiographic parameters (Elevation, Latitude, Longitude):

$$TS_5 = 5496.8 - 0.904*Elev - 53.075*Lat - 2.164*Lon \quad (R^2 = 0.85) \qquad [2]$$

$$DI = 6.23 - 0.00125*Elev - 0.\text{[illegible]}13*Lat - 0.0242*Lon \quad (R^2 = 0.64) \qquad [3]$$

Using the regression equations (2) and (3) and the DEM, we mapped distributions of both climatic indices over the basin. Vegetation was mapped within the basin under the current climate by coupling pixel-by-pixel climatic indices TS_5 and DI with the orobiome model (Fig. 2a). Comparison between observed (Belov et al., 1972) and predicted vegetation using kappa statistics showed good agreement (overall k = 0.44)

Phytomass density for each orobiome was derived from analogous Siberian zonobiome phytomass densities compiled by Monserud et al. (1996) from Russian literature. Phytomass distribution was modeled based on the orobiome distribution. Total phytomass for the basin was calculated as a product of orobiome phytomass density and area (Table 2).

Forest Stand Models

Stand models were developed relating stand characteristics (tree species composition and stocking volume, m^3) to site climate across the mountain range Ulan-Burgasy in Central Transbaikalia (Fig. 1). Data were taken from 500 inventory plots along a transect 38 km long and 34 km wide.

Site climate was calculated for each plot with respect to topography (elevation, aspect, and slope). Temperature Sums and annual precipitation were calculated using different lapse rates for leeward and windward slopes. Radiation balance calculated for the flat surface at any elevation was corrected for aspect and slope.

Stand models were developed applying multiple piece-wise linear regressions to relate stand tree species composition and each tree species stocking volume. Combinations of the two climatic indices explained up to 90% of the variation in stand composition and 70% of the variation in stocking. Total stocking of a stand was calculated as a sum of each tree species portion. Stand wood volume was converted to total live phytomass using a conversion

Table 2. Phytomass under current and a warmed climate.

Orobiome	Current climate			Warmed climate[1]	
	Phytomass density, t ha^{-1}	Area (10^4ha)	Phytomass (10^4ton)	Area (10^4ha)	Phytomass (10^4ton)
Tundra	18.0	53.3	959.4	2.3	41.4
Open Dark-needled and Light-needled taiga	26.3	349.2	9184.0	73.0	1919.9
Dark-needled mountain taiga	143.0	182.1	26040.3	279.0	39897.0
Light-needled mountain taiga	113.0	1474.5	166618.5	269.5	30453.5
Forest-steppe	118.7	373.3	23144.6	1390.1	86186.2
Black taiga (chern)	197.0	3.0	591.0	464.4	91486.7
Steppe	15.3	161.7	2474.0	118.8	1817.6
Total:		2597.1	229011.8	2597.1	251802.4

[1] increase of 2°C to summer temperature;increase of 20% to annual precipitation

coefficient of 0.55 for all three tree species (Alexeyev and Birdsey, 1998).

Our estimates of stand phytomass based on inventory data demonstrated that the numbers obtained were quite realistic (80-160 t ha^{-1}) and comparable to phytomass measured in the field (Bazilevich, 1993).

The DEM (150 m pixel) was coupled to calculated site climates to produce Temperature Sums and Dryness Index surfaces over the transect. Stand models coupled with these climatic surfaces produced both stand composition and total live phytomass distribution across the range.

Climate Change Scenarios

Because GCMs generally operate at low spatial resolution (300 km) it is difficult to predict climate change impacts in detail for mountain regions on a much smaller scale. In our study, in the absence of a specific climate change scenario for the Baikal basin we used values of temperature and precipitation change generalized from several climate change scenarios with high confidence (Climate Change 1995, 1996). These numbers are 2°C increase of summer temperature and 20% increase of annual precipitation.

Using this climate change scenario, TS and DI were recalculated for 90 weather stations in a warmed climate, related to topography and mapped over the basin to get new climatic surfaces. The orobiome model was rerun under changed TS_5 and DI to obtain a new orobiome distribution (Fig. 2b) and phytomass change (Table 2).

The same scenario was applied to the stand models to recalculate stand composition and wood volume distribution over the range.

RESULTS

The 2°C increase in temperature would cause border shifts of about 250 m for the lower orobiomes and 450 m for the upper orobiomes. These results correspond to the "Hopkins bioclimatic law" (Climate Change 1995, 1996).

All orobiomes were predicted to change to some degree (Fig. 2a; Table 2). Tundra and open taiga would nearly disappear, light-needled mountain taiga would decrease to one fifth of its original size and forest-steppe would increase 4 times of its original size and occupy more than one half of the region. Because warming would be accompanied by a 20% increase of annual precipitation, conditions will be better suited for the dark-needled species (mostly *Pinus sibirica* and *Abies sibirica*) than today, but the area occupied would not exceed 20% of the total area.

A remarkable effect of warming will be an impressive expansion of the black taiga, a humid and relatively warm type of taiga that is currently found only on the windward slope of Range Khamar-Daban at the southern extremity of Lake Baikal. A floristically rich ground layer (30-50 species) with large ferns and herbs, sometimes with relic species remaining from the Holocene optimum, is characteristic of the black taiga. Several new black taiga locations can be found under a warmed and humid climate.

Middle elevation mountain taiga would be mostly replaced by less productive forest-steppe resulting in about 45% phytomass loss, although a great expansion of the most productive low elevation black taiga would balance this loss. Total phytomass over the entire area does not show a significant change: predicted 2.5 Pg compared to current 2.3 Pg (Table 2).

Pinus sibirica (Siberian cedar), *Pinus sylvestris* (pine), and *Larix sibirica* (larch) are the main tree species dominating forests across Range Ulan-Burgasy. Stand models coupled with climatic surfaces produce tree species distribution and stocking volume converted to live phytomass across the range Ulan-Burgasy (Fig. 3a). *P. sibirica* dominates forests under moist and cool climates in the highlands and *P. sylvestris* dominates forests under dry and warm climate in lowlands. *L. sibirica* may be equally mixed with pine in dry sites and cedar in cool sites.

Stands having the greatest productivity are found at middle elevations. Under current climate, pine stands are predicted to have some 100-150 m^3 ha^{-1} wood volume; Siberian cedar and larch stands - 50-100 m^3 ha^{-1}. Values for total stocking of all three tree species range result, on average, from 200-250 m^3 ha^{-1} up to 300-350 m^3 ha^{-1} on warm (south-facing slopes) and moist (middle elevations) sites. Total stand phytomass is predicted to increase from 80-160 t ha^{-1} to 200-280 t ha^{-1} under climate change, favorable for woody vegetation (Fig. 3b).

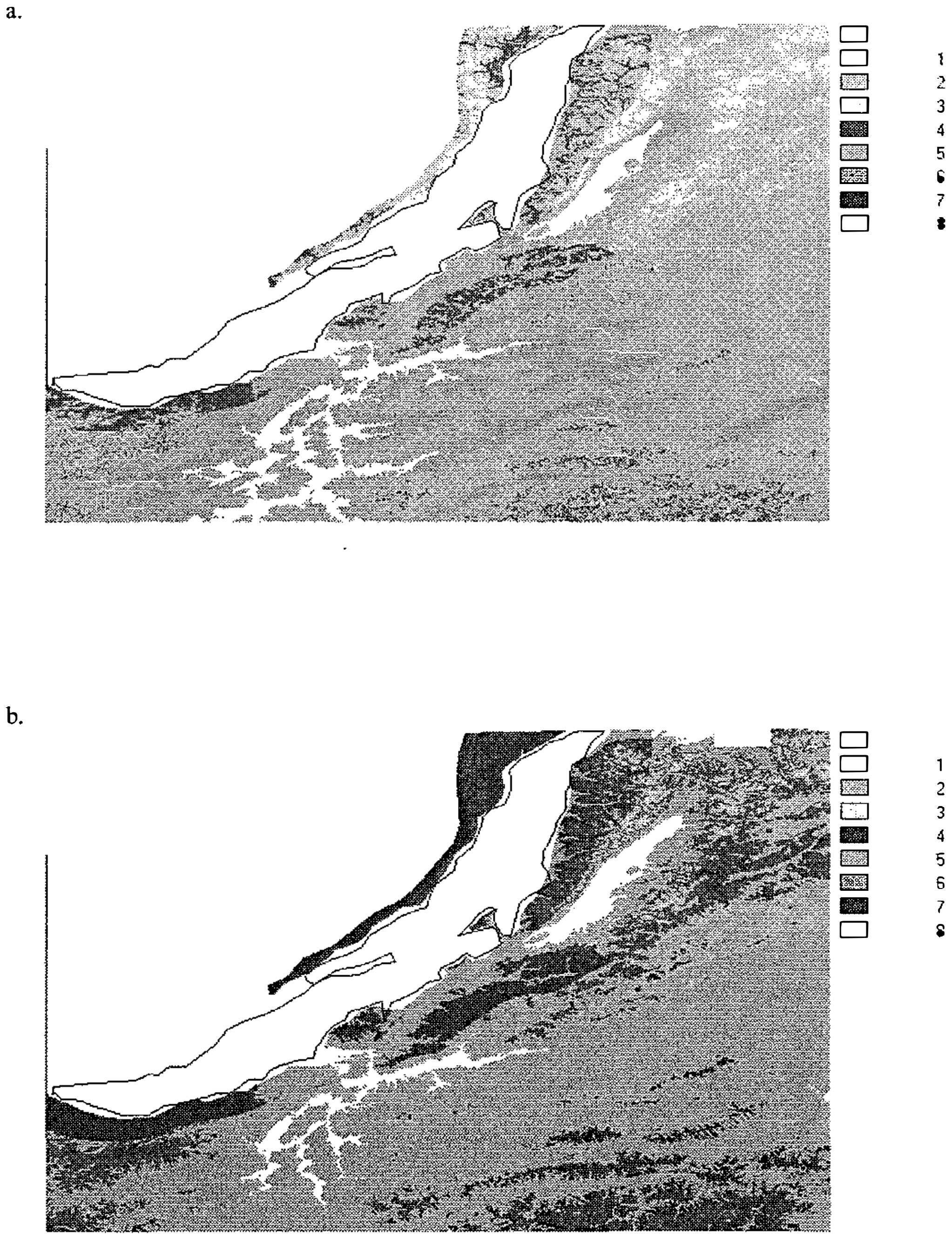

Figure 2. Orobiome distribution under (a.) current climate and (b.) a warmed climate. 1. tundra, 2. dark-needled open taiga, 3. light-needled open taiga, 4. dark-needled mountain taiga, 5. light-needled mountain taiga, 6. subtaiga/forest-steppe, 7. black taiga, 8. steppe.

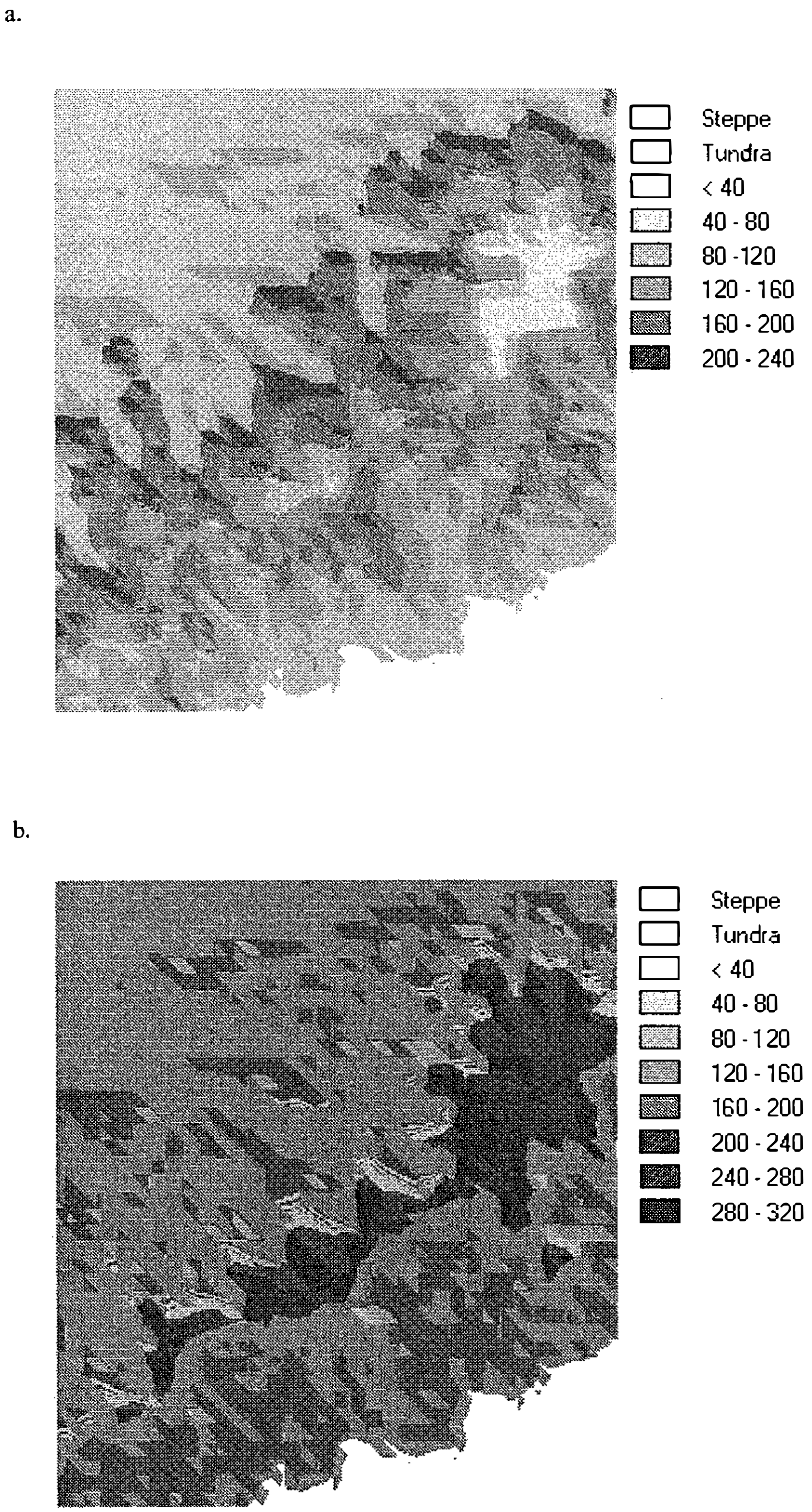

Figure 3. Phytomass distribution over Range Ulan-Burgasy, t h^{-1} under (a.) current climate and (b.) a warmed climate.

DISCUSSION

First assessments of climate change impacts on mountain ecosystems in the Lake Baikal basin have been conducted at two levels - an orobiome and a stand. A simple static model was developed to predict orobiomes from two climatic indices - Temperature Sums (base 5°C) and Dryness Index. Multiple regression models were developed to predict stand characteristics from the same climatic indices. We applied the orobiome model to evaluate how orobiomes may possibly shift and total phytomass may change over the basin in a new climate, and we used a stand level model to predict stand phytomass followed forest structure change in a warmed climate.

There have been few studies of climate change impact in mountains. Because the majority of GCMs are of low resolution (5-7 degrees) and Lake Baikal basin are located in 2 by 2 pixels, we used general numbers for temperature and precipitation change resulting from the GCMs predicitons for the entire territory.

The basin forests play a unique role in protecting the area from erosion and in filtering water that flows into the lake. We predict that under warming taiga orobiomes will be replaced by subtaiga and forest-steppe on one half of the Lake Baikal basin. The filtering capacity of forest-steppe is much weaker and the susceptibility of forest-steppe to erosions is greater than that of mountain forests (Lebedev et al., 1979). Consequently, we may assume that the quality of water in the lake may worsen under warming.

Mosaic mountain habitats, being very heterogeneous, are known to be vulnerable to global warming, but the higher diversity of these habitats creates the opportunity for species to survive in a changing climate. In mountains, plant species are likely to disperse rapidly. Having natural migration rates of 30-50 km/century, they may cover distances of hundreds of meters between adjacent orobiomes. The distances between adjacent zonobiomes require species to migrate across plains for hundreds of kilometers. Therefore, mountains have the capacity to serve as effective conservation reserves under predicted warming regimes.

We are quite aware that our orobiome shift predictions at high elevations may be alted by other climate-forcing factors like wind, snow, and permafrost, which are not included in our model. A dynamic component, when incorporated into the model, would certainly inject some vitality into the model as well. The model is open to new developments.

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RESERVES AND DENSITY OF ORGANIC CARBON IN FORESTS OF RUSSIA

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ABSTRACT

Reserves of organic carbon (C) were estimated for phytomass ($C_{phytomass}$), soil without coarse woody debris (C_{soil}), and as totals (C_{total}). Preliminary estimates for C of coarse wood debris (C_{CWD}) were calculated based on simple assumptions. Estimates were done for three landscape subzones (the northern, middle, and southern) within each of four macro-regions (European-Ural, West Siberia, East Siberia, and Far East) using data from 59 ecoregions in the Asian part, and 56 federal districts in the European part, of Russia. These data include: (a) state forest inventory; (b) phytomass estimates for forest stands; (c) averaged phytomass estimations for meadows, peat bogs, and other non-forest areas, and (d) reserves of C_{SOIL} for stands of the dominant tree species, both for non-forested areas and non-forest lands, according to data of the Russian Forest Fund (RFF).

The total land area in the RFF is 1110.5×10^6 ha. The total reserves of $C_{phytomass}$ and C_{soil} were estimated at 34.35×10^9 t C and 172.43×10^9 t C, respectively. C_{CWD} was estimated at 3.6×10^9 t C. C_{TOTAL} in the RFF was estimated at 210×10^9 t C. If all types of forest land ownership are considered, the land area of Russian forests must be increased by 8%. Therefore, C_{total} may be as high as 238.8×10^9 t C, including $C_{phytomass}$ (36.92×10^9 t C), and C_{soil} (182.82×10^9 t C). The distribution of $C_{phytomass}$, C_{soil}, and C_{total} is shown for four macroregions and three subzones within every macroregion. The density of C_{total} ranges from 170-202 t C ha^{-1} for macroregions and 179-202 t C ha^{-1} for the three landscape subzones. $C_{phytomass}$ estimates range from 24-39 t C ha^{-1} for macroregions and 18-48 t C ha^{-1} for subzones. The ratio $C_{soil}/C_{phytomass}$ exhibits a similar distribution pattern and values ranged from 2.4 to 3.9 for the regions and from 2.6 to 5.5 for the subzones.

INTRODUCTION

Nearly one-quarter of the globe's forest reserves can be found in the Russian Federation (RF). Forests cover 45.3% of the area which includes 774×10^6 ha of closed forests and a growing stock of 82×10^9 m^3 (Lesnoi Fond Rossii, 1999). The contribution of Russia to the world's standing stock is about 50% of coniferous forests and about 10% of deciduous forests.

The resource potential of Russian forests is well-known. Their ecological potential and contribution to processes in the biosphere functioning, including the carbon cycle, is less well understood. The absence of reliable estimates for net primary production (NPP), net ecosystem production (NEP) and, especially, net biome production (NBP) for Russian forests creates uncertainty about published patterns for the current carbon budget and forecasting it's response to global changes.

Understanding the carbon cycle in forests of Russian Eurasia is complex for many reasons, including an inventory of carbon and phytomass (plant biomass) in the territory of the Russian Forest Fund (RFF) is not a primary goal of the forest administration. Therefore, data acquired by soil scientists, geobotanists, and foresters over the course of the International Biological Program (IBP) and in the 1980-1990's, are hitherto basic to understanding the problem of carbon cycling in Russian forests.

Landscape maps, patterns of natural regions in the territory and other means were initially used for estimating pools and carbon fluxes in the former USSR and RF. The areas within different contours were scanned from maps using various techniques. Sometimes GIS-technologies were used. Area parameters were multiplied by averaged estimates of phytomass (carbon) for the same plant contours.

The method of integrating data from the RFF inventories and data on biological productivity for different types of forest ecosystems was later instrumental to estimating phytomass in Russian forests (Mackarevsky, 1991; Isaev et al., 1993, 1995a, 1995b; Alexeyev and Birdsey, 1994, 1998; Kolchugina and Vinson, 1995; Nilsson and Shvidenko, 1998; and others). Surveys for different regions of the RF were used in doing so. Estimations of soil carbon reserves were made using small scale soil survey maps (Orlov et al., 1996; Rozhkov et al., 1997). The same methods were used to estimate carbon pools in peat bogs and in soils of swampy forests (Vomperskij et al., 1996, 1999).

In this study estimates of phytomass carbon ($C_{phytomass}$), soil plus soil litter (C_{soil}) and total carbon (C_{total}) were mated with information from the RFF inventory. We think this approach is more reliable compared with scanning large-scaled maps for determination of land areas.

Data of the RFF inventory concerning areas and wood stocks distributed by age groups for stands of the main tree species or, more rarely, their combinations (coniferous, hardwood, and softwood) and for shrub species were used for estimating $C_{phytomass}$ in closed forests.

In this work carbon pools were estimated in the territory of the RFF divided into landscape subzones and macro-regions, although the former is emphasized in this paper. This method will increase the accuracy of calculations of $C_{phytomass}$ and C_{soil} and therefore, must positively affect the estimation of carbon fluxes within the RFF.

MATERIALS AND METHODS

Data Sources

Our estimates used data from three different sources which include;

1. Inventory of the RFF;
2. Biological Productivity of Forest Ecosystems Database (Utkin et al., 1994) and,
3. Soil Carbon in Various Land Categories of RFF (Chestnykh et al., 1999).

The Inventory of the RFF has been conducted every five years since the latter part of the 1930's. The results are available in computerized form or as handbooks (Lesnoi Fond SSSR, 1990; Lesnoi Fond Rossii, 1995, 1999). The handbook includes various information used by forest managers and is differentiated at three territorial levels:

1. natural macroregions (the European-Ural part, West Siberia, East Siberia, the Far East);
2. natural-economic regions (13 regions) and,
3. federal districts of the RF.

The most abundant information is available for forests and forest lands of the Federal Forest Service of the Russia (FFS) which covers about 92% of the total forest territory.

The computer database "Biological Productivity of Forest Ecosystems" (Utkin et al., 1994) is the second principal source of information. It is a collection of data from the literature (about 750 references) which is continuously updated. It combines information on nearly 1500 forest ecosystems of the former USSR and other states in northern Eurasia with some data from North America. Comparison of phytomass data using allometric equations and regression analysis showed that genetically similar trees on both continents differ insignificantly (Hamburg et al., 1998; Utkin et al., 1998b).

The database provides information on;

1. location and environmental conditions;
2. stand structure and plant communities;
3. phytomass of the stand and the understory;
4. net primary production and,
5. forest litter and humus of the soil layer.

Data for tree age, density, stock, phytomass of stems, branches, foliage, and roots of the main forest-forming species were used to calculate phytomass/stock (Ph/S ratios). The latter were calculated using two approaches: (1) averaged statistical values (Isaev et al., 1993, 1995b) and, (2) values estimated as a function of stocking and age (Zamolodchikov et al., 1998). Furthermore, the possibility of estimating phytomass from the product of the sum of the basal areas and the mean stand height was checked (Utkin et al., 1998a).

Phytomass of the understory in plant communities is not directly dependant upon standing stocks. Therefore, it was estimated for stands as the mean of cuttings taken from sample plots. Age groups of stands were not considered. A similar method was used for non-forested lands (clearcut and burnt areas, and others) and for non-forest lands (meadows, bogs, and others).

The third source of data was the "Database on Soil Organic Carbon" which includes information on more than 1000 soil pits from 300 literature sources (Chestnykh et al., 1999). The database is continually updated and now includes data on forest soils, but information on "other lands" in the RFF is insufficient. Averaged estimates for soil organic matter per unit area were used to extrapolate to the regional scale.

Structure of the RFF Inventory

The inventory of RFF lands is conducted for three large land categories: (1) forested; (2) non-forested, and (3) non-forest lands. Each category is divided further into land types. Land types in the forested area are defined by the dominant species within the stand, non-forested areas by their origin (clearcuts, burns, and others), and in non-forest areas by type of land or landuse (hay-makings, pastures, roads, swamps, and so on). Forest-forming tree species are classified into three groups (1) coniferous, (2) hardwood deciduous (oak, beech, maple, ash, stone birch) or, (3) softwood deciduous (birch, aspen, alder, willows and others). Lands under shrub species are accounted for separately. These lands are often dominated by communities of dwarf Siberian pine (*Pinus pumila* (Pall.) Regel) and occupy an area of 38.3 × 10^6 ha (Utkin and Pryazhnikov, 1999). In addition, oak stands are differentiated into high-forest (seedling origin) and coppice stands.

Stands of the most abundant, or main forest-forming species are divided into six age groups:

1. young stands of the first age class;
2. young stands of the second age class;
3. middle-aged stands;
4. premature ;
5. mature and,
6. over-mature stands.

With the exception of young stands, the actual age of stands within any group may differ significantly depending on the age to stand maturity.

Ph/S ratios were estimated to calculate $C_{phytomass}$ for most of the main forest-forming species. Calculations for other tree species were done using mean Ph/S ratios for the three species groups previously described.

Forest Zones of the RF

The borders of the territories of many federal districts in the RF are meridionally defined and do not harmonize with existing natural forest regions. Therefore, we separated Russian forests into three landscape bands: (1) northern, (2) middle, and (3) southern, according to the landscape map of Isachenko et al. (1988). The boundary between the northern and middle subzones passes through somewhere around 63°- 64° north latitude in Europe and Siberia and 66° in Far East. The boundary between the middle and southern subzones passes through somewhere around 52°-60° north latitude with some shifting in a southward direction in the Far East. Vegetation of the mountains was defined by associated subzones at lower latitudes. Vertical zonality of landscapes was not considered.

Calculations Methods

Correcting areas of RFF lands

Since information on the forest fund of the RFF is basic, it can be applied to forest lands outside of the RFF. In the last two forest inventories correction factors were applied to different land categories to account for these additional forest lands.

The area of the additional forest lands is about 8% of the total area managed by the RFF. Distribution amongst the land categories in the additional forest lands is similar to the RFF, therefore estimates from RFF lands can be readily applied to the additional forest land in order to estimated totals for RF.

Conversion of phytomass and soil organic matter mass to carbon

Conversion coefficients per 1 kg of over-dried phytomass are 0.5 kg C for stems, branches and roots, and 0.45 kg C for needles, foliage, and understorey plants. The unified conversion coefficient equal to 0.57 kg C per 1 kg of soil organic matter was used for soil organic matter together with forest litter.

Calculation of C_{soil}

These values were estimated by analyzing data from corresponding databases and data from the forest inventory in 1993 (Lesnoi Fond Rossii, 1995). Using the database "Reserves of Soil Carbon in Lands of RFF" averaged reserves of organic matter were estimated for all land categories of the RFF including separate estimates for living forest-forming species on forested lands. Reserves of organic matter (t ha^{-1}) were converted to carbon (t C ha^{-1}). The latter values were taken as multipliers when calculating C_{soil} reserves based on the areas of separate lands. Multipliers for C_{soil} of different forest ecosystems were grouped by subzones and macroregions.

Calculation of $C_{phytomass}$

Estimating $C_{phytomass}$ of stands from the database "Biological Productivity of Forest Ecosystems" (Utkin et al., 1994) was preceded by calculations of Ph/S ratios for all main forest-forming species within age groups of stands. First, individual relations for stems, branches, foliage or needles were determined. Their sum was taken as the coefficient for converting stand stocks to aboveground phytomass. Next, the share of stumps and roots within the total stand phytomass was estimated using the ratio of aboveground to belowground phytomass.

A corresponding parameter for estimating total phytomass was calculated after combining aboveground and belowground phytomass of stands and dividing this sum by volume reserves. Multiplying this parameter by 0.5 gave the ratio between carbon reserve and stem stock (C/S ratio) for stands of every forest-forming species within a defined age group. Analogous coefficients can be calculated through equations describing the dependence of C/S ratio upon the age of stands (Zamolodchikov et al., 1999).

$C_{phytomass}$ for non-forested and non-forest lands was estimated in the same manner as C_{soil}.

Calculations of total reserves of $C_{phytomass}$ and C_{soil} for territorial contours at different levels

Carbon reserves were converted from reserves of phytomass and soil organic matter determined separately for 59 ecoregions of Siberia and the Far East and for 56 federal districts in the European part of the RF. Reserves of $C_{phytomass}$ and C_{soil} in macroregions and subzones were determined from total estimates for these 115 units.

Phytomass of all land categories in the RFF was estimated for each of the 115 units. A similar approach was used to calculate carbon reserves for soil organic carbon. Forest soil profiles were related to the forest-forming tree species, and stand age was ignored. Carbon reserves in peat were determined only to a depth of 100 cm instead of the whole peat layer. The integral parameters of $C_{phytomass}$, C_{soil}, and C_{total} for an ecoregion or federal district were calculated by summing phytomass and soil carbon reserves for all types of land categories within a forest fund.

RESULTS

C/S ratios

Values of C/S ratios for stands of the main forest-forming species, with consideration for different age groups, revealed some general tendencies by subzones. The C/S ratio usually decreased in a southward direction and with increasing stand age. Relatively high values of C/S ratio are characteristic for younger stands. Values of C/S ratios for dwarf Siberian pine stands range from 0.8 to 1.2. Calculation of these coefficients took into consideration that long stem segments of *Pinus pumila* are buried under forest litter and in the upper soil layer, whereas estimating volume stock of the coppice was made by aerial photographs and concerned the visible parts of plants only (Utkin and Pryazhnikov, 1999).

Reserves of $C_{phytomass}$ and C_{soil} in the RFF

Data from forests and forest lands of the FFS were used as a basis to estimate $C_{phytomass}$ and C_{soil} within four macroregions and three subzones (for 12 zonal-provincial regions). Although our results focus on the 12 regions, it is worthwhile to briefly consider pools of $C_{phytomass}$ and C_{soil} of different forest land categories depending on age and species structure of the stands.

The total land area in the RFF is 1110.5×10^6 ha, consisting of 64% forests, 10% non-forested (burned areas, clearcuts, etc.), and 26% non-forest lands. The total reserve of $C_{phytomass}$ (34.35×10^9 t C) is distributed between forests (98%), non-forested (1%), and non-forest areas (1%). The total reserve of C_{soil} (172.43×10^9 t C) is distributed between forests (61%), non-forested (8%), and non-forest areas (31%). The C_{total} (206.78×10^9 t C) is distributed as follows: forests (66%), non-forested (8%), and non-forest areas (26%).

Within the forested area (707×10^6 ha), 72% is coniferous, 2% is hardwood deciduous, 16%

is softwood deciduous, and 10% is other tree and shrub species. The distribution of $C_{phytomass}$ in the forested areas is 77% coniferous, 4% hardwood deciduous, 15% softwood deciduous, and 4% other. The distribution of C_{soil} is 69% coniferous, 3% hardwood deciduous, 15% softwood deciduous, and 13% other.

The forested area is distributed by age of stands as follows:

1. young stands of two age groups (18%),
2. middle-aged stands (26%),
3. premature stands (10%),
4. mature and over-mature stands (46%).

$C_{phytomass}$ is distributed between these four groups as 6%, 25%, 12%, and 57%, respectively, and C_{soil} as 16%, 27%, 10%, and 47%, respectively.

The part of the area occupied by older stands clearly increases in an eastward direction. In the European-Ural and West Siberia the percentage of mature and over-mature forests is only 7% and 6%, respectively, but in East Siberia and the Far East these values increase to 15 and 18%, respectively. The younger forests may play the most important role in CO_2 sequestration from the atmosphere in the two western macroregions. In contrast, forests of the two eastern macroregions mainly serve the purpose of carbon storage (Utkin, 1995).

The contribution of separate tree species to the carbon cycle is fully determined by the area they occupy. The most widespread forest-forming tree species in Russia are larch (37% of forested area), followed by pine (16%), birch (12%), spruce (11%), and Siberian pine (5%). Their individual contributions to $C_{phytomass}$ are 36%, 17%, 11%, 12%, and 9%, respectively, and to C_{soil} 34%, 13%, 12%, 12%, and 8%, respectively. In total, the contribution of these five tree species is 81% of the forest area, 85% of $C_{phytomass}$ and 79% of C_{soil}.

Integral zone-macroregion estimates of $C_{phytomass}$ and C_{soil} pools for forested areas of RFF are given in Table 1, and for non-forested together with non-forest lands in Table 2.

The total carbon reserve, without coarse woody debris, within RFF is estimated at nearly $207{\times}10^9$ t C, including forested (66%) and non-forested and non-forest lands (34%). The reserve of C_{soil} is five times greater than $C_{phytomass}$ for Russia as a whole. The degree to which C_{soil} exceeds $C_{phytomass}$ depends on the ecosystem or region under consideration. On average, forest C_{soil} exceeds forest $C_{phytomass}$ by a factor of 3.3, (ranging from 3 to 6 in 15 regions) compared with an average value of 24.3 for non-forested lands (range of 21 - 37). C_{soil}:$C_{phytomass}$ ratios are quantitatively related to the area distribution of different land categories within the RFF. In forests of the northern subzone the ratio C_{soil}:$C_{phytomass}$ is, on average, 20-30% (rarely up to 50%) higher than ratios for the middle and southern subzones. Minimal values of ratios (around 2) are characteristic for the southern subzone of the European-Ural and East Siberia macroregions. These territories have been exposed to the significant impact of forest fires in the past, and in the forest-steppe zone of Siberia this impact is evident today.

The dissimilar structure of RFF lands in separate macroregions of Russia agree with differences in forest cover and bog cover percentages. The forest cover percent, or the percentage of forests within the total area of forests, are as follows by macroregions: European-Ural (39%), West Siberia (37%), East Siberia (56%), and the Far East (46%) (Lesnoi Fond Rossii, 1999). The bog cover percent, or the percentage of non-forest bogs within the RFF, for the same macroregions are 12%, 29%, 7%, and 7%, respectively. The highest values for bog cover percent occur in the European North and lowlands of West Siberia which agrees with the distribution of forest and shrub ecosystems on wet lands. The ratio C_{soil}:$C_{phytomass}$ is highest due to the large C reserves in the bogs.

Carbon Density in C_{soil} and $C_{phytomass}$ Pools

Carbon densities in the phytomass and soil (t C ha^{-1}) were calculated using integral estimates of $C_{phytomass}$ and C_{soil} pools on forested, non-forested and non-forest lands, divided by the area of each zonal-provincial regions. Carbon density of the phytomass varies from 25 to 60 t C ha^{-1} on forest lands over all subzones, with values increasing in a southward direction. Values of phytomass carbon density generalized for non-forested and non-forest lands, have no significant geographical differences and range from 5.4 to 8.2 t C ha^{-1} . Unlike forest ecosystems, carbon density of phytomass on non-forested lands decreases in a southward direction. Such a tendency is consistent with a change in the dominant species of the plant communities. Mainly evergreen species of the families *Vaccinaceae* and *Cyperaceae* are replaced by

Table 1. Carbon reserves and density pools $C_{phytomass}$, C_{soil} and C_{total} in forested lands of the RFF for separate macroregions and subzones (mean values ± SE).

Macroregion	Subzones	Area (10^6 ha)	Carbon reserves (10^6 t C)			Carbon density (t C ha^{-1})		
			$C_{phytomass}$	C_{soil}	C_{total}	$C_{phytomass}$	C_{soil}	C_{total}
1	Northern	53.58	2 121±148	7 921±854	10 041±1 002	39.6±2.8	147.8±15.9	187.4±18.7
	Middle	24.82	1 103±71	2 295±602	3 398±673	44.4±2.8	92.5±24.3	136.9±27.1
	Southern	58.55	3 491±321	6 190±893	9 682±1 214	59.6±5.5	105.7±15.2	165.4±20.7
	As totals	136.95	6 715±540	16 406±2 349	23 121±2 889	49.0±3.9	119.8±17.2	168.8±21.1
2	Northern	17.04	570±46	3 046±668	3 616±714	33.4±2.7	178.7±39.2	212.1±41.9
	Middle	28.32	1 233±80	4 023±716	5 257±796	43.6±2.8	142.1±25.3	185.6±28.1
	Southern	33.40	1 797±142	5 187±703	6 984±845	53.8±4.3	155.3±21.0	209.1±25.3
	As totals	78.76	3 601±268	12 255±2 087	15 856±2 355	45.7±3.4	155.6±26.5	201.3±29.9
3	Northern	14.18	469±37	2 848±574	3 317±612	33.1±2.6	200.8±40.5	233.9±43.1
	Middle	103.83	4 419±343	17 055±3 102	21 474±3 446	42.6±3.3	164.3±29.9	206.8±33.2
	Southern	99.56	5 943±433	15 304±2 270	21 247±2 703	59.7±4.3	153.7±22.8	213.4±27.1
	As totals	217.57	10 831±814	35 207±5 946	46 038±6 760	49.8±3.7	161.8±27.3	211.6±31.1
4	Northern	104.43	2 559±341	16 520±2 407	19 079±2 748	24.5±3.3	158.2±23.0	182.7±26.3
	Middle	132.70	5 882±505	17 063±2 008	22 946±2 514	44.3±3.8	128.6±15.1	172.9±18.9
	Southern	36.60	1 837±163	7 084±944	8 920±1 106	50.2±4.4	193.6±25.8	243.8±30.2
	As totals	273.73	10 278±1 009	40 667±5 359	50 945±6 368	37.5±3.7	148.6±19.6	186.1±23.3
1-4	Northern	189.23	5 719±572	30 333±4 503	36 052±5 076	30.2±3.0	160.3±23.8	190.5±26.8
	Middle	289.67	12 637±999	40 437±6 429	53 074±7 428	43.6±3.5	139.6±22.2	183.2±25.6
	Southern	228.10	13 068±1 059	33 765±4 809	46 833±5 868	57.3±4.6	148.0±21.1	205.3±25.7
	As totals	707.00	31 424±2 631	104 535±15 741	135 959±18 372	44.4±3.7	147.9±22.3	192.3±26.0

Note: Separate estimates for 59 ecoregions in the Asian part of RF and 56 federal districts of RF in the European-Ural region were used in calculating integral characteristics.

Macroregions: 1- European-Ural, 2 - West Siberia, 3 - East Siberia, 4 - the Far East.

Table 2. Carbon reserves and density pools $C_{phytomass}$, C_{soil} and C_{total} in non-forested and non-forest lands of the RFF for separate macroregions and subzones (mean values ± SE).

Macroregion	Subzones	Area (10^6 ha)	Carbon reserves (10^6tC)			Carbon density (t C ha^{-1})		
			$C_{phytomass}$	C_{soil}	C_{total}	$C_{phytomass}$	C_{soil}	C_{total}
1	Northern	21.25	148±21	3 966±437	4 113±458	6.9±1.0	186.7±20.6	193.6±21.6
	Middle	8.54	46±5	1 322±160	1 368±166	5.4±0.6	154.8±18.8	160.1±19.4
	Southern	10.94	71±7	1 509±174	1 579±182	6.5±0.7	138.0±15.9	144.4±16.6
	As totals	40.73	264±34	6 797±772	7 061±805	6.5±0.8	166.9±18.9	173.4±19.8
2	Northern	16.07	123±17	2 940±343	3 063±359	7.7±1.0	183.0±21.3	190.6±22.4
	Middle	23.89	141±16	4 726±523	4 867±539	5.9±0.7	197.8±21.9	203.7±22.6
	Southern	18.33	102±10	3 741±428	3 842±438	5.5±0.5	204.0±23.4	209.6±23.9
	As totals	58.29	366±42	11 407±1 294	11 773±1 336	6.3±0.7	195.7±22.2	202.0±22.9
3	Northern	24.22	198±23	4 456±596	4 655±619	8.2±1.0	184.0±24.6	192.1±25.6
	Middle	36.38	280±32	6 428±886	6 708±918	7.7±0.9	176.7±24.4	184.4±25.2
	Southern	19.24	109±12	3 382±461	3 491±473	5.7±0.6	175.8±24.0	181.5±24.6
	As totals	79.85	588±67	14 266±1 943	14 854±2 010	7.4±0.8	178.7±24.3	186.0±25.2
4	Northern	168.96	1 293±148	26 241±3 212	27 535±3 361	7.7±0.9	155.3±19.0	163.0±19.9
	Middle	47.75	357±39	7 764±1 087	8 121±1 126	7.5±0.8	162.6±22.8	170.1±23.6
	Southern	7.90	54±5	1 422±181	1 477±186	6.9±0.6	179.9±22.9	186.8±23.5
	As totals	224.61	1 704±192	35 428±4 481	37 132±4 672	7.6±0.9	157.7±19.9	165.3±20.8
1-4	Northern	230.50	1 762±209	37 604±4 588	39 366±4 797	7.6±0.9	163.1±19.9	170.8±20.8
	Middle	116.57	825±92	20 240±2 656	21 065±2 748	7.1±0.8	173.6±22.8	180.7±23.6
	Southern	56.41	335±34	10 054±1 245	10 389±1 278	5.9±0.6	178.2±22.1	184.2±22.7
	As totals	403.48	2 922±335	67 898±8 489	70 820±8 824	7.2±0.8	168.3±21.0	175.5±21.9

Note: Separate estimates for 59 ecoregions in the Asian part of RF and 56 federal districts of RF in the European-Ural region were used in calculating integral characteristics.

Macroregions: 1- European-Ural, 2 - West Siberia, 3 - East Siberia, 4 - the Far East.

grasses and herbs.

Carbon density in soils of forest and other land categories of the RFF increases in eastward and northward directions. The tendency of meridional changing C_{soil} is expressed less clearly. This is especially true for East Siberia, where middle and southern subzones constantly lose carbon owing to forest fires in Yakutia and the Far East. Non-forested lands of the RFF have insignificantly higher C_{soil} density compared with forest ecosystems. However, these differences are compensated for by the density of $C_{phytomass}$. As a result, the total carbon density (C_{total}= $C_{phytomass}$+C_{soil}) is rather similar in all ecosystem types within subzones, through zones and through provincial profiles across the whole Russia. This result indicates that it is acceptable to estimate the parameter of carbon density for the $C_{phytomass}$ and C_{soil} pools using the same approach for separate ecoregions and federal districts of the RF, even though they have a different structure.

Figures 1 and 2 show the spatial differentiation of carbon density in $C_{phytomass}$ and C_{soil} respectively. The RFF, especially in the Asian part of Russia, is characterized by low values for the density of the $C_{phytomass}$ pool. Forests growing under the best climatic conditions of the European-Ural macroregion, in the south of Siberia and in Primorsky krai must be considered as exceptions (Figure 1). The spatial distribution of carbon density in the C_{soil} pool is nearly the reverse of that observed for $C_{phytomass}$. The maximum values for carbon density of soils mostly occur throughout the European North, West Siberia, and the northern part of the Far East (Figure 2). The bigh carbon density of C_{soil} in these regions is attributable to the large areas of excessively wet lands and attendant low rates for decomposition of plant residues. In southern Siberia and Primorsky krai significant reserves of soil humus, instead of detritus accumulation, account for the high carbon density of C_{soil} in these regions. Abnormally low values of carbon density for C_{soil} in the middle and southern subzones of Yakutia and Zabaikal'e are undoubtedly related to frequent forest fires. The same can be said about other inland regions of Siberia and the Far East.

Annual Accumulation of Carbon

Information on the eco-physiology of forest ecosystems in Russia is very scarce and devoted to certain forest-forming tree species. Therefore, estimating NPP for all species within forests, with differentiation of stands by age groups and other dendrological divisions, is not yet possible. Calculations of annual carbon accumulation provides a notion of the dynamics of organic matter production in forest stands and other plant communities within the RFF. Methods for the calculations are given in detail by Isaev et al. (1995b). They are based on: (1) the analysis of successive changes in phytomass (carbon) reserves when stands move from a younger to older age group and, (2) the assumption that areas of stands are distributed more or less uniformly by their calendar age, within their age groups.

Young stands are characterized by more intensive increments compared with premature, mature, and especially over-mature stands in accordance with the regularity of forest growth. It is assumed that there is no accumulation of carbon in over-mature forests. In over-mature stands the current accumulation of phytomass compensates for the loss of carbon due to tree mortality (in part or in whole).

Estimates of carbon accumulation in stands of all forest-forming trees by age groups for the macroregions of Russia suggest that forests of the European-Ural and West Siberia macroregions are best suited to maximizing carbon sequestration. The dominance of old stands in the structure of the forest funds of the two other macroregions in eastern Russia allows forests to act only as storage in the form of C_{soil}, from previously accumulated carbon.

The spatial diversity of annual carbon accumulation by ecoregions and federal districts (Figure 3) suggests that forest ecosystems with low potential for sequestration of atmospheric CO_2 are distributed throughout nearly 80% of the territory in the RF. The average annual carbon accumulation is no more than 1 t C ha^{-1} $year^{-1}$. Nearly the same quantity of carbon transfers annually into forest litter and detritus (i.e. into the ecosystem components with the shorter period of carbon containment). As our calculations show, total accumulation of carbon within all lands of the RFF is estimated at 261.64×10^6 t C $year^{-1}$, including forests (243.16×10^6 t C $year^{-1}$).

Carbon accumulation in the RFF is possible, even on a large scale, by creating young stands on large areas in the European

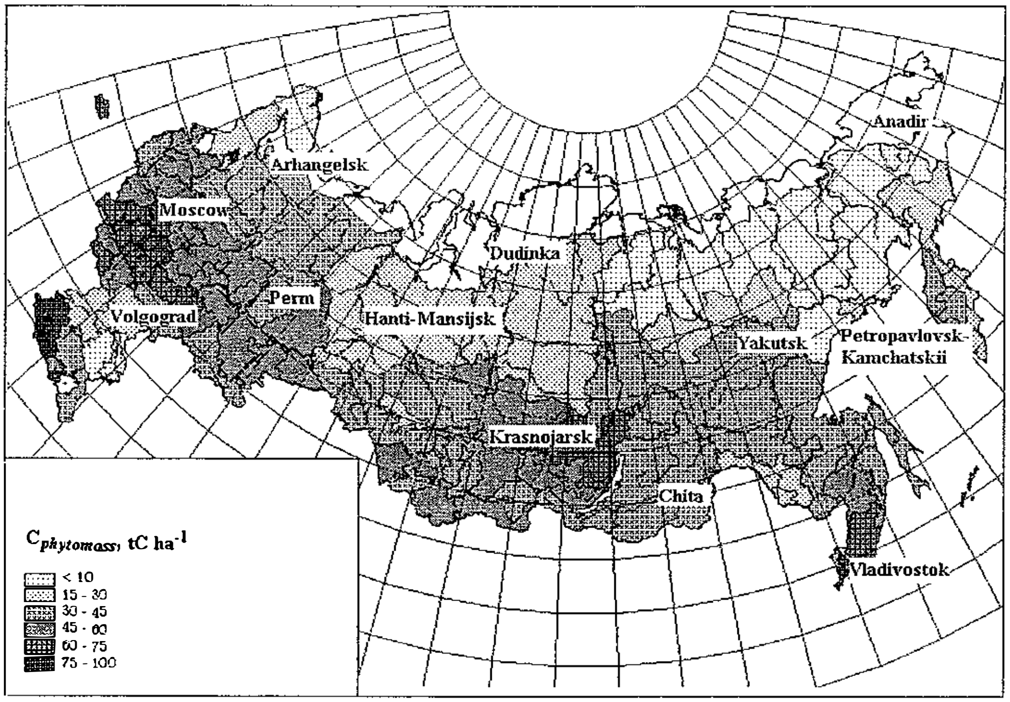

Figure 1. Geographic distribution of $C_{phytomass}$ density within the RFF of Russia.

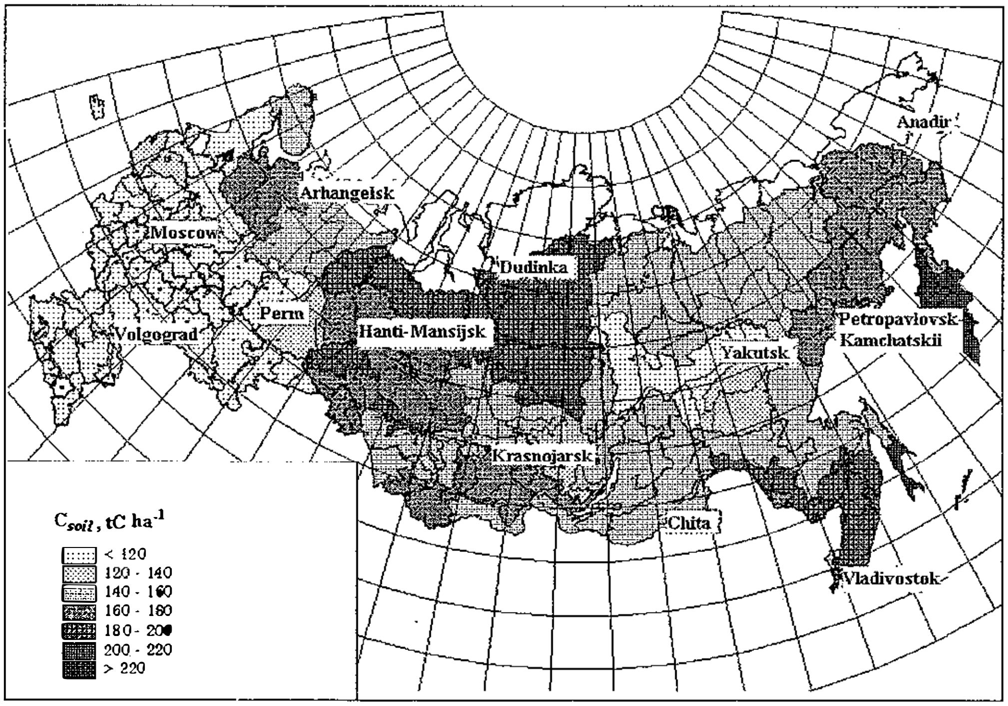

Figure 2. Geographic distribution of C_{soil} density within the RFF of Russia.

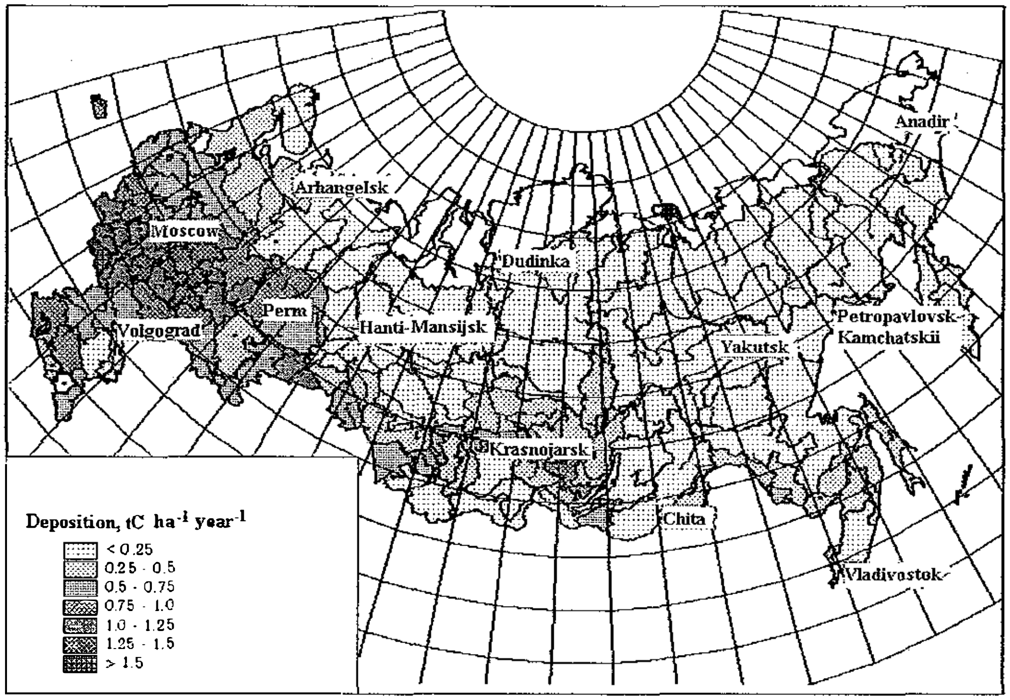

Figure 3. Geographic distribution of annual accumulation of $C_{phytomass}$ density within the RFF of Russia.

part of Russia and in the south of Siberia.

Rejuvenation of the forest fund in the northern and middle subzones of East Siberia and the Far East would be useful for the same purposes. However, large-scale reconstruction of the age structure of stands in these regions is not possible at present due to economic problems.

DISCUSSION

The multistep procedure of calculating carbon reserves in forests managed by the RFF (1110.5×10^6 ha) yielded estimates of 34.35×10^9 t C for $C_{phytomass}$, 172.43×10^9 t C for C_{soil}, and 206.78×10^9 t C for C_{total}.

The absence of data accounting for dead standing and windthrown trees prevents accurate estimation of the coarse woody debris carbon pool (C_{CWD}). A tentative calculation of C_{CWD} was made to approximate actual conditions. The following assumptions were made in doing so:

1. reserves of CWD in forests and on previously forested areas (clearcuts, burns) are equal to 30% of available stand reserves;
2. one third of C reserves in CWD are accounted for by dead-standing trees having a wood density equal to 70% of the standard value for living tree species;
3. the wood density of windthrown trees (2/3 of the total CWD) is averaged at 50% of the standard one;
4. 90% of CWD occurs in forests, 10% occurs on clearcuts and burns.

According to our calculations, the pool of C_{CWD} in the RFF is 3.6×10^9 t C; of which 90% or 3.15×10^9 t C is CWD in forests. Carbon of dead branches in living crowns also needs to be included. Their phytomass is assumed to equal about 10% of the total phytomass of branches and shoots of stands (3.0×10^9 t C) which in turn corresponds to 15% of stem phytomass. Therefore the carbon reserve in dead branches of all stands in the RFF is equal to about 0.2×10^9 t C.

Once estimates for CWD are included, the total reserve of organic carbon in forests managed by the FFS of the RF is 209.95×10^9 tC, or approximately 210×10^9 t C. The estimate for the carbon reserve for all the forests in all of Russia is slightly higher. After applying the correction coefficients of 1.08 the estimate for the total carbon reserve (C_{total}) increases to 223.54×10^9 t C , which includes 36.92×10^9 t C for $C_{phytomass}$, 182.82×10^9 t C for C_{soil}, and 3.80×10^9 t C for C_{CWD}. C_{total} is distributed between 150.15×10^9 t C in forests, 17.18×10^9 t C in non-forested lands, and 56.2×10^9 t C in non-forest lands.

The total estimate of $C_{phytomass}$ is slightly different from the ones we have published earlier (Isaev et al., 1993, 1995a, 1995b). This difference is due to the justification of the conversion coefficients C/S, more accurate evaluation of $C_{phytomass}$ in dwarf pine phytomass (Utkin and Pryaznikov, 1999), and RFF data extrapolation over the entire area of Russian forests. Comparison of our estimates for carbon pools on RFF lands with estimates of other researchers is difficult because of varying definitions for the carbon pools. For example, some authors combine C_{CWD} and carbon into forest litter which is defined as "mortmass", others include forest litter in the C_{soil} pool. Often calculations are made for forested land only, but not for all land categories of the RFF.

Data calculated by separate biomes are also of little use for comparison. Biome areas are defined by "recovered vegetation" and do not necessarily reflect the current regional structure of lands (Kolchugina and Vinson, 1993a, 1993b).

Alexeyev and Birdsey (1994, 1998) estimated $C_{phytomass}$ of Russia as a total land forest area and for forests in the RFF at only 29.53×10^9 and 27.98×10^9 t C, respectively. Nilsson and Shvidenko (1998) estimated $C_{phytomass}$ for the forested land area of RFF on the basis of state forest inventories in 1961-1993. They estimated the area of forested land in 1993 to be 763.5×10^6 ha (lower compared to statistical information, because of forest fires). Their estimates of $C_{phytomass}$ were 32.09×10^9 t C in 1993 and as 32.52×10^9 t C in 1988.

Some authors estimates of $C_{phytomass}$ for the forested land area of RFF are similar and range from 28.5×10^9 to 35×10^9 t C (Alexeyev and Birdsey, 1994, 1998; Nilsson and Shvidenko, 1998; the present work). However, our estimates of $C_{phytomass}$ reflect differences in areas and in values for C/S ratios.

Values for conversion coefficients C/S used in this study are higher than those used by Alexeyev and Birdsey (1994, 1998) and lower than those used by Nilsson and Shvidenko (1998). A very high coefficient of conversion for stand stock (0.53) is given by Kolchugina and Vinson (1993a) for all tree species. Also cited works did not apply a correction of the primary information on forests managed by FFS of RF to the total area of forests of Russia.

Published estimates of C_{soil} are highly variable as well. Kolchugina and Vinson (1993a) estimated C_{soil} of the former USSR at 320×10^9 t C, including forest litter at 338×10^9 t C, based on the areas of biomes. After converting for the RFF these estimates decreased to 314×10^9 t C and 321×10^9 t C, respectively. Alexeyev and Birdsey (1994, 1998), estimated C_{soil}, without forest litter, for the total forest land area at 140.3×10^9 t C (including 54×10^9 t C in peat on open bogs) and 74.16×10^9 t C for the forested land area. Others have estimated C_{soil} at 129.6×10^9 t C to 1m depth of soil in the forested area of the RFF, and 8.72×10^9 t C in forest litter for a total of 138.32×10^9 t C (Nilsson and Shvidenko, 1998). Orlov et al. (1996) estimated C_{soil} of forest soils in Russia, including areas partly utilized by agriculture, at 235.0×10^9 t C on an area of 1383.3×10^6 ha. Rozhkov et al. (1997) estimated the same parameters at 266.7×10^9 t C and 1316.5×10^6ha, respectively.

Estimates of C_{soil} are given in the present work at 172.44×10^9 t C for the total area of the RFF and at 67.2×10^9 t C for the forested area. These estimates are not consistent with estimates from other authors, with both overestimation and underestimation. The scatter of estimates for C_{soil} is related both to variability of initial information on soil carbon density and to contradictory approaches to calculation of C_{soil} in bogs. We think that uncertainty associated with estimates related to bogs is the most influential factor affecting variability in C_{soil} estimates.

Bogs occupy an area of 125.2×10^6 ha within the RFF. They are present on other lands in forest zones (agricultural lands, nature reserves, etc.). If we use a value of 224 t C ha^{-1} as a realistic estimate of carbon density for bogs, the capacity of carbon pool of peats is equal to 28.04×10^9 t C. Alexeyev and Birdsey (1994, 1998) estimated bog carbon at 54×10^9 t C which means, in this case, that the averaged carbon density of peat would be much higher at 431 t C ha^{-1}.

Vompersky et al. (1996) estimated the areas and carbon reserves of peats, using a soil map at a scale of 1:2 500 000. The area of peatlands, both forested and non-forest, in Russia is equal to 369.1×10^6 ha. The area of peatlands is 262.9×10^6 ha in taiga and other forest subzones (i.e. two times greater than in the RFF). Total carbon reserves in bogs with peat layers $\leq$ 30 cm and > 30 cm were estimated by Vompersky et al.(1996, 1999) at 12.6×10^9 and 100.93×10^9 t C, respectively, with average values of 55 and 726 t C ha^{-1}, respectively.

The range of averaged carbon reserves in peat bogs of the RF varies from 224 to 726 t C ha^{-1} and is certain to influence the total estimate of C_{soil} for the RFF since about 12% of lands within the forest fund are bogs. Obviously much effort is required to estimate C_{soil}, with some degree of confidence, for Russian forests as a whole and for their bogs in particular.

The information on $C_{phytomass}$ available for forests and lands within the RFF could be considered as sufficient and rather correct when combined with information from the state forest inventories. The same can not be said about data for C_{soil}. Therefore, calculating carbon macro-fluxes and forecasting changes in the carbon cycle should emphasize the dynamics of $C_{phytomass}$ until such time as C_{soil} estimates can be improved. We recognize that sequestration of CO_2-C in C_{soil} in Russian forests is as important as NPP and carbon accumulated in $C_{phytomass}$.

Data on carbon fluxes in forests of the RFF are scarce, presented by different characteristics, and fall in the category of expert estimates that often ignore carbon sequestration in C_{soil}.

Kudeyarov and Kurganova (1998) estimated the total annual emission of CO_2 for soils in Russia at 4.50×10^9 t C, and Bazilevich (1993) estimated NPP at 4.81×10^9 t C $year^{-1}$ suggesting a steady state between emission loses of CO_2 and carbon sink. Kolchugina and Vinson (1993b, 1993c) estimated NEP for forests of the USSR at 825×10^6 t C $year^{-1}$ and again at 662×10^6 t C $year^{-1}$ (Kolchugina and Vinson, 1995). Carbon sequestration in forests of the former USSR has been estimated at 416×10^6 t C $year^{-1}$ (Sedjo, 1992) and at 660×10^9 t C $year^{-1}$ (Kolchugina and Vinson, 1995).

In accordance with our data annual carbon accumulation is equal to 262×10^6 t C $year^{-1}$ including 243×10^6 t C $year^{-1}$ in forests (Isaev et al., 1995a, 1995b). It remains uncertain if carbon accumulation corresponds to NEP or NPP. Our observations on permanent plots in pine and birch stands over 30 years (from age 30 to 59 years) indicate that NPP is three times and NEP is two times, greater than accumulation. Using data from the "Bioproductivity of Forest Ecosystem: Computer Database" (Utkin et al., 1994) coefficients relating NPP to stand stock were calculated for the main forest-forming tree species (Zamolodchikov and Utkin, 2000). Combining these data with information from the state forest inventories, NPP of the main forest-forming tree species (without shrubs) was estimated at $1910.5\times10^6 \pm 386.5\times10^6$ t C $year^{-1}$ for aboveground phytomass and at $2267.6\times10^6 \pm 487.6\times10^6$ t C $year^{-1}$ for aboveground and belowground phytomass together.

Thus, we consider that the estimates for $C_{phytomass}$ in Russian forests are studied satisfactorily and in sufficient detail. But we cannot say the same about C_{soil}. Nevertheless, available results and patterns of estimating $C_{phytomass}$ with the use of information from state forest inventories is a useful approach to estimating carbon macro-fluxes including NPP, NEP, and NBP. Undoubtedly more reliable information on carbon cycling in boreal forests will become available in the immediate future. At that time, we will be better able to determine the magnitude of the importance of Russian forests in the C cycle of the biosphere. The strategy of forest management in Russia must be determined in light of these studies and their results.

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ROLE OF SOIL ORGANIC MATTER IN THE CARBON CYCLE IN FOREST ECOSYSTEMS IN THE KRASNOYARSK REGION

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ABSTRACT

In forest ecosystems, organic matter is mineralized to inorganic carbon (CO_2) when carbon is released during decomposition of the light-hydrolyzed and resistant to biodegradation fractions of soil organic matter. Light-hydrolyzed organic matter is the main source of carbon mineralized into the atmosphere and for the synthesis of humus substances. In the Krasnoyarsk region the accumulation of carbon in the soil cover of forest ecosystems is estimated at 9493.3 million tonnes. This includes 5662.9 million tonnes (60.8 t ha^{-1}) of carbon storage in the resistant fraction of humus in the mineral horizons to a depth of 0-50 cm, and 3830.4 million tonnes in the light-hydrolyzed organic matter. Soluble forms of light-hydrolyzed organic matter make up 2719.2 million tonnes (29.2 t. ha^{-1}) and the above- and below-ground dead organic matter make up 1111.2 million t (11.9 t ha^{-1}). Most of the stored carbon is accumulated in the top mineral soil (0-20 cm in depth). Carbon of the light-hydrolyzed fraction of soil organic matter makes up 53% of the total carbon storage within the upper soil layer. Newly formed humus accounts for 3 to 6% of the organic matter stored in decomposed residues or 0.2 to 0.6% of the carbon stored in the top 50 cm of forest soil. The total mineralized carbon flux is 75.8 million tonnes C yr^{-1}, which includes 59.6 million tonnes C yr^{-1} in the Krasnoyarsk region within the borders of Middle Siberia.

INTRODUCTION

Soil is an essential component of natural ecosystems. It is related to other components of a given ecosystem as well as to the components of surrounding ecosystems through fluxes. Interaction of these fluxes determines the form and function of the ecosystem. The system of fluxes, which form the carbon cycle in the ecosystem, can be classified into two main groups: the production of organic matter and its destruction (decomposition).

The decomposition segment of the cycle returns carbon to the atmosphere (heterotrophic respiration, mineralized carbon flux) which has been taken from the atmosphere through the process of photosynthesis. Mineralization of carbon occurs in the soil component of the ecosystem. Soil organic matter (SOM) is heterogeneous in terms of genesis and structure. Such diversity confers different susceptibilities of the SOM to biodegradation. Forest litter is an integral part of the accumulated humus form in forest ecosystems. It accumulates all the dead and fallen phytomass, including tree stems. The upper part of the litter is refreshed annually, whereas the lower part of the litter is transformed into the fermentation (F) and humification (H) subhorizons.

In forest ecosystems, the organic matter is mineralized to inorganic carbon through carbon release during the decomposition of light-hydrolyzed (originating from above- and below- ground dead plant and animal residues, microbial biomass and soluble organic components) and resistant to biodegradation (soil humus) fractions of the SOM. Light-hydrolyzed organic matter is the main source of the mineralized carbon flow into the atmosphere and for the synthesis of humus substances.

The aim of this study was to assess the carbon storage and rate of C_{CO2} released from the SOM fractions during its decomposition in forest ecosystems of the Krasnoyarsk region.

MATERIALS AND METHODS

Tasks of the study involved the quantitative estimation of the following parameters:

- soluble (C_{H2O} + $C_{0.1nNaOH}$) and resistant humus (TC-soluble C) storage in forest soils (0 – 50 cm);
- storage and composition of above- and below-ground dead organic matter in larch (*Larix sibirica and L. gmelinii*), cedar (*Pinus sibirica*), pine (*Pinus sylvestris*), spruce (*Picea obovata*), fir (*Abies sibirica*), aspen (*Populus tremula*) and birch (*Betula pendula*) stands;
- carbon concentration in above- and below-ground dead organic matter;
- mineralization rates (kg C . ha^{-1} yr^{-1}) of the main components of above- and below-ground dead organic matter .

The C values for aboveground dead organic matter (dead and fallen phytomass including stems and the LFH) calculated in this work were based on previously described data (Stakanov, 1978; 1990; Korotkov, 1998; Korsunov et al., 1988; Shugaley et al., 1998; Vedrova and Mindeeva, 1998; Vedrova et al., 2000). In addition, the potential accumulation of plant material on the soil surface was calculated based on annual die-off rate data for each phytomass fraction and it's specific rate of decomposition. The stem fall was determined using the tables of forest stand growth for Siberia, and the litterfall data were collected experimentally (Kulagina, 1978; Soil factors ..., 1976; Vedrova, 1980).

Samples for belowground dead organic matter were collected from the soil columns (0-20 cm and 20-40 cm increments, n = 5-20), were washed through a sieve (d> 0.25 mm) and sorted into live roots, dead roots and "other mortmass"(collectively referred to as "root material"). In addition, the store of dead plant residues in the soil was determined, assuming that the ratio of the litterfall and decomposition rates at biochemical transformation of plant residues on soil surface and in soil thickness are the same.

Rates of decomposition for above- and belowground plant residues were represented by the value of the decay constant (k), which was calculated according to results of numerous field experiments as;

$$k = (\ln Ct / Co) / t, \qquad [1]$$

where;

Co = stores of organic matter ($kg.ha^{-1}$) at the beginning of the experiment;
Ct = stores of organic matter ($kg.ha^{-1}$) at the end of the experiment;
t = duration of experiment (yr.) and,
k = a constant of decomposition (yr^{-1}).

The calculation was used to describe decomposition of litter from trees only, and does not take into account residues from ground vegetation. These calculations are in agreement with results of field data on the litter stores of forest stands with or without grass cover. Lichen and green moss forest types prevail in coniferous stands of the northern latitudes. The litter store calculations in larch stands showed that the tree leaffall fraction made up only 20-25% of the litter store and the rest was made up of semi-decomposed moss and (or) lichen residues. In the taiga pine stands 40-60% of tree leaffall litter was morphologically distinguishable. For these reasons, the calculated values of litter store, for larch and taiga pine stands, were corrected to take into account the type of ground vegetation.

The rate of the new humus formation is represented by the value of the humic acid carbon extracted from plant residues by a 0.1N NaOH solution. The characteristics obtained for conifer stands are comparable with the indices of humification specific rate that have been determined in field experiments. Earthworms take an active part in deciduous litter decomposition. Carbon concentrations in their coprolites were 3-4 times higher than in the soil substrate. The experimental results showed that 30-60% of the newly synthesized humus is accumulated due to coprolites in decomposing aspen and birch litter (Vedrova, 1998).

RESULTS AND DISCUSSION

The percent coverage of the main forest forming species in the forested area of the Krasnoyarsk region are as follows: larch (*Larix sibirica* and *L. gmelinii*) -57%, pine (*Pinus sylvestris*) - 9%, cedar (*Pinus sibirica*) - 7%, fir (*Abies sibirica*) - 7%, spruce (*Picea obovata*) - 6%, birch (*Betula pendula*) – 12% and, aspen (*Populus tremula*) – 2 %.

9493.3 million tonnes of carbon is present in the soil cover of the forest ecosystems. This includes 5662.9 million tonnes (60.8 t ha^{-1}) of carbon storage in the resistant fraction of humus in the mineral horizons (0-50 cm), 2719.2 million tonnes (29.2 t. ha^{-1}) in the soluble fraction of the SOM, and 1111. 2 million tonnes (11. 9 t ha^{-1}) in the above- and below-ground dead plant residues (Table 1). Most of the

Table 1. Carbon store in forest soils of Krasnoyarsk region.

	Above- and below ground dead organic matter					Soil humus			
						0 – 20 cm		0 – 50 cm	
Zone (subzone)	leaf-fall	litter	stem wood	dead root material	total	Csol*	Cresist**	Csol	Cresist
Pre-tundra									
mln.t	3.25	109.06	26.41	27.71	166.43	223.38	609.60	300.15	787.35
t.ha^{-1}	0.18	6.20	1.50	1.58	9.47	13.22	34.68	17.08	44.80
Northern taiga									
mln.t	6.33	226.26	40.33	59.22	332.14	276.16	724.42	366.78	935.86
t.ha^{-1}	0.34	11.98	2.14	3.14	17.59	14.63	38.37	19.43	49.57
Middle taiga									
mln.t	13.17	179.06	106.86	55.07	354.16	722.81	1093.29	1125.04	1802.24
t.ha^{-1}	0.43	5.83	3.48	1.79	11.53	23.53	35.59	36.62	58.66
Southern taiga									
mln.t	8.41	55.45	75.96	19.13	158.95	341.10	590.87	537.60	1081.43
t.ha^{-1}	0.64	4.20	5.75	1.45	12.03	25.82	44.73	40.70	81.86
Forest-steppe									
mln.t	2.72	13.96	8.37	6.03	31.08	66.43	333.77	106.00	591.69
t.ha^{-1}	0.62	3.20	1.92	1.38	7.12	15.21	76.43	24.27	135.49
Mountain of Siberian									
mln.t	5.10	24.49	27.05	11.78	68.42	133.50	294.38	283.63	464.36
t.ha^{-1}	0.60	2.91	3.21	1.40	8.12	15.84	34.93	33.66	55.10
Total	38.97	608.28	284.98	178.94	1111.18	1772.38	3646.33	2719.20	5662.93

* - carbon of soluble soil organic matter fraction

** - carbon of resistant soil humus

carbon is stored in the upper layer (0-20 cm in depth) of the soil. Carbon of the light-hydrolyzed fraction of soil organic matter makes up 53% of the total carbon storage within the upper soil layer.

Carbon accumulation in above- and below-ground dead organic matter (t ha^{-1}) is greatest in larch stands, followed by cedar, pine, spruce, fir, aspen and birch stands (Figure 1). Of the coniferous stands, carbon accumulation is highest in the northern and southern taiga. In deciduous stands, carbon accumulation increases slightly from north to south.

Maximum stores of dead organic material are usually found in tree stands in age classes II and III. The accumulation in conifer pre-tundra and northern taiga ecosystems is observed up to age classes IV and V. Biomass estimates for above- and belowground dead organic matter up to age class V are highest for litter, followed by root material and then stem wood.

Like litter, the "dead root" component of the "root material" is heterogeneous in composition, carbon concentration and degree of decomposition. The magnitude of carbon stored in the dead root component is often comparable with that for litter. As reported before, the store of the dead root material under the 25-aged spruce stand is about 19.3 t ha^{-1} (Orlov, 1966). Dead roots (d<0.5 mm) and other organic residues comprise 39% and 54% of this store, respectively. The remaining 7.1% is comprised of mushroom mycelia (7.1 from 10.3 $t.ha^{-1}$).

In the 25-year old stands of the main forest-forming species of Siberia the store of dead root material under conifers ranges from 18.5 to 35.6 $t.ha^{-1}$. Under deciduous leafy species it ranges from 13.4 to 18.4 $t.ha^{-1}$ and exceeds the litter store under these species by 1.5-2 times. Carbon concentration in the underground mortmass (23.3-27.3%) for deciduous forests is almost two time lower than in the litter (37.3- 47.8%). Therefore the carbon store in decomposed plant residues on the surface and in the 20-cm soil layer are equivalent. Dead roots make up only 20-31% of the total root material store in conifer stands and 13-17% in the leafy stands. The fraction called "other mortmass" prevails (up to 80% of root material store). In the conifer stands mushroom mycelium prevails, whereas in the leafy stands it consists of the mixture of chaffy scales of bark, root coverings, and heavily decomposed roots. Decomposition rate of the dead root in soil is lower than on surface (Table 2). The "other mortmass" fraction decomposes slower than many of the other fractions, which accounts for the accumulation of mortmass in the soil (Vedrova, 1996; 1998; Vedrova et al., 2000).

Under mature pre-tundra and northern taiga larch forests, the "other mortmass" fraction also prevails in the underground root

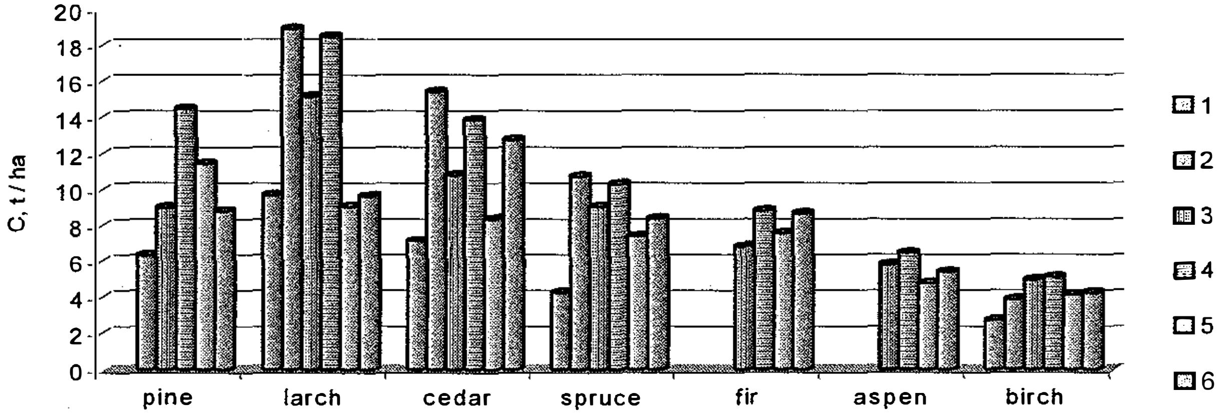

Figure 1. Above- and belowground dead OM, C (t / ha). 1. pre-tundra forests; taiga: 2. northern, 3. middle, 4. southern; 5. forest-steppe; 6. mountains

material. The mushroom mycelium mass is much less than in 25-year old larch stands. The total store of root material in the 20-cm soil layer in pre-tundra and northern taiga (29.2-34.9 and 12.5-21.4 t.ha^{-1}, respectively) is much less than that in the litter store (42.0-78.1 and 36.7-45.1 t.ha^{-1}, respectively). Taking into account the observed difference in the carbon concentration between litter and root material components, as well as the fact that 70-80% of litter store are made up by the slowly decomposed moss (or lichen) residues, it can be concluded that the dead plant residues store on the soil surface and in the soil layer formed by products of tree leaffall (tree falling away) are comparable.

Coniferous stands store three to four times more biomass in the above- and below-ground woody debris than deciduous stands due to a faster rate of decomposition of deciduous materials (Table 2).

This larger storage capacity results in a greater carbon dioxide (CO_2) output, when decomposition occurs. However, it is not the amount of stored carbon, but the biochemical structure of the material being decomposed, which determines the mineralization rate under any given combination of hydrothermic indices. Therefore, at a lower woody debris stores, the amount of carbon released to the atmosphere is as high as in the aspen, birch, and cedar (*Pinus sibirica*) stands (Table 3).

Most of the carbon lost upon decomposition goes back to the atmosphere in the form of carbon dioxide. The remaining part goes to the soil, taking part in the humus profile formation. The newly formed humus is about 8 - 26% of the carbon lost to decomposition. This is equivalent to 3 - 6% of the organic matter stored in decomposed residues or to 0.2 - 0.6% of the carbon stored in the top 50-cm layer of the forest soil. In the soils of mature stands, the young humus compounds renew the peripheral part of the soil humus molecules and, therefore, any additional carbon accumulation in the soil is not measurable. In this case, the carbon loss values at decomposing correspond to total mineralization of the organic matter of plant residues and soil humus. In young stands, newly formed biologically resistant compounds of a specific nature are fixed in soil, not only renewing part of the molecule of the soil-humus acids but also enlarging the humus store in the soil.

The total mineralized carbon flux in the Krasnoyarsk region is 75.8 million t C yr^{-1}, including 59.6 million t C yr^{-1} for the area within the borders of Middle Siberia. Larch stands accounted for more than 90% of the total CO_2 released to the atmosphere in the pre-tundra and northern taiga forests, and almost 50% of CO_2 released in middle taiga forests (Figure 2). Birch stands contributed 17% of the total carbon flux to the atmosphere, which increases along a latitudinal gradient to the south.

Table 2. Decomposition rate (k yr^{-1}).

Fraction	Larch	Pine	Cedar	Fir	Spruce	Aspen	Birch
Litterfall:							
foliage	0.14 - 0.22*	0.14 - 0.23	0.15 - 0.23	0.20 - 0.34	0.17 - 0.29	0.30 - 0.38	0.31- 0.45
branches	0.06 - 0.09	0.08 - 0.11	0.08 - 0.10	0.10 - 0.14	0.11 - 0.14	0.10 - 0.18	0.11- 0.20
bark	0.02	0.03	0.03 - 0.04	0.06 - 0.08	0.05 - 0.09	0.11- 0.13	0.05- 0.09
cone	0.07 - 0.08	0.05 - 0.06	0.05 - 0.06	0.09 - 0.10	0.06 - 0.09	-	-
Forest litter:							
L + F	0.12 - 0.20	0.08 - 0.17	0.08 - 0.20	0.16 - 0.27	0.15 - 0.25	0.17 - 0.25	0.16 - 0.25
H	0.02 - 0.04	0.03 - 0.06					
Root material:							
dead root	0.03 - 0.12	0.03 - 0.06	0.04 - 0.10	0.06 – 0.10	0.06 - 011	0.05 - 0.25	0.09 - 0.14
"other mortmass"	0.03 - 0.09	0.03	0.04 - 0.06	0.06	0.06	0.05	0.09 - 0.12

* - the interval of change based on the hydrothermal conditions of stand growth and its age

Table 3. Intensity of C-CO_2 release during decomposition of the above- and below-ground organic matter (C, t ha^{-1} yr^{-1}).

Stands	Pre-tundra	Boreal forest			Forest-steppe	Mountains
		northern	middle	southern		Siberian south
Pinus sylvestris	--	0.32	0.50	0.76	0.83	0.59
Pinus sibirica	0.39	0.90	0.84	1.07	0.74	0.82
Larix sibirica, L.gmelinii	0.34	0.76	0.65	0.70	0.64	0.58
Picea obovata	0.25	0.59	0.54	0.72	0.67	0.62
Abies sibirica	--	--	0.44	0.85	0.77	0.87
Populus tremula	--	--	0.78	0.97	0.75	0.84
Betula pendula	0.42	0.58	0.65	0.93	0.84	0.84

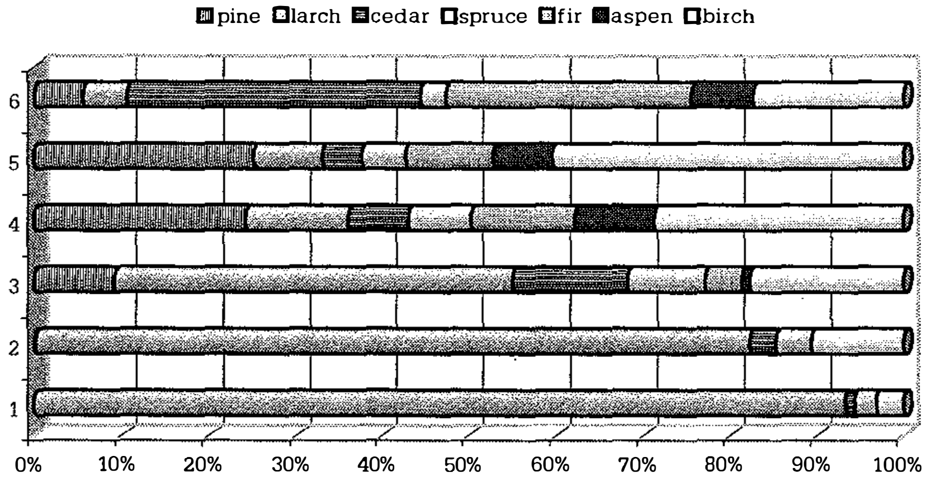

Figure 2. Participation of the stands in the formation of carbon mineralization flow (%). 1. pre-tundra forests; taiga: 2. northern, 3. middle, 4. southern; 5. forest-steppe; 6. mountains

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AN ASSESSMENT OF THE AMOUNT OF BIOMASS CONSUMED FROM WILDLAND FIRES BASED ON VEGETATION FUEL MAPS

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ABSTRACT

Carbon emission from wildland fires is the result of biomass (organic substances) combustion. The amount of biomass consumed in a fire depends upon;

1. the distribution of the biomass stock amongst vegetation fuel classification categories,
2. the period of the fire season in which the fire occurs,
3. the severity of the drought and,
4. the fire development as controlled by wind, topography and air humidity.

A method is proposed to make large-scale vegetation fuel maps based on the classification of vegetation fuels. The dynamics of the biomass consumed, depending on seasonal and weather factors, was investigated for different categories of the vegetation fuel classification and surface fires in boreal forests of Russia.

INTRODUCTION

The amount of carbon per unit of area that is emitted from boreal forests wildland fires is determined by the amount of the biomass consumed by the fire. Biomass consumption depends on:

1. the distribution of the biomass stock amongst vegetation fuel classification categories,
2. the period of the fire season when the fire occurs, and the severity of the drought and,
3. fire development as controlled by wind, topography and air humidity.

We estimated biomass burned using the example of forests in Russia where moss and lichen forest floor types predominate.

Large-scale vegetation fuel (VF) maps are desirable to evaluate the biomass stock distribution on every plot by vegetation fuel classification categories. These maps must characterize, first, the typical VF complexes which form the biogeocoenoses, and second, the factors which control the moistening, drying and burning conditions. We started to develop methods to produce VF maps in 1984. The basis for this work was a system of VF classification.

CLASSIFICATION OF VEGETATION FUELS (VF)

In Russia the VF are classified by their contributions to fire incidence, spread, and behaviour (Table 1) (Kurbatsky, 1962, 1970; Volokitina, 1996; Volokitina and Sofronov, 1996).

The first group of VF are the prime conductors of burning (PCB) which play the leading role in fire incidence and spread. The main principles of PCB classification are shown in Table 2.

FIRE DANGER OF VEGETATED AREAS

A detailed investigation of the critical class of drought for plots (including the plots with atypical conditions) was based on (Table 3);

1. the distribution of rainfall on the forest floor,
2. the relative quantity of solar radiation absorbed by the forest floor and,
3. the possibility of burning of the PCB (Sofronov and Volokitina, 1990; Volokitina, 1995).

VEGETATION FUEL MAPS

Large-scale VF maps were based on forest survey data (Volokitina, 1988). The base maps for this work are the uncolored plans of forest stands. The maps aresupplemented with detailed pyrological descriptions of each forest inventory plot (Figure 1, Table 4).

Table 1. Classification of vegetation fuels (VF).

VF group	VF subgroup	VF type (and subtype)	Character of burning*
I. Cover of moss or lichen, fine litter - the (Prime Conductors of Burning), PCB-fine fuels**	Mosses	Lichen (Lc)	Fl
		Dry moss (Dm)	Fl
		Moist moss (Mm)	Fl and Sm
		Bog moss (Bm)	
		subtype Bm (1)	Fl
		subtype Bm (2)	Wn
	Litter	Cured grass (Cg)	Fl
		Loose litter (Ll)	Fl
		Compact litter (Cl)	Fl and Sm
		«Non- conductor» (Nc)	
		subtype Nc (1)	Sm
		subtype Nc (2)	Wn
II. Duff, humus and peat layers of soil	Duff, Peat and humus	Rough humus	Sm
		Mull	Sm
		Turf	Sm
		Peat horizon	Sm
III. Layer of herbs and low brush (at coverage ratio 0.5 m or more)	Low brush	Vaccinium vitis-idaea (Vv)	Fl
		Arctostaphylos uva-ursi (Au)	Pd
		Swamp-shrub (Bs)	Fl and Pd
		And other types	
	Herb (green)	Grass (Gr)	Pd
		Sedge (Se)	Pd
		«Winter» sedge (Ss)	Fl
		Mixed herb (Mh)	Pd
		And other types	
IV. Large wood remnants (dead branches, snags, limbwood, slash)	Dead-standing and downed trees	Dead-standing trees	Sc, Sm
		Hanging limbwood	Sc
		Downed limbwood	Sc
	Slash	Coniferous foliage-covered slash	Fl
		Foliageless slash	Pd
V. Understory of advance tree growth and shrubs	Coniferous	not classified yet	Fl
	Broad-leaved		Pd and Fl

Table 1. (cont'd) Classification of vegetation fuels (VF).

VF group	VF subgroup	VF type (and subtype)	Character of burning*
VI. Green foliage, foliage-covered twig, and dead limbs of live trees.	Coniferous	Crowns of young tree stands and dwarf Siberian pine thickets	Fl
		Crowns of spruce and fir stands	Fl
		Crowns of pine and larch stands	Fl
	Broad-leaved	Crowns of broadleaved stands	Pd
VII. Trunks of live trees, branches thicker than 7 mm		Healthy trunks	Sc
		Resinous trunks	Fl,Sc
		Rotten or hollow trunks	Sc,Sm

Notes: *Character of burning: Fl - burning with flame, Sm - smouldering, Sc - surface charring, Pd – passive thermal decomposition, Wn - would not burn. **The first group of VF (PCB) - plays the leading role in fire incidence and spread.

Table 2 shows the main principles of PCB classification.

Table 2. Types of the prime conductors of burning (PCB).

[1]Subgroup moss	Lc	Dm	Mm	Bm_1	Bm_2
[1]Subgroup litter	Cg ↔ Ll ↔ Cl ↔ Nc_1 Nc_2				
[2]Critical DC:	I	II	III	IV	can't burn

[1]Subgroup abbreviations are described in Table 1.

[2]Critical DC is the drought class at which the PCB fuel class becomes flammable

Table 3. Estimation of the natural fire danger of vegetated areas.

PCB subgroups		Prevailing tree species				Treeless
moss	litter	pine, spruce, Siberian pine, larch, birch, aspen, fir - with foliage			larch, birch, aspen - without foliage	areas without a thick layer of shrubs
Types and subtypes of prime conductors of burning (PCB)		Relative stocking coefficient (including the second storey)				
		≥0.8	0.5 - 0.7[1]	≤0.4[2]	any density	
		Critical classes of drought				
Lc	Cg	II	I	I	I	I
Dm	Ll	III	II	I	I	I
Mm	Cl	IV	III	III	III	III
Bm_1	Nc_1	V	IV (Nc_1-V)	IV (Nc_1-V)	IV (Nc_1-V)	IV (Nc_1-V)
Bm_2	Nc_2	Would not burn				

[1]for larch stands of the Northern open forest zone ≥0.6

[2]for larch stands of the Northern open forest zone ≤0.5

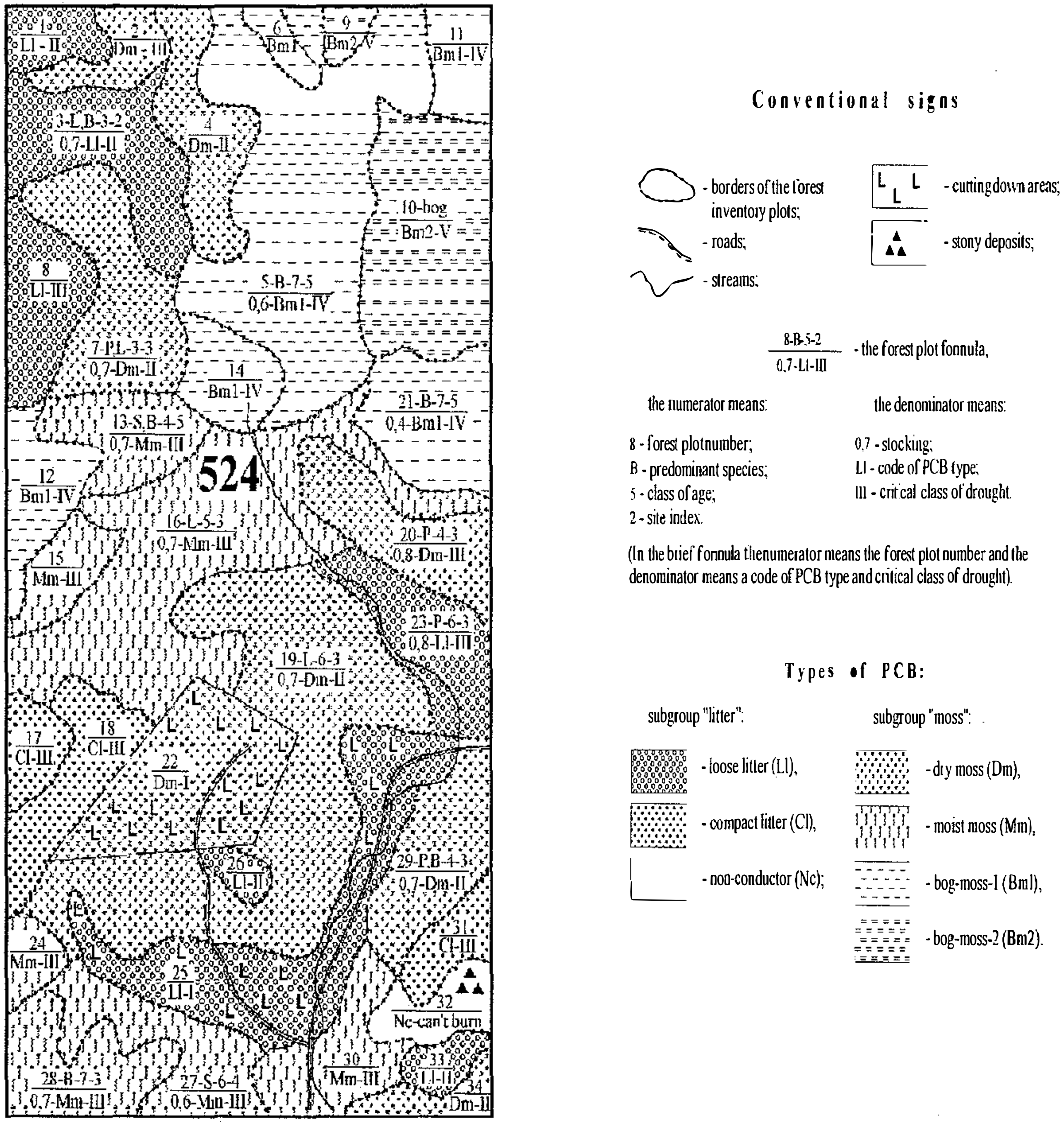

Figure 1. A fragment of the vegetation fuel map (spring and autumn). Scale 1:25 000.

Table 4. Example of data in pyrological descriptions.

Forest Division	7	7	7	7	7	7	7
Forest Inventory Unit	524	524	524	524	524	524	524
Plot Number	1	2	3	4	5	6	7
Area (ha^2)	7.0	9.0	27.0	23.0	83.0	5.0	28.0
Exposure	0	0	0	0	0	0	0
Degree	0	0	0	0	0	0	0
Site Index	4	3	2	3	5	5	3
Fallen Wood	0	0	0	0	0	0	0
Dead Tree Stock	0	0	0	0	0	0	0
Canopy Closure							
spring, autumn	2	3	2	2	4	4	2
summer	2	3	3	3	4	4	3
Type of PCB							
spring, autumn	Ll	Dm	Ll	Dm	Bm1	Bm1	Dm
summer	Ll	Dm	C1	Dm	Bm1	Bm1	Dm
Shading							
spring, autumn	0.7	0.8	0.6	0.6	0.6	0.4	0.6
summer	0.7	0.8	0.8	0.7	0.7	0.5	0.7
Density (understory)							
Saplings	0.0	0.0	0.0	0.0	0.3	0.0	0.5
Woody vegetation	0.1	0.3	0.0	0.1	0.2	0.0	0.1
Understory Saplings							
Evergreen	0	0	0	0	10	0	10
Deciduous	0	0	0	0	0	0	0
Height (m)	0	0	0	0	1	0	5
Canopy 1							
Relative Stocking	0.7	0.8	0.7	0.7	0.6	0.4	0.7
Height (m)	2	16	17	15	12	7	16
Age (years)	15	60	55	55	65	40	60
Evergreen	10	10	1	6	1	0	6
Deciduous	0	0	9	4	9	10	4
Canopy 2							
Relative Stocking	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Height (m)	0	0	0	0	0	0	0
Age (years)	0	0	0	0	0	0	0
Evergreen	0	0	0	0	0	0	0
Deciduous	0	0	0	0	0	0	0

The pyrological description of a forest inventory plot (Table 4) includes the information which makes it possible to judge the probability of burning during different periods of the fire season and at different levels of drought, the probable character of burning, its rate of spread, and the probable impact of the fire. In practice, such information is a set of data characterizing the entire fuel complex (seasonal dynamics included) and its condition of moistening and drying.

The pyrological description (Table 4) includes the;

1. location of the forest inventory plot, it's

area and designation (number);

2. position in the landscape (the exposure and degree of the slope);
3. forest type and the site index;
4. canopy characteristics of the forest as follows,
    - age (the impact of surface fires is largely dependent on age),
    - height (possibility of crowning fire is dependent on the height of conifer stands),
    - species composition of each canopy story, separately for each of the two categories evergreen and deciduous (such separation is of great importance for determining ground surface fuel shading from each canopy story and fuel drying rates for both summer and spring when the canopy is foliage free),
    - relative closure of the stand is correlated with both solar light penetration (a very important factor for fuel drying) and wind velocity under the canopy (one of the most important factors of rate of flame spread). Forest stand closure (in combination with height) may be used as the basis for estimating such factors as foliage load of every tree species, maximum possible dead trees, stock, etc.;
5. dead trees stock and the load of fallen wood;
6. young trees are characterized by height and species separately for evergreen and deciduous species while the understory woody vegetation is characterized only by canopy density (young tree and the understory add to shading);
7. an overall (calculated) shading from all stories of the forest canopy, separately for spring and summer, is expressed as a decimal fraction (tenths) as compared to open place;
8. present PCB type separately for spring and for summer.

Some of the values itemized above can be found in tables or calculated by formulas on the basis of forest inventory descriptions. The automatic creation of VF maps has some difficulties because forest inventory descriptions do not have complete data on prime conductors of burning (i.e. on layers of moss, lichens and organic matter remnants). To characterize the prime conductors of burning, we used indirect techniques that analyze characteristics such as relief, soil and type of forest to predict PCB's where data are missing. This evidence is not always accurate enough, because the main task of the forest inventory is the evaluation of wood stock. Our GIS contains data from different sources of information, which is why it is possible to verify, refine, and to supplement the forest inventory data.

BIOMASS CONSUMED WITHIN EACH VF GROUPS

VF Group I

In Russian taiga forests, the moss and lichen forest floors (i.e. PCB) are typical prime conductors of burning by flame during surface fires. The average biomass stock of this layer (oven dry, 105°C) is about 1 kg/m^2. The biomass of dried branches on the forest floor is not less than 0.5 kg/m^2. The proportion of the soil humus from F and H layers' biomass consumed by fire depends on the level of drought.

VF Group II

When drought is strong, creeping ground fires develop in VG Group II. In this case, flaming PCB and smoldering of the soil F and H layers, located under the moss layer (VF Group II), can be observed. Biomass of the F and H is about 2 kg/m^2. Like VG Group I, the portion of the consumed biomass of the F and H layers depends on the drought level.

VF Group III

In the third group of vegetation fuel, the whole shrub layer is usually consumed during flaming burns of PCB. In the forest the biomass of shrubs is about 0.1 kg/m^2; the biomass of red whortleberry may reach 0.2 kg/ m^2; and the biomass of wild rosemary (Ledum) may be 0.5 kg/m^2. Live grasses have a high moisture content (200-1000%) and are passively burned in the flames of the PCB. If the biomass of live grasses exceeds the burning biomass of PCB, then fire is not possible.

VF Group IV

In VF Group IV windfallen trees lying separately are usually not consumed in the fire but are scorched only on the surface. Trunks of dead trees fallen during the fire can smoulder to ashes within several days. In a typical forest, the biomass of windfallen and dead trees is 1-3 kg/m^2, and their burning biomass is 0.1-0.2 kg/m^2 depending upon the level of drought.

VF Group V

In the fifth VF group, needles and thin branches of coniferous understory are consumed in the fire. Their biomass is not large (less than 0.1 kg/m^2). Outside the forest in the thickets of combustible bushes (e.g. Siberian dwarf-pine) the burning biomass of the canopy is the same as in the forest (0.5-2 kg/m^2).

Thus, the biomass consumed in the flaming regime under surface forest fires in the period of droughts is estimated at, 1 kg/m^2 for the PCB, 0.5 kg/m^2 for fallen down branches, 0.1-0.2 kg/m^2 for the layer of grasses and shrubs, 0.1-0.2 kg/m^2 for the windfallen and dead trees and up to 0.1 kg/m^2 for the understory. In total, the estimate for biomass consumed is 1.8 kg/m^2, in which the PCB constitute over 50% of the burning biomass.

During periods of sufficiently intense droughts, the litter (about 2 kg/m^2) burns in the smouldering regime. When a surface fire turns into a crown fire an additional 0.5-2 kg/m^2 of biomass is burned in the stand canopy.

At the beginning of the drought period, when litter does not burn and there are no crown fires, the burning biomass usually does not exceed 0.2-0.4 kg/m^2 in the forest.

To calculate the amount of carbon in absolutely dry biomass, a conversion coefficient of 0.45 can be used for green biomass (needles, foliage, grasses etc.) and 0.50 for wood, bark etc. (Kobak, 1988).

DYNAMICS OF CONSUMED BIOMASS

VF Group I

Fire experiments were used to investigate the following characteristics of PCB types.

1. probability of fire occurrence,
2. dynamics of burning biomass and,
3. dynamics of heat energy emitted from the fire.

Investigations were carried out for different drought levels. An annular (circular) screen (Wright, 1967) was used together with our heat accumulator (in the mode of a vessel filled with water) (Figure 2). According to our calibration the accuracy of the heat measurement is ±7%.

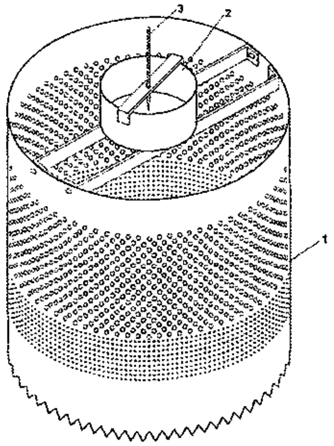

Figure 2. An annular screen for fire experiments.
1. annular screen; 2. vessel with water (accumulator of heat); 3. thermometer.

During several fire seasons 2-3 fire experiments were carried out daily. Before the experiments, samples of the forest floor (moss and lichen) were taken to determine their moisture content. Over 800 fire experiments were carried out and over 3,000 samples were analysed.

The dynamics of the following characteristics of the forest floor were investigated at different drought levels;

1. thickness of the consumed layer (**d**, cm),
2. its stock (**M**, kg/m^2),
3. its moisture content (**W**, %),
4. quantity of the emitted heat (**Q**, MJ/m^2),
5. duration of layer combustion (**T**, s),
6. heat emission intensity (**Is**, **kVt**/m^2).

Data were obtained for four PCB types including the lichen type (Lc), dry moss type (Dm) (Figure 3), moist moss type (Mm), and the loose litter type (Ll).

The dependence of the stock of consumed biomass (M) on the moisture content of the whole PCB layer can be expressed by the inverse linear dependence:

$$M = (1 - W / Wcr) \cdot Mmax$$

where;

M = stock of consumed biomass (kg/m^2),
Mmax = biomass of the whole layer (kg/m^2),
W = moisture content of the whole layer (%) and,

Wcr = critical moisture content of the whole layer (%).

As a result of these analyses, we created tables characterizing the surface burning heat of a layer (Q), the intensity of heat emission (Is) and the stock of consumed biomass (M) (Table 5) for different levels of drought.

VF Group IV

The flammability of large-sized vegetation residues (fallen branches, wind-thrown trees) in relation to weather conditions can be estimated using the U.S. National Fire-Danger Rating System (Deeming et al., 1978).

It must be taken into account that downed branches and tree trunks can burn only when heated by the burning flame carrier fuels (prime conductors of burning - PCB) on the fireline or by the trail of smouldering duff behind the advancing fire front. These fuels can sustain combustion if packed together in heaps or jackstrawed masses.

Thicker branches and boles, take a longer time to burn. Because the fire front moves on, only the wood residues up to 2.5 cm in diameter (fuel class 2 - 10 hours time lag fuels) are consumed within the flaming zone. If the fire front leaves behind it a trail of smouldering duff, class 3 woody fuels (up to 7.5 cm in diameter) can be consumed by fire. Larger boles standing alone and separated from other large residues can not burn down during the burning residence period of the fire front, so they only get surface charring.

Rotten wood matter in the lower parts of standing-dead trees can sustain steady smouldering. As a result such trees would fall down, but the process of smoldering carries on, advancing very slowly from butt to top. Several days later, only a trail of ash is left in the place of such boles. Only those dead-standing trees, which are felled by fire, can smoulder down in such manner.

Thus, the entire load of burn-ready biomass of the vegetation fuel group IV does not get consumed by combustion in surface fires.

VF Group II

The vegetation fuels of Group II (duff, humus and peat) may have a considerable load, but their drying is retarded because such fuels are buried under the layer of the prime conductors of burning. In topographic lows, these fuels may be very moist due to the capillary rise of ground water.

Table 5. Biomass stock consumed (kg/m^2) for PCB's at different drought indices.

Fire danger index of drought[1]			Types of prime conductors of burning					
(+)	typical	(-)	Lc	Dm	Mm	Ll	Cl	Cg*
40	100	300	0.4					
70	200	600	0.7					
100	300	900	0.9					(0.6)
130	400	1200	1.1					(0.7)
160	500	1500	1.2	0.5				(0.7)
200	700	2000	1.2	0.8		0.4		(0.7)
300	1000	3000	1.3	1.2	0.3	0.8	0.3	(0.7)
600	2000	6000	1.3	2.1	1.3	1.3	1.2	(0.7)
1000	3000	9000	1.3	2.4	1.5	1.4	1.5	(0.8)
1300	4000	12000	1.3	2.5	1.8	1.5	1.8	(0.8)
1600	5000	15000	1.3	2.5	1.8	1.5	2.0	(0.8)

[1] (+) less dry than typical; (-) more dry than typical

[2] Lc – lichen, Dm – dry moss, Mm – moist moss, Ll – loose litter, Cl – compact litter, Cg – cured grass. Bog moss and non-conductor PCB types are practically unburned and are not included.

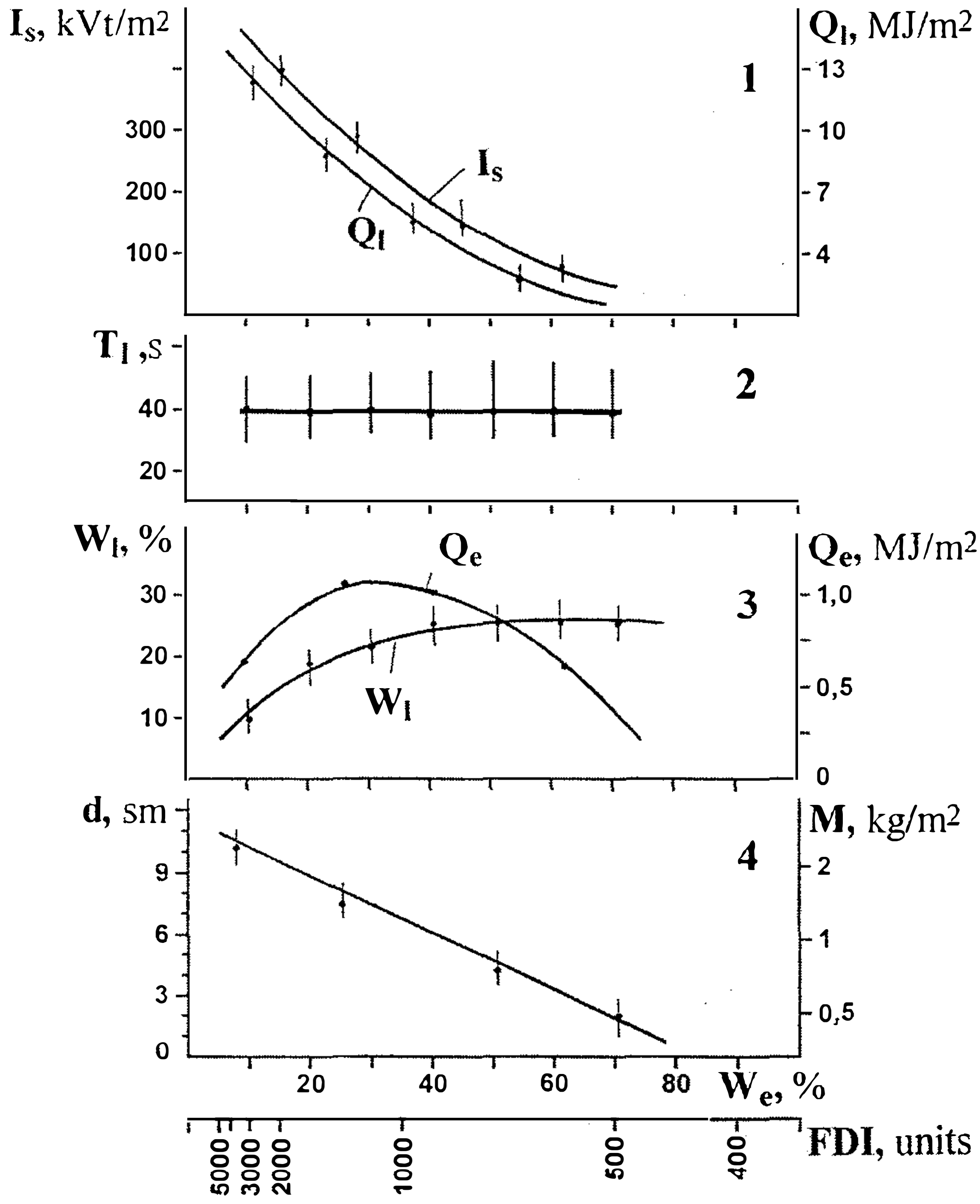

Figure 3. Dependence of the following characteristics upon moisture content (W_e) and fire danger index (FDI, Nesterov's index) using the example of dry moss PCB: 1. heat emission intensity (I_s) and quantity of the heat emitted from the consumed layer (Q_l); 2. duration of layer combustion (T_l); 3. moisture content of the consumed layer (W_l) and quantity of the heat spent for moisture evaporation from the consumed layer (Q_e); 4. thickness of the consumed layer (d) and its stock (M).

In Russia a PV-2 index had been developed for the estimation of the ignition potential of duff and peat. Scales for different forest types have been developed (Vonsky and Zhdanko, 1976). The DC index of the Canadian Forest Fire Danger Rating System represents the moisture content of the duff layer at the depth of 82 -262 mm in the standard fuel complex. The time lag of this layer is 52 days at the midday air temperature 21°C and midday air humidity 45% (Van Wagner, 1987). In the U.S. National Fire-Danger Rating System the duff (peat) layer at a similar depth of 100-300 mm has 1000-hour time lag at a constant temperature (26.5°C) and air humidity (20%).

CONCLUSION

The load of potentially combustible biomass constitutes only a part of the total biomass load. For practical purposes, biomass of the trunks, thick limbs and roots of live trees do not burn.

The biomass that is ready to burn is, in turn, only a part of the potentially combustible biomass. This part is variable over time and depends upon weather and phenological changes. In addition, not all burn-ready biomass will get consumed in forest fires.

Therefore, estimation of the biomass that will actually get consumed in a particular fire is complex. To do this, one needs to use;

1. large-scale fuel maps supplemented by pyrological description of the map's sections;
2. tables of biomass load distribution over fuel groups and fractions, depending upon the forest type and the characteristics of the forest stand and,
3. tables and equations describing the dynamics of fuel type flammability as dependent on weather indices.

A crude estimate may be made with the methods used in the Canadian Forest Fire Behaviour Prediction System (Forestry Canada, 1992), where the biomass load to be consumed by fire is presented in terms of fuel types with dependence on weather indices.

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SYNTHESIS OF FIELD NPP DATA FOR CHINA'S LARIX FORESTS

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ABSTRACT

Forest inventory data (FID) are important resources that can be used to understand the dynamics of forest biomass, net primary productivity (NPP) and carbon cycling at landscape and regional scales, especially to facilitate compliance with the Kyoto Protocol on greenhouse gas reduction and to validate ecosystem dynamic models at regional and global scales. Estimation of forest net primary productivity at regional and global scales is important to the study of global climate change. For this reason, a "biology-climate" model based on forest inventory and climatic data was developed for *Larix* forests that couples the integrated effects of both climatic factors and forest biological features. This new NPP model represents an improvement over estimating forest NPP using a climate-based NPP model which only estimates the potential NPP of vegetation.

The result of this study indicate that satisfactory estimates of forest NPP could not be obtained using a broad suite of empirical and hybrid models based solely on climatic variables. This case study also offers a methodology that can be applied to existing forest inventory data around the world to estimate NPP and evaluate the carbon balance of forests, providing researchers in other forest types with the technology and a methodology for the study of forest NPP.

Keywords: *Larix* forests, forest inventory data, NPP model

INTRODUCTION

Estimation of forest net primary productivity (NPP) at regional and global scales is important to the study of global change. These estimates can be used to validate dynamic global vegetation models developed at the ecosystem level, to assess the carbon balance at regional and global scales, and they may play a role in demonstrating compliance with the Kyoto Protocol on greenhouse gas reduction. However, regional evaluation of forest NPP in China has not been done due to the limited availability of field data and differences between the methods used to measure forest NPP. Recently, data from regional forest inventory surveys in China (1973~1976, 1977~1981, 1984~1988, 1989~1993, 1994~1998) have become available. In this paper, we used these data to develop a model to estimate forest NPP using the example of *Larix* forests. *Larix* forests were chosen because of their sensitivity to global climate change and the availability of a large field observational data set for biomass and inventory data for this forest type (Gu,1987; Wu et al.,1995; Han, 1994; Feng et al., 1994).

Larix forests are important timber resources in China. These forests are typical vegetation of the cold temperate zone where the mean annual temperature is below 0°C and the length of the frost stage ranges from 265 to 275 days. Mean annual precipitation ranges from 350mm to 550mm and occurs mostly in July and August. The zonal soil is a brown forest soil.

The dominant species of *Larix* forests in China include *Larix gmelini*, *L. sibirica* and *L. principis-rupprechtii.* Amongst these, *Larix gmelini* occupies 55% of the total *Larix* forest area and accounts for 75% of the total *Larix* forest volume. It is the zonal vegetation in China north of 49°10' latitude. The altitudinal distribution ranges from 1000m to 1600m, increasing from north to south. *L. sibirica* is found in the

Altai mountains (below 3500m), Xinjiang Autonomy, east Tianshan mountains (3500~4000m) and Baerku mountains (2100~2800m). *L. principis-rupprechtii* occurs in the Hebei Province (Weichang, Wuling mountain, Xiao Wutaishan) and Shanxi Province (Guanqin mountain, Guandi mountain, Wutai mountain, Hengshan).

Regions where *Larix gmelini* grows are characterized by short growing seasons with wide seasonal swings in temperature and precipitation. Thus, the growth and productivity of *L. gmelini* forests is strongly influenced by seasonal climate dynamics. Strong year-to-year variation in climatic condition result in pronounced between-year variation in net primary productivity (NPP) and net ecosystem productivity (NEP). Under the conditions of 2° or 4° increases in air temperature with 20% increase in precipitation, these forests are predicted to shift northward and move out of China (Zhang, 1993a,b). Therefore, global climate change induced by doubled CO_2 may strongly affect *Larix* forests. For these reasons, it is necessary to understand the characteristics of NPP of *Larix* forests in order to predict the possible effects of global change and develop scientific adaptation and/or mitigation strategies.

Often, potential NPP of forests are estimated using climate-based NPP models, such as the Miami model(Lieth, 1975), Thornthwaite Memorial model (Lieth,1975), and Chikugo model (Uchijima and Senio, 1985). These models are used to evaluate the potential distribution of forest NPP in China and its responses to global climate change (He, 1986; Chen, 1987; Zhang and Yang, 1990; Hou & You, 1990; Zhang, 1992; Xu, Guo and Yu, 1994). Some Chinese scientists, such as Jiang (1992), suggest that the development of relationships between forest NPP and environmental factors is necessary in order to predict the possible effects of global change on forest NPP in China, and emphasize the potential to use forest inventory data (FID) to meet this goal. Fang et al. (1996) roughly estimated the distribution of forest NPP in different provinces of China based on the data of the third forest survey (from 1984 to 1998).

Actual NPP of forests depends on both environmental and biological factors. Forest inventory data includes the area and volume of tree species. Volume includes the combined effects of biological and environmental factors, such as forest types, stand ages, soil conditions and forest densities. Thus, it may be possible to estimate forest NPP by combining forest inventory data with a climate-based NPP model in order to reflect the synthetic effects of both climate and biological features of the forest.

The objectives of this paper are to establish a biology-climate NPP model of *Larix* forests that couples the biological features and climatic factors, is based on forest inventory data and makes full use of forest inventory data to estimate accurately the dynamics of NPP in order to enhance our understanding of the responses of *Larix* forests to global change.

DATA COLLECTION

Over the past 40 years, a large body of knowledge about species composition and biomass of *Larix* forests has been obtained, as well as field observational data of biomass, and forest inventory data (Gu,1987; Wu et al.,1995; Han, 1994; Feng et al., 1994). The latter includes data on species composition, distribution area, volume of timber, biomass and productivity.

For this study, 16 data sets were collected from the published literature (Table 1). These data sets include both biological and environmental features, such as stand age, density, stand volume, biomass, productivity, longitude, latitude, altitude and mean annual air temperature and mean annual precipitation. In this study, biomass refers to the amount of living woody components including roots, stem, branches and leaves in a given unit area. Stand volume refers to stem volume. The minimum DBH is 5 cm, and breast height is about 1.3m. Forest NPP is defined as the production of all tissues by photosynthesis after accounting for respiration by the plants themselves.

BIOLOGY-CLIMATE NPP MODEL

In order to develop a biology-climate

Table 1. Climate,volume,biomass and NPP for *Larix* forests.

	Latitude (°)	Longitude (°)	Altitude (m)	MAT (°C)	MAPpt. (mm)	Density (plant hm^{-2})	Stand Volume (m^{3} hm^{-2})	Stand Age (yr)	Total Biomass (t hm^{-2} yr^{-1})	Net Primary Productivity (t hm^{-2} yr^{-1})	References
A	45.27	127.57	350.0	2.8	723.8	1450	139.02	21	145.62	12.423	Liu et al. ,1990
A	45.27	127.57	350.0	2.8	723.8	1323	153.59	24	154.51	14.348	Ding et al, 1990
A	45.27	127.57	350.0	2.8	723.8	1158	173.58	24	179.58	11.505	Ding et al, 1989
A	45.27	127.57	350.0	2.8	723.8	1300	168.87	24	170.19	10.248	Ding et al, 1989
A	45.27	127.57	350.0	2.8	723.8	1358	152.87	24	134.15	8.531	Ding et al, 1989
A	45.27	127.57	350.0	2.8	723.8	1475	119.03	24	134.11	8.225	Ding et al, 1989
B	47.18	128.88	707.3	-0.3	676.0	1160	131.00	25	150.80	11.040	Bai, 1980
C	52.50	124.50	650.0	-3.0	500.0	400	103.62	100	54.58	3.182	Xu et al.,1988
D	50.80	121.50	700.0	-5.0	437.4	1512	229.59	156	199.03	4.220	Feng et al.,1985
D	50.80	121.50	700.0	-5.0	437.0	811	163.66	175	196.08	4.910	Feng et al.,1985
D	50.80	121.50	700.0	-5.0	437.0	2934	75.10	107	115.14	1.560	Feng et al.,1985
E	50.40	121.70	800.0	-5.0	450.0	4237	83.69	26	67.14	6.570	Cheng et al.,1989
F	50.33	123.00	700.0	-1.7	483.0	1340	124.07	55	114.85	8.180	Cheng et al.,1989
E	50.40	121.70	800.0	-5.0	450.0	1654	142.06	91	121.24	6.790	Cheng et al.,1989
G	51.27	121.43	800.0	-5.5	450.0	717	173.53	130	145.12	4.900	Cheng et al.,1989
H	52.20	120.80	800.0	-6.0	400.0	624	180.00	171	140.37	4.950	Cheng et al.,1989

A: Maoershan Forest Farm, Shangzhi country,Heilongjiang Province; B: Liangshui Reserve of Xiaoxinganling,Heilongjiang Province;
C: Daxinganling Forest Area, Inner Mongolia; D: Chaocha Forest Farm, Genhe, Inner Mongolia;
E: Tulihe Forest Area, Daxinganling, Inner Mongolia; F: Jiwen Forest Area, Daxinganling, Inner Mongolia;
G: Jinhe Forest Area, Daxinganling, Inner Mongolia; H:Qiqian Forest Area, Daxinganling, Inner Mongolia

NPP model, we decided to first evaluate the effects of climate on forest NPP and select one of many available climate-based NPP models. Secondly, we incorporated the biological effects based on factors derived from forest inventory data.

Evaluation of a Climate-based NPP Model for *Larix* Forests

In order to select an appropriate climate-based NPP model we compared estimates of NPP for *Larix* forests using 4 different numerical models which are described below (Zhou and Zhang 1995).

1. Miami Model: developed by H. Lieth using 50 data sets from around the world obtained during the IBP (International Biology Project). The Miami model expresses NPP(g m^{-2} yr^{-1}) as the minimum of two functions (Lieth, 1975):

   $$NPP_t = 3000/(1+e^{1.315-0.119t}) \qquad [1]$$

   $$NPP_r = 3000/(1-e^{-0.000664r}) \qquad [2]$$

   where t is the mean annual temperature (°C) and r is the mean annual precipitation (mm).
2. Thornthwaite Memorial Model: Also suggested by H. Lieth, this model calculates NPP (g m^{-2} yr^{-1}) in terms of the mean annual actual evapotranspiration (Lieth, 1975).

   $$NPP_E = 3000\,(1-e^{-0.0009695E}) \qquad [3]$$

   where E is the mean annual actual evapotranspiration estimated from Thornthwaite model (mm).
3. Chikugo Model: This model is based on potential evapotranspiration and water-use efficiency of vegetation, and its parameters were determined using 682 data sets of forest and related climate data obtained during the IBP (Uchijima and Senio, 1985). It can be written as,

   $$NPP= 0.29\,R_n \cdot \exp[-0.216(RDI)^2] \qquad [4]$$

   where NPP is calculated in units of t DW·ha^{-1}·yr^{-1}, RDI is the radiative dryness index ($=R_n/Lr$, L is the latent heat (0.596kcal/g)), r is the mean annual precipitation(cm), R_n is intercepted net radiation at the land surface (Kcal·cm^{-2}·yr^{-1}).
4. Synthetic Model: Developed by Zhou and Zhang(1995), this model is based on an analytical procedure similar to the Chikugo model. However, it replaces potential evapotranspiration by actual evapotranspiration of vegetation.

$$NPP= RDI\,\frac{r\,R_n\,(r^2+R_n^2+rR_n)}{(R_n+r)\,(Rn^2+r^2)}\,\exp[-(9.87+6.25RDI)^{0.5}] \qquad [5]$$

where r is the mean annual precipitation (mm), R_n is the intercepted net radiation at the land surface (Kcal·cm^{-2}·yr^{-1}) and NPP is calculated in units of t·ha^{-1}·yr^{-1}. R_n can be estimated using the relationships between RDI and potential evapotranspiration rate (PER=PET/r) developed by Zhang and Yang(1993) based on 30 year average data (1951~1980) from 647 meteorological stations in China.

$$R_n = RDI \times r \times L \qquad [6]$$

$$RDI = 0.629+0.237PER - 0.00313PER^2 \quad (R^2=0.90) \qquad [7]$$

where Rn is in units of Kcal·cm^{-2}·yr^{-1}, r is in units of mm, and PET is the mean annual potential evapotranspiration (mm) calculated by the Thornthwaite method.

NPP values for *Larix gmelinii* forests from the same climatic conditions were averaged to produce 8 data sets with different combinations of climatic and biological factors. These 8 data sets were used to validate the climate-based NPP models (Table 2). The correlation coefficient (R^2) between the simulated values and the observed values range from 0.443 for the Miami model to 0.523 for the Synthetic model (Table 2). The correlation coefficients of the four climate-based NPP models are very close in value (around 0.5) and considered relatively low. The Thornthwaite Memorial model was selected to reflect the effects of climate on *Larix* forests in further model development, since net radiation (R_n) values, which are necessary for the Synthetic and Chikugo models, are often not observed at climatological observation stations.

Effects of Biological Factors on NPP of *Larix* Forests

Net primary production of forests is a function of the biomass, stand age and environmental conditions. Total forest

Table 2. Comparison between the observed and the simulated NPP data of *Larix* forests in China based on climate-based NPP model ($t \cdot hm^{-2}yr^{-1}$)

Sampling No.	B	A	F	E	H	G	D	C	Correlative efficient (R^2)
Observed values	11.04	10.88	8.18	6.68	4.95	4.9	4.038	3.182	1.0
Miami Model	6.17	8.18	5.40	3.87	3.49	3.67	3.87	7.71	0.443
Thornthwaite Memorial Model	10.05	10.91	7.68	7.23	6.46	7.16	7.10	5.51	0.513
Chikugo Model	7.60	8.46	5.52	5.14	4.51	5.08	5.04	4.95	0.516
Synthetic model	6.88	7.77	5.02	4.69	4.09	4.606	4.61	4.74	0.523

A: Maoershan Forest Farm, Shangzhi country,Heilongjiang Province;
B: Liangshui Reserve of Xiaoxinganling,Heilongjiang Province;
C: Daxinganling Forest Area, Inner Mongolia;
D: Chaocha Forest Farm, Genhe, Inner Mongolia;
E: Tulihe Forest Area, Daxinganling, Inner Mongolia;
F: Jiwen Forest Area, Daxinganling, Inner Mongolia;
G: Jinhe Forest Area, Daxinganling, Inner Mongolia;
H: Qiqian Forest Area, Daxinganling, Inner Mongolia

biomass can be obtained based on stand volume from forest inventory data, since stand volume has a close relationship with total forest biomass (Brown & Lugo, 1984). Stand age of forests reflect different growth stages and has a close relationship with NPP. The relationships between NPP and stand volume and between NPP and stand age of *Larix* forests are shown in Figures 1 and 2, respectively.

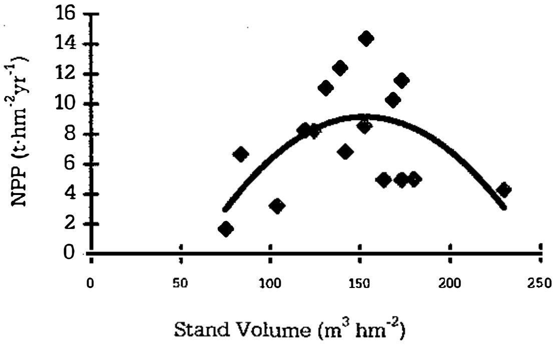

Figure 1. Relationship between NPP and stand volume of *Larix* forests.

Thus, the biological characteristics of *Larix* forests may be expressed as a function of stand volume and stand age. The biology-climate NPP model could be expressed as;

$$NPPa = NPPc \times f(A,V) \quad [8]$$

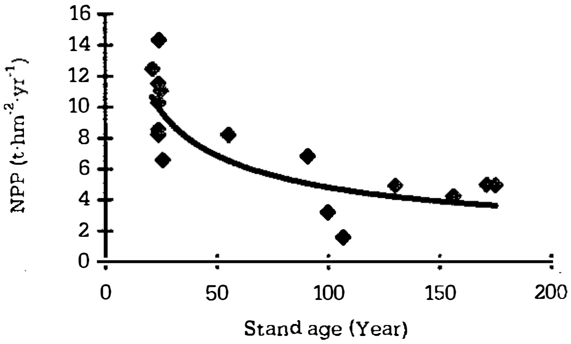

Figure 2. Relationship between NPP and stand age of *Larix* forests.

where NPPa ($t \cdot ha^{-1} \cdot yr^{-1}$) reflects the collective effects of both climatic factors and biological features, NPPc reflects only the effects of climate factors and can be calculated by the vegetation-climate NPP model ($t \cdot ha^{-1} \cdot yr^{-1}$), and f(A,V) reflects only the effects of biological characteristics on forest NPP, which can be estimated by the following equation:

$$f(A,V) = NPPa / NPPc \quad [9]$$

f(A,V) can be obtained using the observed NPP of *Larix* forests and the simulated NPP from Thornthwaite Memorial model. The relationships between f(A,V) and the biological factors from the forest inventory data were analyzed using the same data sets as those used to evaluate the climate-based

NPP models. The results indicate that f(A,V) has a close relationship with mean annual volume increment (the ratio of stand volume to stand age), which includes the integrated effects of forest biomass and stand age. The relationship between f(A,V) and V/A could be expressed as:

$$f(A,V) = 0.3318\,Ln(V/A)+0.4747 \quad (R^2=0.6199) \qquad [10]$$

where V is stand volume ($m^3 \cdot ha^{-1}$), and A is stand age (yr). The relationship between F(A,V) and V/A indicates that NPP of *Larix* forests increases with the increase in mean annual volume increment (Figure 3).

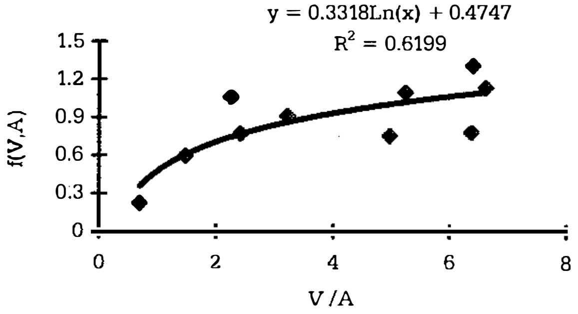

Figure 3. The effect of biological factors on NPP of *Larix* forests.

Biology – Climate NPP Model

Combining equations [3] and [10] into equation [8], the NPP model of *Larix* forests that includes the integrated effects of both climatic factors and biological features is as follows:

$$NPPa = 0.3318Ln(V/A)+0.4747) \times 30(1-e^{-0.0009695E}) \qquad [11]$$

This biology-climate NPP model, and the NPP of *Larix* forest can be easily calculated using climate and forest inventory data. This model may be helpful to calculate NPP values using existing forest inventory data from around the world which can be used to evaluate forest NPP dynamics. These data can be used to validate ecosystem dynamic models at regional and global scales and to evaluate the global carbon balance. Figure 4 shows a comparison of simulated and observed NPP values of *Larix* forests based on data sets that were not used to estimate the parameters of the biology-climate model.

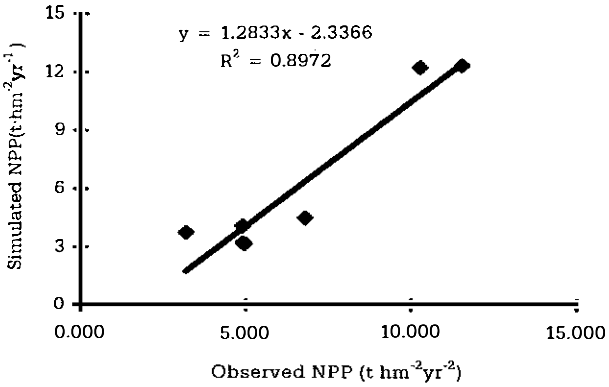

Figure 4. Comparison between observed data and simulated data of NPP of *Larix* forests.

The simulated NPP values correspondent closely to the observed NPP data of *Larix* forests, with a correlation coefficient (R^2) of 0.8972.

DISCUSSION AND CONCLUSIONS

The accurate estimation of net primary production of forests is very important, not only for guiding scientifically sound forest management but also for improving global change studies, for the establishment of a NPP database to validate ecosystem models at regional and global scales, to accurately predict the possible effects of climate change on forest NPP at regional and global scales and for evaluating compliance with the Kyoto Protocol on greenhouse gas reduction. The development of a biology-climate NPP model in this paper, based on the data from *Larix* forests in China, provides a demonstration of estimating forest NPP in terms of climatic factors and forest inventory data. The conclusions related to *Larix* forests are summarized below:

1. Best estimates of NPP for *Larix* forests could not be obtained using only a climate-based NPP model (Miami model, Thornthwaite Memorial model, Chikugo model and Synthetic model). Climate-based NPP models can only give the potential NPP of vegetation or NPP of zonal forests.
2. Forest inventory data can be used to reflect the biological characteristics of forest. A new forest NPP model coupling the integrated effects of both climatic factors and forest biological features, the so-called biology-climate

NPP model, was developed using forest inventory data and climatic factors:

$$NPPa = (0.3318 \mathrm{Ln}(V/A) + 0.4747) \times 30(1 - e^{-0.0009695E})$$

This case study offers a method that may be applied to existing forest inventory data around the world to evaluate NPP and the carbon balance of forests and to provide researchers in other types of forests, a useful technology and methodology for the study of forest NPP.

ACKNOWLEDGEMENTS

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THE ROLE OF CHINA'S TEMPERATE FORESTS IN THE ATMOSPHERIC CARBON BUDGET

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ABSTRACT

Temperate forest ecosystems are believed to be an important sink of atmospheric carbon. China's temperate forests account for nearly one half of the nation's total forest area. Currently there is great potential for, and trend toward increased, development in these forests. Therefore, accounting for the role of China's temperate forests in the global carbon budget is very meaningful.

The carbon emission and sequestration by temperate forests in the northern part (15 provinces) of China in 1990 and the following 60 years were calculated and projected, based on data from China's national forestry inventories (1984-1988, 1989-1993 and 1994-1998), the national forestry development plan, and parameters from ecological research. A computer model F-CARBON was developed by the authors and used for the calculations and projections.

For the F-CARBON model, forests in China were classified into 5 age classes, and it was assumed that 6-12% of the forest in each age class would transfer to next higher age class. The differences amongst biomass density and growth rates for the forests in different age-classes were considered. The life time for biomass oxidization and decomposition was also considered. The soil carbon changes between harvesting and reforestation were estimated.

The calculations indicated that in 1990 temperate forests in China had taken up 52.6 MtC, release 16.5 MtC and accumulate 6.1 MtC in forest soils, resulting in a net absorption of 42.2 MtC which amounts to 7.5% of the national industrial CO_2 emissions in that year. Projections of carbon emissions were made for a 60 year period starting in 1990. In 2050, the net carbon uptake by China's temperate forests is expected to be around 87.7 MtC, almost double the value estimated for 1990. From 1990 to 2050 and 2008 to 2012, accumulated net carbon uptake by China's temperate forests is predicted to be 4251 MtC and 162 MtC, respectively. The North and Northwestern region of China has the greatest potential for carbon sequestration through forest development.

Keywords: China's temperate forest, carbon uptake, projection, F-CARBON model

INTRODUCTION AND BACKGROUND

Global change is a reality that was acknowledged by governments at the Rio Summit in 1992 and at Kyoto in 1997. The Intergovernmental Panel on Climate Change (IPCC) considered it possible to increase afforestation to offset 7.5% of the carbon that is likely to be emitted through burning fossil fuels up to the year 2050. Individual countries are claiming forest carbon sinks to offset some of their requirement to cut fossil fuel emissions (Cannell 1999). Some national and/or regional carbon sinks/sources for the forest sector have been estimated and projected under several mitigation scenarios (Kauppi and Tomppo 1993; Cannell and Milne 1995; Cannell et al 1999; Rodriguez 1994; Schroeder and Winjum 1995a,1995b; Kokorin et al 1996; Borjesson et al 1997). To estimate the magnitude of credits claimed against the requirement to cut fossil fuel emissions, it is necessary to understand the current and projected forest carbon budgets in China.

There are about 200,000 permanent plots established for periodic forest inventories in China, and national forest inventories have been conducted about every 5 years in China, though there is no specified regulatory requirement for periodic inventories. There have been five forest resource inventories conducted since 1973. The results are presented in Table 1.

The definition of forests was based on a 0.3 threshold of canopy density, for inventories prior to 1994. For the period between 1994-

1998 and subsequent forest inventories, the threshold value was changed to 0.2 (Table 1).

Table 1. Forest inventories conducted in China (CMOF, 1996a; CMOF, 2000).

Investigate period	Forest area (M ha)	Forest cover (%)	Living stock (M m^3)
1973-1976 a)	122	12.70	8,700
1977-1981 a)	115	12.00	9,000
1984-1988 a)	125	12.98	9,100
1989-1993 a)	134	13.92	10,100
1994-1998 b)	159	16.55	

a) the definition of forest based on a canopy density with threshold value of 0.3
b) threshold for canopy density of 0.2

According to the 1994-1998 inventory, the area under forest cover in China was 158.94 million hectares (Mha), amounting to 16.55 percent of the total territory, which is much greater than the 145.19 Mha and 15.12% estimated from the 1988-1993 inventory (converting to 0.2 threshold value). The forested area is estimated at 0.13 ha per capita, which is much less than the global average of 0.65 ha per capita. The average living forest stock per capita in China is estimated at 8.6 m^3, only 12 percent of the world average of 71.8 m^3, ranking the lowest in the world (CMOF 1994a).

The timber resource available in China for harvesting is limited and the role of forestry in the national economy is small. However, China has the largest afforested/reforested areas in the world and the rate of plantation development is one of the highest in the world. There were more than 46.67 Mha of plantations established from 1949 to 1998. In the period from 1980 to 1988 the area of newly planted forests increased by 479,000 ha per year, and in the period from 1989 to 1993, it increased at a rate of 2.2 Mha per year, more than four times the previous rate. On average, the annual rate of increase for the area of forest cover, increased by 3.66 Mha from the period 1989-1993 to the period 1994-1998 and the annual rate of increase for forest coverage was as high as 0.29 percent.

Furthermore, China has great potential to enlarge its forest area. The primary objective of the Chinese Ministry of Forestry (CMOF) long-term forestry and afforestation plan, is to increase forest coverage to 19.4, 24 and 26 percent of the geographic area of China by the year 2010, 2030 and 2050, respectively (CMOF, 1999).

Temperate forests are believed to be a net sink of atmospheric carbon (IPCC 2000). Nearly one half of the forests in China are distributed in the temperate zone (i.e.,12 provinces in the Northern and Northwestern China and 3 provinces in the Northeastern China). In addition, there is a large area of land available for forest development. With its large forest areas and rapid forest development, the temperate forest in China has great potential for carbon uptake and to play an important role in the atmospheric carbon budget.

Chinese foresters have never estimated the carbon stocks and budget of China's forests and major forest ecosystems (Liu et al 2000, Zhou et al 2000). Although Xu (1992, 1995,1999) estimated the current and future carbon budget of forests in China, these estimates were mainly based on a relatively old forest development plan (CMOF 1995a) and forest data (CMOF 1994a, 1989). The latest forest data (CMOF, 2000) showed that forested areas in the past ten years in China increased much faster than expected. In the new forest development plan issued by CMOF in 1999, forested areas will be larger than were expected in the old plan. Furthermore, most of the estimation in the past did not identify carbon density in different age classes and the carbon sink/emission of soils and wood utilization. It is necessary to re-estimate the current forest budget and potential sink of China's forests in the future, using the new data and latest forest development plan.

The objective of this paper is to calculate the carbon budget of forests in the temperate region of China in the base year 1990 and subsequent years until 2050. To realize this objective, the F-Carbon Model was conceived and developed by authors.

METHOD

Introduction

The accuracy of Greenhouse Gas (GHG) estimation increases when forests are divided into smaller units, because of improved heterogeneity of growth rates and other parameters within forest types. To reduce the errors in calculations, the temperate region in China was divided into 2 parts: the Northeastern China and the Northern and Northwestern China, because forest types and other parameters in the two regions are quite different. The biomass density and growth rate are very different for forests in different age-classes. The age-class and its change with time are introduced in the calculations of the F-CARBON model. The end uses of wood have different duration periods for oxidation, and they influence the carbon emission processes when the time lag between wood harvesting and carbon emissions is considered. Estimates of parameters and data used for calculations were mainly based on data from forest resource inventories conducted in 1984-1988 (CMOF 1989), 1989-1993 (CMOF 1994a) and 1994-1998 (CMOF, 2000). Some data were used from the China Forestry Statistics 1990-1997 (CMOF, 1991-1998), the China National Ecological Re-establishment Planning – Forest Aspect (CMOF, 1999) and ecological research reports listed in the references. A summary of some of these data used in the F-Carbon model are listed in Table 2.

Table 2. Parameters for calculation of the carbon budget in 2 forest regions of China.

		Region		
Parameter		Northeast	North & Northwest	Temperate Total
Forest area at base year (Mha)[a]		26.042	29.394	55.436
Land available for forest development (Mha)[a]		9.309	48.913	58.222
Harvested area at base year (Mha)[a]		0.70	0.30	1.00
Carbon content (%)		0.45	0.45	
Wood density (t.m^{-3}) [b]		0.44	0.40	
Ratio of above ground biomass/ stem biomass [c]		1.2	1.3	
Total/above-ground biomass ratio [c]		1.3	1.3	
Forest Soil carbon (t.ha^{-1}) [d]		163.8	159.9	
Annual growth (m^3.ha^{-1}yr^{-1}) [a]		2.91	2.91	
Relative growth factors for 5 age classes		0.70(young), 1.15(mid-aged), 1.20(near-matured), 1.15(matured), 0.8(over-matured)		
Area weighting factors for 5 age classes [a]	Young stand	0.3391	0.3577	
	Mid-aged	0.3893	0.3264	
	Near-Mature	0.0953	0.1330	
	Mature	0.1309	0.1200	
	Over-mature	0.0456	0.0630	
Volume weighting factors for 5 age classes [a]	Young stand	0.1312	0.1112	
	Mid-aged	0.3950	0.3202	
	Near-Mature	0.1463	0.1776	
	Mature	0.2378	0.2283	
	Over-mature	0.0896	0.1626	
Decomposition factors		0.10	0.15	
Burnt proportion		0.10	0.10	
Proportion of litterfall		0.025	0.033	
Rotation (year)		80	60	
Proportion of area of lower age class to higher age class		0.06	0.08	

a. calculated based on forestry inventory during 1984-1988 and 1989-1993 (CMOF, 1989 1994a)

b. Internal information

c. shared biomass database with nearly 8000 set of biomass data

d. average organic carbon in forest soils, with sampling number 65 for Northern China and 77 for North and Northwestern China

F-CARBON Model

Our research team developed a computer model, F-CARBON, to calculate the carbon budget of China's forestry sector. The model incorporates most of the methods proposed by the IPCC. The principal advantage of the F-CARBON model is that it is able to account for the variation in biomass densities and growth rates for different age classes of forests. The model uses data of five age-classes in the initial year to calculate the age-classes for the following years. The model assumes the proportion of the forest in one age class which will move to a higher age class each year, varies between different regions.

In the F-CARBON model the lifetime for biomass oxidization and decay is considered. Therefore the gradual process of carbon emissions is addressed. If the long-term average of carbon emissions is calculated, the introduction of lifetime is not necessarily. But when a short time frame (for example less than 50 years) is used, the lifetime for biomass oxidization is meaningful, especially when the pattern of wood consumption is changing.

The F-CARBON model is composed of five sub-models, including area calculation, carbon uptake, carbon emission, soil carbon and carbon budget.

Area Calculation Sub-Model

Since the interval between forest inventories is approximately 5 years, it was assumed that the forest area in each inventory is represented by the data for the last year of the inventory period. That means the areas inventoried in 1984-1988, 1989-1993 and 1994-1998 are the areas for the years 1988, 1993 and 1998, respectively. The data between these years were interpolated and those after 1998 were calculated based on the long-term forestry plan (CMOF 1999). There are data for national total cut area but not broken down for each province or region in the forest inventory. Fortunately, the cut area for each region could be calculated using data for harvested volume and standing volume of mature stands, which are available for each province from the annual forest statistics.

The base year for calculation is 1990. Forest areas in each age class in subsequent years, in each region, were estimated based on the areas in base year, on the age class conversion ratio, cut area and actual/ planned forest development. The five age classes are young stand, mid-age stand, near-mature stand, mature stand and over-mature stand. It was assumed that cutting always occurred in over-mature stands followed by mature stands, after all over-mature stands were cut. It was also assumed that in Northeastern China, 6% of forests in one age class would move to the next higher age class each year, while the ratio is 8% in the Northern and Northwestern China.

Carbon Uptake Sub-Model

Since biomass investigations have never been conducted using the National Forestry Inventories, the carbon uptake has to be estimated through volume stock and volume growth rates as follows;

$$UPTAKEC = \sum_{i=1}^{2}\sum_{j=1}^{5} AREA_{i,j}\cdot GROWTH_{i}\cdot AGEGR_{j}\cdot WODEN_{i}\cdot CRCNT_{i}\cdot ASTEM_{i}\cdot TOTAG_{i} \qquad [1]$$

where;

i and j	are regions and age classes respectively,
$AREA_{i,j}$	is the forest area for each age class in each region (ha),
$GROWTH_i$	is volume growth for each region per year(m^3 ha^{-1} yr^{-1}),
$AGEGR_j$	is the growth factor for each age class,
$WODEN_i$	is wood density for each region ($tB.m^{-3}$),
$CRCNT_i$	is carbon density ($tC.tB^{-1}$),
$ASTEM_i$	is ratio of above-ground biomass to stem biomass, and
$TOTAG_i$	is the conversion factor from total biomass to above-ground biomass.

Carbon Emission Sub-Model

Carbon emissions include on-site burning of slash, oxidation due to timber utilization and decomposition of forest litterfall and slash left after burning. The removed aboveground biomass carbon per year was calculated using the following equation;

$$ABBIOC= \sum_{i=1}^{2} CUTAREA_i\cdot VOLUME_i\cdot WODEN_i\cdot CRCNT_i\cdot ASTEM_i \qquad [2]$$

where;

$CUTAREA_i$ is the cut area for each region (ha), and

$VOLUME_i$ is the stem volume in over-mature stands for each region (m^3ha^{-1}).

The under-ground biomass carbon left on cut areas is calculated as follows,

$$BBIOC = \sum_{i=1}^{2} ABBIOC_i \cdot (TOTAG_i - 1) \quad [3]$$

where;

$ABBIOC_i$ is removed aboveground biomass for each region.

- *Carbon Emission Due to Timber Harvesting*

Approximately 70 percent of the stock volume is taken for commercial timber, while the remaining 30 percent is left on site. There are many kind of timber uses, however 8 categories were developed based on the lifetime of timber use. It was assumed that timber is oxidized at an average rate over its lifetime. Therefore the total carbon emission from commercial timber use can be estimated using the following equation:

$$TIMUSEC = \sum_{i=1}^{2}\sum_{j=1}^{Y}\sum_{k=1}^{8} \frac{ABBIOC_{i,j} \cdot ALLOC_k \cdot 0.7}{ASTEM_i \cdot D_k} \quad [4]$$

where;

$ALLOC_k$ and D_k are percentage and lifetime respectively, for each of the eight timber use categories.

- *Emission From On-site Burning*

Carbon emissions from site burning are significant in subtropical and tropical zones, where slash and burn is a common method for site preparation before regeneration. In the temperate regions, the ratio of on-site burning is relatively small. The carbon emission from this source was estimated as follows:

$$BURNTC = \sum_{i=1}^{2} ABBIOC_i \cdot (1 - 1/ASTEM_i) \cdot BURNT_i \quad [5]$$

where;

$BURNT_i$ is the burnt proportion of slash biomass left on site.

- *Decomposition Emission*

The underground biomass and slash left after burning remains on site to decay and decompose. Their total carbon is estimated as follows:

$$LEFTC = \sum_{i=1}^{2}(ABBIOC_i - TIMUSEC_i) \cdot (1 - BURNT_i) + BBIOC_i \quad [6]$$

and the total carbon in forest litterfall is,

$$LITC = \sum_{i=1}^{2}\sum_{j=1}^{5} AREA_{i,j} \cdot AGEVLM_{i,j} \cdot WODEN_i \cdot CRCNT_i \cdot ASTEM_i \cdot LITFL_i \quad [7]$$

where;

$AGEVLM_{i,j}$ is stand volume for each age class in each region ($m^3\ ha^{-1}$),

$LITFL_i$ is the proportion of litterfall from above-ground biomass.

The slash left, underground biomass and litterfall, decompose using an exponential decay function as follows:

$$DECOMC = \sum_{i=1}^{2}\sum_{j=kk}^{YY} DEC_{i,j} - DEC_{i,j+1} \quad [8]$$

where;

kk is the initial year ,

YY is the end year you wish to calculate, and,

$$DECi,j = (LEFTCi \cdot LITCi) \cdot (DERATi \cdot EXP(-DERATi \cdot (j - kk + 1))) \quad [9]$$

where;

$DERATi$ is the decomposition factor for each region, and

j changes from kk to YY.

Soil Carbon Sub-Model

The net change of soil carbon is the difference of soil carbon release and accumulation. Soil carbon release each year in cut areas is assumed to be a proportion of the original soil carbon stock. The accumulation of soil carbon is assumed to be a proportion of litterfall and underground biomass left on site.

Carbon Budgeting Sub-Model

Net carbon changes of the forest are the sum of all carbon uptake minus the total carbon released.

$$NETC = TIMUSEC + BURNTC + DECOMC + SOILC - SINK \quad [10]$$

RESULTS

The carbon stock in China's temperate forests in 1990 was 11.1 GtC, amounting to 7 percent of the global carbon stock in temperate forests (159GtC, IPCC 2000) and 0.97 percent of global forest carbon stock (1146000MtC, IPCC 2000).

The carbon emissions and uptakes of temperate forests in China from 1990 to 2050, calculated using the F-CARBON model, are listed in Table 3 and illustrated in Figures 1-4. Results indicate that China's temperate forests are a net sink of atmospheric carbon. In 1990 net uptake was 42.2 MtC yr^{-1}, which was the net result of 52.6 MtC yr^{-1} of growth uptake, 16.5 MtC yr^{-1} of emission (biomass burning and decomposition) and 6.1 MtC yr^{-1} of soil carbon accumulation. The net uptake in 1990 amounted to 7.5% of China's industrial CO_2 emission in 1990 (560MtC, China Climate Change Country Study,1999). Forest soil was a net sink of atmospheric carbon (Table 3).

From 1990 to 2050, carbon emissions and soil carbon flux increased slightly, but carbon uptake through biomass growth and forest carbon stocks increased rapidly, particularly before 2030 (Table 3 and Figures 1,2,3). The projected net carbon uptake by the year 2050 in China's temperate forests is 87.7 $MtC.yr^{-1}$, which is double the value for 1990 (Table 3).

4251 MtC is expected to accumulate in China's temperate forests from 1990 to 2050, of which 2720 MtC will accumulate during the period between 2000-2050 (Table 3, Figure 4).

The budget indicates that during the period 2008-2012, China's temperate forests will have a net uptake of 251 MtC (89MtC in Northeastern China and 162 MtC in North and Northwestern China) accounting for 44.8 percent of industrial CO_2 emissions in 1990.

There were no significant differences between the Northeastern China (NE) and North and Northwestern China (NNW) regions for net carbon uptake. However, there is a much greater potential for forest development and carbon uptake in NNW because the land available for forest development is 5 times greater than that in NE (Table 2). From 1990 to 2050, the net carbon uptake by forests in NE and NNW is expected to increase from 20.3 $MtC.yr^{-1}$ and 23.5 $MtC.yr^{-1}$ respectively, to 25.6 $MtC.yr^{-1}$ and 62.1 $MtC.yr^{-1}$ respectively, with corresponding values for accumulated net carbon uptake of 1418 MtC and 2833 MtC (Figure 5).

Table 3. Carbon budget of temperate forests in China from 1990 to 2050 ($MtC.yr^{-1}$)

Year		Emission	Growth Uptake	Soil Carbon Flux	Budget
1990		16.487	-52.590	-6.125	-42.227
1995		16.632	-57.684	-6.285	-47.338
2000		16.776	-63.201	-6.212	-52.638
2005		17.403	-68.400	-6.593	-57.591
2010		18.218	-73.539	-6.961	-62.282
2015		18.883	-78.577	-7.391	-67.085
2020		19.565	-83.494	-7.875	-71.804
2025		20.271	-88.286	-8.404	-76.419
2030		20.992	-92.956	-8.971	-80.935
2035		21.715	-95.264	-9.497	-83.045
2040		22.443	-97.090	-10.010	-84.655
2045		22.781	-98.579	-10.500	-86.302
2050		23.111	-99.844	-10.980	-87.716
Accumulated Flux (MtC)	1990-2030	741	-3002	-294	-2556
	1990-2050	1187	-4943	-495	-4251
	2000-2030	568	-2367	-225	-2024
	2000-2050	1014	-4308	-426	-3720
	2008-2012	73	-296	-28	-251

Note: negative "-" means uptake and positive means "emission"

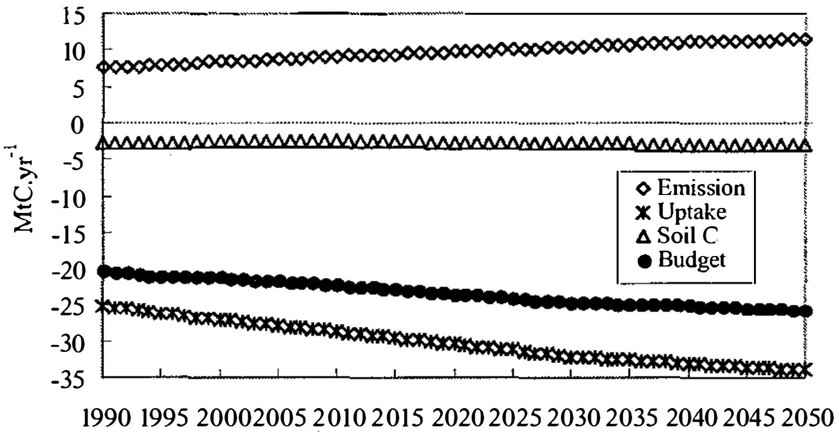

Figure 1. Forest carbon budget in Northeastern China.

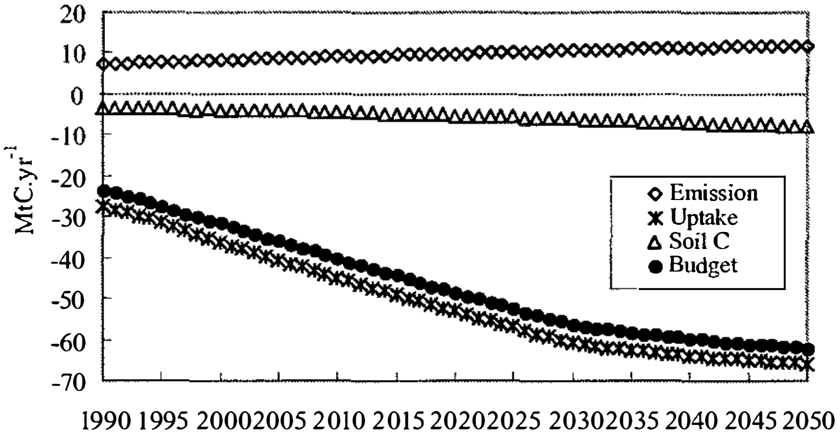

Figure 2. Forest carbon budget in North and Northwestern China.

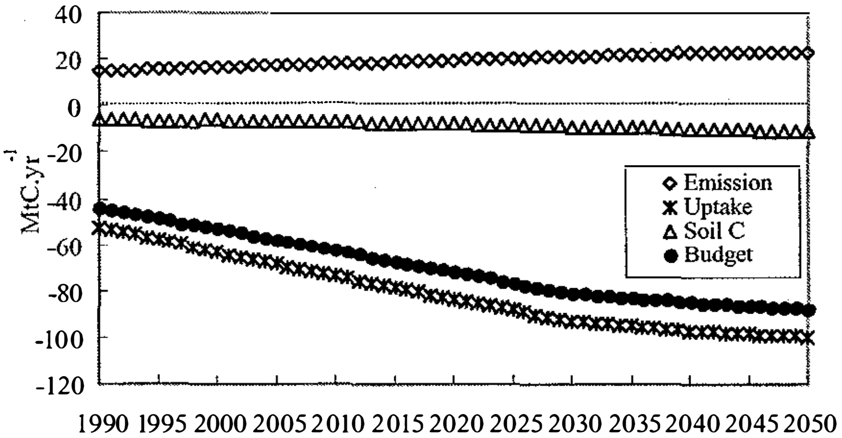

Figure 3. Forest carbon budget in the temperate region of China.

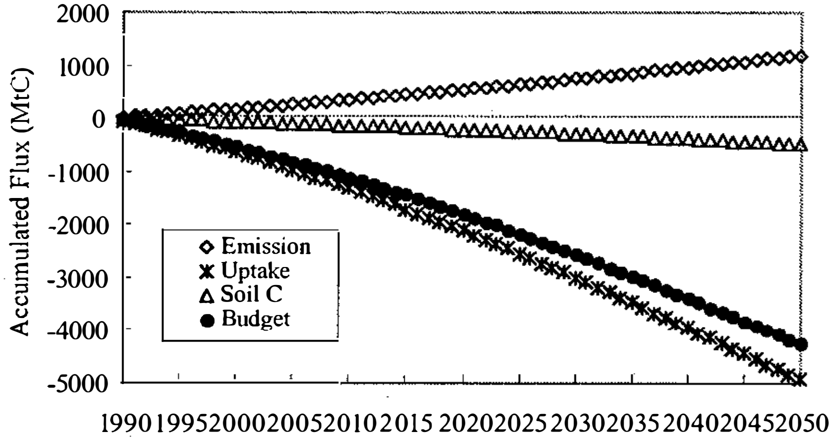

Figure 4. Accumulated forest carbon flux in the temperate region of China from 1990 to 2050.

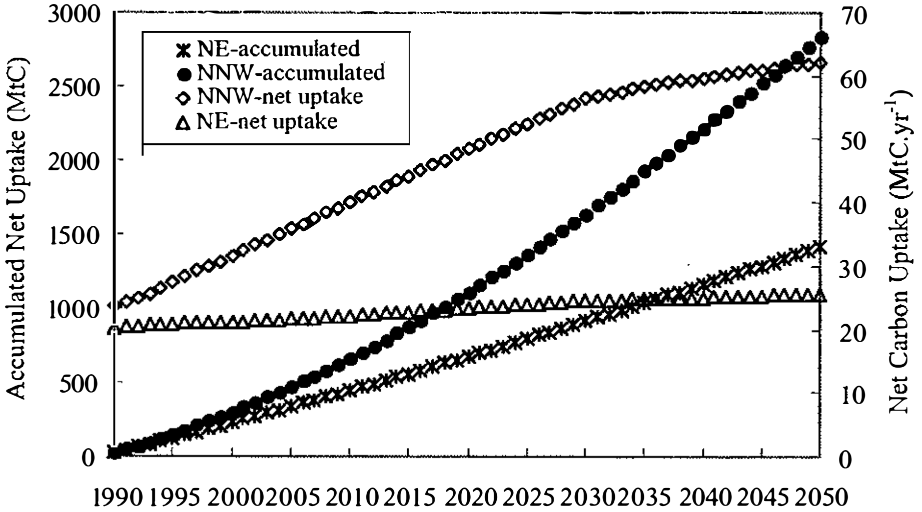

Figure 5. Carbon uptake by forests in two regions of China from 1990 to 2050.

CONCLUSIONS

The results of our forest carbon budget indicate there was 11.1GtC stored in China's temperate forests in 1990, accounting for 7 percent of the carbon in global temperate forests. The temperate forests of China had a net uptake of about 42.2 MtC yr^{-1} in 1990 accounting for 7.5 percent of the national industrial CO_2 emissions, and with forest development, the net uptake is expected to increase gradually up to 87.7 MtC yr^{-1} in 2050. The accumulated net uptake from 1990 to 2050 and 2008 to 2012 would be 4251 MtC and 162 MtC respectively. Forest soils in the temperate region of China may play a significant role as a net sink of atmospheric carbon. The North and Northwestern region of China has the greatest potential for carbon sequestration through forest development, compared with the small potential in Northeastern China.

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IMPACT OF FIRE ON THE CARBON BUDGET IN PINE FORESTS OF CENTRAL SIBERIA

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ABSTRACT

Fire in the pine forests of Central Siberia contributes considerably to the regional carbon budget. Biomass carbon stocks depend on pine stand age, site conditions, and time since the last fire. Mean fire interval decreases from north to south and is determined by latitude, degree of landscape isolation, and anthropogenic impact. Fire intensity, vegetation type, and site conditions all influence the amount of fire emissions. The latter can increase by an order of magnitude during extreme fire seasons.

Keywords: forest fire, carbon budget, mean fire interval, fire intensity, boreal forest, Central Siberia

INTRODUCTION

About 16,000 forest fires occur annually in Russia. Based on official statistics, these fires burn an average of 900,000 ha annually. Fire experts, however, estimate this area to be closer to 7 to 12 million ha (Dixon and Krankina 1993; Conard and Ivanova 1997). Predicted global warming is expected to increase both fire frequency and intensity, which will in turn, worsen site conditions and influence carbon storage in boreal forests.

Most of Russia's pine stands are found in Siberia, where they make up 30% of the total coniferous forest. About 1/3 of the terrestrial carbon in Siberia is accumulated in pine forests (Alexeyev and Birdsey 1994). Because of their vast extent, and the large amounts of aboveground biomass and organic matter in the soil and the forest floor, these pine forests have a key role in the global carbon budget and atmospheric chemistry. Furthermore, pine forests account for up to 60% of the total number of forest fires (Korovin 1996), and fire intensities vary widely in both time and space. Fire type and intensity determine emission amounts and postfire tree mortality and regeneration. Carbon stocks and emissions depend on vegetation type and site conditions, so investigation of fire contribution to carbon storage changes in central Siberian pine forests is of primary importance.

Pine Forest Extent and Type Diversity in Central Siberia

Climatic conditions determine pine forest geography. However, pine can survive under unfavorable regional climate because it is able to grow where local edaphic conditions are favorable. This allows it to grow in many climatic zones (Shimanuk 1962). In Central Siberia, pine forests are most common in the Angara river basin (its upper and middle parts), as well as in the watershed separating the Angara River from the Podkameni Tunguska, Lena, and Nizhni (Lower) Tunguska headwaters. Pine grows here under a wide range of site conditions and covers about 3.85 million ha. In this area, pine forests are made up mainly of three types: pine-lichen, pine-green moss, and pine-mixed herb types. The pine-lichen forest type accounts for not more than 6% of the total pine forest land in the Angara region, while pine-mixed herb and pine-green moss forests cover 67% of the area in approximately equal shares (Forests of the USSR 1969).

In the boreal zone of Central Siberia, pine forest type diversity is fairly wide and depends on soil conditions. Pine-shrub forest types comprise the biggest type group: pine-alder forest grows on carbonate soils rich in loam; pine-dwarf shrub forest is favored on heavy loam; pine-rhododendron stands are limited to sandy soils and loamy sands; pine-small shrub forest (*Vaccinium myrtillus*, *V.uliginosum*, *V.vitis-idaea*) occurs on gentle slopes on moderately moist heavy loam soils; pine-lichen-*V.vitis-idaea* stands prefer shallow stony or sandy soils; the pine-*V.vitis-idaea*-mixed herb forest type needs moist rich loamy soils; and pine-*V.myrtillus* stands are found on poorly drained sandy soils and loamy sands (Shumilova 1962). Another equally important pine forest type group is comprised of pine stands with herbaceous ground cover (*Calamagrostis*-mixed herbs, sedge, and iris). These pine stands develop on well-drained loam, weakly podzolic or soddy soils, as well as on ferruginous or carbonate soils, and are often subject to human impacts. Pine-lichen stands, the simplest in structure, and the most xerophytic, are confined mainly to sandy or skeletal soils. Pine-green moss stands (very often with minor components of larch and dark coniferous species) are a typical type of boreal forest and apparently represent one of the regeneration stages after crown fire (Shumilova 1962). Pine-sphagnum and pine-long moss forest types are represented by boggy stands of small extent.

Carbon Stocks and Fuel Complexes of Different Forest Types

According to the literature, pine forests cover 13.7 million ha in Central Siberia (Shimanuk 1962). These pine forests contain about 67 MgC ha^{-1}, with 44.5 MgC ha^{-1} in the overstory, 2.5 MgC ha^{-1} in the understory, and 20 MgC ha^{-1} in the forest floor plus litter (Alexeyev and Birdsey 1994). Thus regional carbon storage in pine forests is 615 TgC in the overstory, 35 TgC in the understory, and 273 MgC in the forest floor including litter (based on Alexeyev and Birdsey 1994). Tree crowns account for 20-23% of the overstory biomass. Mature and overmature pine stands account for 75% of the accumulated carbon, middle-aged stands for 21.5%, and logging-aged young stands for 4.5%. As was mentioned above, pine forests are represented by three main forest types: pine-lichen, pine-green moss, and pine-mixed grass stands. Other forest types occupy minor areas. In every stand, however, distinct sinusia of ground cover can be differentiated. By way of example, consider the phytocoenotic structure of two pine stands in Central Siberia (Figure 1).

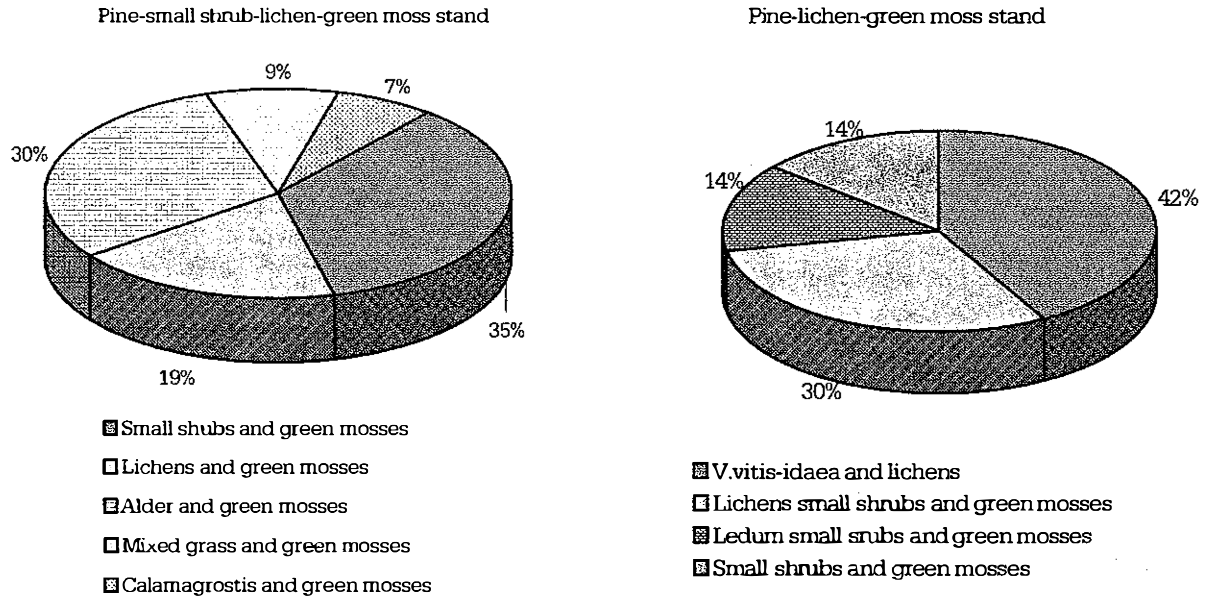

Figure 1. Phytocoenotic structure of pine stands.

Every pine forest type has its specific fuel complex. Structural fuel complex differences are due to edaphic conditions.

The ground fuel complex of pine-lichen stands is made up of various lichens, with *Cladonia* dominating, plus litter and forest floor (humus). Lichens, with included litter elements (needles, bark pieces, small branches, etc.), constitute a loosely packed fuel layer. A long fire-free period can result in accumulation of huge amounts of fuel in these pine stands. This greatly increases the potential for crowning, should a fire occur. Fire danger can remain high throughout the fire season in these stands.

Along with green mosses, the ground fuel complex of pine-green moss stands includes a layer of small shrubs that slows down fuel drying in summer. A deep humus layer acts the same way. This pine forest type is highly flammable for a shorter period during fire season, but fire hazard can increase greatly under drought. This pine forest type usually experiences steady surface fires that can become large and of high intensity during severely dry spells.

In pine-mixed grass stands, the ground fuel complex is represented mainly by litter and cured grass. Small shrubs make the fuel layer loosely packed. This forest type is usually highly flammable in the spring before new grass appears. As grass grows taller, fire probability decreases practically to zero in these stands, since green grass and low shrubs prevent fine fuel from rapid drying and much heat is needed for water to evaporate from them (Kurbatsky and Ivanova 1987).

We estimated carbon accumulation in four pine stands differing in site quality class and belonging to different forest types using the data of Alexeyev and Birdsey (1994). The largest amount of carbon storage occurs in the pine-small shrub-green moss stand (Table 1). Structurally, the proportion of carbon in the overstory is the largest. The heaviest ground fuel load is estimated for the pine-lichen stand, where there was no fire for 37 years.

FIRE IN BOREAL FORESTS OF CENTRAL SIBERIA

Fire has a profound influence on boreal forests and has shaped today's forest landscapes. One can find evidence of at least two or three fires in any forest stand of central Siberia. That fire has been an integral part of Central Siberian forest ecology since prehistoric times is confirmed by the presence of coal (charcoal) in soil and peat (Koshkarova 1986, Firescan 1996). Based on data from peat and charcoal samples, climate is believed to have been drier and warmer in this region 8300 to 8000, 5000 to 6000, and 3200 years ago. High forest fire activity evidence is abundant for these periods (Koshkarova 1986). Records of forest fires have also been observed in lake sediments from between 5000-4200 and 3400-2800 BC (Firescan 1996).

Table 1. Carbon stocks in different pine forest types.

Forest type	Site location	Stand age (yr)	Site quality class	Carbon stocks (Mg ha^{-1}) Overstory	Live ground cover	Litter and humus	Total
Pine-*V. vitis-idaea*-lichen	60°38'N 89°44'E	180	5	55.70	5.30	4.38	65.38
Pine-lichen	60°45'N 89°25'E	130-180	3	72.80	7.95	8.80	89.55
Pine-small shrub-green moss	58°41'N 90°18'E	160	4	111.40	3.36	9.84	124.60
Pine-*V. vitis-idaea*-green moss	56°13'N 93°00'E	100	2	95.30	0.95	17.1	113.5

Siberia is a huge area encompassing many climatic and vegetation zones. This results in a wide diversity of local climates and site conditions, and interannual variations in climatic patterns promote fire occurrence in different parts of the region every year. Three types of fire seasons are characteristic of Central Siberia (Valendik 1990):

- a short, continuous fire season (very high fire danger during 1-3 months) is common in northern and central taiga forest;
- a long fire season with periodical fires over a 4-6 month period prevails in the southern taiga;
- a double-peak fire season (high fire occurrence in spring and fall) occurs in the southern mountain areas.

Forest fires are unevenly distributed across the area where maximum fire occurrence is observed in light coniferous (pine and larch) forests. Lightning fires are the cause of 30% of all forest fires in Central Siberia while the majority of fires are due to human causes.

The 1947-1999 annual dynamic of the total number of large forest fires shows a cyclic character, with high interannual variability (Figure 2). Large forest fires, which are responsible for 90% of the total area burned, occur with a background of numerous small fires during extreme fire seasons. High numbers of large fires were recorded in 1950, 1953, 1956-1958, 1962, 1964-1965, 1967-1969, 1976-77, 1984-85, 1989-1990, 1994, 1996-97, and 1999. The most devastating fires occurred in 1957, when all of central Siberia (50°-70°N) was under severe drought. Starting in about 1990, the average number of fires recorded per year has increased substantially.

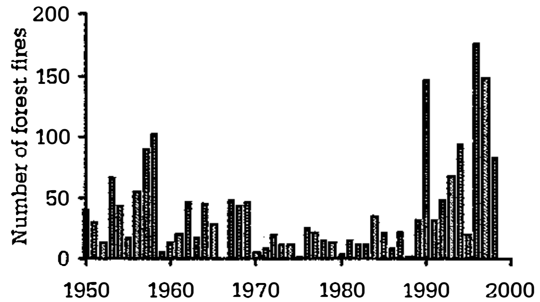

Figure 2. Large forest fire dynamics in Central Siberia (1950-1998).

The periodicity of extreme fire seasons depends on drought frequency and latitude. Drought events promoting extreme fire situations are usually induced by warm, dry air masses coming from Central Asia, Mongolia, and the central part of eastern Siberia. The southern (50°-55°N) and northeastern (56°-62°N) parts of Central Siberia are notable for experiencing the most frequent drought, 7-10 and 5-7 times a decade, respectively. In the rest of the area, 2-3 drought events are observed per decade, and sometimes drought can occur in two successive years. Severe drought, however, is rare.

Spatial and Seasonal Patterns of Fire in Pine Forests of Central Siberia

Fire interval, extent, and severity in a given forest depend on its vegetation pattern, as well as on forest type, location of the forest, remoteness, and climatic patterns. In general, forest fire periodicity is related to the sequence of dry and wet years, seasonal and daily weather changes, forest vegetation dynamics, geographic location of forest stands and human activity (Melekhov 1971). The frequency of severe fire seasons has grown over the several past centuries which is, in part, due to an increase in economic activity.

Patterns of past fire occurrence were estimated in Central Siberia using dendrochronological methods. This allowed us to reconstruct a multi-century history of fire regimes. These data were compared with current fire regimes to obtain spatial and temporal estimates of the contribution of fire to the carbon balance. Reconstruction of past fire regimes was based on fire chronologies built for 20 sites samples along the Yenisey meridian. Sites represented different forest types and vegetation zones of central Siberia. To establish times of past fire occurrence using fire scars, we compared records of fire dates with chronologies from neighboring trees and cross-dating with composite tree-ring chronologies available for the region in question (Swetnam and Madany 1989). Mean fire interval was calculated from the ratio between the tree-ring chronology duration obtained for a given site and the number of fires established for this period of time (Arbatskaya and Vaganov 1977). An example of composite tree-ring chronologies for 2 sites are shown in Figure 3.

Estimates for mean fire interval are reported for each major vegetation zone along a latitudinal gradient (Table 2). Mean fire

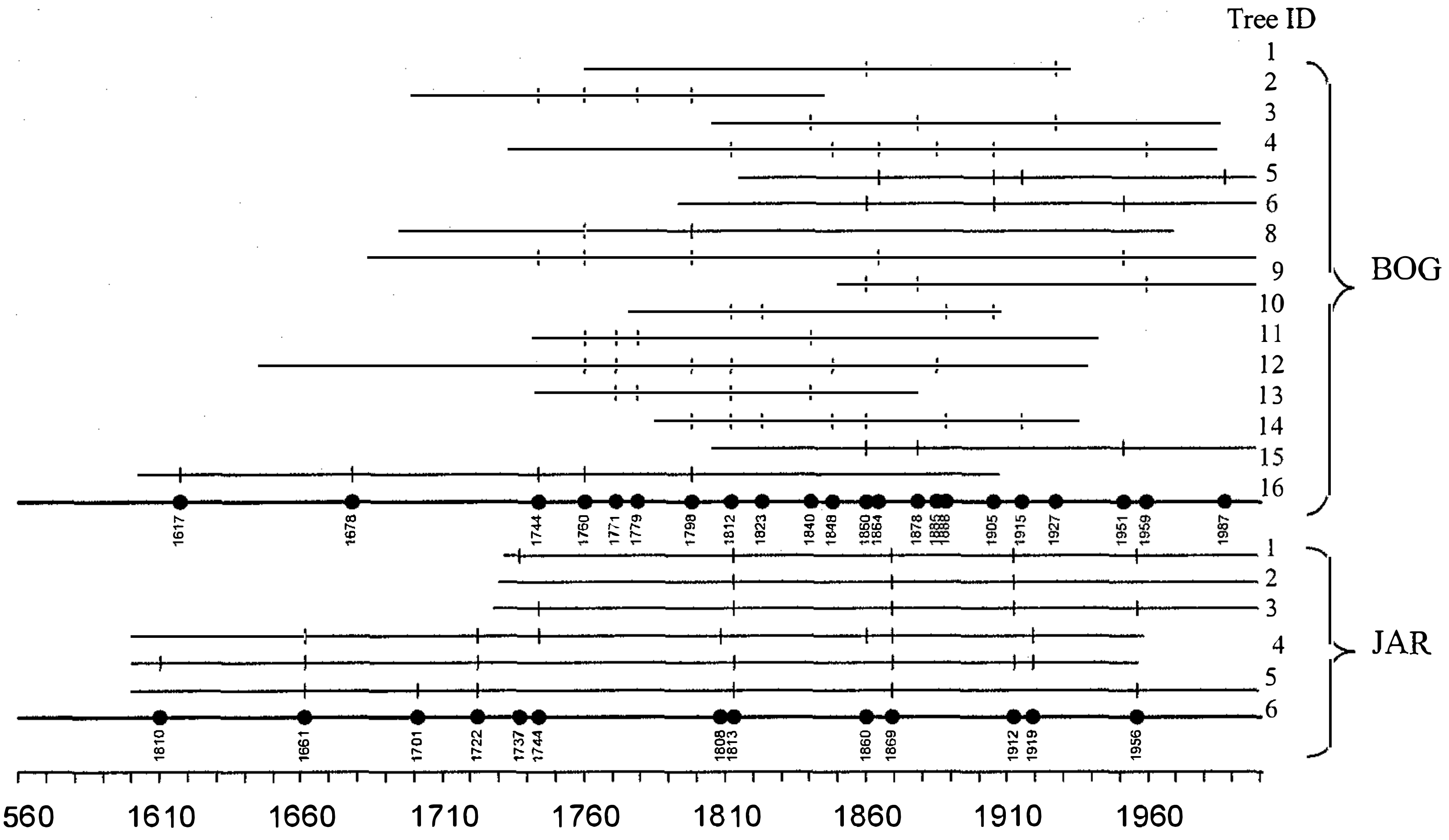

Figure 3. Tree-ring chronologies of fire in pine stands.
BOG – (58°41′ N 90°13′ E). Pine-small shrub green stand (non-isolated sites);
JAR – (60°38′ N 89°44′ E). Pine-*V. vitis idea*-lichen stand (isolated sites);
—●– Composite.

intervals decrease from north to south. In the northern taiga sub zone, a mean fire interval of 53.3 years was estimated for larch stands, while it was 45.0 years at the northern boundary of the pine forest extent (a sample pine-*Vaccinium vitis-idaea*-green moss stand). Mean fire intervals as a function of forest type calculated for the northernmost pine-*V.vitis-idaea*-green moss, pine-*V.myrtillus*, and pine-lichen-*V.vitis-idaea* stands located in the central taiga subzone in the Turukhan and Yeloguy interstream area, were 45.0, 25.7, and 21.0 years, respectively.

Table 2. Mean fire intervals in pine stands of Central Siberia.

Vegetation zone	Mean fire interval (years)
Northern taiga subzone	45-53
Central taiga subzone	20-40
Southern taiga subzone	
north	24-38
south	12-21
Subtaiga zone	6-10
Forest-steppe zone	8-12

In the central taiga subzone, mean fire intervals were established to be shorter on non-isolated sites where fires can easily spread from bigger areas, as compared to those on sites isolated by bog areas. Mean fire intervals of 34.5 to 86.3 years are characteristic of isolated sites, whereas mean fire intervals of 20-40 years are more common in non-isolated sites. These values are confirmed by data published by other authors (Kireyev and Furyaev 1977; Furyaev 1996; Swetnam 1996). Mean fire interval ranged from 24.0 to 38.3 years in pine shrub and pine-green moss forest types in the northern part of the southern taiga subzone. Mean fire intervals in pine stands of the West and East Sayan Mountains were similar and varied from 13.8 to 27.5 years. The same range in mean fire intervals is characteristic of the southern taiga subzone.

Shorter mean fire intervals in the south (12.7-21.4 years) are associated with more recreational activity, high temperatures, and high fire hazard. Very short mean fire intervals (6.6-9.5 years) in the subtaiga pine-mixed herb stands along the Yenisey River are due to a high degree of anthropogenic influence. In the Krasnoyarsk and Kansk forest-steppe areas, where pine stands surround towns and villages, fires occurred very often as a result of intense human activity, and mean fire interval ranges from 8.0 to 12.6 years.

Fire interval duration is thus controlled by latitude, site conditions, and human influence. Mean fire interval also depends on forest type. For example, pine-lichen stands on loam and sandy soils burn most frequently and severely. Mean fire interval here is less than half that in pine-*V.vitis-idaea*-mixed herb and pine-green moss stands.

Our study and data found in the literature (Swetnam 1996; Firescan 1998) suggest that, aside from areas of intense human activity, these spatial fire dynamics have lasted for the past 5-6 centuries and are representative in terms of both mean fire interval and seasonality of fire occurrence. Intense crown fires occurring with a background of numerous surface fires were characteristic of past fire regimes. Surface fire frequency was, to a large extent, determined by interrelation of landscape elements. Isolated sites, such as islands among bogs, burned at a 45-90 year return interval (Figure 3). Non-dissected areas were subject to fire about every 20-40 years. Fires tended to occur mostly in late June-early July, immediately after snowmelt, or with the onset of storm activity. In the Angara Region, Buzikin (1975) observed that fire governed forest age structure to a large extent. Disastrous crown fires usually stimulate development of even-aged light coniferous stands, while surface fires are responsible for uneven-aged stand occurrence. High-intensity crown fires, intermixed with surface fires, have brought about a variety of forest woody species generations to form stands of complex age structure.

Fire Intensity and Severity in Boreal Forests

According to data from the Krasnoyarsk forest protection airbase, the total annual area burned by large forest fires in the period between 1991 and 1999 varied greatly from year to year (Table 3). This variability is attributed to dry conditions and inadequate forest protection funds. While large fires (greater than 200 ha) account for only 2-12% of recorded fires, they are responsible for 63.7 to 96.6% of the total area burned. Surface fires are most common and cover up to 80% of the total area burned. In large fire events, about 17.2% of the area burns in crown fires. This agrees

Table 3. Number of forest fires and area burned in Central Siberia (Krasnoyarsk forest protection airbase, 1991-1999).

Year	Number of forest fires	Area burned, thou ha	Number of large fires	Large fires	Crown fires
				% of the total area burned	
1991	878	36.2	31	77.2	-
1992	1246	73.3	48	77.9	8.2
1993	1314	110.7	64	88.0	17.2
1994	1504	143.0	93	85.5	6.6
1995	970	27.8	20	63.7	4.2
1996	1412	316.7	176	96.6	7.1
1997	1337	126.4	147	86.0	*
1998	1099	32.1	82	67.1	*
1999	1772	132.0	174	65.3	*

* no crown fire data

with published data on fire occurrence across Russia (Korovin 1966). Surface fires of varying intensity are common in pine forests. Crowning is only possible in mountain forests, on round-top hills, where young stands have developed closed canopy, or under severe drought or wind conditions.

Contribution of Fire to the Carbon Balance in Pine Forests

It is widely recognized that fire can contribute considerably to boreal forest carbon sources or sinks in the following ways (Kasischke, Christensen and Stocks 1995):

- direct release of carbon to the atmosphere from biomass burning;
- plant material conversion into charcoal, which does not disintegrate through decomposition;
- postfire biogenic emissions and decomposition;
- postfire vegetation recovery;
- changes of thermal regimes of organic and mineral soil layers;
- increase in soil nutrients available for plants;
- stand age structure changes.

The magnitude of fire influence on carbon stocks through each of the above, depends on fire intensity and severity.

We estimated the potential contribution of fire emissions in the Krasnoyarsk Region to the boreal pine forest carbon budget, depending on fire intensity and forest type. Krasnoyarsk Forest Protection Airbase data indicate that the area burned between 1991 and 1999 amounted to 998.2 thousand ha in central Siberia, or about 100,000 ha annually on average. 60% of fires here occur in pine forests (Ivanov 1996) accounting for 60,000 ha of the total burned area. This area can increase considerably in extreme fire seasons. Moreover, these statistics may substantially underestimate the burned area (see Conard and Ivanova 1997), in part because fire protection does not cover the entire area and many fires are not recorded. Mean fire interval is known to vary with forest vegetation zone from 20 to 40 years in central Siberia. Here, mean fire intervals range from 24 to 38 years. If we assume the total pine forest area in central Siberia equals 13.676 million ha (Shimanuk 1962), then, with a 40-yr mean fire interval, the estimated annual pine area burned will be 342,000 ha, that is 5 times as much as official statistics. However, intensive development of forest fire protection in 1940's to 1990's caused some increases in fire intervals. This is confirmed, in particular, by the absence of fire scars in trees during past few decades. Thus, fire interval decrease is observed in areas under strong anthropogenic impact and, at the same time, fire intervals tend to increase in more remote, but protected forest areas.

Consumption of fine fuels is known to depend on fire intensity. It is estimated to vary from 16% to 100% in North American boreal pine forests (Stocks 1991; Kasischke, Christensen and Stocks 1995). Fire ignition pattern and heavy ground fuel loading led to a high-intensity crown fire, even in absence of a fuel ladder, during a 1993 fire experiment in a pine-lichen stand in central Siberia (FIRESCAN 1996). This fire consumed an average of 48 to 80% of the total ground fuel, with complete lichen layer consumption and partial forest floor consumption. A fast-moving surface lightning fire that occurred in a similar forest type in the same region, consumed only the lichen layer leaving the forest floor intact. In terms of weight, it was some 40% of the total ground fuel load.

Estimates of carbon release from boreal forest fires of varying intensity (surface-crown fire combinations) and different crown scorch levels indicated a potential 4-fold range in emissions as the contribution of crown fire increased from 10 to 70 percent (Conard and Ivanova 1997). These estimates were based on the assumption that crown fire consumes all fine fuel elements (needles, small branches, understory vegetation, and litter) that make up 40% of the total aboveground fuel. Fast-moving low-intensity surface fires were assumed to consume 50% of regrowth and understory, as well as 10% of ground fuel. High-intensity surface fires were assumed to reduce understory vegetation and ground fuel by 90% and 50%, respectively. Based on the percentages above, one can derive the following carbon emission estimates for pine forests: 18.8 Mg ha^{-1} in crown fires, 12.2 Mg ha^{-1} in high-intensity surface fires, 3.2 Mg ha^{-1} in low-intensity surface fires. We used these estimates to compare potential average annual emissions in the Krasnoyarsk Region by combining estimates of burned area and percent of area burned in crown fire (Table 3) with emission estimates from low- and high-intensity surface fires. Crown fire combined with a highly intensive surface fire gives a 3-fold increase in emission amount, when compared to crown fire combined with a low-intensity surface fire (Figure 4). This illustrates the potentially wide interannual differences in emissions that can be expected as a function of variability, both in burned area and in fire intensity, between fire seasons. While these estimates of the magnitude of interannual variability are probably reasonable, it is important to recognize that our calculations are based on official fire statistics and are likely to be a 4 to 10-fold underestimate of actual emissions (see Conard and others, in press).

Estimating emissions from fires of varying intensity requires consideration not only of fire intensity and fuel consumption rate, but also of forest type and site conditions

For example, in a relatively high-intensity experimental fire conducted in a pine-lichen stand in 1993 (FIRESCAN 1996), crowning occurred in 57% of the study area. Canadian specialists estimated consumption of total ground and crown fuels to be 60% and 20% respectively, in this pine forest type, with an estimated carbon emission of 12.06 Mg ha^{-1}. This value is similar to the one we used when estimating emissions from high-intensity fires. On the other hand, in a fast-moving surface lightning fire that occurred in a pine-*V.vitis-idaea*-lichen stand in 1998, shrub, lichen, and surface litter layers burned completely, while only 20% of the humus was consumed. We estimated emissions at 2.95 Mg ha^{-1}, which is quite similar to the value we used above for estimating emissions from low-intensity fires. Based on fine fuel loading and the consumption levels in the previous two examples, emissions from high-intensity fire would be 9.52 Mg ha^{-1} in a typical pine-small shrub-green moss stand (e.g. Table 1), whereas those from low-intensity fire would be 5.33 Mg ha^{-1}. For a pine-*V.vitis-idaea*-mixed grass stand these estimates are 13.0 and 4.37 Mg ha^{-1}, respectively. This suggest that carbon emissions vary with vegetation type and site conditions and the latter two should be considered when calculating forest fire emissions.

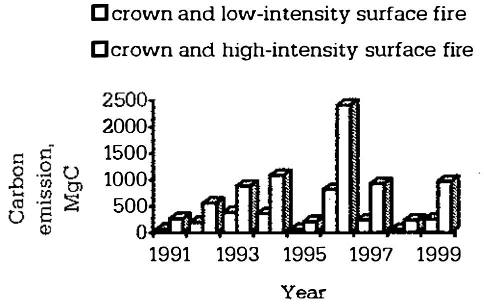

Figure 4. Emission of carbon due to fire in forests of Central Siberia.

CONCLUSION

Our estimates indicate that carbon emissions in pine forests do not depend solely on fire intensity, but also on vegetation type. Fire emissions vary year-to-year in forests of central Siberia and may range over more than an order of magnitude depending on the severity of the fire season. In this paper, we have presented generalized estimates of emissions for pine forests of central Siberia based on a relatively small number of sample sites. Clearly, accurate estimates of fire contribution to the carbon balance in pine forests require improved data on burned area and on fire type and severity, as well as information on specific stand characteristics and vegetation type.

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CARBON BALANCE OF ETHANOL FROM WOOD: THE EFFECT OF FEEDSTOCK SOURCE IN CANADA

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ABSTRACT

Delucchi's full fuel cycle greenhouse gas (GHG) model was used to calculate the amount of GHG emissions from production and combustion of wood-derived ethanol under the following three land-use scenarios: 1) short rotation forestry on land previously covered by pasture (70%), agricultural fields (15%) and forest (15%); 2) short rotation forestry on previously forested land (100%), and 3) using sawmill wood residues with no energy-dedicated wood plantation. The modelling results indicated that in comparison with pure gasoline, utilisation of a gasoline blend containing 10% wood-derived ethanol under the scenarios (1), (2) and (3) could reduce GHG emissions by 7.9%, 1.0% and 6.3%, respectively. The difference in the level of emission reduction is primarily due to the changes in the amount of above-ground biomass and soil organic carbon under each land-use scenario.

INTRODUCTION

The increase in concentration of atmospheric carbon dioxide (CO_2) has been attributed to three major human activities, fossil fuel combustion, cement manufacturing, and disturbance of natural ecosystems through deforestation and practice of intensive agriculture (Lal and Bruce, 1999). According to the National Energy Board of Canada (1999), in 1997 the transportation sector made the largest contribution (28%) to the total greenhouse emissions generated from consumption and production of energy in Canada (Figure 1). The energy sector emissions consist of about 90% CO_2, 7% CH_4 and 3% N_2O (National Energy Board, 1999).

The projections by the Canadian National Energy Board indicate that there will be a steady rise in the amount of energy-related GHG emissions in Canada over the next two decades, pushing the total emissions to 720 mega-tonnes CO_2 equivalents *(CO2-e)* in 2025, a 25% increase compared with 1997. In order to meet its commitment to the Kyoto Protocol (1997) proposed by the United Nations Framework Convention on Climate Change, Canada has to reduce its GHG emissions by 6% by the period 2008 to 2012. Considering the significant contribution of the transportation sector to these emissions, improving the quality of road vehicle fuels is one of the possible ways to reduce Canada's GHG emissions.

It has been shown that blending gasoline with ethanol produced from wood and other types of lignocellulosic feedstocks can help reduce the GHG emissions from motor vehicles (McCloy and O'Connor, 1998). Net sequestration of carbon dioxide during the growth of feedstocks and the high oxygen content of ethanol with a greater combustion

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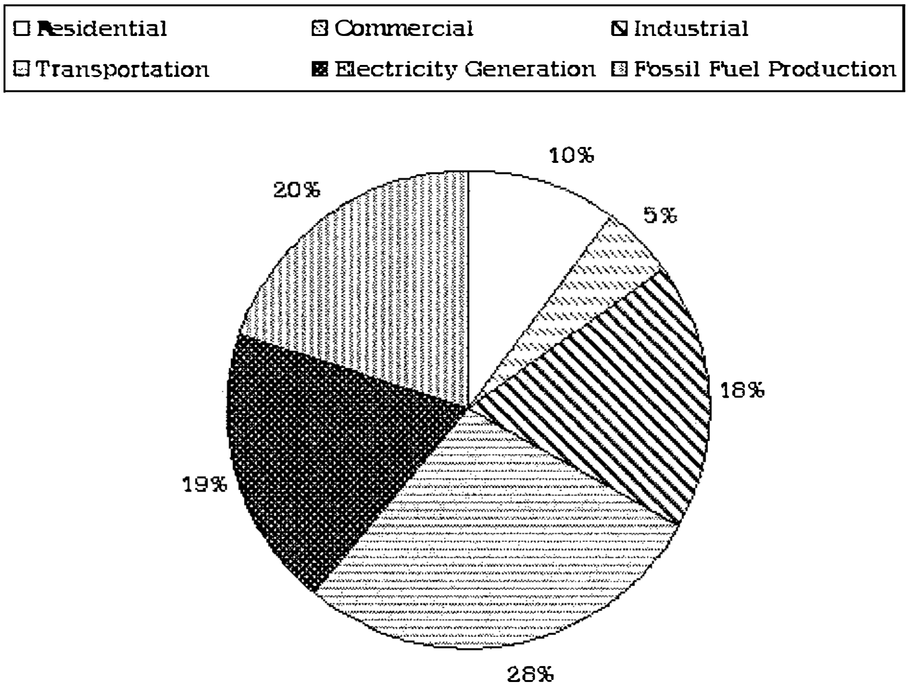

Figure 1. Canadian GHG emissions by sectors. Total of 583 Mt (CO_2-equivalent).

efficiency are the two primary reasons for the lower emissions from ethanol and ethanol-blend fuels (Wyman, 1996). Potential sources of feedstock for the bioconversion process include short rotation woody crops, grasses and agricultural residues (wheat straw, corn stover, rice husk, etc.), starchy crops as well as sawmill, pulp mill and woodland wood residues. In comparison with agricultural practices used to produce starchy row crops (e.g., corn), short rotation plantations of high-yield woody crops (e.g., poplar and willow) are generally favoured, as they require less frequent tillage and lower amounts of chemical input, i.e., fertilizer, pesticides and herbicides (IEA Bioenergy, 1999). The lower tillage and chemical input requirements of short rotation plantations causes less soil disturbance and lowers the release of organic carbon from soil (Delucchi, 1998).

Wood residues are produced throughout Canada in very large amounts, i.e., 18.8×10^6 oven dry ton/year (Hatton, 1999). Currently, 73% of these materials are used to produce energy and other value added products, such as medium density fibreboard and oriented strand board, while 27% remain un-utilised and are often disposed of by landfilling or incineration (Hatton, 1999). The current disposal methods are not devoid of problems. It is known that landfilling can result in contamination of groundwater and surface run-off, and smoke from wood incinerators can cause aesthetic and health problems (McCloy and O'Connor, 1998). The fine particulate matter produced during wood burning has been linked to the occurrence of respiratory health problems in the province of British Columbia (McCloy and O'Connor, 1998). While wood residues produced by sawmills and pulp mills are readily accessible and can be transported to a bioconversion plant, the residues generated by forestry operations in the woodlands (e.g., thinnings, branches, ends, low-value trees, etc.) frequently remain on the logging site due to economic and ecological constraints. There are conflicting opinions about the ecological significance of woodland residues, some suggest that these materials act as a "long-term stabilizing storage pool of nutrients" and have to remain on the forest floor to decay, while others have shown that these residues do not play a significant role in recycling of soil nutrients (e.g., nitrogen and phosphorus) in the coniferous forests (Laiho and Prescott, 1999).

Previous studies have shown that the type and origin of the feedstock (e.g., composition, source, processing requirements, ease of access, etc.) have a significant impact on the overall efficiency and economic feasibility of the bioconversion process (Levelton Eng., 1999). To gain some insight into the effect of land-use change on the generation of GHG emissions from transportation fuels, we used Delucchi's full-fuel cycle model developed at the University of California, Davis (Delucchi, 1998). We calculated and compared the emissions generated throughout the entire life cycle (production-delivery-combustion) of gasoline as a transportation fuel with those resulting from a fuel blend containing 10% bioethanol and 90% gasoline. The bioethanol used in the fuel blend was assumed to have been produced from one of the following sources:

1. Base case: Short rotation forestry on land previously covered by pasture (70%), agricultural fields (15%) and forest (15%);
2. Modified land-use: Short rotation forestry on previously forested land (100%), and
3. Sawmill wood residues: No new wood production.

GREENHOUSE GAS EMISSIONS MODELLING METHODOLOGY

The model used in this study was developed by Mark Delucchi at the Institute of Transportation at University of California in Davis, CA, USA. It is in spreadsheet format and uses a large number of assumptions, which have been described in detail in a very comprehensive users' manual (Delucchi, 1998). The objective of the present work was to shed some light on the issue of land-use change and its impact on the mitigation of greenhouse gas emissions using a comprehensive tool (e.g., Delucchi's model). Thus, it was considered that any extensive discussion on the details and/or validity of the assumptions, or the description of any mathematical procedure used by the model would be beyond the scope and purpose of this manuscript, and therefore, have not been discussed here.

In 1998-1999, the Delucchi model was expanded and modified to include Canada, and has ever since been used in multiple studies for Agriculture & Agri-Food Canada (Levelton Engineering, 1999a and b) and the Transportation Table (Levelton Engineering, 1999c). The Canadian version of Delucchi's model has been further developed and is considered to be a rigorous life-cycle analysis of greenhouse (CO_2, CH_4 and N_2O) and non-greenhouse gases from alternative motor fuels (Levelton Engineering Ltd. and (S&T)[2] Consulting Inc., 1999b).

In this paper, the total emissions are reported as *gram equivalent* CO_2 (CO_2-*e*), in accordance with the model. It is worth noting that carbon dioxide (CO_2) is often the major component of the gaseous emissions from road vehicles, while methane (CH_4) and nitrous oxide (N_2O) rank second and third. On a molecule for molecule basis, however, methane and nitrous oxide have higher Global Warming Potentials (GWP), 21 and 310, respectively. The CO_2-equivalent is calculated as the weighted sum of carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) as shown in Eq. [1];

$$CO_2\text{-e (g)} = (\text{g } CO_2) + (310 \times \text{g } N_2O) + (21 \times \text{g } CH_4) \quad [1]$$

where 310 and 21 are, respectively, the *100-year global warming potentials* of N_2O and CH_4, as prescribed by the Intergovernmental Panel on Climate Change (IPCC, 1996).

The model divides the full life cycle of a fuel into the following four sub-cycles: feedstock production (crude oil or biomass), fuel production (ethanol or gasoline), fuel delivery, and fuel combustion (Figure 2):

1. Feedstock Production: This stage involves all the field operations including tillage, irrigation, maintenance and harvesting. The emissions associated with the production, transportation and application of necessary chemicals, i.e., fertilizers, herbicides, insecticides, as well as those resulting from fossil fuel energy consumption during irrigation, harvesting, and biomass harvest and packing are also included. The changes in soil carbon and biomass are calculated over 15 years. The model uses a fixed wood production yield of 4.8 ton/acre, and treats the plant biomass build-up during the growth phase as a one-time, short-term, negative emission of CO_2. The fertilizer and chemical requirements (lb/ton wood) of a short-rotation plantation were assumed to be as follows, nitrogen: 2.14, phosphorus: 1.71, potassium: 1.18, lime: 41.3, and pesticides: 0.11 lb/ton wood produced. For each land use scenario, the model assumes the following biomass and soil (kg C/m^2) carbon content values: for forests, 11 and 13; for short rotation plantations, 8 and 5.8; for pastures, 6.8 and 0.8; and for general

agriculture, 6 and 0.4 kg C/m^2. This sub-cycle also includes the emissions generated by the transportation and delivery of the feedstock to a bioconversion plant.

2. Bioconversion Process: This sub-cycle includes the GHG emissions associated with the industrial production of ethanol from wood in a bioconversion plant. These emissions can result from plant unit operations, such as fermentation, production of plant utility (heat, steam and electricity), and disposal of ash produced in biomass incinerators. The GHG emissions resulting from the production of various chemicals (acid, alkali, H_2O_2, etc.) and biochemicals (enzymes) used in the bioconversion process are also included. The co-products of the bioconversion process, such as excess electricity, lignin derivatives and acetic acid also provide some emissions savings, and have been taken into account. The lignin is used for electricity generation, and does not factor into the emissions except as a source of electricity, and is credited with the displaced emissions from electricity production by natural gas combustion. The acetic acid is credited with the displaced emissions from conventional acid production.
3. Fuel Ethanol Delivery: Emissions resulting from blending, storage, transportation and distribution of ethanol to fuel stations as well as those released during fuel dispensing are included in this sub-cycle.
4. Fuel Ethanol Combustion: This category includes emissions resulting from vehicle operation (combustion, evaporation, leaks, etc.) and manufacturing (materials and assembly technology). The type of material and manufacturing technology are affected by the choice of fuel.

Further details on various assumptions and technology alternatives used by the model can be found in Model's Users' Guide (Delucchi, 1998).

RESULTS AND DISCUSSION

The GHG emissions, expressed as CO_2-e, generated during the production and use of gasoline and ethanol are shown in Table 1. It appears that the emissions from gasoline and ethanol differed significantly in issues related to land-use change, due to variations in above and below ground biomass, irrigation and cultivation practices as well as harvesting frequency and intensity (item 1); fertilizer production and use (item 2); leaks & flares from the processing operations (item 8); and emissions displaced by co-products, lignin and electricity (item 9). Since extraction and recovery of oil (feedstock for gasoline) is not through biomass cultivation, the land-use change and fertilizer production emissions for gasoline were set to zero.

In the base case scenario, converting an area covered mostly by pasture (70%) and agricultural land (15%) to a short rotation forest plantation, increased the above-ground biomass considerably and reduced the amount of tillage required to maintain the agricultural land. The modelling results indicated that the combination of higher biomass and less frequent occurrence of soil disturbance could produce a relatively large land-use credit and result in mitigation of CO_2 emissions.

However, in the modified land-use scenario, when the land previously covered by natural forest was converted to a short rotation forest, the tillage for site preparation, irrigation and harvesting operations had to be performed more frequently, and a considerable amount of agrochemicals (fertilizers, pesticides and herbicides) had to be used to improve plantation productivity. Both of these items increased the level of carbon emissions. In comparison with natural stands, which have a cutting cycle of about 70 years, the energy-dedicated plantations are harvested once every 10-15 years. Therefore, the change in the land-use, from long-standing forest to short-rotation plantations, increased the net CO_2 emissions, as reflected in Table 1. In summary, it appeared that the increase in CO_2 emissions in the plantations was due to a combination of:

1. reduced above-ground biomass as short-rotation trees contain less biomass in comparison with mature natural stands and,
2. lower soil carbon content in the plantations as these areas are harvested more frequently and receive more chemical inputs in the form of pesticides, herbicides and insecticides.

In the case of sawmill wood residues (scenario 3), wood production was assumed not to have any bearing on energy production, since the wood was produced for commercial purposes with no intention of serving as a

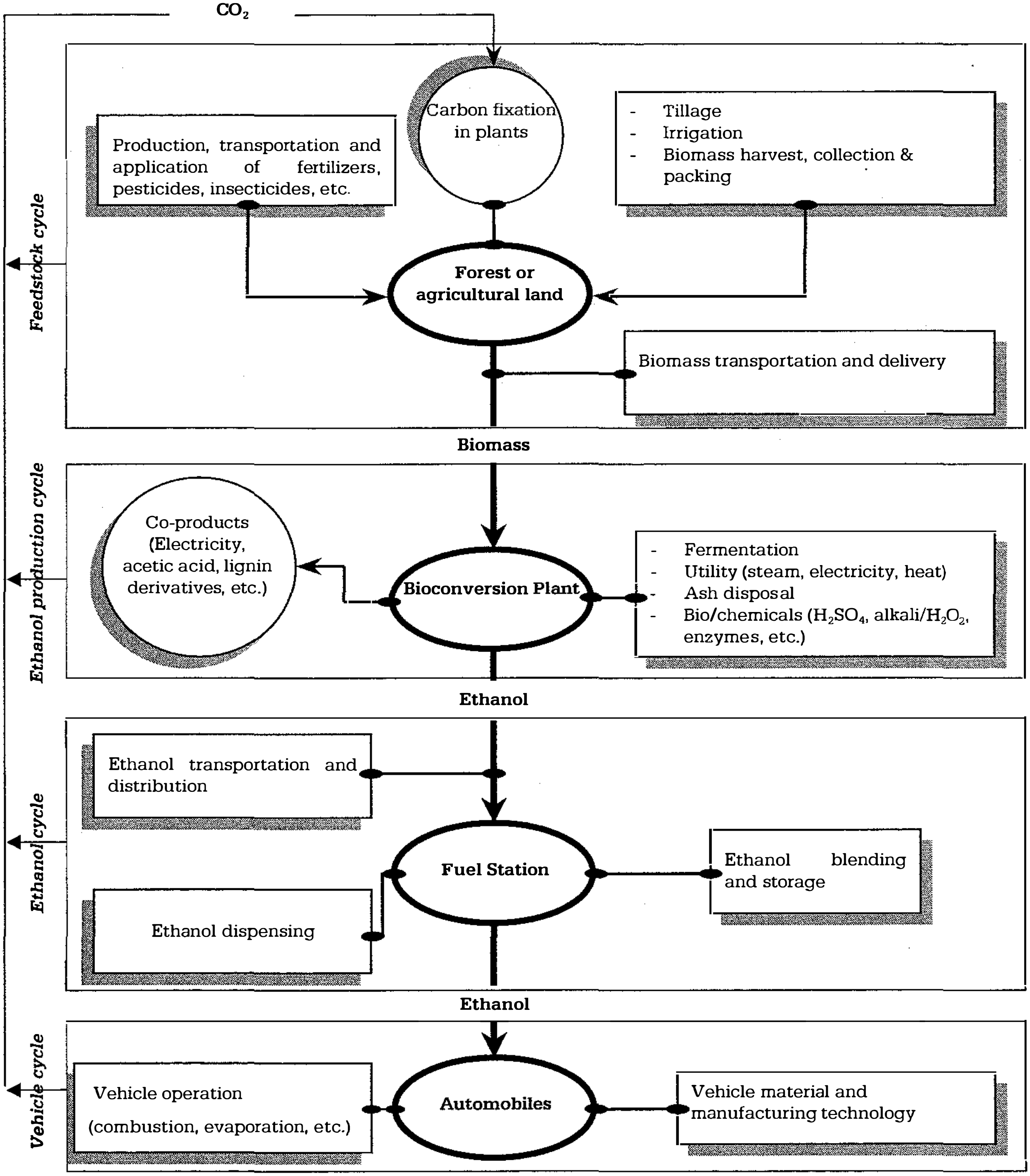

Figure 2. Bioethanol Full Life Cycle. Atmospheric carbon sources are represented by rectangles (•) and carbon sinks by circles (O).

Table 1. The effect of land-use scenario on the amounts of greenhouse gas emissions ($g\ CO_2$-e/ 10^6 BTU energy delivered) produced at different stages of a wood-to-ethanol cycle.

Life cycle step	GASOLINE	ETHANOL* Base Case	Modified Land-use	Sawmill Wood Residues
1. Land use change	0	-27,772	71,890	0
2. Fertiliser production	0	2,037	2,037	0
3. Feedstock recovery	8,219	5,607	5,607	5,607
4. Feedstock transport	371	2,801	2,801	56
5. Fuel production	8,755	8,967	8,967	8,967
6. Fuel storage & distribution	774	1,534	1,534	1,534
7. Fuel dispensing	160	165	165	165
8. Leaks & flares	1,921	0	0	0
9. Emissions displaced by co-products	0	-8,461	-8,461	-8,461
Total**	20,200	-15,122	84,540	7,868

*Base Case: Short rotation forestry on a land previously covered by pasture (70%), agricultural fields (15%) and forest (15%);
Modified Land-use: Short rotation forestry on a previously forested land (100%),
Sawmill Wood Residues: No wood production.
**This excludes the emissions generated by fuel combustion in vehicle engines.

feedstock for energy production. Therefore, in handling wood residues as the bioconversion feedstock, no emissions related to tillage, biomass change, or fertilizer/chemical use was considered. The alternative method for the disposal of wood residues was considered to be incineration at the sawmill, and since the resulting emissions would be from wood biomass they were not be counted in an emissions inventory.

Leaks and flares are produced by the oil refining processes during the production of gasoline, and were set to zero for the wood-derived ethanol. It was also assumed that the bioconversion plant could produce saleable excess electricity as a co-product. By replacing the high efficiency natural gas-derived grid electricity, this co-product was able to generate some emission credits for the bioconversion plant.

The total emission figures in Table 1 indicated that ethanol production and utilisation from wood generated by short rotation forestry on a land previously covered by pasture and agricultural fields (Base case) produced the lowest level of emissions. The sawmill wood residue and modified land-use scenarios followed the base case. It is important to note that the total emission figures in Table 1 exclude the GHG emissions from fuel combustion. In the case of gasoline, an additional 68,150 $g\ CO_2$-e/10^6 BTU should be added to the total emissions to account for the greenhouse gases resulting from gasoline combustion in road vehicle engines, hence bringing the total amount of emissions to 88,350 $g\ CO_2$-e/10^6 BTU. In the case of wood-derived ethanol, however, the combustion emissions need not be added as it was assumed that growth-phase photosynthesis counterbalances the emissions released by the combustion of the biofuels. Any difference between photosynthesis and combustion is reflected as changes in carbon stocks on the land. This clearly indicated how the origin of the feedstock and land-use change could impact the total net emissions from alternative fuels. It appeared that among the three feedstocks studied in this work, the wood produced from short rotation plantations was the best option.

Table 2 provides results for a similar analysis based on the number of miles travelled by a vehicle using a gasoline blend containing 10% ethanol. It was apparent that the base case (establishment of energy-dedicated plantations on pasture and agricultural land) was the most effective option

Table 2. Greenhouse gas emissions (g CO_2-e/mile travelled) at selected stages of a wood-to ethanol cycle for a 10% ethanol blend.

Life cycle step	GASOLINE	10 % ETHANOL BLEND*		
		Base Case	Modified Land-use	Sawmill Wood Residues
1. Land use change	0	-9.8	25.4	0
2. Feedstock & fertiliser production	42.0	41.3	41.3	40.6
3. Feedstock transport	1.9	2.7	2.7	1.8
4. Fuel production	44.7	42.8	42.8	42.8
5. Fuel storage & distribution	4.0	4.2	4.2	4.2
6. Fuel dispensing	0.8	0.8	0.8	0.8
7. Leaks & flares	9.8	9.0	9.0	9.0
8. Vehicle operation	370.8	369.4	369.4	369.4
9. Vehicle assembly and transport	5.6	5.6	5.6	5.6
10. Materials in vehicles	30.7	30.6	30.6	30.6
11. Emissions displaced by co-products	0	-3.0	-3.0	-3.0
12. Carbon in fuel sequestered from air	0	-23.9	-23.9	-23.9
Total	510.3	469.7	504.9	477.9
Emission reduction vs. gasoline (%)	-	-7.9%	-1.0%	-6.3%

*Base Case: Short rotation forestry on a land previously covered by pasture (70%), agricultural fields (15%) and forest (15%);
Modified Land-use: Short rotation forestry on a previously forested land (100%),
Sawmill Wood Residues: No wood production.

in terms of emissions reduction, followed by the use of sawmill residues and short-rotation tree plantation on previously forested landed. As compared to gasoline, the base case scenario (mixed land-use converted to short rotation plantation) had the greatest potential for GHG emissions reduction (8%). It is likely that the reduced emissions from a plantation is due to the fact that over the rotation period, plantation trees accumulate more above ground biomass (AGB) than do the annual or perennial herbaceous plants found in pastures and agricultural lands. Also, it is possible that converting an agricultural field, which requires intensive management (e.g., tillage, harvest and fertilization) on an annual basis, to a plantation that is harvested only every 15 years, can provide reduced emissions from the soil.

The production and use of ethanol from sawmill wood residues was shown to have a slightly lower potential for emission reduction (6.3%), when compared with short-rotation plantations established in lieu of pastures and agricultural fields (8% reduction potential). This could be attributed to the fact that in using sawmill wood residues, the model does not take into account the biomass production phase, therefore, for the wood residues there is no carbon credit associated with the soil carbon content and/or above-ground biomass, as there is in the case of short-rotation plantations.

The modified land-use scenario, converting forested land to short rotation plantations, appeared to have the lowest emission reduction potential (1%). It may be suggested that when a long-standing forest (~70 years) is converted to a short-rotation plantation that is harvested every 15 years, the average amount of above-ground biomass over time is reduced. Also, the higher frequency of tillage, harvesting operations, irrigation and chemical use in a plantation can result in reduced levels of soil organic content, and in turn, reduce the GHG emission reduction potential of short-rotation plantations as compared to long-standing forests.

CONCLUSION

The modelling results demonstrated that the extent of greenhouse gas emission reduction by replacing gasoline with wood-derived ethanol fuel was dependent upon the origin of the feedstock used. It appeared that short-rotation forestry on lands previously covered by grass or agricultural crops had the highest potential for reducing greenhouse gas emissions, while transformation of long-standing forests to short rotation plantations showed marginal benefits. Based on the assumptions used by the model, these differences are primarily due to the differences in the amounts of above-ground biomass and soil organic carbon under each land-use scenario.

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CO_2 AND WINTER TEMPERATURE EFFECTS ON NORWAY SPRUCE AND DOWNY BIRCH. A COMPARATIVE STUDY

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ABSTRACT

Three-year old Norway spruce seedlings (*Picea abies* (L.) Karst.) from three latitudinal populations were kept in a greenhouse during the winter (1992/93) at winter ambient temperatures and and winter temperatures raised by 4°C. The following spring the plants were moved into four open top chambers, pre-set at 350 and 650 μmol mol^{-1} CO_2. The experiment was repeated the next year (1993) and supplemented with one-year old seedlings of downy birch (*Betula pubescens* Ehrh.) from three populations. Birch from subarctic and southern populations in Norway increased their shoot elongation rates and biomass strongly at elevated CO_2. Due to earlier budbreak and a longer growth period, there seemed to be a positive influence of winter temperatures on shoot and biomass growth in the southern birch population while there was a slightly negative effect on plants from the Icelandic population. In spruce ecotypes high winter temperatures seemed to reduce shoot growth the following season. The observed growth reduction and the corresponding premature needle loss in high vs. low temperature treated seedlings was explained as a response to a temporary negative carbon balance in needles during exceptionally warm winters.

INTRODUCTION

One major topic in present-day research is the expected effect of the rise in greenhouse gas emission, particularly CO_2. According to climatological models (Callaghan 1993) a doubling of the CO_2 concentration would cause a 3-4°C increase in winter temperatures and about 2°C in summer temperatures. More recent predictions, e.g. the newly published IPCC WG1 Third Assessment Report (Albritton 2000) indicate that the strongest global heating will occur, and has recently (1976-99) occurred, at mid- and high-latitudes of the continents of the Northern Hemisphere, while there has been a slight cooling tendency in the North Atlantic and North Pacific Oceans. More variable and extreme temperatures are also expected (e.g. Katz and Brown 1992). One side effect of raised temperatures would be more rapid decomposition of soil organic matter and uptake rates of nitrogen from soil (Karlsson and Nordell 1987). The growth increase due to elevated CO_2 is often overestimated because of the CO_2 acclimation that takes place at the same time (e.g. Poorter 1993). Mortensen (1994) found a significant (20%) dry weight increase over the season in silver birch (*Betula pendula*) as a result of a doubling of the CO_2 concentrations, and a similar increase was found in *B. pubescens* (Mortensen 1995). Most CO_2 studies on growth are done in a controlled environment or in greenhouses at relatively high temperatures. Dark respiration is known to decrease with increasing CO_2 concentration (e.g. Mousseau 1993). The effect of CO_2 on plant production rates in growth chamber experiments has been positive, with a few exceptions (Eamus and Jarvis 1989, Poorter 1993). Most of the experiments have been carried out at considerably higher temperatures (15-30°C) and lower PAR than occurred under ambient field conditions at higher latitudes (cf. Mortensen 1998).

In the present study the combined effects of raised winter temperatures and increased CO_2 level on growth in some downy birch and Norway spruce provenances was investigated. Downy birch (*Betula pubescens*) and Norway spruce (*Picea abies*) were chosen as test plants because they are important species in northern forests of Fennoscandia. According to Kellomäki and Kolström (1992) a rise in temperature and CO_2 level is expected to favor more competitive species and provenances. Earlier studies (e.g. Karlsson and Nordell 1987, Skre 1991b, Mortensen 1995) have shown that southern and lowland downy birch provenances have a high ability for taking advantage of increased nitrogen availability in soil and of increased temperature and CO_2 levels. Birch is important also because it is the most common tree-line species in this area (see Skre 1991a). On the other hand, Norway spruce shows predetermined growth with a strong downregulation of photosynthesis at late season, after growth cessation (e.g. Skre and Nes 1996). In a model study based on field observations Beuker et al. (1996) found that the production of Norway spruce and Scots pine in southern Finland would decrease as a result of climate warming.

Because earlier studies indicated that coniferous and deciduous tree species have different abilities to take advantage of the expected rise in CO_2 level and winter temperatures, a comparative study was done on seedlings of Norway spruce and downy birch, grown in a heated greenhouse and open top chambers. The objectives of the present study were as follows:

1. to see how different winter temperature levels may influence growth in downy birch and Norway spruce seedlings during the following season.
2. to compare the responses of downy birch and Norway spruce seedlings from different origins to different temperature and CO_2 levels, in order to investigate possible adaptation mechanisms.
3. to see how nutrient level may influence the CO_2 and winter temperature responses on growth parameters.

In an earlier paper (Skre and Nes 1996) using two of the same spruce provenances as in the present study, responses in growth parameters to raised winter temperatures were investigated in plants grown at ambient CO_2 level. There was substantial needle loss due to exceptionally high winter temperatures during the 1992/93 season in 3-year old spruce seedlings, and downregulation of photosynthesis occurred in non-growing shoots. Similarly, growth and photosynthetic responses to raised CO_2 and winter temperature levels were studied in 1-year old seedlings of the three birch populations used in this study (Skre and Naess 1999). The aim of the present paper, however, is to conduct a comparative study of the responses of shoot and biomass growth in these two species to raised winter temperatures and CO_2 and nutrient levels during the same period. For this reason it has been necessary to include some previously published results.

MATERIAL AND METHODS

Norway Spruce

Three year old spruce seedlings, raised from mixed seed populations originating from three provenances, were transplanted into plastic containers (diameter 12 cm) with fertilized peat in the spring of 1992. For peat composition see Skre (1991a). The provenances were as follows:

H5 - Harz, Germany (52°N), 420-470 meters
C1 - Østfold, Norway (59°N), 50-150 meters
B3 - Hedmark, Norway (61°N), 250-350 meters

The spruce seedlings were placed under a birch canopy (light intensity 3-400 μmolm^{-2}s^{-1} PPFD) at Fana near Bergen (59°N, 5°E, 50 m). Two strengths of SUPERBA nutrient solution (Skre, 1991b), equivalent to 1 or 10 g m^{-1}year^{-1} nitrogen, were added once a week over the season (May-October). In late November the plants were moved into two greenhouse compartments, simulating ambient winter temperatures and winter temperatures raised by 4°C. A constant temperature difference was achieved by means of a differential temperature controller. Control plants were kept outside the greenhouse.

In late April 1993 the plants were moved into six open top chambers for six months (April – October) where they were exposed to normal (350 μmol mol^{-1}) and elevated (650 μmol mol^{-1}) CO_2 levels. The control plants remained outside the chambers. The chambers were made of 1 mm polyethylene sheet over a cylindrical aluminum frame (3.2 m diameter and 5 m height). CO_2 was supplied at a rate of about 1.5 l/min through perforated tubes placed at the bottom of the chambers. The temperature and CO_2 concentration in the open top chambers and in the greenhouse were monitored continuously using a Hartmann and Braun infrared gas analyzer for CO_2 monitoring and a Campbell data logger for monitoring CO_2-concentration, temperature and light. Light (Lambda LI-185B quantum sensor) and humidity was recorded continuously at the top of the canopy in each treatment unit. In addition, data from the light probe (Lambda LI-185B quantum sensor) and the humidity probe were recorded. The probes were placed in one open top chamber during the summer and in the greenhouse during the winter season. The average temperatures inside the open top chambers were 1.8°C above ambient, while the daylight PAR was reduced by about 20% relative to ambient light conditions.

Subsamples of plants and needles were taken for biomass determination throughout the experiment. Shoot length, stem base diameter and needle density on the side branches of the uppermost whirl were measured every two weeks throughout the growing period. The CO_2 treatment continued until early November 1993. The remaining plants, eight per replicate and with three replicates per treatment, were then moved into the two greenhouse compartments, where the winter temperature treatment was repeated. The CO_2 treatment was repeated during the 1994 season (April-October) to investigate the after-effect of the needle damage. Non-destructive measurements of needle density (see Skre and Nes 1996) and apical shoot length were carried out regularly during the season, and in October all remaining plants were harvested for biomass determination. The harvested plants were separated into needle, stem and root biomass and dried 24 hours at 80°C before weighing.

Downy Birch

Seeds from three populations (stands) of *Betula pubescens*, four mother trees per population, were sown in fertilized peat in February 1993 (Skre 1991b) and covered with a thin (2 mm) layer of sand. The populations were as follows:

NB - Blefjell, Norway
(59°80'N, 9°12'E, 750 m)
NH - Hammerfest, Norway
(70°50'N, 23°40'E, 50 m)
IH - Hafnaskogur, Iceland
(64°10'N, 22°N, 40 m)

The downy birch seedlings (50 plants per population) were placed in a greenhouse at 20°C and given continuous daylight with supplementary light (Osram 58W/30) equivalent to 200 $\mu molm^{-2}s^{-1}$. The water and nutrient regime was similar to that for the spruce seedlings. In May the seedlings were transplanted to 15 cm plastic pots with fertilized peat, one plant per pot, and moved to a daylight greenhouse compartment (10-20°C). In early November these plants were divided between ambient and elevated (+4°C) winter temperature treatments and kept there until April 1994, when they were moved into the open top chambers previously described. The CO_2 treatment was equivalent to what was described for the spruce seedlings, with two instead of three replicates (chambers). The two northern birch populations were given supplementary light in order to extend their photoperiod to that of their habitat, but the light level was relatively low (8-10 $\mu molm^{-2}s^{-1}$) and therefore assumed to have no significant effects on overall photosynthesis and carbon balance (cf. Skre 1993).

The lengths of longest shoots were measured throughout the growing season. In addition, records were taken of the number of leaves per plant and maximum leaf length (see Skre and Naess 1999). Four plants per treatment and population were harvested from each replicate chamber after leaf fall in 1994 for biomass determinations. The harvested plants were separated into stem and root tissue and dried for 24 hours at 80°C before weighing.

In the following discussion the birch populations are referred to as "provenances" for comparison with the spruce seedlings.

STATISTICS

The growth and biomass variables were tested by analysis of variance (Goodnight 1976) to detect significant main and interactions effects. The variance due to different variables and their interactions were tested against the residual variance (F-values). The variation between replicates (open top chambers) was not significant for any of the variables, hence the total residual variances were used as error terms in the analysis. Means testing using the least square method were performed subsequent to the variance analysis on all independent variables in order to detect significant differences.

RESULTS

The weekly mean and extreme temperatures during the two subsequent winters (1992/93 and 1993/94) showed that during the first winter the mean temperatures in the heated greenhouse were almost 6°C above the average outdoor temperature, and in the cold greenhouse about 2°C above the outdoor temperature. During the second winter the temperature in the heated greenhouse was close to 4°C above ambient, and in the cold greenhouse close to ambient (Skre and Nes

1996). January temperatures in the second winter ranged from -0.5°C outside the greenhouse to +1.5°C in the cold room and to +5°C in the heated room (about 2°C lower than the first year). The temperatures inside the open top chambers were up to 3°C higher than outside the chambers. The light in the greenhouse stayed below 100 µmol $m^{-2}s^{-1}$ during all three winter months (1993/94).

Norway Spruce

The needle biomass in October 1994 is shown for separate age classes and provenances in Figure 1. In Figure 1 and Table 1 only the data for seedlings grown at the low CO_2 level are included, to facilitate comparison with results from the 1993 season (Skre and Nes 1996). A strong negative effect of winter temperature was found on the needle biomass in both age classes, particularly after the first winter (cf. Skre and Nes 1996). The needle loss during the first winter may have affected stem and root biomass in the second year (Table 1). In addition to the significant temperature effect on the needle biomass there was a strong general nutrient effect on all biomass variables (Table 1).

Although the temperature x provenance interaction was not significant in the ANOVA (Table 1) means were tested for significant differences since one of the main aims of this investigation was to examine how elevated winter temperatures influenced different Norway spruce provenances. The means testing showed a significant negative temperature effects on stem biomass and the biomass of young and old needles of the northernmost B3 provenance (Figure 1), on older needles of the other Norwegian provenance (C1) and on young needles of the German provenance (H5). The biomass of older needles was not affected by temperature in the German provenance, but was affected in the two Norwegian provenances (B3 and C1). The total biomass of C1 plants seemed to be lower than in the two other provenances at low winter temperature. At high temperatures the current needle biomass was significantly lower in the northern B3 provenance than in the German provenance (Figure 1).

When testing the whole plant material for CO_2 treatment effects, a general significant CO_2 response was found on the means of the biomass variables (Table 2).

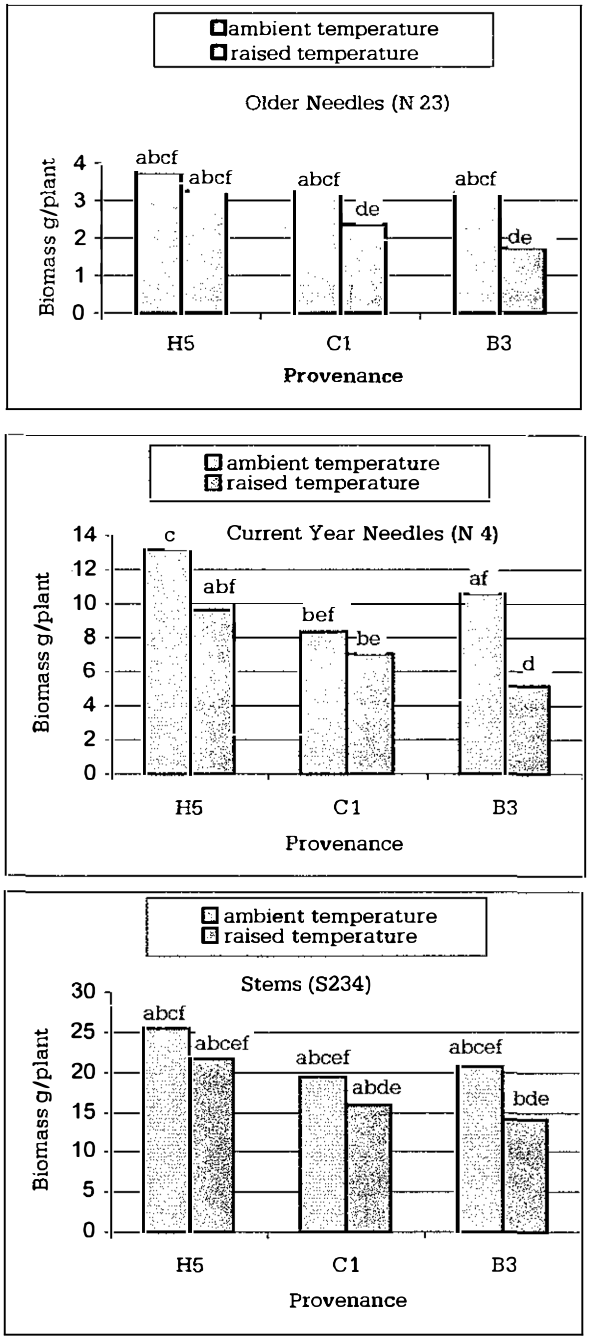

Figure 1. Mean biomass (g/plant) in current year needles , older needles and stems of fertilized plants from the three investigated spruce provenances (H5, C1 and B3) grown at low (ambient) CO_2 levels, harvested 27 Oct 1994. The symbols are: ambient (T1) and +4°C elevated (T2) winter temperatures. Means with the same letters are not signficantly different (p<0.05).

Table 1. Variance ratios (F-values) and significance levels for biomass parameters of 4 year old spruce seedlings, harvested 27. October 1994, as functions of winter temperature, nutrient level and provenance, after two years treatment at low (ambient) CO2 level. Significance levels are *=$p<0.001$; +=$p<0.01$; °=$p<0.05$. The biomass parameters are: Current year needles (N4), older needles (N23), total stem (S) and root (R) tissue. DF=degrees of freedom.
Only significant variables and interactions are included.

		F-values			
Source	DF	N_4	N_{23}	S	R
Provenance	2	6.7+	3.0°	2.1	0.3
Nutrient level	1	49.6*	16.0*	31.5*	12.5+
Temperature	1	9.5+	26.3*	3.9°	1.6
Temp x nut	1	6.7°	2.5	1.2	3.2
Temp x prov	2	3.8°	0.5	1.5	1.1
Nut x prov	2	5.3°	3.5	1.9	2.4
Temp x nut x prov	2	4.0°	3.1	3.8°	0.7
Model	11	9.6*	5.9*	5.0*	2.4
Error ms		6.55	0.90	25.6	7.72
R^2		0.81	0.73	0.70	0.52

Table 2. Variance ratios (F-values) and significance levels for biomass parameters of 4 year old spruce seedlings, harvested 27. October 1994, as functions of CO_2 level, winter temperature, nutrient level and provenance, after two years treatment, the last year also at high CO_2 level. Significance levels are *=$p<0.001$; +=$p<0.01$; °=$p<0.05$. The biomass parameters are: Current year needles (N_4), older needles (N_{23}), total stem (S) and root (R) tissue. DF=degrees of freedom. Only significant variables and interactions are included.

		F-values			
Source	DF	N_4	N_{23}	S	R
Provenance	2	6.7+	1.6	1.4	0.5
CO_2	1	4.4°	5.4°	18.6*	27.7*
Nutrient level	1	87.7*	21.0*	88.2*	68.6*
Temperature	1	25.4*	10.6*	13.0*	5.4°
CO_2 x Nut	1	3.9°	3.2	9.9+	18.8*
CO_2 x Prov	2	0.6	1.9	0.6	0.7
CO_2 x Temp	1	3.0	1.5	2.2	2.5
Temp x nut	1	11.9+	1.1	5.0°	1.0
Temp x prov	2	2.0	2.0	1.3	0.2
Nut x prov	2	2.5	2.4	1.9	2.1
Model	23	11.0*	4.2*	10.5*	9.4*
Error m.s.		7.1	1.6	22.4	8.8
R^2		0.82	0.63	0.82	0.80

There was no significant provenance or temperature interactions with CO_2, but there was a significant CO_2 x nutrient interaction effect for young needle (N4), total stem(S) and root tissue(R) biomass. Results from the means testing show that needle, stem and root biomass of the fertilized, but not unfertilized, plants grown at elevated CO_2 level was higher than in plants grown at ambient levels (Figure 2). The reason for this interaction was a stronger winter temperature response on fertilized vs. non-fertilized spruce seedlings in young needles (Skre, unpubl.)

In accordance with the variation in needle biomass there was a slight reduction in the final shoot length of fertilized plants grown at ambient CO_2 level after the 1994 season due to elevated winter temperatures (Table 3). Means testing also showed that the temperature effect on shoot elongation rates was strongest early and late in the growing season (Figure 3). The seedlings from the heated greenhouse (T2) started growing earlier in spring, but were finally overtaken and surpassed by the seedlings from the cold greenhouse (T1). The C1 provenance showed lower growth rates than the other two and showed a weak response to the CO_2 effect. In addition, the reduction in needle biomass of young shoots (N_4) due to elevated winter temperatures was also less for this provenance compared with the two other provenances (Figure 1).

Downy Birch

The length of the longest shoots in the three provenances is shown as a function of time during the 1994 season in Figure 4. Some previously published results (Skre and Naess 1999), are included in Figures 4 and 5 to enable direct comparisons with the corresponding results (Figures 2 and 3) of shoot and biomass growth in Norway spruce from this study. For downy birch there was a significant temperature effect on growth in the early and late parts of the growing season (Figure 4) (cf. Skre and Naess 1999) which may be attributed to earlier budbreak and different final shoot length in birch seedlings grown in high vs. low winter temperatures.

Due to earlier budbreak the shoot lengths were also considerably higher in seedlings from the southern alpine provenance (NB) than in the northern relatives (NH and IH) in the early part of the growing season. By the end of season the differences were smaller. Means testing (Skre and Naess 1999) showed that there was a positive CO_2 effect on shoot growth in the NB and NH provenances, but not for the Icelandic birch (IH) provenance (Figure 4). In all three provenances increased winter temperatures seemed to lead to earlier budbreak and increased shoot elongation during the first part of the season (Skre and Naess 1999). The effect of raised winter temperatures by the end of season was slightly positive (not significant) in fertilized birch from

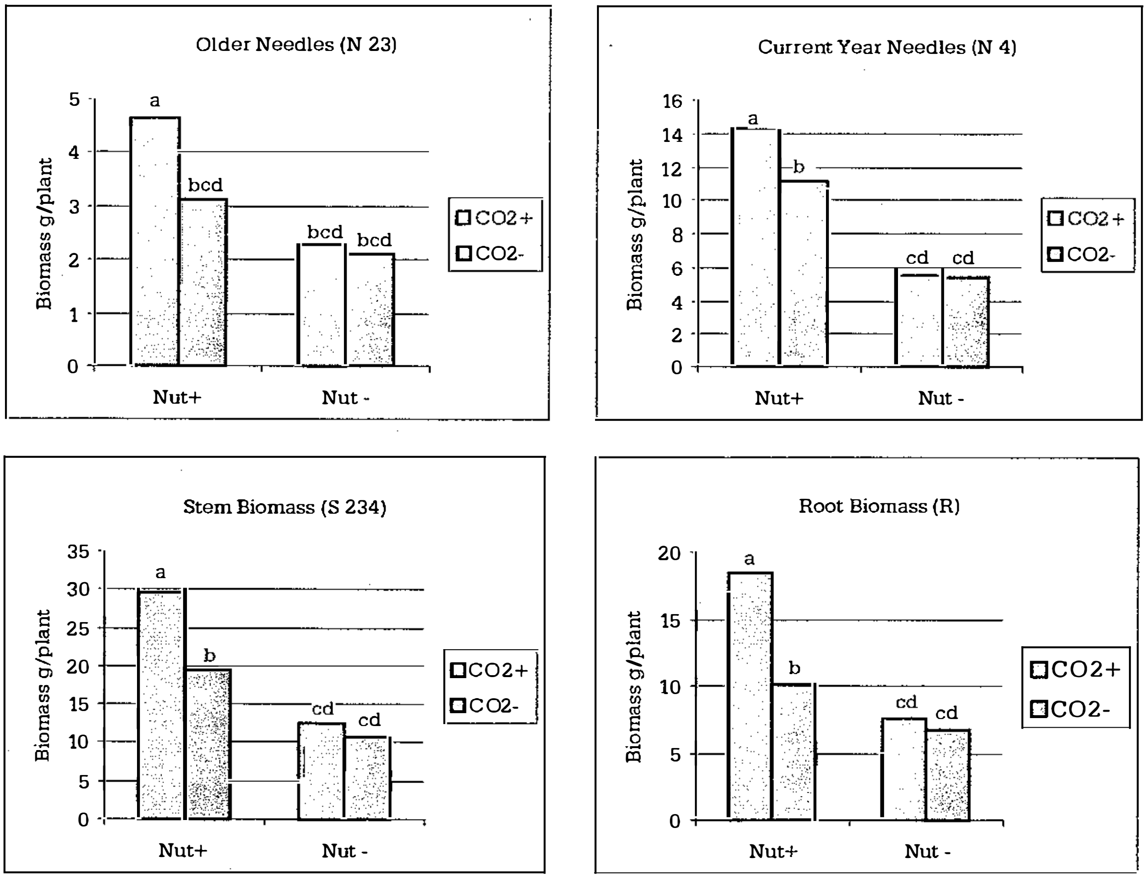

Figure 2. Mean biomass (g/plant) of combined plant samples at low (-Nut) and high (+Nut) nutrient level in current year (N_4) and older (N_{23}) needles, combined stem (S_{234}) and root (R) tissue (top), and in stem tissue of the three investigated spruce provenances, shown separately (bottom) measured 27 Oct 1994, in spruce seedlings grown for one year at ambient (CO_2-) and elevated (CO_2+) carbon dioxide level. Means with the same letters are not significantly different ($p<.01$).

the southern alpine provenance (NB), while the effect on plants from Icelandic origin (IH) was negative (Figure 4).

The stem and root biomass in fertilized and non-fertilized birch (Table 4b) by the end of the 1994 season (October) confirmed the results from the shoot length measurements. Similar to the spruce seedlings, a very strong and significant positive effect of increased nutrient strength was found on the birch biomass variables (Table 4b). There was also a significant provenance x nutrient interaction (Table 4b) where the nutrient effect on growth was much stronger for plants from the southern provenance (NB) than for their northern relatives, particularly the NH provenance. There was no significant effect of winter temperatures on the stem or root biomass of any of the provenances (Table 4a).

Generally, stem biomass in the southern birch provenance (NB) was significantly higher than in the two northern provenance (cf. Skre and Naess 1999). Stem biomass of fertilized plants from the two southern provenances (NB and IH) was higher than for non-fertilized plants from the same provenances, but were lower than stem biomass of fertilized plants from the northern NH provenance (Table 4b). Results from means testing for CO_2 x nutrient interactions are shown in Table 4c. Stem and root biomass of fertilized plants grown at high CO_2 level was significantly higher than in comparable non-fertilized plants, but not in plants grown at ambient CO_2 level. In this table no significant CO_2 effects were found, although Table 4a indicates a significant general CO_2 effect; the reason is that all three provenances have been combined. Results from the ANOVA run for each birch provenance individually (Table 5) indicate strong nutrient effects on the biomass variables in all

Table 3. Variance ratios (F-values) and significance levels for annual shoot length (cf. Figure 3) at different dates and day numbers(in brackets) during the 1994 season, as functions of provenance, winter temperature, nutrient level and CO_2 concentration in the three investigated Norway spruce provenances. Significance levels are: *=$p<0.01$; +=$p<0.05$; °=$p<0.1$. DF = degrees of freedom. Only significant interactions are included. See Figure 3 for means testing of single provenances on winter temperature and CO_2 levels.

Source	DF	13. Apr. (103)	12. May (132)	26. May (146)	7. June (158)	27. June (178)	3. Aug (215)
CO_2	1	2.05	0.04	2.45	1.41	0.31	0.06
Nutrient level	1	0.03	0.02	2.03	9.78	29.74°	290.8+
Temperature	1	53.93°	85.05+	0.76	2.98	85.29+	22.30°
Provenance	2	0.43	0.66	0.39	5.09	11.82°	40.96+
CO_2 x Nut	1	0.77	1.65	2.28	0.77	1.47	36.03°
CO_2 x Prov	2	0.44	13.01°	0.59	0.82	10.96°	4.22
CO_2 x Temp	2	2.05	0.05	2.47	152.1+	0.15	0.22
Nut x Temp	1	0.28	0.02	69.44°	245.4+	9.00	1.00
Prov x Temp	2	0.34	0.67	0.57	6.50°	0.45	1.85
Model	23	2.97*	5.44*	2.23*	2.13*	4.69*	5.85*
Error m.s.		0.17	0.72	2.32	4.16	7.12	12.62
R^2		0.45	0.68	0.46	0.44	0.54	0.60

three provenances and a strong CO_2 effect on stem and root biomass in the southern NB provenance. In the two northern provenances the CO_2 effect was much weaker. In the southern provenance there was also a strong and significant interaction between CO_2 and nutrient level, this was also found in roots of Icelandic birch seedlings (IH). The means testing (Figure 5) partly supported these conclusions. The stem biomass of fertilized plants from the two northern provenances (NH and IH) was significantly higher than in non-fertilized plants at both CO_2 levels. In the Icelandic provenance (IH) the stem and root biomass per plant was also significantly higher at high than at ambient CO_2 levels. Such effects were not found in the southern NB provenance, where the nutrient effect seemed to be stronger than in the two northern provenances (see also Table 5).

DISCUSSION

Norway Spruce

Looking at the growth parameters in Tables 1 and 2, i.e. the biomass of annual and older needles, stem and root tissue, the nutrient level is by far the most significant response factor. Nutrient levels increased growth and biomass in all three provenances (cf. Skre and Nes 1996). The reason is partly prolonged secondary growth in fall (Pümpel et al. 1975) and partly earlier dormancy breaking (Larsen 1976, Murray et al. 1989). The heavy needle loss, mostly of old needles, after the mild 1992/93 winter (Skre and Nes 1996) correlates well with the reduction in growth and biomass parameters the following season (Figure 1), and the reduced needle biomass during the 1994 season may also partly be an after-effect of this needle damage. The growth reduction was particularly strong in plants grown under high winter temperatures (Figures 1 and 3), where also the needle damage was found to be strongest (Skre and Nes 1996). On the other hand, elevated CO_2 level the following summer seemed to partly repair the damage in plants grown at high nutrient level (Figure 2). This effect seemed to be general. The strongest reduction in biomass of old needles was found in the northernmost B3 provenance (Figure 1) where also the strongest needle damage (Skre and Nes 1996) and shoot growth reduction (Figure 3) occurred. Figure 2 also shows that the positive CO_2 effect on growth was strongest for root biomass (cf. Mousseau 1993), while the negative effect of elevated winter temperatures was strongest on the biomass of young needles (Skre and Nes 1996).

In the present study some plants were grown under elevated CO_2 levels during the

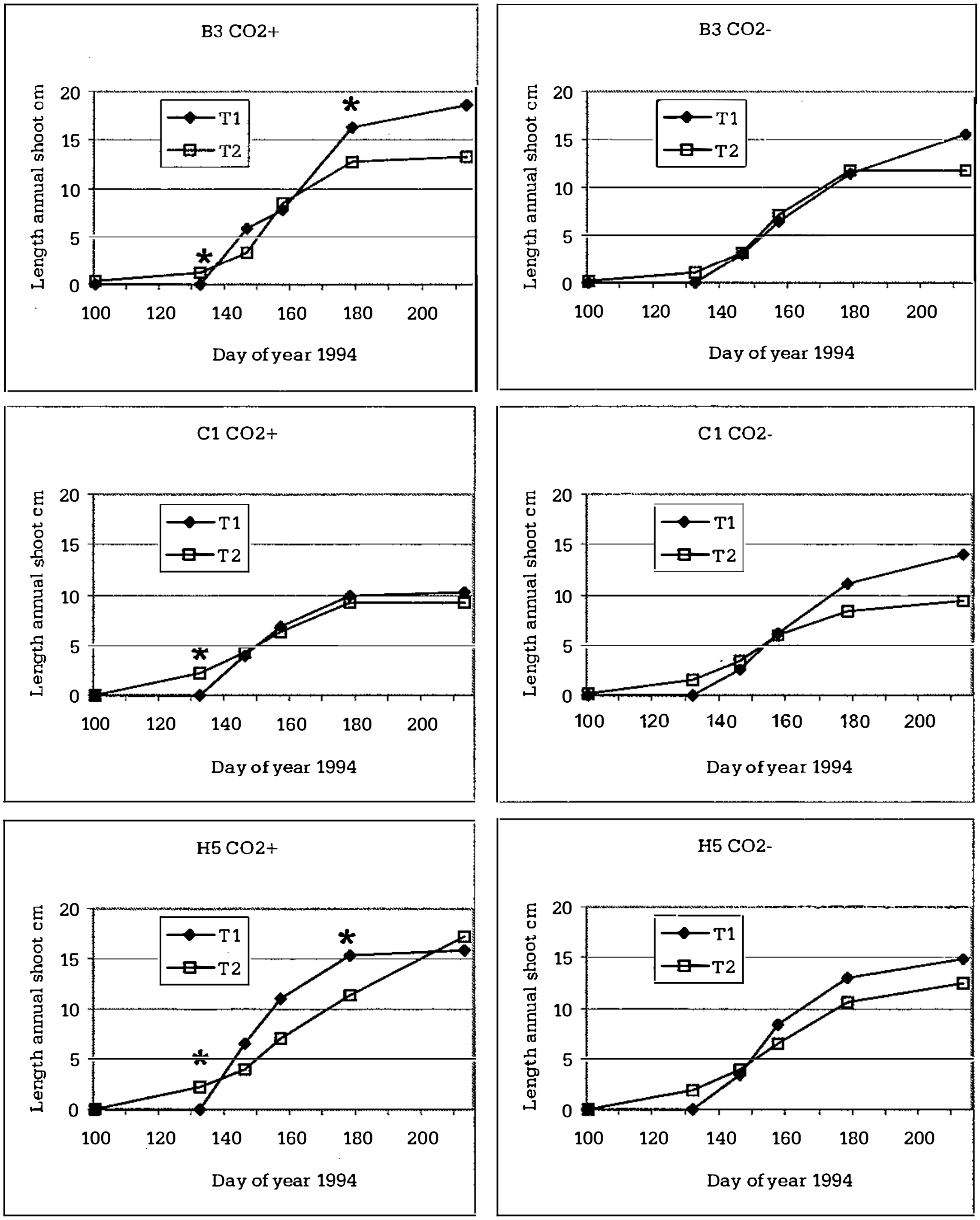

Figure 3. Mean shoot elongation (cm) in fertilized seedlings of the three investigated spruce provenances during the 1994 season. The plants are grown at high (650 ppm) and normal (350 ppm) CO_2 levels (CO2+ and CO2-), and at ambient (T1) and +4 °C elevated (T2) winter temperatures. Significant differences between winter temperature levels are sown as asterisks where $^*p = <0.05$. See Table 3 for overall statistics.

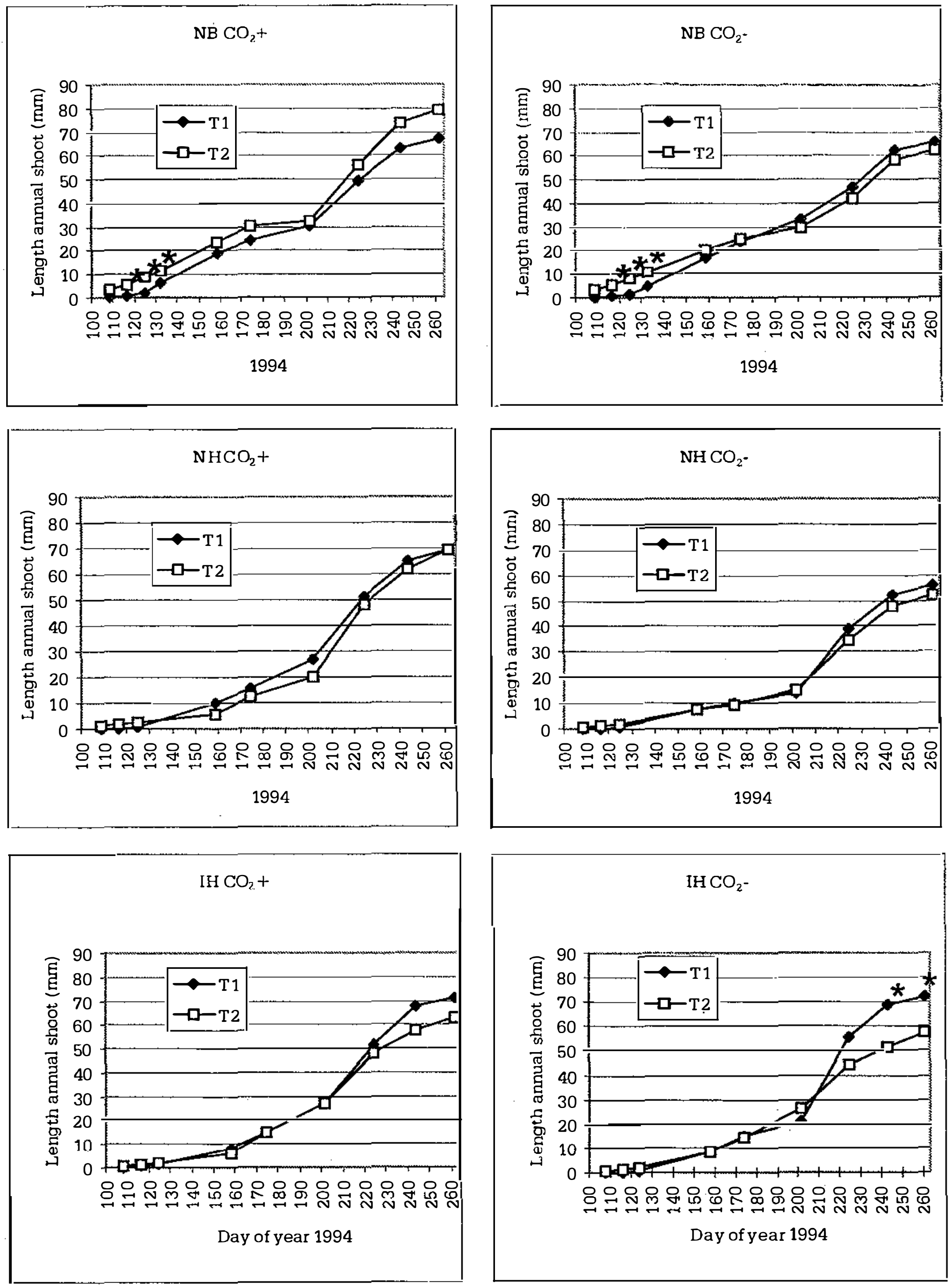

Figure 4. Mean shoot elongation (mean length longest shoot L_{max} in cm) during the 1994 season in fertilized plants of the three investigated birch populations. Significant differences between winter temperature treatments are shown by asterisks on the highest values, where $*p<0.05$. After Skre and Naess (1999). The plants are grown at elevated (left) and ambient (right) CO_2 levels and at two temperature levels: T1 = ambient winter temperatures and T2 = +4°C elevated winter temperatures.

Table 4(a). Variance ratios (F) and significance levels for final maximum shoot length (L_{max}) and stem and root biomass at harvesting in October 1994 in the three investigated downy birch provenances, as functions of winter temperature, nutrient level and CO_2 concentrations, Significance levels are: *=$p<0.001$; +=$p<0.01$; °=$p<0.05$. DF = degrees of freedom. Only significant interactions are listed. See Figure 4 and 5 for means testing of single provenances on CO_2 and temperature levels.

Source	DF	L_{max}	Stem biomass	Root biomass
Provenance	2	87.3*	12.5*	5.9*
CO_2	1	7.5+	28.3*	62.3*
Nut	1	249.2*	257.8*	178.3*
Temp	1	7.4+	0.03	0.12
Prov x Temp	2	3.7°	1.5	0.3
Prov x Nut	2	23.6*	2.7°	3.2°
Prov x CO_2	2	8.7*	1.1	0.7
Temp x Nut	1	4.1°	0.03	0.02
Temp x CO_2	1	1.1	0.1	1.6
Nut x CO_2	1	4.6°	10.3+	16.9+
Model	23	22.1*	9.9*	8.8*
Error m.s.		17.7	20.9	11.6
R^2		0.88	0.54	0.51

(b) Means testing of the Prov x Nut interaction in stem and root biomass. Treatments and provenances with the same letters are not significantly different ($p<0.01$).

Nutrient Level	Provenance	Mean Stem Biomass	Mean Root Biomass
+	NB	15.7 a	10.4 ac
	NH	10.3 bcd	7.1 b
	HI	11.9 bc	10.0 ac
-	NB	6.2 bdef	4.7 def
	NH	4.4 def	4.3 def
	HI	5.0 def	5.1 def

(c) Means testing of the Nut x CO_2 interaction (cf. Figure 5) in stem and root biomass. Treatments with the same letters are not significantly different ($p<0.1$).

Nutrient Level	CO_2 Level	Mean Stem Biomass	Mean Root Biomass
+	+	15.3 ab	11.6 ab
	-	11.1 ab	7.3 abcd
-	+	6.0 bcd	5.4 bcd
	-	4.9 bcd	4.0 bcd

1993/94 winter season. Means testing on shoot growth measurements showed a significant positive effect of continuous vs. non-continuous CO_2 level at low winter temperatures on fertilized plants from two provenances (B3 and H5), while the C1 provenance was unaffected (Naess 1997).

The difference in needle loss and biomass between provenances after the mild winter may be explained as genetic adaptations to the climate at the habitats of the seed populations. Meteorological data (Naess 1997) show that the winter climate at the habitat of the German seed population was more like the climate at the experimental site (Fana) than the climate at the habitats of the two Norwegian populations. According to Johnsen et al. (1996) Norway spruce is able to adapt easily to changes in the climate especially with regards to autumn frost hardiness.

Hagem (1947) carried out controlled experiments in greenhouse at 50% light reduction. He found that when temperature increased to 5-8°C in December/January, there was a slightly negative carbon balance in the Norwegian spruce provenance, and a positive carbon balance in the German provenance, he investigated. This is in agreement with our experiment on Norway spruce seedlings, where the January temperature was about 7°C (Skre and Nes 1996). In contrast the January temperature the following winter (1993/94) was only 5°C, with only a slight reduction in needle loss (Skre and Nes 1996).

Downy Birch

There was a positive effect of elevated CO_2 on all growth parameters in two of the three provenances (Table 5), but strongest in the southern provenance and most evident for the root and stem biomass (Table 4b). The final shoot length (Figure 4) and biomass (Figure 5) was highest in the southern plants, in accordance with the different growth strategies within the provenances. According to Skre (1991b) southern and lowland *Betula pubescens* provenances are adapted to a high-growth strategy for survival in an environment where competition for light and nutrients are the selective factors, while northern and arctic provenances are adapted to a low-growth strategy, probably in order to accumulate storage compounds and defense against climatic and herbivore stress. The present study showed a generally strong and significant effect of nutrient level on all growth parameters, as well as an interaction effect with CO_2. The reductions in shoot and root biomass at the lower fertilizer strength (Table 4c) were about 50% and slightly stronger at

Table 5. Variance ratios (F) and significance levels for shoot and root biomass in separate birch provenances, measured October 1994, as functions of winter temperature, nutrient level and CO_2 concentrations. Significance levels are: * =$p<0.001$; + =$p<0.01$; °=$p<0.05$. DF = degrees of freedom. Only significant interactions are listed. See Figure 5 for means testing of single provenances on CO_2 concentration at different nutrient levels.

		Stem Biomass			Root Biomass		
Source	DF	NB	NH	IH	NB	NH	IH
CO_2	1	13.8*	11.2+	4.4	38.4*	8.4+	9.9+
Nut	1	133.8*	26.7*	51.8*	91.6*	11.7*	28.4*
Temp	1	0.1	0.2	1.0	0.1	0.0	0.0
CO_2x Nut	1	17.2*	0.5	2.8	15.9*	0.0	8.7+
CO_2x Temp	1	0.1	0.3	1.4	2.0	0.3	0.1
Nut x Temp	1	0.1	1.1	0.1	0.1	0.8	0.0
Replicate	1	0.0	0.1	1.4	0.7	0.4	0.2
Model	15	14.1*	3.5+	5.1+	12.8*	2.2°	3.7+
Error m.s.		16.1	26.2	18.2	8.5	13.7	17.3
R^2		0.69	0.41	0.5	0.67	0.3	0.42

higher than at lower CO_2 level.

In contrast to Norway spruce, strong provenance interactions were found with nutrient and CO_2 levels, while the winter temperature effect on biomass parameters was insignificant. The increase in stem and root biomass due to increased nutrient level was strongest in the two southern provenances which is in accordance with Skre (1993), who found the same differences when comparing a subarctic and a southern lowland birch provenance. In contrast to the present study, Pettersson and McDonald (1994) found no significant increases is shoot or biomass growth in *B. pendula* seedlings after 70 days of CO_2 treatment. The explanation may be that their treatment time was too short. On the other hand, Mortensen (1994) found a 7% increase in relative growth rates after *B. pendula* seedlings were grown under elevated CO_2 for only 35-40 days. The results indicate that large differences may exist between populations and individuals in responses to CO_2 level. In contrast to the two others, the plants from the Icelandic provenance (IH) showed no significant response of shoot elongation rates to CO_2 level (Figure 4), although there was a slightly positive effect on stem and root biomass of fertilized plants and of fertilized stems of the northern population (Figure 5). On the other hand, the strong nutrient response in the southern birch provenance was also reflected in the strong CO_2 x nutrient biomass interactions for this provenance (Table 5). However, no significant specific CO_2 effects were found in the means testing of this provenance (Figure 5), presumably because of the strong nutrient effects.

Due to earlier budbreak in plants grown at high winter temperatures, there was a very strong and significant temperature effect on shoot elongation rates early in the growing season (Figure 4), particularly in the southern provenance (NB). The nutrient effects on shoot growth, however, seemed to increase throughout the season (see also Skre and Naess 1999). The reason why there was a slightly positive influence of elevated winter temperature on shoot elongation rates in the southern alpine ecotype (NB), but not in the northern ecotypes, may be earlier budbreak at elevated winter temperatures in the southern plants (Figure 4). On the other hand, the reduced shoot elongation rates that were found in the Icelandic downy birch population at elevated temperatures may be related to its low-growth strategy, as a result of hybridization with slow-growing *Betula nana* in the provenance (Elkington 1968, Anamthawat-Jonsson and Tomasson 1990, Anamthawat-Jonsson et al. 1993). Because of small low temperature requirements for dormancy release, increased winter temperatures may extend the growth season in birch (Myking and Heide 1995), particularly in southern ecotypes, leading to increased growth. Furthermore, even in a dormant state birch may carry out some photosynthesis in

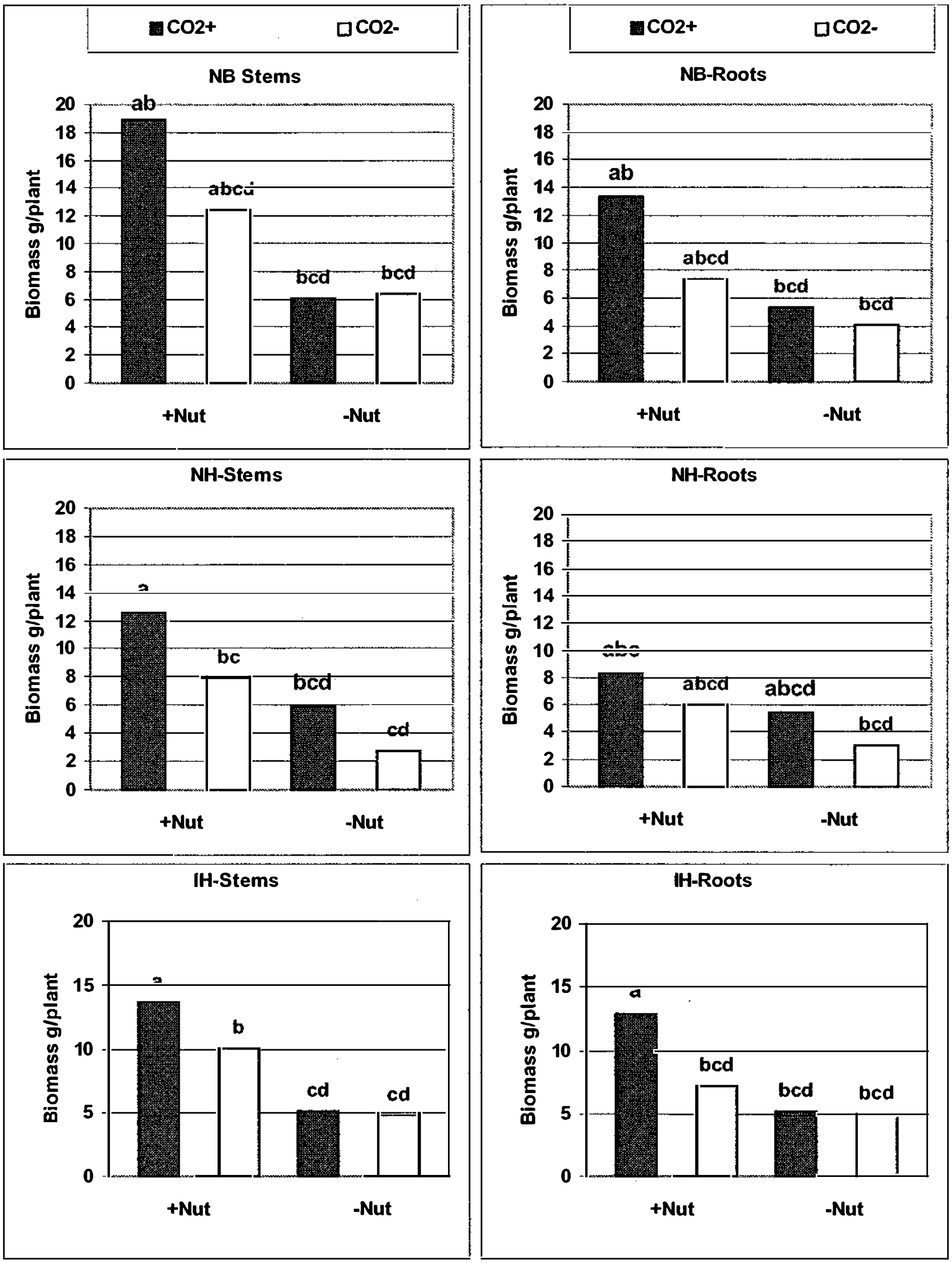

Figure 5. Mean stem and root biomass (g/plant) in separate provenances (NB, NH and IH) grown at different CO_2 level and nutrient levels, harvested October 1994 after one growth season. The symbols are: high (+Nut) and low (-Nut) nutrient level and ambient (CO_2-) and elevated (CO_2+) CO_2 level. Means with the same letters are not significantly different ($p<0.1$).

winter, due to the green chlorophyll layer in the innerbark tissue (Aalvik 1939), and due to its deciduous life form, with leaf abscission and retention of nutrients, respiration rates in birch are very low in winter. This positive carbon balance may be even stronger when temperatures are raised by 4°C in plants with early dormancy release (NB), while plants with late budbreak may have a disadvantage. This negative effect of elevated winter temperatures on growth in northern ecotypes of birch may be caused by a combination of a low-growth strategy (Skre 1993) and a more unfavorable carbon balance during the period with rapid growth in spring (Stewart and Bannister 1973).

Comparison Between Species and Ecotypes

In general, stronger and more significant responses to CO_2 and nutrient level occurred in the birch provenances than in the Norway spruce provenances, indicating birch as a more responsive and opportunistic species than spruce. This difference may be related to life form and growth pattern in the two species (cf. Skre 1991c).

In contrast to Norway spruce, strong provenance interactions were found with nutrient and CO_2 levels in birch, while the winter temperature effect on biomass parameters was insignificant. These results agree well with those from other studies (e.g. Murray et al. 1989, Skre 1993), indicating that ecotypes and species that are competitive and adapted to a high-growth strategy (e.g. southern and lowland ecotypes of both species), in particular *Betula pubescens*, may take advantage of the expected climate change, because they would tend to put more of their carbon and nutrients into shoot growth than slow-growing relatives. On the other hand, plants and ecotypes that are less growth-oriented, would easily be out competed because they tend to put their resources into root growth, storage or defense against winter stress (Chapin 1979). The present study indicates that Norway spruce and downy birch seedlings both respond to raised CO_2 and nutrient level by increasing their shoot and biomass growth. Generally, the significant interactions between nutrient and CO_2 effects, particularly in the southern provenances, indicates higher genetic variation in birch than in Norway spruce, probably as a result of inbreeding with *Betula nana* and *B. pendula* in the different *B. pubescens* provenances (Elkington 1968, Kallio et al. 1983) The strongest CO_2 response in birch occurred in the southern ecotype. This was the only provenance to show positive responses in shoot elongation rates at +4°C raised winter temperatures. This response was mainly due to earlier budbreak and a longer growing season. In Norway spruce the CO_2 effect on growth was general indicating, again, less genetic variation between provenances than in birch.

On the other hand, the two northern birch population and all three Norway spruce provenances showed negative winter temperature responses, and in spruce there was also increased needle loss when mean ambient winter temperatures were above 3-4°C as in 1992/93 (Skre and Nes 1996). The needle loss and growth reduction was particularly strong in the northernmost provenance (B3). The reason for the observed damage may have some relationship to temporary negative carbon balance in winter and early spring. Measurements on CO_2 exchange in spruce shoots during the 1993/94 winter season (Skre, unpublished results) indicate that the compensation light for whole plants varied around 30 $\mu mol\ m^{-2}s^{-1}$ at ambient winter temperatures (+1°C) and around 60 $\mu mol\ m^{-2}s^{-1}$ at elevated winter temperatures (+5°C), equivalent to 6 and 12 weeks with negative carbon balance. Similar negative growth responses were not observed in the birch populations, except for the Icelandic birch, where it may be related to genetic differences caused by hybridization with slow-growing *Betula nana* in the birch populations (cf. Elkington 1968, Anamthawat-Jonsson et al. 1993).

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FORECAST OF CARBON POOLS AND FLUXES IN SIBERIAN AND FAR EASTERN RUSSIAN FORESTS

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ABSTRACT

This paper concerns the methods and results from mathematical modeling of natural and anthropogenic dynamics for Siberian and Far Eastern Forests in Russia. It contains the evaluation of carbon pools and annual carbon uptake as well as the analysis of natural and anthropogenic impacts, as they influence tree species composition, age structure and the carbon budget.

INTRODUCTION

Recently the study of the role of boreal forests as a potential regulator of large-scale biosphere processes has increased in significance, since boreal forests may be instrumental to forming a sustainable development strategy for the world community. In this context the estimation of a carbon budget for boreal forests and developing an understanding of the mechanisms through which they may sequester carbon (C), is critical to discerning the role of boreal forests in mitigating the effects of global climate and environment changes.

Dynamics of carbon pools and fluxes in forest ecosystems are determined by production, death and destruction processes occurring within them. The task of quantitative estimation of a carbon budget in boreal forests is especially difficult because their edaphic and climatic conditions are heterogeneous and the processes of stands growth, evolution and death are dynamic and subject to many biotic and abiotic factors.

More than half of the globe's total boreal forest cover is located within the territories of Siberia and Far East Russia, and involve numerous forest vegetation zones and regions at different scales of economic development. The assessment of their carbon budgets and contribution to the atmosphere's carbon budget is made possible through the study and simulation of long-term forest vegetation dynamics within each of these regions, under different forest management strategies.

DESCRIPTION OF THE RESEARCH AREA

The focus of our research is the forest vegetation of Siberia and Far East Russia within the Forest Fund lands. The following structural characteristics of the forests were evaluated;

1. the Forest Fund area distribution amongst land categories, groups and categories of forest protection and,
2. the area and stand growing stock distribution by forest type and age groups.

The first of these evaluations is aimed at classifying forests and other wooded lands within the Forest Fund. The second evaluation concerns forest type, age structure and productivity.

The Forest Fund of Siberia and Far East Russia is comprised of 63 ecoregions in total. Only those ecoregions with forest land cover were used in this study (total of 58 ecoregions). Their areas vary from 0.2 million ha in the southern districts of Western Siberia to 55.0 million ha in the northern districts of Eastern Siberia.

METHODS

The forests of each region were divided into two forest use regimes: exploitation (harvesting of mature stands permitted) and non-exploitation (harvesting not permitted) forests.

Forest stands with similar characteristics and use regimes were grouped into economic sections. The sum of these sections determined the organizational structure of the Forest Fund within each ecoregion. The attributes used to classify the economic sections for exploitation forests were: predominant tree species, the productivity group (high, low), relief type (plain, mountain) and method and age of clear cutting. The economic sections in non-exploitation forests were classified by using forest characteristic attributes only (predominant tree species, productivity group and relief type). Grouping into homogenous sections allows for the assumption of uniformity of biological processes of stand growth and evolution, as well as their response to impacts external to the bounds of each economic section. Combinations of age and method of clear cutting determined the unity of forest use regimes within each economic section.

The forest cover structure of any ecoregion was assumed to be known for each time step, if the forest type and average stocks of the main forest forming tree species of each age class were known for each economic section. The initial condition for each economic section was derived from the data of the State Forest Inventory (Dynamics of the Siberian Forest Fund and Potential Wood Supply, 1997) and experimental data on phytomass and productivity of the forest ecosystem (Forest Fund of the USSR, 1998).

The dynamics of vegetation cover (i.e. the change of forestland structure, tree species composition and forest age structure) over time were considered the result of the biological processes of stand growth and succession as influenced by disturbance and management impacts (Figure 1). Disturbance impacts include forest fires, pests and forest diseases, extreme weather conditions, and other incidental natural and anthropogenic factors. Management impacts include clear cutting and intermediate use (tender cutting), as well as measures taken for forest protection and regeneration. The model assumes that natural processes in forest ecosystems are deterministic, and estimation of parameters was based on mean values for the variables.

Two types of forest vegetation dynamics were modeled, natural (without management impacts) and anthropogenic. The processes of tree stand growth were described by equations of their mean stock dynamics, using functions of the trees' biological age. Natural forest stand successions were modeled as Markov's chains processes of prevailing tree species changes and stand age changes.

The processes of stand destruction caused by forest fires, pests, forest diseases and other disturbance impacts were described through the matrices of stand death, which allow for the possibility of forest lands moving into the category of other wooded lands under different scales of impact. Forest forming processes on burned or cut areas, as well as on other wooded lands, were modelled in terms of ingrowth by forest vegetation and the tree species composition of forming young forests. To describe the processes, the matrices of tree species composition of natural regeneration were used, as well as rules for transition of other wooded lands to forests lands.

Anthropogenic dynamics of forest vegetation was modeled by imposing management impacts on natural dynamics. These impacts resulted in withdrawal of a certain amount of forest biomass, changes in tree species composition and stand age, and changes to average growing stock per ha. Different strategies of management were reflected in the regime of forest resource use, type of regeneration and level of forest protection.

The forest use regimes were defined by the intensity and recurrence of different methods of clear- and intermediate cutting, while regeneration regimes were expressed through the ratio of natural to artificial regeneration. The latter regime took into account the differing establishment abilities of major tree species, and the abundance and tree species composition of viable undergrowth.

The level of forest protection was expressed through the scale of tree stand death, because of destructive impacts of fires and other elemental factors. It was derived by taking actual areas of burned and dead stands within each ecoregion into account. It was considered as an outer (regulated) model parameter for anthropogenic dynamics of forest vegetation.

Simulations of forest cover dynamics, carbon pools and fluxes were done for all forestlands, burned or cut areas and dead stands within the territories of Siberia and Far East Russia. The initial value for the total area was 540.6 million ha and 58.5 x 10^9 m^3 for total stem wood stock (Table 1). The area of exploitation forests was estimated at 243.8 million ha with a growing stock of 29.0 x 10^9

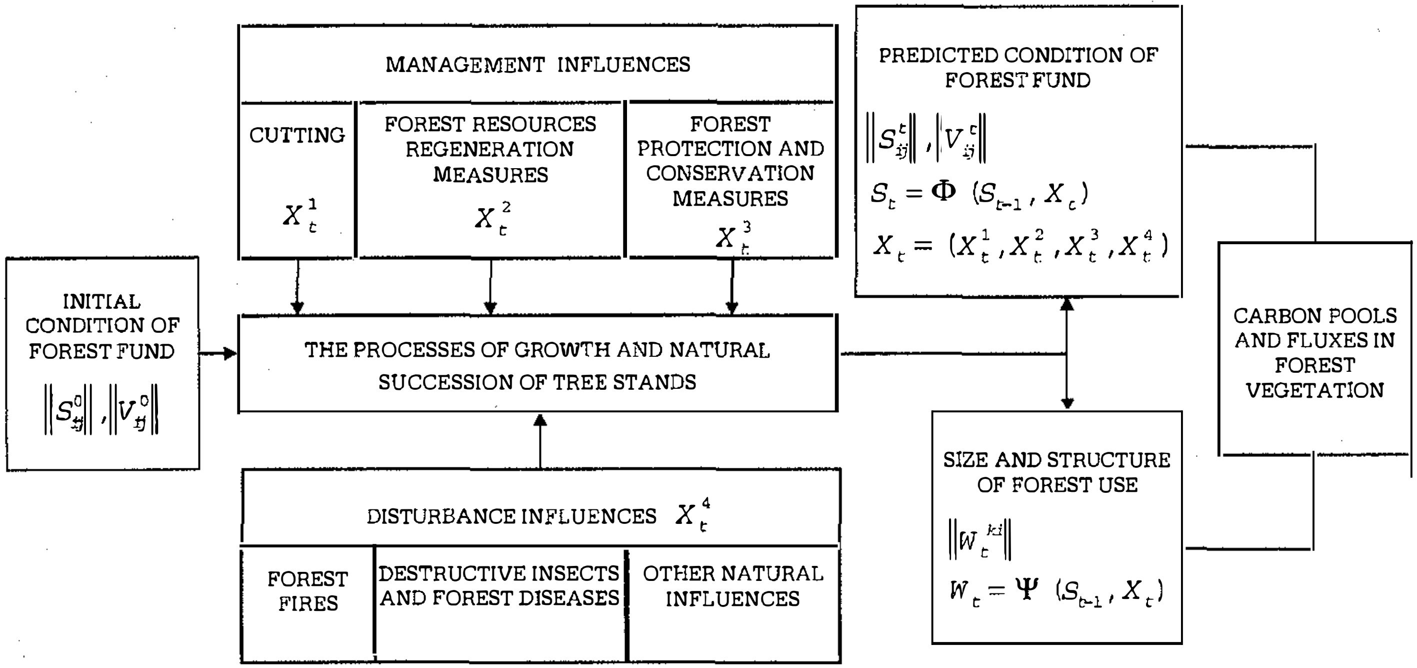


Figure 1. The general scheme for modelling of Forest Fund dynamics.

Table 1. The area (ha) and growing-stock volume (m^3) by economic region and forest use regime for Siberia and Far East Russia.

	Forest Use Regime		
Economic Region	Exploitation Forest	Non-exploitation Forest	Total
	Area of forest land (10^6 ha)		
West Siberia	49.7	23.6	73.4
East Siberia	91.21	123.7	214.9
Far East Russia	102.9	149.4	252.3
Total	243.8	296.7	540.6
	Growing-stock volume (10^9 m^3)		
West Siberia	6.5	3.7	10.2
East Siberia	12.6	15.8	28.4
Far East Russia	9.9	10.2	20.1
Total	29.0	29.7	58.7

m^3. The area of non-exploitation forests was estimated at 296.7 million ha with a growing stock of 29.7 x $10^9 m^3$ (Table 1).

Simulation of long-term forest dynamics for Siberian and Far East Russian carbon pools and fluxes, were run for two "polar" forest management strategies: one of them concerns natural dynamics (without management impacts), and the other describes anthropogenic effects on forest vegetation (including management impacts). The first management strategy represents the existing scale of forest protection with no forest resource use or artificial regeneration. The second management strategy represents the existing scale of forest protection, maximum allowable clear cutting volumes and minimal inputs for forest regeneration.

The maximum allowable cut in forests of each ecoregion was determined using estimators of maximum mean annual increment developed by Isaev and Korovin (1999) and Isaev et al (1993). Environmental limitations were set to ensure conservation of tree species diversity, as well as allowing for artificially regenerated stands within the Forest Fund. Economic limitations guaranteed current timber supply volumes into the future, with the same quality of tree species.

Detailed assessment of the destructive natural and anthropogenic impacts on the carbon budget of forest vegetation was carried out for the test regions by using different levels of annual tree stand death (from 0 to 1% with 0.1% spacing). The test regions were the Angarsky Southern Taiga and Low Amur Mountain Middle Taiga located within the Krasnoyarsk and Khabarovsk Territories.

Simulations of forest cover dynamics and carbon pool fluxes were run for a period of approximately one rotation (180 years) using 20 year time steps. The outputs from the model for each time step of the period modeled include; the distribution of areas and growing stocks for dominant tree species and age classes, the areas of other wooded lands (cut areas, burned out areas, perished stands) and carbon stocks and fluxes of woody vegetation.

Forest vegetation carbon pools were calculated using conversion coefficients developed by Isaev et al (1995a, 1995b). Carbon stocks contained in the vegetation of other wooded lands were calculated using the areas of these lands and data on shared (per 1 ha) phytomass stocks within the test sites. It was assumed that 1 kg dry mass of stems, branches and roots contains 0.5 kg of carbon and that 1 kg dry mass of coniferous needles, leafs and low level plants contains 0.45 kg of carbon.

RESULTS

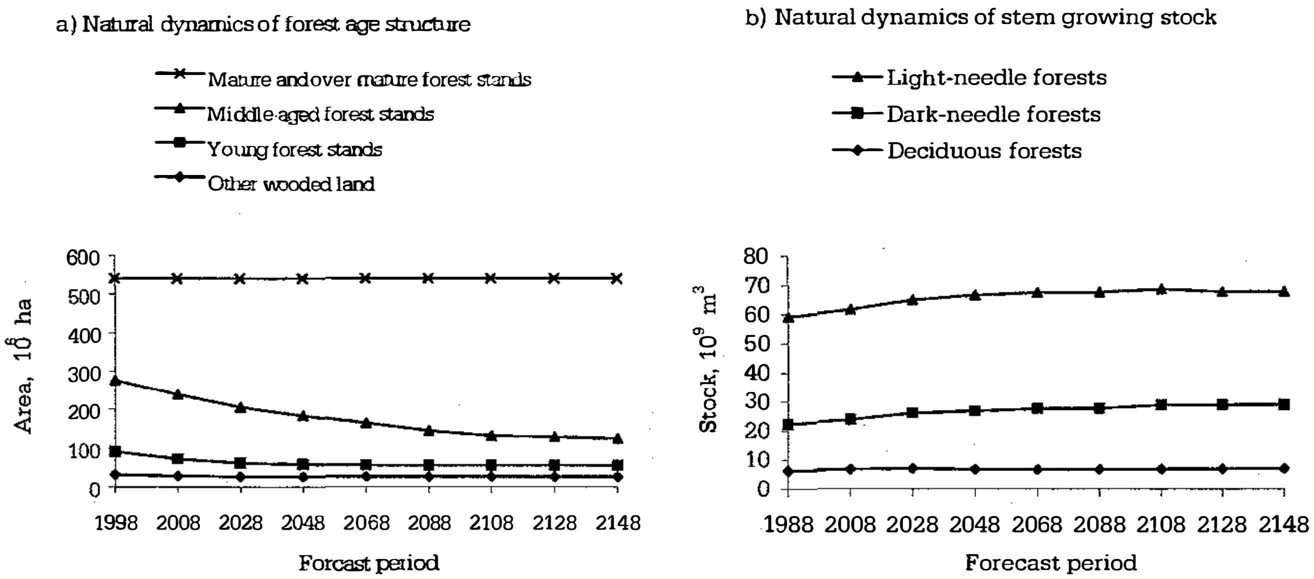
The modeling of forest cover natural dynamics has shown that despite the diversity of climatic conditions and edaphic factors, each ecoregion of Siberia and Far East Russia exhibits common trends over time for growing stock and area distributions, which influence the carbon budgets of boreal forests. The most important trends are the following:

1. increase of forest area due to ingrowth of cut areas and other wooded lands categories;
2. gradual accumulation of mature and overmature forest areas due to lack of cutting (from 263.1 in 1998 up to 415.5 millions ha in 2148) (Fig. 2a);
3. increase of dark coniferous forest area due to replacement of predominant tree species in light coniferous and deciduous forests;
4. accumulation of total wood growing stock due to increasing of the mature and overmature forested area (Fig. 2b).

Modeling of anthropogenic dynamics of Siberian and Far Eastern Russian exploitation forests also shows common trends in changes of forest type and age structures and growing stocks, across all ecoregions and economic sections. As a rule, the direction of these changes is opposite to those shown under the natural dynamics of forest cover. If the strategy for sustainable management of exploitation forests in Siberia and Far East Russia is implemented, an increase in the other wooded land areas, reduction of mature and overmature forest stand stocks resulting from clear cutting (Fig. 3a) and a decrease in total wood growing stock (Fig. 3b) are expected.

The opposing trends for forest type, age structure and growing stocks of exploitation and non-exploitation forests, as well as in anthropogenic dynamics of exploitation forests, leads to their mutual compensation and ensures relative stability of the overall Forest Fund characteristics. The strategy of sustainable management in the Siberian and Far Eastern Russian forests should be implemented so that only small (less than 10 %) variations of forest cover structure occur over the whole forecast period.

These trends of forest type, age structure and growing stock were more pronounced when modeled using data from the test sites. Modelling of natural dynamics resulted in the rapid aging of forests and an increase in the area of dark coniferous stands. Disturbance impacts cause the gradual slowing of these processes resulting in the stabilization of forest type and age structure of forests. Each rate of stand mortality has a corresponding tree species composition and age structure for ecoregion forests functioning under a constant regime. As stand mortality increases the


a) Natural dynamics of forest age structure

b) Natural dynamics of stem growing stock

Figure 2. Natural dynamics of forest age structure and of stem growing stock in forests of Siberia and Far East Russia

proportion of other wooded lands and young forests increases (Fig. 4a), and the area of coniferous forests decreases because of its replacement by deciduous stands (Fig. 4b).

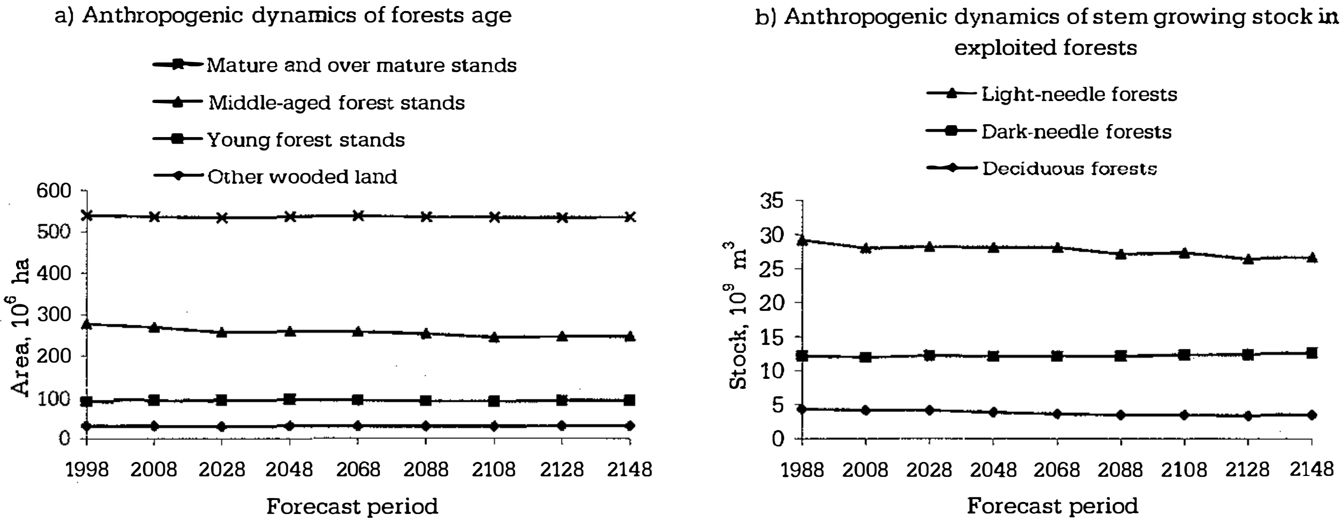


Figure 3. Anthropogenic dynamics of forest age structure and of stem growing stock in forests of Siberia and Far East Russia.

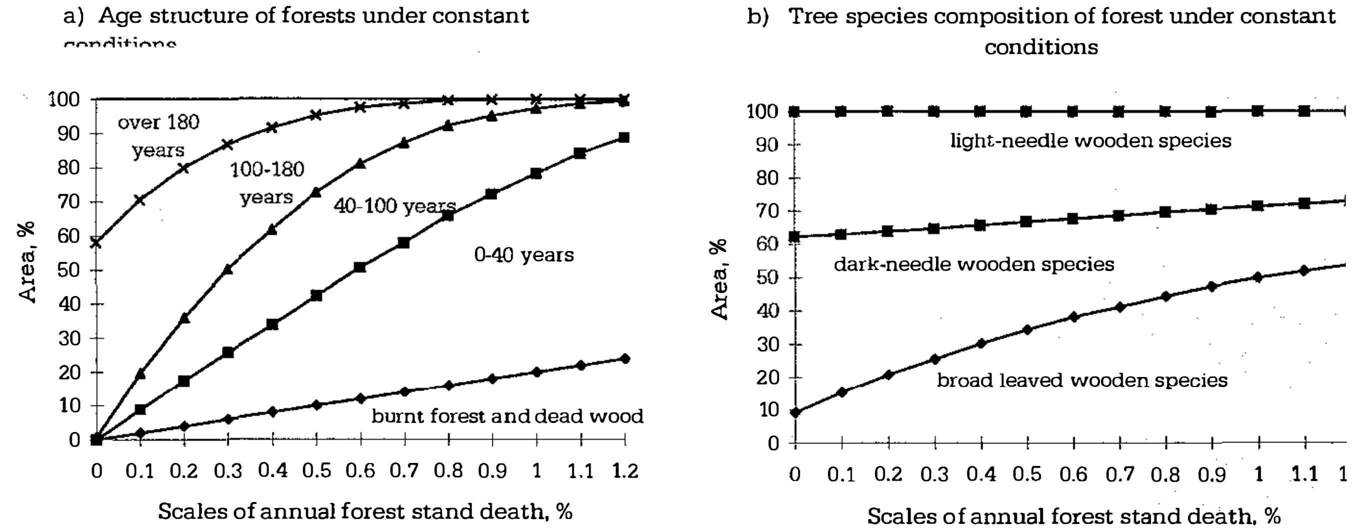


Figure 4. Forest type and age structure of forests in the Angarsky southern taiga ecoregion under constant conditions at different scales of forest stand death.

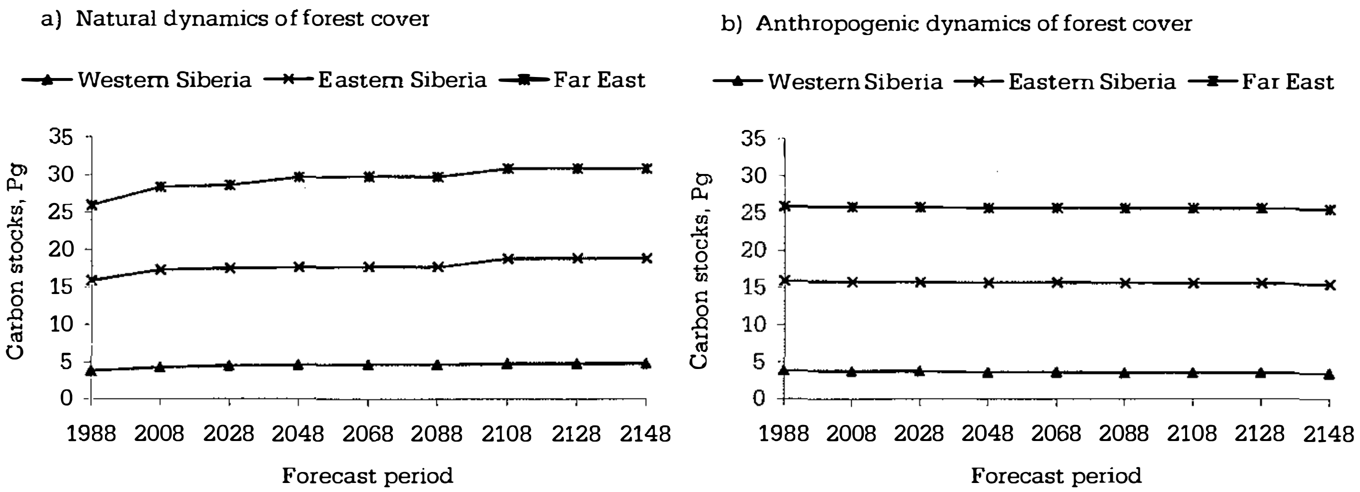


Figure 5. Dynamics of carbon pools in woody vegetation of Siberia and Far East Russia.

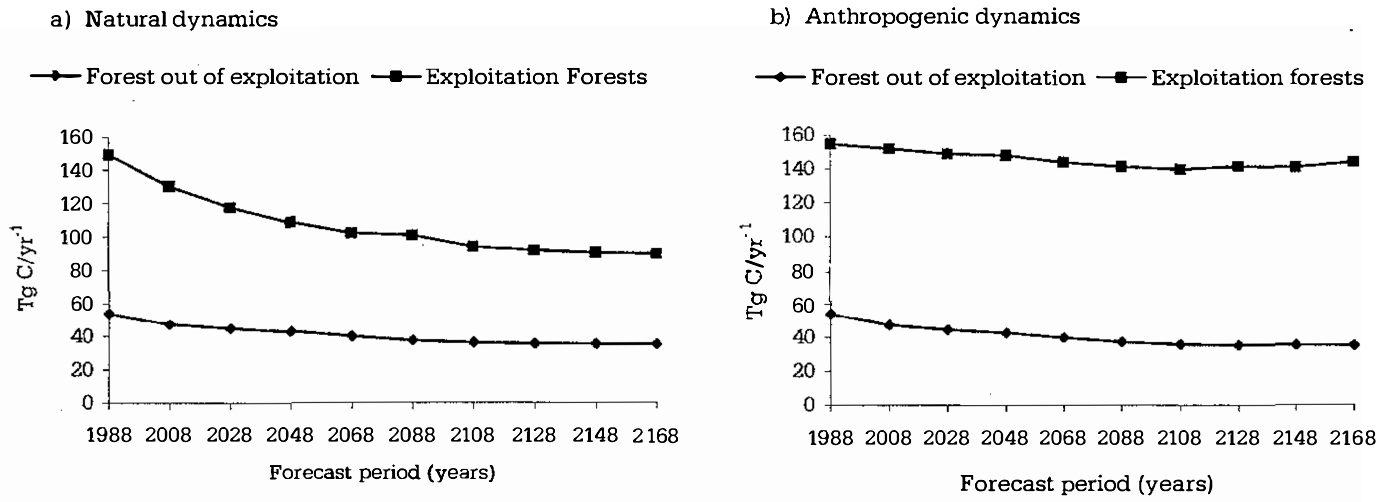
The changes over time in natural and anthropogenic forest type and growing stocks determine the corresponding dynamics of carbon pools and fluxes. The initial carbon pool of woody vegetation within forestlands of Siberia and Far East Russia was 26.0 Pg. Under a no clear-cut scenario the total carbon stocks contained in forest vegetation will gradually increase up to 30.2 Pg by the end of the forecast period (Fig. 5a). Under a maximum sustainable yield forest management strategy the carbon pool contained in Siberian and Far Eastern exploitation forests are projected to gradually decreased from 12.8 Gt to 11.8 Gt (Fig. 5b). Carbon stocks contained in non-exploitation forests will be increased from 13.1 Gt to 14.0 Gt for the forecast period. At the same time the total carbon pool of Siberian and Far Eastern Russian arboreal and bush vegetation remains practically constant because the increase of carbon stock in non-exploitation forests compensates for the decrease of carbon in exploitation forests.

The change in biomass carbon in the forest vegetation of Siberia and Far East Russia was about 150 Tg yr^{-1} at the beginning of the simulation period. Under a no clear-cut scenario the change in biomass carbon gradually decreased (Fig. 6a) to about 90 Tg yr^{-1} at the end of the simulation period. A maximum sustainable yield management strategy in Siberian and Far Eastern Russian forests lead to the stabilization of change in biomass carbon at the level of 140-145 Tg yr^{-1}. This is close to the current rate of 150 Tg yr^{-1} (Fig. 6b). In this case, the decreasing amount of change in biomass carbon in non-exploitation forests because of their aging, will be compensated for by the change in biomass carbon in exploitation forests, due to their rejuvenation.

Patterns for carbon fluxes are typical for the test ecoregions. Increasing the mortality rate due to forest fires, pests and forest diseases leads to a decrease in carbon pools and an increase in fluxes of carbon to forest vegetation. The carbon fluxes for each mortality rate tend toward a constant value (Fig. 7a). The carbon fluxes in forest vegetation under a constant regime, at high rates of tree mortality, may exceed (by more than 3 times), the fluxes of carbon when destructive impacts on forests are absent (Fig. 7b).

DISCUSSION

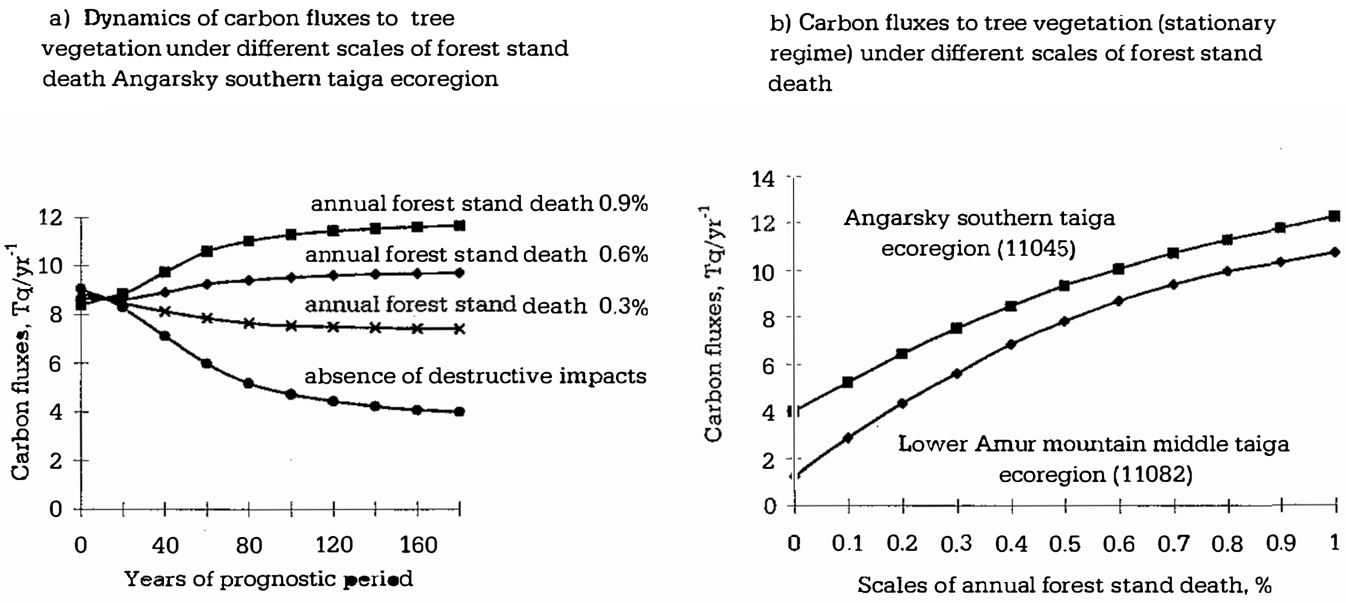
Estimates of carbon pools and changes in carbon pools in Siberian and Far Eastern Russian forest vegetation at the beginning of the simulation period, differ from those published earlier (Korovin and Karpov, 1995; Utkin et al, 1994). The carbon stock contained in forest vegetation at the beginning of the simulation is estimated at 26.0 Pg instead of 30.0 Pg (earlier estimate). This difference is explained by the exclusion of all other wooded and some forest lands (natural open woodlands, wastelands and glades). The change in the biomass carbon pool was 150 Tg/year (instead of 134 Tg/year) at the beginning of the simulation period. The new estimate for the change in biomass carbon is based on new information about growing stock inventories, that are presumed to provide a more accurate estimate than inventory data used in previous calculations.



a) Natural dynamics

b) Anthropogenic dynamics

Figure 6. Change in biomass carbon in tree vegetation of Siberia and Far East Russia.



a) Dynamics of carbon fluxes to tree vegetation under different scales of forest stand death Angarsky southern taiga ecoregion

b) Carbon fluxes to tree vegetation (stationary regime) under different scales of forest stand death

Figure 7. Carbon fluxes to tree vegetation at different scales of forest stand death.

The results of modeling of carbon pool fluxes in arboreal vegetation in Siberia and Far East Russia, indicate that forest age structure is an important factor influencing estimates of the forest carbon budgets. When the area of mature and overmature forest stands with high growing stocks and low or zero rates of biomass accumulation is extensive, the forest vegetation carbon stock is large and fluxes are low.

Natural dynamics of Siberian and Far Eastern Russian forests without cutting are characterized by growth of carbon pools and decreasing fluxes to arboreal vegetation. Anthropogenic dynamics of exploitation forests managed under a maximum allowable clear-cut scenario, will decrease carbon pools and increase fluxes to arboreal vegetation. Management strategies that allow for both natural dynamics of non-exploitation forests and anthropogenic dynamics of exploitation forests, will ensure the stabilization of carbon pools mentioned above, for Siberian and Far Eastern Russian forest vegetation, at a level close to the initial value for the simulation period. Such stabilization is the result of the compensating effect of opposing trends in changes to carbon pools fluxes of arboreal vegetation, in exploitation and non-exploitation forests.

For the purpose of estimating a total boreal forest carbon budget it is necessary to account for the dynamics of phytodetritus as well as phytomass. Dynamics of biomass carbon accumulation in forest vegetation, dynamics of carbon emissions due to forest death caused by destructive impacts (cuttings, forest fires, etc.) and phytodetritus destruction should all be estimated. To address these tasks, further development of mathematical models of forest vegetation dynamics are necessary that include modules describing the processes of exfoliation and wood fall in stands, as well as phytodetritus decomposition.

The development of mathematical models describing natural and anthropogenic dynamics of boreal forests carbon pools and fluxes in forest ecosystems, that account for global climate changes, is especially important. These models are needed to assess climatic change impacts on forest forming processes, tree-species composition and age structure, as well as stand productivity within different forest vegetation zones.

Long term forecasts of boreal forest dynamics, carbon pools and fluxes are required to form scientifically grounded strategies of sustainable forest use and forest management in Siberia and Far East Russia. The results presented in this work represent one possible approach to solving this important and serious problem.

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Appendix I

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